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Author-produced version of the article published in Journal of Environmental Management, 2013, 117, 150-161
Original publication available at http://www.sciencedirect.com/
doi : 10.1016/j.jenvman.2012.12.006

Wildfire frequency varies with the size and shape of fuel types in southeastern France: implications for environmental management

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Author-produced version of the article published in Journal of Environmental Management, 2013, 117, 150-161
Original publication available at http://www.sciencedirect.com/
doi : 10.1016/j.jenvman.2012.12.006

Abstract

Characterizing time intervals between successive fires in the recent history is of main interest for fire hazard prevention and sustainable environmental management as it indicates what the typical fire return interval for each type of ecosystem is. We tested the extent to which fire return intervals (FRI) depend on fuel type and age, and we compared FRI values between two fire-prone areas of south-eastern France (Provence). These areas had similar weather and roughly similar fuel types but fuels occurred in patches with different sizes and shapes in the landscape. We built a fire database (1960-2010) and we fitted Weibull distributions of FRI in order to compute the probability density function and the hazard of burning. Our results indicate maximal probability of burning again for shrublands (garrigues and maquis), and minimal values for mixed broadleaf-conifer forests and broadleaved forests. Most fuel types of Provence showed no effect of fuel age on the probability of burning again. Only the unmanaged maquis showed a linear increase of fire hazard in time due to a rapid postfire fuel build up. Rather long fire-free intervals and low age-dependency for most forest fuels of Provence suggest that reducing their biomass may not be sufficient to reduce fire risk. In contrast, the flammable shrublands have rather short fire intervals and represent a high fire hazard for the whole study area. The two areas had statistically significant difference of fire return intervals for a same fuel type (e.g. 18 to 22 years for shrublands, 20 to 24 years for pine forests, and 24 to 27 years for oak forests). This suggested that size, shape and connectivity of fuels play a major role in the probability of burning again and should be taken into account for fire management. The present policy of fire prevention puts efforts into public information and prevention, and preferential management of fuels at risk in the vicinity of roads and wildland-urban interfaces where fires occur preferentially. However, fire suppression may also take advantage of favouring low-flammable fuels with low age-dependency on strategic places in the landscape.

Keywords: fire regime; fire interval; Weibull model; Mediterranean-type ecosystems; landscape management

Author-produced version of the article published in Journal of Environmental Management, 2013, 117, 150-161
Original publication available at http://www.sciencedirect.com/
doi : 10.1016/j.jenvman.2012.12.006

1. Introduction

South-eastern part of France (so-called Provence) is a fire hotspot with ca. 35,000 fires burned during the 1973-2006 period and 8,500 ha burned annually on average (JRC-EFFIS, 2006). Provence also belongs to the biodiversity hotspot of the Mediterranean basin (Myers *et al.*, 2000) and has a wide range of Mediterranean type ecosystems (MTEs) with shrublands, forests, and grasslands. Literature has long stated that fire is a key disturbance in such MTEs, which has a major impact on Humans, ecosystems, and landscapes (Moreno and Oechel, 1994; Pausas *et al.*, 2008; Keeley *et al.*, 2012). Previous studies in Provence have indicated that fire is an essential element in the vegetation dynamics, thus shaping the landscape mosaic (Curt *et al*., 2011; Schaffhauser *et al*., 2011). In Provence as in many MTEs of southern Europe, abandonment of former agricultural practices, afforestation policy and the increase of population were major drivers of fire risk during the past decades (*In* Moreira *et al.*, 2011). Shrublands have expanded in the past decades because of the abandonment of former agropastoral management, and they sometimes constitute large tracts with high fire hazard (Curt *et al*., 2011; Schaffhauser *et al*., 2011). Extensive afforestation with conifer forests (mainly *Pinus halepensis* and *P. pinaster*) and their spontaneous expansion from planting (Barbéro *et al.*, 2000) have strongly increased the fuel biomass and the connectivity of fuels on landscape scale. Thus, pine forests have been claimed to increase fire hazard in Provence, because they favoured intense crown fires (see Fernandes and Rigolot, 2007). Mixed pine-oak forests have a reputation for being highly flammable because of the vertical connectivity between the different species (see *In* Pausas *et al*., 2008). Only oak forests are considered to be low flammable and resilient due to a good resprouting ability (Pausas et al., 2008; Curt et al., 2009). Urban sprawl and the strong increase of population (Moreira *et al*., 2011) have favoured the development of road corridors and wildland-urban interfaces (i.e. the area where houses and the wildland vegetation coincide) where vegetation is generally extensively managed, with possible incidence on fire risk. The Provence area includes two neighbouring fire-prone areas with the same climate and roughly similar fuel types but with different size, shape and connectivity of fuels across the landscape: the Aix-Marseille area and the Maures massif. This configuration provided a unique opportunity to investigate if fire frequency was similar for a given fuel type but different patch size and connectivity. This has implications for management, because if fire frequency is similar for a certain fuel type in both areas then similar management should be applied.

In this context, it is crucial for sustainable management to characterize the fire frequency specific to each type of ecosystem (hereafter referred to as 'fuel types', i.e. identifiable associations of fuel elements of distinctive species that will cause a predictable fire behavior; Pyne *et al.*, 1996). It is noteworthy that although the Provence area is a fire hotspot, no georeferenced fire database existed yet. This prevented any accurate fire frequency analysis whereas it existed in neighbouring countries such as Spain (e.g. Diaz-Delgado *et al*., 2001) or Portugal (e.g. Oliveira et al., 2011). This may explain why controversy still persists among land managers in Provence about the role of fuel age and the typical fire return interval for the most common fuel types. Literature stated that characterizing the past fire regime, i.e. spatial pattern, frequency, intensity and seasonality of fires prevailing in an area (Gill, 1975; Pyne *et al.*, 1996) is necessary to assess fire risk and the ecosystem vulnerability. Fire frequency is a main feature of fire regime which is notably characterized by fire intervals, i.e. time in years between two successive fires in a designated area. First, this key component has major implications for fire risk assessment since it is important to predict the mean fire return interval or the probability of a new fire after having burned (Moritz *et al.*, 2009). Secondly, fire intervals control in part the survival and regeneration of many species (Pausas *et al*., 2008) and thus the dynamics and sustainability of wildland and forests. As a consequence, the characterization of fire return intervals (FRI) has been increasingly used to guide forest management to prevent fires by fuel treatment (Keeley *et al*., 1999; Fernandes and Bothelho 2003), to mimic the natural disturbance regime in order to maintain biodiversity (Martin and Sapsis, 1992; Burrows 2008), or to

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limit the risk of species extinction in fire-prone ecosystems (Allen *et al.*, 2002). It has been proved that fires at inappropriate time intervals may change the quality of habitats (Burrows, 2008) or lead to the extinction of interval-sensitive species (Eugenio *et al.*, 2006; Pausas, 2006; Russell-Smith *et al.*, 2010).

The distribution of FRI in a landscape results from the recurrence and the patterning of fires which in turn results from a complex stochastic process of ignition, spread, and regrowth dynamics (Moritz *et al*., 2009). Interactions between ignitions sources, weather, topography and land cover explain why some locations burn more often than others in a landscape (Mermoz *et al.*, 2005; Moreira *et al.*, 2011). In many MTEs, ignition is man-induced (Moreira *et al.*, 2011) then favoured by weather. As a result, most points of ignition aggregate in road corridors (Curt and Delcros, 2010) or wildland-urban interfaces (Lampin-Maillet *et al.*, 2010) and fuel types located in the vicinity of these areas are likely to burn more frequently than others. FRI should also strongly depend on factors controlling fire spread within the landscape, including physiography and fuels. In French or Spanish MTEs it has been proved that fire recurrence is higher on certain topographic positions (e.g. south facing slopes and crests; Mouillot *et al*., 2003; Vazquez and Moreno, 2001) due to a combination of factors linked to ignition, topoclimate, and fuels (Moreira *et al.*, 2011). Fuels can play a major role in variations of FRI as certain fuel types or land covers are especially flammable because of their composition, biomass, structure, and moisture. Higher fire selectivity for shrublands rather than for forests and almost agricultural lands has been stated in many MTEs (see *In* Moreira *et al.*, 2011). Some fuel management practices such as forest thinning, planting or shrub-clearing would also affect the probability to burn (or to reburn) and to propagate fire because they modify the biomass and spatial arrangement of fuel. As FRI express the probability of reburning within a certain time period, it also depends upon the ability of a fuel to recover after fire or to shift to another fuel type (Moritz *et al*., 2009). Some fuel types exhibit rapid postfire fuel build-up whereas others can stay for a long time with fuel load insufficient to carry a new fire ('fuel limitation' stage). A feedback exists between fuels and fire (i.e. fuels produce fires and, in turn, are affected by fires), and a typical FRI is expected to settle down for a certain duration in a certain landscape (Turner, 1989).

In the study we assessed the fire return intervals for two fire-prone areas in Provence, based on an original georeferenced fire database from 1960 to 2010 including all fires larger than ca. 3 ha. First, we tried to find out what the typical FRI for each fuel type was. This knowledge is crucial to guide the type of management to apply to ensure the ecosystem sustainability. For that purpose we computed the FRI for each fuel type and area using both censored and uncensored data. Censored data take only into account fire intervals between two fire dates known precisely while uncensored include all data. Then, we tested the hypothesis that FRI could differ from one area to the other for a same fuel type (and a similar climate) due to a difference of size, shape and connectivity of fuels. For that purpose we compared FRI for a same fuel type in the two areas. Our final objective was to advise what type of fuel management could be applied to the main fuel types existing in Provence.

2. Materials and methods

2.1 Study areas

We selected two study areas within the Provence fire hotspot since it enabled us to compare FRI with similar climate and fuel types but with different size, shape and connectivity of fuels across the landscape. The Aix-Marseille area (called below AIXM; central point: 43° 20N, 5° 23E; area 510,593 ha) has ecosystems on limestone-derived soils dominated by *Pinus halepensis*, *Quercus ilex* and *Quercus pubescens*, and *Quercus coccifera* shrublands (Appendix 1A). The Maures massif (called below MAUR; central point 43.3° N, 6.3° E; area 145,686 ha) is made of a gneissic substratum and has

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siliceous soils. Its ecosystems are dominated by *Pinus pinaster, Quercus suber (with Q. pubescens and Q. ilex)*, *Erica-Cistus* shrublands (Appendix 1B). Fuels are roughly similar in these two areas with the exception of mixed forests (Appendices 1A-1B; Fig. 3).

Conversely, the two areas exhibit a high contrast of size, shape and connectivity of fuels in the landscape. The Maures massif is dominated by wildland with sparse cork oak woodlands intermingled with large extents of flammable *Erica-Cistus* maquis (Fig. 1B). In contrast, the Aix-Marseille area is a matrix of fragmented and interspersed pine and oaks forests, with small patches of *Quercus coccifera* garrigue. In addition, wildland and forests are almost absent in the western part (Fig. 1A) which is dominated by agriculture, industries, and salt pans. The two areas are close to each other and have the same climate making them conducive to frequent and intense summer fires (JRC-EFFIS, 2006). Both areas had roughly similar fuel types including shrublands, pine forests, oak forests, and mixed pine-oak forests. Shrublands are fire-prone ecosystems dominated by seeders (e.g. *Cistus* spp., *Ulex parviflorus*) or resprouting species (e.g. *Quercus coccifera*, *Erica arborea*) that have evolved to adapt to fire (Pausas 2006) and are generally flammable (Curt *et al*., 2011). They have low fuel moisture content during the summer season (Schaffhauser *et al*., 2011). Pine forests can generate surface fires of moderate intensity (fireline intensity < 2000 $kW.m^{-1}$) or high-severity crowning fires depending upon the pine species and the stand structure (Rigolot, 2004; Fernandes *et al.*, 2008). In both areas, the climate is typically Mediterranean, classed as subhumid xerothermic (Quézel and Médail, 2003) with a mean annual rainfall of ca. 550 mm and a mean annual temperature of 15.9°C, and high interannual and seasonal variability. The summer period is characterized by severe droughts associated with strong wind, which favour fire ignition and propagation.

2.2 Assessment of fuel and landscape characteristics

Fuel composition, fuel biomass and fuel patterns in the landscape have been studied extensively in previous studies in both areas. First, we established 20*20 m plots in the different fuel types, and we described live and dead fuels according to a standard protocol. Fuel particles of different size were then collected and oven-dried to get the dry biomass. Detailed information on field and laboratory protocols are available in previous studies (Ganteaume *et al.*, 2009; Curt *et al.*, 2011; Schaffhauser *et al.*, 2011). The main characteristics of fuels are described in Appendices 1A and 1B. In order to make simpler comparisons we regrouped the fuels into four main fuel groups having different FRI values: shrublands, pine forests (including those of WUI and road corridors), mixed forests (pine-oak mixings), and oak forests (Fig. 3).

Secondly, we characterized spatial patterns of fuel and fires on landscape scale using the Geographical Information System *ArcGis 9.3* (ESRI, Redlands, CA, USA) and the Patch Analyst and Patch Analyst for grid programs v. 4.2.13. Thanks to it, we could compute landscape metrics for the different fuel types: (i) the mean patch size, which indicates how large the patches of each fuel type are; (ii) the mean perimeter/area ratio, which indicates if the fuels patches are irregular or not; and (iii) the area weighted mean shape index which increases as the patch shape becomes more irregular (Table 1). These indices were used to investigate if the size and connectivity of the fuels patches and the burned scars were different in the two study areas. Indeed, we hypothesized that the Aix-Marseille had a very patchy mosaic of fuels with a smaller grain than the Maures area, which would limit the probability of having very large fires burning frequently the same fuel patches. We also used the FragStat software (McGarigal *et al.*, 2002) to compute a fuel contagion index which reflects dispersion (i.e. the spatial distribution) and intermixing of patch types on raster maps (pixel size 100 m) over each landscape. The index is computed from the frequencies on which pairs of fuels occur as adjacent pixels and it varies from 0 (maximal dispersal of pixels of fuels over the landscape) to 100

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Original publication available at http://www.sciencedirect.com/
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(all pixels of fuel are adjacent). For this purpose we coded all fuel types (shrublands and forests) as 1 and non-fuels (including agricultural lands, cities, roads) as 0. The contagion index was intended to indicate to which degree fires may theoretically propagate throughout the flammable fuels within each study area (Table 1). We also computed a contagion index specific to the shrublands and one specific to the forest, for each study area. These indices were computed in order to test to which degree shrublands and forests may favour the propagation of fire in each area. We also computed the MESH and SPLIT indices. MESH equals the sum of fuel patch area squared, summed across all patches of the corresponding patch type, divided by the total landscape area (m^2). It is maximum when the landscape consists of a single patch. The SPLIT index equals the total landscape area (m^2) squared divided by the sum of patch area (m^2) squared, summed across all patches of the corresponding fuel patch type. SPLIT is equal to 1 when the landscape consists of single patch, and increases as the focal fuel patch type is increasingly reduced in area and subdivided into smaller patches. We compared the values of the different landscape metrics between the two areas using the Mann-Whitney-Wilcoxon *W* test or the Tukey HSD test with $\alpha < 0.05$, using the R statistical software (R Development Core Team, 2011).

2.3 Georeferenced fire database

We built fire recurrence maps for both study areas (AIXM and MAUR) on the basis of a comprehensive study of burn scars visible on a set of satellite images (1960-2010; Fig. 1). A first set of georeferenced burn scars had been drawn in the field by foresters and fire-fighters for the older fires (1960-1985). They have been gathered, checked and redrawn individually, completed using Landsat images with MultiSpectral Scanner (MSS, spatial resolution 80 m, 1973-1983), then using TM and ETM (Enhanced Thematic Mapper, resolution 30 m) since 1983. Since 1999 we have also used satellite images (MODIS, resolution 250 m) using semi-automatic recognition by the multi-dates comparison of the normalized vegetation index (NDVI) (see Curt *et al*., 2011 and Schaffhauser *et al*., 2011 for details). In practice, we checked for every fire listed in the French fire database 'Prométhée' (http://www.promethee.com) since this database gives the date, the point of ignition and the estimated size of all fires since 1973. From this information, we assessed the NDVI value all around the likely location of the fire. NDVI data were extracted from the so-called 16 days Vegetation Index Products (MOD13Q1) with 250m resolution images one month before and after the fire. This VI compositing algorithm includes the maximum value composite (MVC) and a constraint on view angle – maximum value composite (CV-MVC). This technique of compositing data reduces noise in the surface reflection signal including QA data sets with statistical data indicating the quality of the VI product (Gu *et al.*, 2009). The comparison of the value before and after the fire confirmed if a fire occurred; a threshold of 0.5 for the Δ_{NDVI} was considered as indicative of a fire. The final contour of the burn scar was drawn using Landsat images. Finally, we used only the fires larger than 3 ha because smaller fires may have not been detected (and thus, underestimated). Fire maps were created for each year under analysis (1960-2010, then pooled together using the Geographical Information System *ArcGis 9.3* (ESRI, Redlands, CA, USA). The time intervals between successive fires have been extracted for all fires using a 25x25 meters regular grid. In example, if a certain pixel burned in 1990, then in 2003 and 2010, the two times intervals were 13 and 7 years respectively. For each pixel of the 25*25 meters grid we collected the information on the fuel type at different dates to account for changes that could have occurred over time in a pixel. For this purpose we used maps of vegetation (National Forest Inventory, spatial resolution 60 to 30 m; http://www.ifn.fr/spip/) and land cover (Corine Land Cover, spatial resolution 100 m; http://www.eea.europa.eu/fr) for 1975, 1984, and 2004. To each pixel was assigned the vegetation type existing at the time just before the date of fire.

2.4 Modelling the fire return intervals

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doi : 10.1016/j.jenvman.2012.12.006

Fire frequency analysis is considered as a branch of pyrostatistics (Moritz *et al.*, 2009). When characterizing FRI using a survival analysis (Smith, 2002; Lawless, 2003), fires are considered 'deaths' and fire intervals represent survival time (Johnson and Gutsell, 1994). Thanks to that we can assess FRI on time series for a given landscape by computing hazard functions which reflect how the probability of fire changes with the age of fuels. Such analysis requires accurate fire data during a time period sufficient to let successive fires to burn some locations and to account for temporal heterogeneity, i.e. a minimum of several decades for many MTEs experiencing frequent fires (Oliveira *et al.*, 2011). Few FRI studies of are available in MTEs due to the lack of such fire databases. Fire-frequency analysis has been proposed to test the extent to which fuel age could influence the fire incidence (e.g. Moritz *et al.*, 2004b). Actually, controversy about the environmental drivers of fire intervals in many MTEs has risen in the past years. For some authors, fire occurrence in shrublands and conifer forests is time-dependent and fire hazard would increase with the age of fuels (Minnich, 1983; Minnich and Chou, 1997), while others suggest little role for fuel age (Moritz *et al.*, 2004b; Keeley and Zedler, 2009). This debate expanded to different MTEs such as the South-African *fynbos* (van Wilgen *et al.*, 2010) or the Australian woodlands and shrublands (O'Donnell *et al.*, 2011). It has ecological and management implications: if fire hazard increases with fuel age, then fuel management would be effective to limit future fires thus justifying extensive fuel treatment.

Fire data are typically a combination of uncensored and censored fire intervals (Moritz *et al*. 2009), censored values being those prior to the first fire on record, and those following the last fire on record. Thus, censored values (Polakow and Dunne, 1999) take only into account fire intervals between two fire dates known precisely. This censoring limits the size of the fire database but this is essential for estimating correctly the fire return interval and to avoid biases in the *b* and *c* parameters of the Weibull function (see below; Moritz *et al.*, 2009). The fire interval can be complete (i.e. bounded on both ends with known dates) or not (Polakow and Dunne, 1999). Moritz *et al*. (2009) have explored the consequences of using censored or uncensored fire data and concluded that disregarding censored data may cause distortion of the real fire hazard. Recently, Oliveira et al. (2011) have also tested this effect for FRI during the 1975-2005 period for the whole Portugal and confirmed that censoring fire data has a predominant effect on FRI. It is likely to be especially important when the database comprises few fire intervals (e.g. a short record coming from a frequent fire system or a long record in a system with very infrequent fires). In this paper we used both complete censored data (called below 'censored') and uncensored data in order to investigate the extent to which this could change the results.

We fitted a Weibull model to the distribution of fire intervals at our sites, using our fire atlas. The Weibull model (see Johnson and Gutsell, 1994) has been proved flexible and able to provide superior fit to fire history data than most other distributions (see Grissino-Mayer *et al.*, 2004). Using this model, the distribution of fire intervals can be described in two complementary forms: F(t) gives the probability of fire occurrence before or at a time t, and f(t) is the probability density function reflecting the frequency of burning in a given time interval (Moritz, 2003; Moritz *et al.*, 2009) as follows:

$$F(t) = 1 - \exp[-(t/b)^c] \qquad \text{(eq 1)}$$

$$f(t) = (ct^{c-1}/b^c)\exp[-(t/b)^c] \qquad \text{(eq 2)}$$

where t>0, b>0 and c≥0. The parameter *b* is the scale parameter related to the expected interval between fires, while *c* is the shape parameter, which shows how the hazard of burning changes with time since the last fire. The parameters *b* and *c* have ecological meaning: the scale parameter *b* is the typical 63.2 percentile of fire intervals (i.e. the typical fire return interval surpassed 36.8% of the

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time, in years; Polakow and Dunne, 1999) while the *c* shape parameter indicates the change in fire probability through time (see below). The hazard of burning function $\lambda(t)$ gives the probability of a fire to occur within a specific time interval (Moritz *et al.*, 2004a):

$$\lambda(t) = ct^{c-1}/b^{c} \qquad \text{(eq 3)}$$

where *t* is the time since the last fire, and *b* and *c* are estimates of the Weibull model. This hazard of burning function is useful because it typically reflects how the probability of fire changes with the age of fuels. Actually there is still a debate on whether fuel age (or the time-since-the-last-fire) may have an impact on fire recurrence or not. In short, some authors (Minnich, 1997) argue that large fires are fuel-driven while others argue that they would be mostly climate-driven (Keeley *et al.*, 1999; Moritz *et al.*, 2004a). This issue can be solved using the Weibull's *c* shape parameter. Indeed, values of $c = 1$ reflect no change in hazard of burning with time, while $1 < c < 2$ indicates a hazard growing with a diminishing rate, $c = 2$ indicating a linear increase, and $c > 2$ indicating an increasing hazard with time (see Moritz *et al.*, 2004a). A *c* value of 1.42 is considered the maximum estimate for a combined fuel-weather effect for Californian chaparral (Polakow and Dunne, 1999; Moritz *et al.*, 2004a).

We computed these Weibull equations on fire intervals between 1960 and 2010. The *b* and *c* parameters were estimated by maximizing the likelihood function, and the Kaplan-Meier reliabilities to the positions of the points plotted by the survival analysis (Dodson, 2006). We tested the goodness-of-fit using a Kolmogorov-Smirnov test. In addition, we also computed the median fire return interval (also called the median fire-free interval, MEI; Grissino-Mayer, 1999) for each fuel type, using the estimate on which the proportion of the area surviving without a successive fire is 0.5 in the generalized Weibull model (van Wilgen *et al.*, 2010), accounting for censored values. MEI is computed as:

$$\mathrm{MEI} = b(\ln 2)^{(1/c)} \qquad \text{(eq. 4)}$$

For each area we also computed the fire cycle (Johnson *et al.*, 1999; Crow et al., 1994) or the natural fire rotation period (NFR) as the length of time required to burn the equivalent of a specified area (Heinselman, 1973; Agee, 1993):

$$NFR = \frac{N}{A/S} \qquad \text{(eq. 5)}$$

Where *N* is the number of years during the period studied, *A* is the total area burned, and *S* is the total study area.

3. Results

During the 1960-2010 period, a total of 1,451 fires larger than 3 ha have been considered for the AIXM area (sum of 78,860 ha burned), and 217 fires (sum of 59,606 ha burned) for the MAUR area (Fig. 2). The two areas had contrasted fire and fuel patterns: MAUR had much less fires but they were much larger than AIXM, and MAUR burned more extensively and frequently. As a result, the fire cycle was much lower for MAUR (Table 1). This area had also larger fuel patches, especially for shrublands although the proportion of shrublands in the landscape was similar in the two areas (ca. 55%, Table 1). The resulting fuel contagion index was clearly higher for MAUR than for AIXM for shrublands, forests, and the whole landscape (Fig. 4). The contrast of fire patterns in the landscape was clearly visible on Fig. 1. In the MAUR area, large and superimposed fires correspond to large patches of maquis located in the western and eastern parts of the massif, while a part of the central massif covered with mature oak forests and chestnut coppices has remained unburned (Fig. 1B). In the AIXM area, fires are much smaller and fragmented, and they are mostly located on the eastern

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part of the region dominated by wildland and forest. The characteristics of the main fuel groups were roughly similar in the two study areas (Fig. 3). Only the mixed fuels (mainly pine-oaks mixings) had significantly higher tree cover, fuel bed depth and 1-h fuel load in AIXM.

The mean time interval between fires (FRI) varied not much across the study areas (ca. 18 to 31 years; Appendices 2A and 2B), although differences between fuel types were higher for MAUR. For most fuel types of both areas, censoring the fire data clearly increased the value and the range of the *b* parameter while it decreased the value of *c* (Appendices 2A and 2B). The mean FRI and median fire-free intervals (MEI, censored values) were minimal for shrublands in both study areas (Appendices 2A and 2B). The AIXM area showed clear differences of fire intervals and of the Weibull's *b* parameter between fuel types (Appendix 2A). The lowest MEI values were recorded in garrigues. Intermediate values (ca. 30-35 years) were for pine forests, sparse woodlands and spontaneous afforestation (dominated by pines), and fuels of road corridors dominated by shrubs and grasses. Oak forests, pine-oak mixings and fuels of WUI (often shrub-cleared) had clearly longer fire-free intervals and *b* parameter. The *c* parameter was strikingly low for all fuel types of AIXM, which resulted in slow increase of hazard of burning with time in comparison to MAUR (Fig. 5). In the MAUR area, MEI and the *b* parameter varied much less (Appendix 2B). However, the different subtypes of maquis had lower values than the forests types including cork oak, mixed oaks or pine-oak mixes. The *c* parameter was clearly low for all forest fuels (1.19 to 1.37) while it was close to 2 for all maquis (Appendix 2B). The figure 5 displayed the hazard of burning $\lambda(t)$ for the four main groups of fuels in both areas. It showed that the MAUR maquis had clearly higher hazard of burning over time than all the other fuels, and especially than the AIXM garrigues. Indeed, the hazard of burning strongly increases in the Maures maquis beyond 10 years after a fire.

4. Discussion

4.1 The interplay between landscape, fuels and fire

Our study areas showed contrasted patterns of fires (burned scars) and fuels on the landscape scale. The Maures massif was 3.5 times smaller than the Aix-Marseille area. It had large homogeneous fuel patches of shrublands and forests favouring the contagion of fire across the landscape. Indeed, it had large fires distributed evenly across the landscape and often superimposed although having a low fire occurrence. This resulted in a short fire cycle. This also corresponds to a specific patterning of ignitions: most of them occur around the Maures massif in the vicinity of roads, WUI and resorts, then large fires spread throughout connected maquis and cork oak forests within the massif (Curt *et al*. 2011). This feature has favoured a positive fire-fuel feedback and the expansion of maquis during the past decades (Schaffhauser *et al*. 2011). In contrast, the Aix-Marseille area had fuels fragmented with a fine grain across the landscape, numerous but small fires, and these fires were less superimposed. This logically resulted in a much longer fire cycle. In this area, ignitions occur throughout the whole area along the dense network of roads and WUI, but fire spread is limited by the fine-grained fragmentation of shrubland and forest patches.

This interplay between fires and landscape generated a striking contrast between the two neighbouring areas: ca. 41% of the MAUR area was burned at least once during the last five decades against only ca. 15% for AIXM. These two areas had similar weather and causes of fires (and ignition rates), and the limited differences between the characteristics of fuel types suggest that this may not be sufficient to generate such contrast with fire recurrence. We suggest that the size, shape and connectivity of fuels across the landscape may have affected fire recurrence. This result is in line with the findings of O'Donnell et al. (2011) in semi-arid shrublands and woodlands of Australia, where the spatial variation of fuels and their connectivity across the landscape control the fire intervals.

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Original publication available at http://www.sciencedirect.com/
doi : 10.1016/j.jenvman.2012.12.006

The use of a same distribution type (i.e. the Weibull model) applied to fire interval data on a same time span provided a standard system for comparing the fire regimes of AIXM and MAUR. For AIXM, censored data showed increasing fire intervals from garrigues to pine forests, oak forests, then and mixed forests. But fuel age has a low effect on fire hazard whatever the fuel type is, even for the garrigues which recover their fuel biomass rapidly after a fire. The low values for the Weibull's *c* parameters ($1 < c < 2$) indicate a hazard growing with a diminishing rate, and thus suggest a limited role of fuel characteristics and the predominance of weather (Keeley *et al*., 1999; Moritz *et al.*, 2004a). Only the maquis in MAUR exhibits a *c* parameter close to 2, which indicates a linear increase of fire hazard over time.

Shrublands of both study areas are undoubtedly the most likely fuels to reburn since they have low fire intervals in comparison to forests. Actually, shrubby fuels are long acknowledged fire-prone in many MTEs throughout the world (Moreira *et al.*, 2001; Mouillot *et al.*, 2002; Baeza *et al.*, 2006; Syphard *et al.*, 2007; Saura-Mas *et al.*, 2010). Many shrubs common to MTEs (*Quercus coccifera*, *Erica arborea*, *Cistus* spp., *Ulex*) have been proved highly flammable due to a high amount of fine and dead fuel particles, low fuel moisture content during summer, and chemical compounds (Baeza *et al.*, 2006; Saura-Mas *et al.*, 2010). In addition, shrublands of Provence contain high proportion of flammable grasses such as *Brachypodium retusum* which favour ignition (Curt and Delcros, 2010; Curt *et al*., 2011). Once ignited, these fuel types generally generate rapid crown fires of medium- to high intensity which can propagate over large areas (Pausas *et al*., 2008; Curt *et al*., 2011). The MAUR maquis had an almost linear effect of fuel age, which fits with the fact that most shrublands are renowned for having rapid post-disturbance recovery of fine biomass (generally within 9 to 26 years, Trabaud, 1998; Fernandes *et al.*, 2010) that would favour rapid reburning. In Portugal, shrub-dominated areas exhibit rapidly increasing hazard of burning with fuel age; *c* values ranging from 2.6 to 4.8 (Fernandes *et al*., 2010). In the MAUR maquis, postfire fuel accumulation is very rapid due to the predominance of efficient seeder shrubs (notably *Cistus*) which colonize the gaps after fire, and of efficient resprouters such as *Erica arborea*. Both have flammable live fuels (Schaffhauser *et al*., 2011) and dead fuels (Curt *et al*., 2011). A positive feedback likely exists between fire and shrubs abundance since most areas burned recurrently turn into flammable shrublands (Curt *et al*. 2011; Moreira *et al.*, 2011; Mouillot *et al.*, 2003; Diaz-Delgado *et al.*, 2004) due to the predominance of auto-succession (Trabaud 1998; Lloret, 2002; Acácio *et al.*, 2009). In contrast, the hazard of burning tends to decrease over time in the AIXM area. First, this counter-intuitive finding may be explained by the fact that many already burned areas turn into bare soil. Secondly, limestone-derived soils are low fertile and postfire recovery of *Quercus coccifera* garrigue is rather long (Trabaud 1998), thus making shrublands on limestone substrate less flammable than those on siliceous soils (Fernandes and Botelho 2003). Thirdly, the recent policy of fire prevention has promoted shrub clearing in strategic places to establish firebreaks, notably in areas already burned. This study defends the strategy of reducing the biomass of shrublands in strategic places (WUI, road corridors, and fuelbreaks) to limit future fires, using notably prescribed fire and sylvopastoralism. However, fuel reduction has been proved effective to limit reburning only during few years in most shrublands (generally 2-4 years; Fernandes and Botelho, 2003; Moritz *et al.*, 2004b; van Wilgen *et al.*, 2010), and only if fire weather is not severe or extreme (Moreira *et al*., 2011). In the MAUR massif particularly, the expansion of maquis challenges the efforts for fire prevention and fire-fighting.

Conifer forests were renowned among forest managers for conferring high fire hazard over the Provence landscape. This study confirms they have rather short fire intervals due to their high flammability and combustibility but low hazard of reburning, indicating that the fuel rebuilding after a fire is rather slow. On the one hand, *P. halepensis* in AIXM and *P. pinaster* forests in MAUR are highly flammable due to a heavy fuel load and high amount of fine particles which burn readily (Ganteaume et al., 2011; see also Dimitrakopoulos *et al.*, 2007; Alessio *et al.*, 2008; De Lillis *et al.*,

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doi : 10.1016/j.jenvman.2012.12.006

2009). Many Mediterranean pines thus cause high-intensity fires, including crown fires (Fernandes and Rigolot 2007). On the other hand, pine forests of Provence fortunately exhibit low effect of fuel age after a fire. This is coherent with the low fuel accumulation over time in the overstory: understory grasses and shrubby fuels decrease rapidly with stand closure and maturation, thus reducing the fire hazard (Fernandes, 2009). The reputation of *Pinus halepensis* forests to generate fire risk is likely due to its large expansion throughout Provence during the past decades (Barbéro *et al*., 2000), especially after grazing cessation. In that case, pines expanded in shrubby and grassy fuels, thus creating very flammable fuels. A firewise way to manage these fuels is to thin and prune trees, clear the shrubs understory, and favour the establishment of oaks. This may direct them towards less flammable fuels in the medium-term (see Pausas *et al*., 2008). In the Maures massif, although being especially fire-resistant (see Fernandes and Rigolot 2007), *Pinus pinaster* has strongly retracted after recurrent fires and postfire pests and diseases. *P. pinaster* is only a strong danger when mixed with maquis, because it can generate high-intensity crowning fires.

Fuel types typical of WUI and road corridors have lower fire intervals than forests, confirming their fire-proneness. WUI and road corridors are preferential areas for ignitions due to the proximity of human activities causing most fires. However, the effect of fuel age for these fuels remains low owing to the fact that these areas are often intensively managed to prevent fire spread. We also suspect our database to be biased for these fuels since most fires are very small due to the rapid intervention of fire fighters and the small size of fuel patches. Actually, very small fires (e.g. < 1 ha) were not mapped in our database.

Mixed pine-oak forests and almost broadleaved forests (*Quercus ilex*, *Q. pubescens*, *Castanea sativa*) have long fire intervals and no effect of fuel age on fire hazard. This suggests a low and slowly increasing hazard after a fire, coherent with the low flammability of these fuels due to the low ignitability of their litter (Curt *et al*. 2011) and to the slow increase of fuel biomass with stand age (Schaffhauser *et al*. 2011). Where no fire occurred over the past 50 years, most mixed conifer-oak forests have progressively turned into oak-dominated forests (Curt *et al.*, 2009; Schaffhauser *et al*., 2011). Some have suffered few fires of low to medium intensity, which have opened windows of regeneration for oaks (e.g. Pausas *et al*., 2008 for Spain). Cork oak (*Q. suber*) forests of the MAUR area burned more frequently due to the expansion of surrounding maquis. As stated by Acácio et al. (2009) in Portugal, this protected habitat tends to shift towards shrublands due to a combination of droughts, fires and plant competition (Curt *et al*., 2009).

4.2 Comparison with other MTEs

Our study provided knowledge of the range of FRI for Provence, which is vital for the ecosystem management and sustainability. Provence is undoubtedly a fire hotspot for France (JRC-EFFIS, 2006), rather alike Corsica (mean FRI = 20 years, Mouillot *et al.*, 2003). We acknowledge that fire frequency depends critically on the area over which it was computed (Johnson and Gutsell 1994), thus inter-regional comparisons must be done cautiously. However, our FRI values for shrublands were intermediate between those of many MTEs including the Portuguese shrublands (12-16 yrs., Fernandes *et al*. 2010), South Africa *fynbos* (10-13 yrs., Van Wilgen *et al*. 2010) or grasslands (2-10 yrs., Archibald *et al.*, 2011), and those of southern California chaparral (33-42 yrs., Moritz 2003; Moritz *et al*. 2004) or shrublands of south-western Australia (47 yrs., O'Donnell *et al.*, 2011). Fire intervals in forests were much higher than those of ponderosa pine forests of Colorado (2-20 yrs., Grissino-Mayer *et al.*, 2004) but much lower than those of Australian forests (310 years, (O'Donnell *et al*., 2011). They fall within the range for many MTEs (20-50 years, *In* Diaz-Delgado *et al.*, 2004). Literature on natural fire rotation (NFR) period is scarce but we compared our data with those of neighbouring countries. The Maures massif had a short NFR value similar to that of Sierra de Gredos

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doi : 10.1016/j.jenvman.2012.12.006

in Spain (64 years, Diaz-Delgado and Pons, 2001) while the Aix-Marseille area had a longer NFR value than that of the similar region of Catalonia (Spain, 133 years, Diaz-Delgado and Pons, 2001) but much lower than those of some areas or Portugal (maximum 762 years, Oliveira et al., 2011). Indeed, the authors have shown that the fire rotation period can range from 31 to 762 years according to the regions of Portugal.

Low age-dependency for most fuels of Provence suggested interplay between weather and fuels, according to Moritz et al. (2004a). Most fires occur during the dry and windy summer period (source: Prométhée database), indicating a harsh summer drought as a prominent driver of fires. The assumption that fuel age could limit reburning rests on the fact that fuel would be limited in a first step (i.e. insufficient to carry a new fire) then it would accumulate more or less continuously over time. Most of our fuels have short 'fuel limitation' stage: shrubs and grasses replenish in less than 15-20 years even in the understory of pine or oak forests. Fire can thus ignite and spread in many fuels few years after a fire. An implication of these findings is that fuel management may not be sufficient to prevent fire recurrence, although it can reduce fire intensity and rate or spread, thus fire severity. Time-since-fire is not the predominant driver of new fires and cannot be the pivotal criterion for fuel management on landscape scale. This is confirmed by our flammability experiments and fire modelling exercises indicating that fire risk increases after fire until 20-30 years but tends to flatten out then to decrease in many forest fuels (Curt *et al.*, 2011; Schaffhauser *et al.*, 2011). Some studies have stated higher age-dependency of fire hazard for similar fuel types: Fernandes *et al*. (2010) have shown that *Erica* or *Ulex* shrublands and *Pinus pinaster* or *Eucalyptus globulus* forests of Portugal exhibit extreme fire frequency for Western Europe with Weibull's *c* ranging from 2.6 to 4.8. They suggested that fires are weather-independent and that fuel management may be a priority to limit future fires. Such differences with our study can be explained by the fact that Portugal is the most fire-prone country in Europe whereas burned areas tend to decrease in France since the 1990's. Oliveira et al. (2011) found lower age dependency in regions of Portugal with high portions of shrublands, and higher age dependency for extensively forested regions. This difference with our results can result from the fact that areas with shrublands in Portugal are regularly burned then grazed and have small fires.

4.2 Methodological considerations

As our georeferenced fire database was unique for France, we tested the extent to which FRI and the Weibull parameters could change when censoring the fire data. Moritz *et al*. (2009) have explored the methodological limits and the sensitivity of the Weibull analysis to fire data, and stated that MEI and the *b* and *c* parameters are notably highly sensitive to censoring in fire intervals distribution. Our study confirmed that censoring data has strong effect on parameters estimates, in line with Oliveira *et al*. (2011) for Portuguese data. First, censored data tend to flatten out the Weibull density function and decrease the *c* parameter, thus emphasizing the age-independency of fire hazard for most fuels (as stated by Moritz *et al*. 2009; Oliveira *et al*., 2011). Secondly, censored data revealed differences between the two areas, with stronger differences among fuels in the Aix-Marseille area. Although censored data are more restrictive, they provide calculations on time intervals really observed and thus provide a sound basis for the management of ecosystems. A typical feature of fires in Provence, especially in the Maures massif, is the existence of large fires burning frequently the same areas and thus generated similar time intervals for most fuels (see *In* Johnson and Gutsell 1994). This should increase because of the recent tendency to rare but large fires during exceptional conditions such as in 1990 and 2003 (Fig. 2).

4.4 Conclusions

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doi : 10.1016/j.jenvman.2012.12.006

In the study we have shown that two neighbouring areas with similar weather and roughly similar fuel types can have different fire intervals due to size, shape and connectivity of fuels in landscape, and the fire-fuel feedback. In the Maures massif, large fires propagate throughout extents of wildland and favour the expansion of maquis, which, in turn, favour new fires in the same location. This process has undoubtedly homogenized fire intervals across the landscape. In a more fragmented landscape with a large proportion of non-flammable fuels such as agriculture lands and orchards (Aix-Marseille area), fires propagate less easily through different fuel types, and fire intervals can be linked to fuel characteristics in a better way. This landscape contrast between the Maures massif and the Aix-Marseille area exist throughout the south-eastern France, where urbanized areas with small-grained forests and wildland (such in AIXM) are side by side with areas dominated by large extents of forest and wildland (such in MAUR). Overall, this suggests that fires in Provence are predominantly triggered off by weather and human activity, and that fuel availability in the landscape is generally sufficient to let new fires propagate for some years (ca. 10 years for most fuels) after a previous event, depending upon the size and connectivity of fuels.

This study has implication for environmental management and fire prevention. First, it showed that a same fuel type may not face the same probability of burning again according to the area studied. Then, we showed that the hazard of burning varies weakly with the age of fuel (except for maquis). Thus, a systematic, intensive and costly management of all fuels would not automatically reduce fire risk and the probability of reburning in Provence, even if it is of interest to reduce fire intensity. The present policy of fire prevention uses intensive fuel management (shrub-clearing) only in areas at high risk, and a systematic surveillance of the most likely points of ignition (WUI, road corridors). When a fire occurs, the fire suppression services practice a rapid, hard-hitting initial attack on all ignitions, regardless of the fuel type and the weather conditions. However, Provence is likely to face large and intense fires (such as those of 1990 and 2003) in the future. These fires occur when the summers are exceptionally dry and when fire suppression capacities are overwhelmed. They paradoxically result from the efficient policy of fire prevention and fire fighting set up during the 1980's, which has favoured fuel accumulation in shrublands and many pine forests (Sande Silva *et al.*, 2010). In this context, the distribution of low-flammable fuel types such as mature oak forests or immature shrublands with low fuel biomass could be crucial to wildland conservation and to limit future fires or fire damage in the context of forecasted weather conditions (Moreira *et al*., 2011).

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Original publication available at http://www.sciencedirect.com/
doi : 10.1016/j.jenvman.2012.12.006

Figures Captions

Figure 1. Maps of burned areas and fire frequency (1960-2010) for: A. AIXM (Aix-Marseille area). B. MAUR (Maures massif). The contour of each study area is drawn in light grey.

Figure 2. Burned area and number of fires larger than 3 ha (1960-2010) for AIXM (Aix-Marseille area, 510,593 ha) and MAUR (Maures massif, 145,686 ha).

Figure 3. Main characteristics of the fuel groups and the two study areas (AIXM: Aix-Marseille area; MAUR: Maures massif). The overstory covering is the percentage of covering by the tree canopy (height of trees > 10 m). The understory covering is the percentage of covering by shrubs (height < 1 m). The fuel bed depth is the height of surface fuels contained in the combustion zone of a spreading fire front, including notably low branches, shrubs, and grasses. The 1-h fuel load is the dry biomass of fine fuels (diameter < 6 mm). Large grey bars are mean values, and small black bars are 95% confidence intervals. Different letter for a same variable but different study indicate a statistically different difference using the Tukey HSD test (with $\alpha < 0.05$). Sample size is 8 (SHRUB), 9 (PINE), 12 (MIXED) and 10 (OAK) in AIXM, and 31 (SHRUB), 20 (PINE), 19 (MIXED) and 32 (OAK) in MAUR

Figure 4. Fuel contagion index for AIXM (Aix-Marseille area) and MAUR (Maures massif). The contagion index has been computed for shrublands, forests, and the whole landscape within each area. High values (close to 100) indicate that fires may theoretically propagate easily whereas low values (close to 0) indicate low probability of fire propagation

Figure 5. Hazard of burning for the four main fuel groups in AIXM (Aix-Marseille area) and MAUR (Maures massif). Values were computed using the Weibull model with censored values. For complete data see Appendices 2A and 2B

Figure 1. A

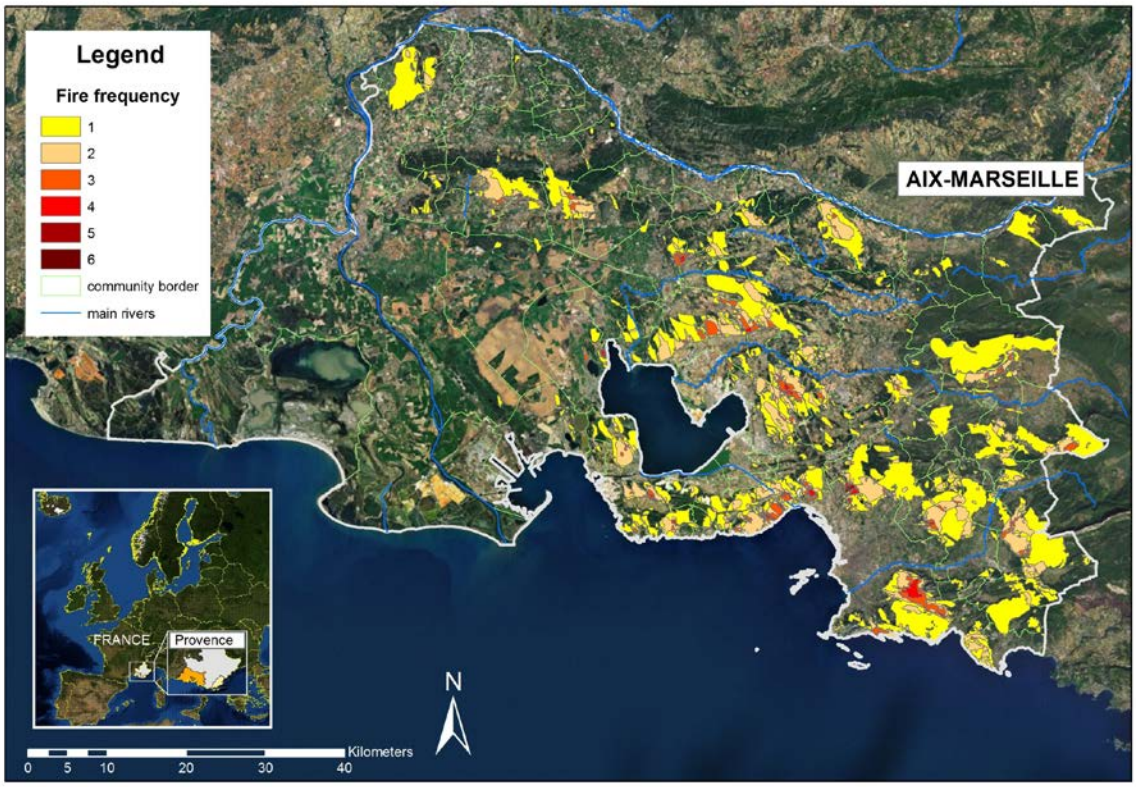

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Figure 1. B

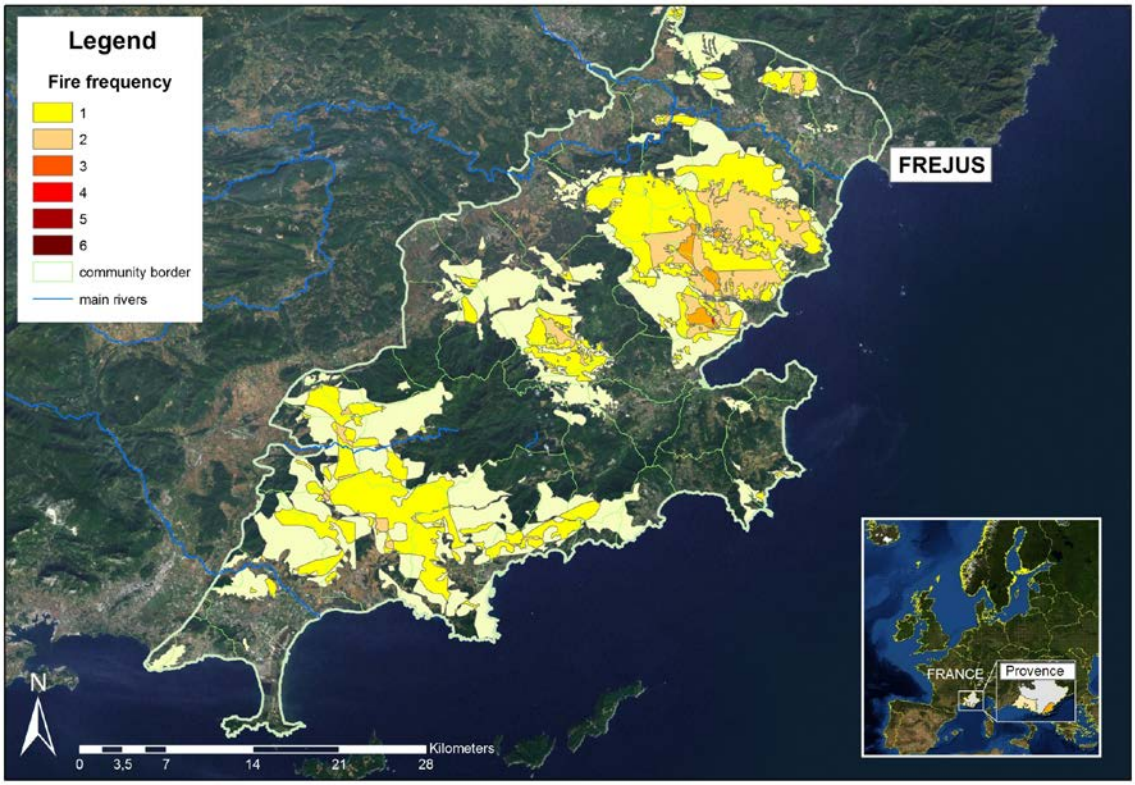

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Figure 2

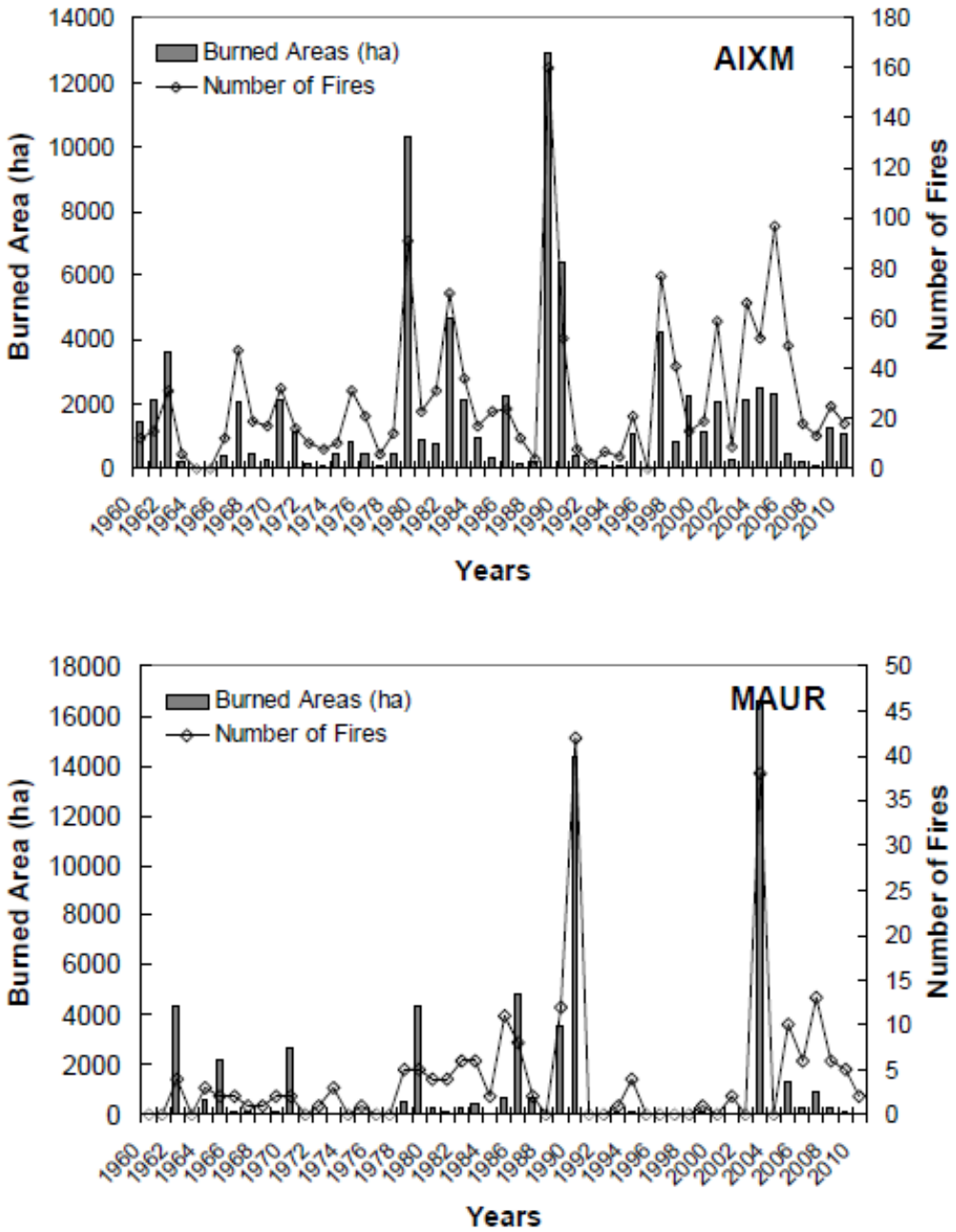

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Figure 3

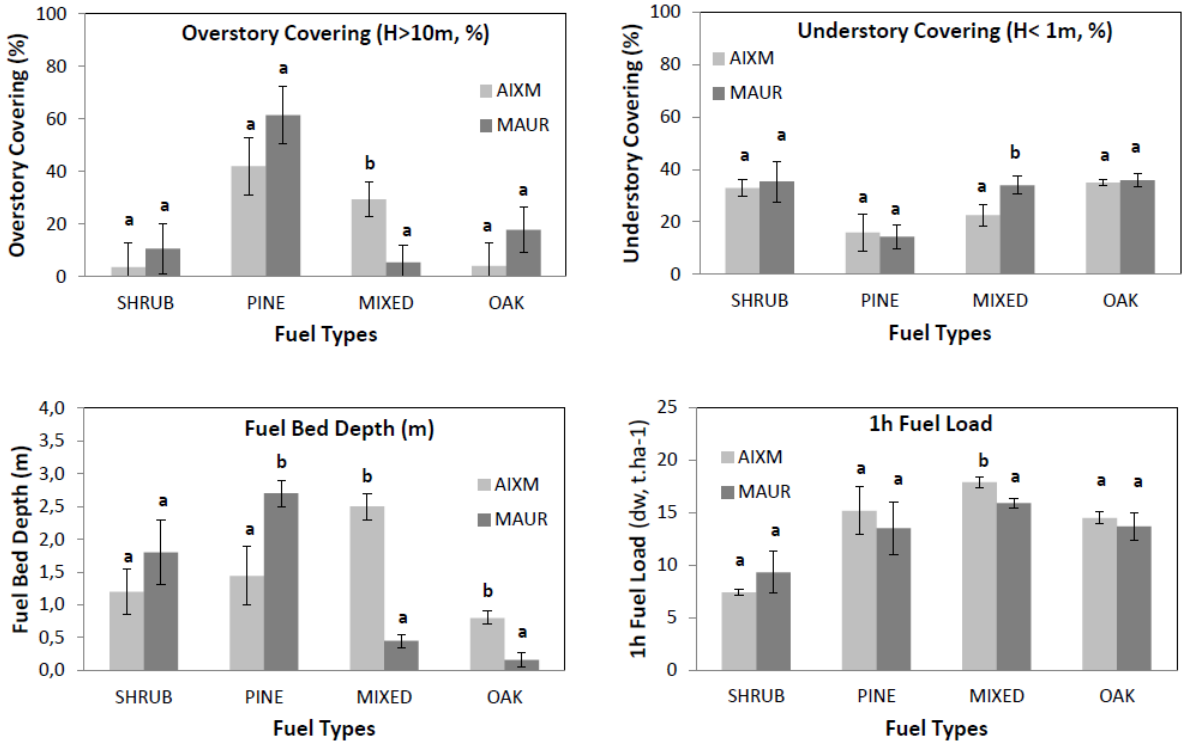

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Figure 4

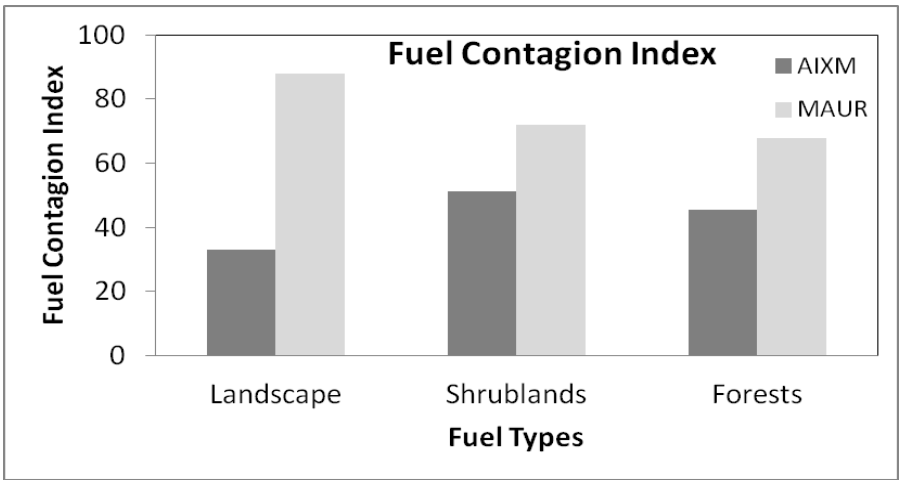

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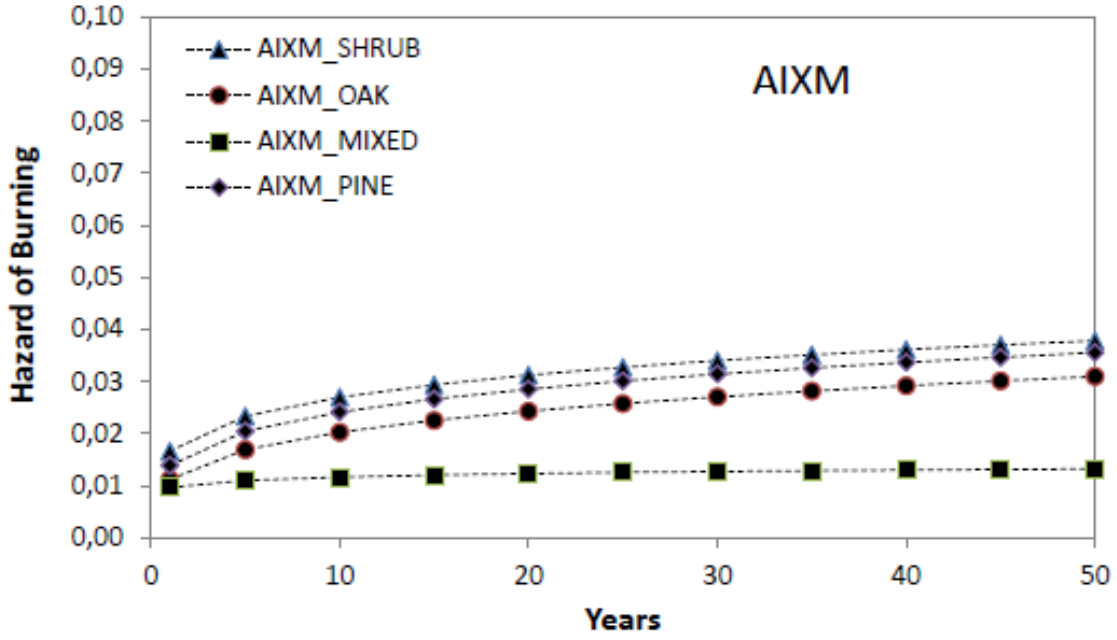
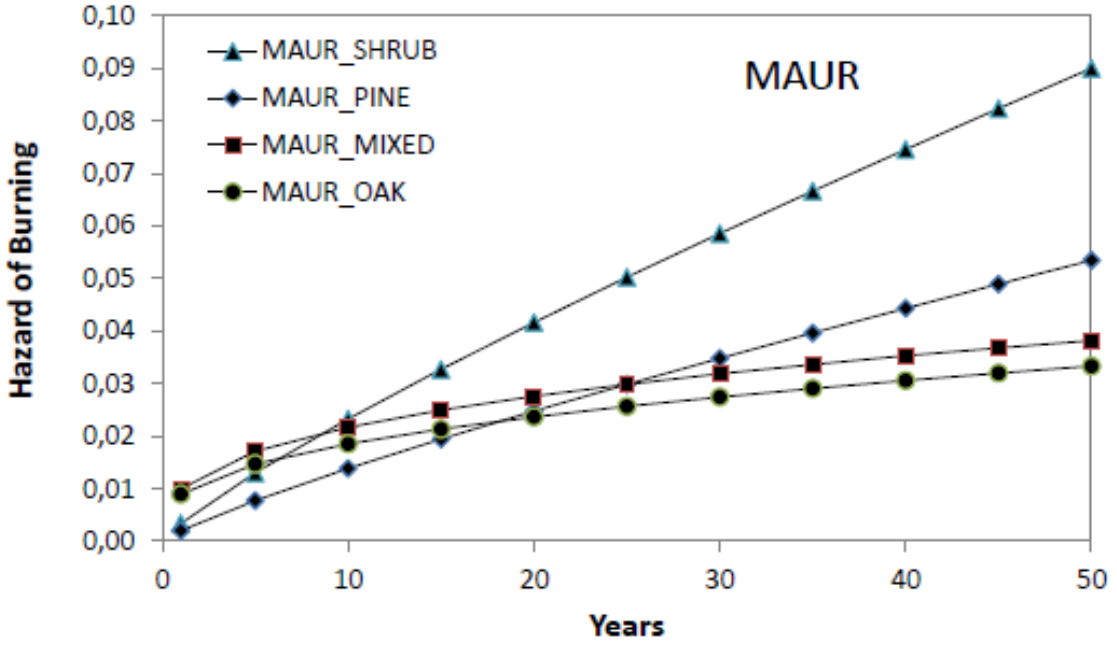
Figure 5





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Original publication available at http://www.sciencedirect.com/
doi : 10.1016/j.jenvman.2012.12.006

Table 1. Main spatial characteristics of fires and fuel types in the study areas, for fires larger than 3 ha (1960-2010). Different letters indicate statistically different values (Mann-Whitney test ; Wilcoxon, P < 0.05)

Fire Characteristics	Aix-Marseille Area (AIXM)	Maures Area (MAUR)
Fire Patterns		
Number of fires	1,451	217
Fire size (mean ± SE, ha)	56 ± 200 *a*	292 ± 717 *b*
Number of fires per a year (mean ± SE)	28 ± 30 *b*	4 ± 8 *a*
Area burned annually (mean ± SE, ha)	1,592 ± 2,452 *a*	1,177 ± 3,170 *a*
Percent area burned annually (%)	0.31	0.81
Cumulated area burned (ha)	104,171	112,377
Total extension of the area burned (ha)	78,860	59,606
Percentage area burned at least once (%)[§]	14.7	40.9
Fraction burned 1, 2, 3, 4 and 5 times *	71.8, 22.6, 4.5, 1.0, 0.1	60.0, 30.5, 8.7, 0.9, 0.002
Fire Cycle or Natural Fire Rotation (years)	235	62
Fuel and Landscape Patterns		
Size of the study area (ha)	510,593	145,686
Number of fuel patches	9,594	2,966
Mean fuel patch size (ha)	55 ± 971 *a*	49 ± 402 *a*
Area Weighted Mean Shape Index	7 *b*	5 *a*
Mean perimeter/area ratio per fuel patch	208 *b*	169 *a*
Mean shrubland patch size (ha)	21 ± 39 *a*	50 ± 27 *b*
Mean forest patch size (ha)	26 ± 99 *a*	37 ± 76 *a*
Mesh (shrublands)	315	2133
Mesh (forests)	283	413
Split (shrublands)	2890	165
Split (forests)	3499	867
Fuel contagion index (whole landscape)	33	88
Fuel contagion index (shrublands)	51.3	71.9
Fuel contagion index (forests)	45.3	67.9

[§] This percentage was calculated on the total area burned
*The fractions were computed among the area burned at least once during the 1960-2010 period

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Original publication available at http://www.sciencedirect.com/
doi : 10.1016/j.jenvman.2012.12.006

Appendix 1A. Main characteristics of the fuel types for the Provence area (AIXM, limestone-derived soils).

Vegetation Type	Dominant Species	Overstory Covering [§] (H>10 m; %)	Understory Covering [§] (H< 1m, %)	Fuel Bed Depth (m)	1h Fuel Load (dw, $t.ha^{-1}$)
Garrigues	*Quercus coccifera, Juniperus oxycedrus, Ulex parviflorus, Cistus* spp*., Rosmarinus officinalis, Brachypodium retusum*	4.0	35.0	1.2	7.4
Pine forests	*Pinus halepensis, Phyllirea angustifolia, Quercus coccifera*	44.0	33.0	1.0	16.4
Mixed pine-oak forests	*Pinus halepensis, Quercus pubescens, Quercus ilex*	35.0	23.0	2.5	17.9
Mixed oak forests	*Quercus pubescens, Quercus ilex, Ulex parviflorus*	42.0	10.0	0.8	15.5
Spontaneous afforestation	*Pinus halepensis*	13.0	26.0	3.0	6.7
Sparse woodlands	*Pinus halepensis, Quercus* spp*., Rosaceae*	12.0	33.0	1.5	7.0
Vegetation WUI	All species (pines, oaks) with shrub-cleared understory	78.0	8.0	1.2	12.8
Road corridors	Graminoids and dicots, *Quercus coccifera*	0.0	13.0	0.5	8.0

[§] Standard errors were not computed for covering values

Appendix 1B. Main characteristics of the fuel types for the Provence area (MAUR, siliceous soils).

Vegetation Type	Dominant Species	Overstory Covering (H>10 m; %)	Understory Covering (H< 1m) (%)	Fuel Bed Depth (m)	1-h Fuel Load (dw, $t.ha^{-1}$)
Maquis	*Erica arborea, Cistus* spp*., Calycotome spinosa*	0.5 ± 1.0	34.7 ± 6.1	0.8	7.5
Maquis with cork oak	*Quercus suber, Arbutus unedo, Erica arborea, Cistus* spp*., Calycotome spinosa*	27.0 ± 5.0	37.0 ± 10.1	1.8	9.0
Maquis with mixed oak woodlands	*Calycotome spinosa, Cistus* spp*., Quercus suber, Q. ilex, Q. pubescens,*	26.0 ± 4.2	36.1 ± 9.0	2.8	11.5
Maquis with pine	*Pinus pinaster, Cistus* spp*., Calycotome spinosa*	24.0 ± 5.3	34.0 ± 10.4	2.7	11.5
Cork oak forests	*Quercus suber, Brachypodium retusum, Calycotome spinosa*	49.0 ± 5.2	18.3 ± 6.0	0.3	14.6
Mixed pine-oak forests	*Pinus pinaster, Quercus ilex, Q. pubescens, Q. suber*	70.6 ± 6.5	12.4 ± 2.3	0.1	15.9
Mixed oak forests	*Quercus ilex, Q. pubescens, Q. suber, Brachypodium retusum*	75 ± 2.3	8.3 ± 3.3	0.1	16.0

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Original publication available at http://www.sciencedirect.com/
doi : 10.1016/j.jenvman.2012.12.006

Chestnut forests	*Castenea sativa, Brachypodium retusum*	70.1 ± 4.0	4.5 ± 4.0	0.1	13.5

Appendix 2A. Weibull model parameters *a* and *b* for the hazard of burning function, mean fire intervals (FRI), and median fire-free intervals (MEI) for the fuel types of the Aix-Marseille area (AIXM, limestone-derived soils). Different letters for mean fire intervals indicate statistically significant differences (P < 0.001)

Fuel Types	Area (ha)	Mean fire interval FRI (mean ± SE)	Uncensored			Censored		
			MEI (years)	*b*	*c*	MEI (years)	*b*	*c*
Garrigues	50,076	18.3 ± 12.2 a	17.2 a	20.2	1.87	23.5 a	30.5	1.21
Pine forests	13,941	19.6 ± 12.9 ab	17.5 a	21.8	1.53	34.8 b	45.9	1.24
Mixed pine-oak forests	12,195	23.6 ± 12.2 bc	22.6 b	26.5	1.99	56.2 c	78.5	1.08
Mixed oak forests	10,521	24.4 ± 11.8 bc	21.9 ab	25.2	1.94	62.2 d	85.4	1.08
Spontaneous afforestation	8,242	19.9 ± 11.2 ab	19.5 b	22.3	1.81	33.2 b	43.8	1.28
Sparse woodlands	5,101	20.7 ± 12.4 ab	19.3 b	23.2	1.70	29.9 b	37.4	1.24
Vegetation WUI	3,517	20.8 ± 12.5 ab	19.2 b	23.3	1.67	52.5 c	70.4	1.25
Road corridors	578	22.2 ± 14.4 b	19.5 b	24.5	1.49	30.4 b	42.1	1.36
Total Area	104,171	20.3 ± 12.6	18.2	22.6	1.61	33.3	44.5	1.20

Appendix 2B. Weibull model parameters *a* and *b* for the hazard of burning function, mean fire intervals (FRI), and median fire-free intervals (MEI) for the fuel types of the Maures massif (MAUR, siliceous soils). Different letters for mean fire intervals indicate statistically significant differences (P < 0.001)

Fuel Types	Area (ha)	Mean fire interval FRI (mean ± SE)	Uncensored			Censored		
			MEI (years)	*b*	*c*	MEI (years)	*b*	*c*
Maquis	18,048	22.2 ± 10.5 a	20.2 a	24.8	2.12	21.8 a	26.8	1.88
Maquis with cork oak	28,599	21.4 ± 11.4 a	19.5 a	24.0	1.88	28.5 b	34.7	1.79
Maquis with mixed oak woodlands	2,971	22.2 ± 16.5 a	20.4 a	26.0	1.54	22.8 a	27.8	1.83
Maquis with pine	7,941	24.0 ± 11.5 ab	23.3 b	27.2	2.23	32.0 b	38.7	1.84
Cork oak forests	27,961	28.8 ± 8.5 c	25.5 c	21.1	1.83	35.0 c	28.8	1.37
Mixed pine-oak forests	15,573	28.0 ± 8.2 c	25.1 c	26.3	1.65	29.9 b	38.5	1.27

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Original publication available at http://www.sciencedirect.com/
doi : 10.1016/j.jenvman.2012.12.006

Mixed oak forests	8,941	27.4 ± 14.7 c	24.8 bc	23.8	1.57	28.9 b	37.9	1.19
Chestnut forests	940	31.3 ± 11.5 d	23.1 b	24.1	1.26	25.0 ab	32.3	1.23
Vegetation WUI	1,403	27.2 ± 15.9 c	24.4 bc	23.9	1.41	24.1 ab	30.7	1.29
Total Area	112,377	22.2 ± 12.3	20.2	24.9	1.82	30.0	38.1	1.53

Author-produced version of the article published in Journal of Environmental Management, 2013, 117, 150-161
Original publication available at http://www.sciencedirect.com/
doi : 10.1016/j.jenvman.2012.12.006

Acknowledgments

We greatly acknowledge colleagues from Cemagref, National Forest Service and Regional Forest Service (Provence Alpes Côte-d'Azur Region) for their help building the fire database.

Author-produced version of the article published in Journal of Environmental Management, 2013, 117, 150-161
Original publication available at http://www.sciencedirect.com/
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