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Tree type and forest management effects on the structure of stream wood following wildfires

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A B S T R A C T

Wildfires are an increasingly common disturbance influencing wood recruitment to streams, and thereby affecting their physical and biological condition. Mediterranean countries such as Portugal, where more than 25% of the land area has burned since 1990, are ideal areas to study impacts of wildfire effects on streams. We evaluated the physical structure of 2206 downed wood pieces (DWP) across 27 first- to third-order streams in central Portugal, all of which had experienced recent wildfires. The streams flowed through monospecific upland forests of Eucalyptus, Maritime pines, or Cork oaks and were fringed by a mixture of riparian tree species. DWP structure differed between tree types and between burned and unburned pieces. Post-fire timber-production forests (Maritime pines and Eucalyptus) contributed a higher quantity of thinner, longer and straighter DWP to streams than Cork oak stands. Pieces from Maritime pines had more rootwads and branches than DWP from the other tree types. Pieces from Cork oak and riparian species generally had a bent form, were shorter and had no rootwads. Burned DWP in streams were often from riparian trees. Relative to unburned DWP, the burned DWP occurred more frequently, were larger and straighter, had branches less often, and were more decayed. With more complex branches, rootwads, and a larger diameter, inputs from burned Maritime pine forests are more likely to change stream hydraulics and habitat complexity, relative to inputs from Eucalyptus forests with their simpler structure. This study shows that, less than a decade after wildfires, structure of downed wood in and near streams is strongly influenced by wildfire, but also still reflects intrinsic species characteristics and respective silviculture practices, even after the effects of fire have been accounted for. Under an anticipated shift in landscape cover with higher shrubland proportions and more mixing of Maritime pine and Eucalyptus forests, our results suggest that instream large wood will become scarcer and more structurally homogeneous.

1. Introduction

The amount and characteristics of wood delivered from forests to streams depends on the type of forest supplying the wood (Evans et al., 1993) and the processes that introduce it into the stream channels. Processes that can affect wood recruitment vary by region, but generally include biological processes such as insect outbreaks and disease, and abiotic processes such as fire, floods, bank erosion, wind storms, ice storms, and snow avalanches (Naiman et al., 2005; Resh et al., 1988). The volume and type of wood entering streams is a function of the synergy between dominant input processes and the susceptibility of riparian trees to those processes (Bendix and Cowell, 2010a). Ultimately, the recruitment, characteristics, transport and storage of wood can affect a stream's physical and biological condition, through direct and indirect mechanisms (Chen et al., 2008; Everett and Ruiz, 1993; Gurnell et al., 2002; Schneider and Winemiller, 2008). The current study focuses on the impacts of wildfire, an increasingly common disturbance, on the characteristics of wood recruited to streams. In addition to changing wood mass, fire changes the form of wood pieces and alters post-fire decomposition rates (Harmon, 1992). The specific effects of fire on characteristics of wood recruited to streams remain poorly understood.

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A changing global climate and continued anthropogenic activities are combining to increase the probability and severity of fire in Europe and around the world (Flannigan et al., 2009; Moriondo et al., 2006). Fire is a disturbance process affecting trees in both riparian zones and adjacent slopes. Trees injured by fire are more susceptible to mortality via windthrow and disease, and thereby enter fluvial systems more readily (Benda et al., 2003; Pinto et al., 2009). Given the importance of large wood in streams (Gregory et al., 2003), fire-induced changes to the quantity and character of large wood inputs have the potential to cause shifts in stream ecosystems. The degree to which fire affects wood recruitment is dependent on a number of factors, including the type (surface or crown) and intensity of fire, the tree species resistance to fire, age and size of the trees, and the time since the previous fire (Brais et al., 2005; Harper et al., 2005). Variability in the quantity, size, decay and wood form of downed wood pieces (DWP) is influenced by the characteristics of forest fuels including flammability, combustibility and wood density (Boulanger et al., 2011; Browne, 1958; DeLuca and Aplet, 2008) and, after the fire, by elements like post-fire tree resistance and forest management (Brassard and Chen, 2008; Tinker and Knight, 2000). Therefore, the post-fire physical characteristics and quantity of DWP on forest floors (Pedlar et al., 2002) and in "burned streams" differs according to forest type.

Among the structural characteristics of DWP in streams, there is a set of core variables that can be easily measured for comparative purposes and that interact with the stream to influence the function and transport potential of a given piece of wood (Bocchiola et al., 2006; Gregory et al., 2003; Wohl et al., 2010). Wood size can influence the degree to which wood affects biological diversity and biota abundance (Lester et al., 2009), and wood size in relation to the channel width and depth is a primary control on wood stability in streams (Cadol and Wohl, 2010; Haga et al., 2002; Merten et al., 2010, 2011), which, in turn, influences channel morphology (Andreoli et al., 2007; Comiti et al., 2008; Jackson and Sturm, 2002). Abbe and Montgomery (2003) found that wood longer than half the bankfull width tends to form key pieces in logjams. Beyond size, the shape of wood is also important (O'Connor, 1991). A rootwad, for example, raises the center of mass of a wood piece and is therefore a fundamental control on log stability (Braudrick and Grant, 2000, 2001). DWP with stout branches have a geometry that extends well beyond their bole diameter, whereas pieces without branches may be transported more readily and routed through river systems.

Others have evaluated the impact of fire on wood loading to streams (Arseneault et al., 2007; Jones and Daniels, 2008; Young, 1994; Zelt and Wohl, 2004), however, most have studied a single large fire event and thus had little replication. This lack of replication decreases the applicability and generalization of results to different forest situations, where factors such as forest age, time since the last fire, methods of post-fire logging and silviculture practices often differ. No previous studies have focused on differences in the abundance and structure of DWP in streams across multiple single-species forest stands impacted by fire. Others have examined fire susceptibility of different species and input in streams of mixed-forest systems (Arseneault et al., 2007; Jones and Daniels, 2008; Young, 1994; Zelt and Wohl 2004), but not in single-species stands where differences between forests may be more clear.

In this study, we evaluated the physical structure of wood in 27 streams in central Portugal. All streams experienced recent fires and they encompassed three different upland forest types. This is the first study to quantify species-specific differences in DWP that eventually is recruited into streams. Contrary to previous studies neglecting source trees, we hypothesized that, once burned, downed wood from separate species retain some differences in terms of their potential effect on stream ecosystem structure and function. We aimed to address the following specific questions: in areas where fire occurred less than 10 years prior, (i) To what extent does DWP still retain species-specific physical architecture of the pre-burned trees? (ii) How does the physical structure of DWP that moves into streams reflect prior production silviculture practices? and (iii) For DWP within streams, how does wood structure differ between burned and unburned pieces and what are the long-term potential implications for stream wood function and movement after wildfires?

2. Methods

2.1. Study area

This study was conducted from fall 2009 to fall 2010 in nine sub-basins of the Tagus River, which experienced wildfires between 2003 and 2007. Sub-basins are located in east-central Portugal between latitude 39°16′–39°39′N and longitude 7°30′–8°14′W. The resident population within the study area was ca. 60 thousand people, although across the selected sub-basins the human presence is scattered. Climate is Mediterranean with hot, dry summers and cool, wet winters. The mean annual precipitation from 2005 to 2010 was 512 mm (ranging from 3 mm in July to 82 mm in November) and the mean annual temperature was 15.8 °C (range: 9° in December–January to 23 °C in July–August). The area has gentle slopes with altitudes ranging from 19 to 643 m (mean elevation ~266 m). Land cover is dominated by forests, shrublands and agriculture (Table 1). Burned areas of maritime pine (*Pinus pinaster*) are now in a shrubland-like structure, with dense growth of *Erica* spp., *Cistus* spp. and *Ulex* spp. along with young post-fire maritime pine. Less dense shrublands, mainly *Cistus* spp. and *Ulex* spp., are present in the understory of burned (but already recovered) stands of eucalyptus (*Eucalyptus globulus*). Ferns (*Pteridium aquilinum*) are common understory plants in more humid burned zones. In contrast, cork oak (*Quercus suber*) drainage areas usually have bare soil surface with some low understory grasses.

Table 1
Characteristics of the sub-basins from where sites were selected. Ec = Eucalyptus; MP = Maritime pine; CO = Cork oak.

Sub-basin	Maximum stream order	Drainage area (km^2)	Mean stream gradient (%)	Percentage burned	Year 2000 forest/shrubs/agriculture (%)	Dominant forest
Abrançalha	3	26	4.5	75	68/20/11	Ec
Alferreira	3	59	4.5	92	53/23/24	
Palhais	3	49	3.8	47	35/27/38	
Arcês	4	50	5.4	39	40/26/34	MP
Rio Frio	3	37	5.1	71	55/22/22	
Eiras	4	143	5.6	64	59/28/12	
Fouvel	3	50	5.2	66	41/32/27	CO
Salgueira	3	81	2.3	86	40/26/33	
Vale da Lama	3	36	3.1	83	82/4/15	

2.2. Study sites

We selected nine burned sub-basins representing three dominate forest types: eucalyptus (Ec), maritime pine (MP) and cork oak (CO), with three replicate sub-basins in each forest type. Although two of the selected sub-basins have a maximum stream order of 4 (Table 1), we restricted our assessments to stream orders 1, 2 and 3 within each sub-basin. In total, 27 burned reaches of ~500 m each were assessed, totaling ~13,460 m of stream channel (Fig. 1).

For a statistical analysis comparing wood characteristics, there would, ideally, be an equal number of DWP that were burned and not burned for each species. However, it is difficult to have a well-balanced design to study the effects of large natural disturbances (Wiens and Parker, 1995; Reich et al., 2001; Parker and Wiens, 2005), and we did not expect *a priori* to find equal frequencies of instream DWP originating from riparian trees compared to upland forests. To circumvent this instream bias in sample size, we added 100–200 m transects perpendicular to the stream and immediately beyond the riparian area for each study reach when possible. We considered these transects to include wood that could potentially enter the stream.

2.3. Tree types characterizations

In the study area, MP is grown for timber in pure (monoculture) stands. These trees have a pyramidal structure with branches forming an acute angle with the trunk. Trees reach ~5–25 m tall and ~7–50 cm diameter at breast height (DBH) (Catry et al., 2010). MP wood is heavy and moderately hard (Carvalho, 1997). Each year, the crown increases one or two increments, each consisting of 5–7 branches. Trunks have a cylindrical straight shape and branches are relatively weak. The bark is thick and has fissures. Roots penetrate poorly in compacted soils and are easily impeded by obstacles below ground (Correia et al., 2007), making trees susceptible to falling on steep hillslopes. In Portugal, MP is the species most affected by fire (Moreira et al., 2009; Silva et al. 2009), with high flammability resulting from volatile compounds (e.g. resin), fuel accumulation, including a well-developed shrub understory, and stand structure. Although thick bark enables MP to withstand low to moderate intensity fires (Fernandes and Rigolot, 2007), trees usually die (especially younger trees) as a result of intense wildfires and fall within 1–3 years, with almost unaltered overall structure. As far as we could observe in the MP study sites, no post-fire logging was carried out and fire-killed trees were left on the ground on stream side-slopes.

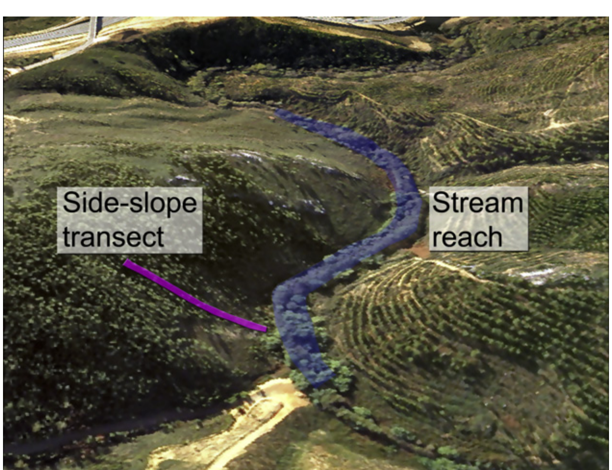

Fig. 1. Digital terrain model of a typical study site, including the stream reach and a perpendicular side-slope transect.

Trees of Ec are planted as pure stands for paper pulp production. These trees have a slender and upright trunk (~6–31 m tall; 5–60 cm DBH) which, when mature, contains almost 90% of the tree biomass (Pereira, 2007). The bark is smooth and thin and is replaced over time. About 90% of the roots are very thin and highly branched at the surface. The wood is denser and harder than MP (Carvalho, 1997). Eucalyptus stands are also strongly affected by fire (Moreira et al., 2009; Silva et al., 2009), but their rapid growth allows fast recovery. The stumps coppice method of regeneration was the most common management strategy in our study area for trees that were top-killed by fire.

At the study area, CO stands (~4–14 m tall; ~10–45 cm DBH) were mainly managed according to an agro-forestry system named "montado" (*dehesa* in Spain). In this system trees are relatively scattered (30–60 trees per ha), are harvested for cork each 9–12 years without felling (Bugalho et al., 2011), and grow among pastures and cereal crops (Silva et al., 2009). CO trees have branches with an irregular architecture, a short trunk of very hard wood (Carvalho, 1997) and fissured, thick cork bark with insulating properties. CO has a very deep root system (Pausas et al., 2009), anchoring trees against uprooting. When compared with MP and Ec in the study area, the CO affected by fire are generally much older trees. Although CO trees are the least prone to burning and are very resilient to fire, many trees were killed in the study area in recent fires (2003–2007). CO is the only native European tree with above-ground sprouting capability similar to Ec (Silva and Catry, 2006). Branch pruning (with slash usually left on the ground), plowing and shrub clearing were the most common management practices in our study area for the rehabilitation of trees that were burned but not killed.

Most streams in the maritime pine and eucalyptus forests had riparian buffers in place, however, gaps longer than 10 m adjacent to the stream channel where managed forests occur along stream margins were common. Where present, buffers were fairly narrow (3–15 m from each margin) and were generally equal to or less than the height of managed forest trees prior to the fire. In southern parts, riparian buffers are narrower with only semi-continuous 'riparian galleries' (Ferreira et al., 2005). There was occasional evidence of logging of riparian species (e.g., saw cuts on DWP). The riparian vegetation was dominated by ash (*Fraxinus angustifolia*), alder (*Alnus glatinosa*), black poplar (*Populus nigra*) and willow (*Salix atrocinerea*, *S. alba*, *S. salvifolia*), frequently surrounded by edges of bramble-thicket (*Rubus ulmifolius*). In most southern areas, hawthorn (*Crataegus monogyna*) is another common species (Aguiar et al., 2000). Besides indigenous species, silver wattle (*Acacia dealbata*), an exotic invasive tree and highly resilient to fire, is widespread across the surveyed riparian zones (Silva et al., 2011).

2.4. Fire data, land-cover and streams

The years 2003 and 2005 were the two highest fire years in the existing record for Portugal (e.g. Viegas et al., 2006), particularly in the east-central areas where this study was conducted. These fires, and the fact that they affected vast areas of MP, Ec and CO forest types, created a unique opportunity to conduct this study across basins but within a relatively uniform climatic region. Yearly maps of fire scars (⩾5 ha) during the period 2003–2007 were assessed from cartography available in vector format from the National Forest Authority. Fire polygons were overlapped with land-cover maps of the region. Our base maps included the 1990 land-cover map for Portugal (reference scale 1:25 000) developed by the National Center for Geographic Information, the CORINE (Coordination of Information on the Environment) land-cover data (1:100 000) from European Environment Agency (years 2000 and

2006) and data points from National Forest Inventories from 1995 and 2005. Computations for the sub-basins and stream orders (Strahler, 1957) were made using GIS processing of a 25-m digital elevation dataset and a 1:25 000-scale hydrography network. Finally, the information on these layers was overlaid and cross-tabulated, and sites checked with field reconnaissance.

2.5. Data collected on DWP

Each study site included one representative 500-m reach complemented whenever possible with a 100–200 m perpendicular transect onto the adjacent hillside. Census techniques (Diez et al., 2001; Elosegi et al., 1999) and the line-intercept method (Van Wagner, 1968) were used along stream reaches and corresponding burned valleys, respectively. Thus, for pieces that lie along the slope of the valley, only those intercepting the line transects were considered and its total length was measured. We included for measurement of DWP (diameter $\geqslant$0.05 m; length $\geqslant$0.5 m) wood pieces that were still rooted but entirely dead, or still alive but entirely uprooted (Merten et al., 2010). However, we excluded snags (following Young et al. (2006), defined as pieces leaning or suspended over the stream at an angle greater than 30°), stumps (which would be outliers, at least in terms of diameter and length) and jams (>2 logs). Inside each stream reach, only DWP intercepting bankfull boundaries were included in the tallies, but the total length of each piece was recorded. Thus, for each DWP, for transects inside or outside of streams, a single person recorded the following:

(i) The total estimated *length* (excluding rootwads) of the piece for the portion over 0.01 m in diameter;
(ii) A single *diameter* measurement taken from a central point. Wood lengths were estimated to the nearest 0.2 m and diameters to the nearest 0.005 m. All estimates were verified for the first 20 pieces in each transect with a tape. Field tapes were also used for all larger pieces (lengths >6 m or diameters >15 cm), where errors were likely to be greater;
(iii) Effects of fire on the DWP (*burned status*) assessed by the amount of charred bark and sapwood (following Jones and Daniels (2008), where 0 = no char, 1 = charred bark but outermost ring present in at least one part of the circumference, and 2 = charred bark and sapwood resulting in significant ring loss);
(iv) *Tree type* (maritime pine, eucalyptus, cork oak, or "riparian species") was identified by assessing morphological characteristics of the DWP;
(v) *Decay* classes were adapted from Jones and Daniels (2008) during the fieldwork data collection but were later simplified to sound or decayed wood;
(vi) Class of the DWP *form* (straight; bent; strongly bent);
(vii) Presence of *rootwads* (yes/no);
(viii) Presence of *branches* (yes/no).

2.6. Statistical analysis

To make the design more balanced and results more interpretable, the variables burned status, form and decay were ultimately reduced from a series of categories to simple binary criteria: unburned/burned wood, straight/bent wood and sound/decayed wood. All analyses were made using the statistical software R (available online at http://www.r-project.org/). The Box–Cox family of transformations was used to find the best transformation for meeting normality and homoscedasticity assumptions when necessary (Quinn and Keough, 2002; Sokal and Rohlf, 2009). The analysis had three components:

(i) *Comparing proportions of burned/unburned DWP according to tree type*: A frequency analysis was conducted comparing patterns in counts of burned and unburned DWP across tree types in a 4 × 2 contingency table. We then explored the pattern of standardized residuals to reveal which cross classifications deviated the most and in what direction from the expected values, thus contributing the most to the lack of independence between burned status and species of DWP.
(ii) *Comparing key characteristics according to DWP tree type and DWP burned status*: Contingency tables were also performed for each categorical variable (branches, rootwads, form and decay), but separating burned from unburned counts of each tree type (resulting in individual 8 × 2 contingency tables). For variables diameter and length, a two-factor unbalanced model I ANOVA (using type III sums of squares in *F*-ratios) was used to test main effects and interactions of tree type and burned status.
(iii) *Comparing key characteristics of DWP inside streams according to DWP burned status*: Besides individual contingency tables for categorical variables, differences in diameter and length between burned/unburned DWP in streams were investigated by two randomization *t*-tests (with 5000 randomizations).

3. Results

A total of 2206 DWP were tallied, with counts distributed as shown in Table 2 by tree type, location and burned status. As expected, the number of DWP was unbalanced among tree types and burned status. The number of CO pieces was lower than the others and frequency of pieces from riparian trees (Ri) was higher. Also, the number of burned DWP was more than double the unburned DWP. In MP, the number of unburned DWP was particularly low, with 6 unburned pieces and 556 burned (Table 2).

3.1. Comparing the proportions of burned/unburned DWP according to tree type

There was a clear lack of independence between DWP tree type and burned status (Fig. 2, χ^2 = 548.5, P < 0.001). Proportionally, only Ri had more unburned than burned pieces, which, along with the high proportion of burned pieces for MP, contributed greatly to the association of tree type and burned status. The proportion of unburned pieces was higher in Ec than in MP (having comparable sample sizes), whereas Ec and CO had equal proportions of burned/unburned pieces (but unequal samples).

Table 2
Counts of down wood pieces ($\geqslant$ 0.05 m; total length $\geqslant$0.5 m) by tree type, location and burned status (unknown burned status refers to inconclusive visual assessments of this variable). Values represent the number of pieces of each tree type across all sites intercepting either the 100–200 m transects perpendicular to the 500 m stream reaches (Side-slope rows) or the bankfulls (In-stream rows). Tree types included are Eucalyptus (Ec), Maritime pines (MP), Cork oaks (CO) and Riparian trees (Ri).

Burned status	Location	Number of pieces by tree type				*Grand total*
		CO	Ec	MP	Ri	
Unburned	In-stream	13	34	2	522	*571*
	Side-slope	3	30	4	0	*37*
	Total	*16*	*64*	*6*	*522*	*608*
Burned	In-stream	50	150	116	422	*738*
	Side-slope	31	175	440	0	*646*
	Total	*81*	*325*	*556*	*422*	*1384*
Unknown	In-stream	1	27	5	141	*174*
	Side-slope	0	40	0	0	*40*
	Total	*1*	*67*	*5*	*141*	*214*
	Grand total	*98*	*456*	*567*	*1085*	*2206*

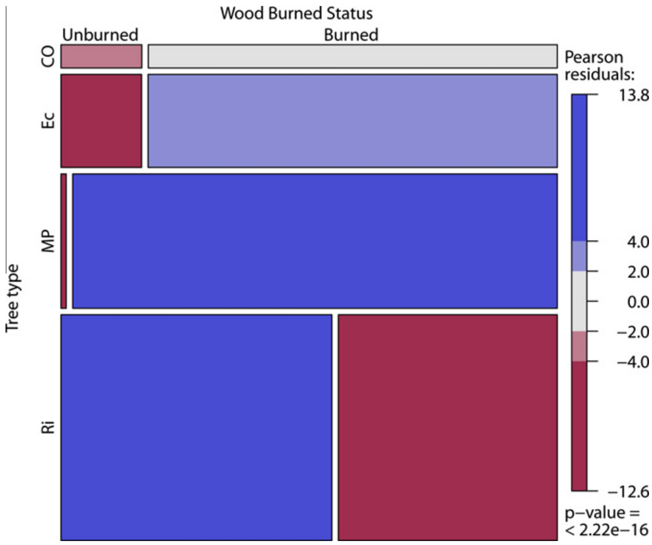

Fig. 2. Mosaic plot associating *tree types* and *burned status* of the different wood pieces. Rectangles are proportional to observed frequencies and color reflects the magnitude and significance of residuals from a contingency table test. Tree types included are Eucalyptus (Ec), Maritime pines (MP), Cork oaks (CO) and Riparian trees (Ri).

3.2. Comparing key characteristics of DWP according to tree types and DWP burned status

3.2.1. Diameters and lengths

Diameters were significantly different ($F = 29.1$, $P < 0.001$) between tree types (means: CO = 0.10 m; Ri = 0.09 m; MP = 0.08 m; Ec = 0.07 m), with burned pieces being thickest ($F = 8.9$, $P = 0.003$). There was no evidence of an interaction between tree type and burned status ($P > 0.05$), suggesting that the effect of fire was consistent across tree types (Fig. 3a). Lengths were also different ($F = 47.2$, $P < 0.001$) between tree types (means: MP = 5.2 m; Ec = 4.3 m; Ri = 3.2 m; CO = 1.2 m). Burned pieces were significantly longer ($F = 10.9$, $P = 0.001$), except for DWP from riparian trees (Fig. 3b).

3.2.2. Branches, rootwads, form and decay

The contingency table tests rejected the null hypothesis of no association between DWP of different tree types separated by burned status and branches ($\chi^2 = 670.7$, $P < 0.001$), rootwads ($\chi^2 = 191.7$, $P < 0.001$), form ($\chi^2 = 404.9$, $P < 0.001$) or decay ($\chi^2 = 186.1$, $P < 0.001$) in all cases (Fig. 3c–f). Branches were present more often than expected (i.e., Pearson residuals were positive) for MP but not for the other tree types. Only unburned Ri showed no difference between proportions of DWP with and without branches, although burned Ri pieces had branches less often than expected.

Regarding the presence of rootwads, the standardized residuals revealed that the percentage of burned MP pieces with rootwads was clearly higher than expected (Fig. 3d). No CO pieces had rootwads. In addition, the frequency of bent DWP was higher than expected in Ri and burned CO (Fig. 3e). For the decay status of DWP (Fig. 3f) it is worth noting that, in general, burned wood was most often decayed. Sound burned DWP of Ec, Ri and CO were rare, whereas unburned sound DWP of Ec and Ri, and burned sound MP pieces, all appear more often than expected (positive Pearson residuals).

3.3. Comparing key characteristics of DWP in all streams according to DWP burned status

3.3.1. Diameters and lengths

Regarding wood in all streams combined, burned DWP was significantly thicker (Fig 4a; $t = -4.9$, $R = 5000$, $P < 0.001$) but not significantly longer ($t = -0.5325$, $R = 5000$, $P = 0.5945$) than unburned wood (Fig. 4b).

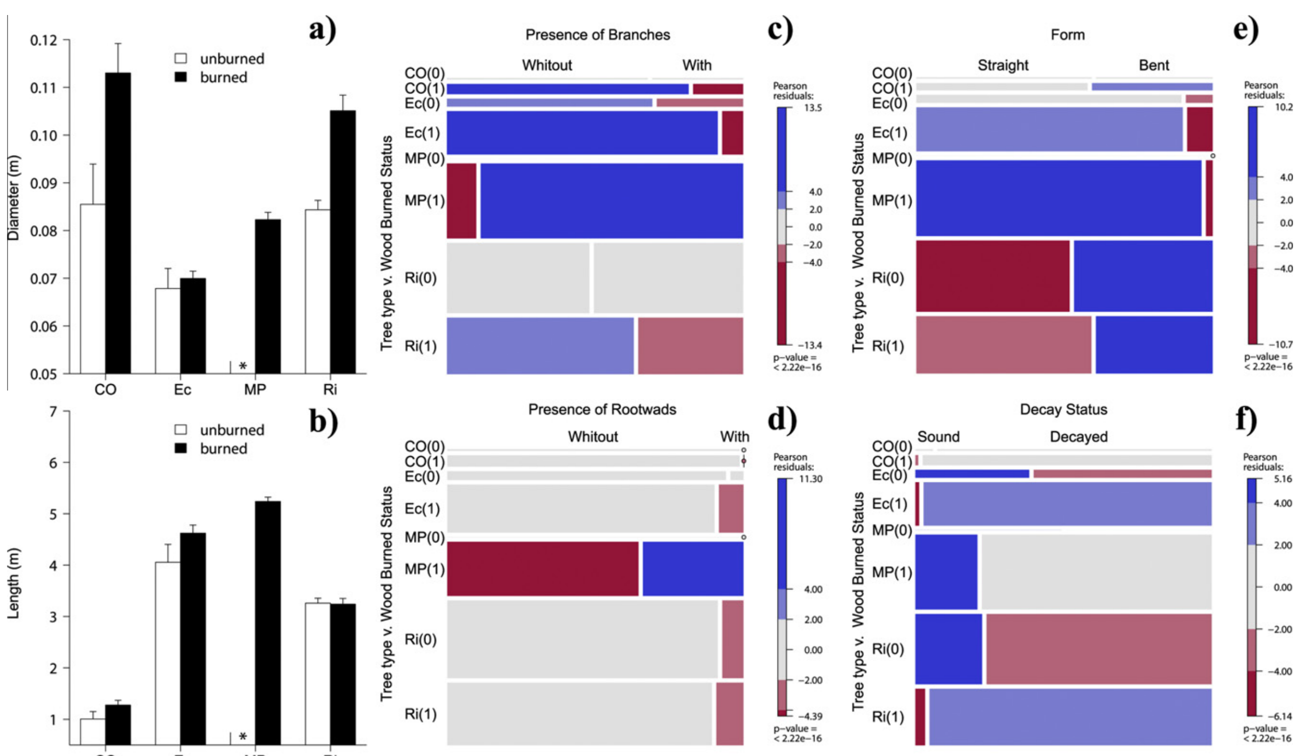

Fig. 3. (*a*) and (*b*): Bar plots of mean diameters and lengths (error bars are 95% confidence intervals) of unburned and burned wood pieces for the four tree types. *bars of unburned MP were omitted due to a very low (n = 6) sample size. (*c*) to (*f*): Mosaic plots associating *tree types* and *burned status* (0 = unburned; 1 = burned) of the different wood pieces with the *presence of branches*, *presence of rootwads*, *wood form* and *decay status*. Rectangles are proportional to observed frequencies and color reflects the magnitude and significance of residuals from contingency table tests. Tree types included are Eucalyptus (Ec), Maritime pines (MP), Cork oaks (CO) and Riparian trees (Ri).

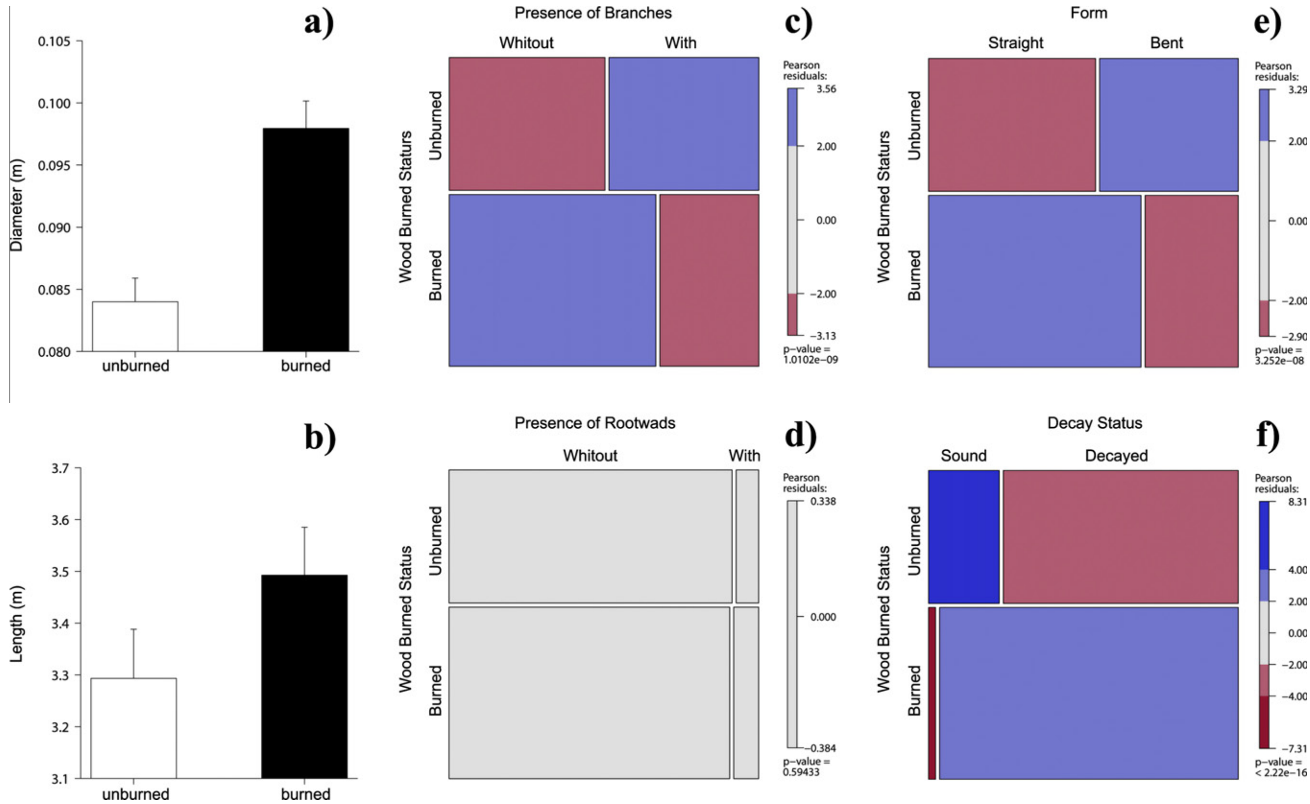

Fig. 4. (*a*) and (*b*): Bar plots of mean diameters and lengths (error bars equal 95% confidence intervals) of unburned and burned wood pieces in the study streams. (*c*) to (*f*): Mosaic plots associating *burned status* of the wood pieces in stream reaches with the *presence of branches*, *presence of rootwads*, *wood form* and *decay status*. Rectangles are proportional to observed frequencies and color reflects the magnitude and significance of residuals from contingency table tests. Tree types included are Eucalyptus (Ec), Maritime pines (MP), Cork oaks (CO) and Riparian trees (Ri).

3.3.2. Branches, rootwads, form and decay

The presence of branches was not independent ($\chi^2 = 37.3$, $P < 0.001$) of burned status (Fig. 4c), with burned DWP being less likely to have branches. Conversely, the presence of rootwads (Fig 4d) was not associated with burned status ($\chi^2 = 0.3$, $P = 0.594$). The form of DWP was not independent ($\chi^2 = 30.6$, $P < 0.001$) from burned status; burned pieces were most often straight (Fig. 4e). Regarding decay interactions with burned status ($\chi^2 = 138.3$, $P < 0.001$), burned DWP were rarely sound and, with residuals of opposite sign in Fig. 4f, unburned sound DWP appeared more frequently than expected.

4. Discussion and conclusions

In this study, we demonstrated that the physical characteristics of post-fire wood delivered to streams were markedly different depending on the source tree type. Many of the wood pieces retain their species-specific characteristics even after burning. There were also structural differences between burned and unburned pieces of wood reaching streams, which were generally consistent across the studied tree types. When filtering the analysis for DWP already recruited to streams (mainly from riparian trees), it became clear that diameter, presence of branches, wood form and decay status were influenced by fire. Regarding the quantity of DWP in "burned streams" and eventual recruits from the side-slope, we discuss below which tree types contributed more DWP to standing stocks in streams and, proportionally, which tree types supplied more burned wood.

4.1. Differences between post-fire DWP originating from cork oak montados and timber-production forests

Overall patterns in wood size and architecture among the Mediterranean forests studied reflect intrinsic species characteristics that are inseparable from silviculture practices (Brin et al., 2008). In general, timber-production forests (MP and Ec) contributed thinner, longer and straighter DWP to streams than CO, independent of burned status. In the study sites, timber-production forests of Ec and MP were young stands when they burned (which may be an increasingly common situation on production forests in Portugal), thus it is not surprising that their downed wood had thinner pieces than Ri and the older CO forests (Benda et al., 2002). This is consistent with work from the Pacific Northwest region of North America where Rot et al. (2000) reported diameters of wood in streams of old-growth forests exceeding the diameter of trees in riparian forests. CO pieces also share with riparian DWP a bent form when compared to timber-production forests. DWP of CO, much shorter and always without rootwads, should reflect their origin as slash from branch pruning, along with the fact that the trees themselves are shorter and very difficult to uproot.

Maritime pine and eucalyptus pieces also showed well-marked structural differences from each other. Our data indicate that post-fire DWP from MP have more branches and rootwads than Ec. This could be due to differences in rooting structure and branching patterns. Living Ec invest more energy in the trunk than in the branches (Pereira, 2007). The presence of rootwads in DWP of MP probably highlights the tendency for the trees to uproot easily from side-slopes in the study sites. Another pattern observed was that MP pieces were longer and thicker than Ec. Although there are differences in growth form and growth rates, this probably has more to do with post-fire management (or lack thereof) than specific characteristics of the trees. A number of burned MP wood pieces were whole trees, while most Ec pieces were slash left behind after post-fire clear-cutting along with fallen wood from young recovered trees.

4.2. Relations between DWP quantities and tree type post-fire recovery

Although a complete analysis of wood standing stocks is beyond the scope of this study, differences in wood frequency among study forests were conspicuous (Table 2). First, despite larger and more connected riparian buffers, and therefore a low probability of upland wood entering the streams, timber-production stands contributed more wood to streams following fire than CO stands with more limited buffers. Discounting pieces originated from riparian trees, the average number of recruited pieces of MP (2.3 pieces/100 m) and Ec (4.5 pieces/100 m) were ~2 and ~3 times higher than the frequency of CO wood (1.4 pieces/100 m) in CO sub-basins (Table 3). This likely reflects a combination of the low density of CO trees in *montado* systems, their higher fire resistance (e.g., cork insulation), post-fire resilience and salvage logging in post-fire management. Second, another contrast between tree types is linked with proportions of unburned pieces (Fig. 2). Only DWP of Ri trees had more unburned than burned pieces, which can be interpreted as a result of burned wood transport downstream since fire occurrence, while at the same time there was a sustained input of unburned wood from the riparian zone. Arseneault et al. (2007) conceptually stressed these asynchronies when "source and sink systems" recover at different rates.

Another explanation for the prevalence of unburned Ri pieces may be that riparian vegetation presents a lower fire proneness (Fernandes et al., 2010; Moreira et al., 2001) or that riparian zones experienced less severe wildfires due to their lower elevation and more hydric environment (Bendix and Cowell 2010b; Everett et al., 2003). Unfortunately, few data are available regarding fire and riparian vegetation (Pettit and Naiman, 2007); some authors refer to riparian vegetation as being subjected to frequent fires (Kobziar and McBride, 2006) while, as stated by Dwire and Kauffman (2003), others ignore fire effects in riparian zones because of the belief that riparian areas are too wet to burn. In our data the majority of burned wood inside streams came from riparian trees but DWP from this tree type had the highest proportion of unburned pieces. Riparian areas are clearly not immune to the presence or impacts of fire but continued recruitment is a confounding factor to some degree. Furthermore, the wood from side-slope forests that did enter the stream following the fire is within the riparian zone of influence if not the riparian buffer. This suggests that species-specific differences in fire susceptibility are more important to riparian zone impacts than location in the landscape influence. More work is clearly needed to address the question of fire impacts in riparian zones.

Finally, regarding contrasts between MP and Ec, the higher proportion of unburned DWP that can be found in burned Ec stands (equal to CO, with 20%, against 1% in MP) is likely due to the rapid coppice regrowth in an Ec forest after fire – with subsequent wood recruitment – as opposed to the slower natural stand-replacement in MP stands (Calvo et al., 2008). The growth of Ec is striking and we observed recruitment of new wood from coppiced growth within the two years over which these data were collected.

4.3. What can streams expect from fires in the long-term?

This study showed that fire contributed toward larger DWP in streams. This included a number of potential "key" pieces (Abbe and Montgomery, 1996, 2003) with the capacity to entrap other wood and form logjams (Bocchiola et al., 2008; Nakamura and Swanson, 1993). Smaller pieces may have been consumed by fires or were the first to be transported downstream. Since decay rate (Hassan et al., 2005) and probability of displacement (Merten et al., 2010, 2011; Warren and Kraft 2008) are functions of size, large pieces have a more sustained long-term influence on habitat and physical processes than small pieces (Dolloff and Warren, 2003).

Burned pieces within the stream channel also tended to lack branches (Fig. 4c), which when present increase surface area and can promote habitat complexity that improves conditions for aquatic organisms (Sundbaum and Naslund, 1998). It is known that tree boles tend to survive fire while most branches are consumed in the blaze (Agee, 1993). We therefore suggest for future studies in fire-prone areas that each DWP be assigned to a class (trunk/branch) to address this point. A related system by Newbrey et al. (2005) implemented a branching complexity for each piece, where higher complexity corresponded to a greater number of branches and twigs. In this context our data suggest a fire-driven reduction in stream wood complexity in burned streams. In addition, fire seems to have promoted the presence of straight wood in the study reaches. Straight wood pieces are less likely to become trapped or snagged in river channels than more irregular wood pieces of a similar size (Gurnell, 2003).

Fire also appears to increase the number of decayed DWP pieces in streams (Fig. 4f). Decayed pieces may be more prone to breakage (Hassan et al., 2005) and shorter pieces are depleted more readily via downstream transport (Merten et al., 2010, 2011). Zelt and Wohl (2004), compared characteristics of wood in two adjacent burned and unburned streams a decade after wildfire and, contrary to our results, reported smaller average piece sizes in the burned stream. The authors explained this difference by age distinctions among source trees on both streams. In our study, although fire-

Table 3
Frequency, volume and burned pieces proportion of in-stream downed wood (mean ± SE) across 27 first- to third-order streams in central Portugal following wildfires. Stocks are reported by upland forest: Maritime pine (MP), Eucalyptus (Ec) or Cork oak trees (CO). For each forest type, 9 stream reaches of ~500 m each were surveyed. The contribution of the upland forest for the respective total stock of in-stream wood (pieces intercepting the bankfull) is also given. Results are presented by our size criteria ($\geqslant$ 0.05 m; length $\geqslant$ 0.5 m) and the standard definition for large wood ($\geqslant$ 0.1 m; length $\geqslant$ 1 m).

Side-slope forest	In-stream wood species	$\geqslant$ 0.05 m; length $\geqslant$ 0.5 m			$\geqslant$ 0.1 m; length $\geqslant$ 1 m		
		Proportion of burned pieces	Frequency (# 100 m^{-1})	In-stream volume (m^3/100 m)	Proportion of burned pieces	Frequency (# 100 m^{-1})	In-stream volume (m^3/100 m)
MP	MP	1.0 ± <0.1	2.3 ± 1.0	0.12 ± 0.06	1.0 ± <0.1	1.1 ± 0.5	0.11 ± 0.05
	All species	0.7 ± 0.1	12.6 ± 2.1	0.65 ± 0.18	0.8 ± 0.1	5.0 ± 1.1	0.56 ± 0.17
Ec	Ec	0.7 ± 0.1	4.5 ± 2.3	0.11 ± 0.05	0.7 ± 0.2	0.3 ± 0.2	0.05 ± 0.02
	All species	0.7 ± 0.1	10.8 ± 3.5	0.38 ± 0.17	0.7 ± 0.1	2.9 ± 1.4	0.28 ± 0.15
CO	CO	0.8 ± 0.1	1.4 ± 0.6	0.02 ± 0.01	0.9 ± 0.1	0.4 ± 0.2	0.01 ± 0.01
	All species	0.6 ± 0.1	9.6 ± 2.9	0.22 ± 0.08	0.7 ± 0.1	2.0 ± 0.6	0.15 ± 0.05
Total across 27 streams		*0.7 ± 0.1*	*11.0 ± 1.6*	*0.42 ± 0.09*	*0.7 ± 0.1*	*3.3 ± 0.7*	*0.33 ± 0.08*

impacted wood was larger, data also indicated that burned wood was often more decayed than unburned wood. This sets up an interesting contrast. Does the larger size of wood reduce decay to a greater degree than the impacts of burning may promote decay? If burning wood substantially increases the susceptibility of stream wood to decay, the input of wood following fires may not persist. Our study could have been strengthened if DWP from unburned streams were included for comparison to those of the burned sections, especially if we had been able to find unburned forest stands of the same age as burned stands when the fire happened. This design would prevent stand age from becoming a confounding factor when comparing attributes such as DWP dimensions (Zelt and Wohl, 2004), particularly for fast-growing species such as Ec.

It is possible to predict the implications of future DWP inputs from upland Ec, MP and CO trees following fire. More MP pieces should change stream hydraulics and habitat complexity, due to the frequency of branches and rootwads. By comparison, Ec pieces are often straight pieces without rootwads, which may be transported more readily downstream (Braudrick and Grant, 2001) or stay in positions (e.g., bridges) with little contribution to stream morphology and function (Jones and Daniels, 2008). Nevertheless, the individual ease of transport of Ec pieces can be hindered by the higher quantities of DWP coming from upland Ec relative to MP. In agro-systems, Elosegi and Johnson (2003) suggested that thinner, longer wood pieces were more common than in other streams. In this sense, post-fire pieces from CO agro-systems delivered to their streams can predictably contrast with other pre-existent wood.

4.4. Management concerns

Less than a decade after large scale wildfires, structural characteristics of downed wood in and near streams are clearly fire-driven and directly influenced by silviculture practices. Changes in forest management will not only affect standing stocks of instream wood in the short term following a fire, but will change the susceptibility of wood pieces to downstream transport and decay, thus changing stocks in the long term as well. Changes in wood stocks can have dramatic effects on stream ecosystems (Gregory et al., 2003) and warrants consideration when decisions are being made regarding forest management. Reduced stocks, for example, can lead to decreased in habitat for fish, substrate for invertebrates and biofilms, leaf litter retention, transient storage, and hyporheic exchange (Gregory et al., 2003). Maintaining riparian buffers – probably less susceptible to fire than surrounding forest stands – may reduce, but will not eliminate, the impacts of fire on stream wood.

In a previous study (Silva et al., 2011), we concluded that wildfires strongly influenced the landscape dynamics of three fire-prone areas across Portugal over ~14 years. Fire-driven transitions revealed that land abandonment led to increases in shrublands (encroaching into previously forested areas) and more mixed forests of MP with Ec over time. Under this scenario, our results suggest that large pieces of wood will slightly become less common in these streams and that stream reaches will become more homogeneous in terms of wood characteristics. At the global scale, the combination of increasing wildfire disturbance along with changing forest composition (from older, more traditional mixed uses to single-use, single-species forests managed for timber) can alter the characteristics of wood entering streams with a shift from large DWP to thinner and straighter wood. We reiterate that DWP, even after burning, still retain species-specific physical architecture of the pre-burned trees and could not be lumped together in terms of their effect on stream ecosystem structure and function or in-stream wood movement. The interaction between the unique physical traits of a species and production silviculture practices are responsible for long-term implications for stream function and structure after wildfires.

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