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## Abstract

The effect of two post-fire stabilization techniques (Seeding and Mulching) on trace elements (Al, B, Co, Cu, Fe, Mn, Mo and Zn) losses with eroded sediments was evaluated during 13 months after an experimental fire in a steep shrubland of a temperate-humid region (NW Spain). With time, concentration of extractable Mn, Zn and Cu in sediments decreased, Fe tended to increase and Al, Co, B and Mo varied without a clear trend. Most sediments and trace element losses occurred during the first three months post-fire. Compared with the available elements in ash+burned topsoil, the fraction lost with sediments was highest for Mo (10-16%), intermediate for Mn (4%) and Zn (3%) and low for the rest (0.4-1.2%). Although minor effects of stabilization techniques on element concentrations were found, accumulated mass losses of trace elements decreased 6-12 times in Mulching due to its 10-fold lower soil erosion rate; no significant changes were found in Seeding. Sediment nutrient losses are probably more important than those published for smoke, leaching or volatilization. Our results suggest that the Zn and Cu enrichment in sediments from the first erosion events increase the risk of downslope water and soil contamination. In conclusion, soil stabilization techniques are useful to prevent post-fire ecosystem damage.

**Keywords:** Burned Area Emergency Response, experimental fire, sediment, shrubland.

## Introduction

Increasing frequency and extent of anthropogenic wildfires are a global concern due to their ecological effects. Fire impacts on soils modifying their physical, chemical and biochemical properties. Soils characteristics are variably affected by fire depending on the burn severity. Burning can reduce organic matter, alter soil structure and porosity, trigger considerable losses of nutrients (through volatilisation, ash entrapment in smoke columns, leaching and erosion), and modify soil biological communities (Certini 2005). Most studies on the effects of fire on soil chemical quality have been focussed on evaluating changes in organic matter (C, N) and available macro-nutrients (P, Ca, Mg, Na and K) (Khanna and Raison 1986; Carballas 1997; Couto-Vázquez and González-Prieto 2006; Úbeda et al. 2006).

Micro-nutrients, generally present in small quantities but essential for plant development, are of interest in the recovery of burned soils and

vegetation. González-Parra et al. (1996) found that both the total content and the easily reducible forms of Mn increased significantly after fire, due to Mn supplied by the ash, while the exchangeable Mn did not show any variation. Presumably, Fe, Cu and Zn could behave similarly to Mn, although specific works were lacking until recently (Certini 2005). Knowledge of fire effects on micro-nutrients has increased substantially in recent years.

An increase of extractable Mn and Zn after prescribed fires was found by several authors (García-Marco and González-Prieto 2008; Pivello et al. 2010; Close et al. 2011), although Pereira et al. (2011) reported a significant decrease in water-soluble Mn and Zn in ashes compared with the original *Quercus robur* and *Quercus suber* litter. For extractable Fe, the published results varied from decrease (García-Marco and González-Prieto 2008; Pereira et al. 2011) to no clear tendency (Close et al. 2011) and even increase (Pivello et al. 2010). In the case of Cu, while García-Marco and González-Prieto (2008) did not find a significant change, Pivello et al. (2010)

reported higher levels in burned soils, and for B it has been reported that its concentrations were higher in soil from burned forest than from unburned forests (Ponder et al. 2009). Despite these recent advances, Smith et al. (2011) and Stankov Jovanovic et al. (2011) claimed that, until now, influence of fire on micro-nutrients behaviour and export in sediments is not completely solved, primarily due to the lack of specific studies.

Erosion is widely recognized as a prevalent post-fire process (DeBano et al. 1998; Robichaud and Brown 1999; Martin and Moody 2001; Cerdá and Lasanta 2005), but whether it affects or not the long-term soil quality and site productivity is still under controversy (Helvey et al. 1985; Emmerich 1999; Thomas et al. 1999; Shakesby et al. 2002; Robichaud et al. 2006) and little is known concerning nutrient losses associated with the eroded material (De Koff 2006; Gómez-Rey et al. 2013), especially in the case of micro-nutrients.

To reduce the increased erosion rates, post-fire stabilization techniques (e.g. seeding and mulching) of soils must be applied as soon as possible after the wildfire to accelerate development of ground and vegetation cover and to reduce surface runoff and prevent soil erosion (Pinaya et al. 2000; Robichaud et al. 2000; Vega et al. 2005; Wagenbrenner et al. 2006; Groen and Woods 2008; Robichaud 2009; Fernández et al. 2011, 2012). Despite its undoubted interest, few studies deal with the effects of fire on soil nutrient losses by erosion (DeBano et al. 1998; Gabet et al. 2005; Robichaud et al. 2006; Spigel and Robichaud 2007) and their reduction by post-fire stabilization techniques (Thomas et al. 2000; Robichaud et al. 2006; Gómez-Rey et al. 2013).

Post-fire stabilization techniques can potentially alter trace element concentration in the eroded sediments. Enhanced vegetation recovery by grass seeding can increase plant uptake of nutrients and thus reduce their availability in soils and sediments. If the tested techniques modify the runoff-infiltration ratio and/or reduce preferentially the erosion of some soil fractions, they can, consequently, alter the chemical composition of sediments. Therefore, we hypothesized that seeding and mulching might lead to changes on trace elements concentration of eroded sediments and that changes might depend on the effectiveness of such techniques. Accordingly, the objective of the present study was to evaluate, during 13 months, the effects of two post-fire stabilization

techniques (seeding and mulching) on the losses of eight trace elements (Al, B, Co, Cu, Fe, Mn, Mo and Zn) with the eroded sediments following an experimental fire of low intensity in a steep shrubland. The study site was in the temperate-humid region of NW Spain, one of the areas with highest fire incidence in Europe (Biro 2009).

## Material and methods

### *Site description and experimental design*

The experimental field was located in the Monte Cabalar (42° 38' 58" N; 8° 29' 31" W, 660 m a.s.l.) at A Estrada (Galicia, NW Spain). The area has a temperate and rainy climate. During the study period (October 2009 - December 2010), accumulated precipitation monitored in the area was 3036 mm. The soil, developed over granite and with a slope of 38-54%, is acidic, with sandy loam texture, relatively high organic-matter content and low C/N ratio. Vegetation cover is dominated by gorse *Ulex europaeus* L. with some *Pteridium aquilinum* (L.) Kuhn., *Ulex gallii* Planch., *Daboecia cantabrica* (Huds.) K. Koch and *Pseudoarrenhaterum longifolium* Rouy, with an average height of 123 cm and 100% of ground cover.

After selecting an area of homogeneous slope, orientation, soil type and vegetation cover, 12 experimental plots (30 x 10 m each) were established with the longest dimension parallel to the maximum slope. On June 2009, the shrub was cut and laid down directly on the ground, to favour litter and duff combustion, and three months later (October 14th and 15th) plots were burned with the backfire technique to favour soil organic cover consumption and soil heating. The rate of fire spread was slow (0.30-0.33 m min<sup>-1</sup>) and the soil temperature reached during fires, monitored with ten thermocouples (K type) per plot, was moderate at the mineral soil surface (153 °C) and low at 2 cm soil depth (34 °C). The fire reduced by 3.5 ± 0.4 kg m<sup>-2</sup> the initial fuel loading of 3.8 ± 0.2 kg m<sup>-2</sup>.

To monitor post-fire erosion, the plots were delimited by a geotextile fabric fixed to posts. Sediment fences, made from a geotextile fabric similar to that described by Robichaud and Brown (2002) and located at the downhill portion of the plots, were used for periodic collection of sediments. After the experimental fire, three treatments were arranged in a fully randomized design with four replications: a) burned plots, considered as Control

treatment; b) Mulching, burned plots with 230 g m<sup>-2</sup> of straw mulch; and c) Seeding, burned plots with a mixture of seeds at a rate of 45 g m<sup>-2</sup> (*Lolium multiflorum*, 35%; *Trifolium repens*, 25%; *Dactylis glomerata*, 20%; *Festuca arundinacea*, 10%; *Festuca rubra*, 5%; *Agrostis tenuis*, 5%). The soil stabilization treatments were applied the 6th of November 2009 manually in order to minimize soil perturbation.

#### *Sampling and chemical analysis of soil, ashes and sediments*

Soil samples were taken from the top soil layer (0-5 cm depth) immediately before (US) and after burning (BS). Immediately after the fire, but before the application of soil stabilization treatments, "ash

samples" (ashes+charred plant and litter debris) were randomly taken in ten squares (15 x 15 cm) of each plot, weighted, thoroughly homogenized and combined to give a composite ash sample per plot; then, soil samples were taken from the top soil layer (BS), weighted and combined to give a composite soil sample per plot. The same was done for unburned soil (US) after removing the plant litter. Composite soil samples were sieved (< 2 mm) and homogenized; then, sub-samples were air-dried for chemical analyses. The average concentrations of extractable elements found in these samples (ash, US and BS) are shown in Table 1.

Table 1. Concentration of extractable nutrients (mg kg<sup>-1</sup> d.w.) in unburned soil (US), burned soil (BS) and ash at the beginning of the study. Values are mean ± standard deviation (n=12).

	US	BS	Ash
Extractable Al	370 ± 65	385 ± 95	208 ± 102
Extractable Fe	747 ± 194	470 ± 148	314 ± 84
Extractable Mn	1.4 ± 0.8	1.8 ± 1.6	153 ± 45
Extractable Zn	0.92 ± 0.58	1.50 ± 0.45	17.2 ± 5.6
Extractable Cu	0.09 ± 0.17	0.03 ± 0.03	7.5 ± 3.1
Extractable Co	0.09 ± 0.02	0.07 ± 0.02	0.14 ± 0.03
Extractable B	0.11 ± 0.05	0.11 ± 0.03	2.48 ± 0.74
Extractable Mo	0.00 ± 0.00	0.00 ± 0.01	0.04 ± 0.04

After each sediment-producing rain event, eroded sediments were collected and carried to laboratory, where they were dried, thoroughly homogenized and weighed. A sub-sample was taken for chemical analysis.

For extractable trace element analyses, soil, ash or sediments (10 g; two replicates per sample) were shaken for 2 h with a mixture of 1 M NH<sub>4</sub>Ac and 0.005 M DTPA (1:5 soil, ashes or sediments: solution ratio); the extracts were filtered through cellulose filter paper and then analysed for trace elements (Al, B, Co, Cu, Fe, Mn, Mo and Zn) by simultaneous ICP-OES (Varian Vista Pro, Mulgrave, Australia). Available B and Mo levels were below analytical equipment detection limits in more than 60% of soil samples analysed.

#### *Statistical analysis*

Data on concentrations and mass of trace elements in the sediments from each erosion event were statistically analysed by repeated measures ANOVA (RM-ANOVA), with time as within-subjects factor and treatment as between-subjects factor, using Greenhouse-Geisser corrections if assumption of sphericity was not met according to Mauchly's sphericity test. The effects of soil stabilization treatments on the accumulated mass of eroded sediments and trace elements were statistically analysed by one-way ANOVA. The Levene's test was used for verifying the equality of variances among groups. In the case of homocedasticity, significant differences among the mean groups were established at p < 0.05 using the Bonferroni's test for multiple comparisons (RM-ANOVA) or the Tukey's test (one-way ANOVA). In the case of unequal variances,

the original data were subjected to the Tukey's ladder of power or to Box-Cox transformations to obtain equality of variances and then significant differences among the mean groups were established as previously explained

Simple regression models between soil properties as dependent variables and the inverse of the accumulated precipitation as independent variable were explored after checking the fulfilment of the assumptions of linear regressions (independence, homocedasticity and normality), and the best models were selected maximizing the adjusted  $r^2$  and minimizing the standard error of the estimated residues. Statistical procedures were performed with SPSS 15.0 for Windows.

## Results

### *Extractable nutrient concentration in sediments*

The extractable Al concentration of the sediments varied without a clear trend during the study period (Fig. 1a), with values close to that of ashes in the first sediments collected and similar to or higher than those of burned and unburned soil in the other samplings (Table 1); in most cases, Al levels in Control and Seeding sediments were even higher than that of burned topsoil and 2-fold that in ashes. No significant treatment or treatment  $\times$  date interaction was shown by the RM-ANOVA.

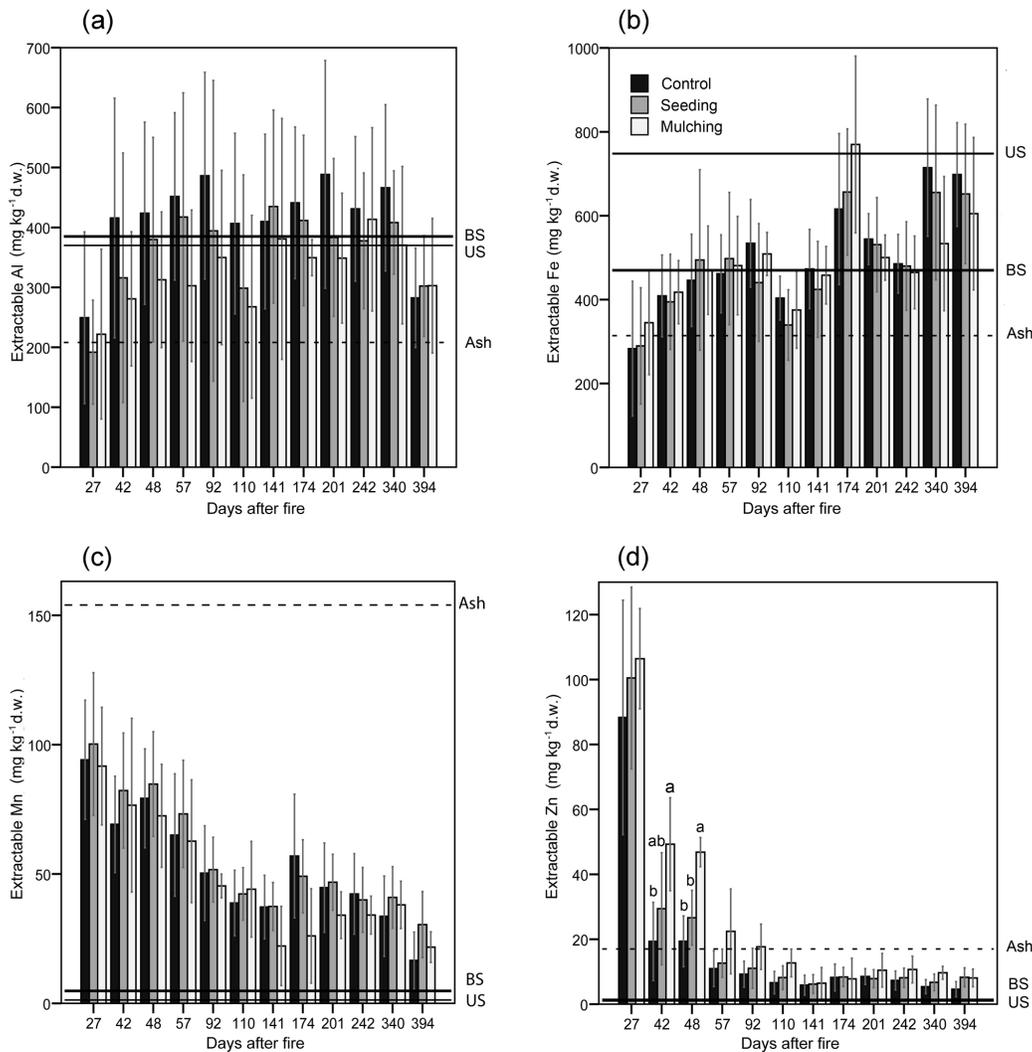


Fig. 1. Evolution of extractable Al (a), Fe (b), Mn (c) and Zn concentration (d) of the sediments eroded along the first 13 months after the experimental fire in the Control, Seeding and Mulching treatments. Horizontal lines show the mean values in ash, unburned soil (US) and burned soil (BS) at the beginning of the study. Error bars refer to SD and for a given sampling date, different letters (a, b, c,...) indicate statistically significant differences ( $p < 0.05$ ) among treatments.

Along the study period, the concentration of extractable Fe in sediments (Fig. 1b) tended to increase but the fitting to either linear or curvilinear regression models was middling (< 30% of variance explained, data not shown). Although the sampling date explained 65% of the variance (Table 2), neither treatment nor interaction effect on sediment-Fe concentration was found with the RM-ANOVA. Comparing the sediments collected in the first 5 and the last 8 months, the Fe concentration in the former tended to be lower (most values between those of ash and burned soil), while in the latter Fe concentration was almost always between those of unburned and burned topsoil, largely doubling that in ash (Table 1).

The initial extractable Mn concentration in

sediments was two thirds that in ashes but 20 and 70 times higher, respectively, than those in burned and unburned topsoils and, despite a steady decline with time, at the end of the study it was still 4-6 fold higher than in the surface layer of burned soil (Table 1, Fig. 1c). Extractable Mn concentration was neither affected by the emergency stabilization treatment considered nor by the treatment x date interaction, but it was strongly influenced by the sampling date (78% of variance explained; Table 2). In all cases with the accumulated precipitation as independent variable, linear and curvilinear (inverse and logarithmic) regression models, explaining 50-60% of the variance, were found for extractable Mn in the sediments (Table 3).

Table 2. Results of the repeated measures ANOVA for trace element contents in sediments ( $\text{mg kg}^{-1}$ ) with treatment (T) as between-subjects and date (D) as within-subjects factor. Note: \*  $p < 0.05$ ; \*\*  $p < 0.01$ ; \*\*\*  $p < 0.001$ ; n.s. no significant.

	Treatment		Date		Interaction (TxD)	
	partial $\eta^2$	p	partial $\eta^2$	p	partial $\eta^2$	p
Extractable Al	0.096	n.s.	0.445	***	0.150	n.s.
Extractable Fe	0.010	n.s.	0.645	***	0.165	n.s.
Extractable Mn	0.143	n.s.	0.782	***	0.124	n.s.
Extractable Zn	0.162	n.s.	0.943	***	0.357	*
Extractable Cu	0.364	n.s.	0.634	***	0.204	n.s.
Extractable Co	0.011	n.s.	0.873	***	0.190	n.s.
Extractable B	0.293	n.s.	0.975	***	0.223	n.s.
Extractable Mo	0.353	n.s.	0.808	***	0.225	n.s.

Table 3. Significant regression models ( $p < 0.0005$ ) between trace element contents in sediment ( $\text{mg kg}^{-1}$ ) and the inverse of the accumulated precipitation (in mm) as the independent variable.

	Regression model	$R^2$	Equation
Extractable Mn	Linear	0.549	$y = 88.61 - (0.024 \cdot x)$
	Inverse	0.494	$y = 33.16 + (17039 \cdot x^{-1})$
	Logarithmic	0.598	$y = 251.8 - (28.06 \cdot \log x)$
Extractable Zn	Inverse	0.836	$y = -4.958 + (22638 \cdot x^{-1})$
Extractable Cu	Inverse	0.461	$y = -3.239 + (4855 \cdot x^{-1})$

Very high extractable Zn concentrations were found in the first sediments collected: 5-6 fold that in ash and 60-100 fold those in burned and unburned topsoils (Table 1, Fig. 1d). Zinc concentrations decreased steadily, their values being above those of ash during 2-3 months. The RM-ANOVA showed a small, but significant, effect of treatment x date interaction and a strong effect of sampling date on sediment-Zn concentration (36% and 94% of variance explained, respectively; Table 2), with highest values in Mulching and lowest in Control plots for the sediments collected 42 ( $p < 0.05$ ), 48 ( $p < 0.01$ ) and 340 ( $p = 0.07$ ) days after fire (Fig. 1d). The evolution of extractable Zn concentration in the sediments fits very well to a curvilinear inverse regression model with the accumulated precipitation as independent variable that explained 84% of the variance (Table 3).

In the first sediments collected, extractable Cu concentration was from slightly higher (Mulching) to more than 3-fold higher (Control, Seeding) than in ashes and 100-270 times higher than in the topsoils (Table 1, Fig. 2a). These very high initial concentrations fall suddenly two orders of magnitude in the second sampling of sediments, varying then without a clear trend around values 2-6 fold higher than in topsoils. Like for extractable Mn, extractable Cu was influenced by the sampling date (63% of variance explained; Table 2), but not by treatments or treatment x date interaction. The evolution of extractable Cu concentration in the sediments fits to a curvilinear inverse regression model with the accumulated precipitation as independent variable that explained 46% of the variance (Table 3).

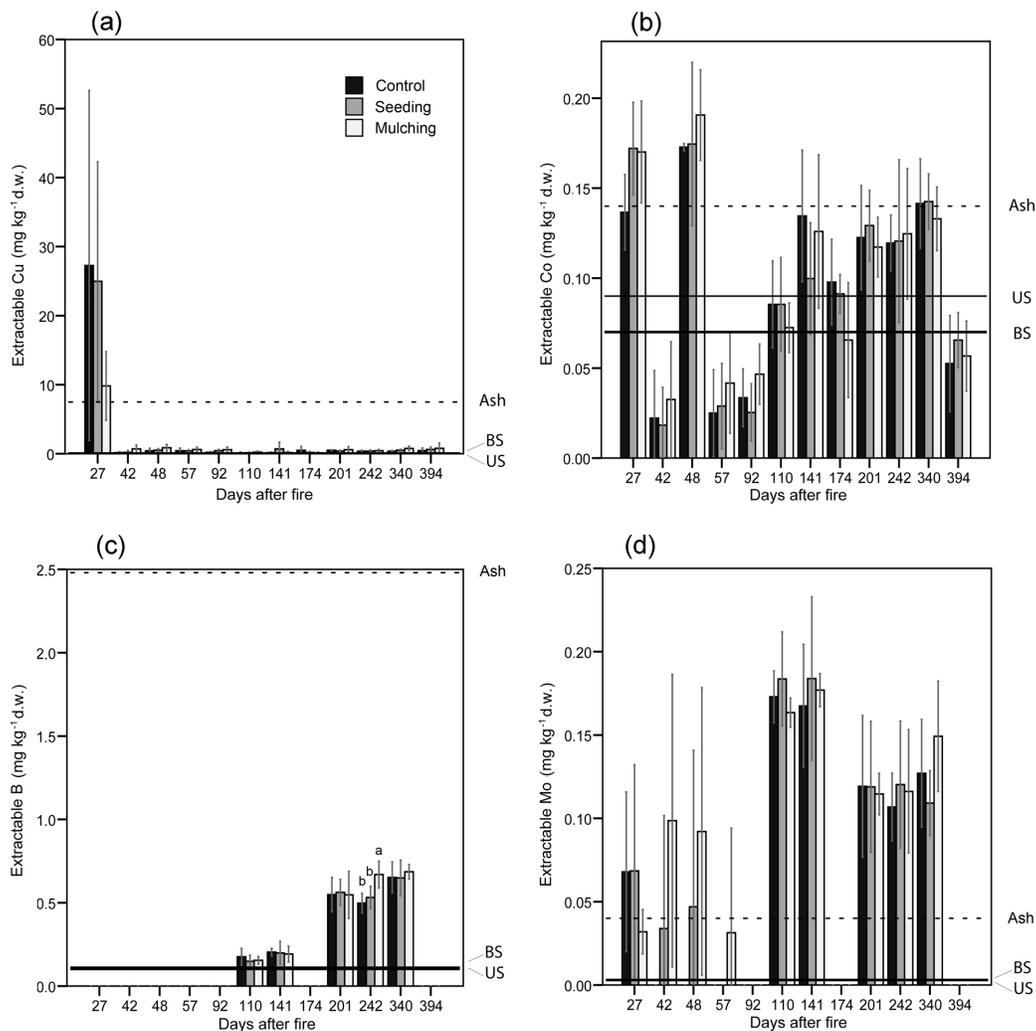


Fig. 2. Evolution extractable Cu (a), Co (b), B (c) and Mo concentration (d) of the sediments eroded along the first 13 months after the experimental fire in the Control, Seeding and Mulching treatments. Horizontal lines show the mean values in ash, unburned soil (US) and burned soil (BS) at the beginning of the study. Error bars refer to SD and for a given sampling date, different letters (a, b, c,...) indicate statistically significant differences ( $p < 0.05$ ) among treatments.

Extractable Co concentration in sediments varied, without any trend during the study, from values above to below the range found for ashes and topsoils (Table 1, Fig. 2b). As for extractable Mn and Cu, the extractable Co was influenced by the sampling date (87% of variance explained; Table 2), but not by treatment or treatment x date interaction.

In seven of the sampling dates, the levels of extractable B in the sediments were below the detection limits and in the other five samplings their concentrations were much closer to those in topsoils than to those in ashes (Table 1, Fig. 2c). As a result, the RM-ANOVA showed that most of variance (98%) was explained by the date of sediments collection (Table 2).

Data on Mo concentration in sediments showed continuous oscillations during the study, ranging from values below the detection limit to other 4- and 40-fold those of ashes and topsoils, respectively (Table 1, Fig. 2d), date of sediments collection explaining 81% of the variance according to the RM-ANOVA.

#### *Mass of nutrients lost with sediments*

The RM-ANOVA showed that the mass of eroded sediments was strongly influenced by the treatment and sampling date (91% and 95% of variance explained, respectively), but their interaction explained 83% of the variance, showing a stronger effect for sediments in Control and Seeding than for those in Mulching, with more stable values during the study period. The accumulated mass of sediments at the end of the study decreased ( $p < 0.05$ ) in the

order: Control (3897 kg ha<sup>-1</sup>)  $\approx$  Seeding (3117 kg ha<sup>-1</sup>) > Mulching (381 kg ha<sup>-1</sup>). Very similar results were found for the extractable trace element losses with the sediments and a strong influence of treatment (76-97% of the variance explained,  $p < 0.001$ ), date (81-93% of the variance explained,  $p < 0.001$ ) and their interaction (64-73% of the variance explained,  $p < 0.001$ ) was observed. However, the effect was somewhat less important for Cu (71%, 58% and 34% of the variance explained for treatment, date and treatment x date interaction, respectively), and the interaction between factors was weaker for Co (53% of the variance explained,  $p < 0.001$ ).

Relative to Control, the accumulated mass of trace elements lost with sediments at the end of the study period were 59-98% in Seeding but only 7-18% in Mulching (Table 4); differences of Mulching with Control and Seeding were statistically significant ( $p < 0.05$  to  $p < 0.01$ ).

The trace elements lost by erosion accounted for 1.4-67.6% and 1.0-44.3% of the amounts supplied by ash in Control and Seeding treatments, respectively, figures significantly higher than the 0.25-7.5% found in Mulching (Table 4). In all treatments, the percentages were higher for Mo, Al, Fe and Co, intermediate for Mn and Zn, and lower for Cu and B. Taking as reference the extractable nutrients in ash+topsoil, losses by erosion were low in Mulching for all elements (0.08 to 1.7%); the same was true in Seeding and Control for Al, Fe, Cu, Co and B (0.37 to 1.2%), but not for Zn and Mn (3-4%) and, especially, Mo (10-16%).

Table 4. Accumulated mass of trace elements (g ha<sup>-1</sup>) lost by erosion in the plots under each treatment (C, Control; S, Seeding; M, Mulching), and percentage of elements in ash and ash+topsoil they accounted for. In each row, different letters showed statistically significant differences between treatments for each variable: a) mass of trace elements; b) % of ash trace elements lost by erosion; and c) % of ash+topsoil trace elements lost by erosion.

	Mass of trace elements			% of ash trace elements lost by erosion			% of ash+topsoil trace elements lost by erosion		
	C	S	M	C	S	M	C	S	M
Extractable Al	1783 <sup>a</sup>	1094 <sup>a</sup>	152 <sup>b</sup>	66.34 <sup>a</sup>	40.71 <sup>a</sup>	5.66 <sup>b</sup>	0.91 <sup>a</sup>	0.56 <sup>a</sup>	0.08 <sup>b</sup>
Extractable Fe	1900 <sup>a</sup>	1232 <sup>a</sup>	208 <sup>b</sup>	46.81 <sup>a</sup>	30.35 <sup>a</sup>	5.12 <sup>b</sup>	0.80 <sup>a</sup>	0.52 <sup>a</sup>	0.09 <sup>b</sup>
Extractable Mn	182 <sup>a</sup>	171 <sup>a</sup>	12.7 <sup>b</sup>	9.74 <sup>a</sup>	9.15 <sup>a</sup>	0.68 <sup>b</sup>	4.26 <sup>a</sup>	4.00 <sup>a</sup>	0.30 <sup>b</sup>
Extractable Zn	30.2 <sup>a</sup>	28.2 <sup>a</sup>	5.34 <sup>b</sup>	13.62 <sup>a</sup>	12.71 <sup>a</sup>	2.41 <sup>b</sup>	3.10 <sup>a</sup>	2.90 <sup>a</sup>	0.55 <sup>b</sup>
Extractable Cu	1.35 <sup>a</sup>	1.29 <sup>a</sup>	0.25 <sup>b</sup>	1.39 <sup>a</sup>	1.33 <sup>a</sup>	0.26 <sup>b</sup>	1.23 <sup>a</sup>	1.18 <sup>a</sup>	0.23 <sup>b</sup>
Extractable Co	0.42 <sup>a</sup>	0.41 <sup>a</sup>	0.04 <sup>b</sup>	24.60 <sup>a</sup>	24.01 <sup>a</sup>	2.34 <sup>b</sup>	1.18 <sup>a</sup>	1.15 <sup>a</sup>	0.11 <sup>b</sup>
Extractable B	0.56 <sup>a</sup>	0.33 <sup>a</sup>	0.08 <sup>b</sup>	1.75 <sup>a</sup>	1.03 <sup>a</sup>	0.25 <sup>b</sup>	0.62 <sup>a</sup>	0.37 <sup>a</sup>	0.09 <sup>b</sup>
Extractable Mo	0.36 <sup>a</sup>	0.24 <sup>a</sup>	0.04 <sup>b</sup>	67.64 <sup>a</sup>	44.32 <sup>a</sup>	7.46 <sup>b</sup>	15.59 <sup>a</sup>	10.21 <sup>a</sup>	1.72 <sup>b</sup>

$p < 0.05$  for Cu, Co, B, Mo;  $p < 0.01$  for Al, Fe, Mn and Zn

Agreeing with most studies about fire effects on soil total and chemically extractable Mn (Chambers and Attiwill 1994, in Smith et al. 2011; González-Parra et al. 1996; De Marco et al. 2005; García-Marco and González-Prieto 2008; Close et al. 2011), we found higher extractable Mn concentrations in the burned soil ( $p < 0.005$ ), and especially in sediments and ashes, than in the unburned soil. Those authors pointed out that such post-fire increase was due to the contribution of ash from burned vegetation and the physiochemical breakdown of Mn complexed with organic matter. This is also likely our case because Gómez-Rey et al. (2013) found that sediments of present study were mainly constituted by charred plant and litter material. While Pivello et al. (2010) found a negative correlation in burned soils between pH and Mn, in sediments of present study  $\text{pH}_{\text{KCl}}$  and extractable Mn were positively correlated ( $r = 0.633$ ;  $p < 0.001$ ). It has been reported that one year after a prescribed fire there is no difference in extractable Mn between unburned and burned soils (García-Marco and González-Prieto 2008; Ponder et al. 2009). In the same way, we found that the extractable Mn concentration in sediments, unaffected by the emergency stabilization treatments, decreased steadily with time, indicating that Mn is easily lost; however, one year after the fire the sediments were still notably enriched in Mn compared with the burned topsoil. Consequently, at the end of the study, the amount of extractable Mn lost by erosion ( $182 \text{ g ha}^{-1}$  in Control; Table 4) was nine-fold that in suspended sediments reported by Townsend and Douglas (2004) and accounted for 9-10% of the Mn supplied by ashes and around 4% of ash+ burned topsoil Mn reserves in Control and Seeding, with much lower figures (0.7% and 0.3%) in Mulching.

Fire increases the availability of Zn in soils (García-Marco and González-Prieto 2008; Close et al. 2011; Stankov-Jovanovic et al. 2011) and the same was true in present study for ashes and sediments ( $p < 0.005$ ), the effects lasting for at least one year in the latter as also found Stankov-Jovanovic et al. (2011). The increased Zn concentration in sediments from mulching plots could be associated with changes in pH ( $r = 0.70$  to  $0.77$ ,  $p < 0.01$ , for  $\text{pH}_{\text{H}_2\text{O}}$  and  $\text{pH}_{\text{KCl}}$ , respectively) and/or to the inputs with the added straw (wheat straw contains  $15\text{-}70 \text{ mg Zn kg}^{-1} \text{ d.w.}$ ; Plank 1989; Johnson et al. 2000). Most of the variance of Zn concentration in sediments was explained by a curvilinear inverse regression model with the accumulated precipitation (Table 3). This result could be due either to a fast leaching from burned soil and

ashes if Zn is highly soluble, as found Zhan et al. (1996) in wood ash for a similar pH, or to the preferential erosion of Zn-rich particles if Zn is slowly leached from ash, as reported by Menzies and Gillman (2003) and Abdul et al. (2004). In the present study, the extractable Zn lost by erosion ( $30 \text{ g ha}^{-1}$  in Control) accounted for 13.6% and 3.1% of extractable Zn in ashes and ashes+burned topsoil, respectively; these important losses were only significantly reduced (by a factor of six) in the Mulching treatment.

Post-fire Cu dynamics was similar to that of Zn: its availability increased (García-Marco and González-Prieto 2008; Close et al. 2011; Stankov-Jovanovic et al. 2011) and sediments remained enriched in Cu compared with the burned topsoil for at least one year. However, the fast and sudden changes of Cu availability in the initial sediments collected (from higher than in ashes in the first erosion event to two orders of magnitude lower in the second erosion event) were only observed for this element and are likely related to the evolution of pH ( $r = 0.609$ ;  $p < 0.001$ ) and also to the accumulated precipitation, although Zhan et al. (1996), Menzies and Gillman (2003) and Abdul et al. (2004) reported that Cu is leached slowly from ash. The strong losses of Cu that occurred during the first erosion event could be due to a flush of organic matter, clay or silt in the runoff following the fire, as suggested by Quinton and Catt (2007) for peaks in Cu concentration observed in some erosion events from agricultural soils. Despite the low absolute amount of Cu lost by erosion ( $1.3 \text{ g ha}^{-1}$  in Control), it accounted for 1.4% and 1.2% of the extractable Cu in ashes and ashes+ burned topsoil, respectively, agreeing with the opinion that erosion of ash could cause relevant losses of Cu from the burned system (Abdul et al. 2004; Menzies and Gillman 2003).

No a clear trend could be identified for Co concentration in sediments during the study, meanwhile fire effects on Co availability (ash > unburned soil = burned soil) are compatible with those (burned soil = unburned soil) reported by García-Marco and González-Prieto (2008) considering that these authors jointly sampled ashes and burned soil. One year after forest fires, Aref et al. (2001) found no effects on Co availability in two out three soils and a significant increase in the third. Although quantitatively small ( $0.42 \text{ g ha}^{-1}$  in Control, Table 4), the amount of Co lost with the eroded sediments was qualitatively important because it accounted for one fourth of the extractable Co supplied by ashes and around 1.2% of the extractable Co in ashes+burned

topsoil. As for most studied elements, Co losses decreased by one order of magnitude in Mulching due to the reduction in the amount of eroded sediments.

Boron concentration did not differ between unburned and burned soil and was 20-fold higher in ashes, while the only published reference we found reported higher levels of extractable B in burned forest soils than in unburned ones (Ponder et al. 2009). The wide difference between extractable B concentration in ashes and sediments from the first erosion event could be due either to high losses in soluble form, as this element is quickly leached from ashes (Khanna et al. 1994) or to a preferential erosion of ash or soil particles impoverished in B. The evolution of extractable B in sediments can not be discussed because in half of sampling dates it was below the detection limits of the method employed.

The analytical constraints do not allow a sound discussion of Mo results; however, it seemed that, while experimental fire increases only slightly the extractable Mo in soils, ashes were 12-15 times enriched in Mo compared with the burned and the unburned soil. Aref et al. (2001) found an enhanced Mo availability in one of three studied soils, and no changes in the other two, one year after the wildfires. Our results also suggested that Mo losses with the eroded sediments were quantitatively small ( $= 0.36 \text{ g ha}^{-1}$ , Table 4), but they could be qualitatively important: up to 68% and 15.6% of Mo in ash and ash+burned topsoil in Control, values being around one third and 9-fold lower in Seeding and Mulching, respectively.

Moreover, soil available Mo in these plots was below the detection limit (Gomez-Rey et al. 2013), highlighting the importance of the losses of Mo-enriched sediments that might led to a significant depletion of this micronutrient essential for atmospheric  $\text{N}_2$  fixation by legumes (Rosolem and Caires 1998) which play an important role in the recovery of N cycling in burnt ecosystems (Hendricks and Boring 1999).

Sediments of present study were enriched in some trace elements (Zn, Cu, Co and Mo) compared to ashes, a feature not observed for macro-nutrients (Gómez-Rey et al. 2013). This enrichment might be related to the preferential transport of fine sized particles, such as clay or silt, and of organic matter with which these metals could be associated, as also has been suggested for Zn and Cu enrichment in sediments from agricultural soils (Quinton and Catt

2007; Fernandez-Calviño et al. 2008, 2012); moreover, this result is consistent with the enrichment in clay and silt reported for sediments from burned soils (Thomas et al. 1999). The increase in these trace elements in sediments was mainly observed during the first months after the fire (but during a year for Mo) and it could had several important environmental repercussions because it can adversely affect: a) the revegetation of burned areas, since essential nutrients for plant growth are removed from the soil; b) water quality if sediments with toxic concentrations of trace elements reach the surrounding aquatic systems; and c) soil quality where sediments are deposited if soil contamination thresholds are surpassed. With regard to the last point, during the first months after the fire, concentration of total Mn, Zn and Cu in sediments might exceed the maximum metal concentrations allowed by the EU in soils treated with sewage sludge (CEC 1986) and the reference levels for ecosystems protection established in Galicia (Macías-Vazquez and Calvo de Anta 2009), Australia (DEC 2010) and the USA (US EPA 2003-2007, see Table 5).

Nutrient losses by erosion are relevant compared with those during the fire, either by volatilization or ash entrapment in smoke columns, as well as post-fire losses by leaching. On the basis of the data supplied by Alves et al. (2010), we estimated that total particulate losses of trace elements with the smoke from the experimental fire they performed in a Portuguese Mediterranean shrubland with a fuel load of  $15.6 \text{ Mg ha}^{-1}$  amounted to 382, 50, 13 and  $0.8 \text{ g ha}^{-1}$  for Fe, Zn and Mo and Co, respectively. Such values were lower than the estimated total amounts of these trace elements lost by erosion in Control and Seeding treatment (Table 5), although it should be taken into account the higher fuel loading consumed in our plots ( $35 \text{ Mg ha}^{-1}$ ). Losses by erosion we found were also higher than those of total particulate Cu ( $\sim 15 \text{ g ha}^{-1}$ ) and Zn ( $\sim 70 \text{ g ha}^{-1}$ ) by stormwater runoff reported by Stein et al. (2012) one to two years following wildfires in California. Nutrient losses by erosion we measured also exceeded the particulated losses with runoff from the Cerro Grande Fire area (New Mexico), calculated on the basis of Gallaher and Koch (2004) data assuming that, relatively to pre-fire values, the additional mass of metals transported in runoff during the first year come from the burned area. Except that for B, the calculated amounts [Zn  $23 \text{ g ha}^{-1}$ ; Mo  $0.34 \text{ g ha}^{-1}$ ; Mn  $805 \text{ g ha}^{-1}$ ; Al  $3282 \text{ g ha}^{-1}$ ; B  $11 \text{ g ha}^{-1}$ ; Co  $2.4 \text{ g ha}^{-1}$ ; Cu  $4.8 \text{ g ha}^{-1}$  and Fe  $2070 \text{ g}$

ha<sup>-1</sup>] were below those we observed in Control and Seeding treatments (see Table 5). For most of the studied nutrients, substantial losses by volatilization are not expected because their volatilization

temperatures are higher than the flaming temperatures of woody fuels (1100 °C), except in the case of Zn (407-907 °C).

Table 5. Range of estimated concentration (mg kg<sup>-1</sup> d.w.) and accumulated mass (g ha<sup>-1</sup>) of trace elements in eroded sediments of the present study compared to reference levels (mg kg<sup>-1</sup> d.w.).

	Present study <sup>A</sup>		Reference levels			
	(mg kg <sup>-1</sup> )	(g ha <sup>-1</sup> )	CEC <sup>B</sup>	Galicia <sup>C</sup>	USA <sup>D</sup>	Australia <sup>E</sup>
Total Al	1337-15330	3040-35660				
Total Fe	1946-19263	4160-38000				
Total Mn	75-2720	254-3640		850	220-4300	500
Total Zn	0.30-1013	107-604	150-300	200	46-160	200
Total Cu	19-2787	5-27	50-140	50	28-80	100
Total Co	0.1-4.8	0.8-8.4		40	13-230	50
Total B	2.1-15.7	1.6-11.2				
Total Mo	0.2-4.7	0.8-7.2		5		40

<sup>A</sup> Values estimated considering that 5% of total soil nutrient was removed by DTPA and EDTA extractants (Chen *et al.* 1996; Fernández-Calviño *et al.* 2012).

<sup>B</sup> Limit value for concentrations of heavy metals in soils receiving sewage sludge application in the European Union (CEC 1986).

<sup>C</sup> Generic reference levels for protection of ecosystems established for soils in Galicia (Macías-Vazquez and Calvo de Anta, 2009).

<sup>D</sup> Ecological soil screening levels regional for superfund sites (US EPA, 2003-2007).

<sup>E</sup> Reference levels for ecological investigation established for soils in Australia (DEC, 2010).

Except minor effects on Zn concentration, the stabilization techniques used in this study did not change trace element concentration in sediments, but Mulching greatly reduced nutrient mass losses by decreasing the amount of eroded sediments. Therefore, it seems that the effectiveness of stabilization techniques for reducing nutrient losses rely almost completely on their efficacy in controlling erosion. Although the available information on the effectiveness of post-fire stabilization treatments for erosion reduction is scarce and contradictory, it is well known that at least 60 % of ground cover is needed to mitigate erosion rates (Robichaud *et al.*, 2000). Seeding has limited ability to reduce erosion, especially in the first post-fire year, because of the time it needs to provide a sufficient ground cover (Groen and Woods, 2008) while mulching provides immediate ground cover and protection from raindrop impact and overland flow. Under similar climatic conditions that

present study (Díaz-Raviña *et al.*, 2012), the reduction of erosion rates with straw mulching (73-94%) was twice that with ryegrass seeding (34-42%) during the first 4 months after application. However, mulch effectiveness depends on strand length, ground cover amount, thickness of application and longevity (Bautista *et al.*, 1996; Wagenbrenner *et al.*, 2006; Groen and Woods, 2008; Robichaud *et al.*, 2010; Fernández *et al.*, 2011; Robichaud *et al.*, 2013a, b).

## Conclusions

No impacts of post-fire soil stabilization techniques on trace element concentration in sediments were found, except minor effects on Zn. During the study period, the concentration of Fe in sediments tended to increase while those of Mn, Zn and Cu decreased. No clear trend was identified for Co, B and Mo.

Expressed as percentages of these trace element supplied by ashes, losses by erosion in Control and Seeding were high for Mo, Al and Fe, intermediate for Co, Zn and Mn, and low for Cu and B. Compared with the available elements in ash+burned topsoil, the fraction lost with sediments was highest for Mo, intermediate for Mn and Zn and low for the rest. Our results suggest that sediment losses are probably more important than that reported in the literature for smoke, leaching or volatilization. Unless effective post-fire soil stabilization techniques (Mulching) are applied for reducing trace element losses with sediments, fires can affect medium-term soil quality *in situ* (burned soil) and *ex situ* (sediment deposition areas).

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### References

Abdul G, Koch CB, Borggaard OK (2004) Weathering intensity controlling sustainability of ultisols under shifting cultivation in the Chittagong Hill Tracts of Bangladesh. *Soil Science* 169, 663-674.

Alves CA, Gonçalves C, Pio CA, Mirante F, Caseiro A, Tarelho L, Freitas MC, Viegas DX (2010) Smoke emissions from biomass burning in a Mediterranean shrubland. *Atmospheric Environment* 44, 3024-3033.

Aref IM, El Atta HA, Al Ghamde ARM (2011) Effect of forest fires on tree diversity and some soil properties. *International Journal of Agriculture and Biology* 13, 659-664.

Biro, Y (Ed.) (2009) 'Living with wildfires: what science can tell us.' In 'EFI Discussion Paper'. (European Forest Institute)

Carballas T (1997) Effects of fires on soil quality.

Biochemical aspects. In 'Forest fire risk and management.' (Eds P Balabanis, G Eftichidis, R Fantech.) pp. 249-261. (European Commission: Brussels, Belgium)

CEC (1986) Council Directive 12 June 1986 on the protection of the environment, and in particular the soil when sewage sludge is used in agriculture. Commission of the European Communities. Official Journal of the European Communities, Brussels. No L.181 (86/278/CEE): 6.

Cerdá A, Lasanta T (2005) Long-term erosional responses after fire in the Central Spanish Pyrenees - 1. Water and sediment yield. *CATENA* 60, 59-80.

Certini G (2005) Effects of fire on properties of forest soils: a review. *Oecologia* 143, 1-10.

Close D, Davidson N, Swanborough P, Corkrey R (2011) Does low-intensity surface fire increase water- and nutrient-availability to overstorey Eucalyptus gomphocephala? *Plant and Soil* 349, 203-214.

Couto-Vázquez A, González-Prieto SJ (2006) Short- and medium-term effects of three fire fighting chemicals on the properties of a burnt soil. *Science of the Total Environment* 371, 353-361.

De Koff JP, Graham RC, Hubbert KR, Wohlgemuth PM (2006) Prefire and postfire erosion of soil nutrients within a chaparral watershed. *Soil Science* 171, 915-928.

De Marco A, Gentile AE, Arena C, De Santo AV (2005) Organic matter, nutrient content and biological activity in burned and unburned soils of a Mediterranean maquis area of southern Italy. *International Journal of Wildland Fire* 14, 365-377.

DeBano LF, Neary D, Ffolliott PF (Eds) (1998) 'Fires effects on ecosystems'. (John Wiley and Sons, New York, New York, USA)

DEC (2010) Assessment Levels for Soil, Sediment and Water. Environmental Regulation Division, Department of Environment and Conservation, Contaminated Sites Management Series (Perth, WA).

Demeyer A, Voundi Nkana JC, Verloo MG (2001) Characteristics of wood ash and influence on soil properties and nutrient uptake: An overview. *Bioresource Technology* 77, 287-295.

Emmerich WE (1999) Nutrient dynamics of rangeland burns in Southeastern Arizona. *Journal of Range Management* 52, 606-614.

Fernández-Calviño D, Pateiro-Moure M, López-Periago E, Arias-Estévez M, Nóvoa-Muñoz JC (2008) Copper distribution and acid-base mobilization in vineyard soils and sediments from Galicia (NW Spain). *European Journal of Soil Science* 59, 315-326.

- Fernández-Calviño D, Pateiro-Moure M, Nóvoa-Muñoz JC, Garrido-Rodríguez B, Arias-Estévez M (2012) Zinc distribution and acid-base mobilisation in vineyard soils and sediments. *Science of the Total Environment* 414, 470-479.
- Fernandez C, Vega JA, Jimenez E, Fonturbel T (2011) Effectiveness of three post-fire treatments at reducing soil erosion in Galicia (NW Spain). *International Journal of Wildland Fire* 20, 104-114.
- Fernández C, Vega JA, Jiménez E, Vieira DCS, Merino A, Ferreiro A, Fonturbel T (2012) Seeding and mulching+seeding effects on post-fire runoff, soil erosion and species diversity in Galicia (NW Spain). *Land Degradation & Development* 23:150-156.
- Gabet EJ, Fierer N, Chadwick OA (2005) Prediction of sediment-bound nutrient delivery from semi-arid California watersheds. *Journal of Geophysical Research-Biogeosciences* 110,
- Gallaher BM, Koch RJ (2004) Cerro Grande fire impacts to water quality and stream flow near Los Alamos National Laboratory: Results of four years of monitoring. Los Alamos National Laboratory Report LA-14177. (Los Alamos, NM)
- García-Marco S, González-Prieto S (2008) Short- and medium-term effects of fire and fire-fighting chemicals on soil micronutrient availability. *Science of the Total Environment* 407, 297-303.
- Gómez-Rey MX, Couto-Vázquez A, García-Marco S, Vega JA, González-Prieto SJ (2013) Reduction of nutrient losses with eroded sediments by post-fire soil stabilization techniques. *International Journal of Wildland Fire* (available online: <http://dx.doi.org/10.1071/WF12079>)
- González Parra J, Cala Rivero V, Iglesias Lopez T (1996) Forms of Mn in soils affected by a forest fire. *Science of the Total Environment* 181, 231-236.
- Groen AH, Woods SW (2008) Effectiveness of aerial seeding and straw mulch for reducing post-wildfire erosion, north-western Montana, USA. *International Journal of Wildland Fire* 17, 559-571.
- Groeschl D, Johnson J, Smith D (1993) Wildfire Effects on Forest Floor and Surface Soil in a Table Mountain Pine-Pitch Pine Forest. *International Journal of Wildland Fire* 3, 149-154.
- Helvey JD, Tiedemann AR, Anderson TD (1985) Plant nutrient losses by soil-erosion and mass movement after wildfire. *Journal of Soil and Water Conservation* 40, 168-173.
- Hendricks JJ, Boring LR (1999) N-2-fixation by native herbaceous legumes in burned pine ecosystems of the southeastern United States. *Forest Ecology and Management* 113, 167-177.
- Johnson GV, Raun WR, Zhang H and Hattey JA (2000) *Oklahoma Soil Fertility Handbook*. Oklahoma State University. (Stillwater, OK).
- Khanna PK, Raison RJ (1986) Effect of fire intensity on solution chemistry of surface soil under a *Eucalyptus pauciflora* forest. *Australian Journal of Soil Research* 24, 423-434.
- Khanna PK, Raison RJ, Falkiner RA (1994) Chemical properties of ash derived from eucalyptus litter and its effects on forest soils. *Forest Ecology and Management* 66, 107-125.
- Macías-Vázquez F, Calvo de Anta R (2009) Niveles genéricos de referencia de metales pesados y otros elementos traza en suelos de Galicia. Consellería de Medio Ambiente e Desenvolvemento Sostible. Xunta de Galicia. (Santiago de Compostela, Galicia)
- Martin DA, Moody JA (2001) Comparison of soil infiltration rates in burned and unburned mountainous watersheds. *Hydrological Processes* 15, 2893-2903.
- Menzies NW, Gillman GP (2003) Plant growth limitation and nutrient loss following piled burning in slash and burn agriculture. *Nutrient Cycling in Agroecosystems* 65, 23-33.
- Pereira P, Ubeda X, Martin D, Mataix-Solera J, Guerrero C (2011) Effects of a low severity prescribed fire on water-soluble elements in ash from a cork oak (*Quercus suber*) forest located in the northeast of the Iberian Peninsula. *Environmental Research* 111, 237-247.
- Pinaya I, Soto B, Arias M, Díaz-Fierros F (2000) Revegetation of burnt areas: Relative effectiveness of native and commercial seed mixtures. *Land Degradation & Development* 11, 93-98.
- Pivello VR, Oliveras I, Miranda HS, Haridasan M, Sato MN, Meirelles ST (2010) Effect of fires on soil nutrient availability in an open savanna in Central Brazil. *Plant and Soil* 337, 111-123.
- Plank CO (1989) *Plant analysis handbook for Georgia*. University of Georgia Cooperative Extension Service. (Athens, GA).
- Ponder F, Tadros M, Loewenstein EF (2009) Microbial properties and litter and soil nutrients after two prescribed fires in developing savannas in an upland Missouri Ozark Forest. *Forest Ecology and Management* 257, 755-763.
- Quinton JN, Catt JA (2007) Enrichment of Heavy Metals in Sediment Resulting from Soil Erosion on Agricultural Fields. *Environmental Science & Technology* 41, 3495-3500.
- Robichaud PR (2009) Post-fire stabilization and rehabilitation. In 'Fire effects on soils and restoration strategies' (Ed. A Cerdá, Robichaud, P.R.) pp.

- 299-320. (Science Publishers: Enfield, NH, USA)
- Robichaud PR, Beyers JL, Neary DG (2000) Evaluating the effectiveness of postfire rehabilitation treatments. USDA Forest Service General Technical Report RM-GTR-63. (Fort Collins, CO)
- Robichaud PR, Brown RE (Eds DS Olsen, JP Potyondy (1999) 'What happened after the smoke cleared: Onsite erosion rates after a wildfire in Eastern Oregon.'
- Robichaud PR, Brown RE (2002) Silt fences: an economical technique for measuring hillslope soil erosion. USDA Forest Service, Rocky Mountain Research Station General Technical Report RMRS-GTR-94. (Fort Collins, CO)
- Robichaud PR, Lillybridge TR, Wagenbrenner JW (2006) Effects of postfire seeding and fertilizing on hillslope erosion in north-central Washington, USA. *CATENA* 67, 56-67.
- Rosolem CA, Caires EF (1998) Yield and nitrogen uptake of peanuts as affected by lime, cobalt, and molybdenum. *Journal of Plant Nutrition* 21, 827-35.
- Shakesby RA, Coelho COA, Ferreira AJD, Walsh RPD (2002) Ground-level changes after wildfire and ploughing in eucalyptus and pine forests, Portugal: Implications for soil microtopographical development and soil longevity. *Land Degradation & Development* 13, 111-127.
- Smith HG, Sheridan GJ, Lane PNJ, Nyman P, Haydon S (2011) Wildfire effects on water quality in forest catchments: A review with implications for water supply. *Journal of Hydrology* 396, 170-192.
- Spigel KM, Robichaud PR (2007) First-year post-fire erosion rates in Bitterroot National Forest, Montana. *Hydrological Processes* 21, 998-1005.
- Stankov Jovanovic VP, Ilic MD, Markovic MS, Mitic VD, Nikolic Mandic SD, Stojanovic GS (2011) Wild fire impact on copper, zinc, lead and cadmium distribution in soil and relation with abundance in selected plants of Lamiaceae family from Vidlic Mountain (Serbia). *Chemosphere* 84, 1584-1591.
- Stein ED, Brown JS, Hogue TS, Burke MP, Kinoshita A (2012) Stormwater contaminant loading following southern California wildfires. *Environmental Toxicology and Chemistry* 31, 2625-2638.
- Thomas AD, Walsh RPD, Shakesby RA (1999) Nutrient losses in eroded sediment after fire in eucalyptus and pine forests in the wet Mediterranean environment of northern Portugal. *CATENA* 36, 283-302.
- Thomas AD, Walsh RPD, Shakesby RA (2000) Post-fire forestry management and nutrient losses in eucalyptus and pine plantations, northern Portugal. *Land Degradation & Development* 11, 257-271.
- Townsend SA, Douglas MM (2004) The effect of a wildfire on stream water quality and catchment water yield in a tropical savanna excluded from fire for 10 years (Kakadu National Park, North Australia). *Water Research* 38, 3051-3058.
- Úbeda X, Outeiro LR, Sala M (2006) Vegetation regrowth after a differential intensity forest fire in a Mediterranean environment, northeast Spain. *Land Degradation & Development* 17, 429-440.
- US EPA (2003-2007) Ecological soil screening levels. <http://www.epa.gov/ecotox/ecossl/> [accessed 11 March 2013]
- Vega JA, Fernandez C, Fonturbel T (2005) Throughfall, runoff and soil erosion after prescribed burning in gorse shrubland in Galicia (NW Spain). *Land Degradation & Development* 16, 37-51.
- Wagenbrenner JW, MacDonald LH, Rough D (2006) Effectiveness of three post-fire rehabilitation treatments in the Colorado Front Range. *Hydrological Processes* 20, 2989-3006.
- Zhan G, Erich MS, Ohno T (1996) Release of trace elements from wood ash by nitric acid. *Water, Air, & Soil Pollution* 88, 297-311.