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# Changes in wildfire severity from maritime pine woodland to contiguous forest types in the mountains of northwestern Portugal

Paulo M. Fernandes[a,b,*], Ana Luz[b], Carlos Loureiro[a,b]

[a] Centro de Investigação e de Tecnologias Agro-Ambientais e Tecnológicas (CITAB), Universidade de Trás-os-Montes e Alto Douro, Apartado 1013, 5001-801 Vila Real, Portugal
[b] Departamento de Ciências Florestais e Arquitectura Paisagista, Universidade de Trás-os-Montes e Alto Douro, Apartado 1013, 5001-801 Vila Real, Portugal

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ABSTRACT

Large and severe wildfires are now widespread in the Mediterranean Basin. Fire severity is important to ecosystem properties and processes and to forest management but it has been neglected by wildland fire research in Europe. In this study, we compare fire severity between maritime pine (PS) woodland and other forest (OF) types, identify other variables influent on fire severity, and describe its variation. We sampled contiguous, paired stands of PS and OF cover types – including deciduous and evergreen broadleaves and short-needled mountain conifers – that burned under very high to extreme fire danger in northwestern Portugal. Data on stand characteristics and fire severity metrics were collected in plots along transects perpendicular to the PS–OF boundary. Fire severity was rated in separate for the tree canopy, understorey vegetation and forest floor layers, and then an average (composite) fire severity rating was calculated. Fire intensity inferred from stem char height (adjusted for the effects of other factors) was highest in PS, followed by deciduous broadleaved woodland and short-needled conifer forest. With a few exceptions, all fire severity ratings were significantly different between PS and OF at all sites. Most fire severity metrics and ratings were correlated. The distance for fire severity minimization did not differ between OF types (median = 21 m). Variation in composite fire severity was accounted for by a classification tree ($R^2 = 0.44$) based on cover type (contributing with 51% to the overall explanation), stand variables, aspect, distance to the PS–OF edge and fire spread pattern. Except for a more immediate decline in deciduous broadleaves, fire severity rating was not affected by OF type and tended to decrease in more mature stands and moister aspects. The fire severity moderation from PS to OF was compounded by a dominant pattern of down slope fire propagation into moister topographical positions, exacerbating the fuel effect implicit in the cover type change. The results are consistent with fire hazard and fire incidence studies and support conventional knowledge that advocates the expansion of broadleaved deciduous or evergreen forest as a means to achieve more fire-resilient ecosystems and landscapes.


## 1. Introduction

Fire severity describes the immediate effects of fire produced by the aboveground and belowground heat release pulses, and its assessment is based on biomass consumption and visual evidences of heating and heat-inflicted injury to vegetation (Ryan and Noste, 1985; Keeley, 2009). Both the interpretation of fire severity and its implications on post-fire response are expected to vary among ecosystem types (Keeley, 2009). Highly severe fires can have great impact on ecosystem attributes and processes, namely soil erosion and sedimentation, habitat fragmentation and availability, patterns of vegetation and community recovery, alien plant invasion and carbon dynamics (Keeley, 2009; Miller et al., 2009). As a consequence, the evaluation and prediction of fire severity is of interest to scientists and managers given its ramifications to ecological integrity and forest management.

Climate, fuel and topography influence fire regimes at different scales. Climate is the driving force at coarse spatial and temporal scales, whereas fine-scale variation in vegetation and terrain control fire behaviour and severity locally (Heyerdahl et al., 2001). Forest vegetation affects fire behaviour through its intrinsic fuel characteristics (e.g. Rothermel, 1972), but stand structure and position in the landscape will interact with atmospheric conditions and modify fire behaviour accordingly. When weather is conducive to large fires, these might expand in the landscape regardless of land cover, i.e. not selecting for more flammable vegetation types, e.g. Podur and Martell (2009). Spatial patterns in fire severity, however, are determined by a complex interplay between forest composi-

* Corresponding author at: Centro de Investigação e de Tecnologias Agro-Ambientais e Tecnológicas (CITAB), Universidade de Trás-os-Montes e Alto Douro, Apartado 1013, 5001-801 Vila Real, Portugal. Tel.: +351 259 350885; fax: +351 259 350480.
*E-mail addresses:* pfern@utad.pt (P.M. Fernandes), luzanalu@gmail.com (A. Luz), clour@utad.pt (C. Loureiro).

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tion and structure, weather and topography, even under extreme meteorological conditions (Finney et al., 2005; Lentile et al., 2006; Oliveras et al., 2009). In many circumstances the environmental and societal impacts of fire may well be better gauged by the amount of land burned by high-severity fire than by the total surface burned (Reinhardt et al., 2008). Accordingly, fuel management strategies primarily designed to mitigate landscape fire spread – including conversion to less flammable cover types (Pyne et al., 1996) – may in fact be more successful at decreasing fire severity.

Large wildland fires have gained importance in Mediterranean Europe since the 1970s, which is ascribed to changes in forest management and composition and to the abandonment of marginal agricultural land (Schelhaas et al., 2003). The northwestern Iberian Peninsula is particularly affected, given the coincidence of favourable conditions for plant growth, summer drought and abundant ignition sources (Moreno et al., 1998). Recent studies in the Mediterranean Basin focus on the weather-driven nature of area burned and examine how fire incidence varies with forest cover type or with cover type in combination with forest structure descriptors (Moreira et al., 2001, 2009; Diaz-Delgado et al., 2004; Nunes et al., 2005; Pereira et al., 2005; González et al., 2006; Silva et al., 2009). Conversely, there is a general paucity of studies tackling the determinants of wildfire severity in Europe, which are restricted to Catalonia, Spain: Broncano and Retana (2004) and Oliveras et al. (2009) address the issue through remote-sensing approaches, while González et al. (2007) have examined how tree mortality varies with stand composition and structure. Other ground-based studies have not been carried out to date and the patterns of fire severity in relation to forest composition, among other relevant factors, remain poorly understood.

Maritime pine (*Pinus pinaster* Aiton) and shrubland are the dominant wildland cover types in the mountains and plateaus of northern and central Portugal. Both maritime pine stands (Cruz et al., 2008; Fernandes et al., 2009) and shrubland (Fernandes et al., 2000; Fernandes, 2001) are known for their high flammability and will usually experience stand-replacing wildfire. Fire selectivity studies (Nunes et al., 2005; Moreira et al., 2009; Silva et al., 2009) show that these cover types tend to be preferred by fire and burn proportionally more than other vegetation types.

Changes in forest type composition to decrease landscape-level fire hazard are expected to limit the extent of wildfire in southern Europe. However, the implications to fire severity remain to be determined. The main goal of this study is to analyze how fires impelled by severe weather burn in different forest types in the mountains of northwest Portugal. Our specific objectives are (i) to assess and compare fire severity between maritime pine woodland and cover types associated with lower fire hazard (Fernandes, 2009a) and incidence (Moreira et al., 2009; Silva et al., 2009), i.e. deciduous and evergreen broadleaves and short-needled conifers; (ii) to determine if fire severity is influenced by other factors, namely by stand structure; and (iii) to describe variation in fire severity. To achieve the stated goals we sampled plots along transects located in paired, adjacent stands of maritime pine and other forest types.

## 2. Methods

### 2.1. Study area and study sites

Our study area is a 4556-km$^2$ fraction of northwest Portugal bounded by latitudes 41°53′N and 41°16′N and longitudes 8°16′W and 7°28′W. Climate is transitional between Mediterranean and oceanic. Average annual precipitation ranges from 1000 mm at the lower elevations to more than 2800 mm at the upper elevations in the western side of the region, mostly falling between October and April and with 0–2 rainless months, and mean annual temperature varies between 7.5 and 15 °C (APA, 2003). Terrain is highly dissected with steep slopes. Soils are cambisols and podzolic rankers derived from granite or schist parent material.

Most of the mountains within the study area are communal lands densely covered by forest and shrubland that were afforested between the 1930s and the 1970s, mainly with maritime pine (*P. pinaster*). However, forest occupation has been reduced from 60 to 20% between 1972 and 2000 due to wildfires fuelled by conflicts over land use (Rego, 2001), thus shifting most of the landscape from pine-dominated to shrub-dominated. Shrubs of the genus *Erica*, *Ulex*, *Pterospartum* and *Cytisus* are prevalent in the shrubland communities and in the understorey of maritime pine stands. Wildlands within the study area have a high (>0.30) or very high (>0.40) probability to burn over a 30-year period (Pereira and Santos, 2003), although shrubland burns more than would be expected by chance in relation to forest (Moreira et al., 2009). Other vegetation types cover about 10% of the wildland area. The most represented are deciduous oaks (*Quercus pyrenaica* and *Quercus robur*) – as fragments of the original forest cover – followed by other deciduous broadleaved species (*Castanea sativa*, *Betula alba*) and by montane pines (*Pinus sylvestris* and *Pinus nigra*). Small patches of Mediterranean evergreen broadleaves (*Quercus suber* and *Arbutus unedo*) and North American conifers (*Pseudotsuga menziesii* and *Chamaecyparis lawsoniana*) occur locally.

Sites for sampling were selected by inspecting areas burned in the summers of 2005 and 2006. We searched for stands of maritime pine, hereafter referred to as PS (pine–shrub), adjacent to other forest types, hereafter referred to as OF. We selected sites where (i) fire suppression activity was not evident, (ii) fire had moved from PS into OF, and (iii) overstorey fire severity was different between PS and OF in general terms (burned, scorched, green). Consequently, fire severity and forest type were intentionally confounded to some extent, which is acceptable because our main interest was to describe variation in fire severity where differences in fire severity between PS and OF were apparent. The paired PS and OF stands had equal aspect and similar slope whenever possible. Our focus was to collect data on a variety of stand types and locations, rather than to concentrate on a particular forest type or large fire. Accordingly, only one sampling location was selected per fire, unless additional OF types were found bordering PS stands. The study locations are presented in Fig. 1.

Fire danger rating in Portugal is based on the CFFWIS, the Canadian Forest Fire Weather Index System (Van Wagner, 1987). The Fire Weather Index (FWI) indicates potential fire intensity. The Drought Code (DC) indicates the moisture status of deep organic layers and slow-drying fuels in general, including live shrub foliage (Viegas et al., 2001), and should be useful to gauge the potential for severe ground and belowground fire effects. The FWI and DC from the nearest weather station were averaged for the duration (number of days) of every fire. We assigned a fire danger class to each wildfire by using the FWI breakdown of Palheiro et al. (2006), which categorizes the difficulty of fire suppression in maritime pine forest. Table 1 displays the CFFWIS indexes and official information from the National Forest Authority on the 10 sampled wildfires. All fires occurred on very-high or extreme fire danger days. Fire size variation is wide (44–7086 ha) and the four largest fires ever recorded in the study area are included. With two exceptions (fires 6 and 10), the DC was well above 500, which indicates ground fire activity and persistent smouldering (Alexander and Cole, 2001).

Table 2 presents the physiographical characteristics and forested cover types adjoining PS in the 13 sampled sites. Deciduous broadleaves (*Q. pyrenaica*, *Q. robur*, *B. alba*, *C. sativa*), short-needled conifers (*P. sylvestris*, *P. menziesii*, *C. lawsoniana*) and evergreen broadleaves (*Q. suber*, *A. unedo*) comprise OF vegetation at 7, 4 and 2 sites, respectively. Site elevation ranged from 380 to 1050 m. Slopes facing west and north prevailed in the study sites. The pattern of fire

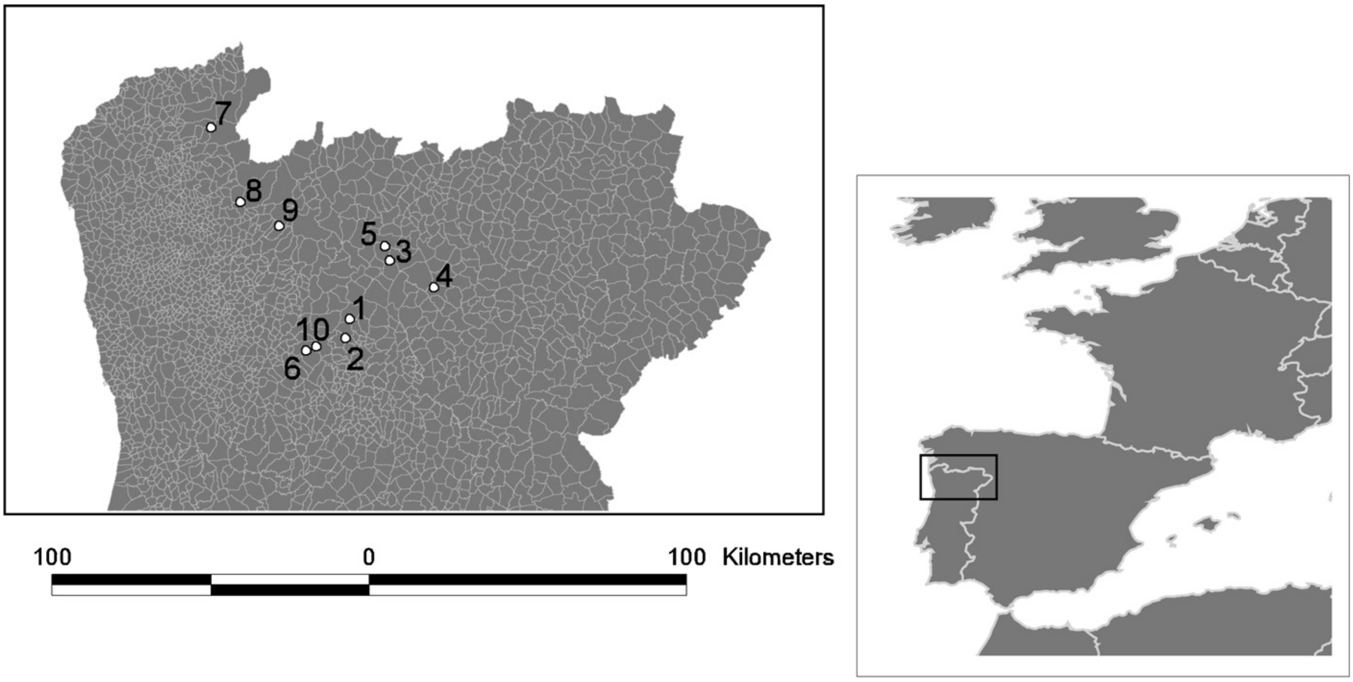


**Fig. 1.** Location of the study area and study sites. Numbers correspond to the wildfires in Table 2.

**Table 1**
Sampled wildfires.

| Fire # | Start date | Mountain range | Fire size (ha) | % Forest | Fire danger rating | | |
|---|---|---|---|---|---|---|---|
| | | | | | Class | FWI | DC |
| 1 | 15.08.2005 | Alvão | 4356 | 18.5 | Very high | 38 | 717 |
| 2 | 17.08.2005 | Alvão | 146 | 80.1 | Extreme | 40 | 734 |
| 3 | 30.08.2005 | Alvão | 599 | 100.0 | Extreme | 39 | 835 |
| 4 | 14.08.2005 | Padrela | 3603 | 34.9 | Extreme | 43 | 952 |
| 5 | 07.08.2005 | Alvão | 7086 | 93.9 | Extreme | 43 | 702 |
| 6 | 04.06.2006 | Alvão | 283 | 100.0 | Very high | 38 | 282 |
| 7 | 06.08.2006 | Soajo | 5590 | 3.6 | Very high | 35 | 621 |
| 8 | 03.09.2006 | Gerês | 184 | 98.9 | Very high | 29 | 568 |
| 9 | 10.08.2006 | Cabreira | 636 | 0.3 | Extreme | 51 | 576 |
| 10 | 15.07.2006 | Marão | 44 | 11.9 | Very high | 28 | 414 |

spread was down slope in both PS and OF (eight sites), upslope in PS and down slope in OF (two sites), down slope in PS and upslope in OF (one site), and slope neutral at two sites, i.e. the directions of fire spread and terrain slope were approximately perpendicular in both PS and OF.

### 2.2. *Sampling scheme*

The experimental design tried to capture the spatial heterogeneity in fire severity at the PS–OF interface. The boundary between PS and OF was defined as the vertical projection of the outermost

**Table 2**
Sampled sites characteristics, with aspect and slope discriminated by maritime pine (PS) and other forest (OF) types.

| Fire, site | Location | OF type | Elevation (m) | Aspect | | Mean slope (%)[a] | |
|---|---|---|---|---|---|---|---|
| | | | | PS | OF | PS | OF |
| 1, 1 | Lamas de Olo | *Quercus pyrenaica – Betula alba* | 1050 | S | N | 6 | −14 |
| 2, 1 | Sirarelhos | *Quercus pyrenaica – Castanea sativa* | 640 | N | N | −15 | −35 |
| 3, 1 | Cidadelha | *Pinus sylvestris* | 760 | SE | NW | 7 | −16 |
| 3, 2 | Cidadelha | *Castanea sativa – Betula alba* | 740 | SE | NW | −33 | 30 |
| 4, 1 | Mascanho | *Quercus pyrenaica* | 700 | SW | SW | 15 (0) | 15 (0) |
| 5, 1 | Bragado | *Quercus suber – Arbutus unedo* | 570 | E | E | −59 | −59 |
| 6, 1 | Covelo do Monte | *Pinus sylvestris* | 910 | NW | NW | −15 | −15 |
| 7, 1 | Vilar de Soente | *Betula alba* | 870 | W | W | −53 | −53 |
| 7, 2 | Vilar de Soente | *Pseudotsuga menziesii* | 810 | W | W | −45 | −45 |
| 7, 3 | Vilar de Soente | *Castanea sativa – Betula alba* | 790 | W | W | −28 | −28 |
| 8, 1 | S. Bento da Porta Aberta | *Arbutus unedo* | 380 | W | W | 50 (0) | 50 (0) |
| 9, 1 | Anjos | *Pinus sylvestris – Chamaecyparis lawsoniana* | 1040 | NW | N | −12 | −5 |
| 10, 1 | Campeã | *Quercus pyrenaica – Q. robur* | 940 | NW | NW | −32 | −32 |

[a] Negative slope values indicate downslope fire spread. (0) denotes transects that were perpendicular to slope orientation, i.e. the boundary between the two forest types was aligned with slope orientation.

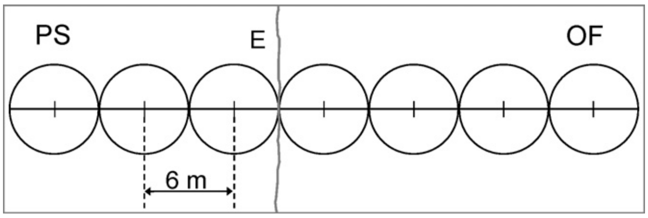


**Fig. 2.** Transect sampling scheme. PS: maritime pine stand; OF: other forest types; E: PS–OF edge.

continuous canopy of OF. On each study site linear transects with at least 30 m spacing were laid out perpendicularly to the PS–OF boundary. The location of each transect was subjectively chosen and avoided uneven boundaries and the existence of a transition zone between PS and OF. We installed three transects per site, except where the length of contact between PS and OF was insufficient to comply with the former requirements.

Contiguous and circular 3-m radius plots (28.3 $m^2$) were systematically located on each transect (Fig. 2). The initial two plots centres were marked at 3 m to both sides of the PS–OF edge (the reference to distance measurement), and the others followed successively at 6-m intervals. Heterogeneity in fire effects at small spatial scales is expected to decrease as fire intensity increases (Knapp and Keeley, 2006; Van Mantgem and Schwilk, 2009), which in general was visually apparent when the study sites were selected. Accordingly, we assumed less variable fire severity on the PS-transect segment and adopted a constant sampling distance of 18 m, i.e. three plots. The OF segment was variable in length and number of plots, from a minimum of three plots up to a maximum determined by the extent of the apparent spatial gradient in fire severity. Sampling ceased either when fire self-extinguished or when the fire severity level reached a minimum and ceased to vary with distance from the edge. Hence, transects ran from PS (−18 m) through the boundary (0 m) to a variable distance into OF. We sampled 200 plots between 1 and 3 months after fire, distributed by 56 transect segments located in the 13 study sites.

### 2.3. Data collection and fire severity classification

Slope was measured for each transect segment with a clinometer. All other assessments were plot-based. Tree stems taller than 1.3 m were measured for diameter at breast height (1.3 m) (dbh) and height. We calculated mean dbh, stem density, basal area and mean height of the dominant and codominant trees (hereafter referred to as stand height) for each plot, under the understanding that a 3-m radius plot will only provide rough estimates of tree density. Tree characteristics were also summarized for transect segments (PS or OF).

Fire was categorized as a surface fire when the overstorey did not burn or its combustion was limited to the lowest branches and as a crown fire otherwise. The assessment of fire severity individualized the overstorey, understorey and ground (organic substrate) layers. Overstorey fire severity was based on metrics – crown scorch height, stem char height, bark char depth – measured in each of the dominant and codominant trees in a plot. Potential fire severity was expected to be better depicted by following this procedure rather than by measuring all trees in a plot because completely burned or scorched trees underestimate fire intensity. Pre-burn crown base height for the dominant and codominant trees was measured in unburnt trees, i.e. was equal to post-fire crown base height. The crown base height of burned trees was estimated by assuming that (i) only dead branches are totally consumed by fire, and (ii) branches with residual buds supported foliage. Stem char and crown scorch heights – surrogates for flame size and fire intensity – were measured from the ground to respectively the highest uninterrupted mark of fire in the stem, and to the top of the line separating green and brown (fire-killed) foliage in the crown. All heights on trees were measured with a laser rangefinder to the nearest 0.1 m. After dividing the tree bole into slope-oriented quadrants, bark char depth (BCD) at 0.5 m was assessed for each quadrant and was rated as per Ryan (1982) and averaged.

The classification of fire severity followed Table 3. All fire severity components range from 1 (highest severity) to 4 (lowest severity). Overstorey fire severity was based on BCD, stem char ratio (SCR) and crown scorch ratio (CSR). SCR was calculated by expressing stem char height as a proportion of tree height. CSR was calculated from tree height, crown base height and crown scorch height as the proportion of crown length scorched or consumed by fire. After averaging these metrics for each plot the overstorey severity rating was computed as (BCD + SCR + 2 CSR)/4; CSR has a double weight in the equation because crown scorch is more readily correlated with fire behaviour (e.g. Van Wagner, 1973) and canopy kill is the most important cause of tree death (Fowler and Sieg, 2004; Fernandes et al., 2008). The classification of Ryan and Noste (1985) was adapted to score the prevailing fire severity in the understorey and ground layers and its evaluation was carried out by the same individual throughout the study. A composite fire severity score was determined by averaging the fire severities of the existing vegetation layers, rounded to the nearest integer and qualified as very high (1), high (2), moderate (3) or low (4).

### 2.4. Data analysis

The confidence level for statistical significance was set at 95% ($\alpha = 0.05$). Data variables were tested for normality with normal quantile plots and the Shapiro–Wilk test. Since almost all variables were non-normal, non-parametric statistical tests were used. Strength of the relationships among fire severity metrics and between fire severity ratings was assessed using the rank correlation coefficients of Spearman ($\rho$) and Kendall ($\tau$), respectively. The type of fire (surface or crown) and fire severity rating assigned to each plot were summarized by vegetation type. Wilcoxon two-sample tests were conducted to test for pairwise differences in fire severity between PS and OF stands. The analysis of correlation between fire severity ratings in PS and OF is based on fire severities averaged over the length of the transect segment (PS or OF). All other analysis are plot-based.

**Table 3**
Description and classification of fire severity components.

| Component | Fire severity classes | | | |
|---|---|---|---|---|
| Overstorey | | | | |
| Bark char depth | Unburned | Light | Moderate | Deep |
| Stem char ratio | <0.25 | 0.25–0.49 | 0.50–0.74 | ≥0.75 |
| Crown scorch ratio | Green | Low-moderately scorched (<0.50) | Severely scorched (≥0.50) | Burned |
| Understorey | Scorched | Lightly burned | Moderately burned | Severely burned |
| Ground | Scorched | Lightly burned | Moderately burned | Severely burned |
| Fire severity rating | 4 (low) | 3 (moderate) | 2 (high) | 1 (very high) |

**Table 4**
Median and range (min.–max.) of stand variables ($n = 56$) and fire severity metrics ($n = 200$) by forest cover type.

| Variable | PS[a] | SNC[b] | EB[c] | DB[d] |
|---|---|---|---|---|
| Stand variables | | | | |
| *n* (transects) | 28 | 9 | 4 | 15 |
| Mean transect length (m) | 18 | 27 | 21 | 24 |
| Stems $ha^{-1}$ | 463 (0–3056) | 667 (278–1111) | 903 (694–2917) | 1111 (741–1889) |
| Dbh (cm) | 15.5 (5.2–49.8) | 22.9 (16.9–57.7) | 23.6 (6.7–27.4) | 13.8 (7.5–25.2) |
| Basal area ($m^2$ $ha^{-1}$) | 8.9 (0.0–54.3) | 32.9 (16.6–72.6) | 44.0 (11.1–45.9) | 15.8 (7.4–44.9) |
| Stand height (m) | 11.8 (2.7–19.7) | 13.7 (9.8–27.8) | 9.3 (6.2–9.8) | 9.9 (6.9–21.2) |
| Fire severity variables | | | | |
| *n* (plots) | 84 | 41 | 14 | 61 |
| Stem char height (m) | 5.0 (1.0–15.2) | 2.3 (0.0–9.7) | 3.6 (0.1–11.3) | 0.8 (0.0–7.9) |
| Stem char ratio | 0.4 (0.1–1.0) | 0.2 (0–0.7) | 0.5 (0.0–1.0) | 0.1 (0.0–0.6) |
| Crown scorch height (m) | 10.2 (1.8–21.7) | 12.7 (2.0–22.0) | 8.0 (1.7–11.3) | 7.2 (1.3–18.5) |
| Crown scorch ratio | 1.0 (0.0–1.0) | 0.7 (0.0–1.0) | 0.6 (0.0–1.0) | 0.4 (0.0–1.0) |
| Bark char depth | 2 (1–3) | 2 (2–4) | 2 (2–4) | 2 (1–4) |

[a] Maritime pine.
[b] Short-needle conifers.
[c] Evergreen broadleaves.
[d] Deciduous broadleaves.

The sampling scheme allowed assessing the variability in fire severity within PS and OF areas, while decreasing the impact of differences in weather and topography between PS and OF. However, it also implied non-independent data and pseudo-replication (Hurlbert, 1984). To offset this difficulty and unequivocally determine whether fire behaviour differed between cover types we fitted linear mixed-effects models to crown scorch and stem char heights using restricted maximum likelihood (REML) (Bolker et al., 2008). Models included random effects due to site and transect within site. The significance of fixed effects was evaluated with *F*-tests. Least squares means of fire behaviour surrogates per cover type were compared with the Tukey–Kramer HSD test.

Classification tree analysis (Death and Fabricius, 2000) divides a dataset into increasingly homogeneous sub-groups and is well suited to model a categorical response variable from multiple variables, particularly in the presence of non-parametric or unbalanced data and non-linearity. Classification trees deal satisfactorily with autocorrelated data (Calbk et al., 2002) and are able to disclose complex interactions among predictor variables and to quantify their relative importance. We build a classification tree to relate composite fire severity to site conditions (forest cover type, stand characteristics, aspect, terrain slope, fire spread pattern and distance from the edge). Aspect was categorized as N/W (225–44°) or S/E (45–224°), respectively differentiating the moister and the drier aspects. Selection of independent variables was automatic and model overfitting was prevented by basing the number of splits on a 10-fold cross-validation.

## 3. Results

### 3.1. Fire severity in relation to cover type

Most (67%) study plots were burned by surface fire, with percentages of 53, 79, 90 and 97% for PS, evergreen broadleaves (EB), short-needled conifers (SNC) and deciduous broadleaves (DB) cover types, respectively. Fire self-extinguished in four sites, in *Q. pyrenaica* – *B. alba*, *Q. pyrenaica* – *C. sativa*, *B. alba* and *P. sylvestris* – *C. lawsoniana* stands, always in mesic conditions near water streams. Table 4 presents the median values and ranges for stand characteristics and fire severity metrics, which were highly overlapped between cover types. Linear mixed-effects modelling determined that cover type, among other variables, had a significant effect on both crown scorch height and stem char height (Table 5). The models accounted respectively for 61 and 70% of the existing variation. Crown scorch height is significantly lower in DB stands than in PS stands when the other fixed effects in the model are controlled by being set to neutral values (Table 6). However, crown scorch height does not reflect the full fire behaviour range, i.e. plots without scorched trees are excluded from the analysis. The least squares means for stem char height confirm the difference in fire intensity between PS and DB stands and furthermore identify SNC as the least flammable cover type (Table 6).

**Table 5**
Linear mixed models for fire behaviour surrogates: significance (*p*-values) of the fixed effects.

| Fixed effects | Crown scorch height ($n = 144$) | Stem char height ($n = 169$) |
|---|---|---|
| Cover type | <0.001 | <0.001 |
| Stand height | <0.001 | – |
| Crown base height | – | <0.001 |
| Dbh | – | 0.023 |
| Distance to edge | – | 0.002 |
| Aspect | 0.003 | – |

Fig. 3 shows the plot distribution of composite and individual fire severity ratings per cover type, with PS and DB stands occupying the extremes of the fire severity spectrum for all of its components, and EB and SNC stands in an intermediate position. Even though fire crowned in nearly half of PS plots, the fire severity contrast between PS and the other cover types is more evident in ground and understorey layers. Pairwise comparisons – restricted to sites where three transects were sampled – identify a significant effect of cover type (i.e. PS versus OF) on fire severity ratings at all study sites with a few exceptions identified in Table 7.

The distance for reaching the minimum fire severity level in OF was not affected by cover type ($p = 0.229$, median = 21 m). Correlation between the distance for minimum fire severity in OF and the PS-transect composite fire severity suggests ($p = 0.069$) the former increases with the preceding level of fire behaviour. Likewise, positive correlations were found between transect-based fire severities in PS and OF – $p = 0.030$ for ground fire severity, $p < 0.001$

**Table 6**
Least squares means (±standard errors) of fire behaviour surrogates by forest cover type. Means followed by the same letter are not significantly different ($p > 0.05$).

| Cover type | Crown scorch height | Stem char height |
|---|---|---|
| PS | 11.3 ± 0.7 a | 4.8 ± 0.7 a |
| SNC | 9.0 ± 1.1 ab | 0.8 ± 0.9 b |
| EB | 9.2 ± 1.5 ab | 3.5 ± 1.3 abc |
| DB | 8.2 ± 0.8 b | 3.1 ± 0.8 c |

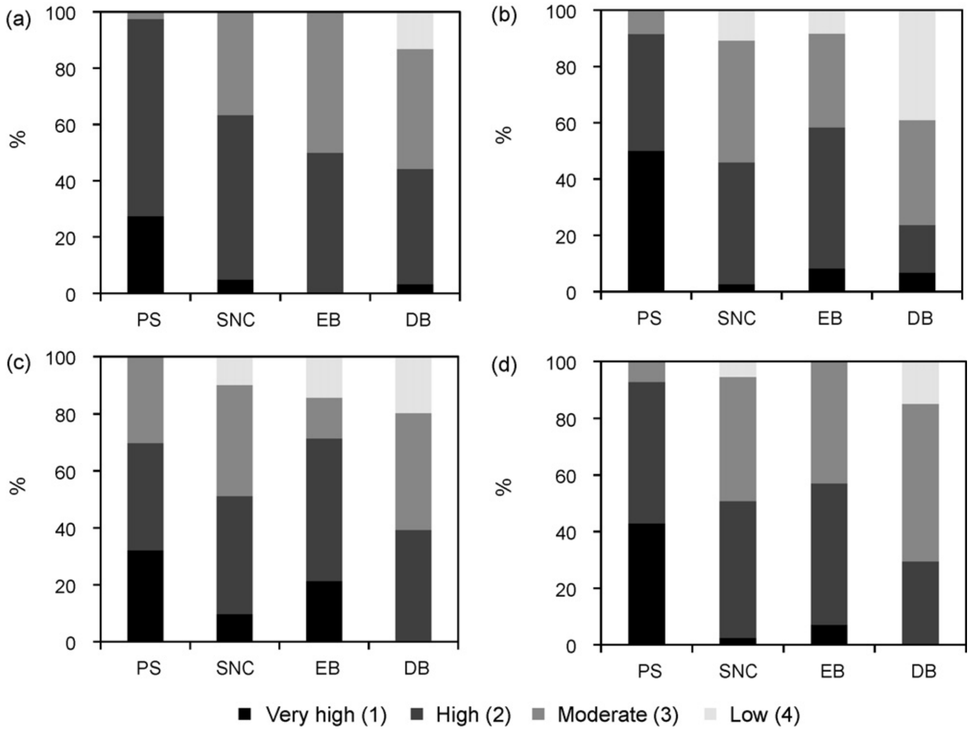


**Fig. 3.** Percentage of plots in each fire severity class by cover type and fire severity component. (a) Ground, (b) understorey, (c) overstorey and (d) composite. PS: maritime pine; SNC: short-needled conifers; EB: evergreen broadleaves; DB: deciduous broadleaves.

for understorey, overstorey and composite fire severities – suggesting limitations in the mitigating effect of OF cover types.

### *3.2. Correlation between fire severity metrics and ratings*

The evidence of association between most fire severity metrics was very strong ($p < 0.001$). Analysis by cover type confirms the correlation patterns and in addition does not provide support for the existence of association between stem char depth and crown scorch height for any of the cover types. Correlations among individual fire severity ratings (ground, understorey and overstorey) were highly significant ($p < 0.001$), but analysis by cover type negates ($p = 0.395$) association between overstorey and ground fire severities in PS stands. The best indicator of composite fire severity was its understorey component ($\tau = 0.87$, $p < 0.001$), except in EB stands, where composite severity was more closely related with ground fire severity ($\tau = 0.83$, $p = 0.002$).

### *3.3. Classification tree analysis of composite fire severity*

The classification tree analysis accounted for 44% of the observed variation in composite fire severity and produced a fifteen-outcome classification tree (Fig. 4). Consistent with the previous analysis, cover type was by far the most relevant variable in explaining fire severity. The plots were split in two groups that matched the PS and OF cover types. Very high to high fire severity is largely prevalent in the PS sub-tree, while all fire severity categories are represented in the OF sub-tree. The first splitting level for PS woodland was based on basal area; fire severity was highest at basal areas <7.0 $m^2$ $ha^{-1}$ while it ranged from moderate to very high when basal area ≥7.0 $m^2$ $ha^{-1}$. Stand height and terrain aspect ranked next in importance, respectively, for basal area above and below 7.0 $m^2$ $ha^{-1}$: fire severity decreased in stands taller than 15.6 m or facing N/W aspects. When basal area ≥7.0 $m^2$ $ha^{-1}$ and stand height <15.6 m, N/W aspects were again a factor associated to fire severity mitigation. Finally, fires in poorly stocked stands (<7.0 $m^2$ $ha^{-1}$) that moved down slope in N/W aspects were less severe than their upslope counterparts.

In OF types the first division is based on whether distance from the edge was ≥9 m, a criterion associated to the prevalence of low or moderate fire severities. Further partitions on the left side of the regression tree are based on tree density, distance from the edge, stand height and basal area, with decreases in fire severity associated to increases in these variables. The lowest fire severity level in OF occurred for tree density ≥1389 $ha^{-1}$, stand height ≥8 m and basal area ≥32.4 $m^2$ $ha^{-1}$. Cover type was the most significant factor explaining fire severity variation when distance from the edge was lower than 9 m; less severe fire was generally experienced in DB stands than in SNC and EB stands, which implies a more immediate change in fire severity. The next splits were based on stem density and aspect; fire severity in DB woodland increased for density ≥1111 stems $ha^{-1}$, while in SNC and EB it was somehow mitigated by N/W aspects. The highest fire severity level in OF happened in the immediate vicinity of PS when SNC and EB stands coincided with S-E aspects. Overall, forest composition, stand characteristics, terrain aspect, distance to the PS–OF edge and fire spread pattern contributed to the tree sum of squares in percentages of 51.3, 28.3, 9.3, 9.1 and 4.5%, respectively.

**Table 7**
Significance of the local differences in plot-level fire severity ratings between maritime pine woodland and other cover types. The *p*-values refer to Wilcoxon tests. Values in bold are non-significant.

| Fire, site | Fire severity | | | |
|---|---|---|---|---|
| | Ground | Understorey | Overstorey | Composite |
| 1, 1 | **0.070** | <0.001 | 0.001 | <0.001 |
| 2, 1 | 0.033 | <0.001 | **0.110** | <0.001 |
| 3, 1 | <0.001 | <0.001 | 0.012 | <0.001 |
| 3, 2 | <0.001 | 0.002 | 0.034 | <0.001 |
| 4, 1 | 0.015 | <0.001 | 0.005 | <0.001 |
| 5, 1 | <0.001 | <0.001 | **0.198** | <0.001 |
| 6, 1 | **0.175** | 0.002 | 0.034 | 0.003 |

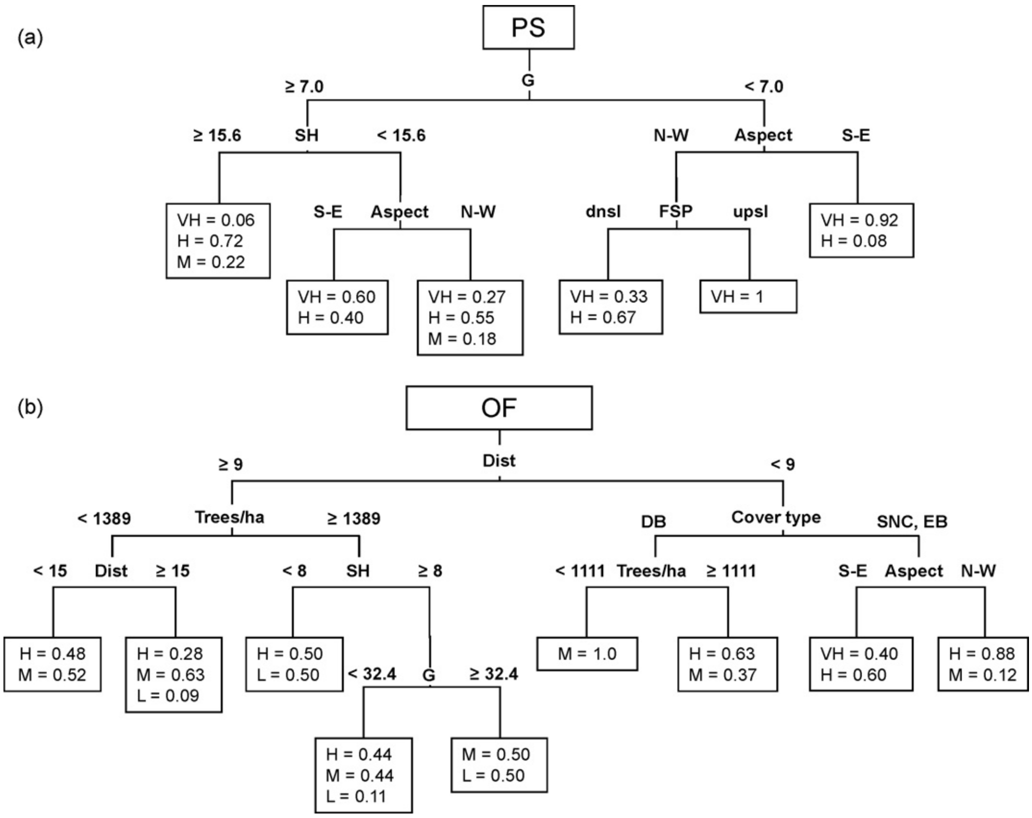


**Fig. 4.** Classification tree for the variables explaining composite fire severity at the plot level. (a) Maritime pine sub-tree and (b) other forest types sub-tree. Numbers at the ends of terminal nodes are the proportion of plots per severity class (VH = very high; H = high; M = moderate; L = low). G: basal area ($m^2\ ha^{-1}$), SH: stand height (m); FSP: fire spread pattern (dnsl = down slope; upsl = upslope and slope neutral); Dist = distance to PS–OF edge (m).

## 4. Discussion

In this study, fire severity decreased significantly from maritime pine woodland to the adjacent broadleaved (deciduous or evergreen) or short-needled conifer forest types. The observed transition in fire severity was often dramatic, namely when fire moved into deciduous forest located on lower slope positions. Although the existing evidence for comparison is limited (Ritchie et al., 2007; Safford et al., 2009), it appears that a change in forest cover type modifies fire behaviour and severity faster than fuel treatments applied in flammable conifer forests. The results agree with reports of more severe fire behaviour and effects in pine forest than in deciduous forest (Kafka et al., 2001; Wang, 2002; Hély et al., 2003; Choung et al., 2004; Epting and Verbyla, 2005; Lee et al., 2009). The differences in fire severity found between the contiguous stands of maritime pine and other forest types are due to dissimilar fire environments, i.e. the compounded effects of topography, weather and fuel on fire behaviour. Terrain aspect differed between the adjacent cover types at three sites only (Table 2), implying that the pattern of fire spread in relation to wind and slope was shared between the stands of maritime pine and other forest types in 10 out of the 13 study sites. Very high to extreme fire danger was common to all sampled wildfires and the adopted sampling procedure was expected to minimize variation in fire severity caused by the spatial and temporal variation in wind and fuel moisture on each site. Consequently, we had anticipated that fuels and stand structure would be the major drivers of within-site variability in fire severity.

Consideration of the effects of local fuel characteristics on fire severity was precluded by the opportunistic nature of the study. However, the results are consistent with a fire hazard modelling analysis for the Portuguese forest cover types (Fernandes, 2009a): while potential fire behaviour in maritime pine stands varies considerably with surface fuel accumulation and stand structure, dense stands of broadleaved species and short-needled conifers tend to be less fire prone, especially when tall. The factors involved include lower in-stand wind speed, higher dead fuel moisture content, higher foliar moisture content, and less flammable litter due to differences in drying and decomposition rates, compactness and chemical composition (Neyisci and Intini, 2006; Drever et al., 2008; Fernandes, 2009a). Mature plantations of short-needled conifers in Portugal usually lack understorey vegetation which combined with the more compact litter probably explains why stem char height (i.e. fire intensity) was higher in deciduous broadleaves than in the former. Regardless of cover type, fire severity will increase as the shrub layer becomes more conspicuous and is more involved in fire spread (Stephens et al., 2008; Thompson and Spies, 2009). However, higher shading levels may change understorey composition by promoting the replacement of sclerophylous species by shade-tolerant and less flammable species (Nowacki and Abrams, 2008).

The broadleaved and short-needled conifer types were sampled at lower slope positions than the contiguous pine stands in all but three study sites (3, 2; 4, 1; 8, 1). The implication is that the observed differences in fire severity likely include an additional meteorological influence which is confounded with the effects of changing the cover type. In comparison with upland forest, the microclimate in riparian areas is cooler, moister and calmer, and vegetation is more mesophytic (Dwire and Kauffman, 2003; Pettit and Naiman, 2007). Deeper organic layers in riparian areas can reduce ground fire severity (Halofsky and Hibbs, 2008). Fire severity in North America has been reported to decrease in moister or riparian forest (Wang, 2002; Jain et al., 2006; Halofsky and Hibbs, 2008), to the point of fire self-extinction (Kobziar and McBride (2006) as we have observed. Ground and understorey fire severities in the adjacent forests were in general more contrasted than overstorey fire severity, which in part may reflect the mesic effect induced by slope position, as suggested by Halofsky and Hibbs (2008). Consequently, the prevailing topographical setting has probably exacerbated the fire severity moderation caused by the modified fuel characteristics and stand structure.

Similarly to other studies (Jain and Graham, 2007; Halofsky and Hibbs, 2009) we have found that fire severity can be dissociated among vegetation layers. Distinct fuel availabilities or the constrainment of crown scorch expression by tree height (Van Wagner, 1973; Michaletz and Johnson, 2006) are plausible explanations for this. The former can be caused by a vertical gradient in dead fuel moisture content (Finney and Martin, 1993; Hille and den Ouden, 2005) or by a gap in fuel continuity precluding vertical fire development (Van Wagner, 1977; Cruz et al., 2004). The fact that understorey fire severity was the fire severity component best related with composite fire severity does not imply the other components can be overlooked. The results indicate that an incomplete or inaccurate depiction of fire severity patterns may arise from an assessment based on information from a single vegetation layer.

The classification tree confirmed and expanded the previous findings. Performance of the classification tree may seem modest ($R^2 = 0.44$) but it is satisfactory, considering that a plot-based assessment of fire severity reflects fine-scale variation in micrometeorology and fuels, and that these factors can be addressed only under experimental conditions. There were no differences in composite fire severity between the cover types bordering maritime pine stands other than the stronger initial impact observed in deciduous forest (Fig. 4), in spite of distinct fire intensity levels in stands of short-needled conifers and deciduous broadleaves. Evidence was found of fire severity decreases in stands with more mature conditions, i.e. lower fire severity was associated to stands with larger trees and higher basal area (Fig. 4). Higher tree density favoured lower fire severity in more mature OF stands but was associated to more severe fire in younger deciduous stands. Forest stands that either are more mature or have higher canopy cover have been associated to lower fire severity or to unburned patches (Odion et al., 2004; Oliveras et al., 2009; Róman-Cuesta et al., 2009). Increasingly dense tree canopies in conifer stands reduce the exposure of surface fuels to wind and solar radiation and minimize understorey vegetation development, hence decreasing surface fire intensity. In turn, denser canopies facilitate crown fire spread (Van Wagner, 1977; Cruz et al., 2005). A multitude of fire modelling studies (e.g. Stephens et al., 2009) and empirical evidence from North-American conifer dry forests (Pollet and Omi, 2002; Agee and Skinner, 2005; Lentile et al., 2006; Ritchie et al., 2007; Safford et al., 2009) indicate that fire severity is lower in open stands, especially when thinning is concurrent with surface fuel treatment. The post-burn proportion of dead pines decreased with increases in both basal area and tree diameter in Catalonia, Spain, but the relative effect of the former was quite low (González et al., 2007). In this study, fire severity was to some extent mitigated in PS stands with higher basal area and taller trees (Fig. 4), but an assessment of the fire severity implications of a more open canopy is not warranted.

Fire severity is dependent on fire behaviour, but because the latter determines how effective fire suppression is, it should be relevant for fire incidence also. Therefore, it is only natural that our fire severity results parallel the findings of fire incidence research in the Iberian Peninsula. Diaz-Delgado et al. (2004) report a trend of decreasing fire occurrence from pine to evergreen broadleaved to deciduous broadleaved forests in Catalonia, Spain. In the same region, fire is less likely to occur in hardwoods (*Q. robur*, *Q. ilex*) and short-needled pines (*P. sylvestris*, *P. uncinata*) than in more flammable pine species, including *P. pinaster*, and stand-level burn probability decreases as tree size and the proportion of hardwood trees increase (González et al., 2006). A general study for Portugal indicates that fire preference is far more affected by forest composition than by stand structure, but the burn probability of deciduous woodland decreases sharply with cumulative vegetation cover (Silva et al., 2009), although this variable may not always relate with basal area. Finally, wildfires in the study region tend to avoid broadleaved woodland and select pine forest and especially shrubland (Moreira et al., 2009).

Terrain aspect affects solar irradiance and water regimes (Swanson et al., 1988), resulting in higher fuel moisture contents and often more mesic plant communities and environments in northern aspects in relation to southern aspects (Schroeder and Buck, 1970). The higher flammability of south-facing slopes generates more severe fires (e.g. Skinner and Taylor, 1998; Alexander et al., 2006; Oliveras et al., 2009). In our study, such pattern was more evident at the high end of the fire severity range and was restricted to maritime pine, short-needled conifers and evergreen broadleaves (Fig. 4); deciduous woodland was always associated with the moister N/W aspect class (Table 2).

## 5. Conclusion

We have shown that significant, and often abrupt, decreases in fire severity can occur when wildfire moves from maritime pine woodland to adjacent broadleaved (deciduous and evergreen) and short-needled conifer forest types. These changes took place despite generalized drought and very high to extreme fire danger. The most immediate decrease in fire severity was revealed in deciduous broadleaved forest, most likely due to moisture-related changes in flammability. However, the final fire severity level was reached at similar distances from the PS–OF boundary and did not change with OF type.

Fire severity was mainly determined by forest cover type (accounting for more than half of the variation explained by the regression tree), but it was also manifest that severe fire is less likely in stands with a more mature structure. Results of this study cannot be generalized to the whole wildfire area, and reflect primarily a fire severity mitigation effect ensuing from down slope fire spread into moister aspects and topographical positions. Further understanding of fire severity patterns in different cover types and stand structures – to disentangle the role of each contributing factor – can be attained through fire modelling, based on representative micrometeorological and fuel data.

Surface and canopy fuel treatments have been demonstrated to be effective at mitigating fire behaviour and severity in maritime pine stands (Fernandes and Rigolot, 2007; Fernandes, 2009b). However, landscape scale implementation of fuel treatments is intrinsically difficult and costly. Our results support a combined strategy where a more prominent role could be played by the expansion of broadleaved deciduous or evergreen tree species and strategic conversion of maritime pine stands to mixed or pure stands of those species. The current successional stage of most oak woodlands in Portugal implies vertical continuity and low stature (Godinho-Ferreira et al., 2005). Consequently, acquisition of stand maturity leading to less severe fire requires a combination of silvicultural practices and increased protection from wildfire disturbance. The benefits would be seen not just in decreased fire severity (and presumably in decreased fire incidence) but also in more fire-resilient ecosystems and landscapes.

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## References

Agee, J.K., Skinner, C.N., 2005. Basic principles of forest fuel reduction treatments. Forest Ecology and Management 211, 83–96.

Alexander, J.D., Seavy, N.E., Ralph, C.J., Hogoboom, B., 2006. Vegetation and topographical correlates of fire severity from two fires in the Klamath–Siskiyou region of Oregon and California. International Journal of Wildland Fire 15, 237–245.

Alexander, M.E., Cole, F.V., 2001. Rating fire danger in Alaska ecosystems. Fireline 12, 2–3.

APA, 2003. Atlas do Ambiente. http://www2.apambiente.pt/atlas/est/index.jsp?zona=continente (Last accessed on 07/10/2009).

Bolker, B.M., Brooks, M.E., Clark, C.J., Geange, S.W., Poulsen, J.R., Henry, M., Stevens, H., White, J.S., 2008. Generalized linear mixed models: a practical guide for ecology and evolution. Trends in Ecology and Evolution 24, 127–135.

Broncano, M.J., Retana, J., 2004. Topography and forest composition affecting the variability in fire severity and post-fire regeneration occurring after a large fire in the Mediterranean basin. International Journal of Wildland Fire 13, 209–216.

Calbk, M.E., White, D., Kiester, A.R., 2002. Assessment of spatial autocorrelation in empirical models of ecology. In: Scott, J.M., Heglund, P.J., Morrison, M.L., Haufler, J.B., Raphael, M.G., Wall, W.A., Samson, F.B. (Eds.), Predicting Species Occurrences: Issues of Scale and Accuracy. Island Press, Washington, DC, pp. 429–440.

Choung, Y., Lee, B., Cho, J., Lee, K., Jang, I., Kim, S., Hong, S., Jung, H., Choung, H., 2004. Forest responses to the large-scale east coast fires in Korea. Ecological Research 19, 43–54.

Cruz, M.G., Alexander, M.E., Fernandes, P.A.M., 2008. Development of a model system to predict wildfire behaviour in pine plantations. Australian Forestry 71, 113–121.

Cruz, M.G., Alexander, M.E., Wakimoto, R.H., 2004. Modeling the likelihood of crown fire occurrence in conifer forest stands. Forest Science 50, 640–658.

Cruz, M.G., Alexander, M.E., Wakimoto, R.H., 2005. Development and testing of models for predicting crown fire rate of spread in conifer forest stands. Canadian Journal of Forest Research 35, 1626–1639.

Death, C., Fabricius, K., 2000. Classification and regression trees: a powerful yet simple technique for ecological data analysis. Ecology 81, 3178–3192.

Diaz-Delgado, R., Lloret, F., Pons, X., 2004. Spatial patterns of fire occurrence in Catalonia, NE, Spain. Landscape Ecology 19, 731–745.

Drever, C.R., Drever, M.C., Messier, C., Bergeron, Y., Flannigan, M., 2008. Fire and the relative roles of weather, climate and landscape characteristics in the Great Lakes St. Lawrence forest of Canada. Journal of Vegetation Science 19, 57–66.

Dwire, K.A., Kauffman, J.B., 2003. Fire and riparian ecosystems in landscapes of the western USA. Forest Ecology and Management 178, 61–74.

Epting, J., Verbyla, D., 2005. Landscape-level interactions of prefire vegetation, burn severity, and postfire vegetation over a 16-year period in interior Alaska. Canadian Journal of Forest Research 35, 1367–1377.

Fernandes, P.M., 2001. Fire spread prediction in shrub fuels in Portugal. Forest Ecology and Management 144, 67–74.

Fernandes, P.M., 2009a. Combining forest structure data and fuel modelling to assess fire hazard in Portugal. Annals of Forest Science 66, 415p1–415p9.

Fernandes, P.M., 2009b. Examining fuel treatment longevity through experimental and simulated surface fire behaviour: a maritime pine case study. Canadian Journal of Forest Research 39, 2529–2535.

Fernandes, P.M., Botelho, H.S., Rego, F.C., Loureiro, C., 2009. Empirical modelling of surface fire behaviour in maritime pine stands. International Journal of Wildland Fire 18, 698–710.

Fernandes, P.M., Catchpole, W.R., Rego, F.C., 2000. Shrubland fire behaviour modelling with microplot data. Canadian Journal of Forest Research 30, 889–899.

Fernandes, P.M., Rigolot, E., 2007. Fire ecology and management of maritime pine (*Pinus pinaster* Ait.). Forest Ecology and Management 241, 1–13.

Fernandes, P.M., Vega, J.A., Jimenez, E., Rigolot, E., 2008. Fire resistance of European pines. Forest Ecology and Management 256, 246–255.

Finney, M.A., Martin, R.E., 1993. Modeling effects of prescribed fire on young-growth coast redwood trees. Canadian Journal of Forest Research 23, 1125–1135.

Finney, M.A., McHugh, C.W., Grenfell, I.C., 2005. Stand- and landscape-level effects of prescribed burning on two Arizona wildfires. Canadian Journal of Forest Research 35, 1714–1722.

Fowler, J.F., Sieg, C.H., 2004. Postfire Mortality of Ponderosa Pine and Douglas-fir: A Review of Methods to Predict Tree Death. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO (General Technical Report RMRS-GTR-132).

Godinho-Ferreira, P., Azevedo, A., Rego, F., 2005. Carta da tipologia florestal de Portugal Continental. Silva Lusitana 13, 1–34.

González, J.R., Palahí, M., Trasobares, A., Pukkala, T., 2006. A fire probability model for forest stands in Catalonia (north-east Spain). Annals of Forest Science 63, 169–176.

González, J.R., Trasobares, A., Palahi, M., Pukkala, T., 2007. Predicting stand damage and tree survival in burned forests in Catalonia (North-East Spain). Annals of Forest Science 64, 733–742.

Halofsky, J.E., Hibbs, D.E., 2008. Determinants of riparian fire severity in two Oregon fires, USA. Canadian Journal of Forest Research 38, 1959–1973.

Halofsky, J.E., Hibbs, D.E., 2009. Relationships among indices of fire severity in riparian zones. International Journal of Wildland Fire 18, 584–593.

Hély, C., Flannigan, M., Bergeron, Y., 2003. Modeling tree mortality following wildfire in the Southeastern Canadian mixed-wood boreal forest. Forest Science 49, 566–576.

Heyerdahl, E.K., Brubaker, L.B., Agee, J.K., 2001. Spatial controls of historical fire regimes: a multiscale example from the interior west, USA. Ecology 82, 660–678.

Hille, M., den Ouden, J., 2005. Fuel load, humus consumption and humus moisture dynamics in Central European Scots pine stands. International Journal of Wildland Fire 14, 153–159.

Hurlbert, S.H., 1984. Pseudoreplication and the design of ecological field experiments. Ecological Monographs 54, 187–211.

Jain, T.B., Graham, R.T., 2007. The relation between tree burn severity and forest structure in the Rocky Mountains. In: Powers, R. (Ed.), Restoring Fire-Adapted Forested Ecosystems: Proceedings of the 2005 National Silviculture Workshop. USDA Forest Service, Pacific Southwest Research Station, Albany, CA, pp. 213–250 (General Technical Report PSW-GTR-203).

Jain, T.B., Graham, R.T., Pilliod, D.S., 2006. The relation between forest structure and soil burn severity. In: Andrews, P., Butler, B. (Comps.), Fuel Management—How to Measure Success: Conference Proceedings. USDA Forest Service Proceedings RMRS-P-41, Fort Collins, CO.

Kafka, V., Gauthier, S., Bergeron, Y., 2001. Fire impacts and crowning in the boreal forest: study of a large wildfire in western Quebec. International Journal of Wildland Fire 10, 119–127.

Knapp, E.E., Keeley, J.E., 2006. Heterogeneity in fire severity within early season and late season prescribed burns in a mixed conifer forest. International Journal of Wildland Fire 15, 1–9.

Keeley, J.E., 2009. Fire intensity, fire severity and burn severity: a brief review and suggested usage. International Journal of Wildland Fire 18, 116–126.

Kobziar, L.N., McBride, J.R., 2006. Wildfire burn patterns and riparian vegetation response along two northern Sierra Nevada streams. Forest Ecology and Management 222, 254–265.

Lee, S., Lee, M., Lee, Y., Won, M., Kim, J., Hong, S., 2009. Relationship between landscape structure and burn severity at the landscape and class levels in Samchuck, South Korea. Forest Ecology and Management 258, 1594–1604.

Lentile, L.B., Smith, F.W., Shepperd, W.D., 2006. Influence of topography and forest structure on patterns of mixed severity fire in ponderosa pine forests of the South Dakota Black Hills, USA. International Journal of Wildland Fire 15, 557–566.

Michaletz, S.T., Johnson, E.A., 2006. A heat transfer model of crown scorch in forest fires. Canadian Journal of Forest Research 36, 2839–2851.

Miller, J.D., Safford, H.D., Crimmins, M., Thode, A.E., 2009. Quantitative evidence for increasing forest fire severity in the Sierra Nevada and Southern Cascade mountains, California and Nevada, USA. Ecosystems 12, 16–32.

Moreira, F., Rego, F.C., Ferreira, P.G., 2001. Temporal (1958-1995) pattern of change in a cultural landscape of northwestern Portugal: implications for fire occurrence. Landscape Ecology 16, 557–567.

Moreira, F., Vaz, P., Catry, F., Silva, J.S., 2009. Regional variations in wildfire susceptibility of land-cover types in Portugal: implications for landscape management to minimize fire hazard. International Journal of Wildland Fire 18, 563–574.

Moreno, J.M., Vásquez, A., Vélez, R., 1998. Recent history of forest fires in Spain. In: Moreno, J. (Ed.), Large Forest Fires. Backhuys Publishers, Leiden, pp. 159–185.

Neyisci, T., Intini, M., 2006. The use of cypress barriers for limiting fires in Mediterranean countries. In: Il Cipresso e gli incendi. Arsia, Firenze, pp. 3–18.

Nowacki, G.J., Abrams, M.D., 2008. The demise of fire and "mesophication" of forests in the eastern United States. BioScience 58, 123–138.

Nunes, M.C.S., Vasconcelos, M.J., Pereira, J.M.C., Dasgupta, N., Alldredge, R.J., Rego, F.C., 2005. Land cover type and fire in Portugal: do fires burn land cover selectively? Landscape Ecology 20, 661–673.

Odion, D.C., Frost, E.J., Strittholt, J.R., Jiang, H., Dellasala, D.A., Moritz, M.A., 2004. Patterns of fire severity and forest conditions in the western Klamath Mountains, California. Conservation Biology 18, 927–936.

Oliveras, I., Gracia, M., Moré, G., Retana, J., 2009. Factors influencing the pattern of fire severities in a large wildfire under extreme meteorological conditions in the Mediterranean basin. International Journal of Wildland Fire 18, 755–764.

Palheiro, P., Fernandes, P., Cruz, M.G., 2006. A fire behaviour-based fire danger classification for maritime pine stands: comparison of two approaches. Forest Ecology and Management 234, S1.

Pereira, J.M.C., Santos, M.T.N., 2003. Áreas queimadas e risco de incêndio em Portugal. MADRP, Direcção-Geral das Florestas, Lisboa.

Pereira, M.G., Trigo, R.M., da Câmara, C.C., Pereira, J.M.C., Leite, S.M., 2005. Synoptic patterns associated with large summer forest fires in Portugal. Agricultural and Forest Meteorology 129, 11–25.

Pettit, N.E., Naiman, R.J., 2007. Fire in the riparian zone: characteristics and ecological consequences. Ecosystems 10, 673–687.

Podur, J.J., Martell, D.L., 2009. The influence of weather and fuel type on the fuel composition of the area burned by forest fires in Ontario, 1996–2006. Ecological Applications 19, 1246–1253.

Pollet, J., Omi, P.N., 2002. Effect of thinning and prescribed burning on crown fire severity in ponderosa pine forests. International Journal of Wildland Fire 11, 1–10.

Pyne, S., Andrews, P.L., Laven, R.D., 1996. Introduction to Wildland Fire. John Wiley & Sons, Inc., New York.

Rego, F.C., 2001. Florestas Públicas. MADRP, Direcção-Geral das Florestas, Lisboa.

Reinhardt, E.D., Keane, R.E., Calkin, D.E., Cohen, J.D., 2008. Objectives and considerations for wildland fuel treatment in forested ecosystems of the interior western United States. Forest Ecology and Management 256, 1997–2006.

Ritchie, M.W., Skinner, C.N., Hamilton, T.A., 2007. Probability of tree survival after wildfire in an interior pine forest of northern California: effects of thinning and prescribed fire. Forest Ecology and Management 247, 200–208.

Róman-Cuesta, R.M., Gracia, M., Retana, J., 2009. Factors influencing the formation of unburned forest islands within the perimeter of a large fire. Forest Ecology and Management 258, 71–80.

Rothermel, R.C., 1972. A Mathematical Model for Predicting Fire Spread in Wildland Fuels. USDA Forest Service, Intermountain Forest and Range Experiment Station, Ogden, UT (Research Paper INT-115).

Ryan, K.C., 1982. Techniques for assessing fire damage to trees. In: Lotan, J. (Ed.), Proceedings of the Symposium: Fire, its Field Effects. Intermountain Fire Council, pp. 1–11.

Ryan, K.C., Noste, N.V., 1985. Evaluating prescribed fires. In: Lotan, J.E., Kilgore, B., Fischer, W., Mutch, R. (Tech. Coords.), Proceedings—Symposium and Workshop on Wilderness Fire. USDA Forest Service, General Technical Report INT-182, Intermountain Forest and Range Experimental Station, Ogden, UT, pp. 230–238.

Safford, H.D., Schmidt, D.A., Carlson, C.H., 2009. Effects of fuel treatments on fire severity in an area of wildland-urban interface, Angora Fire, Lake Tahoe Basin, California. Forest Ecology and Management 258, 773–787.

Schelhaas, M., Nabuurs, G., Schuck, A., 2003. Natural disturbances in the European forests in the 19th and 20th centuries. Global Change Biology 9, 1620–1633.

Schroeder, M.J., Buck, C.C., 1970. Fire weather: a guide for application of meteorological information to forest fire control operations. Agriculture Handbook, vol. 360. USDA Forest Service, Washington.

Silva, J.S., Moreira, F., Vaz, P., Catry, F., Godinho-Ferreira, P., 2009. Assessing the relative fire proneness of different forest types in Portugal. Plant Biosystems 143, 597–608.

Skinner, C.N., Taylor, A.H., 1998. Fire history and landscape dynamics in a late-successional reserve, Klamath Mountains, California, USA. Forest Ecology and Management 111, 285–301.

Stephens, S.L., Fry, D.L., Franco-Vizcaino, E., 2008. Wildfire and spatial patterns in forests in Northwestern Mexico: The United States wishes it had similar fire problems. Ecology and Society 13 (2), 10 [online] URL: http://www.ecologyandsociety.org/vol13/iss2/art10/.

Stephens, S.L., Moghaddas, J.J., Edminster, C., Fiedler, C.E., Haase, S., Harrington, M., Keeley, J.E., Knapp, E.E., McIver, J.D., Metlen, K., Skinner, C.N., Youngblood, A., 2009. Fire treatment effects on vegetation structure, fuels, and potential fire severity in western U.S. forests. Ecological Applications 19, 305–320.

Swanson, F.J., Kratz, T.K., Caine, N., Woodmansee, R.G., 1988. Landform effects on ecosystem patterns and processes. BioScience 38, 92–98.

Thompson, J.R., Spies, T.A., 2009. Vegetation and weather explain variation in crown damage within a large mixed-severity wildfire. Forest Ecology and Management 258, 1684–1694.

Van Mantgem, P.J., Schwilk, D.W., 2009. Negligible influence of spatial autocorrelation in the assessment of fire effects in a mixed conifer forest. Fire Ecology 5, 116–125.

Van Wagner, C.E., 1973. Height of crown scorch in forest fires. Canadian Journal of Forest Research 3, 373–378.

Van Wagner, C.E., 1977. Conditions for the start and spread of crown fire. Canadian Journal of Forest Research 7, 23–34.

Van Wagner, C.E., 1987. Development and Structure of the Canadian Forest Fire Weather Index System. Canadian Forestry Service, Ottawa, Ontario.

Viegas, D.X., Piñol, J., Viegas, M.T., Ogaya, R., 2001. Estimating live fine fuels moisture content using meteorologically-based indices. International Journal of Wildland Fire 10, 223–240.

Wang, G.G., 2002. Fire severity in relation to canopy composition within burned boreal mixedwood stands. Forest Ecology and Management 163, 85–92.