

# Reduction of nutrient losses with eroded sediments by post-fire soil stabilisation techniques

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**Abstract.** After an experimental fire in steep shrubland in a temperate-humid region (north-west Spain), the effects of two post-fire stabilisation treatments (grass seeding and straw mulching) on the chemical properties of eroded sediments, and the amount of nutrients lost with them, were evaluated relative to control burnt soil, over a period of 13 months. Total C and N concentrations, and  $\delta^{13}\text{C}$ , indicated that sediments were mainly contributed by charred plant and litter material. The highest concentrations of extractable base cations in the sediments occurred during the first 3 months following fire, especially for Na and K. As treatments had little or no effect on nutrient concentration in sediments, differences in nutrient losses were due to the 10-fold lower sediment production in mulching compared with other treatments. In control and seeding treatments, the accumulated amounts of nutrients lost with sediments were 989-1028 kg ha<sup>-1</sup> (C), 77 kg ha<sup>-1</sup> (N), 1.9-2.4 kg ha<sup>-1</sup> (Ca), 0.9-1.1 kg ha<sup>-1</sup> (Mg), 0.48-0.55 kg ha<sup>-1</sup> (NH<sub>4</sub>-N), 0.39-0.56 kg ha<sup>-1</sup> (K), 0.19-0.34 kg ha<sup>-1</sup> (Na) and <0.1 kg ha<sup>-1</sup> (P and NO<sub>3</sub>-N). These values accounted for 22-25% (total C and N) and 5-12% (NH<sub>4</sub>-N, Ca, P and Mg) of available nutrients in ash, and 1.0-2.4% of those in ash+topsoil. As nutrient and sediment losses were strongly correlated, the reduction of the latter by mulching application leads to an effective decrease of post-fire nutrient losses.

## Introduction

Wildfires destroy the vegetation cover and can greatly accelerate the processes of soil degradation (physical, chemical and biological) and nutrient losses through volatilisation, leaching and erosion (Certini 2005). Soil erosion rates typically increase after a wildfire due to reduced ground cover (Diaz-Fierros *et al.* 1987; Robichaud and Brown 1999) and changes in soil physical characteristics (Martin and Moody 2001; Cerda and Lasanta 2005). After fire, the highly available nutrients accumulated in the ash and the upper soil layer are vulnerable to losses by leaching, wind and runoff (DeBano *et al.* 1998). Despite universal recognition of erosion as a prevalent post-fire process, most studies have focussed on nutrient losses in solution, with less emphasis on the importance of nutrients attached to sediments and, therefore, little is known concerning nutrient losses associated with the eroded material (de Koff *et al.* 2006). Shortly after a fire, compared with soil surface, debris leaving a burnt area is enriched in organic matter, P, mineral N and exchangeable cations (Carreira and Niell 1995; Andreu *et al.* 1996; Thomas *et al.* 1999) and, consequently, nutrient losses with sediments can substantially exceed those lost in solution (DeBano and Conrad 1978; Gimeno-Garcia *et al.* 2000;

Smith *et al.* 2011). The increase in nutrients in eroded sediments has been related to fire intensity (Andreu *et al.* 1996; Gimeno-Garcia *et al.* 2000), but it was also observed after low intensity prescribed fires (de Koff *et al.* 2006).

Although post-fire erosion effects on downstream water quality and sedimentation (Smith *et al.* 2011) are recognised, several authors consider that nutrient losses in eroded sediments are unlikely to affect long-term site productivity (Helvey *et al.* 1985; Emmerich 1999; Robichaud *et al.* 2006) taking into account the small amounts of nutrients in sediments compared with those lost from burning, or accumulated in soils and the post-fire ash-layer (Grier 1975; Feller 1988; Means *et al.* 1992; Busse 1994; Baird *et al.* 1999). By contrast, other authors consider that soil quality might well be seriously affected through preferential loss of fines and organic matter, both of which contain more nutrients than does the remainder of the soil (Thomas *et al.* 1999; Shakesby *et al.* 2002). Moreover, the fate of eroded C (e.g. sequestration or oxidation) could affect the global C budget, and N and P delivery to waterbodies can lead to eutrophication (Gabet *et al.* 2005).

Post-fire erosion rates are highly variable (reported values range from 1 to 240 Mg ha<sup>-1</sup>; Robichaud *et al.* 2006; Moody and

Martin 2009) and are strongly related to the percentage of bare soil (Vega *et al.* 2005); they generally decrease by an order of magnitude with each year of recovery (Robichaud *et al.* 2000, 2006), leading to the greatest nutrient losses in the first 4-12 months after fire (Gimeno-Garcia *et al.* 2000; de Koff *et al.* 2006; Robichaud *et al.* 2006). Therefore, post-fire soil stabilisation techniques (e.g. seeding and mulching) must be applied as soon as possible to accelerate development of ground and vegetation cover, reduce surface runoff and prevent soil erosion (Pinaya *et al.* 2000; Robichaud *et al.* 2000; Vega *et al.* 2005; Wagenbrenner *et al.* 2006; Robichaud 2009; Fernandez *et al.* 2011, 2012). In many countries, seeding of fast-growing non-native grass species is the most common post-fire stabilisation technique to rapidly increase plant cover because of its relatively low cost and easy aircraft application (Robichaud *et al.* 2000; Beyers 2004). However, this practice has been questioned because it can introduce exotic species (Kruse *et al.* 2004) and displace native species, interfering with natural vegetation recovery (Beyers 2004). Furthermore, grass seeding could be ineffective in increasing ground cover or reducing erosion rates during the first critical rain events after a fire (Robichaud *et al.* 2006; Wagenbrenner *et al.* 2006; Groen and Woods 2008; Peppin *et al.* 2010; Fernandez *et al.* 2012). As an alternative, straw mulch is increasingly being applied to burnt areas with high value due to its immediate effectiveness in increasing ground cover and erosion control (Bautista *et al.* 2009; Robichaud *et al.* 2010). Previous studies have shown that mulching can reduce sediment production by 30-94% during the first critical months following fire (Wagenbrenner *et al.* 2006; Groen and Woods 2008; Fernandez *et al.* 2011; Diaz-Ravifia *et al.* 2012).

Despite its great importance, few studies have focussed on the effects of fire on soil nutrients removed in eroded soil material (DeBano *et al.* 1998; Gabet *et al.* 2005; Robichaud *et al.* 2006; Spigel and Robichaud 2007) and even fewer have dealt with the effectiveness of post-fire stabilisation techniques to reduce nutrient losses by erosion (Thomas *et al.* 2000; Robichaud *et al.* 2006). In this context, the aim of the present study was to evaluate, over a period of 13 months, the effects of two post-fire stabilisation techniques (seeding and mulching) on nutrient losses ((total carbon (C) and nitrogen (N), ammonium (NH<sub>4</sub><sup>+</sup>), nitrate (NO<sub>3</sub><sup>-</sup>), sodium (Na), potassium (K), magnesium (Mg), calcium (Ca) and phosphorous (P)) with the eroded sediments following an experimental fire in a shrubland ecosystem from the temperate-humid region of north-west Spain, an area with one of the highest fire incidence in Europe (Biro 2009). A complementary objective was to monitor the evolution of pH and the natural abundance of <sup>13</sup>C and <sup>15</sup>N in the eroded sediments.

## Material and methods

### Site description and experimental design

The experimental site was located in the Monte Cabalar (42°38'58"N; 8°29'31"W, 660 m above sea level) at A Estrada (Galicia, north-west Spain). The area has a temperate climate with 3036mm of accumulated precipitation during the study (October 2009-December 2010). The soil, developed over granite, is acidic with sandy loam texture and relatively high

organic matter content. Vegetation cover is dominated by gorse *Ulex europaeus*, with some *Pteridium aquilinum*, *U. gallii*, *Daboecia cantabrica* and *Pseudoarhenatherum longifolium*, with an average height of 123 cm and 100% of ground cover.

Before the experimental fire, there were no significant differences among plots assigned to the different treatments for any of the main characteristics, showing an acceptable spatial homogeneity of the study area for slope (38-54%), orientation (345-359° N-NW), vegetation cover (Vega *et al.* 2010), fuel loading (shrub stratum: 3.1±0.19kgm<sup>-2</sup>; litter layer: 1.7 ± 0.26 kg m<sup>-2</sup>; Fonturbel *et al.* 2012) and soil characteristics: pH<sub>KCl</sub> (3.5 ±0.1), total N (14.0 ± 0.9gkg<sup>-1</sup>), <sup>8</sup>15N (2.4 ±0.5‰), NH<sub>4</sub>-N (29±15mgkg<sup>-1</sup>), NO<sub>3</sub>-N (64± 15mgkg<sup>-1</sup>), total C (millgkg<sup>-1</sup>), <sup>8</sup>13C (-27.0±0.1‰), Na (61±7mgkg<sup>-1</sup>), K (132±25mgkg<sup>-1</sup>), Mg (39± 7mgkg<sup>-1</sup>), Ca (45 ± 7mgkg<sup>-1</sup>) and P (14±2mgkg<sup>-1</sup>). Twelve experimental plots (30 x 10 m each), with the longest dimension parallel to the maximum slope, were established within a total area of 1.73 ha. During June 2009, the shrub was cut and laid down directly on the ground, to facilitate litter and duff combustion, and 3 months later (14 and 15 October) the plots were burnt with the backfire technique to favour soil organic cover consumption and soil heating. The rate of fire spread was slow (0.30-0.33 mmin<sup>-1</sup>) and the soil temperature (monitored with 10 type K thermocouples per plot), reached 153°C below the mineral soil surface and 34°C at 2-cm soil depth.

To monitor post-fire erosion, the plots were delimited by a geotextile fabric similar to that described by Robichaud and Brown (2002), fixed to posts. Sediment fences made of this same fabric were established at the downhill side of each plot for periodic collection of sediments. After the experimental fire, three soil treatments were arranged in a fully randomised design with four replications: (a) burnt soil (BS), considered as control treatment; (b) mulching (M), burnt soil with 230 g vrT of straw mulch, which ensured a soil cover —60%; (c) seeding (S), burnt soil with a mixture of seeds at a rate of 45gm<sup>-2</sup> (*Lolium multiflorum*, 35%; *Trifolium repens*, 25%; *Dactylis glomerata*, 20%; *Festuca arundinacea*, 10%; *F. rubra*, 5%; *Agrostis tenuis*, 5%). The soil stabilisation treatments were applied on 6 November 2009 manually in order to minimise soil perturbation.

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

Immediately after the experimental fire, but before the application of soil stabilisation treatments, 'ash samples' (ashes+ charred plant and litter debris) were taken in 10 squares (15 x 15 cm) uniformly distributed inside each plot and combined to obtain a composite sample. The same was done for topsoils (0-5-cm depth) that were sampled after removing the plant litter in the case of soils before the experimental fire (US) or the ash layer of burnt soils (BS). The field-moist soil was sieved (<2 mm), homogenised and stored at 4°C for inorganic N determination. Sub-samples were air-dried for chemical analyses.

After each sediment-producing rain event, eroded sediments were collected and transported to the laboratory, where they were dried and weighed (see Fig. 1). A sub-sample was taken for chemical analysis.

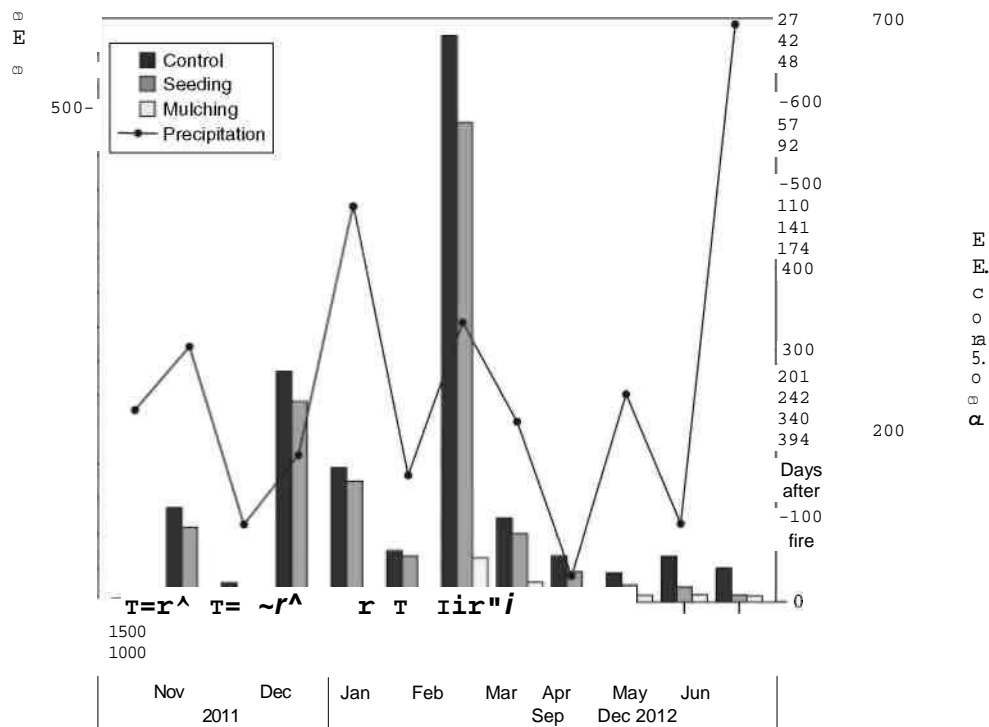


Fig. 1. Amount of sediments ( $\text{kg ha}^{-1}$ ) eroded during the first 13 months after the experimental fire in the control, seeding and mulching treatments and precipitation (mm) during each sediment-producing rain event.

The pH was measured with a pH meter (Metrohm, Switzerland) in  $\text{H}_2\text{O}$  and  $\text{KCl}$  employing a soil: solution ratio of 1:2.5 and a sediment or ash: solution ratio of 1: 6. Total C, total N,  $\delta^{13}\text{C}$  and  $\delta^{15}\text{N}$  were measured on finely ground soil, sediments and ash samples ( $< 100 \mu\text{m}$ ) with an elemental analyser (Carlo Erba CNS 1508, Milan, Italy) coupled online with an isotopic ratio mass spectrometer (Finnigan Mat, delta C, Bremen, Germany). Inorganic N was extracted by shaking 5 g (ash) or 20 g (soil or sediment) of sample for 1 h with 2 M  $\text{KCl}$  (1:10 ash: solution ratio; 1: 5 soils or sediments: solution ratio). Extracts were passed through glass microfibre filters (Whatman GF/A, 125-mm diameter) and  $\text{NO}^-\text{N}$  and  $\text{NH}_4^+\text{N}$  were analysed by a modified diffusion method (Khan *et al.* 1997). In order to sequentially liberate  $\text{NH}_4^+\text{N}$  and  $\text{NO}_2^- + \text{NO}_3^-\text{N}$  in the form of  $\text{NH}_3$ , 20-mL aliquots of extracts, placed in a 500-mL wide-mouth glass jar, were treated successively (24-h periods at  $50^\circ\text{C}$ ) with  $\text{MgO}$  (0.1 g) and  $\text{MgO}$  (0.1 g) plus Devarda's alloy (0.4 g). Gaseous  $\text{NH}_3$  was trapped into 10 mL of 4mM  $\text{H}_2\text{SO}_4$  in a Teflon bottle attached to the glass jar and measured by back titration of the excess of  $\text{H}_2\text{SO}_4$  with 0.004 M  $\text{NaOH}$ . For extractable Na, K, Mg, Ca and P analyses, soil, ash or sediments (10 g) were shaken for 2h with a mixture of 1 M  $\text{NH}_4\text{Ac}$  and 0.005 M DTPA as extracting solution (1:5 soil, ash or sediment: solution ratio); the extracts were filtered through cellulose paper and then analysed by simultaneous ICP-OES (Varian Vista Pro, Melbourne). All analyses were carried out in duplicate and the mean of both analyses was used in the statistical procedure.

### Statistical analysis

Data on concentration and mass of nutrients in sediments from each erosion event were analysed by two-way ANOVA (with treatment and days after fire as factors), whereas the effects of soil stabilisation treatments on the accumulated mass of eroded sediments and nutrients were analysed by one-way ANOVA. After checking the equality of variances among groups with Levene's test, significant differences among the mean groups were established at  $P < 0.05$  using the Bonferroni's test for multiple comparisons (ANOVA-2) or the Tukey's test (ANOVA-1). With unequal variances, the original data were subjected to Cox-Box transformations to obtain homoscedasticity and then significant differences among groups were assessed as previously explained. Simple regression models including soil properties (dependent variables) and the inverse of the accumulated precipitation (independent variable) were explored after checking the fulfilment of independence, homoscedasticity and normality assumptions. The best models were selected maximising the adjusted minimising the standard error of the estimated residues. Statistical procedures were performed with SPSS 15.0.

### Results

#### Main characteristics of sediments

The two-way ANOVA for the pH of sediments, in  $\text{H}_2\text{O}$  and  $\text{KCl}$ , showed that 72-73% of their variation was explained by time since fire, and the emergency stabilisation treatments have a

**Table 1. Results of the two-way ANOVA for the main sediment characteristics (pH,  $^{13}\text{C}$ ,  $^{15}\text{N}$  and nutrient concentrations) with treatment (T) and days after fire (D) as factors**  
Probabilities are significant at \*,  $P < 0.05$ ; \*\*,  $P < 0.01$ ; \*\*\*,  $P < 0.001$

Variable (TxD)	Treatment	Days after fire	Interaction
	partial $r^2$	partial $r^2$	partial $r^2$
pHH <sub>2</sub> O	0.058*	0.732***	
pHKCl	0.057*	0.717***	
Total C (g kg <sup>-1</sup> )	0.071*		
$^{13}\text{C}$ (‰)	0.089**	0.200*	
Total N (g kg <sup>-1</sup> )	0.110**		
$^{15}\text{N}$ (‰)		0.810***	
NH <sub>4</sub> <sup>+</sup> -N (mg kg <sup>-1</sup> )	0.091**	0.521***	
NO <sub>3</sub> <sup>-</sup> -N (mg kg <sup>-1</sup> )	0.136**	0.248**	
Extractable Na (mg kg <sup>-1</sup> )		0.861***	
Extractable K (mg kg <sup>-1</sup> )		0.885***	
Extractable Mg (mg kg <sup>-1</sup> )		0.594***	
Extractable Ca (mg kg <sup>-1</sup> )		0.676***	
Extractable P (mg kg <sup>-1</sup> )		0.592***	

**Table 2. Significant regression models ( $P < 0.0005$ ) between sediment properties as dependent variables (pH,  $^{15}\text{N}$ , ‰, extractable nutrients, mg kg<sup>-1</sup>) and the accumulated precipitation (x, mm) or its inverse ( $x^{-1}$ ) as the independent variable**

Variable	$R^2$	Equation
pHH <sub>2</sub> O	0.52	$y = 4.568 + (271.1x^{-1})$
pHKCl	0.75	$y = 3.585 + (312.4x^{-1})$
$^{15}\text{N}$	0.36	$y = 1.385 - (0.449 \times 10^{-3} x) + (3.529 \times 10^{-7} x^2)$
Extractable Na	0.74	$y = 32.73 + (38500x^{-1})$
Extractable K	0.71	$y = 47.03 + (130054x^{-1})$
Extractable Mg	0.58	$y = 683.8 - (0.219x)$
	0.45	$y = 189.9 + (144666x^{-1})$
Extractable Ca	0.56	$y = 1500 - (0.458x)$
	0.58	$y = 416.4 + (348329x^{-1})$

small but significant effect (<6% of variance explained: Table 1). The evolution of sediment pH during the study fitted to an inverse model with the accumulated precipitation as the independent variable, which explained 51 and 75% (pH in H<sub>2</sub>O and KCl respectively) of the variance (Table 2). The pH of sediments, initially closer to that of ashes than to the burnt soil, decreased 1.2 units during the study, but after 13 months it remained slightly higher than that of the 0-5-cm soil layer of the burnt plots just after the fire (Fig. 2a).

The C concentration of the sediments varied without a clear trend during the first 200 days of the study, with values close to that of ashes and 2-fold higher than those of unburnt and burnt topsoils (Fig. 2b). Despite a slight decrease in the last 6 months, C values in the sediments were still 70% higher than those in the topsoil at 13 months after the experimental fire. The two-way ANOVA showed no effect of date and a small, but significant, effect of treatment on sediment C (7.1% of variance explained; Table 1), with the highest values in S (320±49gkg<sup>-1</sup>), the lowest in M (285 ±39gkg<sup>-1</sup>) and intermediate in BS plots (304 ± 70gkg<sup>-1</sup>). In all sediments,

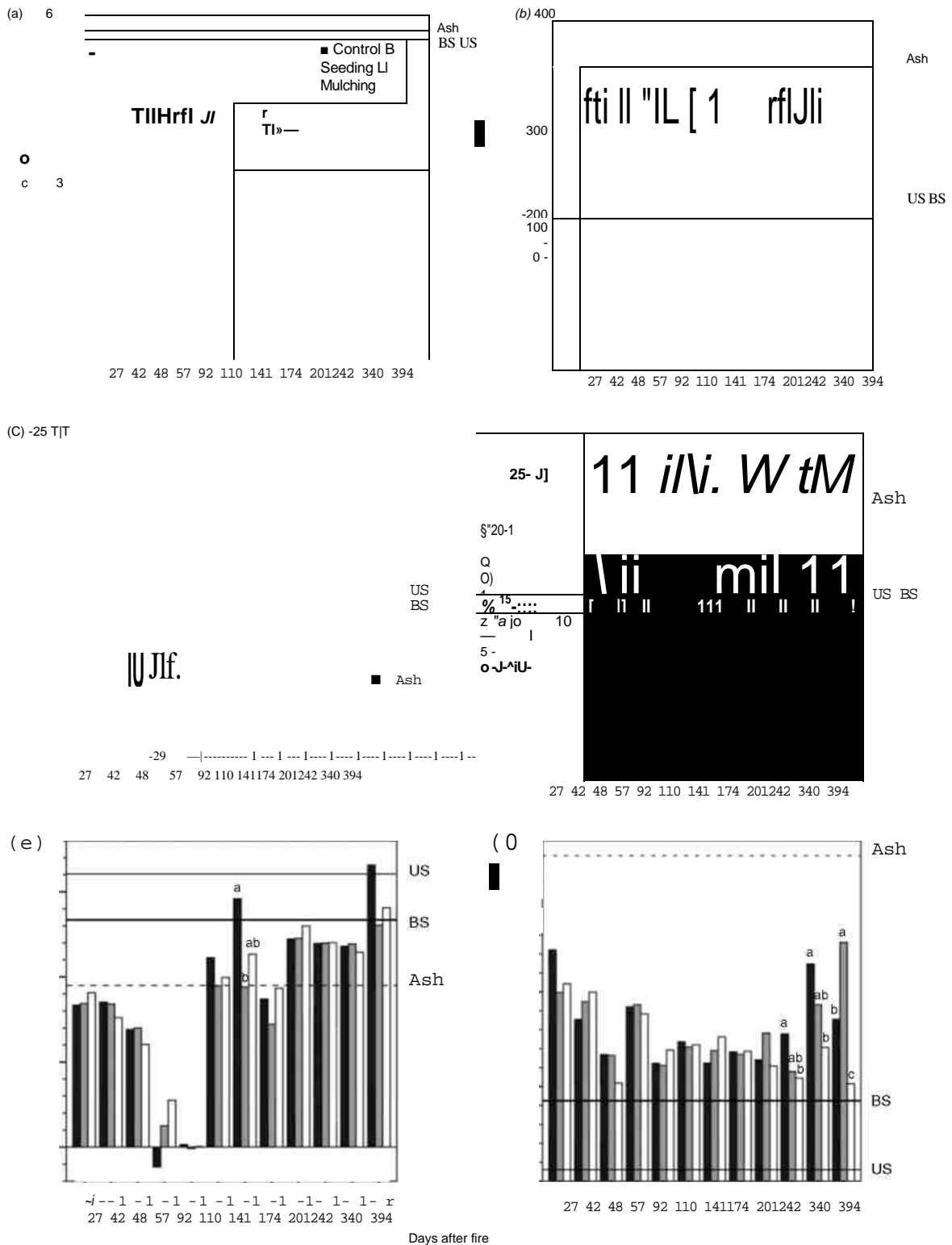
except the last collected, the  $^{13}\text{C}$  was approximately that of ashes, with values ~1‰ more negative than in the 0-5-cm layer of unburnt and burnt soil (Fig. 2c). Treatment and days after fire had significant effects on sediment  $^{13}\text{C}$ , respectively explaining 8.9 and 20.0% of its variance (Table 1). The highest  $^{13}\text{C}$  values were found in M (-27.9 ± 0.4‰) and the lowest in S plots (-28.1 ± 0.3‰), with BS plots (-27.9 ± 0.3‰) in an intermediate position; only at the end of the study the sediments  $^{13}\text{C}$  became less negative, approaching values for burnt and unburnt topsoils.

Only small fluctuations were detected in sediment N concentration during the whole study, most values being similar to those of ashes and 50-70% higher than those of burnt and unburnt topsoils (0-5 cm) (Fig. 2d). As also happens with C values, the two-way ANOVA showed no effect of time after fire and a small, but significant, effect of treatment on sediment N concentration (11% of variance explained; Table 1), with the highest values in S (23.9 ± 3.2 gkg<sup>-1</sup>), the lowest in M (21.1 ± 3.1 gkg<sup>-1</sup>) and intermediate in BS plots (22.7 ± 4.8 g kg<sup>-1</sup>). Conversely, the  $^{15}\text{N}$  of sediments was not affected by the emergency stabilisation treatments considered, but it was largely influenced by days after fire (81% of variance explained; Table 1), rising ~ 1 ‰ from values similar to those of ashes in the first three samplings until values progressively closer to those of the burnt soil in the second half of the study, with an intermediate and sudden decrease in months two to three (Fig. 2e). Such reduction in late autumn-early winter, not related to the amount of eroded sediments, seemed to be influenced by the high rainfall in this period; without runoff analyses it is not possible to verify if this reduction in sediments  $^{15}\text{N}$  is related with changes in dissolved organic matter quantity or quality. For the whole study, the evolution of sediment  $^{15}\text{N}$  fitted to a quadratic model with accumulated precipitation as the independent variable, explaining 36% of the variance (Table 2). Except at  $t = 141$  days, when BS > S, there were no significant differences for  $^{15}\text{N}$  among treatments.

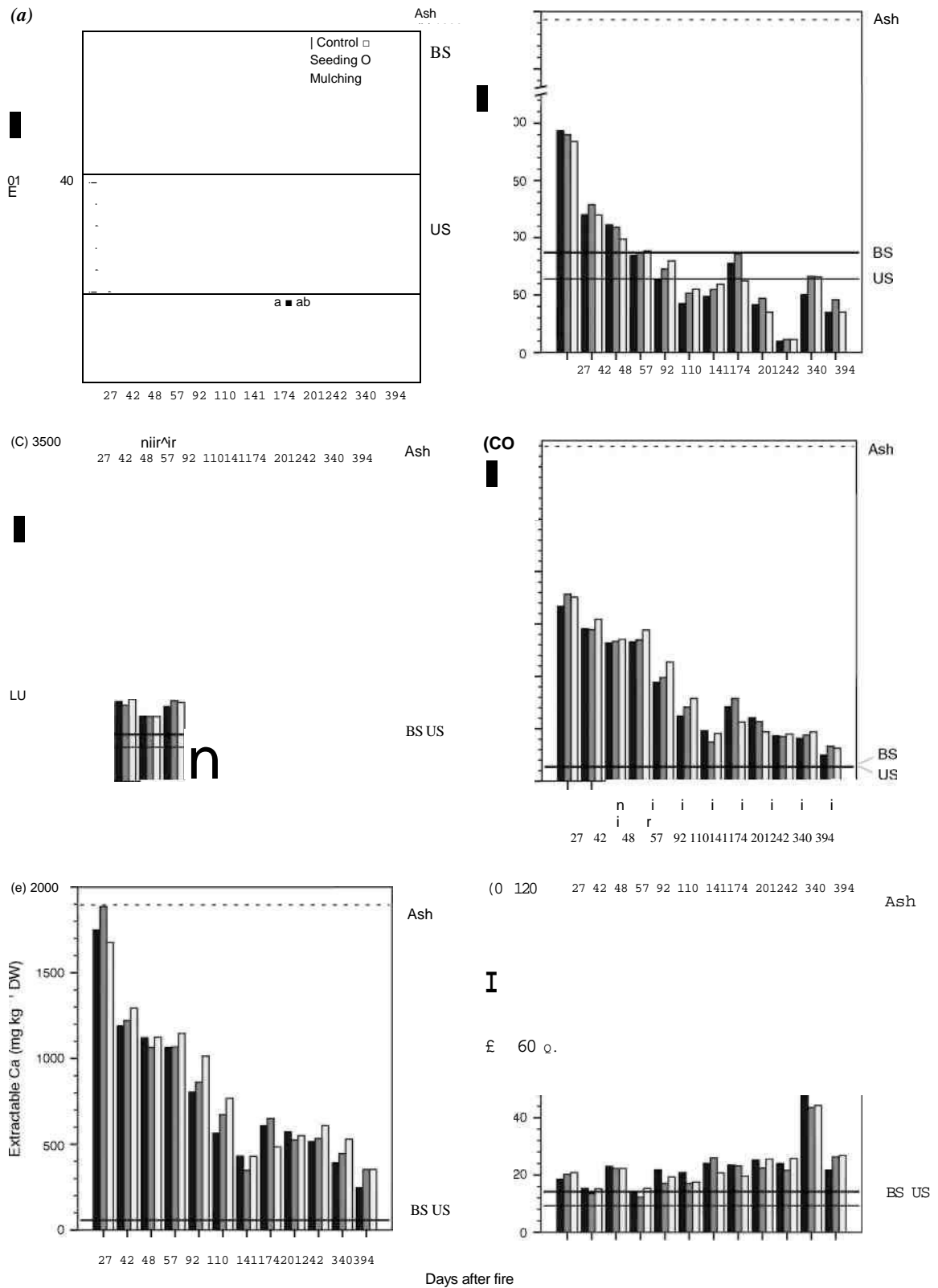
The NFL<sup>-</sup>-N was the dominant form of inorganic N in sediments (>96%) and during most of the study its concentration (100-250mgkg<sup>-1</sup>) largely exceeded that in the unburnt topsoil (12mgkg<sup>-1</sup>), being closer to the burnt topsoil (85mgkg<sup>-1</sup>) than to ashes (344mgkg<sup>-1</sup>). The two-way ANOVA showed significant effects of treatment, time after fire and their interaction (with respectively 9.1, 52.1 and 36.9% of variance explained), with significant differences among treatments (lowest values in M) only in the last three erosion events (Fig. 2f).

Nitrate concentration in sediments was low and always lower than in unburnt and burnt topsoils and ashes (Fig. 3a). Both treatment and days after fire had a significant effect on NO<sub>3</sub><sup>-</sup>-N concentration (13.6 and 24.8% of variance explained, Table 1), with the highest values in control (8.6±4.0mgkg<sup>-1</sup>), the lowest in M (6.2 ± 3.3 mgkg<sup>-1</sup>) and intermediate in S plots (7.7±3.9mgkg<sup>-1</sup>).

The most abundant extractable base cation in the sediments was Ca, followed by Mg and K. Extractable base cation concentrations were affected neither by the emergency stabilisation treatment nor by the treatment x days after fire interaction, but they were largely influenced by the time since fire (59-89% of variance explained; Table 1). During the first samplings,



**Fig. 2.** Evolution of main characteristics of the sediments eroded during the first 13 months after the experimental fire in the control, seeding and mulching treatments: (a) pH<sub>rcb</sub> (& total C concentration); (c) 8 C; (d) total N concentration; (e) 8 N and (f) NH<sub>4</sub>-N concentration. Horizontal lines show the reference values in ash, unburnt soil (US) and burnt soil (BS) at the beginning of the study (i.e. after the experimental fire and before the application of soil stabilisation treatments). For a given date, different letters (a, b, c,...) indicate statistically significant differences ( $P < 0.05$ ) among treatments.



**Fig. 3.** Evolution of nutrient concentrations in the sediments eroded during the first 13 months after the experimental fire in the control, seeding and mulching treatments: (a) NO<sup>3</sup>-N; (b) extractable Na; (c) extractable K; (d) extractable Mg; (e) extractable Ca and (f) extractable P. Horizontal lines show the reference values in ash, unburnt soil (US) and burnt soil (BS) at the beginning of the study (i.e. after the experimental fire and before the application of soil stabilisation treatments). For a given date, different letters (a, b, c,...) indicate statistically significant differences ( $P < 0.05$ ) among treatments.

extractable Na and K concentrations in sediments were above those in burnt topsoils but, from 92 days after the experimental fire, most of the values were below those of the topsoils (Fig. 3b, c). The initial extractable Mg concentration in sediments was half of that in ashes but 13-fold that in topsoils and, despite a steady decline with time, at the end of the study it was twice that in the topsoil (Fig. 3d). Similar temporal evolution was observed for extractable Ca concentration in sediments, from initial concentrations similar to those of ashes (35 times that of topsoil) until values at the end of the study that were still 5.5 times higher than in the topsoil (Fig. 3e). Curvilinear inverse regression models, explaining 56-74% of the variance, were found for extractable Na, K, Mg and Ca in sediments with the accumulated precipitation as the independent variable (Table 2). For extractable Mg and Ca, linear regression models were also found with the accumulated precipitation, which explain approximately half of the variance (Table 2). Contents of extractable P in sediments were low (six times less than in ashes) and showed little variation during the study, except for a transient increase 340 days after the fire (i.e. in

autumn, a temperate humid period in the study area), most values being slightly higher than in burnt topsoils (Fig. 3f). Time since fire, but not treatment, influenced the sediment extractable P concentrations (59% of variance explained: Table 1).

#### Mass of nutrients lost with sediments

The sediment mass was largely influenced by the treatment and time after fire (81 and 88% of respective variance explained), but the interaction of these variables, which explained 61% of the variance, showed a stronger effect for B S and S plots than for M, whose values were more stable during the study period (Table 3). Very similar results were obtained for the amount of extractable Na and K lost with sediments (Table 3). The same was true for the other nutrients, although the interaction between factors was less important (29-44% of variance explained); for total N the treatment effect was also lower (53% of variance explained; Table 3).

Compared with the BS control, at the end of the study period the accumulated mass of sediments (BS,  $3897 \pm 1012 \text{ kg ha}^{-1}$ ; S,  $3117 \pm 612 \text{ kg ha}^{-1}$ ; M,  $381 \pm 194 \text{ kg ha}^{-1}$ ) and extractable nutrients lost with them (Table 4) were only 9-15% in M and 66-88% (but 96-100% for total C and N) in S; differences between M and BS or S were significant at  $P < 0.01$  ( $P < 0.05$  for NOF-N).

As shown in Table 4, the extractable nutrients lost by erosion accounted for 0.14-2.2% of ash extractable nutrients in M, significantly less than the 0.88-25.2% and 1.25-25.2% respectively found in S and BS; in all treatments, the percentages were high for total C and N, intermediate for NH<sub>4</sub>-N, Ca, P and Mg, and low for NO<sup>-</sup>-N, Na and K. If extractable nutrients in ash+topsoil was taken as a reference, nutrient losses by erosion were negligible in M (from 0.01% to 0.39%); the same was true for NO<sup>-</sup>-N, Na and K in S and BS plots (from 0.13% to 0.44%), whereas values were ~ 1% for total C, total N, NFF<sub>4</sub>-N and P, and 2-4 times higher for Mg and Ca.

**Table 3. Results of the two-way ANOVA for the mass of sediments and nutrients with treatment (T) and days after fire (D) as factors**

Probabilities are significant at \*,  $P < 0.05$ ; \*\*,  $P < 0.01$ ; \*\*\*,  $P < 0.001$

	Treatment	Days after fire	Interaction (T x D)
	partial $r^2$	partial $r^2$	partial $r^2$
Sediments (kg ha <sup>-1</sup> )	0.808***	0.882***	0.607***
Total C (kg ha <sup>-1</sup> )	0.802***	0.830***	0.44***
Total N (kg ha <sup>-1</sup> )	0.534***	0.793***	0.330*
NH <sub>4</sub> -N (g ha <sup>-1</sup> )	0.756***	0.786***	0.423***
NO <sup>-</sup> -N (g ha <sup>-1</sup> )	0.695***	0.742***	0.355***
Extractable Na (g ha <sup>-1</sup> )	0.732***	0.829***	0.560***
Extractable K (g ha <sup>-1</sup> )	0.734***	0.817***	0.565***
Extractable Mg (g ha <sup>-1</sup> )	0.681***	0.768***	0.289**
Extractable Ca (g ha <sup>-1</sup> )	0.696***	0.749***	0.314***
Extractable P (g ha <sup>-1</sup> )	0.720***	0.774***	0.389***

**Table 4. Accumulated mass of sediments and nutrients lost by erosion in the plots under each treatment (C, control; S, seeding; M, mulching) and percentage of nutrients in ash and ash+topsoil they accounted for**

In each row, different letters showed statistically significant differences between treatments for each variable: (i) mass of sediments or nutrients; (ii) percentage of ash nutrients lost by erosion and (iii) percentage of ash+topsoil nutrients lost by erosion

Variable	Mass of sediments and nutrients lost by erosion			Ash nutrients lost by erosion			Ash+topsoil nutrients by erosion		
	C	S (kg ha <sup>-1</sup> )	M	C	S (%)	M	C	S (%)	M
Sediments	3897 <sup>a</sup>	3117 <sup>b</sup>	381 <sup>c</sup>						
Total C	1028 <sup>a</sup>	989 <sup>a</sup>	100 <sup>b</sup>	22.90 <sup>a</sup>	22.0 <sup>a</sup>	2.23 <sup>b</sup>	1.15 <sup>c</sup>	1.11 <sup>c</sup>	0.11 <sup>d</sup>
Total N	76.8 <sup>a</sup>	77.0 <sup>a</sup>	7.67 <sup>b</sup>	25.2 <sup>a</sup>	25.2 <sup>a</sup>	2.51 <sup>b</sup>	1.04 <sup>c</sup>	1.04 <sup>c</sup>	0.10 <sup>d</sup>
NH <sub>4</sub> -N	548 <sup>a</sup>	476 <sup>a</sup>	53.5 <sup>b</sup>	12.3 <sup>a</sup>	10.7 <sup>a</sup>	1.20 <sup>b</sup>	1.16 <sup>c</sup>	1.01 <sup>c</sup>	0.11 <sup>d</sup>
NO <sup>-</sup> -N	37.1 <sup>a</sup>	32.6 <sup>a</sup>	2.92 <sup>b</sup>	3.66 <sup>a</sup>	3.21 <sup>a</sup>	0.29 <sup>b</sup>	0.15 <sup>c</sup>	0.13 <sup>c</sup>	0.01 <sup>d</sup>
Extractable Na	232 <sup>a</sup>	194 <sup>a</sup>	21.5 <sup>b</sup>	1.53 <sup>a</sup>	1.28 <sup>a</sup>	0.14 <sup>b</sup>	0.39 <sup>c</sup>	0.33 <sup>c</sup>	0.04 <sup>d</sup>
Extractable K	557 <sup>a</sup>	393 <sup>a</sup>	51.2 <sup>b</sup>	1.25 <sup>a</sup>	0.88 <sup>a</sup>	0.12 <sup>b</sup>	0.44 <sup>c</sup>	0.31 <sup>c</sup>	0.04 <sup>d</sup>
Extractable Mg	1101 <sup>a</sup>	861 <sup>a</sup>	94.6 <sup>b</sup>	6.68 <sup>a</sup>	5.22 <sup>a</sup>	0.57 <sup>b</sup>	2.50 <sup>c</sup>	1.95 <sup>c</sup>	0.21 <sup>d</sup>
Extractable Ca	2387 <sup>a</sup>	1900 <sup>a</sup>	211 <sup>b</sup>	9.76 <sup>a</sup>	7.77 <sup>a</sup>	0.86 <sup>b</sup>	4.41 <sup>c</sup>	3.51 <sup>c</sup>	0.39 <sup>d</sup>
Extractable P	89.4 <sup>a</sup>	58.8 <sup>a</sup>	8.82 <sup>b</sup>	7.44 <sup>a</sup>	4.89 <sup>a</sup>	0.73 <sup>b</sup>	1.08 <sup>c</sup>	0.71 <sup>c</sup>	0.11 <sup>d</sup>

## Discussion

Considering the strong relationship between pH and accumulated precipitation, the progressive acidification of sediments during the study (0.5 units higher than that reported by Robichaud *et al.* 2006) showed that the oxides and carbonates of basic ions supplied by ashes (Certini 2005), as well as soil exchangeable cations, were steadily lost by leaching and runoff. A minor effect of emergency stabilisation treatments on sediment pH coincides with results of Robichaud *et al.* (2006).

The C concentration in sediments, which exceed the range of 110-244 gkg<sup>-1</sup> reported by Thomas *et al.* (1999), was higher than in burnt topsoils and close to that in ashes. This result suggests that, during most of the study period, sediments were mainly contributed by charred plant and litter material, agreeing with de Koff *et al.* (2006) who indicated that C concentration increased immediately after a low severity prescribed fire, probably because of C-rich ashes in sediments. Stabilisation treatments had small effects on sediment C concentration, with the lowest values in mulching plots. Except for with mulching, the accumulated amount of C in the sediments was 5-fold those after high and moderate intensity experimental fires in eastern Spain (Gimeno-Garcia *et al.* 2000) and it slightly exceeded the highest value (900kgha<sup>-1</sup>) observed the first year after high severity wildfires (Baird *et al.* 1999; Spigel and Robichaud 2007). This result suggested that, whereas low intensity prescribed fires reduce nutrient losses by volatilisation relative to wildfires, they expose available nutrients to erosion (de Koff *et al.* 2006). In seeding and control plots, the eroded C during the study accounted for 25% of the ash C and 1% of the ash+topsoil C pool, these values being 10-fold lower in mulching plots. Baird *et al.* (1999) also estimated that 1% of C from the A-horizon was lost through post-fire erosion. Data on  $\delta^{13}\text{C}$  in sediments, similar to that of ashes and different from those of unburnt and burnt topsoils, also suggested that our sediments were mainly charred plant and litter material.

Nitrogen concentration in sediments (similar to that in ashes and higher than in burnt and unburnt topsoils) was at least twice that reported by Thomas *et al.* (1999) for eucalypt and pine plantations, likely due to differences in vegetation cover in the present study (leguminous shrubland). As for C, stabilisation treatments had little effect on sediment N concentration, as also indicated Thomas *et al.* (2000) for a litter treatment, lowest values being observed with mulching. Accumulated losses of N in seeding and control plots largely exceeded the usual range in burnt soils (14-57kggha<sup>-1</sup> year<sup>-1</sup>; Debono and Conrad 1978; Helvey *et al.* 1985; Soto *et al.* 1997; Baird *et al.* 1999; Thomas *et al.* 1999; Gimeno-Garcia *et al.* 2000; Lane *et al.* 2008), being only below the highest values reported for high severity wildfires (100-129kg ha<sup>-1</sup>; SotoefaZ. 1997; Spigel and Robichaud 2007); with mulching the mass of total N eroded was only half of the lowest values. Because there was little treatment effect on sediment N concentration, the mulching effectiveness in reducing the amount of N lost was mainly due to the reduction in soil erosion as reported by Soto *et al.* (1997) and Spigel and Robichaud (2007). As for C, the eroded N during the first year accounted for 25% of ash N and 1% of ash+topsoil N pool in seeding and control plots, but 10-fold less in mulching plots; the latter figure for seeding and control plots was below the 2%

reported for topsoils (0-10 cm) after high severity wildfires (Baird *et al.* 1999; Thomas *et al.* 1999). By contrast with data on C and N concentrations and  $\delta^{13}\text{C}$ , the sediment  $\delta^{15}\text{N}$  was strongly influenced by days after fire, increasing from initial values similar to ashes to final values close to the burnt topsoil following a quadratic model with the accumulated precipitation as the independent variable. These results suggest that, over time, sediment  $\delta^{15}\text{N}$  was affected by losses of soluble N (likely NO<sub>3</sub><sup>-</sup>) relatively depleted in <sup>15</sup>N.

The NH<sub>4</sub><sup>+</sup>-N accounted for most of the inorganic N in burnt topsoil, ashes and sediments, agreeing with the results of Gimeno-Garcia *et al.* (2000), but not with those of Robichaud and Brown (1999) and Robichaud *et al.* (2006). As also reported by Andreu *et al.* (1996) for mineral N, sediments were enriched in NH<sub>4</sub><sup>+</sup>-N relative to burnt (1.2-3 times) and unburnt (12-23 times) topsoil. Differences in NH<sub>4</sub><sup>+</sup>-N concentration among treatments (lowest values for mulching) were only observed in the second half of our study and, despite the significant effect of time after fire, no clear temporal trend was identified. During the first year, accumulated losses of NH<sub>4</sub><sup>+</sup>-N by erosion with mulching were within the usual range (50-140 mgkg<sup>-1</sup>; Robichaud and Brown 1999; Gimeno-Garcia *et al.* 2000; Robichaud *et al.* 2006), whereas values in control and seeding plots exceeded 3-4 times that range. Although 14% lower, total NH<sub>4</sub><sup>+</sup>-N losses with seeding did not differ significantly from those in control plots; Robichaud *et al.* (2006) also found no effects of seeding.

The low NO<sub>3</sub><sup>-</sup>-N concentration in the sediments contrasted with the results of Robichaud and Brown (1999) and Robichaud *et al.* (2006), but coincided with those of Gimeno-Garcia *et al.* (2000) and was an expected result: the runoff water that triggers erosion likely washed the sediments in the trap, carrying off the highly soluble NO<sub>3</sub><sup>-</sup>-N. The lack of significant effects of seeding and of a discernible trend with time for nitrate concentration in sediments agreed with the findings of Robichaud *et al.* (2006). Compared with other published data (50-810kggha<sup>-1</sup>year<sup>-1</sup>; Robichaud and Brown 1999; Gimeno-Garcia *et al.* 2000; Robichaud *et al.* 2006), the accumulated amount of eroded NO<sub>3</sub><sup>-</sup>-N in control and seeding plots were 2-20 times lower (15-250 fold lower with seeding); this result can be explained by the wide differences among study areas for annual precipitation, which was 2-7 times higher in our case (3036mm v. 400-1600 mm) and likely washed the NO<sub>3</sub><sup>-</sup>-N of sediments retained in the sediment fences.

The concentration of soluble base cations in the sediments was lowest for Na, intermediate for K and Mg, and highest for Ca, following the typical order of magnitude (de Koff *et al.* 2006) and reflecting the high solubility of Na and K relative to Ca and Mg, in ashes (Khanna *et al.* 1994).

In the first sediments collected, the extractable Na concentration was three times higher than in the unburnt topsoil, agreeing with findings of de Koff *et al.* (2006). Likely due to its high solubility, the extractable Na concentration in sediments decreased rapidly as a function of accumulated precipitation and from 3 months after fire it fell below that in recently burnt topsoil. The accumulated amount of extractable Na lost with sediments in control and seeding plots was higher than that reported by Gimeno-Garcia *et al.* (2000), but 15 times lower



than that observed by Deban and Conrad (1978) after a prescribed fire. Over the first year, particulate losses of extractable Na were less than 2% of the ash Na and 0.4% of the ash+topsoil Na extractable pool in seeding and control, and negligible (ash Na, 0.14%; ash+topsoil Na, 0.01%) in mulching plots.

Although the extractable K concentration in ashes was 10-fold more than that reported by Thomas *et al.* (1999), in the first sediments it was in the upper part of the range (200-520 mg kg<sup>-1</sup>) found by Robichaud and Brown (1999) and Thomas *et al.* (1999), but below the 1619 mg kg<sup>-1</sup> observed by de Koff *et al.* (2006) immediately after fire. Like Na, the highly soluble extractable K in sediments decreased rapidly as a function of accumulated precipitation during the first 3 months after fire. One year after fire, the accumulated amount of extractable K lost with sediments was medium to low compared with the usual range (0.3-2.9 kg ha<sup>-1</sup>: Robichaud and Brown 1999; Thomas *et al.* 1999; Gimeno-Garcia *et al.* 2000; Robichaud *et al.* 2006), but far from the highest losses that have been reported (19-30 kg ha<sup>-1</sup>: Deban and Conrad 1978; Spigel and Robichaud 2007). Agreeing with Thomas *et al.* (2000) and Robichaud *et al.* (2006) respectively, mulching and seeding had little effect on sediment K concentrations; consequently, mulching reduced K losses with sediments by reducing soil erosion, as found by Thomas *et al.* (2000). In the study of Thomas *et al.* (1999), the particulate losses of K with sediments during the first year accounted for 11.0 and 3.7% of total K (in the 0-3- and 0-10-cm soil layers respectively), whereas in the present study they represented only 1 % of K in ashes and less than 0.5% of that in ashes+topsoil, being negligible in mulching (<0.12%).

The extractable Mg concentration in the first sediments was 13-fold higher than that in burnt or unburnt topsoils; this enrichment ratio decreased to 10 in the three subsequent erosion events, still well above that (4-fold) obtained by de Koff *et al.* (2006) for sediments before and immediately after fire. Unlike for monovalent cations, the progressive decrease of Mg concentration in sediments can be adjusted to curvilinear inverse and linear regression models with the accumulated precipitation, and sediment Mg concentration 1 year after fire was still twice that in topsoils, disagreeing with de Koff *et al.* (2006) who found that exchangeable base cations resumed pre-fire levels 7 months after fire. Accumulated losses of extractable Mg in control and seeding treatments were within the range (0.8-1.4 kg ha<sup>-1</sup> year<sup>-1</sup>) observed by Gimeno-Garcia *et al.* (2000), but below the highest values reported (10-28 kg ha<sup>-1</sup> year<sup>-1</sup>: Spigel and Robichaud 2007; Deban and Conrad 1978). Magnesium losses with sediments from control and seeding plots accounted for 5.2-6.7% and 2-2.5% of the respective extractable Mg pool in ashes and ashes+topsoil, whereas in mulching plots they were 10-fold lower.

In the sediments eroded the first month after fire, irrespective of treatment, the extractable Ca concentration was 30-fold higher than that in burnt or unburnt topsoils; during the following 2 months this enrichment ratio decreased to 20, still much higher than that indicated by de Koff *et al.* (2006) for sediments before and immediately after fire. Like Mg, the steady decline with time of sediment Ca concentration fitted to curvilinear inverse and linear regression models with the accumulated precipitation and after 1 year it was still five times higher than

that in topsoils, showing a much longer-lasting effect of the prescribed fire than that reported by de Koff *et al.* (2006). The accumulated amounts of extractable Ca lost with sediments in control and seeding plots were at the bottom of the published range (2-200 kg ha<sup>-1</sup> year<sup>-1</sup>: Deban and Conrad 1978; Gimeno-Garcia *et al.* 2000; Spigel and Robichaud 2007). Nevertheless, the accumulated Ca losses had a high relative importance in our study, accounting for 7.8-9.8% and 3.5-1.4% of the respective extractable Ca pool in ashes and ashes+topsoil; as with other nutrients, Ca losses with mulching were 10-fold lower.

Except for one of the last samplings, the extractable P concentration in sediments was approximately half the lowest value (50-300 mg kg<sup>-1</sup>) observed by Thomas *et al.* (1999) and it did not decrease over time, contrary to other studies (Thomas *et al.* 1999; Robichaud *et al.* 2006). As also reported by Saa *et al.* (1994), sediments were 3-5 times enriched in available P relative to soil. However, extractable P concentration in sediments was lower than in ashes, possibly due to P insolubilisation, as a consequence of increased cation concentration and pH, or to P adsorption on clay or Fe-oxide surfaces, as Stephens *et al.* (2004) suggested for the lack of P increase in stream water after a prescribed fire. Although higher than the 14 g ha<sup>-1</sup> year<sup>-1</sup> found by Helvey *et al.* (1985), the accumulated losses of extractable P were below the reported range (117-830 g ha<sup>-1</sup> year<sup>-1</sup>: Thomas *et al.* 1999; Gimeno-Garcia *et al.* 2000; Robichaud *et al.* 2006). However, these losses were relevant in control and seeding plots because they accounted for 4.9-7.4% of the extractable P pool in ashes.

It is generally held that post-fire erosion could affect downstream water quality and sedimentation, but not long-term site productivity (Helvey *et al.* 1985; Emmerich 1999; Robichaud *et al.* 2006), because nutrients lost with the eroded sediments are typically negligible compared with soil reserves (Means *et al.* 1992; Busse 1994; Baird *et al.* 1999), outputs from burning (Feller 1988), or inputs with ash after wildfires (Grier 1975; Robichaud *et al.* 2006). However, the present study showed that, at least in steep areas in a humid climate, even a fire of low to moderate severity can lead to substantial accumulated losses of C, N and NH<sup>+</sup>-N with the eroded sediments, as well as a noticeable fraction of the extractable Ca and Mg pools in ashes and topsoil. Under the described scenario, particulate losses of NO<sup>+</sup>-N, Na and K were less important. However, it must be highlighted that this can be due to high losses of these nutrients in soluble forms, as suggested by the differences (10-fold for NO<sup>+</sup>-N; 6-fold for Na and K) between their respective concentrations in ashes and sediments from the first erosion event, which were much higher than those for less soluble divalent cations (no difference for Ca; 2-fold for Mg). Consequently, our results agreed with the advice that post-fire soil quality might be seriously affected through preferential loss of fines and organic matter, both of which contain more nutrients than does the remainder of the soil (Thomas *et al.* 1999; Shakesby *et al.* 2002), especially in the case of recurrent fires in areas where they are not a natural disturbance for ecosystems.

## Conclusions

Little or no effect of treatments on nutrient concentration in sediments was found. Data on C, N and <sup>13</sup>C indicate that our

sediments were mainly charred plant and litter material. During the study, the high concentrations of C, N and NH<sup>4</sup>-N in sediments varied little. Those of Mg and Ca decreased progressively without reaching their reference topsoil values, which were reached by Na and K within 3 months; NO<sup>3</sup>-N concentration in sediments was low and stable.

In all treatments, the fraction of available ash nutrient lost by erosion was high for C and N (22-25%), intermediate for NH<sub>4</sub>-N, Ca, P and Mg (5-12%), and low for NO<sup>3</sup>-N, Na and K (1-4%), although the result for the latter three nutrients may have been due to high losses in solution. Compared with control plots, seeding slightly reduced the accumulated losses of nutrients with sediments, which were decreased 10-fold by mulching. Because emergency stabilisation treatments had little effect on cation concentrations in sediments, the low cation losses in mulching plots was a consequence of erosion reduction.

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