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Abstract

Mulching is increasingly employed to stabilize burnt areas, making necessary to elucidate where and how it should be used. The effects of mulching and the efficiency of two straw application strategies in reducing nutrient losses were evaluated in a steep area (burnt with moderate severity) with twelve experimental plots split into three sets: control burnt plots (BS), burnt plots with straw mulching in narrow bands along the contour lines (NM, global dose 800 kg ha^{-1}) and in wide bands (WM, global dose 1000 kg ha^{-1}). None of the mulching strategies had a significant effect on most of the 16 soil and sediment variables analysed (pH, nutrient and trace element concentrations). The principal component analyses show that soil and sediment samples change with time after the fire regardless treatment, decreasing progressively the differences between successive sampling dates. In sediments, pH_{KCl}, Ca, Mg, Mn and Zn fitted to curvilinear regression models with time after fire as independent variable, while the other variables showed no clear temporal trend. During the first post-fire year, <500 kg ha^{-1} of sediments were eroded and mulching had no effect on the total mass of lost sediments and nutrients. We conclude that the erosion rate was rather low that year due to moderate precipitation rates and therefore mulching did not significantly reduce soil erosion. Nevertheless, the concentration of Mo, Mn and Zn in sediments exceeded reference levels for ecosystem protection and can lead to deficiency problems for on-site vegetation and to soil and water pollution off-site.

Keywords: wildfire - burnt area emergency response (BAER) – sediment –soil quality

1. Introduction

Fire is a global phenomenon affecting more land area than any other natural disturbance (Bento-Gonçalves et al., 2012) and one of the major causes of forest destruction and soil degradation (Certini, 2005; Shakesby, 2011).Although fire is a natural driving force in some ecosystems, humans are responsible for most of the current large-scale and intense wildfires (Bento-Gonçalves et al., 2012). In the foreseeing scenarios of climate change and a drastic shift in land use due to rural exodus and socio-economic factors, the number and severity of wildfires will probably increase substantially (Bento-Gonçalves et al., 2012; Birot, 2009; Pereira et al., 2011). Lately, extensive research on the effects of fire on soil properties has been done (see list of comprehensive reviews in Bento-Gonçalves et al., 2012). The impacts on soils depend on fire severity (duration and intensity), frequency, as well as on the season of fire occurrence and the characteristics of the forest floor (Bento-Gonçalves et al., 2012; Certini, 2005). Accordingly, fire effects on the soil can either be reversible or permanent, being climate, vegetation and topography of the burnt area the factors controlling the resilience of the soil system (Certini, 2005). Wildfires often reduce the amount of organic matter in the soil, alter its structure, modify soil biological communities and increase nutrient losses through volatilization, ash entrapment in smoke columns, leaching and erosion (Shakesby, 2011). Research on the effects of fire on soil chemical quality has been mainly focused on assessing the changes in soil organicmatter (SOM) and available macronutrients, and to a lesser extent on micronutrients and trace elements.

Fires lead to an immediate increase in soil concentrations of most nutrients and trace elements due to the substantial quantities of these elements present in ashes from burnt vegetation and also to the release from SOM after its breakdown by the

fire (Certini, 2005, and references therein). Fire effects on soil nutrients and trace elements may last for a few months or for years, depending on how fast they are lost from soil (or taken up by plants). plants). Nutrient losses are conditioned by the solubility of the elements in ashes (Certini, 2005) and by the erosion rate, which is highly dependent on the percentage of bare-ground areas and the post-fire weather pattern (Thomas et al., 1999).

Although SOM losses are the most common phenomena in the short-term (Certini, 2005; Certini et al., 2011; Couto-Vázquez and González-Prieto, 2006; Gómez-Rey and González-Prieto, 2014), increases of SOM in the long term have also been reported in several experiments (meta-analysis by Johnson and Curtis, 2001). The main effects of wildfires on the N cycle are: a significant removal of organic N due to SOM combustion; a redistribution of the remaining soil organic N; the conversion of organic to inorganic N; losses due to leaching, erosion and volatilisation; and a marked alteration of both quantity and specific composition of microbial communities (Certini, 2005; Fisher and Binkley, 2000; Prieto-Fernández et al., 2004). Increased soil inorganic N concentrations after fires have been widely reported. Both measurements of ^{15}N natural abundance in soils and plants (Aranibar et al., 2003; Couto-Vázquez and González-Prieto, 2006) and ^{15}N-tracing experiments have been used to study the effects of burning on the N cycle (Gómez-Rey and González-Prieto, 2013, and references therein) but no common response of gross N transformations to fires was found, most probably due to the large number of influencing factors. Organic P in soils is depleted by fire at a higher rate than the supplied from mineral weathering (DeBano et al., 1998); however, burning is also responsible of the transient increase in P availability resulting from the mineralization of organic P to orthophosphate (Cade-Menun et al., 2000; Saá et al., 1994).

Fewer studies have assessed the effects of burning on soil micronutrients and trace elements, although they are essential for plant development and post-fire changes in their soil availability could lead to deficiencies or toxicity effects on plants (García-Marco and González-Prieto, 2008; Gómez-Rey et al., 2014). The scarce available data on micronutrients and trace elements in burnt ecosystems is often contradictory due to the differences among the studies in terms of fire type and characteristics (wild or prescribed, severity and frequency), climate conditions, vegetation cover, soil type and sampling design (soil/ash/sediment samples, time after the fire, fraction of the elements measured) (Brye et al., 2002; Close et al., 2011; García-Marco and González-Prieto, 2008; Gómez-Rey et al., 2013a; González-Parra et al., 1996; Pereira et al., 2011; Pivello et al., 2010; Ponder et al., 2009; Stankov-Jovanovic et al., 2011).

Erosion is widely recognized as a common process after wildfires (DeBano et al., 1998; Robichaud and Brown, 1999) and post-fire erosion effects on downstream water quality and sedimentation are recognised (Smith et al., 2011). However, it is still controversial whether nutrient losses in eroded sediments affect the long-term soil quality and site productivity (Baird et al., 1999; Gómez-Rey et al., 2013b, 2014; Robichaud et al., 2006; Shakesby et al., 2002; Thomas et al., 1999). Although studies have been mainly focused on nutrient losses in solution rather than on losses with the eroded material (de Koff et al., 2006), nutrients attached to sediments can substantially exceed those lost in solution (Gimeno-Garcia et al., 2000; Smith et al., 2011). Reported annual post-fire erosion rates are highly variable, ranging from 1 to 240 Mg ha^{-1} (Moody and Martin, 2009; Robichaud et al., 2006), as they depend on fire intensity, climate, topography and vegetation of the site, the percentage of bare soil and its infiltration capacity (Fisher and Binkley, 2000; Vega et al., 2005). The greater the intensity of the fire, the higher the amount of nutrients found in eroded sediments (Andreu et al., 1996; Gimeno-Garcia et al., 2000), although high nutrient losses in sediments have also been reported in low intensity fires (de Koff et al., 2006; Gómez-Rey et al., 2013b, 2014). Usually, the greatest nutrient losses are recorded within the first 4-12 months after the fire and they generally decrease by an order of magnitude per year (Gómez-Rey et al., 2013b, 2014; Robichaud et al., 2006).

Different measures have been implemented after wildfires to minimize fire impacts (Bento-Gonçalves et al., 2012) in the frame of the so-called burnt area emergency response (BAER). Post-fire stabilization techniques must be applied as soon as possible after a wildfire, especially when the vegetation cover is completely destroyed, in order to prevent surface runoff and erosion and to accelerate development of vegetation cover (Robichaud, 2009; Robichaud et al., 2000; Vega et al., 2005). Besides seeding of

fast-growing grass species, the most effective and widely used post-fire stabilization technique is straw mulching (Bento-Gonçalves et al., 2012), which has shown an immediate effectiveness in increasing ground cover and thus it alleviates erosion during the first critical months after the fire (Bautista et al., 2009; Díaz-Raviña et al., 2012; Groen and Woods, 2008; Robichaud et al., 2010).

Until recently, the available information about the effects of post-fire straw mulching on chemical properties of soils and sediments was very scarce (Díaz-Raviña et al., 2012; Fontúrbel et al., 2012). Mulching can alter the MIT turnover (by supplying a substrate with high C/N ratio), enhance vegetation recovery (and thus nutrient uptake by plants), modify the runoff-infiltration ratio, or preferentially reduce the erosion of some soil fractions; consequently, this stabilization technique could alter the chemical composition of soils and sediments. However, some studies have already shown that mulching does not affect the nutrient concentrations in soils and sediments when compared to a control burnt soil, but it does effectively reduce erosion. For instance, Vega et al. (2014) conducted a prescribed fire in a steep area with rainy climate and concluded that erosion was significantly reduced by straw mulching on burnt plots during the first year. A similar experiment carried out after a wildfire in another steep area (up to one year after the fire) showed that mulching did not modify the concentration of nutrients in soils and that mulched plots were more similar to the unburnt control than burnt plots without mulching (Gómez-Rey and González-Prieto, 2014).

Despite the fact that Galicia (NW Spain) has a temperate-humid climate, it is one of the European regions with the highest fire incidence (Birot, 2009). The causes are a complex mixture of traditional use of fire as a silvopasture agroforestry tool, deep economic and demographic changes in rural areas and incendiarism as protest (Gómez-Rey et al., 2013a). In a region like Galicia, where the area affected by wildfires is so large, the compromise among applying straw to burnt soils in order to prevent erosion, the costs and difficulties of its application in large or remote burnt areas and the potential competence with other uses (livestock fodder) of this limited resource arises as a controversial issue. Therefore, it is paramount to elucidate where and how is convenient to use straw mulching as a BAER technique. To ensure the 60–70% of ground cover needed to effectively protect the burnt soil against erosion (Robichaud, 2009), straw doses of 2300–2500 kg ha^{-1} have been used in Galicia (Díaz-Raviña et al., 2012; Fernández et al., 2011; Fernández et al., 2012; Gómez-Rey et al., 2013a). Instead of mulching the entire burnt area, a possible strategy to reduce these doses, could be to apply the straw on bands along the contour lines alternating with bands without straw. In such a design, we need to determine how wide the mulched bands should be to effectively protect the unmulched bands against the erosion. The aims of the present study, conducted in an area with a steep slope affected by a medium severity wildfire, are: a) to evaluate the effects of straw mulching on soil and sediment properties; and b) to assess the efficiency of two straw mulching application strategies (narrow and wide bands with a global straw dose of 800 and 1000 kg ha^{-1}, respectively) to reduce nutrient losses in eroded sediments.

2. Materials and methods

2.1. Site description and experimental design

The study area was located in Escairón (Lugo, NW Spain;42º40'2" N, 7º37'1" W, 510-530 m a.s.l.; Fig. 1), under temperate (mean annual temperature 12.7 ºC) and moderately rainy climate (827 mm y^{-1}). In September 2012 a medium severity wildfire burnt 85 ha of forest and shrubland and twelve plots were set up in an experimental field of 1.3 ha with homogeneous slope (33-38%), orientation (S-SW), soil type (Entisol developed over slates) and vegetation cover (dominated by the shrubs *Erica arborea, Ulex europaeus* and *Pterospartum tridentatum*). As it was a wildfire, all plots were distributed in the only suitable homogeneous area (1.3 ha) found in the whole burnt zone (see Fig. 1) and, thus, pseudoreplication can restrict the generalization of our results. The plots were 40x10 m each, with the longest dimension parallel to the maximum slope. To monitor post-fire erosion, the plots were delimited by a geotextile fabric fixed to posts following the design suggested by Robichaud and Brown (2002). One month after the fire, three treatments were arranged in a fully randomized design with four replications: a) control (BS), burnt plots without any stabilization treatment; b) wide mulching (WM), burnt plots with straw applied in the upper half of the plot at 200 g m^{-2} (global plot dose: 1000 kg ha^{-1}); and c) narrow mulching (NM),

burnt plots with straw applied in two alternated 8 m wide bands at 200 g m^{-2} (global plot dose: 800 kg ha^{-1}).

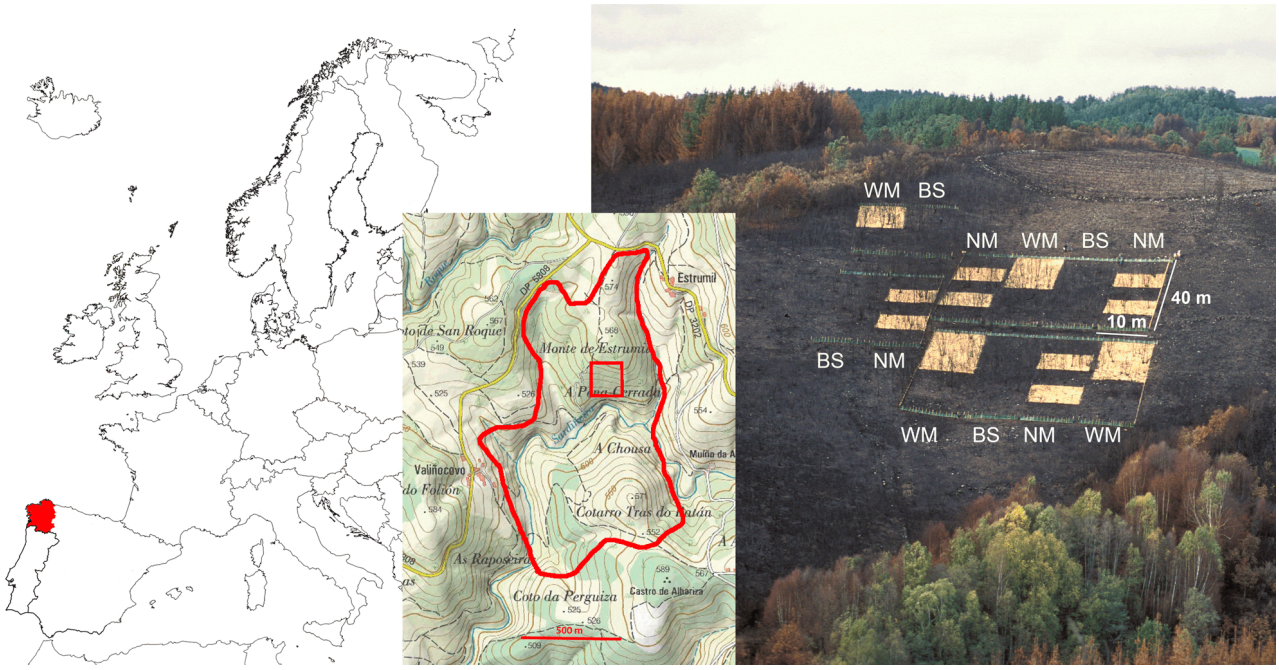

Fig. 1. Map showing the location of the experimental field and the arrangement of treatments.

2.2. *Sampling and chemical analysis of soils, ashes and sediments*

One month after the fire but prior to treatment application, ash samples (ashes and charred plant and litter debris, thereafter referred to as "ashes") and soil samples (0-2.5 cm depth) were collected from four of the corridors between plots (not within the plots to prevent trampling on them). The central corridor in each of the four blocks of plots (see Fig. 1) was selected and 10 squares of 15 × 15 cm per corridor were sampled for ashes and combined in a composite sample per corridor, weighed, homogenized, air dried; the same was done for soil. Once the treatments were applied, in each plot and at each sampling date (3, 6, 9 and 12 months after treatment application) soil samples were collected from 10 uniformly distributed sampling squares. These 10 subsamples were combined to yield one composite sample per plot and sampling date. In plots with mulching, samples were taken separately in the area with straw (NM-1 and WM-1) and without straw (NM-0 and WM-0). We did not find straw on the NM-0 and WM-0 areas, nor in the sediments collectors, in any of the samplings. All soil samples were sieved (2 mm) and homogenised in the laboratory, then they were divided into fresh subsamples, which were kept at 4 ˚C for inorganic N measurements, and air-dried subsamples for the other analyses. After each sediment-producing rain event (December 2012, February 2013,March 2013, April 2013, June 2013 and November 2013), eroded sediments were collected from the geotextile, homogenized and weighed in situ and aliquots of each plot were transported to the laboratory, where they were air-dried for further analyses.

The dry matter content of soils and sediments was assessed by oven-drying sub-samples at 105 ºC for 5 h. Due to the wide differences in water holding capacities of soils, sediments and ashes, different sample:solution ratioswere used to obtain the "solid suspension" needed for pH measurements, aswell as a volume of extract large enough for inorganic N analyses. Soil pH was measured with a pH meter (Metröhm, Switzerland) in water and 1 M KCl employing the standard soil:solution ratio of 1:2.5 , while for sediment and ashes a sample:solution ratio of 1:6 was used. For total N and $\delta^{15}N$ determination, aliquots of air-dried soils, sediments and ashes were finely ground (<100 μm) in a

planetary ball mill (Retsch PM100, Retsch GmbH, Haan, Germany) and then combusted with an elemental analyser (Carlo Erba, Milano, Italy) coupled on-line with an isotopic ratio mass spectrometer (Finnigan Mat, delta C, Bremen, Germany). An elemental reference material (Soil 3 from Eurovector, Milano, Italy) and isotopic standards (IAEA-N1 and IAEA-N2, alternately, from the International Atomic Energy Agency, Vienna, Austria) were included in each set of 10 samples to check the accuracy of the results; if necessary, drift correction was made against internal standards during the run. Inorganic N was extracted by shaking 20 g of soils and sediments or 5 g of ashes for 1 h with 2 M KCl (1:5 soils and sediments:solution ratio and 1:10 ash:solution ratio). Extracts were passed through glass microfibre filters (Whatman GF/A, 125-mm diameter). For NH_4^+-N and NO_3^--N analysis, an extraction-diffusion method described in Fernández-Fernández et al. (2015) was used. For extractable element analyses, soil, sediment and ash samples were shaken for 2 hours with a solution of NH_4Ac 1 M and DTPA 0.005 M (sample:solution ratio 1:5). The extracts and the corresponding blanks were then filtered through cellulose paper (Filter-Laboratory 1242, 90-mm diameter) and analysed by simultaneous inductively coupled plasma optical emission spectrometry (ICP-OES, Varian Vista Pro, Mulgrave, Australia) to quantify Al, Ca, Cu, Fe, K, Mg, Mn, Mo, Na, P and Zn concentrations. A calibration curve prepared with certified standards of all elements was measured beforehand and one of the calibration solutions was routinely included in each set of 30 samples as a quality control and, when necessary, the calibration curve was measured again. Analytical-grade chemicals and type I water (ASTM 2008) were used for analyses.

To ensure the reproducibility and traceability of the results, certified standardswere included in each batch of samples and all analyses were carried out in duplicate, being the mean of both measurements used in the statistical procedures (after doing a third analysis if the coefficient ofvariation was higher than 5%).

2.3. Statistical analyses

The working hypothesis of no effects of straw addition on soil properties regardless sampling date was tested by means of two sets of one-way ANOVAs: a) straw addition does not modify soil properties in-situ (one-way ANOVAs including BS, WM-1 and NM-1 subplots); and b) straw addition does not affect soil properties of the adjacent bare subplots (one-way ANOVAs including BS, WM-0 and NM-0). These two ANOVAs were done for each sampling date and t = 0 values were not included as no treatments were applied yet. Similarly, the hypothesis of no effects of straw addition on sediment composition regardless sampling date was tested by one-way ANOVAs comparing the sediments retrieved in BS, WM and NM plots. Moreover, sediments of each erosion event were compared with soils (t = 0) and ashes as reference values (one-way ANOVA and Dunnett's test) because they are the likeliest sources of sediments. Curvilinear regression models were also developed to check trends on sediment composition throughout the study period. The effectiveness of mulching dose on erosion rates and nutrient losses was assessed by comparing the accumulated mass of sediments and of each nutrient in BS, WM and NM plots with one-way ANOVAs. For all performed ANOVAs, significant differences among the group means were established at $p< 0.05$ (using Tukey's or Dunnett's test) after checking the normal distribution of variables (Shapiro-Wilk's W test) and the equality of variances among treatment groups (Levene's test). A Principal Component Analysis (PCA, based on the correlation matrix, to extract the factors, plus Varimax rotation with Kaiser normalization) was performed to assess the relationships among the studied variables and whether samples are grouped together according to treatment or sampling date. The anti-image correlation matrix (comprising the negative values of the partial correlation coefficients) was analysed to detect soil variables less suitable for the factor analysis (Cu and Mo in soils; ^{15}N and Fe in sediments). After this procedure, the Kaiser-Meyer-Olkin measure of sampling adequacy increased from 0.730 to 0.784 (soils) and from 0.742 to 0.795 (sediments), and the Barlett's test of sphericity was highly significant ($p< 0.0005$). All statistical analyses were performed with SPSS 15.0 software (SPSS Inc., Chicago, IL).

3. Results

The one-way ANOVA checking the effect of straw on soils (Supplementary material, Table A) yielded significant differences among treatments only for NH_4^+-N_{soil} (BS > WM-1 = NM-1) and NO_3^--N_{soil} (BS ≥ WM-1 ≥ NM-1) at t=6, $\delta^{15}N_{soil}$ at t=9 (WM-1 ≥ BS ≥ NM-1) and Ca_{soil} at t=12 (WM-1 ≥ NM-1 ≥ BS) and Mg_{soil} (WM-1 > NM-1 = BS) at t = 12 months. When checking the effect of straw on adjacent unmulched soils (Supplementary material, Table A), there were small but significant differences for Ca_{soil} at t=3 (WM-0 ≥ BS ≥ NM-0) and Fe_{soil} (NM-0 ≥ Control ≥ NM-0) and K_{soil} (WM-0 . NM-0 . BS) at t = 6 months. Most of the studied variables oscillated with time after the fire, although the initial values were higher than the final ones except for Na_{soil} (Table 1).

In the PCA for soil samples (Fig. 2A and Table 2), the first three components jointly explained 70% of the total variance (41%, 19% and 10%, respectively). Soil extractable Mg, K, Zn, Ca, Mn and pH_{KCl} were strongly associated with the positive side of factor 1 (r > 0.7). Regarding factor 2, its positive side was mostly defined by Na and P (r > 0.7) and its negative side by Al and $\delta^{15}N$ (r < -0.6). Inorganic N (both NH_4^+ and NO_3^-) was strongly associated with the positive side of factor 3 (r > 0.7). The distribution of soil samples on the plane defined by the first two factors shows a change of the samples with time after fire, whereas there is no difference among plots from different treatments within each sampling date. Samples from t= 3 and t= 6 months are clearly separated from each other, as well as from those collected at t= 9 and t=12 months, although these last two samplings are intermingled.

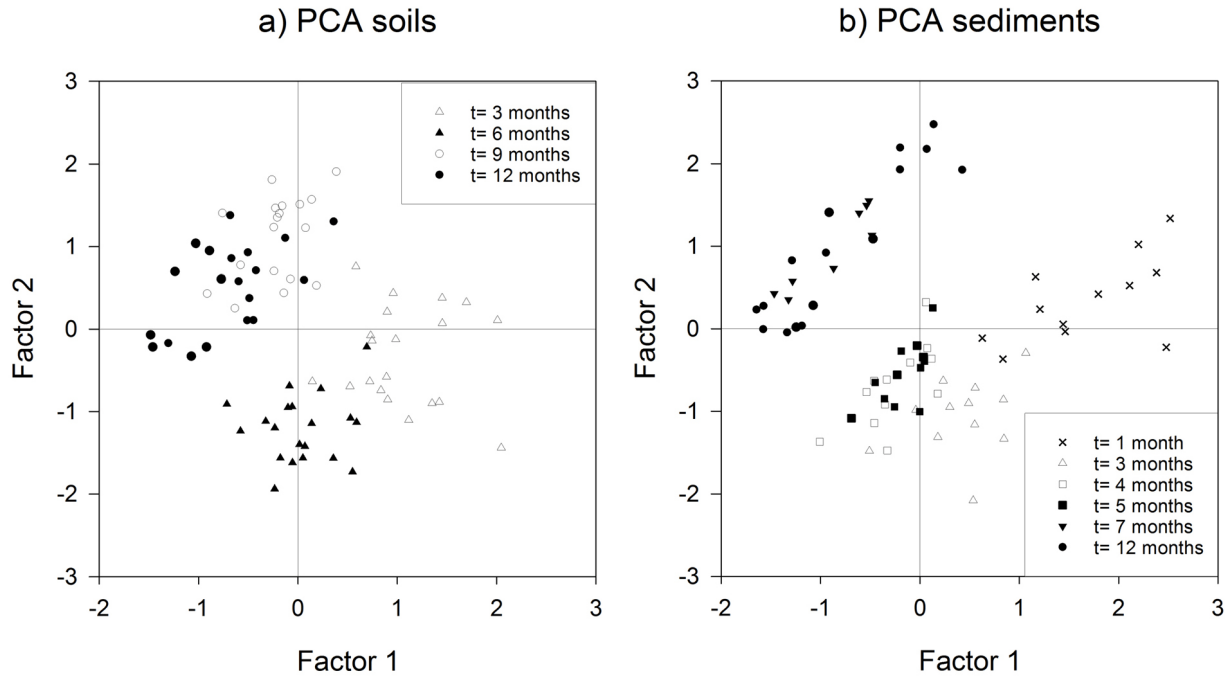

Fig. 2. Score plots from the principal component analyses (PCAs) performed with soil (a) and sediment (b) data for each treatment and sampling time. Symbols represent different sampling times regardless treatment.

Table 1. Mean values ± standard error of pH_{KCl}, $\delta^{15}N$ and concentrations of main nutrients and trace elements in soils for the different treatments and sampling times after the fire. Key: BS, control burnt plots; WM-0, wide-mulched burnt plots, section without straw; WM-1, wide-mulched burnt plots, section with straw; NM-0, narrow-mulched burnt plots, section without straw; NM-1, narrow-mulched burnt plots, section with straw

Sampling time (months)	Plots	pH_{KCl}	Total N (g kg^{-1})	^{15}N (‰)	NH_4^+-N (mg kg^{-1})	NO_3^--N (mg kg^{-1})	Al (mg kg^{-1})	Ca (mg kg^{-1})	Cu (mg kg^{-1})	Fe (mg kg^{-1})	K (mg kg^{-1})	Mg (mg kg^{-1})	Mn (mg kg^{-1})	Mo (mg kg^{-1})	Na (mg kg^{-1})	P (mg kg^{-1})	Zn (mg kg^{-1})
3	BS	3.9 ± 0.1	10.4 ± 0.7	2.0 ± 0.1	98 ± 20	19.2 ± 6.8	570 ± 4	343 ± 13	2.4 ± 0.2	364 ± 28	150 ± 6	77 ± 10	33 ± 3	1.4 ± 0.1	45 ± 4	15 ± 2	6.0 ± 0.5
	WM-0	3.9 ± 0.1	11.0 ± 0.3	1.8 ± 0.2	103 ± 13	16.4 ± 5.3	555 ± 22	391 ± 5	2.4 ± 0.3	355 ± 38	166 ± 8	82 ± 6	38 ± 5	1.4 ± 0.1	55 ± 5	15 ± 3	7.2 ± 0.9
	WM-1	4.0 ± 0.2	10.8 ± 0.5	1.9 ± 0.2	90 ± 12	13.8 ± 3.0	568 ± 18	430 ± 101	2.6 ± 0.2	324 ± 24	180 ± 19	75 ± 14	46 ± 10	1.4 ± 0.1	56 ± 4	17 ± 1	6.7 ± 0.9
	NM-0	3.9 ± 0.1	10.6 ± 0.4	1.8 ± 0.1	105 ± 18	7.9 ± 3.0	563 ± 20	324 ± 21	3.0 ± 0.3	372 ± 34	151 ± 8	74 ± 4	48 ± 19	1.4 ± 0.1	50 ± 2	16 ± 2	6.4 ± 0.5
	NM-1	3.8 ± 0.1	10.7 ± 0.1	1.8 ± 0.1	81 ± 14	8.0 ± 3.2	559 ± 22	282 ± 16	2.7 ± 0.1	386 ± 53	170 ± 16	74 ± 9	40 ± 16	1.4 ± 0.1	46 ± 1	16 ± 2	6.7 ± 0.5
6	BS	3.4 ± 0.1	10.0 ± 0.9	2.1 ± 0.1	66 ± 12	11.1 ± 3.7	538 ± 7	253 ± 27	2.3 ± 0.1	330 ± 14	103 ± 11	49 ± 7	19 ± 2	1.4 ± 0.1	27 ± 4	12 ± 1	4.6 ± 0.5
	WM-0	3.5 ± 0.1	10.0 ± 0.7	2.0 ± 0.2	93 ± 8	12.9 ± 2.3	566 ± 37	284 ± 52	2.5 ± 0.3	294 ± 17	134 ± 4	53 ± 6	24 ± 5	1.4 ± 0.1	31 ± 3	11 ± 1	5.2 ± 0.2
	WM-1	3.4 ± 0.1	9.5 ± 0.5	2.0 ± 0.2	32 ± 8	2.4 ± 0.9	564 ± 26	276 ± 41	2.7 ± 0.2	321 ± 47	97 ± 3	47 ± 5	27 ± 6	1.4 ± 0.1	23 ± 2	14 ± 1	4.6 ± 0.2
	NM-0	3.4 ± 0.1	10.1 ± 0.3	2.1 ± 0.3	59 ± 17	6.1 ± 2.2	541 ± 26	231 ± 31	2.7 ± 0.2	336 ± 45	110 ± 8	53 ± 6	29 ± 13	1.3 ± 0.1	28 ± 3	12 ± 1	5.0 ± 0.5
	NM-1	3.4 ± 0.1	9.8 ± 0.3	2.2 ± 0.1	29 ± 2	2.0 ± 0.4	551 ± 33	227 ± 14	2.6 ± 0.2	337 ± 55	105 ± 2	49 ± 3	22 ± 8	1.3 ± 0.1	27 ± 1	13 ± 1	4.8 ± 0.5
9	BS	3.5 ± 0.1	9.8 ± 0.5	1.8 ± 0.1	76 ± 19	12.3 ± 4.5	444 ± 4	240 ± 23	2.2 ± 0.2	343 ± 16	114 ± 4	54 ± 6	30 ± 4	0.9 ± 0.1	71 ± 5	18 ± 2	3.9 ± 0.5
	WM-0	3.5 ± 0.1	10.2 ± 0.2	1.8 ± 0.1	91 ± 18	16.1 ± 3.8	464 ± 11	227 ± 34	2.4 ± 0.1	323 ± 36	125 ± 15	52 ± 3	30 ± 8	1.0 ± 0.1	73 ± 7	20 ± 4	3.7 ± 0.3
	WM-1	3.5 ± 0.1	9.3 ± 0.5	1.9 ± 0.1	44 ± 14	8.8 ± 3.8	460 ± 15	217 ± 31	2.7 ± 0.3	324 ± 35	112 ± 8	47 ± 8	29 ± 6	0.9 ± 0.1	65 ± 5	22 ± 2	3.8 ± 0.4
	NM-0	3.5 ± 0.1	9.9 ± 0.1	1.5 ± 0.1	74 ± 19	10.0 ± 4.3	449 ± 14	230 ± 27	2.5 ± 0.3	344 ± 58	105 ± 4	57 ± 5	41 ± 18	0.9 ± 0.1	72 ± 4	21 ± 1	4.3 ± 0.3
	NM-1	3.5 ± 0.1	9.9 ± 0.3	1.5 ± 0.1	42 ± 7	3.0 ± 0.8	441 ± 14	241 ± 20	2.6 ± 0.3	357 ± 67	125 ± 6	58 ± 8	42 ± 18	1.0 ± 0.1	77 ± 8	19 ± 1	4.4 ± 0.2
12	BS	3.4 ± 0.1	10.1 ± 0.9	1.8 ± 0.1	32 ± 8	5.1 ± 0.9	401 ± 8	232 ± 31	1.4 ± 0.1	301 ± 12	74 ± 7	37 ± 2	20 ± 4	0.1 ± 0.1	60 ± 6	16 ± 1	2.5 ± 0.5
	WM-0	3.4 ± 0.1	10.4 ± 0.6	1.9 ± 0.2	62 ± 26	9.5 ± 3.0	416 ± 9	202 ± 27	1.5 ± 0.1	275 ± 34	81 ± 11	30 ± 2	18 ± 4	0.1 ± 0.1	61 ± 3	14 ± 2	2.3 ± 0.1
	WM-1	3.6 ± 0.1	11.4 ± 1.1	1.7 ± 0.1	36 ± 12	6.5 ± 2.1	384 ± 17	378 ± 49	1.6 ± 0.2	224 ± 11	93 ± 7	49 ± 9	33 ± 6	0.1 ± 0.1	72 ± 5	16 ± 1	3.1 ± 0.4
	NM-0	3.4 ± 0.1	9.6 ± 0.6	2.1 ± 0.1	54 ± 20	4.9 ± 0.9	414 ± 13	162 ± 36	1.5 ± 0.2	299 ± 49	60 ± 2	29 ± 5	25 ± 16	0.1 ± 0.1	55 ± 7	16 ± 1	2.2 ± 0.2
	NM-1	3.5 ± 0.1	10.3 ± 0.4	1.9 ± 0.1	25 ± 3	4.0 ± 0.4	405 ± 19	273 ± 14	1.6 ± 0.2	297 ± 55	92 ± 10	42 ± 4	27 ± 9	0.1 ± 0.1	69 ± 3	16 ± 1	2.9 ± 0.4

Table 2. Matrix of correlations of soil and sediment variables with the first 3 factors (soils) and 2 factors (sediments) obtained by principal component analyses (PCA).

	Soil components			Sediment components	
Variable	1	2	3	1	2
pH_{KCl}	0.767	0.144	0.242	0.912	0.048
Total N	0.529	0.212	0.247	0.674	0.282
$\delta\ ^{15}N$	-0.341	-0.647	0.304		
NH_4^+-N	0.496	-0.125	0.767	0.034	0.732
NO_3^--N	0.122	-0.038	0.922	-0.468	0.240
Al	0.032	-0.678	0.357	-0.147	-0.849
Ca	0.865	0.182	-0.006	0.885	0.296
Cu				0.658	-0.654
Fe	-0.101	-0.055	-0.050		
K	0.914	0.001	0.154	0.566	0.624
Mg	0.931	0.141	0.051	0.905	0.199
Mn	0.768	0.273	0.128	0.821	0.158
Mo				0.741	-0.583
Na	-0.089	0.800	-0.032	0.674	0.477
P	0.233	0.738	0.236	0.089	0.853
Zn	0.871	-0.232	0.220	0.846	-0.134

The one-way ANOVAs comparing sediments from BS, WM and NM plots for each sampling date yielded no significant differences among treatments for any of the variables (Supplementary material, Table B). Irrespectively of the treatment considered, sediment pH_{KCl}, extractable Ca, Mg, Mn and Zn fitted to curvilinear models with time after treatment application as the independent variable, explaining 50-75% of the variance (Figs. 3 and 4). For the rest of the sediment variables, no suitable regression models were found to explain data variability.

Compared with that of ashes, the pH_{KCl} of the sediments was similar in the first erosion event and lower in those successively eroded (Table 3 and Supplementary material, Table C). Sediments experienced a progressive acidification with time, but their pH_{KCl} was always higher than that of the soil at t=0 (Table 3 and Supplementary material, Table D). Irrespectively of the erosion event, total $N_{sediments}$ and $\delta^{15}N_{sediments}$ were similar to those in ashes but, in most cases, they were higher (total N) or lower ($\delta^{15}N$) than in soil. The concentration of NH_4^+-$N_{sediments}$ (except for those collected at t = 3 months) was well over the ash and soil levels, while almost no differences between sediments, ashes and soils were found for NO_3^--N. Levels of K and Na in sediments were similar or lower than in ashes, but always higher than in soil. Extractable $Ca_{sediments}$ and $Mg_{sediments}$ were lower that Ca_{ashes} and Mg_{ashes} in the first erosion event, but similar in the subsequent events, and much higher than Ca_{soil} and Mg_{soil}. In almost all cases, extractable $Al_{sediments}$ and $Fe_{sediments}$ were higher than in ashes and lower than in soil. Depending on the erosion event considered, sediments showed Mn levels lower or similar to ashes, and higher or similar to soil. If ashes and soil values are taken as a reference, $Zn_{sediments}$ levels were higher in the first 3–4 erosion events and similar in the rest, while Cu levels were lower in the two last erosion events and similar in the previous ones. Finally, the extractable Mo levels in sediments were similar or lower than in ashes but always higher than in soil, whereas $P_{sediments}$ was lower than P_{ashes} except in the last two erosion events but similar to P_{soils} in most cases.

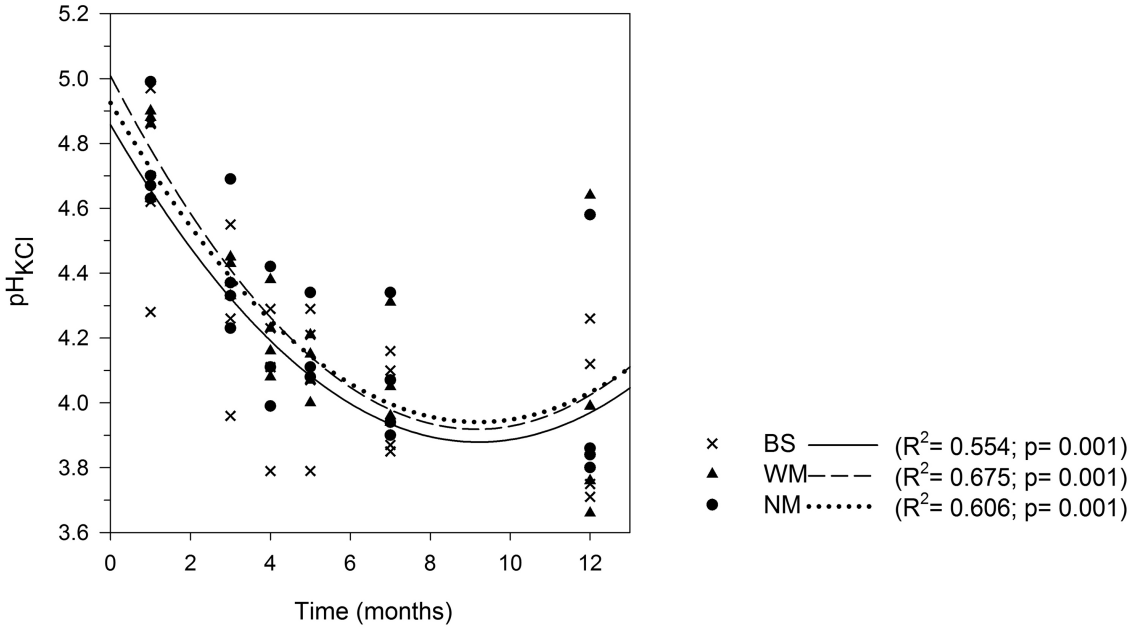

Fig. 3. Significant curvilinear regressionmodels for sediment pHKClwith time after the fire as independent variable. Key: BS, control burnt plots; WM, wide-mulched burnt plots; NM, narrow-mulched burnt plots.

In the PCA for sediment samples (Fig. 2B and Table 2), the first two components jointly explained 71% of the variance (46% and 25%, respectively). The variables most strongly correlated with the positive side of factor 1 were pH_{KCl}, Mg, Ca, Zn, Mn and Mo ($r > 0.7$), whereas P and NH_4^+ were strongly associated ($r > 0.7$) with the positive side of factor 2 and Al and Cu with the negative one ($r < -0.7$). The distribution of sediment samples on the plane defined by the first two factors showed a change with time after fire and lack of differences among treatments within each sampling date. Sediments from t= 1 month are well separated from all the other erosion events, samples from t= 3 are slightly apart from the preceding (t= 1 month) and the two subsequent events (t= 4 and 5 months), the latter being intermingled. Sediments from t=7 and 12 months are close together and further away from the others.

During the first year after the wildfire, less than 500 kg ha^{-1} of sediments were eroded, being the losses of elements lowest for NO_3^--N, Cu, Mo, P and Zn (< 10 g ha^{-1}), intermediate for NH_4^+-N, Al, Fe, K, Mg, Mn and Na (20-100 g ha^{-1}) and highest for Ca (around 400 g ha^{-1}) and total N (5 kg ha^{-1}) (Table 4). While NO_3^--N, Al, Fe and Zn losses accounted for around 10% of the corresponding ash extractable elements, the percentage of the other elements accounted for 5% (total N, NH_4^+-N, Ca and Mg) or less (Cu, K, Mn, Mo, Na and P) (Table 4). If extractable nutrients in ash+topsoil were taken as a reference, the nutrient losses in eroded sediments never exceeded 0.7%, having Al and Fe the lowest losses (0.04 and 0.06% respectively) (Table 4). According to the one-way ANOVA, neither the total mass of sediments nor the mass of any nutrient lost by erosion was affected by the tested treatments (Supplementary material, Table B).

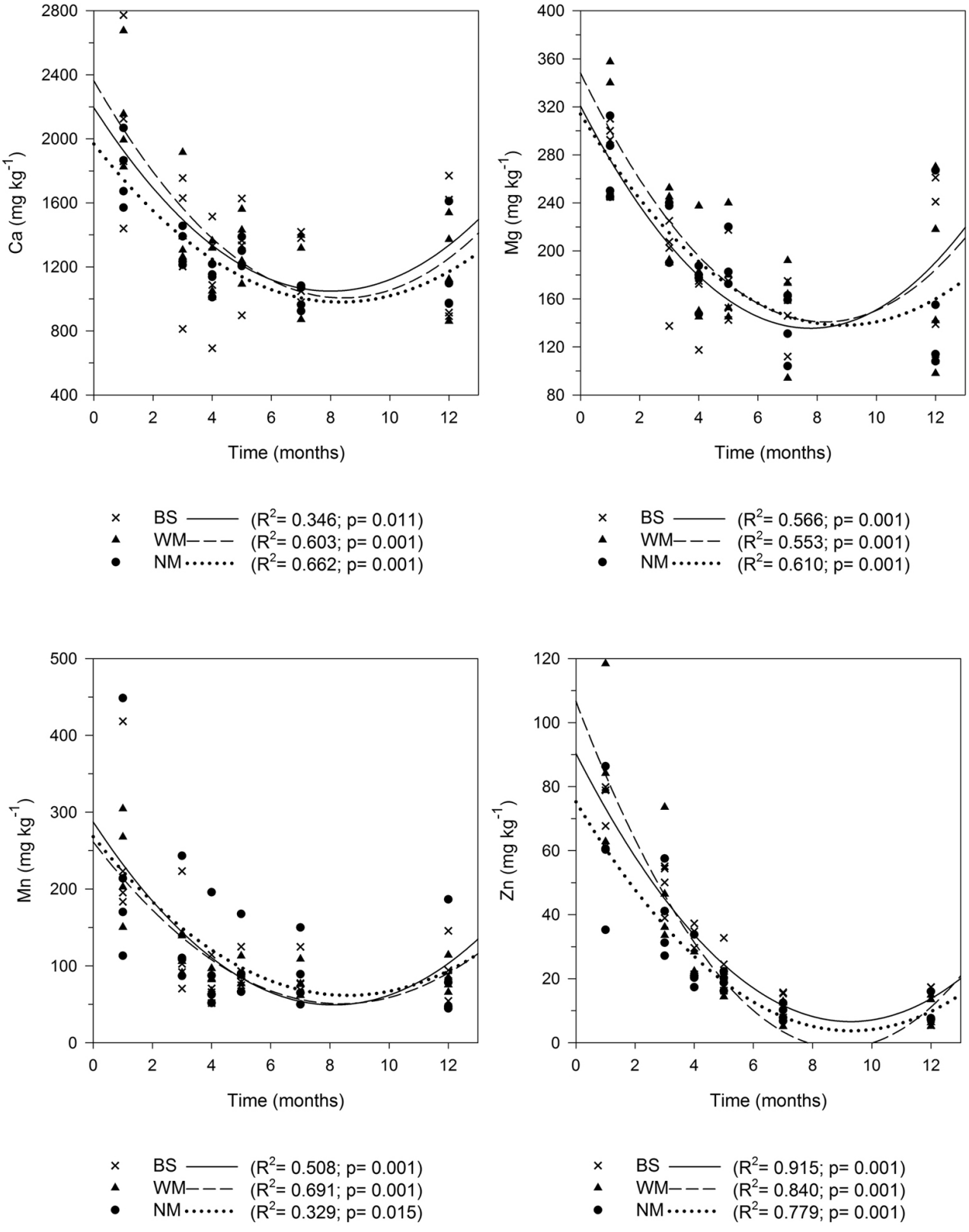

Fig. 4. Significant curvilinear regression models for sediment Ca, Mg, Mn and Zn with time after the fire as independent variable. Key: BS, control burnt plots;WM, wide-mulched burnt plots; NM, narrow-mulched burnt plots.

Table 3 Mean values ± standard error of pH_{KCl}, $\delta^{15}N$ and concentrations of main nutrients and trace elements in sediments for the different plots and erosion events. Key: BS, control burnt plots; WM, wide-mulched burnt plots; NM, narrow-mulched burnt plots. For comparison purposes, soil (t=0) and ash values are also provided

Sampling time (months)	Plots	pH_{KCl}	Total N (g kg^{-1})	^{15}N (‰)	NH_4^+-N (mg kg^{-1})	NO_3^--N (mg kg^{-1})	Al (mg kg^{-1})	Ca (mg kg^{-1})	Cu (mg kg^{-1})	Fe (mg kg^{-1})	K (mg kg^{-1})	Mg (mg kg^{-1})	Mn (mg kg^{-1})	Mo (mg kg^{-1})	Na (mg kg^{-1})	P (mg kg^{-1})	Zn (mg kg^{-1})
1	BS	4.7 ± 0.2	26.7 ± 2.6	0.5 ± 0.1	233 ± 24	10.5 ± 1.9	153 ± 18	2049 ± 279	2.8 ± 0.3	129 ± 15	442 ± 25	287 ± 14	255 ± 55	2.6 ± 0.2	88 ± 13	19 ± 2	72 ± 4
	WM	4.8 ± 0.1	27.8 ± 3.8	0.1 ± 0.3	222 ± 23	6.1 ± 1.0	131 ± 19	2162 ± 184	3.0 ± 0.2	110 ± 12	491 ± 56	308 ± 25	231 ± 34	3.0 ± 0.3	94 ± 11	21 ± 3	86 ± 12
	NM	4.7 ± 0.1	25.6 ± 3.0	0.6 ± 0.3	238 ± 24	5.3 ± 1.0	162 ± 12	1794 ± 110	3.2 ± 0.3	122 ± 9	429 ± 12	274 ± 16	236 ± 74	2.8 ± 0.4	84 ± 3	13 ± 1	61 ± 10
3	BS	4.3 ± 0.1	18.3 ± 1.7	0.9 ± 0.1	74 ± 6	3.9 ± 0.5	222 ± 15	1350 ± 215	2.5 ± 0.2	184 ± 11	212 ± 23	193 ± 19	122 ± 34	2.2 ± 0.1	68 ± 7	11 ± 0	50 ± 4
	WM	4.4 ± 0.1	21.3 ± 3.4	0.7 ± 0.3	79 ± 4	2.6 ± 0.2	233 ± 30	1468 ± 151	3.0 ± 0.2	188 ± 13	258 ± 31	220 ± 17	120 ± 13	2.3 ± 0.1	75 ± 6	12 ± 1	47 ± 9
	NM	4.4 ± 0.1	19.8 ± 1.0	0.9 ± 0.2	89 ± 5	2.5 ± 0.4	293 ± 69	1323 ± 59	3.5 ± 0.6	210 ± 55	282 ± 26	226 ± 12	137 ± 36	2.2 ± 0.2	73 ± 4	11 ± 1	39 ± 7
4	BS	4.1 ± 0.1	17.3 ± 2.4	0.7 ± 0.1	189 ± 22	10.2 ± 0.5	214 ± 12	1156 ± 178	2.5 ± 0.2	166 ± 9	205 ± 20	160 ± 14	76 ± 14	2.1 ± 0.1	50 ± 8	13 ± 1	31 ± 4
	WM	4.2 ± 0.1	20.2 ± 3.9	0.6 ± 0.3	224 ± 16	10.6 ± 1.3	226 ± 32	1185 ± 90	2.4 ± 0.1	147 ± 9	250 ± 26	179 ± 21	82 ± 6	2.1 ± 0.1	62 ± 8	14 ± 2	24 ± 2
	NM	4.2 ± 0.1	15.9 ± 1.8	1.0 ± 0.2	193 ± 9	7.4 ± 1.2	269 ± 19	1129 ± 43	3.0 ± 0.3	164 ± 16	237 ± 8	173 ± 9	99 ± 33	2.1 ± 0.1	61 ± 6	11 ± 1	23 ± 4
5	BS	4.1 ± 0.1	24.3 ± 2.1	0.4 ± 0.1	194 ± 14	12.2 ± 1.3	233 ± 20	1306 ± 151	2.4 ± 0.2	161 ± 14	222 ± 20	172 ± 17	93 ± 11	2.1 ± 0.1	49 ± 5	14 ± 1	24 ± 4
	WM	4.1 ± 0.1	25.4 ± 3.9	0.3 ± 0.3	207 ± 9	9.0 ± 1.3	225 ± 26	1330 ± 103	2.3 ± 0.1	143 ± 6	246 ± 18	179 ± 22	91 ± 8	2.1 ± 0.1	54 ± 5	14 ± 1	19 ± 2
	NM	4.2 ± 0.1	22.0 ± 2.5	0.5 ± 0.3	207 ± 7	7.8 ± 1.3	265 ± 22	1278 ± 42	2.7 ± 0.2	156 ± 13	247 ± 11	187 ± 11	103 ± 22	2.1 ± 0.1	55 ± 4	13 ± 1	19 ± 1
7	BS	4.0 ± 0.1	17.7 ± 1.6	0.9 ± 0.1	208 ± 7	16.4 ± 6.6	124 ± 17	1207 ± 112	1.0 ± 0.1	121 ± 10	310 ± 30	148 ± 13	87 ± 13	0.2 ± 0.1	60 ± 6	21 ± 4	12 ± 2
	WM	4.1 ± 0.1	16.6 ± 2.0	0.7 ± 0.3	229 ± 13	15.9 ± 6.3	111 ± 11	1165 ± 120	1.0 ± 0.1	100 ± 4	337 ± 68	156 ± 21	81 ± 10	0.2 ± 0.1	64 ± 6	20 ± 4	10 ± 2
	NM	4.1 ± 0.1	16.1 ± 1.2	0.8 ± 0.3	237 ± 54	9.4 ± 4.1	139 ± 14	1010 ± 39	1.3 ± 0.1	108 ± 12	321 ± 44	139 ± 14	88 ± 22	0.2 ± 0.1	59 ± 2	18 ± 3	9 ± 1
12	BS	4.0 ± 0.1	19.4 ± 2.4	1.3 ± 0.1	288 ± 43	21.2 ± 5.0	158 ± 33	1300 ± 230	1.2 ± 0.1	203 ± 17	400 ± 71	188 ± 37	96 ± 19	0.2 ± 0.1	83 ± 17	23 ± 4	14 ± 3
	WM	4.0 ± 0.2	19.7 ± 2.1	1.1 ± 0.3	259 ± 78	52.8 ± 24.8	151 ± 50	1224 ± 148	1.2 ± 0.1	153 ± 16	315 ± 52	182 ± 38	85 ± 10	0.2 ± 0.1	69 ± 9	20 ± 3	10 ± 2
	NM	4.0 ± 0.2	18.0 ± 1.5	1.3 ± 0.3	290 ± 73	34.0 ± 11.1	205 ± 22	1163 ± 152	1.7 ± 0.2	202 ± 9	284 ± 52	161 ± 37	89 ± 33	0.1 ± 0.1	71 ± 12	18 ± 2	10 ± 2
Ashes	Mean	4.8 ± 0.1	17.9 ± 0.5	1.2 ± 0.1	145 ± 15	3.9 ± 2.4	64 ± 6	1150 ± 95	2.6 ± 0.2	54 ± 11	351 ± 12	172 ± 11	146 ± 11	1.9 ± 0.2	86 ± 5	28 ± 1	12 ± 1
Soil (t=0)	Mean	3.3 ± 0.1	10.7 ± 0.8	2.6 ± 0.4	110 ± 27	11.2 ± 10.4	540 ± 23	282 ± 89	2.4 ± 0.2	292 ± 70	121 ± 17	46 ± 16	37 ± 28	1.6 ± 0.2	35 ± 3	13 ± 2	6 ± 1

Table 4 Accumulated mass of sediments and nutrients lost by erosion (mean values ± standard error), ash nutrients lost by erosion (%) and ash+topsoil nutrients lost by erosion (%) for the different plots. Key: BS, control burnt plots; WM, wide-mulched burnt plots; NM, narrow-mulched burnt plots

	Mass of sediments and nutrients lost by erosion			Ash nutrients lost by erosion (%)			Ash+topsoil nutrients lost by erosion (%)		
	BS	WM	NM	BS	WM	NM	BS	WM	NM
Sediments (kg ha^{-1})	347 ± 372	310 ± 172	414 ± 309						
Total N (kg ha^{-1})	4.81 ± 1.09	6.40 ± 1.37	5.81 ± 0.34	3.86	5.13	4.66	0.17	0.23	0.21
NH_4^+-N (g ha^{-1})	44.2 ± 12.3	50.4 ± 11.6	56.0 ± 7.9	4.36	4.98	5.54	0.15	0.18	0.20
NO_3^--N (g ha^{-1})	2.94 ± 1.23	8.94 ± 5.05	3.28 ± 1.05	10.74	32.63	11.97	0.10	0.32	0.12
Extractable Al (g ha^{-1})	45.4 ± 17.1	61.2 ± 20.7	65.0 ± 13.3	10.19	13.71	14.58	0.03	0.05	0.05
Extractable Ca (g ha^{-1})	336 ± 87	430 ± 94	394 ± 36	4.19	5.36	4.92	0.43	0.55	0.50
Extractable Cu (g ha^{-1})	0.48 ± 0.09	0.69 ± 0.17	0.81 ± 0.16	2.64	3.77	4.42	0.08	0.11	0.13
Extractable Fe (g ha^{-1})	39.3 ± 14.6	45.6 ± 13.0	47.7 ± 10.4	10.45	12.10	12.66	0.05	0.06	0.06
Extractable K (g ha^{-1})	71.4 ± 19.7	96.1 ± 26.7	89.8 ± 11.4	2.92	3.92	3.67	0.22	0.29	0.28
Extractable Mg (g ha^{-1})	47.5 ± 13.7	61.9 ± 15.9	59.9 ± 5.8	3.96	5.16	5.00	0.35	0.45	0.44
Extractable Mn (g ha^{-1})	32.5 ± 9.0	37.1 ± 8.7	39.8 ± 10.2	3.19	3.65	3.91	0.31	0.36	0.39
Extractable Mo (g ha^{-1})	0.41 ± 0.11	0.51 ± 0.11	0.52 ± 0.07	3.06	3.84	3.92	0.10	0.13	0.13
Extractable Na (g ha^{-1})	15.3 ± 3.5	21.7 ± 6.2	20.2 ± 1.9	2.56	3.62	3.36	0.16	0.23	0.21
Extractable P (g ha^{-1})	3.57 ± 0.81	4.63 ± 0.89	3.77 ± 0.34	1.86	2.41	1.97	0.10	0.13	0.11
Extractable Zn (g ha^{-1})	8.78 ± 1.38	10.33 ± 0.95	9.19 ± 0.55	10.44	12.28	10.92	0.59	0.70	0.62

4. Discussion

Fires lead to an increased soil pH as a result of the oxides and carbonates of basic ions supplied by ashes (Certini, 2005). Although soil alkalinity due to burning might persist during several years (Antos et al., 2003), it is supposed to slowly return to pre-fire values due to leaching of basic ions during thewet season (see references in Couto-Vázquez et al., 2011). Although we do not have pre-fire values, our results for soil pH seem to agree with other studies in the same region (Couto-Vazquez and González-Prieto, 2006; Gomez-Rey and González-Prieto, 2014), in which soil pH of burnt plots decreased to values similar to unburnt plots within the first year after the fire. Concurrently, there was a significant acidification of sediments with time which can be explained by the loss of soil exchangeable cations by leaching and runoff as reported elsewhere (Gomez-Rey et al., 2013b; Robichaud et al., 2006). As in other experiments testing mulching effectiveness, there was no effect of straw addition on the pH of either soils or sediments (Gomez-Rey et al., 2013a,b; Gomez-Rey and González-Prieto, 2014; Robichaud et al., 2006).

Soil and sediment total N concentrations neither showed a clear trend during the study nor were affected by mulching, as also reported Gomez-Rey et al. (2013a,b). As these authors suggest, although N-rich sediments are continuously eroded, these losses are likely not reflected in the successive measurements of soil total N concentration because the big soil N pool masks those losses. Similarly, neither $\delta^{15}N_{soil}$ nor $\delta^{15}N_{sediment}$ showed a trend with time and they were not affected by treatments, although Gomez-Rey et al. (2013a,b) found that both variables were influenced by time after the fire due to enhanced N outputs (nitrates) depleted in ^{15}N. It seems that in our case N losses through erosion and leaching are not that important to modify ^{15}N, likely due to the four-fold lower annual precipitation in our study area. Sediments and ashes shared similar concentrations of total N, which were 1.5-2 fold higher than soil values, whereas $\delta^{15}N_{sediments}$ and $\delta^{15}N_{ashes}$ were also similar but slightly below soil levels; both patterns agree with the data presented in Gomez-Rey et al. (2013b) and shows that the

contribution of ashes to the eroded sediments was higher than that of soil.

The oscillating values of soil NH_4^+-N contrast with the decrease with time recorded in other studies (Gimeno-Garcia et al., 2000; Gomez-Rey et al., 2013a; Gomez-Rey and Gonzalez-Prieto, 2014), most probably due to differences in erosion rates as the soil NH_4^+-N decrease and the extent of erosion of N-rich ashes and sediments are tightly coupled (Gomez-Rey et al., 2013a,b). In sediments the concentration of extractable NH_4^+ was within the same range as in a prescribed burnt area (Gomez-Rey et al., 2013b) and no clear trend with time was found in any case, most probably because the NH_4^+-N pool is subjected to simultaneous and counteracting processes. Extractable NH_4^+-N in sediments is often abundant right after burning because of the high concentration of extractable NH_4^+-N in the ash, but it decreases with time due to ash loss by erosion, translocation and condensation in deeper soil layers, uptake by newly sprouted vegetation, and nitrification (see de Koff et al., 2006, and references therein). In our case, the concentration of NH_4^+-N in sediments was higher than in soils and ashes, while Gomez-Rey et al. (2013b) found that NH_4^+-N concentrations in sediments were also higher than those of burnt soils, but only half than in ashes. Soil NH_4^+-N was not significantly affected by mulching application, as also reported in other studies (Gomez-Rey et al., 2013a; Gomez-Rey and González-Prieto,2014). For NH_4^+-N in sediments we did not record an effect of mulching whereas Gomez-Rey et al. (2013b) found that it reduced sediment NH_4^+-N during the second half of the study.

Post-fire increases in soil NO_3^--N of different magnitude and timing have been reported (Couto-Vázquez and González-Prieto, 2006; Chandler et al., 1983; Gómez-Rey and González-Prieto, 2014; Prieto-Fernández et al., 1993). Soil NO_3^--N concentrations in our experiment were similar to those reported after a wildfire nearby (Gómez-Rey and González-Prieto, 2014) but much lower than in burnt plots after a prescribed fire in the same region (Gómez-Rey et al., 2013a). The concentrations of NO_3^--N in the sediments collected throughout the first post-fire year agree with the results of several authors (Gimeno-García et al., 2000; Gómez-Rey et al., 2013b; Robichaud and Brown, 1999; Robichaud et al., 2006), but the concentration of NO_3^--N in ashes was rather low (25 times lower than in Gómez-Rey et al., 2013b) and within the sediment's range instead of well above them. There was no effect of mulching on soil NO_3^--N (same as in Gómez-Rey et al., 2013a, and Gómez-Rey and González-Prieto, 2014) and sediment NO_3^--N was not affected by straw addition either, although other studies showed lower sediment NO_3^--N concentrations in mulched plots (Gómez-Rey et al., 2013b). These results confirm the hypothesis of Gómez-Rey et al. (2013a) that mulching does not have an effect on net mineralization and nitrification, as neither NH_4^+-N nor NO_3^--N concentrations are affected by treatment application.

Soil P concentration was within the same range reported in other studies (Gómez-Rey et al., 2013a; Gómez-Rey and González-Prieto, 2014), but instead of decreasing with time it oscillates during the study period. High post-fire P_{soil} concentrations are a result of the increased mineralisation of organic P (Cade-Menun et al., 2000; Saá et al., 1994) and the higher solubility of inorganic P when pH increases towards neutrality (Certini, 2005). Soil P tends to decrease afterwards due to sediment losses (Andreu et al., 1996; Gómez-Rey et al., 2013b; Saá et al., 1994) and to orthophosphate chemisorption on Al, Fe and Mn oxides if the soils are acidic (Certini, 2005), but most probably sediment losses are too low in our experiment to cause a significant P depletion in soils. The concentration of $P_{sediments}$ was almost the same as in ashes and slightly higher than in soils. Compared to our data, in Thomas et al. (1999) soils had similar concentration of P, but the sediments and ashes showed much higher values most probably due to the higher intensity of the fire. Our data on $P_{sediments}$ agrees with the values recorded by Gómez-Rey et al. (2013b), although in the latter study ash levels of P were 6 times higher than those in sediments. Like for N, the application of mulching had no effect on soil and sediment P as also reported by Gómez-Rey et al. (2013b) and Gómez-Rey and González-Prieto (2014).

In general the availability of basic cations and micronutrients increases due to the combustion of soil organic matter and to the accumulation of ashes, but eventually the concentration of these nutrients decreases (Certini, 2005; Khanna et al., 1994). The concentrations of basic cations in sediments were similar to those recorded elsewhere (de Koff et al., 2006; Gómez-Rey et al., 2013b; Robichaud and Brown, 1999; Thomas et al., 1999),

although in our case only $Ca_{sediments}$ and $Mg_{sediments}$ decreased with time. While in Gómez-Rey et al. (2013b) ashes were always richer in basic cations than sediments, in the present study ash and sediment levels were within the same range. The addition of mulching does not affect the concentration of basic cations in burnt soils and sediments, although Gómez-Rey et al. (2013a,b) concluded that mulching had little but positive effects on soil K, Mg and Ca mostly by reducing erosion. On the contrary, these authors found that the concentration of basic cations in sediments was not affected by mulching.

In our soils the values for Al, Mn, Mo and Zn were similar to those reported elsewhere (García-Marco and González-Prieto, 2008; Gómez-Rey et al., 2013a; Gómez-Rey and González-Prieto, 2014), whereas Cu was slightly higher compared to these studies and Fe was slightly lower than in Gómez-Rey et al. (2013a) and Gómez-Rey and González-Prieto (2014). For sediments, Mn and Zn were the only trace elements that significantly decreased with time; and when compared to the data from Gómez-Rey et al. (2014), the concentrations in our experiment are similar for Zn, lower for Al, Cu and Fe and higher for Mn. It must be highlighted that, in our case, Mo, Mn and Zn concentrations in sediments exceeded the reference levels for ecosystem protection (CEC, 1986; DEC, 2010; EPA, 2007a,b;Macías Vázquez and Calvo de Anta, 2009), and that might negatively impact the vegetation recolonizing the burnt area and also lead to toxicity risks in the sedimentation area (water and soil) as also suggested Gómez-Rey et al. (2014). Twelve months after the fire, our sediments had less Al and Fe and more Mn and Zn compared to soils, whereas in Gómez-Rey et al. (2013b) sediments and soil concentrations were similar except for Cu, Mn and Zn, which had higher concentrations in sediments. Levels of trace elements in ashes were in general within the sediment range both in our experiment and in Gómez-Rey et al. (2014), but the absolute concentrations in ashes were 2–6 fold higher in the latter than in present study. The lack of mulching effects on the concentration of micronutrients and trace elements in soils and sediments is in agreement with the data provided by Gómez-Rey et al. (2014).

The PCAs show that both soil and sediment samples change with time according to the measured variables, being the differences among sampling dates more pronounced during the first half of the experiment. Soils sampled 9 and 12 months after the fire cannot be distinguished between them, contrasting with the differences between samples taken 8 and 12 months after the fire in a similar field experiment carried out by Gómez-Rey and González-Prieto (2014). These contrasting results are likely due to differences in site characteristics (soil quality, fire severity, dominance of resprouters or obligate seeders, altitude and orientation of the area), weather conditions (temperature and rain regimes) and relative importance of erosion, as all these factors affect the speed of recovery of the vegetation cover and the characteristics of the soil remaining in situ. Due to a more severe fire and harsher climatic conditions in the field site studied by Gómez-Rey and González-Prieto (2014), burning effects on different soil properties were still noticeable 4 years after the fire (authors unpublished data), while in the present study site values from 2 years after the fire indicate no differences on the measured soil properties compared to t = 12 months (data not shown). The variables most strongly associated with the first two factors are rather similar between the PCA_{soils} and $PCA_{sediments}$ and they are also similar to the PCA loadings in Gómez-Rey and González-Prieto (2014).

The lack of differences among control andmulched plots in the accumulated mass of sediments is most probably due to low erosion rates in our site, where the annual precipitation was 827 mm. These results contrast with those from a similar experiment carried out nearby but under much rainier conditions (3036 mm y^{-1}, Gómez-Rey et al., 2013b), where the total mass in the accumulated sediments of most of the analysed elements was much higher in their control and seeding plots than: a) in their mulched plots; and b) the amount retrieved in all plots from our experiment (up to 35-fold).Moreover, the mass of nutrients lost by erosion in their mulching plots was similar in magnitude to the losses in all our plots (slightly lower for K, Ca, Mn, Zn, Cu and Mo; similar for total N, NH_4^+, NO_3^- and Na; and slightly higher for Mg, P, Al and Fe). Therefore, it can be concluded that with moderate precipitation rates, erosion of burnt areas is not significantly reduced by mulching addition. This fact should be taken into account when planning where to use this post-fire

stabilization technique, especially considering the costs and difficulties of its application in large or remote burnt areas and also the potential competence with straw use as livestock fodder. Despite the low erosion rates compared to other fires, it should be highlighted again that in the present study burning lead to exceed the reference levels for ecosystem protection of Mo, Mn and Zn in sediments.

The percentage of ash elements lost by erosion could be useful when assessing the potential recovery of vegetation, especially for micronutrients in repeatedly burnt areas where they can become limiting. In our case, $Fe_{sediments}$ and $Zn_{sediments}$ accounted for a high percentage of ash elements lost by erosion. All the studied elements but Mo accounted for a higher percentage of ash elements lost by erosion in our experiment than in the mulching plots from the prescribed fire studied by Gómez-Rey et al. (2013b) and Gómez-Rey and González-Prieto (2014). Despite the much lower erosion rates recorded in our experiment when compared to the prescribed fire, $NO_{3\ sediments}^{-}$, $Cu_{sediments}$, $K_{sediments}$ and $Na_{sediments}$ in our plots accounted for an even higher percentage of ash elements than in the control and seeding plots from the prescribed fire. Nonetheless, to compare the two studies in a comprehensive way is rather difficult due to several dissimilarities between both sites: a) slates vs. granitic parent material (i.e. fine textured soil with good aggregation vs. coarse textured soil with weak aggregates, leading to differences in water holding capacity, heat transfer, organic matter stabilization, etc.); b) S vs. NW orientation, which usually exert a great influence on litter accumulation; c) wildfire vs. prescribed fire with fuel addition; and d) 0.70 kg m^{-2} of ashes collected one month after the fire vs. 1.29 kg m^{-2} of ashes sampled just after the fire. Although precipitation in our study area before ash sampling was low and likely prevented significant losses by lixiviation and runoff, a fraction of ash enriched in some nutrients might have been wind-eroded before ash sampling and treatment application as suggests the rather different ash elemental composition and the proportion ash:sediments for the different elements between the two studies. If this occurred in our experimental site, the percentages of ash elements lost by runoff might be overestimated when compared to other studies without ash losses before sampling.

For most elements, the erosion losses expressed as percentage of nutrient pools in ash + topsoil were within the same range in our study and in the mulched plots of Gómez-Rey et al. (2013b) and Gómez-Rey and González-Prieto (2014), indicating again how low was the erosion rate in the present study.

5. Conclusions

This field experiment after a medium-severity wildfire in a steep area confirms the already suggested hypothesis that mulching has no impact on soil and sediment concentrations of nutrients and trace elements and it does not affect net soil N rates (mineralisation and nitrification). Both soils and sediments change with time in a similar way and it seems that the PCAs including the studied variables can be a useful tool to assess whether burnt soils and the eroded sediments in the area are changing with time and the speed of the change. Despite the low erosion rates due to moderate precipitation, the concentration of Mo, Mn and Zn in sediments is above reference levels for ecosystem protection. However, straw mulching does not have a significant effect in reducing soil erosion and the subsequent loss of nutrients with such low erosion rates; and we found no differences between the plots with straw mulch applied in wide and narrow bands along the contour lines (higher and lower straw dose per plot respectively).

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Table A. Statistics of ANOVA-1 to check: A) the effect of mulch (WM-1 and NM-1 vs BS) on pH_{KCl}, $\delta^{15}N$ and concentrations of main nutrients and trace elements in soils from each sampling time after the fire (3, 6, 9 and 12 months); and B) the effect of mulching on the downslope unmulched areas (WM-0 and NM-0 vs BS). Key: BS, control burnt plots; WM-0, wide-mulched burnt plots, section without straw; WM-1, wide-mulched burnt plots, section with straw; NM-0, narrow-mulched burnt plots, section without straw; NM-1, narrow-mulched burnt plots, section with straw. Significant differences are highlighted in bold

	A) WM-1 and NM-1 vs BS								**B) WM-0 and NM-0 vs BS**							
	3 months		6 months		9 months		12 months		3 months		6 months		9 months		12 months	
	F	Sig.	F	Sig.	F	Sig.	F	Sig.	F	Sig.	F	Sig.	F	Sig.	F	Sig.
pH_{KCl}	0.311	0.741	0.159	0.855	0.059	0.943	2.425	0.144	0.039	0.962	0.797	0.480	0.068	0.935	0.031	0.969
Total N	0.135	0.876	0.673	0.537	0.492	0.627	0.622	0.558	0.297	0.750	0.006	0.994	0.426	0.666	0.380	0.694
$\delta^{15}N$	0.529	0.606	0.384	0.692	4.422	**0.046**	0.536	0.603	0.608	0.565	0.797	0.484	3.034	0.098	0.900	0.440
NH_4-N	0.299	0.749	6.015	**0.022**	1.812	0.218	0.403	0.680	0.042	0.959	2.017	0.189	0.241	0.790	0.081	0.923
NO_3-N	1.439	0.287	5.371	**0.029**	1.907	0.204	0.834	0.465	1.261	0.329	1.539	0.266	0.536	0.603	1.917	0.203
Al	0.132	0.878	0.285	0.759	0.701	0.521	0.528	0.607	0.186	0.833	0.332	0.726	1.002	0.405	0.601	0.569
Ca	1.796	0.227	0.686	0.528	0.299	0.748	4.815	**0.038**	5.803	**0.024**	0.493	0.627	0.054	0.948	1.201	0.345
Cu	0.689	0.527	1.409	0.293	0.897	0.442	0.619	0.560	1.688	0.239	0.822	0.470	0.465	0.642	0.209	0.816
Fe	0.742	0.503	0.035	0.966	0.141	0.870	1.722	0.233	0.071	0.932	8.126	**0.012**	0.082	0.922	0.170	0.846
K	1.054	0.388	0.415	0.672	1.393	0.297	1.751	0.228	1.335	0.311	4.346	**0.048**	0.807	0.479	1.912	0.203
Mg	0.017	0.983	0.052	0.949	0.601	0.569	10.456	**0.006**	0.316	0.737	0.119	0.889	0.216	0.810	1.870	0.209
Mn	2.525	0.141	2.426	0.150	0.312	0.741	2.640	0.132	1.209	0.348	1.177	0.356	0.288	0.757	2.113	0.183
Mo	0.043	0.958	0.168	0.848	1.009	0.402	0.413	0.674	1.067	0.384	0.336	0.723	3.703	0.067	0.720	0.513
Na	4.136	0.053	0.948	0.423	0.441	0.658	1.591	0.256	2.002	0.191	0.463	0.643	0.042	0.959	0.276	0.765
P	0.788	0.484	1.190	0.348	1.629	0.249	0.093	0.912	0.075	0.929	1.073	0.382	0.951	0.426	0.484	0.632
Zn	0.425	0.666	0.099	0.907	0.706	0.519	0.606	0.566	0.794	0.481	0.752	0.499	0.600	0.570	0.406	0.678

Degrees of freedom: among groups, 2; between groups, 9.

Table B. Statistics of ANOVA-1 to check the effect of mulch (WM and NM vs BS) on pH_{KCl}, $\delta^{15}N$ and concentrations of main nutrients and trace elements in sediments from each erosion event (1, 3, 4, 5, 7 and 12 months after the fire), as well as for the accumulated mass of nutrients lost by erosion. Key: BS, control burnt plots; WM, wide-mulched burnt plots; NM, narrow-mulched burnt plots

	Erosion event												Accumulated mass of nutrients	
	1 month		3 months		4 months		5 months		7 months		12 months			
	F	Sig.	F	Sig.	F	Sig.	F	Sig.	F	Sig.	F	Sig.	F	Sig.
pH_{KCl}	0.402	0.681	0.163	0.852	0.348	0.715	0.172	0.845	0.190	0.830	0.032	0.968	----	----
Total N	0.120	0.888	0.442	0.656	0.586	0.576	0.342	0.719	0.254	0.781	0.201	0.822	0.607	0.566
$\delta^{15}N$	0.833	0.466	0.163	0.852	0.722	0.512	0.197	0.825	0.102	0.904	0.268	0.771	----	----
NH_4-N	0.125	0.884	2.236	0.163	1.344	0.309	0.533	0.604	0.212	0.813	0.069	0.934	0.305	0.744
NO_3-N	4.176	0.052	3.461	0.077	2.656	0.124	3.019	0.099	0.459	0.646	0.993	0.408	1.212	0.342
Al	0.927	0.430	0.744	0.502	1.576	0.259	0.880	0.448	0.933	0.428	0.620	0.559	0.359	0.708
Ca	0.860	0.455	0.247	0.786	0.056	0.946	0.058	0.944	1.129	0.365	0.143	0.869	0.377	0.696
Cu	0.585	0.577	1.697	0.237	2.582	0.130	1.613	0.252	2.401	0.146	3.676	0.068	1.289	0.322
Fe	0.597	0.571	0.174	0.843	0.797	0.480	0.661	0.540	1.386	0.299	3.904	0.060	0.114	0.893
K	0.825	0.469	1.773	0.224	1.404	0.295	0.778	0.488	0.073	0.930	1.034	0.394	0.402	0.681
Mg	0.781	0.487	1.110	0.371	0.383	0.693	0.200	0.822	0.246	0.787	0.139	0.872	0.385	0.691
Mn	0.049	0.953	0.105	0.901	0.343	0.719	0.177	0.841	0.064	0.938	0.055	0.947	0.158	0.856
Mo	0.361	0.706	0.171	0.845	0.061	0.941	0.163	0.852	0.402	0.680	0.100	0.906	0.391	0.687
Na	0.236	0.795	0.403	0.680	0.767	0.493	0.493	0.626	0.199	0.823	0.313	0.739	0.603	0.568
P	3.424	0.078	0.549	0.596	0.633	0.553	0.074	0.929	0.124	0.885	0.900	0.440	0.605	0.567
Zn	1.833	0.215	0.624	0.558	1.982	0.193	1.200	0.345	0.719	0.513	0.910	0.436	0.622	0.558

Degrees of freedom: among groups, 2; between groups, 9.

Table C. Statistics of ANOVA-1 to compare the main properties of sediments from each erosion event with those of ashes (Dunnett's test). Only significant differences are shown, in bold when sediments > ashes and in regular type when sediments < ashes. Key: BS, control burnt plots; WM, wide-mulched burnt plots; NM, narrow-mulched burnt plots.

		sed. 1 month		sed. 3 months		sed. 4 months		sed. 5 months		sed. 7 months		sed. 12 months	
		F	sig.	F	sig.	F	sig.	F	sig.	F		F	sig.
	BS				0.010		0.000		0.000		0.000		0.008
pH_{KCl}	WM	0.42		5.40	0.023	14.10	0.001	20.06	0.000	20.33	0.000	6.26	0.012
	NM				0.026		0.001		0.000		0.000		0.013
	BS												
Total N	WM	2.65		0.61		0.51		1.70		0.35		0.28	
	NM												
	BS												
$\delta\ ^{15}N$	WM	3.69	0.018	0.87		1.49		3.54	0.030	0.75		0.31	
	NM												
	BS		**0.041**		0.001				**0.046**				
NH_4-N	WM	3.81		11.32	0.001	3.78	**0.015**	5.46	**0.012**	1.98		1.38	
	NM		**0.031**		0.004				**0.012**				
	BS		**0.018**				**0.010**		**0.003**				
NO_3-N	WM	3.94		0.82		6.31	**0.006**	6.04		1.37		2.22	
	NM												
	BS		**0.003**		**0.033**		**0.000**		**0.000**		**0.011**		
Al	WM	9.38	**0.017**	6.48	**0.023**	20.46	**0.000**	20.47	**0.000**	7.13	**0.043**	3.37	
	NM		**0.001**		**0.003**		**0.000**		**0.000**		**0.003**		**0.023**
	BS		**0.015**										
Ca	WM	5.67	**0.007**	0.74		0.03		0.46		0.59		0.16	
	NM												
	BS										0.000		0.000
Cu	WM	1.03		1.77		2.16		1.59		62.92	0.000	23.89	0.000
	NM										0.000		0.001
	BS		**0.001**		**0.021**		**0.000**		**0.000**		**0.000**		**0.000**
Fe	WM	9.93	**0.008**	5.98	**0.018**	24.66	**0.000**	23.22	**0.000**	11.67	**0.007**	29.43	**0.000**
	NM		**0.002**		**0.007**		**0.000**		**0.000**		**0.002**		**0.000**
	BS				0.008		0.001		0.001				
K	WM	3.00	**0.031**	4.84		9.22	0.013	9.01	0.006	0.16		0.90	
	NM						0.006		0.006				
	BS		**0.006**										
Mg	WM	8.38	**0.002**	1.77		0.18		0.14		0.53		0.11	
	NM		**0.013**										
	BS								0.058		0.041		
Mn	WM	0.97		0.23		2.76		3.17	0.050	4.13	0.025	1.86	
	NM										0.047		
	BS										0.000		0.000
Mo	WM	3.07	**0.038**	1.61		1.73		1.68		174.84	0.000	172.11	0.000
	NM										0.000		0.000
	BS						0.007		0.000		0.008		
Na	WM	0.23		1.96		5.17	0.064	14.50	0.001	6.65	0.019	0.53	
	NM						0.050		0.001		0.006		
	BS		0.032		0.000		0.000		0.000				
P	WM	7.64		42.63	0.000	22.92	0.000	25.28	0.000	1.37		2.31	
	NM		0.001		0.000		0.000		0.000				
	BS		**0.001**		**0.002**		**0.001**		**0.007**				
Zn	WM	15.44	**0.000**	8.24	**0.004**	8.00	**0.030**	4.81		0.85		0.80	
	NM		**0.003**		**0.020**		**0.037**		**0.084**				

Degrees of freedom: among groups. 3; between groups. 12.

Table D. Statistics of ANOVA-1 to compare the main properties of sediments from each erosion event with those of burnt soil just after the fire (Dunnett's test). Only significant differences are shown, in bold when sediments > soil and in regular type when sediments < soil. Key: BS, control burnt plots; WM, wide-mulched burnt plots; NM, narrow-mulched burnt plots.

		sed. 1 month		sed. 3 months		sed. 4 months		sed. 5 months		sed. 7 months		sed. 12 months	
		F	sig.	F	sig.	F	sig.	F	sig.	F		F	sig.
	BS		**0.001**		**0.001**		**0.001**		**0.001**		**0.001**		**0.041**
pH_{KCl}	WM	54.98	**0.001**	33.17	**0.001**	25.21	**0.001**	29.73	**0.001**	19.41	**0.001**	4.39	**0.028**
	NM		**0.001**		**0.001**		**0.001**		**0.001**		**0.001**		**0.025**
	BS		**0.004**		**0.047**				**0.007**		**0.012**		**0.012**
Total N	WM	8.53	**0.002**	5.67	**0.007**	2.56	**0.048**	7.10	**0.004**	4.80	**0.034**	5.73	**0.010**
	NM		**0.006**		**0.018**				**0.021**		**0.049**		**0.033**
	BS		0.001		0.001		0.001		0.001		0.001		0.008
$\delta\ ^{15}N$	WM	21.55	0.001	17.48	0.001	15.76	0.001	24.02	0.001	13.92	0.001	7.88	0.002
	NM		0.001		0.001		0.001		0.001		0.001		0.008
	BS		**0.005**		0.021		**0.011**		**0.001**				
NH_4-N	WM	7.94	**0.009**	3.96	0.045	9.46	**0.001**	16.96	**0.001**	4.07	**0.034**	2.18	
	NM		**0.003**				**0.008**		**0.001**		**0.024**		
	BS												
NO_3-N	WM	1.11		2.54		0.38		0.51		0.38		1.63	
	NM												
	BS		0.001		0.001		0.001		0.001		0.001		0.001
Al	WM	159.27	0.001	14.72	0.001	56.20	0.001	52.91	0.001	242.33	0.001	32.87	0.001
	NM		0.001		0.002		0.001		0.001		0.001		0.001
	BS		**0.001**		**0.001**		**0.001**		**0.001**		**0.001**		**0.002**
Ca	WM	24.26	**0.001**	16.39	**0.001**	17.61	**0.001**	28.07	**0.001**	24.37	**0.001**	9.08	**0.003**
	NM		**0.001**		**0.001**		**0.001**		**0.001**		**0.001**		**0.005**
	BS										0.001		0.001
Cu	WM	2.14		2.37		2.75		1.44		44.87	0.001	16.45	0.001
	NM										0.001		0.012
	BS		0.001				0.002		0.002		0.001		0.031
Fe	WM	18.07	0.001	2.24		10.94	0.001	12.05	0.001	22.83	0.001	7.30	0.002
	NM		0.001				0.002		0.001		0.001		0.030
	BS		**0.001**		**0.047**		**0.013**		**0.002**		**0.024**		**0.006**
K	WM	28.95	**0.001**	9.00	**0.004**	11.27	**0.001**	15.13	**0.001**	5.49	**0.011**	5.21	**0.050**
	NM		**0.001**		**0.001**		**0.001**		**0.001**		**0.017**		
	BS		**0.001**		**0.001**		**0.001**		**0.001**		**0.001**		**0.030**
Mg	WM	49.66	**0.001**	32.21	**0.001**	18.72	**0.001**	18.18	**0.001**	11.00	**0.001**	3.89	**0.037**
	NM		**0.001**		**0.001**		**0.001**		**0.001**		**0.003**		
	BS		**0.023**						**0.050**				
Mn	WM	4.27	**0.043**	2.91		1.84		4.08	**0.058**	2.53		1.62	
	NM		**0.038**		**0.050**				**0.021**				
	BS		**0.041**		**0.013**		**0.001**		**0.001**		0.001		0.001
Mo	WM	5.63	**0.007**	7.14	**0.004**	15.35	**0.001**	15.46	**0.001**	158.70	0.001	154.52	0.001
	NM		**0.018**		**0.007**		**0.001**		**0.001**		0.001		0.001
	BS		**0.002**		**0.002**						**0.007**		**0.032**
Na	WM	10.14	**0.001**	12.88	**0.001**	3.67	**0.033**	4.84	**0.020**	7.91	**0.003**	3.17	
	NM		**0.004**		**0.001**		**0.042**		**0.014**		**0.009**		
	BS												**0.049**
P	WM	3.97	**0.041**	1.37		0.54		0.08		1.14		2.61	
	NM												
	BS		**0.001**		**0.001**		**0.001**		**0.001**		**0.033**		**0.050**
Zn	WM	18.61	**0.001**	11.59	**0.001**	15.92	**0.001**	13.69	**0.002**	2.88		2.40	
	NM		**0.001**		**0.005**		**0.002**		**0.002**				

Degrees of freedom: among groups. 3; between groups. 12.