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ORIGINAL PAPER

Vulnerability of forest ecosystems to fire in the French Alps

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Abstract
Forest fires are expected to be more frequent and more intense with climate change, including in temperate and mountain forest ecosystems. In the Alps, forest vulnerability to fire resulting from interactions between climate, fuel types, vegetation structure and tree resistance to fire is little understood. This paper aims at identifying trends in the vulnerability of Alpine forest ecosystems to fire at different scales (tree species, stand level and biogeographic level) and according to three different climatic conditions (cold season, average summer and extremely dry summer). To explore Alpine forest vulnerability to fire, we used surface fuel measurements, forest inventory and fire weather data to simulate fire behaviour and ultimately post-fire tree mortality across 4438 forest plots in the French Alps. The results showed that cold season fires (about 50% of the fires in the French Alps) have a limited impact except on low-elevation forests of the Southern Alps (mainly Oak, Scots pine). In average summer conditions, mixed and broadleaved forests of low elevations suffer the highest mortality rates (up to 75% in coppices). Finally, summer fires occurring in extremely dry conditions promote high mortality across all forest communities. Lowest mortality rates were observed in high forest stands composed of tree species presenting adaptation to surface fires (e.g. thick bark, high canopy) such as Larch forests of the internal Alps. This study provides insights on the vulnerability of the main tree species and forest ecosystems of the French Alps useful for the adaptation of forest management practices to climate changes.

Keywords Mountain forest · Vulnerability · Forest fire · Tree mortality · Climate change · Alps

Introduction

Mountain forests provide a wide range of ecosystem services (ES) such as renewable wood resources, biodiversity or protection of human beings and infrastructures against natural hazards (Briner et al. 2013). However, this ES provisioning might be disturbed by climate changes which are especially perceptible in mountain areas (Pachauri et al. 2014; Kohler et al. 2010; Beniston 2005). Amongst climate changes impacts, the increase in forest fire risk has been demonstrated in European mountains (Bedia et al. 2014; Wastl et al. 2012). In the French Alps, longer fire seasons and more frequent extreme drought episodes have been observed since the 1960s (Dupire et al. 2017). Although wildfires have always been present in the Alpine area (Power et al. 2008; Wick and Möhl 2006; Tinner et al. 1999), their potential impacts on the current forests and tree species are little known. Vulnerability to fire (Miller and Ager 2013) of forest communities results from complex interactions between climate, fuel availability (fuel load and moisture content of the vegetation) and ignition sources (Moritz et al. 2011). Investigating these interactions is a prerequisite for understanding the vulnerability of species and forest ecosystems to fire and anticipating the effects of global changes. In particular, it is unknown how mountain-range variability in fuel moisture and fire ignition (Zumbrunnen et al. 2012), vegetation flammability (Fréjaville et al. 2016), fire weather

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(Dupire et al. 2017) and fire resistance of trees (Fréjaville et al. 2018b) interact to shape broadscale patterns in Alpine forest vulnerability to fire.

In the French Alps, the number of fires is equally distributed between cold season (November–April) and summer but 65% of the burnt area results from summer fires (Dupire et al. 2017). Surface fires, which burn and propagate only in the surface fuels (litter, herbs and shrubs), are the most common fire types (Conedera et al. 2018; Valese et al. 2014). Depending on their intensity, surface fires can affect different tree components: mainly tree bole in low-intensity fires, bole and crown in moderate- to high-intensity fires (Dickinson and Johnson 2001). Roots can also be damaged if the residence time of the fire is long enough. Crown fires have a low occurrence in Alpine forests (Curt et al. 2016) and are generally concentrated in the southernmost part. Tree mortality following forest fires results from direct (damaged meristems due to heat transfers) or indirect (insect attack, altered physiology, etc.) processes (Michaletz and Johnson 2007). In practice, the different methods to assess post-fire tree mortality consist in estimating whether the lethal temperature of the different tree components is reached according to the fire behaviour and the tree characteristics (Dickinson and Johnson 2004; Gutsell and Johnson 1996; Brown and DeByle 1987; Peterson and Ryan 1986). Generally, the critical threshold is fixed at 60 °C lasting for 60 s at the meristem level. This lethal threshold can theoretically be reached for any fire intensity if the flame residence time is long enough (Bauer et al. 2010). Thus, long flame residence times are needed in low-intensity fires in order to damage the vegetative tissues while shorter time lapse can produce the same damages with high-intensity fires (Bova and Dickinson 2005).

Several studies contribute to the understanding of resistance or resilience strategies to fire of different Alpine species. Amongst them, Maringer et al. (2016b) investigated direct and indirect mortality in beech forests, Fréjaville et al. (2013) focused on the ecological traits affecting the resistance to fire of the main subalpine species, Moris et al. (2017) studied the resilience of European Larch and Tinner et al. (2005) followed tree species responses to fires during the Holocene. Beside their valuable inputs, these studies are not sufficient to draw an overview of the vulnerability to fire of the main Alpine forest ecosystems due to the high climatic and biogeographic variability in the area. Furthermore, unlike other regions of the world, no method has been proposed to estimate post-fire mortality of Alpine species and ecosystems.

The objective of our study was to identify the ranges in vulnerability to fire of the main tree species and forest ecosystems of the French Alps under three different climatic conditions (cold season, average summer and extreme summer). To explore Alpine forest vulnerability to fire, we used surface fuel measurements, forest inventory and fire weather data to simulate fire behaviour and ultimately post-fire tree mortality. Due to the scarcity of post-fire field observations covering the study area, a modelling approach was performed in three consecutive steps on 4438 National Forest Inventory (NFI) plots located across the French Alps. First, fire simulations were conducted using the fire behaviour program FlamMap (Finney 2006) which has been extensively used worldwide for simulating fires, both in Mediterranean environments (Mitsopoulos et al. 2016) or in mountain environments (Ager et al. 2011). Second, different mortality equations were used to estimate single tree cambial mortality from fire intensity and residence time. Van Wagner's (1977) crown fire transition criteria were used to assess crown fire transition. The range of predicted mortality was compared to observed mortality reported in previous studies and field observations. Finally, post-fire mortality was used as metric to assess the vulnerability to fire of the main tree species, forest types (stand level) and biogeographic areas of the French Alps.

Material and methods

Study area

This study focuses on the French Alpine area (Fig. 1) as defined by the French Institutions (Decree no. 2016-1208). Located at the South-East of France, it represents about 40,600 km^2 with an elevation ranging from 15 m to 4809 m (Mont Blanc). The territory was divided into four biogeographic zones corresponding to Northern and Southern Alps and low elevation (≤ 800 m a.s.l.) and high elevation (> 800 m a.s.l.) areas (Dupire et al. 2017).

Table 1 shows information on each biogeographic zone regarding fire regime and forest cover. Northern Alps are characterized by a mountain climate with degraded oceanic influences (homogeneous rainfall distribution along the year). Low elevations are covered by coppices and mixed stands dominated by broadleaved species such as deciduous *Quercus* sp., *Tilia* sp., *Acer* sp., *Castanea sativa* and *Carpinus betulus*. Forests of the high elevations are mainly mixed stands and pure coniferous stands with predominance of *Picea abies*, *Abies alba* and *Fagus sylvatica*. Southern Alps are marked by a mountain climate with a strong Mediterranean influence (dry summer and wet autumn and winter). Low elevations are mainly covered by puny coppices sometimes mixed with pines and pure coniferous stands also dominated by *Pinus sylvestris* and *Pinus nigra*. Forests of high elevations are mainly pure coniferous stands of *Pinus* sp. and *Larix decidua*.

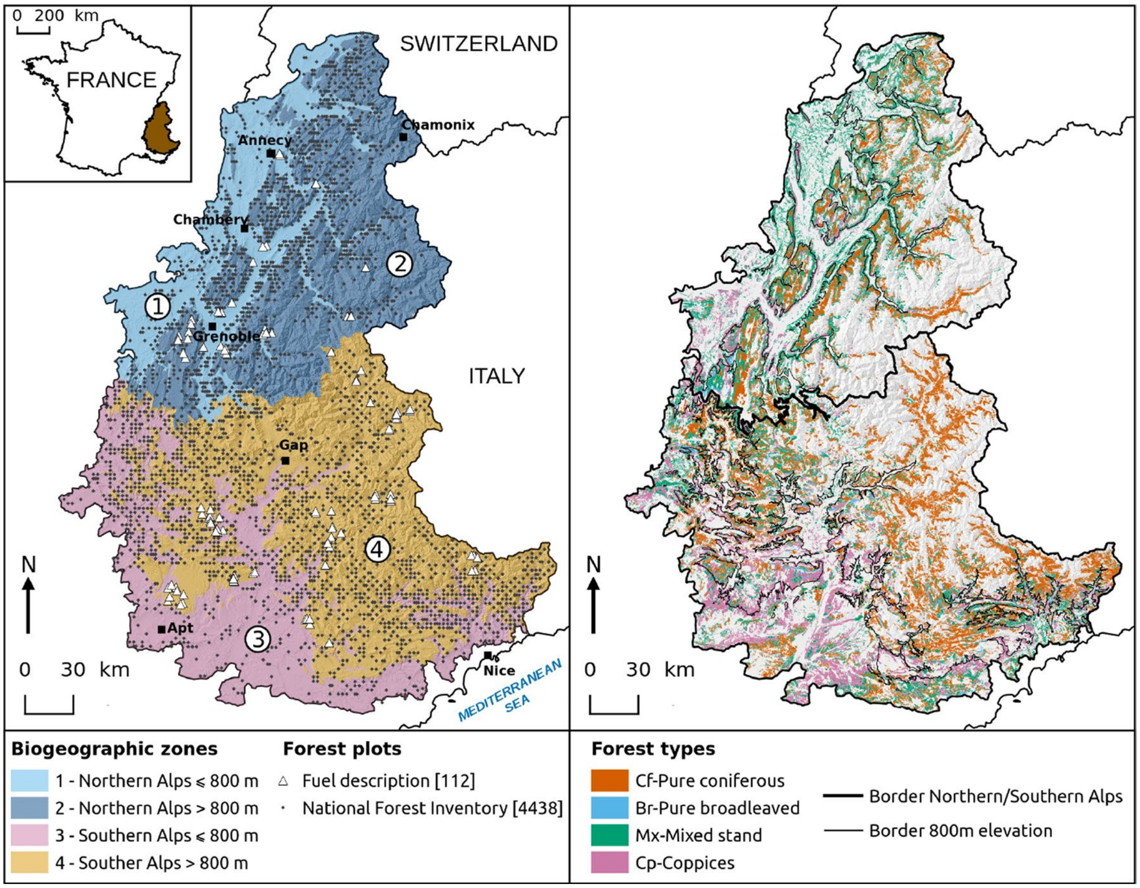


Fig. 1 Map of the French Alpine area and locations of the forest plots from the different sources. The four biogeographic zones used for analysing the results are also shown. The forest area divided into four forest types is displayed on the right map

Table 1 Total area, proportion of forested area and fire regime information of the four biogeographic zones used in the study

Biogeographic zone	Area (km^2)	Proportion of forest (%)	Number of fires ($z > 100$ ha) (fires year^{-1})	Burned area (% forest area) (km^2 year^{-1})
1. Northern Alps ≤ 800 m	5850	46	21.1 (0.13)	0.6 (0.02)
2. Northern Alps > 800 m	10,800	49	5.3 (0)	0.2 (< 0.01)
3. Southern Alps ≤ 800 m	8975	71	113.7 (0.54)	8.0 (0.13)
4. Southern Alps > 800 m	15,035	66	44.9 (0.25)	4.1 (0.04)

Annual values of number of fires and burned area refer to the period 2002–2015

Fuel parameters and fire resistance traits

To explore the broadscale variability in vegetation flammability (Fréjaville et al. 2016), we first carried out an extensive sampling of surface fuel parameters across the main forest ecosystems of the French Alps. Measurements were taken in late spring on 96 plots (fuel plots) covering the French Alps (Fréjaville et al. 2016) plus 16 complementary plots located only in the Northernmost part (2017 field campaign). Fuel measurement was based on standard procedures (Bessie and Johnson 1995) following the methods of McRae et al. (1979) and Brown et al. (1982) along a 20-m equilateral triangle. On each fuel plot, measurements concerned the canopy and surface fuel covers, the duff bulk density, the height and load of litter, herbs and shrubs and the surface area-to-volume ratios (SAV) of the different fuel compartments (Fréjaville et al. 2016). Supplementary Material A shows the distributions of fuel loads and SAV for the different fuel compartments.

To quantify fire resistance of trees, we measured bark thickness, tree height and canopy base height on dominant species for a range of tree diameters (Fréjaville et al. 2018b).

In addition, we used individual tree data (species, total and first branch heights) of the French National Forest Inventory (NFI, 2005-2015) that were collected on trees with a diameter at breast height (DBH) greater than or equal to 7.5 cm in 4438 forest plots. NFI data are based on a systematic grid of 1 km × 1 km across the country. Each measurement is collected on circular plots with a radius of 25 m (Robert et al. 2010). Bark thickness of the NFI trees was estimated from allometric equations according to DBH and tree species (Fréjaville et al. 2018b).

Fire weather scenarios

To assess the broadscale effect of fire weather and climatic conditions on vegetation flammability, we used climatic data from the SAFRAN analysis system implemented by Météo France (Vidal et al. 2010; Quintana-Segui et al. 2008) to calculate the daily fine fuel moisture code (*FFMC*) over the French Alps from 1959 to 2015 for each of the 112 fuel plots and the 4438 NFI plots (see Dupire et al. (2017) for more information). *FFMC* is one of the six meteorological-based indices of the Canadian Fire Weather Danger Rating System (Van Wagner 1987). It rates the moisture of live fine fuels including grass, fine shrubs and the surface litter (5–10 cm depth) and therefore evaluates the ease of ignition of these light fuels. *FFMC* can be related to the 10-hr fuel moisture content (*mc*, % of dry mass) of the US National Fire Danger Rating System (see Eq. 1, Wotton (2009)) which is a key input of many fire simulation models.

$$mc = 147.2 \times \frac{101 - FFMC}{59.5 + FFMC} \quad (1)$$

Three fire weather scenarios corresponding to different climatic conditions have been considered in this study (Table 2). The first corresponds to the cold season (November–April) where 50% of the vegetation fires occur in the French Alps (Dupire et al. 2017). Those low-intensity surface fires are mainly due to human activities (agriculture and forest cleaning). The second corresponds to the summer months (June–September) where the most intense wildfires are observed. Finally, the values observed during the summer 2003 were used to account for extreme climatic conditions (Poumadère et al. 2005). The 95th percentile of *FFMC* values was used for each climatic condition in order to focus on the most severe conditions that drive fire activity in mountain areas (Fréjaville and Curt 2015). If fine fuel moisture contents were derived from *FFMC* values, *mc* of live herb and live woody compartments were determined according to climatic conditions from field measures carried out on three fuel plots located in the Grenoble area along the year 2016.

Although fuel loads (and SAV) may vary according to the season (i.e. between winter and summer), especially in forests dominated by broadleaved species, we consider similar fuel parameters (load and SAV) for all climatic conditions because data about seasonal fuel variation were not available in the study area. We thus assume that fire behaviour variation amongst seasons is mostly driven by climate and fire weather conditions (Bessie and Johnson 1995).

Simulate fire behaviour across Alpine forests

Figure 2 shows the general workflow followed for the simulation of the fire behaviour. Fire simulations were first computed on the 112 fuel plots using the program FlamMap 5.0 (Finney 2006). Then, fire behaviour on the NFI plots was estimated from the outputs of the fire simulations using a k-NN algorithm.

Fire simulations on the 112 fuel plots

To assess the effects of vegetation flammability and climatic conditions on the broadscale variation of fire behaviour, we used the program FlamMap version 5.0 (Finney 2006) which simulates fire behaviour and propagation on defined landscape based on topography, fuel characteristics and weather data. Potential fire behaviour calculations include surface fire spread (Rothermel et al. 1972), crown fire initiation (Van Wagner 1987) and crown fire spread (Rothermel 1991).

In this study, FlamMap was used on a virtual landscape in order to perform fire simulation simultaneously on the 112 fuel plots. To do so, each one of the 112 plots has

Table 2 Description of the fire weather scenarios and their associated *FFMC*. $P^y_{95}(FFMC_{Months})$ is the annual (year *y*) 95th percentile of daily FFMC values for the months considered. The ranges of fuel moisture contents (*mc*) and wind speed used for fire behaviour simulations for each climatic condition are also given

Fire weather scenario	Average summer	Extreme summer	Average cold season
Months	June–September	June–September	November–April
Years	1959–2015	2003	1959–2015
Corresponding FFMC	$\frac{\sum_{y=1959}^{2015} P^y_{95}(FFMC_{Ju-Se})}{57}$	$P^{2003}_{95}(FFMC_{Ju-Se})$	$\frac{\sum_{y=1959}^{2015} P^y_{95}(FFMC_{No-Ap})}{57}$
10-hr dead *mc*	5–14%	4–13%	9–17%
Live herb *mc*	60%	40%	90%
Live woody *mc*	90%	70%	120%
Wind speed	0, 3, 8, 14, 20 km h^{-1}	0, 3, 8, 14, 20 km h^{-1}	0, 3, 8, 14, 20 km h^{-1}

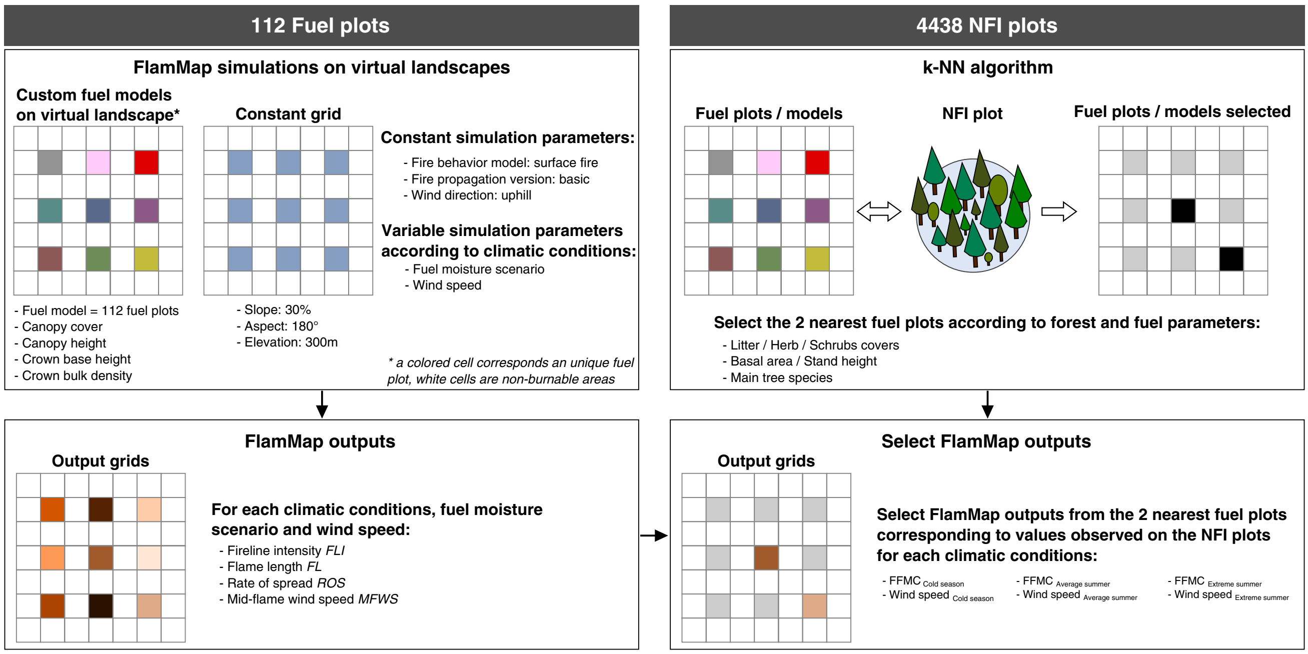


Fig. 2 Workflow chart of the fire modelling. The program FlamMap 5.0 (Finney 2006) was used for the fire simulations. The fireline intensity (FLI), flame length (FL), rate of spread (ROS) and mid-flame wind speed (MFWS) were extracted for each fire simulation

been defined as a custom fuel model with the fuel characteristics measured on the field. These custom fuel models were then dispatched in virtual landscapes on a regular grid so that each fuel model is surrounded by non-burnable areas. The canopy total and base heights, canopy cover and canopy bulk density measured on the fuel plots were dispatched on the same regular grid in order to match their associated fuel model. In order to focus on the effect of fuel characteristics and climatic conditions on fire behaviour, the input grids of slope, elevation and aspect were constant for all fuel models. The slope of the terrain was set to 30% which corresponds to the minimum slope of the NFI plots and the upper bound of Rothermel model calibration (Rothermel et al. 1972). Elevation was set to 300 m and aspect to 180°. All simulations used the surface fire behaviour model with a basic fire propagation version. For conservative matter, the wind direction was set to blow uphill.

Finally, for each fuel plot (or custom fuel model), 145 fire simulations are performed: 45 for the cold season (9 different *mc* × 5 wind speeds), 50 for average summer (10 different *mc* × 5 wind speeds) and 50 for extreme summer (10 different *mc* × 5 wind speeds). 10-hr dead fuel moisture contents (*mc*) were obtained from Eq. 1 using the range of *FFMC* values observed on both NFI and fuel plots for each climatic condition. Moisture contents of other dead fuel classes were assumed to be *mc* − 1 for 1 hr and *mc* + 1 for 100 hr as recommended in Finney (2006). Live herb and live woody moisture contents were fixed for each climatic condition according to fuel collection around the Grenoble region along the year 2016 (see Table 2).

FlamMap computes four metrics that were used to estimate the vulnerability to fire of forest ecosystems of the French Alps. The first is the fireline intensity (*FLI*, $\mathrm{kW\,m^{-1}}$ Byram (1959)) which indicates the energy (heat) release per unit time per unit length of the fire front. The second is the flame length (*FL*, m) which accounts for the height of scorching in trees, and the possibility of flame transition to the tree crown. The third is the rate of spread (*ROS*, $\mathrm{min\,m^{-1}}$) or the speed of the fire front. *ROS* can be related to the flame residence time (τ_f) if the width of the burning strip (D) is known: $\tau_f = \frac{D}{ROS}$ (Alexander 1982). Finally, the mid-flame wind speed (*MFWS*, $\mathrm{km\,h^{-1}}$) is used in the calculation of the fraction of crown volume killed.

Fire behaviour on the 4438 NFI plots

Fuel information on the NFI plots was insufficient to perform direct fire simulation with FlamMap. Thus, we used a k-NN algorithm (Mucherino et al. 2009) to estimate the broadscale variability in fire behaviour as a function of vegetation and climatic conditions, using both field measurements of fuel parameters, fire weather observations and NFI data (Fig. 2). Vegetation properties used in the k-NN algorithm were the horizontal covers of surface and understory fuel strata (litters, herbs and shrubs), forest structure (tree basal area and stand height) and composition (main tree species) that largely explain fire intensity variation amongst

Alpine forests (Fréjaville et al. 2018a). For each NFI plot, the outputs of fire simulations of the two fuel plots with the nearest vegetation properties ($k = 2$) and corresponding to its climatic conditions (FFMC, wind speed) were used to assign fire behaviour values across Alpine forests.

No previous work was found in the study area to compare our modelled fire behaviour metrics. Arpaci et al. (2011) is the only reference returning information on maximum fireline intensity calculated with FlamMap under different moisture and wind scenarios for two forest types that can be found in the French Alps (*Pinus sylvestris* and *Castanea sativa* stands). Due to lack of information on the other forest ecosystems, only the results concerning these two forest types were compared.

Assessing vulnerability to fire from post-fire mortality

Cambial mortality

Cambial mortality has been shown to be strongly linked to the insulation capability of the bark (Spalt and Reifsnyder 1962) which mainly depends on its thickness and, to a lesser extent, on its moisture content and density (Bauer et al. 2010). Different approaches were tested to estimate the post-fire cambial mortality on the forest plots.

Cambial mortality from temperature gradient and heat duration

A common method to estimate cambium necrosis was first proposed by Spalt and Reifsnyder (1962) who modelled it as a one-dimensional conduction into a semi-infinite solid (Fourier's law) as shown in Eq. 2:

$$\frac{T_c - T_f}{T_a - T_f} = \mathrm{erfc}\left(\frac{\mathrm{bthi}}{2\sqrt{\alpha \times \tau/60}}\right) \tag{2}$$

T_c: lethal temperature of cambium (°C)
T_f: temperature of the flame (°C)
T_a: air temperature (°C)
erfc: complement of the Gauss' error function
bthi: bark thickness (cm)
α: bark thermal diffusivity (cm^2 min^{-1})
τ: flame residence time (v)

According to this approach, cambium necrosis can be related to a temperature gradient and a heating duration. However, most of the classical fire behaviour models do not return information about temperature and give a rough estimation of the flame residence time. Therefore, flame temperature is generally assumed to be constant over time and equal to the maximum temperature of the flaming fireline. For sample, Peterson and Ryan (1986) derived Eq. 2 by fixing $T_c = 60\,°\mathrm{C}, T_f = 500\,°\mathrm{C}, T_a = 20\,°\mathrm{C}$ and $\alpha = 0.06\,\mathrm{cm}^2\,\mathrm{min}^{-1}$; with these parameters, the equation becomes:

$$t_{c-PR} = 174 \times \mathrm{bthi}^2 \Leftrightarrow \mathrm{bthi}_{c-PR} = \sqrt{\frac{\tau}{174}} \tag{3}$$

where t_{c-PR} (s) is the time required to reach the lethal conditions in the cambium (or critical time) and bthi_{c-PR} is the minimum bark thickness (or critical bark thickness) required to prevent reaching the cambium lethal temperature according to the fixed parameters.

Cambial mortality from bark burning experiments

Bauer et al. (2010) were particularly interested in the insulation capability of the bark under low-intensity fires. They designed an experiment to measure the time needed to reach the lethal cambial temperature (t_{c-Ba}, s) for different European tree species (*Abies alba, Fagus sylvatica, Larix decidua, Pinus sylvestris*) with a constant flame temperature of 214 °C and different bark moisture contents. They produced a general equation that included all the tree species studied for wet and dry bark. Only the equation for the dry bark is presented here:

$$t_{c-Ba} = 9.1 \times \mathrm{bthi}^{1.401} \Leftrightarrow \mathrm{bthi}_{c-Ba} = 0.2068 \times \tau^{0.7138} \tag{4}$$

where bthi_{c-Ba} (mm) is the critical bark thickness required to prevent reaching the cambium lethal temperature according to the authors experiments and a flame residence time τ (s).

Cambial mortality from heat flux

Bova and Dickinson (2005) were particularly interested in linking outputs of fire behaviour models with stem heating and tissue necrosis. They developed equations based on physical relationships between fire behaviour and surface heat flux and between surface heat flux and tissue necrosis. Although only two tree species were studied (*Acer rubrum* and *Quercus prinus*), they produced an equation linking the depth of necrosis (or critical bark thickness: bthi_{c-Bo}, mm) with fireline intensity (*FLI*, kW m^{-1}) and flame residence time (τ, s) independent of the tree species:

$$t_{c-Bova} = 12.57 \times \mathrm{bthi}^{1.56} \times FLI^{-0.31}$$
$$or\ \mathrm{bthi}_{c-Bova} = 0.21 \times FLI^{0.2} \times \tau^{0.64} \tag{5}$$

Crown mortality

Crown mortality was modelled according to Peterson and Ryan (1986) who estimated the fraction of crown killed from tree characteristics (tree height and crown proportion), fireline intensity and wind speed. This method presented the advantage of being directly compatible with the outputs of FlamMap simulation and the available tree parameters.

$$C_k = \frac{(h_k - h_t + c_l) \times (h_t - h_k + c_l)}{c_l^2} \quad (6)$$

with:

$$h_k = \frac{3.94 \times FLI^{1.17}}{(T_c - T_a) \times (0.11 \times FLI + MFWS^3)^{0.5}} \quad (7)$$

C_k: fraction of crown volume killed ($0 \le C_k \le 1$)
h_t: tree height (m)
c_l: crown length (m)
h_k: height of crown kill (m)
$MFWS$: mid-flame wind speed ($\mathrm{m\,s^{-1}}$)

Probability of post-fire mortality of a tree

The probability of post-fire mortality of a tree P_m was assessed by combining the critical time for cambial kill (τ_c) for each method exposed above and the fraction of crown volume killed (C_k) as proposed in Peterson and Ryan (1986):

$$P_m = C_k^{\left(\frac{\tau_c}{\tau_f} - 0.5\right)} \quad (8)$$

P_m: Probability of mortality ($0 \le P_m \le 1$)
C_k: fraction of crown volume killed ($0 \le C_k \le 1$)
τ_c: critical time for cambial kill (s)
τ_f: flame residence time (s)

The probability of mortality strongly depends on the flame residence time which is difficult to get from fire simulations. For this reason, we chose to mark off P_m by calculating it for two τ_f thresholds. The lowest threshold was fixed at 30 seconds which corresponds to a minimum time for a low-intensity surface fire to maintain itself (Peterson and Ryan 1986). The highest threshold corresponded to the theoretical maximum duration of lethal bole heating defined in Peterson and Ryan (1986) which depends on the moisture content and the loads of the different fuel size classes (1 h, 10 h, 100 h). For our data, it ranged from 1 min to 5 min depending on the fuel bed loading and moisture content (see Supplementary Material B.1).

Post-fire tree mortality of the main tree species was calculated at the French Alps scale with Eqs. 3 to 5 by averaging simulated P_m values across NFI plots. The modelled post-fire mortality rates were compared, when available, to observed data taken in the literature or to field observations in order to check the validity of the cambium mortality equations in the local forest ecosystems and climatic conditions (Table 3).

Corresponding climatic conditions were attributed to each study or field observations used for validation according to the fire information available. Field observations were directly linked to the climatic condition that corresponds to their date of occurrence. For literature data, when information about fire intensity was present, the climatic conditions with similar modelled fire intensity were chosen. If the fire information only concerns fire severity with no information on fire behaviour nor season (Maringer et al. 2016b), predicted mortality rates were

Table 3 Literature studies and field observations used to compare modelled tree mortality with observed post-fire mortality for European tree species present in the French Alps

	Authors/Fire event	Fire information	Tree species	Corresponding climatic conditions		
				Cold season	Average summer	Extreme summer
Literature	Maringer et al. (2016b)	Fire severity	*Fagus sylvatica*	X	X	X
	Linder et al. (1998)	*SF*: low-to-moderate intensity	*Picea abies*		X	
	Kolström and Kellomäki (1993)	Summer *SF* of moderate to high intensity	*Picea abies*			X
			Pinus sylvestris			X
	Ordóñez et al. (2005)	Large summer *SF*	*Pinus nigra*			X
	Valor et al. (2017)	*SF*: Low-to-moderate intensity occurring in spring and fall	*Pinus nigra*	X		
			Pinus sylvestris	X		
	Sidoroff et al. (2007)	*SF*: Low-to-moderate intensity	*Pinus sylvestris*	X	X	
	Forradellas et al. (2016)	*SF*: Low-to-high intensity	*Pinus uncinata*	X		
Field observations	Champagny-en-Vanoise, Aug 2003	*SF* with torching (100ha, 3 days)	*Picea abies*			X
	Néron mountain, July–August 2003	*SF* and *CF* (300 ha, 33 days)	*Quercus* sp.			X
	Marlens Ugine, November 2016	Low-intensity *SF* (100 ha, 4 days)	*Quercus* sp.	X		

SF and *CF* stand for surface fire and crown fire, respectively compared to real observations for each climatic scenario but differentiating the different levels of fire severity.

Probability of mortality at the forest community scale

Post-fire tree mortality in the main forest types was finally computed for the four biogeographic zones of the French Alps using all the cambial mortality equations in order to use the entire range of mortality modelled. In France, the National Forest Inventory attributes a weight to each tree according to the distance between the tree and the centre of the plot (Robert et al. 2010). These weights were used to calculate the mortality rate at the NFI plot level as described in Eq. 9.

$$\text{Stand mortality rate} = \frac{\sum_{i=1}^{n} w_i \times P_m(i)}{\sum_{i=1}^{n} w_i} \tag{9}$$

$P_m(i)$: Probability of mortality of the tree i
w_i: weight attributed to the tree i
n: number of trees measured on the NFI plot

Results

Fire simulations

The results of the fire simulations amongst forest stand types and biogeographic zones of the French Alps are presented in Fig. 3. Detailed modelled fire behaviour within forest types is presented in Supplementary Material B. As expected, an increase in fireline intensity and rate of spread was observed from cold season to summer 2003. Decreasing gradients of *FLI*, *FL* and *ROS* were generally observed from Southern Alps to Northern Alps and from low elevation to high elevation.

Forest fires occurring during the cold season had very low intensity (< 200 kW m^{-1} in the Northern Alps and < 300 kW m^{-1} in the Southern Alps) and low *ROS*. Nonetheless, coppices forests and pure coniferous stands might experience more severe fire behaviour in the cold season.

In average summer conditions, the modelled forest fires presented a low-to-moderate intensity (100–400 kW m^{-1}) with an accentuated difference between low and high elevations and between Northern and Southern Alps. The fire front also had a greater speed (mostly between 0.5 and 1.5 m min^{-1}) than in the cold season. The forest types that already showed higher fire intensities in the cold season also had greater values during average summer conditions. Pure and mixed stands composed of *Pinus* sp., *Quercus* sp. and *Larix decidua* were characterized by higher values of *FLI*, *FL* and *ROS*. Although most of the fire behaviour variables are typical to surface fires with a low-to-moderate intensity, some forests characterized by a small stand height or presenting a vertical continuity of the vegetation (e.g. coppices) may experience crown fires during average summer conditions, especially at low elevation in the Southern Alps.

In extreme summer conditions like summer 2003, simulated forest fires presented moderate intensity from 150 to 600 kW m^{-1} with a significant difference according to the biogeographic zones. The hierarchy observed between forest stand types for average summer was emphasized and the probability of crown fires increased with several forest stands showing $FLI > 500$ kW m^{-1}.

The maximum *FLI* from literature (Table B.1 in Supplementary material) for *Pinus sylvestris* and *Castanea sativa* stands have the same order of magnitude as the maximum FLI computed in our study for the different climatic conditions. Our modelled maximum *FLI* for summer conditions concerning *Pinus sylvestris* stands are lightly greater than those provided by Arpaci et al. (2011). For *Castanea sativa* (all climatic conditions) and *Pinus sylvestris* in cold season, our results are within the range of maximum *FLI* found in the literature and generally close to the upper bound.

Vulnerability to fire of the main tree species

Figure 4 presents the post-fire mortality rates per species according to *DBH*, cambial mortality method of calculation and climatic conditions. For each tree species, post-fire mortality is bounded between two flame residence times (τ). The bounds were assumed to correspond to a τ of 30 s for the lowest and to the maximum theoretical flame residence time τ_{max} for the highest ($\tau_{max} \in [60 - 300]$ s according to fuel load and fuel moisture content, see Supplementary Material B.).

As expected, modelled post-fire tree mortality differs according to the cambial mortality equations, but to a lesser extent compared to the effect of the flame residence time. A gradient in mortality rates is observed from cold season to extreme summer conditions with higher mortality in drier conditions. Bauer et al. (2010) return the lowest mortality rates with a quick convergence with Peterson and Ryan (1986) for *DBH* above 20 cm. Mortality modelled from Bova and Dickinson (2005) equation is always the highest. With the exception of *Fagus sylvatica* which presents a very wide range of mortality rates for every climatic conditions, the different tree species mortality curves follow some general trends. The range of modelled mortality rates is generally wider for the lowest *DBH*, and it narrows progressively close to large *DBH*. Modelled mortality rates underline differences in vulnerability to fire depending on the tree species. Species with thin bark (*Fagus sylvatica*, *Abies alba*) have the highest and the widest ranges of mortality rates between minimum and maximum residence times, while thick-bark species

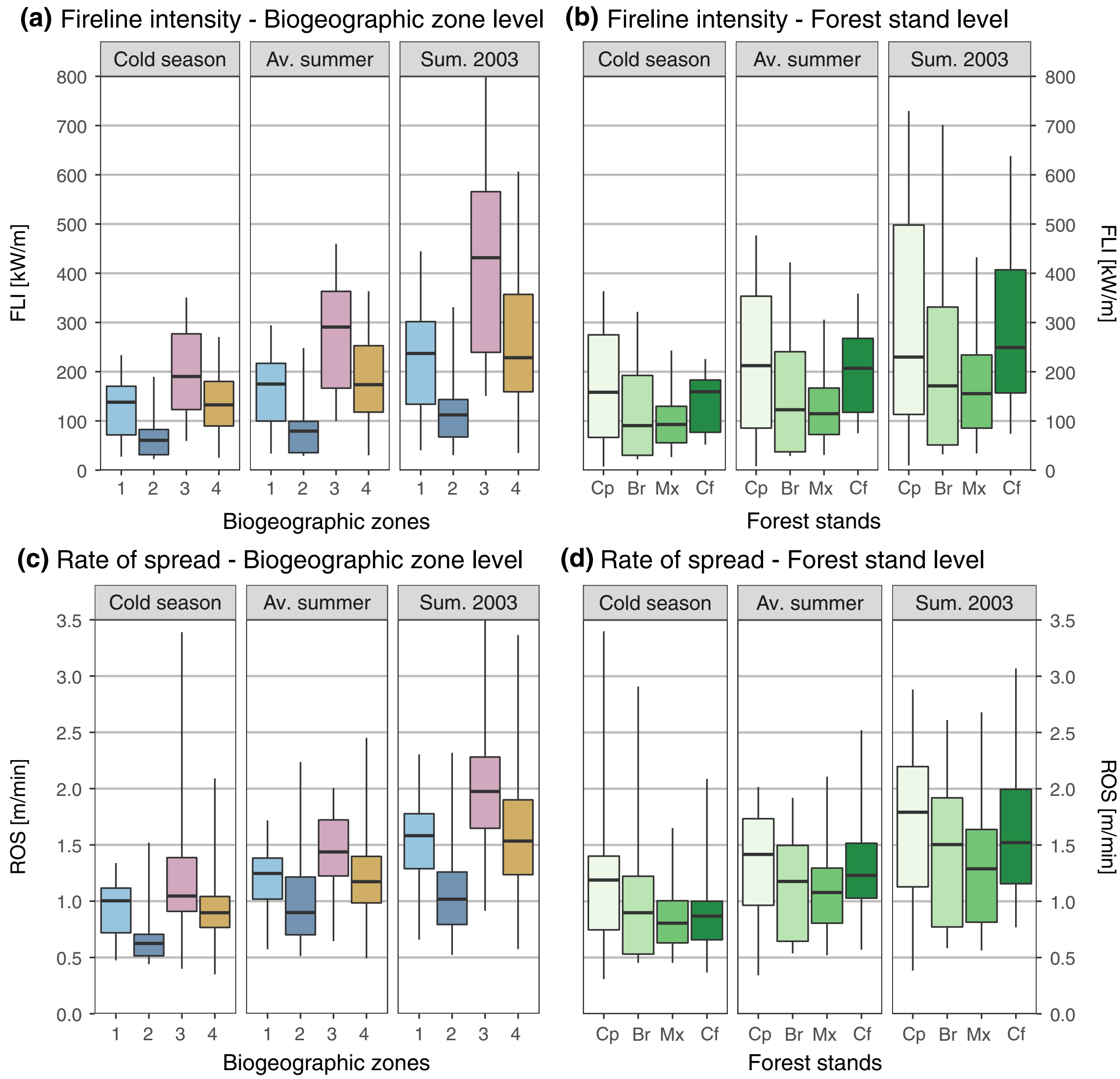


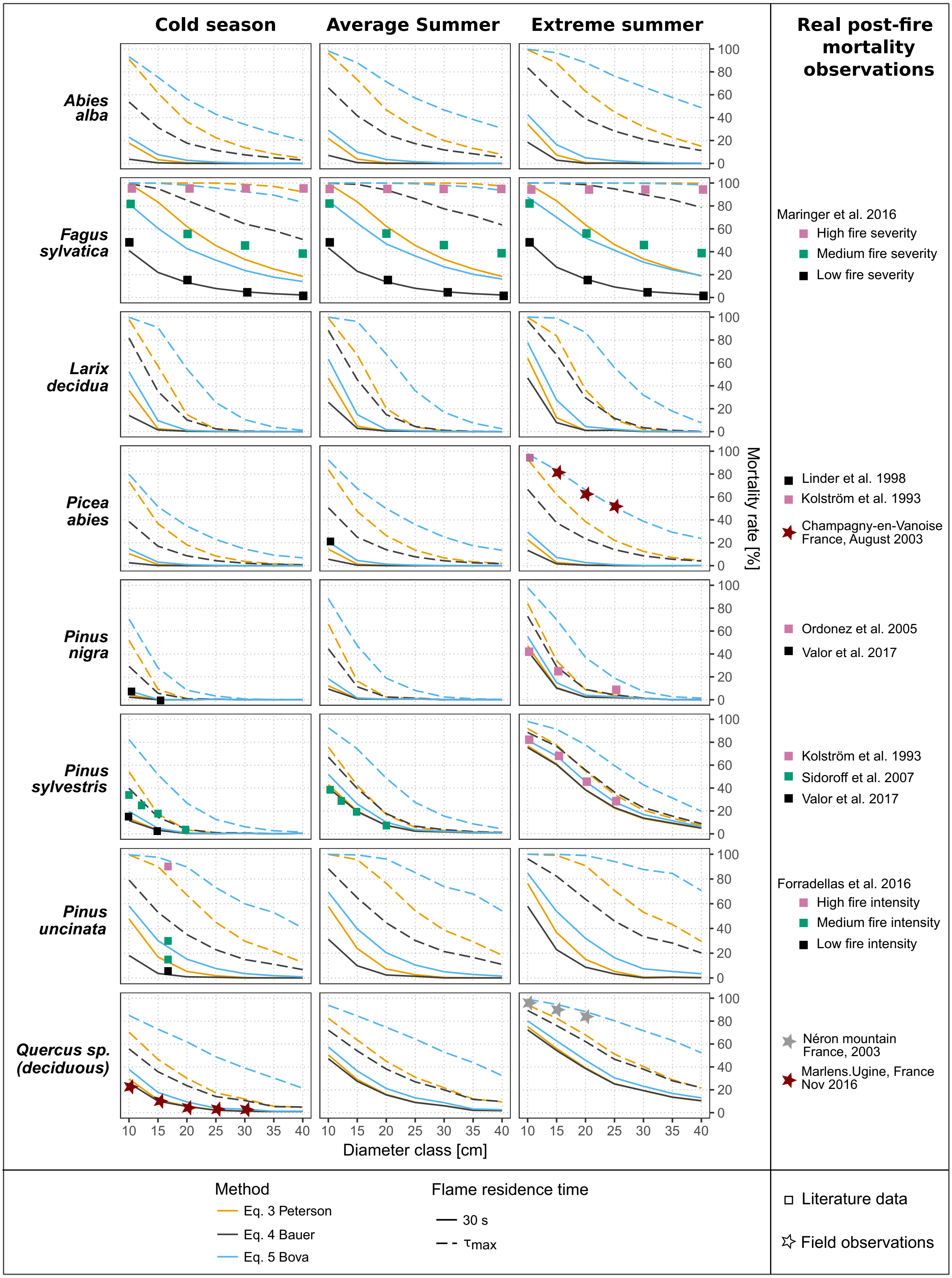
Fig. 3 Fireline intensity (FLI) and rate of spread (ROS) modelled on the NFI plots according to climatic conditions (Table 2). Results are given for four biogeographic zones (Fig. 1): 1—Northern Alps ≤ 800 m, 2—Northern Alps > 800 m, 3—Southern Alps ≤ 800 m, 4—Southern Alps > 800 m (a. and c.)—and four forest stand types: Cp—coppice, Br—pure broadleaved stands, Mx—mixed stands, Cf—pure coniferous stands (b. and d.). Boxplots represent the 5th, 25th, 50th, 75th and 95th percentiles, respectively

(*Larix decidua*, *Pinus nigra*, *Pinus sylvestris*, *Quercus* sp.) showed lower ranges of mortality rates. Moreover, for the two coldest climatic conditions, the thin-bark species may experiment post-fire mortality for all trees with a *DBH* up to 40 cm while only trees with a *DBH* under 25 cm are concerned for thick-bark species.

The post-fire data from literature or field observations are always displayed within the range of modelled mortality, suggesting a good representation of real observations in our predictions. Real observations from cold season are generally closer to mortality rate curves corresponding to the minimum flame residence time, while those from extreme summer events are closer to the maximum theoretical flame residence time. Mortality rates observed during average summer conditions are in between the two flame residence times.

Vulnerability to fire of the main forest types according to their biogeographic zones

The modelled post-fire mortality at the biogeographic zone level and forest stand level is displayed in Fig. 5. Predicted post-fire mortality according to climatic conditions increased from cold season fires (median mortality rates < 30%) to extreme summer fires (median mortality rates above 60% for all biogeographic zones). Mortality rates decreased with high altitude or high latitude. Moreover, forest stands mainly



Cold season
Average Summer
Extreme summer
Real post-fire mortality observations
Abies alba
Fagus sylvatica
Larix decidua
Picea abies
Pinus nigra
Pinus sylvestris
Pinus uncinata
Quercus sp. (deciduous)
Maringer et al. 2016
High fire severity
Medium fire severity
Low fire severity
Linder et al. 1998
Kolström et al. 1993
Champagny-en-Vanoise France, August 2003
Ordonez et al. 2005
Valor et al. 2017
Kolström et al. 1993
Sidoroff et al. 2007
Valor et al. 2017
Forradellas et al. 2016
High fire intensity
Medium fire intensity
Low fire intensity
Néron mountain France, 2003
Marlens.Ugine, France Nov 2016
Mortality rate [%]
Diameter class [cm]
Method
Eq. 3 Peterson
Eq. 4 Bauer
Eq. 5 Bova
Flame residence time
30 s
τmax
Literature data
Field observations

◂**Fig. 4** Average mortality rate of the main tree species according to diameter classes and climatic conditions. When available, mortality data from real fire events are also displayed. Post-fire mortality corresponding to the minimum flame residence time (30 s) is shown with solid line. Post-fire mortality corresponding to the theoretical maximum flame residence time is shown with dashed line

composed of broadleaved species (especially coppices) show higher mortality rates than mixed or pure coniferous stands.

Figure 6 details the mortality rates computed for the main forest types of the French Alps, the three climatic conditions and the four biogeographic zones. In the pure coniferous stands (see Dupire et al. (2016b) for definition), the mortality rates modelled in the cold season were quite low (median < 15%) except for pure stands of *Pinus uncinata/cembra* in the Southern Alps at high elevation (Fig. 6a). In average summer conditions, only pure stands of *Pinus sylvestris* and *Pinus uncinata/cembra* showed a majority of plots with a median mortality rate above 50%. Finally, in extreme summer conditions only pure stands of *Picea abies*, *Pinus nigra* and *Larix decidua* did not experiment a median mortality rate above 50%.

In mixed stands (Fig. 6b), modelled mortality rates were in between pure coniferous and pure broadleaved stands. In the cold season, the mortality rates modelled in the cold season were moderate (median < 30%) except for *Mixed Broadleaved* stands, especially in the Southern Alps (> 60%). In average summer conditions, mixed broadleaved stands have a median mortality rate above 50% in all biogeographic zones. They are followed by *Abies/Picea/Fagus* and *Larix/Broadleaved* stands. Finally, in extreme summer conditions, all forest types showed a median mortality rate above 50%.

In the pure broadleaved stands (Fig. 6c), modelled mortality rates were higher. In the cold season, only *Castanea sativa* stands stayed under 50% of mortality rate. In average summer conditions, all forest types showed a majority of plots with a mortality rate above 50%. Finally, in extreme summer conditions, the median mortality rate observed is generally above 75%.

Coppices (Fig. 6d) showed the highest mortality rates. All coppices reached the threshold of 50% in the cold season at least in one of the biogeographic zones. In the summer, mixed coppices showed a high mortality rate (> 60%) in average summer and very high mortality (> 80%) in extreme summer condition. Values for *Quercus* sp. and *Fagus sylvatica* coppices are slightly lower in average summer but also reached very high level in summer 2003 conditions (> 80%).

Discussion and conclusion

Vulnerability to fire of the main Alpine tree species

The following section gathers information from literature combined with our results in order to provide significant information on vulnerability to fire of the main tree species of the Alps.

Abies alba

Abies alba is characterized as a “fire-intolerant” species (Tinner et al. 2000) due to its relatively small bark thickness (Fréjaville et al. 2018b, 2013) and because of its low resilience ability several years after fire (Tinner et al. 2005). It has even been assumed that its extinction in the mid-Holocene in the Southern Alps was due to an increase in forest fires frequency (Wick and Möhl 2006; Tinner et al. 2005, 1999). This species is mostly located in the moist areas of the Northern Alps characterized by low vegetation flammability (Fréjaville et al. 2018a) and low fire weather risk (Dupire et al. 2017). In forests dominated by *Abies alba*, the spatial arrangement of fuel components and particles is very compact conferring this forest type a high resistance to the spread of fire. Our results agreed with this statements with a low post-fire mortality in cold season fire (very unfavourable climatic conditions). However, in drier conditions, the mortality rates can reach values above 50%, especially within the smallest trees (*DBH* < 20cm) where a high mortality was observed.

Fagus sylvatica

Fagus sylvatica is classified as a “fire-sensitive” species (Tinner et al. 2000) due to its very thin bark (Fréjaville 2015). High fire frequency generally leads to a dramatic abundance decreases in the species (Tinner et al. 2000). However, several studies suggested that *Fagus sylvatica* shows a high regeneration rate for low-to-medium fire severity but suffers for high burn severity or high fire frequency (Maringer et al. 2016c, b; Conedera et al. 2010). This species has a rather similar biogeographic distribution than *Abies alba* across low-flammability forests except in its driest margin in the southern Alps (Fréjaville et al. 2018a). According to the fire simulations, the immediate post-fire mortality is the highest observed amongst the studied species. Its thin bark makes the species, especially sensitive even for the biggest trees. Our results were compared to the post-fire observations in Swiss and Italian Alps from Maringer et al. (2016b). If it was not possible to link the different fire severity levels to one of our climatic conditions, observed

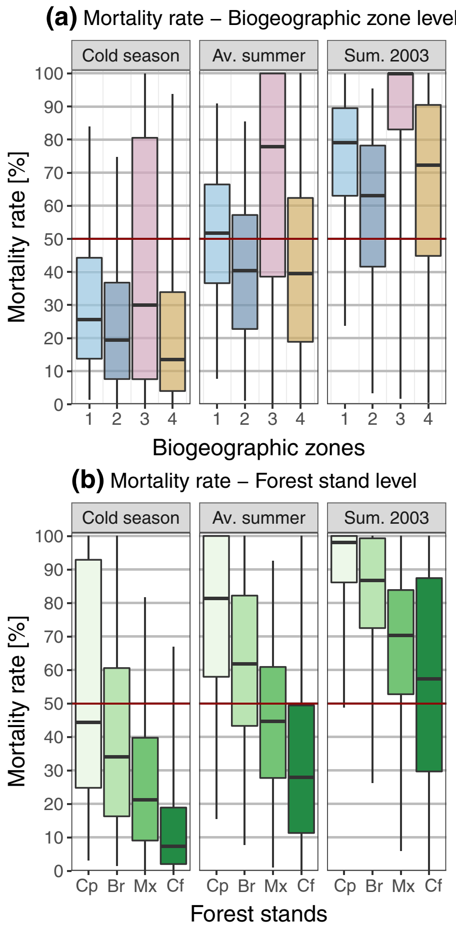


Fig. 5 Mortality rate modelled on the NFI plots according to climatic conditions (Table 2) and biogeographic zones (Fig. 1): 1—Northern Alps ≤ 800 m, 2—Northern Alps > 800 m, 3—Southern Alps ≤ 800 m, 4—Southern Alps > 800 m (a.) and forest stand types: Cp—coppice, Br—pure broadleaved stands, Mx—mixed stands, Cf—pure coniferous stands (b.). The boxplots gather results with the two flame residence times and with the three cambial mortality equations. The symbolic threshold of 50% was used in order to compare the mortality rate of the different forest types

mortality was always displayed within the range of modelled mortality. Assuming that fire severity correlates with fire intensity (Moris et al. 2017; Maringer et al. 2016b; Ascoli et al. 2013; Catry et al. 2010), high fire severity is more susceptible to happen under extreme climatic conditions, moderate severity under average summer conditions and low severity during cold season fires. This suggests that *Fagus sylvatica* may suffer high post-mortality in the next decade during extreme drought period expected to be more frequent.

Larix decidua

Larix decidua is a "fire-indifferent" species (Tinner et al. 2000) due to its thick bark, especially for the biggest trees (Fréjaville et al. 2013) and its high resilience characterized by a strong recruitment after fire (Moris et al. 2017). Nonetheless, high mortality rates can be observed in open forests due to higher continuity between surface fuels and the base of the crown (Fréjaville et al. 2018b) or in young stands defined by thinner bark (Moris et al. 2017). According to the fire simulations, the immediate post-fire mortality can be relatively high in the lowest diameter class (*DBH* < 20 cm) especially if the flame residence time is long. However, the mortality rate quickly decreased for higher *DBH* which is consistent with the quick increase in bark thickness with diameter observed in this species (Fréjaville et al. 2018b). At the stand level, our results suggest that stand dominated by this species can only experiment high mortality rates under extreme climatic conditions.

Picea abies

Picea abies has a similar behaviour than *Abies alba* (Tinner et al. 2000). In Sweden, it has been shown that forest fires can reach high mortality rates in young stands (Wallenius et al. 2005). According to the fire simulations, the immediate post-fire mortality is quite similar to the one of *Abies alba*. The post-fire mortality following a forest fire that occurred in August 2003 in Champagny-en-Vanoise, Savoie, France (1600–1900 m a.s.l.), was added to the plot of extreme climatic conditions. Observed mortality was in agreement with simulations calculated according to Bova and Dickinson (2005) under a high flame residence time. Two studies were found with reference of post-fire mortality of small-diameter *Picea abies*, the first one concerned summer prescribed burnings in Sweden (Linder et al. 1998) with moderate fire intensity and the second one a summer surface fire in Finland (Kolström and Kellomäki 1993). These observations classified as average and extreme summer conditions were in agreement with simulations based on the Bova's model under low and high flame residence times, respectively.

Pinus nigra and *Pinus sylvestris*

These two *Pinus* sp. respond similarly to fires (Fernandes et al. 2008). They are "fire-resistant" tree species which means that they can survive several surface fires from low to moderate intensities (Granström 2001), especially when the trees *DBH* exceed 20 cm. Both species exhibit the same range of fire resistance traits when accounting for tree diameter and stand density (i.e. bark thickness, crown top and base heights; Fréjaville et al. (2018b)). However, a recent study has shown that Pinus sylvestris may be less resistant than Pinus nigra according to a lower protection of needles (thinner) to heat (Valor et al. 2017). Fire may also favour the recruitment and early growth of *Pinus sylvestris* (Hille and den Ouden 2004) conferring it also good resilience abilities in addition to its resistance traits.

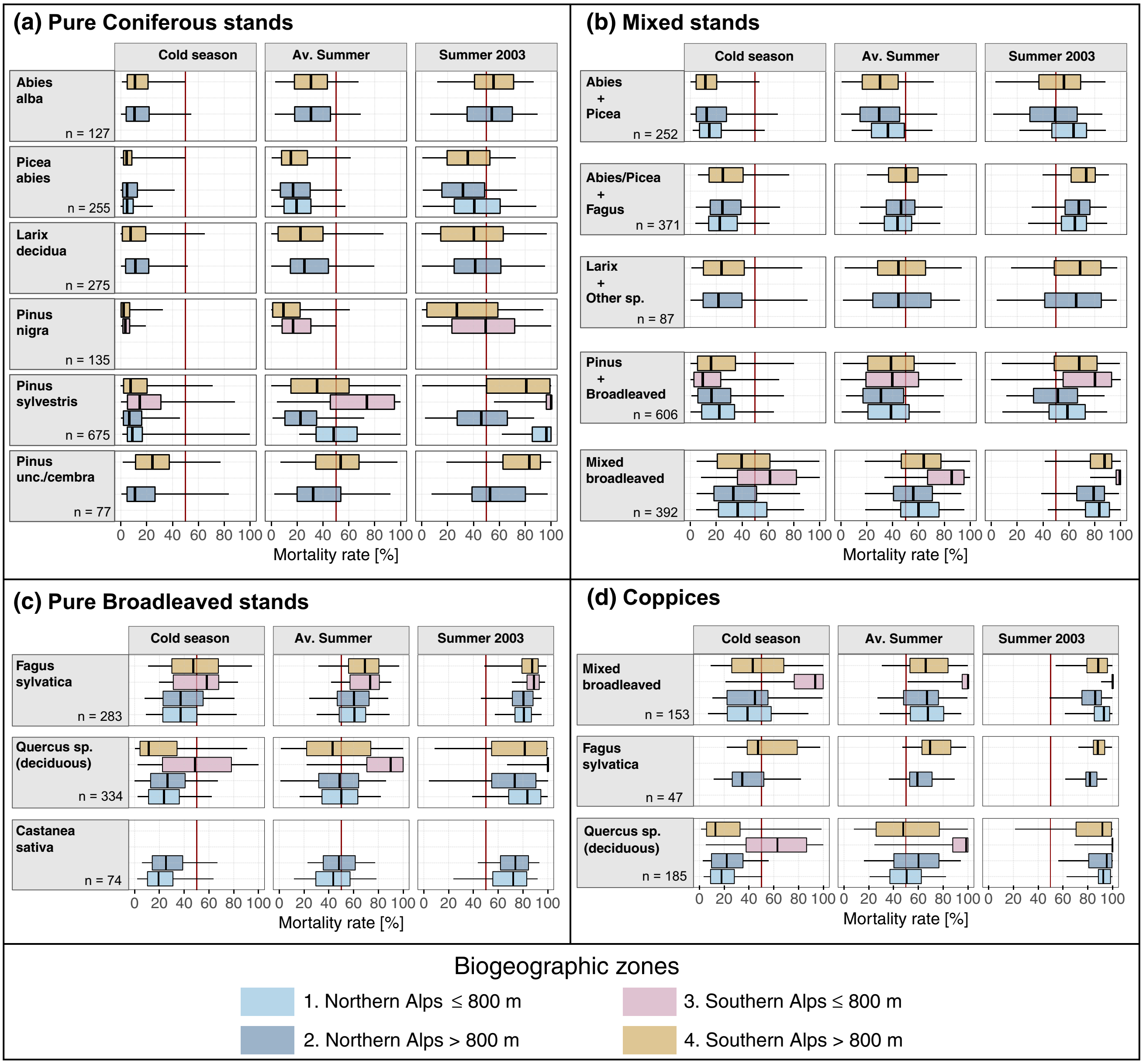


Fig. 6 Mortality rate modelled on the NFI plots according to forest types, climatic conditions (Table 2) and biogeographic zones (Fig. 1): 1—Northern Alps ≤ 800 m, 2—Northern Alps > 800 m, 3—Southern Alps ≤ 800 m, 4—Southern Alps > 800 m. Forest types are shown only if at least ten plots in the biogeographic zone are present. The boxplots gather results with the two flame residence times and with the three cambial mortality equations. The symbolic threshold of 50% was used in order to compare the mortality rate of the different forest types

Our results showed good prediction of the real post-fire observations from four different studies (Valor et al. 2017; Sidoroff et al. 2007; Ordóñez et al. 2005; Kolström and Kellomäki 1993). Despite their similar resistance to surface fires (Fernandes et al. 2008), higher mortality rates are observed in *Pinus sylvestris* stands. In fact, *Pinus sylvestris* is widely distributed in the French Alps compared to *Pinus nigra*; therefore, it experiments higher fire intensities with higher mortality rates, especially at the lowest elevations and latitudes which corresponds to its dry range margin (Fréjaville et al. 2018a)

Pinus uncinata and *Pinus cembra*

These *Pinus* sp. that co-dominate with *Larix decidua* the subalpine forests of the inner Alps are amongst the most fire-sensitive species due to their thin barks and low crown base heights (Fréjaville et al. 2018b; Fernandes et al. 2008), in addition to highly flammable barks (Fréjaville et al. 2013). The modelled mortality rates are in agreement with these specific traits with higher values than in others *Pinus sp*. Our results were compared to post-fire observations concerning a winter fire in *Pinus uncinata* forest in central Pyrenees (Forradellas et al. 2016). Mortality observed for low-to-moderate fire intensity is in agreement with low flame residence time simulations, while mortality for high fire intensity is close to the most severe simulation model. This result supports the idea that very dry conditions (even in winter) may promote high mortality for these species because they are mostly distributed in open forests of high flammability (Fréjaville et al. 2018a).

Deciduous Quercus sp.

Quercus pubescens, *Quercus robur* and *Quercus petraea* are fire-indifferent species such as *Larix decidua* (Tinner et al. 2000). Conedera et al. (2010) described *Quercus robur* and *Quercus petraea* as highly resistant to low fire intensity but sensitive to moderate fire intensity, especially because of a high foliage loss and significant bark failures.

The modelled mortality rates agreed with this description with low values in the cold season and moderate to high values in extreme climatic conditions. Mortality rates observed after two events are displayed in Fig. 4. The first event burned about 100 ha of mixed coppices in November 2016 near Ugine, Savoie, France (450–900 m a.s.l.). Tree mortality one year after fire is plotted in the cold season graph and followed simulated trends for flame residence time of 30 s. The second event burned about 380 ha of shrubs and oak coppices on the Néron mountain near Grenoble, France, in August 2003 (250–1300 m a.s.l.). Almost all trees died and the observed mortality followed the most extreme modelled trend for summer 2003. Our results show that these species are especially vulnerable in coppices that exhibit high vertical continuity between surface and crown fuels, and are mostly located downslope where fire weather danger can reach high to extreme levels.

Importance of combining climatic conditions, forest flammability and ecological traits to estimate forest vulnerability to fire

The vulnerability to fire of forest communities results from complex interactions between different drivers (Dalziel and Perera 2009). Our results underline (1) the high mortality expected under extreme climatic conditions, (2) higher mortality in low elevations and low latitudes (exposed to higher fire weather danger) mainly covered by broadleaved forests and (3) higher mortality for thin-bark species such as *Fagus sylvatica* and *Pinus cembra/uncinata*. Post-fire mortality should be interpreted in the light of fire behaviour and ecological traits of the different tree species. For example, *Picea abies* presents similar mortality rates than *Larix decidua* (Fig. 4) despite a significant thinner bark. This is mainly due to lower fire intensities in *Picea abies* stands (Fig. B.2 in Supplementary material). The high mortality rates observed in broadleaved stands result from (1) more severe climatic conditions as those species are mostly located in the lowest elevations (Dupire et al. 2017) and (2) a higher proportion of small trees presenting thinner bark and lower crowns (especially in coppices).

Furthermore, it is important to detail the tree mortality according to the size of the trees. The same tree species can be vulnerable at the young age but very resistant at the mature age such as *Larix decidua* (Moris et al. 2017). At the contrary, some species are vulnerable at almost all maturing stages such as *Fagus sylvatica* (Maringer et al. 2016b). Moreover, our fire simulations, based on Rothermel equations, did not account for the effects of plant architectural and leaf traits on fire severity (Zylstra et al. 2016) that might differ amongst seasons. In this study, only mortality rate as function of *DBH* is shown but a similar result could be obtain regarding the height of the trees (Rigolot 2004).

Finally, only the immediate post-fire tree mortality was considered in this work. Therefore, only direct mortality processes were taken into account, i.e. damages to the vegetative tissues resulting from their exposition to a heat source. However, indirect mortality (e.g. fungus or insects attacks, soil degradation, etc.) might sometimes cause more damages to burnt forests in the long term (Varner et al. 2009; McHugh et al. 2003; Swezy and Agee 1991). Furthermore, we did not account for the resilience abilities of the different tree species. For example, broadleaved species might present resprouting capabilities several years after fire (Conedera et al. 2010) and others species might have a strong recruitment after fire such as *Larix decidua* (Moris et al. 2017). This point could be improved with the follow-up of several pilot forests across the Alps that were touched by a fire. A common method could be applied for the main forest types inspired by the pre-cited existing works. Doing so, the knowledge about both direct and indirect mortality together with the resilience ability of the main tree species of the Alps could be significantly improved.

Predictive capabilities of post-fire mortality models

This work provided a complete method that returns a range of post-fire tree mortality at both tree and stand levels. All real post-fire data from literature and field observations were included in the ranges of modelled tree mortality underlining the robustness of the method. Therefore, using three different mortality equations and two flame residence times allowed to limit the uncertainties inherent in any modelling work. The predicted ranges of post-fire mortality can sometimes be quite wide (e.g. pure stands of *Abies alba* and coppices of *Fagus sylvatica*). This point is linked to the lack of information concerning the real fire behaviours that occurred in the different post-fire mortality studies. In fact, most of post-fire studies speak in terms of fire severity, e.g. the percentage of trees killed and not in fire intensity (Keeley 2009). It is therefore hard to have a direct link between modelled fire behaviours (which depends on climatic conditions and fuel properties) and fire impacts. This shortcoming could be improved with a better documentation on the wildfire. In France, at the moment, the fire databases Prométhée (2017) and BDIFF (2017) only provide information about the date of ignition, the location, the burnt surface and sometimes the cause. It would be very interesting to have more information (even rough) on fire behaviour such as flame length, speed of the fire front, width of the flaming area and on the type of fire (understory, surface or crown fire) which are of critical importance to enhance the reliability of such modelling approaches at a broadscale.

Implications in the context of climate changes and consequences on the vulnerability of forest ecosystems

Ongoing and expected climate changes might promote favourable climatic conditions for large and intense fires in the Alps with temperature increase, decrease in summer rainfall, and more frequent episodes of severe drought (Dupire et al. 2017; Gobiet et al. 2014). Thus, fires occurring during extreme summer conditions, such as in 2003, that are expected to promote high mortality in all forest communities of the French Alps could become more and more frequent in the next decades, even at the highest altitudes. Moreover, our findings highlight that under less extreme conditions, some forests (e.g. coppices and Scots pine forests in the Southern Alps) may experience substantial post-fire mortality due to a relatively high aridity and vegetation flammability despite the high fire resistance of dominant trees. In addition to climate changes, the sources of fire ignition must also be considered for future predictions as it has been demonstrated that human activities might be a stronger determinant of fire regime than climate in some Alpine valleys (Conedera et al. 2018; Zumbrunnen et al. 2009) and in mountain and Mediterranean landscapes of Southern France (Fréjaville and Curt 2017; Curt et al. 2016) that makes highly challenging a precise evaluation of the spatial patterns in post-fire tree mortality.

This work allows to predict a conditional fire loss (Alcasena et al. 2016) under different fire weather scenarios assuming the same probability of ignition for each forest plot. However, in the Western Europe, about 85% of wildfires result from human activities (Curt et al. 2016; Reineking et al. 2010) and only 15% from natural ignitions such as lightnings (Arndt et al. 2013). The spatial distribution of fire ignitions is also heterogeneous as most of the fires occur near human infrastructures and populated areas (Ganteaume et al. 2013; Moreira et al. 2011) which are mainly located in the foothills in mountain areas. Therefore, forest stands the most likely exposed to a fire ignition are also those that showed the highest mortality rates such as downhill coppices and high stands of broadleaved species.

Different measures could be taken by forest managers in order to reduce the overall vulnerability to fire of their forests. For example, species with thick bark should be favoured when possible, especially in the driest locations. Second, the vertical continuity of the fuel should be broken, especially in foothill coppices (that are currently poorly managed), in order to avoid crown fires in case of extreme climatic conditions. In addition, forest managers and fire fighting policies should anticipate unwanted disasters with the establishment of firefighting road networks and water reserves in the foothills of the Alps (including in the Northern part where fire is not a part of forest management plans). Moreover, the possibility to close the access to forest in case of extreme weather conditions as it is currently practised in the French Mediterranean area could also be implemented during extreme events like summer 2003. Finally, the restoration of natural fire regime and the use of prescribed fires to reduce fuel loads, especially in low-elevation stands, could be an efficient measure to reduce vulnerability to fire in the Alps. However, reintroducing prescribed fires is at the opposite to the objective of reducing air pollution in the Alpine valleys, which is a major concern of public health (Saura et al. 2018).

Some mountain forests also protect against other natural hazards (Dupire et al. 2016a). In this study, we showed that some Alpine forests are especially vulnerable to fire which may have high impacts on their protective effects against rockfalls (Maringer et al. 2016a) or erosion (Inbar et al. 1998), and in turn, to human assets located downslope. Fighting a fire in steep terrain can be technically difficult and economically expensive. Targeting fighting operations to the forests that play an important protection role might be a cost-effective way of reducing the overall potential damages.

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