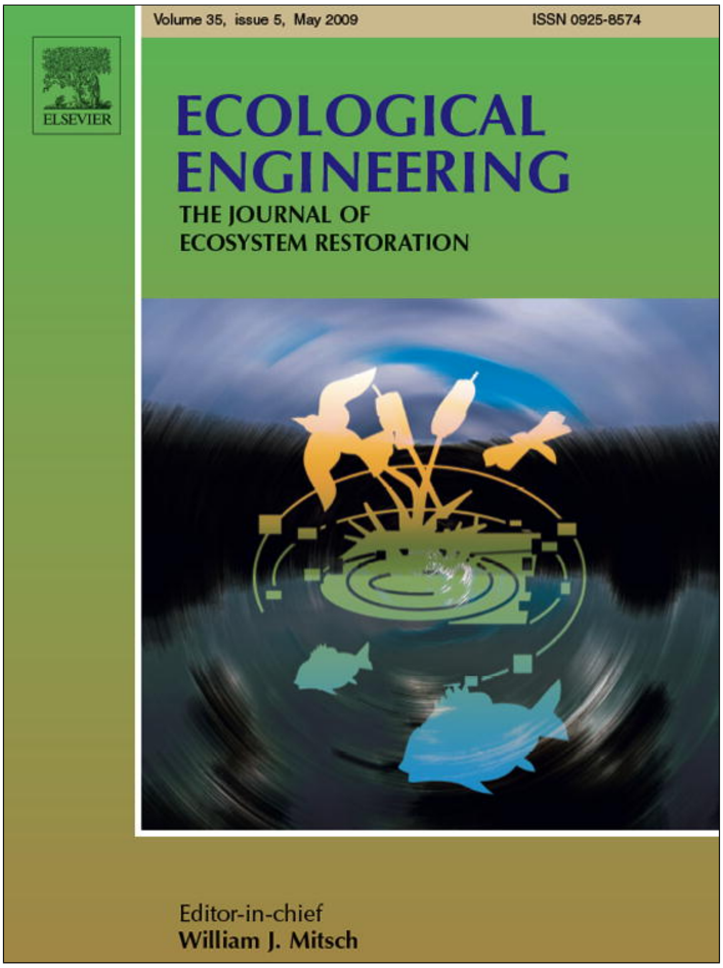
Volume 35, issue 5, May 2009
ISSN 0925-8574
ELSEVIER
ECOLOGICAL ENGINEERING
THE JOURNAL OF ECOSYSTEM RESTORATION
Editor-in-chief
William J. Mitsch

ECOLOGICAL ENGINEERING 35 (2009) 820–828

available at www.sciencedirect.com

ScienceDirect

journal homepage: www.elsevier.com/locate/ecoleng

Short-term effects of experimental burning on soil nutrients in the Cantabrian heathlands

*E. Marcos**, *C. Villalón, L. Calvo, E. Luis-Calabuig*

Área de Ecología, Fac. Ciencias Biológicas y Ambientales, Universidad de León, 24071 León, Spain

ARTICLE INFO

Article history:
Received 30 June 2008
Received in revised form
21 November 2008
Accepted 7 December 2008

Keywords:
Calluna vulgaris
Experimental burning
Nitrogen
Phosphorus
Soils

ABSTRACT

The aim of this study is to determine the short-term effects of fire on nitrogen and phosphorus soil concentration in heathland sites dominated by *Calluna vulgaris* in the Cantabrian Mountain range (NW Spain). Three *C. vulgaris* heathlands sites (San Isidro, Riopinos I and Riopinos II) were selected. In June 2005, one plot (20 m × 20 m) per site was subjected to an experimental fire and the other was used as a control. Immediately after the fire, ten ash samples and ten soil samples (at a depth of 5 cm) were collected and thoroughly mixed. Soil moisture, temperature, total N, NH_4^+, NO_3^-, total P, available P and pH were determined in each sample. The quantity of ashes deposited was 300 g/m^2, with a pH of 9, low N content but higher P concentrations. Significant differences in temperature and soil moisture were detected between the fire-treated and control plots. No significant differences for soil pH, total and available P, total N and NO_3^- concentration were found between the treatments. However, the concentration of ammoniacal-N indicated a significant increase 11 months post-fire and was produced by the changes in environmental soil conditions after the fire. Our results show that low intensity fires do not modify the concentration of N and P in the soil. However, post-fire conditions favour an increase in ammoniacal-N one year later.

1. Introduction

One of the characteristic communities in the Cantabrian Mountains (NW Spain) is heathland dominated by *Calluna vulgaris*. It is a shrub up to 60 cm in height. It is widely distributed in Europe and Russia and often dominant in northwest Europe, from western Scandinavia to northern Spain (Gimingham, 1960, 1972). These ecosystems in Spain represent the most southerly examples of this type of heathland in Western Europe. Heathlands were recognised as an important habitat at a European level by the European Union Habitats Directive in 1992 (Anon., 1992). The conservation of heathland is important because rare plant, bird and invertebrate species depend on the habitat for their survival (Evans et al., 1994). This has contributed to making these habitats the subject of many research studies to help to conserve them (Marcos et al., 2003; Dorland et al., 2004; Niemeyer et al., 2005; Calvo et al., 2007).

Generally, heathlands occur on nutrient-poor acidic soils and are limited by nitrogen and phosphorus (Gimingham, 1972; Groves, 1981); any increase in soil resources is likely to lead to changes in soil functioning and vegetation characteristics. The Cantabrian Mountain range where *Calluna* heathlands occur is characterised by podzolized soils with few nutrients, phosphorus being the most limited element (Fernández, 2002). Traditionally, these areas in NW Spain were used to graze sheep, goats, cattle and horses in transhumant pastoral systems. Burning and cutting were regularly used to provide pasture as a part of management (Fernández, 2002; Calvo et al., 2007). Traditional land use has perpetuated

* *Corresponding author*. Fax: +34 987 29 1501.
E-mail address: elena.marcos@unileon.es (E. Marcos).
0925-8574/$ – see front matter
doi:10.1016/j.ecoleng.2008.12.011

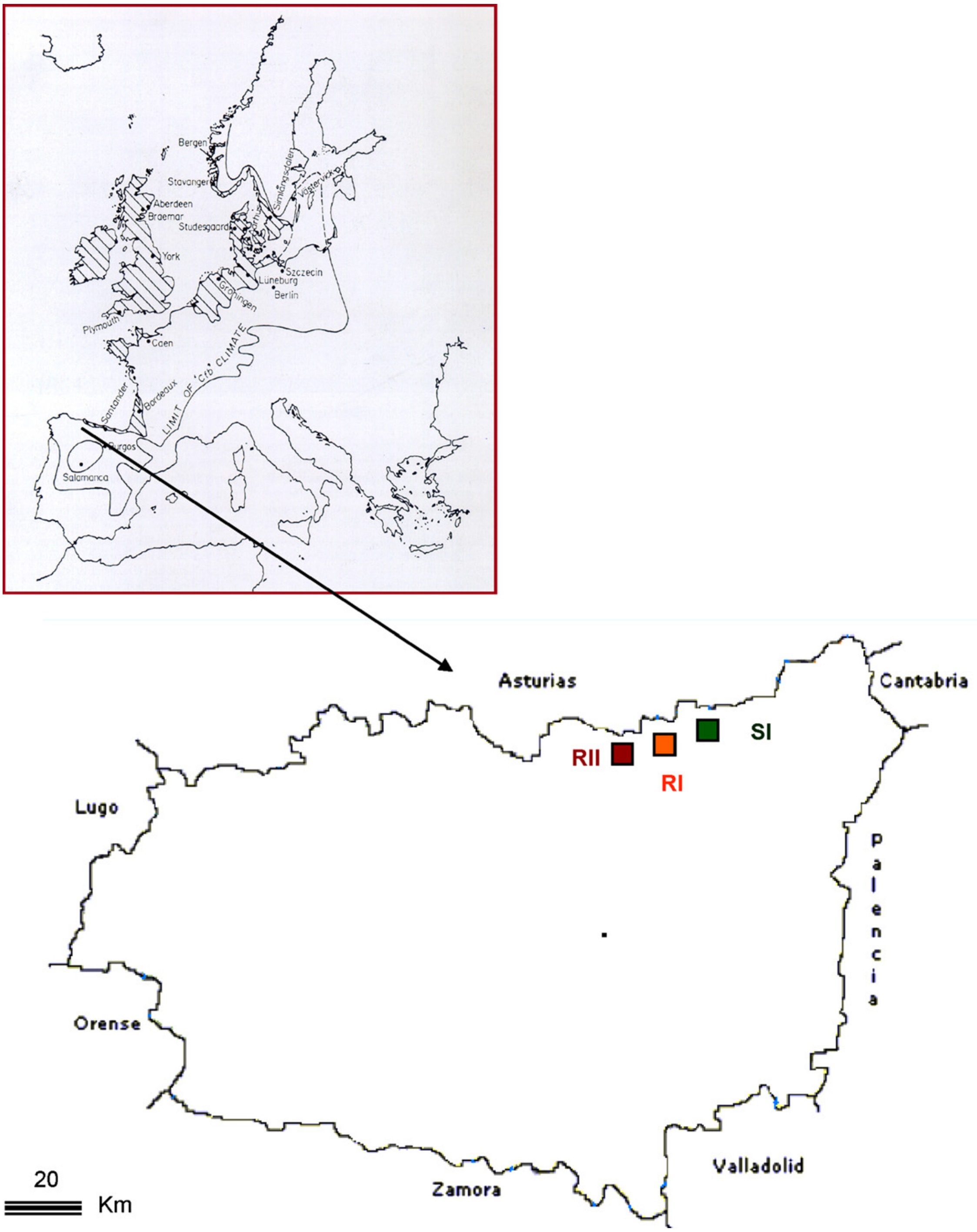

Fig. 1 – Geographical location of the study areas (RI: Riopinos I; RII: Riopinos II; SI: San Isidro). The hatched area indicates the location of the study areas in León province.

ecosystems of a low nutrient status in which plant succession is arrested (Webb, 1998). However, at present traditional management has nearly disappeared in most countries as well as in the Cantabrian Mountains due to changes in agricultural practices and for socio-economic reasons. As a result of these changes, patches characterised by *C. vulgaris* and other dwarf ericaceous species, such as *Erica tetralix* L., are scarce in the Cantabrian Mountains (Calvo et al., 2002, 2005). Also, the increasing amount of nutrient input by atmospheric deposition can produce changes in soil fertility (Mitchell et al., 1999) and increase the ability of grasses to compete with the *Calluna* vegetation (Kristensen and McCarty, 1999).

In order to preserve this type of ecosystem, the employment of management practices to remove nutrients has increased in importance (Power et al., 2001). Prescribed burning is still the predominant measure in the management of lowland heaths (Pakeman et al., 2003). The burning of heather and other vegetation has been practised on both lowland and upland heaths in Britain since at least the seventeenth century (Tubbs, 1974). Small and superficial burnings, as occurring on military training areas, seem to be useful as tools in regeneration of heathlands (Wanner and Xylander, 2003). Together with other traditional methods of heathland management, such as grazing (Gallet and Roze, 2001), turf-stripping and cutting of scrub and bracken, burning prevents tree and scrub colonisation, halts degeneration of the shrub layer and maintains low soil nutrient concentrations (Webb and Haskins, 1980). Fires are generally used on dryer heaths dominated by *C. vulgaris*, where the aim is to burn off ageing growth of the dwarf shrubs to allow their regeneration (particularly to provide food for grouse or livestock), to kill invading scrub and, on a larger scale, to create a mosaic of heathland of different ages (Khoon and Gimingham, 1984; Gimingham, 1992; Michael, 1993).

There has been no burning in the Cantabrian Mountains for some time due to the decrease in livestock grazing. This has caused the present heathlands to degenerate (>40 years). We based our work on the hypothesis that the use of burning is an appropriate measure for eliminating nutrients from the system whilst allowing to maintain the plant community in a young, non-degenerative state. As N and P are known to be the most important nutrients limiting growth of heathlands, we focused on the effects of burning on these nutrients. So the aim of this study was to determine the short-term (one year) effects of fire on nitrogen and phosphorus soil concentration in heathlands dominated by *C. vulgaris* in the Cantabrian Mountain range.

2. Materials and methods

2.1. Study area

The study site is located on the León (south) side of the Cantabrian Mountain range (NW Spain) (Fig. 1). This area has a Eurosiberian climate characterised by a dry period of less than 2 months in summer and usually presents late snow which remains until the end of May. Mean annual precipitation is 1319.5 mm and the mean annual temperature is 5.5 °C. Three *C. vulgaris* heathlands sites (2–4 ha) at least 2.5 km apart were selected. Each site was considered a replicate. San Isidro (1636 m a.s.l., 43°03′N, 5°21′W) represents a flat, continuous heathland area facing north and exposed to winds. Riopinos I (1653 m a.s.l., 43°02′N, 5°24′W) is characterised by a discontinuous heathland area with a high proportion of bare soil and north exposure on a steep slope. Riopinos II (1567 m a.s.l., 43°02′N, 5°26′W) is north-facing and exposed to winds but with a less steep slope. The study zone presents a high degree of geological complexity although the sites are characterised by shales and sandstones (San Isidro) of Carboniferous and Ordovicic quartzite rocks (Riopinos I and Riopinos II). Prevailing soil types are acidic soils of low fertility, and often sandy that suffers podsolitation process. Vegetation is represented by *C. vulgaris* as a dominant species, *E. tetralix* and *Vaccinium mirtyllus* as accompanying ones.

Table 1 – Weather data and average fuel loading at the time of ignition.

	Riopinos I	Riopinos II	San Isidro
Relative moisture (%)	40	56	62
Wind speed ($m\,s^{-1}$)	2.4	1.6	–
Ambient temperature (°C)	20	17	16
Average fuel loading ($kg\,m^{-2}$)			
Calluna vulgaris	2.02	2.10	1.95
Erica tetralix	0.00	0.17	0.10

2.2. Sampling and monitoring

We placed two permanent plots (20 m × 20 m) in each heathland site. In June 2005, one plot per site was subjected to an experimental fire and the other was used as a control. During the fire, the Forestry Service measured environmental data such as air temperature, relative moisture and wind speed (Table 1). Average fuel loading was also estimated in each study site for the two most representative species (*C. vulgaris* and *E. tetralix*). So, five plots (1 m^2) per site were established and the above-ground biomass was harvested. Then, biomass was dried (40 °C-48 h) and weighed in the laboratory.

Initial measurements were made in order to establish the baseline characteristics of the soils at the experimental sites. In every site, ten soil samples were taken at a depth of 5 cm and the following properties were measured: particle size analysis of the fraction <2.0 mm; bulk density; total organic carbon; total N, NH_4^+, NO_3^-, total P, available P and pH.

Immediately after the fire, ten ash samples (10 cm × 10 cm) were taken to determine the level of nutrient input as a result of ash deposition. Then ten soil samples (at a depth of 5 cm) were collected in each plot and thoroughly mixed (from this sample we analysed three subsamples). Litter was removed prior to sampling. This procedure was repeated 4, 11, 14 and 17 months after treatments. Soil total N, NH_4^+, NO_3^-, total P, available P and pH were determined in each sample. Ash samples were characterised for total N, NH_4^+, NO_3^-, available P and pH.

One part of the soil samples was dried, sieved to remove material of a diameter >2 mm and stored until analysis. Total organic carbon in these samples was measured with a TOC 5000 analyser. Particle size analysis was carried out with the Bouyoucos densimeter (MAPA, 1994). The soil pH was determined potentiometrically at a 1:2.5 ratio in H_2O. Available P was determined by the Bray–Kurtz method (Kalra and Maynard, 1991) and total P was dissolved in $HNO_3/HCl/H_2O_2$ solution using microwave digestion. Digests were analysed with ICP–AES. Total N was determined with the Kjeldahl method (Bremner and Mulvaney, 1982). The rest of the soil samples were frozen at −18 °C until extraction. Soil samples were shaken in 2 M KCl (soil:solution ratio 1:10) for 1 h, centrifuged at 3500 rpm and passed through a rinsed GF/F filter. The filtrate was stored for a maximum of one week at 4 °C until mineral N analysis. Then NH_4^+ and NO_3^- were distilled with MgO and Devarda's alloy, respectively, and valued with H_2SO_4 0.005N in a Büchi.

In addition, from May to October 2006 the variation in temperature and soil moisture was measured at monthly intervals. The TDR (Time Domain Reflectometry) technique was used for this (Tapias Pantebre et al., 2001).

Table 2 – General soil characteristics in the study sites. Standard deviation is shown in parentheses.

	Riopinos I	Riopinos II	San Isidro
Sand (%)	92.40 (2.97)	82.80 (3.63)	74.80 (1.79)
Silt (%)	7.60 (2.97)	17.20 (3.63)	24.80 (2.28)
Clay (%)	0.00 (0.00)	0.00 (0.00)	0.40 (0.89)
Texture	Sandy	Sandy	Loamy-sand
Bulk density ($mg\,cm^{-3}$)	0.87 (0.22)	0.55 (0.12)	0.67 (0.12)
pH	4.09 (0.31)	3.77 (0.11)	4.21 (0.19)
TOC ($g\,kg^{-1}$)	116.62 (95.46)	143.90 (90.55)	68.73 (23.70)
TOC ($g\,cm^{-3}$)	92.72 (68.00)	86.02 (63.97)	44.62 (17.45)
Total N ($g\,kg^{-1}$)	0.85 (0.51)	1.96 (0.89)	1.14 (0.40)
Total N ($g\,cm^{-3}$)	0.69 (0.34)	1.14 (0.65)	0.74 (0.31)
NH_4^+ ($mg\,kg^{-1}$)	12.45 (1.64)	10.15 (0.98)	17.16 (7.05)
NH_4^+ ($mg\,cm^{-3}$)	10.67 (2.76)	5.78 (1.26)	11.06 (4.94)
NO_3^- ($mg\,kg^{-1}$)	0.00 (0.00)	0.00 (0.00)	7.74 (2.52)
NO_3^- ($mg\,cm^{-3}$)	0.00 (0.00)	0.00 (0.00)	5.03 (2.07)
Total P ($mg\,kg^{-1}$)	206.20 (55.26)	764.60 (368.89)	578.70 (160.26)
Total P ($mg\,cm^{-3}$)	177.51 (72.15)	446.99 (267.21)	368.83 (112.86)
Available P ($mg\,kg^{-1}$)	5.33 (6.08)	5.20 (1.89)	20.55 (18.82)
Available P ($mg\,cm^{-3}$)	4.10 (3.43)	2.91 (1.13)	13.78 (15.61)

2.3. *Data analysis*

Data from soil variables were analysed by means of factorial ANOVA. For statistical analysis, data expressed as a percentage were first arcsine-square root transformed. The logarithmic transformation $y = \log x + 1$ was used for the rest of the variables. Soil data were submitted to two-way ANOVA, factor one: time as the repeated measures, and factor two: experimental treatments (burn and control). If a repeated measures test detected significant interactions between time and treatment for each dependent variable, one-way ANOVA tests were carried out for each sample date separately. Post hoc Tukey tests were carried out to determine the significance of the differences. Statistical tests were carried out using the STATISTICA 6.0 program from Statsoft 1984-2001.

3. Results

3.1. *Pre-treatment characteristics of soils*

The soils in Riopinos I and Riopinos II are characterised by their loamy sand texture, whilst that in San Isidro is sandy loam (Table 2) due to its lower sand content. They were generally very permeable soils, as can be seen by the low apparent density values. The available nutrient content always had the highest values in the San Isidro mountain pass, whilst the total nitrogen and phosphorus content was always higher in Riopinos II. Riopinos II had a larger reserve of organic nutrients but very little available for plants, whilst the characteristics of Riopinos I were between the other two.

3.2. *Ash composition*

The quantity of ashes settled in the study areas was approximately 300 g/m^2 (Table 3). These ashes came mainly from the herbaceous, litter and woody species such as *E. tetralix*, *Vaccinium myrtillus* and thinner branches of *C. vulgaris*, since the rest of the vegetation was not consumed. The ashes were characterised by a basic pH of around 9. This increase in the pH reflects an important quantity of soluble cations present in the ashes. Total and mineral nitrogen content was very low in all sites. However, the available P content was high in all the study zones, above all in Riopinos I. These results indicated that the experimental fire was of low intensity and vegetation combustion was not complete.

3.3. *Temperature and moisture after burning*

During the sampling period the soil temperature varied between 7 °C and 23 °C in the control plots and between 6 °C and 26 °C in the burned ones (Fig. 2). The temperature in the burned plots was always higher than that in the controls ($F_{1,4} = 15.08$; $P < 0.05$). The highest temperatures were recorded between May and August with September and October showing significantly lower ones ($F_{5,20} = 29.41$; $P < 0.01$), due to a decrease in environmental temperature. As regards soil moisture, it is of note that it was always greater in the burned plots than the controls, with significant differences being detected between treatments ($F_{1,4} = 10.20$; $P < 0.05$). These differences were accentuated in July and August (Fig. 2) when the vegetation had higher water demands. The soil moisture followed the same trend in both treatments over time so the high-

Table 3 – Weight and chemical characteristics of the ashes in the study areas. Standard deviation is shown in parentheses.

	Riopinos I	Riopinos II	San Isidro
Weight ($g\,m^{-2}$)	337.60 (218.70)	302.00 (92.30)	354.80 (138.90)
pH	8.56 (0.15)	9.47 (0.03)	9.10 (0.20)
Total nitrogen ($g\,kg^{-1}$)	9.0 (0.2)	8.9 (0.4)	10.5 (0.5)
NH_4^+-N ($mg\,kg^{-1}$)	1.11 (1.92)	0.00 (0.00)	0.00 (0.00)
NO_3^--N ($mg\,kg^{-1}$)	0.00 (0.00)	0.00 (0.00)	5.34 (9.24)
Available P ($mg\,kg^{-1}$)	233.08 (121.85)	24.25 (21.12)	32.67 (1.01)

est moisture content appeared in September and October ($F_{5,20} = 46.12$; $P < 0.01$) and the lowest in July and August.

3.4. *Effects of burning on pH and phosphorus forms*

Although the soil pH increased slightly 4 months post-fire (Fig. 3), no significant differences were observed between the burned and control plots ($F_{1,4} = 0.01$; $P > 0.05$). The pH-values of the control and treatment plots did not change significantly ($F_{4,16} = 2.83$; $P > 0.05$) throughout the sampling period.

The total P concentration was not significantly affected ($F_{1,4} = 0.05$; $P > 0.05$) after the fire, although it decreased slightly in the burned plots (Fig. 3) due to organic P combustion during the fire. An increase in total P was observed in both treatments and reached its maximum value after 14 months, with significant differences when it was compared with the first two samplings ($F_{4,16} = 4.97$; $P < 0.01$). In contrast, the available P content was greater in the burned plots than in the control, although the differences were not significant ($F_{1,4} = 0.008$; $P > 0.05$). Fig. 3 shows a different temporal response between the treatments: the phosphorus in the burned plots tended to decrease until the last sampling, whilst the control plots varied more over time. No significant differences were detected between the different samplings ($F_{4,16} = 1.92$; $P > 0.05$).

3.5. *Effects of burning on nitrogen forms*

No significant changes were observed in the total N content after fire ($F_{1,4} = 0.001$; $P > 0.05$); this could be related to the deposition of partially burned vegetation on the soil surface. Three months after fire, total nitrogen decreased in the control and burned plots (Table 4). However, a significant increase ($F_{4,16} = 9.48$; $P < 0.05$) was observed in the year after burning in both the burned and control plots.

No differences were detected between the burned and control plots in NH_4^+ concentration immediately after the fire ($F_{1,4} = 2.75$; $P > 0.05$) or in the first few months (Table 4). However, when comparing control and burning separately for every sampling period, there was a significant increase in NH_4^+ in the burned plots after 11 months ($F_{1,16} = 10.99$; $P < 0.01$) and 14 months ($F_{1,16} = 25.97$; $P < 0.05$), although there was a clear decrease after 17 months with values similar to those of the initial sampling. The NH_4^+ concentration in the control plots remained lower than in the burned ones during the study period; although a slight increase was detected over time.

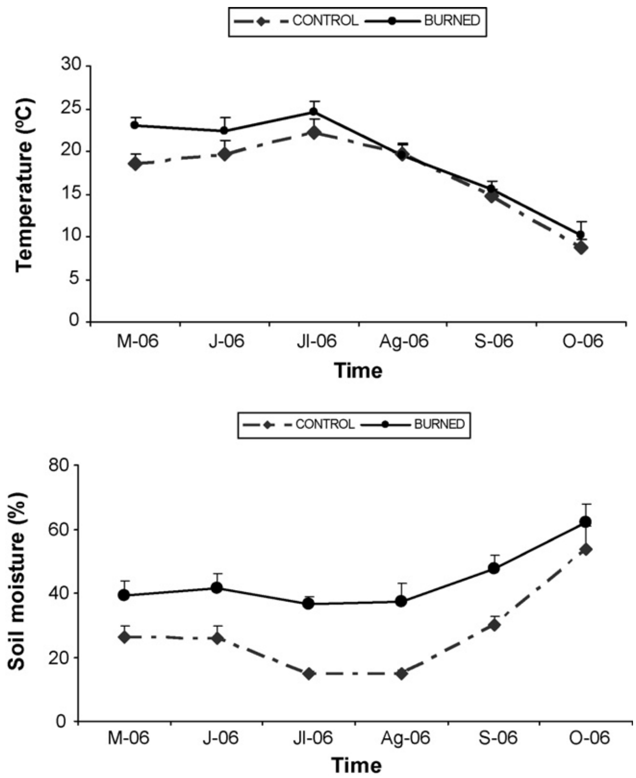

Fig. 2 – Mean values and standard error of the temperature and moisture in the soil in the burned and control plots during the vegetative activity period in 2006.

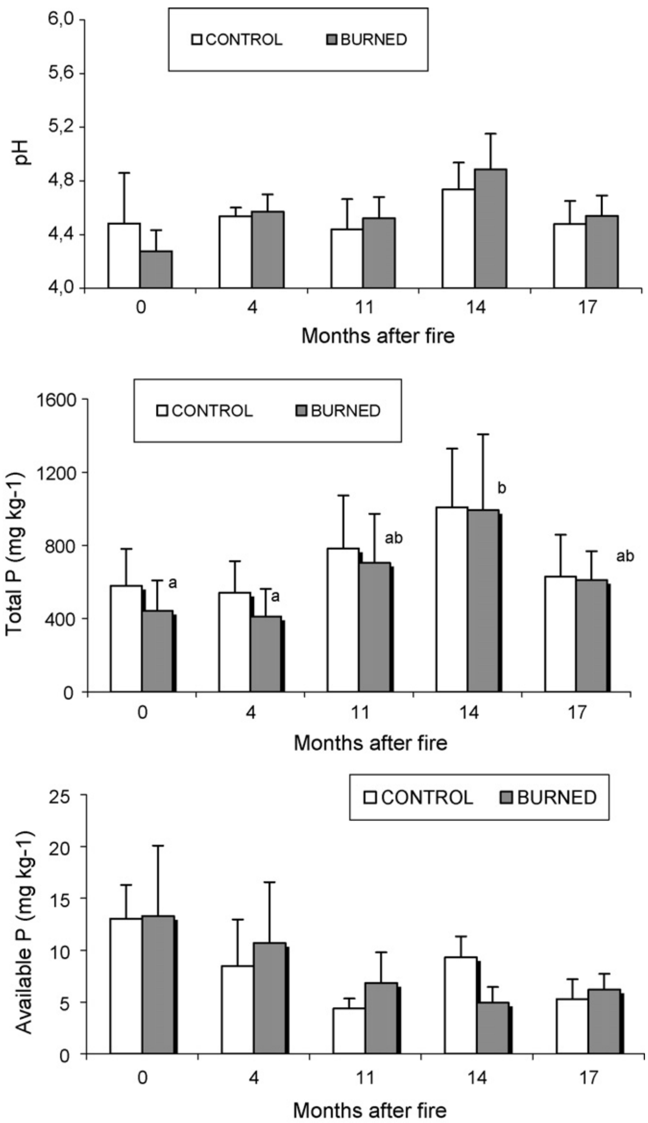

Fig. 3 – Mean values and standard error of soil pH and available and total phosphorus in the soil of the burned and control plots throughout the study period.

Table 4 – Means and standard errors ($n = 3$) for concentrations (p/p and p/v) of total nitrogen, NH_4^+ and NO_3^- in control and burned plots.

	Total N ($g\,kg^{-1}$) Total N ($g\,cm^{-3}$)		NO_3^- ($mg\,kg^{-1}$) NO_3^- ($mg\,cm^{-3}$)		NH_4^+ ($mg\,kg^{-1}$) NH_4^+ ($mg\,cm^{-3}$)	
	Control	Burned	Control	Burned	Control	Burned
0 months	4.66 (1.16)ab 2.98 (0.51)a	4.18 (1.10)ab 2.90 (0.61)a	3.67 (1.47)a 2.17 (1.47)a	0.64 (0.52)a 0.34 (0.52)a	29.85 (18.46)a 16.78 (18.00)a	26.46 (14.08)a 15.74 (15.60)a
4 months	2.48 (0.58)a 1.58 (0.18)b	2.58 (0.52)a 1.75 (0.26)b	8.21 (2.45)a 4.90 (3.04)a	6.89 (5.35)a 3.71 (5.37)a	15.36 (6.36)a 9.16 (5.89)a	21.04 (8.91)a 14.20 (8.16)a
11 months	5.28 (1.13) b 3.40 (0.36)a	5.329 (1.63)b 3.49 (0.79)a	6.78 (3.69)a 4.79 (2.65)a	1.33 (0.55)a 0.88 (0.55)a	16.29 (5.02)a 12.45 (10.47)a	72.53 (23.65)a 46.84 (23.92)a
14 months	6.53 (2.01)b 4.12 (0.71)a	5.51 (1.73)b 3.64 (0.88)a	1.01 (0.21)a 0.68 (0.13)a	4.22 (2.16)a 3.41 (3.73)a	21.36 (7.66)a 16.33 (13.84)a	79.07 (15.63)a 52.94 (4.29)a
17 months	6.10 (2.02)b 3.78 (0.80)a	7.17 (1.98)b 4.78 (0.97)a	6.58 (5.18)a 5.74 (8.51)a	0.80 (0.28)a 0.62 (0.29)a	27.80 (11.35)a 19.63 (9.40)a	41.68 (6.75)a 28.34 (2.49)a

Different letters in the same column indicate significant differences ($P < 0.05$) between sampling periods.

Except for the sampling carried out 14 months post-fire, the NO_3^- concentration was always lower in the burned plots than in the controls (Table 4), although these differences were not significant ($F_{1,4} = 0.001$; $P > 0.05$). No trend was detected in the burned plots throughout the study period, although a tendency to increase their NO_3^- content in autumn and decrease in summer was observed in the controls. No significant differences were detected among the different samplings ($F_{4,16} = 0.45$; $P > 0.05$).

4. Discussion and conclusions

Fire characteristics depend on the type of vegetation as well as the weather conditions at the time of burning. Post-fire changes in soil nutrients are closely related to the energy released during fire and to the amount of ash deposited on the soil (Romanyá et al., 2001; Härdtle et al., 2007). According to Gimingham (1972) *C. vulgaris* burns produce higher temperatures than other dwarf shrubs and grasses. However, this experimental fire can be considered low intensity, due to the environmental conditions present at the time of burning, which propitiated moderate temperatures. The low fire intensity was reflected in the type of ashes deposited, which presented a large quantity of partially carbonised debris, and in their chemical composition. These ashes had slightly lower pH-values than those detected in other studies (Giovannini, 1994; Marcos, 1997). Nevertheless, the N and P content was similar to that recorded by Soto and Díaz-Fierros (1993) in ashes derived from burning *Ulex* in north-west Spain, obtained by heating in a laboratory.

The changes produced after a fire in the soil moisture and soil temperature were closely related to the degree of alteration in the soil surface layer and the reduction in plant cover (Pritchet, 1991; Vermeire et al., 2005). One year after the fire we found that burned plots were generally warmer than controls in all the sampling months, with the difference being more marked in May, June and July. Warming of the soil following fire is well-supported and widely accepted (Viro, 1974; Sharrow and Wright, 1977; Bremer and Ham, 1999; Vermeire et al., 2005; Mohamed et al., 2007). This temperature increase is a consequence of the alteration suffered by the litter after burning and the greater solar energy input to the soil, which results in the deposition of dark ashes (Pritchet, 1991). In contrast, the effects of fire on soil moisture were not well defined as there is disagreement on this aspect among researchers. Thus, some authors found important decreases of soil moisture in burned plots (Martínez-Fernández and Díaz-Pereira, 1994; Josa et al., 1994). Others, like González-Pelayo et al. (2006), detect higher values of water content in samples after fire treatments than those of control samples, or simply find no differences (Soto and Díaz-Fierros, 1997; Vermeire et al., 2005). Our results show an increase in soil moisture in the burned plots in comparison with the control throughout the sampling period, which is accentuated in July and August. This increase could be due to the fact that reduced plant cover allowed more precipitation to reach the soil and, given that the fire was of low intensity, the surface organic layer was not completely destroyed, thus permitting greater retention of the water reaching the soil. On the other hand, the fact that there is no woody vegetation means that transpiration of water the deepest soil layers decreases (Soto and Díaz-Fierros, 1997).

The concentration of nutrients observed after burning followed very different trends according to the observed variable. Although one would expect elevated pH-values owing to the deposition of basic ash (Marcos, 1997; Arocena and Opio, 2003), we have found that soil pH-values showed no significant change and this coincides with the results of Mohamed et al. (2007). The fact that there is no significant increase in the soil pH may be due to an incomplete incorporation of ash into the soil and to the low temperatures reached during the fire, thus not increasing it (Giovannini, 1994; Marcos et al., 2007; Terefe et al., 2008). No significant changes were observed in the concentration of total and available P between the burned plots and the control. Total P in burned plots decreased slightly in comparison with the control plots the first year post-fire, as recorded by Kutiel and Shaviv (1989) in Mediterranean soils, to then present values similar to those of the control. Likewise, a non-significant increase in available P occurred in this period of time on comparison with the control plots. This coincides to a greater or lesser extent with most authors carrying out studies on burned soils (Giovannini, 1994; Sánchez et al., 1994; Ilstedt et al., 2003; Gimeno-García et al., 2004; Johnson et al., 2007; Mohamed et al., 2007), who attribute it to the release of P fixed in the burned biomass, which is then deposited with the ashes (Niemeyer et al., 2005) and whose quantity depends on fire intensity (Gimeno-García et al., 2000). The available P increase was produced after moderate fires (Dimitrakopoulos et al., 1994; Marcos et al., 2007), and with higher temperatures it even decreased. In a soil heating programme Giovannini (1994) recorded a decrease in available P at above 460 °C. The increase detected in this study is small due to the low temperatures reached during the fire, the limited incorporation of ashes and the maintenance of low pH-values (De Jong and Klinkhamer, 1983). When the concentration of available P is compared over time, a clear tendency to decrease was observed, above all in the burned plots. These phosphorus losses are common in very sandy soils (Mitchell et al., 2000), as in this case. In addition, the decrease in available P in the burned plots after 14 months coincided with the high losses by washing detected on lysimeters (unpublished data) placed in those plots. These losses were also recorded by Johnson et al. (2007) two years post-fire.

Immediately after burning no significant changes were detected in total N concentration. The fact that no changes occur in total N concentration or even that it increases after the fire is typical of low intensity fires (Martínez-Fernández and Díaz-Pereira, 1994; Sánchez et al., 1994; Gimeno-García et al., 2004; Del Moral et al., 2007), due to the increased organic matter content. However, other studies have found that the post-fire increase in ammoniacal-N is closely related to fire intensity (Romanyá et al., 2001; Gimeno-García et al., 2004). In contrast to what was found by most authors (Raison, 1979; Prieto-Fernández et al., 2004) no changes in ammoniacal-N concentration were detected in the first few months after the fire (Ilstedt et al., 2003). However, high values of ammoniacal-N were detected 11 months after the fire and reached a maximum after 14 months, coinciding with the higher temperature values recorded in the soil and the higher moisture content when compared with the control plots. Prieto-Fernández et al. (2004) also found the highest ammoniacal-N values 12 months post-fire in September and suggest that the rise in soil temperature during the summer increases ammonification. Mohamed et al. (2007) found that after prescribed fires in winter in Germany the most important rise in ammoniacal-N occurred in summer and one of the reasons was the increase in soil temperature. As in this study Prieto-Fernández et al. (2004) detected small quantities of nitrate in burned and unburned soils in Galicia, and, in addition, the fire did not favour an increase in nitrate. These authors attribute this to reduced nitrification and to the scarce number of nitrifying microorganisms, which are, besides, harmed by burning, particularly if the soil pH does not increase. The soils in this study are similar to those in Galicia and very low and even zero nitrification rates have been observed; no losses of nitrate due to washing have been recorded (unpublished data).

Our results showed that low intensity fires practically do not modify total N or P concentration in the soil, since the temperatures reached during combustion are low and even, in most cases, have left part of the organic horizon unconsumed. However, the post-fire conditions allowed an increase in ammoniacal-N concentration to occur in the year following the fire at the time when the environmental conditions were more favourable. If we took into consideration the changes in total nutrient content at ecosystem level, nutrient losses would occur due to vegetation combustion, when fire temperature is high and thus significant N losses would be produced by volatilization. Although the fires may be of low intensity, most of the light fuel with high N content is burnt, whilst coarser parts are less affected (Härdtle et al., 2007). For this reason, experimental burning has the potential to remove nutrients from the ecosystem. It would be interesting to establish the short-term nutrient balance after an experimental fire and carry out longer-term studies to evaluate the ecosystem recovery capacity. So, studies carried out in these areas, comparing different management measures (burning, cutting and ploughing) showed that burning is a good measure to maintain heathlands (Calvo et al., 2002). In these areas *Calluna* is too old to regenerate by vegetative sprouting, only can germinate. Cutting favoured *E. tetralix* which becomes the dominant woody species and it did not germinate because of competition from herbaceous species. Regeneration after ploughing was very slow. In burning treatment a good response was observed, probably due to germination being stimulated by increase in temperature (Calvo et al., 2002). According to different studies (Calvo et al., 2005, 2007) in these areas, burning is the most appropriate management measure to maintain dwarf shrub dominated heaths.

Acknowledgements

This study was supported by the C.I.C.Y.T. (Research Project GGL2006-10998-CO2-01) and by the Junta de Castilla y León (Research Project Ref. LEO39A05).

REFERENCES

Anon., 1992. Council Directive of 21 May 1992 on the Conservation of Natural Habitats and of Wild Fauna and Flora (92/43/EEC). European Commission, Brussels, BE.

Arocena, J.M., Opio, C., 2003. Prescribed fire-induced changes in properties of subboreal forests soils. Geoderma 113, 1–16.

Bremner, J.M., Mulvaney, C.S., 1982. Nitrogen total. In: Page, A.L. (Ed.), Methods of Soil Analysis: Part: 2. Chemical and Microbiological Properties. American Society of Agronomy, Madison, WI, pp. 595–624.

Bremer, D.J., Ham, J.M., 1999. Effect of spring burning on the surface energy balance in a tallgrass prairie. Agric. Forest Metereol. 97, 43–54.

Calvo, L., Tárrega, R., Luis, E., 2002. Regeneration patterns in a *Calluna vulgaris* heathland in the Cantabrian mountains (NW Spain): effects of burning, cutting and ploughing. Acta Oecol. 23, 81–90.

Calvo, L., Alonso, I., Fernández, A.J., Luis-Calabuig, E., 2005. Short term study of effects of fertilization and cutting treatments on the vegetation dynamics of mountain heathlands in Spain. Plant Ecol. 179, 181–191.

Calvo, L., Alonso, I., Marcos, E., De Luis, E., 2007. Effects of cutting and nitrogen deposition on biodiversity in Cantabrian heathlands. Appl. Veg. Sci. 10, 43–52.

De Jong, T.J., Klinkhamer, P.G.L., 1983. A simulation model for the effects of burning on the phosphorus and nitrogen cycle of a heathland ecosystem. Ecol. Model. 19, 263–284.

Del Moral, F., De Haro, S., Sánchez, J.A., Simón, M., García, I., 2007. Efectos en los suelos en incendios de baja intensidad en ambiente mediterráneo. In: Bellinfante, N., Jordán, A. (Eds.), Tendencias actuales de la Ciencia del Suelo. Universidad de Sevilla, Sevilla, pp. 554–562.

Dimitrakopoulos, A.P., Martin, R.E., Papamichos, N.T., 1994. A simulation model of soil heating during wildland fires. In: Sala, M., Rubio, J.L. (Eds.), Soil Erosion and Degradation as a Consequence of Forest Fires. Geoforma Ediciones, Logroño, Spain, pp. 199–206.

Dorland, E., van den Berg, L.J.L., van den Berg, A.J., Vermeer, M.L., Roelofs, J.G.M., Bobbink, R., 2004. The effects of sod cutting and additional liming on potential net nitrification in heathland soils. Plant Soil 265, 267–277.

Evans, A., Painter, M., Wynde, R., Micheal, N., 1994. An inventory of lowland heathland in England: a foundation for an effective conservation strategy. RSPB Conserv. Rev. 8, 24–30.

Fernández, A.J., 2002. Efecto de la concentración de nitrógeno en las comunidades de callunar de alta montaña. In: Memoria de Licenciatura. Universidad de León.

Gallet, S., Roze, F., 2001. Conservation of heathland by sheep grazing in Brittany (France): importance of grazing period on dry and mesophilous heathlands. Ecol. Eng. 17, 333–344.

Gimeno-García, E., Andreu, V., Rubio, J.L., 2000. Changes in organic matter, nitrogen, phosphorus and cations as a result of fire and water erosion in a Mediterranean landscape. Eur. J. Soil Sci. 51, 201–210.

Gimeno-García, E., Andreu, V., Rubio, J.L., 2004. Spatial patterns of soil temperatures during experimental fires. Geoderma 118, 17–38.

Gimingham, C.H., 1960. Biological flora of the British Isles. *Calluna Salisb*. J. Ecol. 48, 455–483.

Gimingham, C.H., 1972. Ecology of Heathlands. Chapman and Hall, London, UK.

Gimingham, C.H., 1992. The lowland heathland management handbook. English Nature Science Ser. 8. Peterborough, UK.

Giovannini, G., 1994. The effect of fire on soil quality. In: Sala, M., Rubio, J.L. (Eds.), Soil Erosion and Degradation as a Consequence of Forest Fires. Geoforma Ediciones, Logroño, Spain, pp. 15–28.

González-Pelayo, O., Andreu, V., Campo, J., Gimeno-García, E., Rubio, J.L., 2006. Hydrological properties of a Mediterranean soil burned with different fire intensities. Catena 68, 186–193.

Groves, R.H., 1981. Heathland soils and their fertility status. In: Specht, R.L. (Ed.), Ecosystems of the World, 9B. Heathland and Related Shrublands. Analytical Studies. Elsevier, Amsterdam, NL, pp. 143–150.

Härdtle, W., Niemeyer, T., Assmann, T., Meyer, H., Von Oheimb, G., 2007. Can prescribed burning compensate for atmospheric nutrient loads in wet heathlands? Phytocoenologia 37 (2), 161–174.

Ilstedt, U., Giesler, R., Nordgren, A., Malmer, A., 2003. Changes in soil chemical and microbial properties after a wildfire in a tropical rainforest in sabah, Malaysia. Soil Biol. Biochem. 35, 1071–1078.

Johnson, D., Murphy, J.D., Walker, R.G., Glass, D.W., Miller, W.W., 2007. Wildfire effects on forest carbon and nutrient budgets. Ecol. Eng. 31, 183–192.

Josa, R., Arias, X., Solé, A., 1994. Effects of slashburning on some soil physical properties in a holm-oak coppice. In: Sala, M., Rubio, J.L. (Eds.), Soil Erosion and Degradation as a Consequence of Forest Fires. Geoforma Ediciones, Logroño, Spain, pp. 29–42.

Kalra, Y.P., Maynard, D.G., 1991. Methods manual for forest soil and plant analysis. Informatoin Report NOR-X-319. Northern Forestry Centre, Alberta, Canada.

Khoon, W.G., Gimingham, C.H., 1984. Birch regeneration in heath vegetation. In: Proceedings of Royal Society Edinburgh 85B, pp. 73–81.

Kristensen, H.L., McCarty, G.W., 1999. Mineralization and immobilization of nitrogen in heath soil under intact *Calluna*, after heather beetle infestation and nitrogen fertilization. Appl. Soil Ecol. 13, 187–198.

Kutiel, P., Shaviv, A., 1989. Effect of simulated forest fire on the availability of N and P in Mediterranean soils. Plant Soil 120, 57–63.

MAPA, 1994. Métodos oficiales de análisis. Tomo III. Servicio de Publicaciones del Ministerio de Agricultura, Pesca y Alimentación. Madrid.

Marcos, E., 1997. Procesos edáficos en comunidades alteradas por el fuego. Tesis Doctoral. Universidad de León.

Marcos, E., Calvo, L., Luis-Calabuig, E., 2003. Effects of fertilization and cutting treatments on the chemical composition of the vegetation and soils of mountain heathlands in Spain. J. Veg. Sci. 14, 417–424.

Marcos, E., Tárrega, R., Luis, E., 2007. Changes in a Humic Cambisol heated (100–500 °C) under laboratory conditions: the significance of heating time. Geoderma 138, 237–243.

Martínez-Fernández, J., Díaz-Pereira, E., 1994. Changes of the physical and chemical properties in a soil affected by forest fire in Sierra Larga (Murcia, Spain). In: Sala, M., Rubio, J.L. (Eds.), Soil Erosion and Degradation as a Consequence of Forest Fires. Geoforma Ediciones, Logroño, Spain, pp. 67–77.

Michael, N., 1993. The Lowland Heathland Management Booklet. English Nature, Peterbourgh.

Mitchell, R.J., Marrs, R.H., Le Duc, M.G., Auld, M.H.D., 1999. A study of the restoration of heathland on successional sites: changes in vegetation and soil chemical properties. J. Appl. Ecol. 36, 770–783.

Mitchell, R.J., Auld, M.H.D., Hughes, J.M., Marrs, R.H., 2000. Estimates of nutrient removal during heathland restoration on successional sites in Dorset, southern England. Biol. Conserv. 95, 233–246.

Mohamed, A., Härdtle, W., Jirjahn, B., Niemeyer, T., von Oheimb, G., 2007. Effects of prescribed burning on plant available nutrients in dry heathland ecosystems. Plant Ecol. 189, 279–289.

Niemeyer, T., Niemeyer, M., Mohamed, A., Fottner, S., Härdtle, W., 2005. Impact of prescribed burning on the nutrient balance of heathlands with particular reference to nitrogen and phosphorus. Appl. Veg. Sci. 8, 183–192.

Pakeman, R.J., Hulme, P.D., Torvell, L., Fisher, J.M., 2003. Rehabilitation of degraded dry heather (*Calluna vulgaris* (L.) Hull) moorland by controlled sheep grazing. Biol. Conserv. 114, 389–400.

Prieto-Fernández, A., Carballas, M., Carballas, T., 2004. Inorganic and organic N pools in soils Burneo or heated: immediate alterations and evolution after forest wildfires. Geoderma 121, 291–306.

Power, S.A., Barker, C.G., Allchin, E.A., Ashmore, M.R., Bell, J.N.B., 2001. Habitat management: a tool to modify ecosystem impacts of nitrogen deposition? Sci. World J. 1, 714–721.

Pritchet, W.L., 1991. Suelos Forestales. Editoral Limusa, México, DF.

Raison, R.J., 1979. Modification of the soil environment by vegetation fires, with particular reference to nitrogen transformations: a review. Plant Soil 51, 73–108.

Romanyá, J., Casals, P., Vallejo, V.R., 2001. Short-term effects of fire on soil nitrogen availability in Mediterranean grasslands and shrublands growing in old fields. Forest Ecol. Manage. 147, 39–53.

Sánchez, J.R., Mangas, V.J., Ortiz, C., Bellot, J., 1994. Forest fire effect on soil chemical properties and runoff. In: Sala, M., Rubio, J.L. (Eds.), Soil Erosion and Degradation as a Consequence of Forest Fires. Geoforma Ediciones, Logroño, Spain, pp. 53–65.

Sharrow, S.H., Wright, H.A., 1977. Effects of fire, ash, and litter on soil nitrate, temperature, moisture and tobosagrass production in the Rolling Plains. J. Range Manage. 30, 266–270.

Soto, B., Díaz-Fierros, F., 1993. Interactions between plant ash leachates and soil. Int. J. Wildland Fire 3 (4), 207–216.

Soto, B., Díaz-Fierros, F., 1997. Soil water balance as affected by throughfall in gorse (*Ulex europaeus*, L.) shrubland after burning. J. Hydrol. 195, 218–231.

Tapias Pantebre, J.C., Salgot de Marçay, M., Casas Pnsati, A., 2001. Utilidad de la técnica TDR para la medida de la variación espacial y temporal de la humedad del suelo en campos de golf. Edafología 8 (3), 1–10.

Terefe, T., Mariscal-Sancho, I., Peregrina, F., Espejo, R., 2008. Influence of heating on various properties of six Mediterranean soils. A laboratory study. Geoderma 143, 273–280.

Tubbs, C.R., 1974. Heathland management in the New Forest, Hampshire, England. Biol. Conserv. 4, 303–306.

Vermeire, L.T., Wester, D.B., Mitchell, R.B., Fuhlendorf, S.D., 2005. Fire and grazing effects on wind erosion, soil water content, and soil temperature. J. Environ. Qual. 34, 1559–1565.

Viro, P.J., 1974. Effects of forest fire on soil. In: Kozlowski, T.T., Algren, C.E. (Eds.), Fire and Ecosystems. Academic Press, Inc., New York.

Wanner, M., Xylander, W.E.R., 2003. Transient fires useful for habitat-management do not affect soil microfauna (testate amoebae)—a study on an active military training area in eastern Germany. Ecol. Eng. 20, 113–119.

Webb, N.R., 1998. The traditional management of European heathlands. J. Appl. Ecol. 35, 987–990.

Webb, N.R., Haskins, L.E., 1980. An ecological survey of heathlands in the Poole basin, Dorset, England in 1978. Biol. Conserv. 17, 281–296.