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THROUGHFALL, RUNOFF AND SOIL EROSION AFTER PRESCRIBED BURNING IN GORSE SHRUBLAND IN GALICIA (NW SPAIN)

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ABSTRACT

The first-year effect of two different prescribed burning treatments on throughfall, runoff and soil erosion was evaluated in gorse shrubland (*Ulex europaeus* L.) in Galicia (NW Spain). The treatments compared were: intense burn, light burn and control (no burn).

Accumulated annual throughfall represented between the 81 and 87 per cent of total rainfall in intensely burned and lightly burned areas, respectively, whereas in the unburnt areas it was 60 per cent. No significant differences between burning treatments were found for the annual throughfall. However, runoff was significantly greater in intensely burned plots (1·5-times) than in lightly burned plots. Burning also resulted in a significant increase in runoff (between 2·5 and 1·7-times, respectively) compared with controls. Total soil losses were small in all treatments, but the intense burn caused significantly greater soil erosion (5·8-times) compared with the unburned areas. Soil losses after the light burn did not significantly differ from the control although they were higher (2·3-times). The relationships obtained between erosion and several rainfall parameters were significantly different in burned areas compared to the control. The same response was observed for runoff. Annual erosion losses showed a strong dependence on percentage of bare soil even for small values of this variable. Litter thickness was also a very important variable influencing on erosion rates.

This study indicated that by combining ignition techniques and high litter moisture content to maintain the percentage of bare soil below 85 per cent, soil erosion was low. Nevertheless, this result was constrained by the low rainfall that occurred during the study. Copyright © 2005 John Wiley & Sons, Ltd.

KEY WORDS: soil losses; fire intensity; fire hydrologic impacts; *Ulex europaeus*; fire hazard reduction

INTRODUCTION

Prescribed burning is frequently used as a vegetation-manipulation technique in shrubland and forested areas for a wide variety of purposes. These include wildlife habitat and plant diversity improvement, silvicultural activities, stimulation of selected plants species regeneration, and fire-regime restoration. However, the most common motivation for its use is the reduction of fuel accumulation and the alteration of fuel continuity (Biswell, 1989; Pyne *et al.*, 1996; Conard and Weise, 1998; Vega *et al.*, 2000; Baeza *et al.*, 2002). Ultimately, both these fuel-complex modifications attempt to decrease the intensity of a subsequent wildfire affecting the treated area and reduce its negative impact, especially post-fire soil erosion.

Shrub communities in Galicia cover more than a half of its total area (Ministerio de Medio Ambiente, 2001). Most of these areas are covered by different gorse species, usually dominated by *Ulex europaeus*. In these shrubland areas fuel loadings of 40–50 Mg ha^{-1} are common (Casal *et al.*, 1990; Vega *et al.*, 2001). Over the last

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11 years, 10 245 ± 1874 fires have burned anually, destroying 19 000 ± 6828 ha of shrubland areas (Ministerio de Medio Ambiente, 2002). Prescribed burning, alone or combined with other techniques, is currently used as an effective and economic tool for wildfire hazard reduction.

However, fire also removes plant cover and may eliminate the mulching effect of the soil organic layer, leaving the soil exposed to raindrop impact and decreasing the infiltration rate. Soil water repellency may also result, thereby increasing runoff and erosion. Increases in runoff and erosion after shrub burning have been measured in different shrubland-type ecosystems; including heathlands (Imeson, 1971; Kinako and Gimingham, 1980; Marcos *et al.*, 2000), Mediterranean type shrublands (De Bano *et al.*, 1979; Wells, 1981; Sánchez *et al.*, 1994; Gimeno *et al.*, 2000) and gorse (Díaz-Fierros *et al.*, 1990; Soto *et al.*, 1993, 1994; Carreira and Niell, 1995; Soto and Díaz-Fierros, 1997, 1998; García-Cano *et al.*, 2000; De Luis *et al.*, 2003). However, studies on changes in throughfall after wildfires or prescribed burning applications are scarce (Soto *et al.*, 1993; Soto and Díaz-Fierros 1997).

There is a lack of information on how different ignition techniques, combined with environmental conditions during burning, such as fuel- and soil-moisture content and wind speed, etc., can affect rainfall interception and post-burn runoff and erosion. This information is critical in determining whether refined prescribed burning prescriptions are compatible with fire-hazard reduction and soil conservation. Understanding the changes in levels of throughfall, runoff and erosion caused by fires of different intensity is also important in increasing our knowledge of the effects of fires on hydrologic parameters, in improving hydrological responses modelling and in validating current runoff and erosion predictive models.

The objective of this study was to evaluate the short-term (one year) changes in runoff, throughfall and erosion caused by two prescribed-burning types of markedly different intensities conducted in a gorse shrub community, compared to an adjacent undisturbed shrub area, and to explore the influence of certain selected variables on these parameters.

MATERIALS AND METHODS

Study Site

The study was carried out on the northwest facing 30 per cent slope (42° 25′ 49″ N; 8° 44′ 30″ W; 225 m a.s.l.) of Monte Coirego, a hill located in the Cotobade Mountains (Pontevedra, NW Spain) within the Lerez River basin.

The shrub community was dominated by *Ulex europaeus* which made up 92–95 per cent of the total phytomass. Other woody species were *Erica cinerea*, *Daboecia polifolia*, *Halimium alyssoides* and *Rubus ulmifolius*. *Pteridium aquilinum*, *Pseudoarrenatherum longifolium*, *Agrostis curtisii* and *A. capillaris* were also present.

The climate is oceanic. Average rainfall is about 1800 mm y^{-1}, with a marked dry period of one to two months in the summer. Mean annual temperature is 12·5°C. Soils are Humic Cambisols (FAO, 1990) with a sandy-loam texture. Soil depth is 0·30–0·40 m over a granite bedrock. The physico-chemical properties of the soil include a pH of 4·4, an organic carbon content of 11·15 per cent and total N of 0·95 per cent.

Experimental Design

Six plots (4 × 20 m each) were installed oriented parallel to the slope and separated by firebreaks. Plots were delimited with thin galvanized iron strip plates and runoff collectors, located at their lower ends, were connected to sedimentation tanks. Each sedimentation tank was then connected to another downslope runoff collection tank through a 1/6 divider. Precipitation was measured with two recording rain-gauges placed close to the plots. Immediately after burning, two troughs (0·11 × 4 m), connected to a tank, were positioned within each plot for throughfall estimation.

Throughfall (22-times), runoff (21-times) and erosion (8-times) were measured after each rainfall event resulting in throughfall and runoff. The study lasted from October 1988 to October 1989. Data were grouped for analysis in four periods: October 1988–December 1988, January 1989–March 1989 April 1989–June 1989 and July 1989–October 1989. Samples of eroded soil were oven dried (105°C) for 24 h.

Plots were treated with prescribed burning using a randomized block design. There were three treatments: (1) ‘intense burn’; (2) ‘light burn’; and (3) no burn ‘control’; and two replicates for each treatment.

Fuel Inventory

The pre-burn shrub fuel loading for each plot was estimated by cutting and weighing all the standing biomass within two rectangular plots (20 m long × 2 m wide), adjacent to the both sides of each erosion plot. Subsamples of 10 per cent of the weight were taken for fuel moisture content determinations and these figures were used to obtain oven-dried fuel loadings. In the laboratory, shrub subsamples were sorted by means of gauges into two size classes: fine (thickness 0–6 mm); and coarse (6–24 mm). Live and dead biomass were also discriminated for each size class. Pre-burn litter and duff mass was measured by destructive sampling of twenty-four (0·30 × 0·30 m) quadrats in each treated plot. Twelve quadrats were randomly located within each 20 × 2 m plot ($n = 24$ per treated plot). This removed material was oven-dried and subsamples were combusted at 450°C for 4 h to determine mineral soil content.

After burning residual woody plant material was sampled using five (1 × 1 m) quadrats randomly located within each 20 × 4 m treated plot. Remaining litter mass was estimated on four (0·30 × 0·30 m) quadrats nested within each 1 × 1 m quadrat ($n = 20$ per treatment plot).

Shrub and Litter Cover and Height

The percentage of shrub cover was estimated by using two 20 m long transects placed in each treatment plot 0·75 m from the boundary. Vegetation height was measured on these transects every 50 cm. The percentage covered by litter and duff and bare soil were also measured along the same transect and expressed as a percentage of the transect length. These measurements were made immediately before burning and one day, 3, 6, 9 and 12 months after burning. Changes in litter depth resulting from the burning were measured using metal pins placed flush with the litter surface at 1 m intervals along the transects. Just after fire, the emergent portion of the pins and the residual litter depth were measured to determine the (absolute and relative) change in the litter thickness. At 3, 6, 9 and 12 months after burning measurements of the remaining litter depth were made at 1 m intervals lengthwise in two different transects placed within the treatment plot and parallel to the maximum slope at random distances from the plot boundaries ($n = 40$).

Prescribed Burns

Burns were conducted in September 1988 with the objective of simulating rather unfavourable (but not extreme) conditions for burning. The meteorological window was fixed as follows: air temperature > 20°C, relative humidity between 40 and 60 per cent and low wind velocity (< 1·5 m s^{-1}). The prescribed moisture content of elevated dead fuels was between 5–10 per cent for the intense burn, 10–15 per cent for the light burn, relatively high for duff (> 90 per cent) in the light burn, and from moderate (70 per cent) to low (< 40 per cent), for the intense burn. Soil moisture content was kept at about field capacity (30 per cent). Fuel and soil moisture content were checked for ten days before burning and meteorological conditions were surveyed with a portable meteorological station placed close to the experimental site. The soil thermal regime during burns was monitored using thermocouples (chromel alumel K type; inconel sheath 1 mm diameter), connected to a datalogger and positioned at eight or ten randomly selected points within each burning plot. On each point, three thermocouples were inserted at the litter and duff interface, at the mineral soil surface and at 2·5 cm depth below the mineral soil surface. Relative air humidity, temperature and wind velocity at 2 m height during burns were continuously measured by an automatic station. Downslope fire was used for the light burns and upslope fire for the intense ones. Flame length was estimated by visual comparison with the known height of the shrubs, aided by photographs and videos. Just before burning, samples from different fuel portions (elevated live and dead fractions, litter and duff) and mineral soil (0–10 cm depth) were randomly taken from ten points per plot to determine moisture content. Fireline intensity was calculated following Byram's equation (Byram, 1959). Available fuel loading was estimated as the difference between pre- and post-fire fine (diameter < 6 mm) standing fuel. The low heat content of *U. europaeus* (19·086 kJ kg^{-1}), used in that equation, was determined by calorimetry.

Statistical Analysis

Throughfall, runoff, erosion, vegetation cover, percentage of bare soil and litter depth data, for each three-month period, were analysed by a repeated measures analysis of variance. The between-subject factor was the treatment with three levels: light burn; intense burn; and control, except the percentage of bare soil, which had only two (light and intense burn). The within-subject factor was the date with four levels: 3; 6; 9; and 12 months after burning. The immediate effect of burn on litter depth was tested using a repeated measures ANOVA. The between-factor had two levels: light and intense burn, and the within-factor two levels: pre- and post-burn values. An ANOVA was also used to test if there were significant differences between the annual values of throughfall, runoff and erosion for the three treatments. In all the cases, the null hypothesis was a lack of significant differences between the mean values for each treatment level. Residuals were tested for normality and variance homogeneity. When significant ($P < 0{\cdot}05$) differences among mean values of the treatment levels were detected, the Student–Newman–Keuls multiple range test was used to compare treatment means.

A simple linear regression technique was used to examine the influence of a set of variables on throughfall, runoff and erosion. Mean values of throughfall, runoff and soil erosion, for each treatment level, and for each measurement period, were the dependent variables. The accumulated precipitation, rainfall kinetic energy, the rainfall erosivity factor and 30 minutes maximum rainfall intensity for each measurement period, vegetation cover, litter depth and percentage of bare soil were used as independent variables. For these latter three variables, interpolated values from each three-month period survey were used. An ANOVA of regression coefficients over groups was also developed to test if the slopes of the regression relationships obtained differed between the three levels of treatment. Correlations between annual values of runoff and erosion and litter depth, percentage of bare soil and vegetation cover were also tested.

Forward stepwise regression analysis was also used to explore the combined effect of the same set of variables influencing throughfall and runoff. The BMDP (1990) package was used for statistical analyses.

RESULTS

Burn Characteristics

The shrub biomass reduction was similar (Table I) in both burning treatments. Relative consumption of litter in the intense burn was 69 per cent, slightly higher than the 54 per cent in the light burn (Table I). Mean rate of fire spread for the intense burns was more than five-times greater than in the light burns (Table II). Consequently, mean fireline intensity (Table II) was also five-times higher in the intense burns compared to the light ones. Marked differences in thermal regime during the fires were observed between burn treatments (Table III) at the litter/humus interface. At this layer, the mean maximum temperature registered in the intense burn was 463°C, five-times greater than in the light burn. The mean duration of temperatures greater than 200°C in the intense burn was 3·8 minutes, consistently higher than the 0·5 minutes registered in the light burn. Nevertheless, those differences drastically dropped at the mineral soil surface where, contrary to expectations, maximum temperatures were not high, reaching just 73°C in the intense burn and 37°C in the light burn. At 2·5 cm, mean maximum temperatures were similar during both burning types, 32°C in the intense burn and 27°C in the light burn.

Table I. Mean fuel loading ($kg\,m^{-2}$) and shrub height (m) in treated and control plots

	Intense burn			Light burn			Control
	Pre-burn	Post-burn	Relative reduction (%)	Pre-burn	Post-burn	Relative reduction (%)	
Shrub stratum loading	4·1 (0·15)	2·1 (0·01)	49	4·6 (0·17)	2·5 (0·11)	46	4·3 (0·43)
Shrub height	2·3 (0·1)	0·8 (0·03)	65	2·4 (0·1)	0·7 (0·03)	71	2·4 (0·1)
Litter layer loading	0·5 (0·02)	0·2 (0·03)	69	0·6 (0·02)	0·3 (0·04)	54	0·6 (0·07)

(Standard error in brackets).

Table II. Mean moisture content of different fuel fractions and mineral soil (%) just before burning, weather conditions and fire behaviour parameters during burns

	Intense burn	Light burn
Moisture contents		
Elevated dead fuel	8·7 (0·8)	12·1 (1·1)
Elevated live fuel	186·0 (15·0)	173·5 (16·5)
Upper litter	6·4 (0·2)	15·3 (1·1)
Lower litter	62·1 (0·1)	92·4 (1·8)
Mineral soil (0–10 cm)	32·1 (0·8)	36·6 (0·6)
Meteorological conditions		
Air temperature (°C)	31·0 (1·0)	30·5 (0·5)
Relative humidity (%)	61·5 (3·5)	41·0 (1·0)
Wind speed ($m\,s^{-1}$)	0·99 (0·1)	1·25 (0·2)
Fire behaviour		
Fire rate of spread ($m\,min^{-1}$)	8·8 (0·5)	1·7 (0·1)
Fireline intensity ($kW\,m^{-1}$)	4003·2 (394·1)	846·3 (5·1)
Flame height (m)	5·5 (0·5)	3·5 (0·5)

(Standard error in brackets).

Vegetation Recovery After Fire

Immediately after burns there was a drastic shrub-cover reduction (Table IV). A quick recovery of vegetation cover was observed during the first year after treatment. *Ulex europaeus* sprouts, *Agrostis* sp. and *Pteridium aquilinum* were mostly responsible for this rapid restoration of cover. Nine months after burning no significant differences were found between treatment levels in vegetation cover.

The bare-soil percentage immediately after fires was not very high, between 14 and 31 per cent (Table IV), in both burn treatments, this difference being not significant ($P = 0{\cdot}0787$).

The reduction in litter depth after intense burn was 70·6 per cent, a value significantly greater than the 56 per cent measured after the light burn (Table IV). However, the differences in the remaining litter depth between burning treatments only became significant from six months onwards after the fires.

Throughfall, Runoff and Erosion Effects

Precipitation during the study period (October 1988–October 1989) was 1412 mm (Table V). This value is 22·6 per cent lower than the annual mean for this area (1824 mm). Thirty minutes maximum rainfall intensity resulted; low and homogeneously distributed along the year as is common under oceanic climate. The annual rainfall erosivity factor (Wischmeier and Smith, 1978) was 105 $J\,m^{-2}\,cm\,h^{-1}$ (Table V), relatively low during that year if compared with the mean value of 250 $J\,m^{-2}\,cm\,h^{-1}$ established for this area (ICONA, 1988).

Table III. Temperature regime during burns

Position	Intense burn	Light burn
	Maximum temperatures (°C)	
Litter–humus interface	463 (115)	94 (54)
Mineral soil surface	73 (30)	37 (8)
−2·5 cm below mineral soil surface	32 (4)	27 (1)
	Duration of temperatures > 200°C (minutes)	
Litter–humus interface	3·8 (1·6)	0·5 (0·4)
	Duration of temperatures > 100°C (minutes)	
Mineral soil surface	3·1 (3·1)	0·0 (0·0)

(Standard error in brackets).

Table IV. Percentage of soil covered by vegetation, bare soil and litter layer depth immediately before burns and in different moments after burnings

	Before burn	Immediately after burn	Months after burn			
			3	6	9	12
			Vegetation cover (%)			
Intense burn	94 (3·1)	43 a (2·6)	51 a (2·8)	63 a (3·1)	73 a (3·2)	82 a (4·0)
Light burn	91 (3·8)	37 a (0·4)	48 a (1·5)	65 a (6·8)	75 a (6·3)	83 a (6·7)
Control	92 (5·6)	92 b (5·4)	91 b (5·8)	90 b (5·2)	93 a (5·7)	93 a (5·5)
			Bare soil (%)			
Intense burn	0	31 a (4·5)	25 a (6·0)	21 a (5·5)	13 a (4·5)	10 a (3·5)
Light burn	0	14 a (2·0)	12 a (2·0)	8 a (0·0)	7 a (1·5)	6 a (0·5)
			Litter depth (cm)			
Intense burn	3·4 (0·10)	1·0* a (0·10)	0·7 a (0·10)	0·6 a (0·10)	0·5 a (0·10)	0·5 a (0·10)
Light burn	3·9 (0·10)	1·7* a (0·10)	1·5 a (0·10)	1·5 b (0·10)	1·4 b (0·10)	1·3 b (0·10)
Control	3·8 (0·15)	3·8 b (0·30)	3·7 b (0·30)	3·7 c (0·30)	3·7 c (0·30)	3·7 c (0·25)

(Standard error in brackets).
Values followed by the same letter in the same column, means did not differ statistically ($p < 0{\cdot}05$).
For litter layer depth data. Values followed by an asterisk, immediately post-fire and pre-fire values differed statistically ($p < 0{\cdot}05$).

Fire significantly affected the accumulated mean throughfall during the first year after treatment (Table VI). Intensely and lightly burned soils received 306 and 223 mm of annual rainfall more than the control, respectively. The differences between the throughfall in light burn and control (Figure 1a) had disappeared in nine months after burning (April–June), while differences between the intense burn and the control were still evident at the end of the study. No significant differences between burning treatments were found in the accumulated mean throughfall throughout the study period. By season (Figure 1a), they were evident in the periods January–March and April–June.

Table V. Rainfall, mean rainfall kinetic energy, thirty-minutes maximum rainfall intensity, and rainfall erosivity factor during each period considered

	October–December	January–March	April–June	July–October	Annual
Rainfall (mm)	521	425	318	148	1412
Mean accumulated rainfall kinetic energy ($J\,m^{-2}$)	1158 (139–1824)	1027 (334–2030)	403 (62–1400)	61 (61–97)	685 (61–2030)
Mean 30-minutes maximum rainfall intensity ($mm\,h^{-1}$)	2·23 (0·76–3·14)	1·85 (0·98–3·14)	1·60 (0·30–4·76)	0·47 (0·46–0·74)	1·67 (0·30–4·76)
Accumulated 30-minutes maximum rainfall intensity ($mm\,h^{-1}$)	11.16	8.20	11.44	4.60	—
Rainfall erosivity ($J\,m^{-2}\,cm\,h^{-1}$)	45	28	31	1	105

(Range of variation in brackets).

Table VI. Mean throughfall, runoff and soil losses in different levels of treatment for the first year following burning

	Intense burn	Light burn	Control
Throughfall (mm)	1224 a (10)	1141 a (8)	918 b (56)
Runoff (mm)	89 a (5)	59 b (1)	35 c (3)
Erosion ($kg\,ha^{-1}$)	560 a (71)	222 b (28)	96 b (15)

(Standard error in brackets).
Values followed by the same letter in the same row, means did not differ statistically ($p < 0{\cdot}05$).

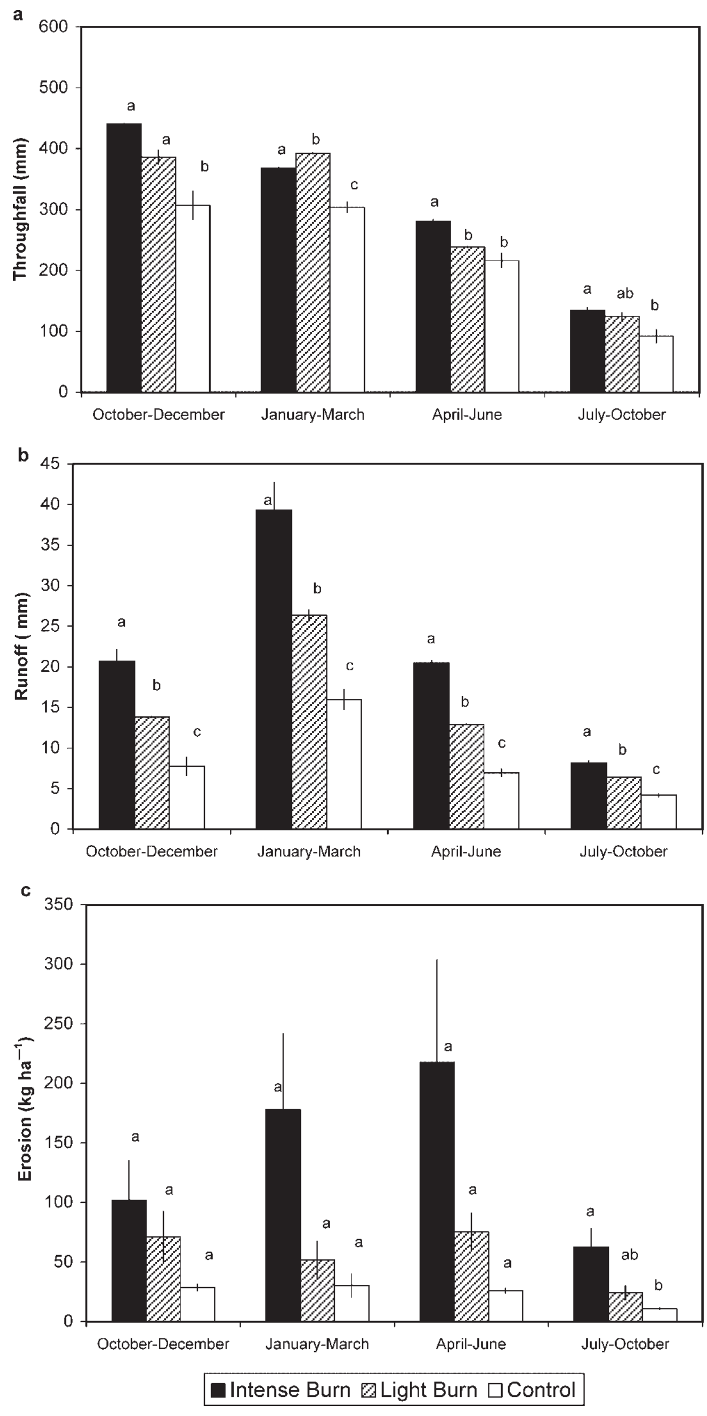

Figure 1. Mean throughfall (a), runoff (b) and soil erosion losses (c) for burned treatments and unburned control. Means topped with the same letter within evaluation dates are not significantly different ($p > 0{\cdot}05$). Vertical bars, standard error.

Mean accumulated runoff throughout the study in the intense burn (89 mm) was significantly ($P = 0{\cdot}0015$) higher than the 35 mm measured in the control (Table VI). Annual runoff in lightly burned plots (59 mm) was also significantly greater than in the control ($P = 0{\cdot}0142$). The mean annual runoff generated was 2·5- and 1·7-times greater than the control for intense and light burn, respectively. In turn, measured runoff in the light burn was statistically lower than in the intense burn ($P = 0{\cdot}0088$). The higher responses in runoff were detected in winter (January–March) in both burning treatments and in the control (Figure 1b).

Accumulated soil losses measured during the study (Table VI) were low in all treatments, but especially so in control plots. Nevertheless, intense-burn treatment resulted in the greatest soil erosion loss which was 2·5-times higher than those measured after light burn and 5·8-times greater than the control. Light burn resulted in 2·3-times more erosion than the control, although these results did not differ statistically ($P = 0{\cdot}1406$). Erosion distribution by periods (Figure 1c) showed a slightly different pattern than that observed in runoff, with its maximum in April–June. In that season, the absolute and relative differences between levels of treatment reached their highest values.

Variables Influencing Throughfall, Runoff and Erosion Responses

A significant and positive linear relationship between the mean throughfall, in each measurement period, throughout the first year ($n = 22$), and the precipitation was observed (Figure 2a) for the intense burn, light burn and control. A significant difference between the slope of both burned relationships and the control was also found.

Runoff for each measurement period was significantly correlated (Figure 2b) with the accumulated rainfall kinetic energy in the same period for all treatments. However, the slopes of the different relationships did differ statistically, the highest value being that of the intense burn followed by the light burn and the control. Similar relationships were obtained with the precipitation for intense burn ($r = 0{\cdot}84$; $n = 22$; $SE = 3{\cdot}04$; $P < 0{\cdot}001$), light burn ($r = 0{\cdot}82$; $n = 22$; $SE = 2{\cdot}11$; $P < 0{\cdot}001$) and control ($r = 0{\cdot}81$; $n = 22$; $SE = 1{\cdot}24$; $P < 0{\cdot}001$) but, in this case, the slopes of the relationships for intense and light burns did not differ.

Stepwise multiple regression analysis (Table VII) revealed the negative influence of litter depth in runoff generation, which explained between 5·0 and 7·3 per cent of runoff variability. This contribution was more pronounced for the intense burn but was still apparent for light burns and the control. At the plot scale, runoff accumulated at the end of the study was significantly correlated with the mean value of litter depth the first year after burning ($r^2 = 0{\cdot}97$; $n = 6$; $SE = 4{\cdot}62$; $P < 0{\cdot}005$) and the mean percentage of bare soil during that time ($r^2 = 0{\cdot}90$; $n = 6$; $SE = 6{\cdot}10$; $P < 0{\cdot}02$). Vegetation cover showed a lower correlation coefficient with runoff ($r^2 = 0{\cdot}74$; $n = 6$; $SE = 0{\cdot}25$; $P < 0{\cdot}10$).

Soil-erosion losses for intensely burnt, lightly burnt and the control were significantly correlated (Figure 2c) with the accumulated thirty minutes maximum rainfall intensity for each measurement period. The slopes of the different relationships did differ statistically. The same occurred with precipitation for intense burn ($r = 0{\cdot}95$; $n = 8$; $SE = 28{\cdot}40$; $P < 0{\cdot}001$), light burn ($r = 0{\cdot}93$; $n = 8$; $SE = 9{\cdot}52$; $P < 0{\cdot}001$) and control ($r = 0{\cdot}94$; $n = 8$; $SE = 3{\cdot}17$; $P < 0{\cdot}001$). Accumulated rainfall kinetic energy was also significantly related to erosion for intense burn ($r = 0{\cdot}85$; $n = 8$; $SE = 46{\cdot}64$; $P < 0{\cdot}01$), light burn ($r = 0{\cdot}79$; $n = 8$; $SE = 15{\cdot}91$; $P < 0{\cdot}05$) and control ($r = 0{\cdot}89$; $n = 8$; $SE = 4{\cdot}39$; $P < 0{\cdot}005$). In this case, the slopes of the relationships for intense and light burns did not differ.

At the plot scale, total erosion at the end of the study was negatively and significantly correlated (Figure 3) with the mean value of the percentage of bare soil for the first year after burning. Mean litter thickness during that time showed also a clear negative influence on erosion ($r^2 = 0{\cdot}90$; $n = 6$; $SE = 76{\cdot}89$; $P < 0{\cdot}05$), which was greater than vegetation cover effect ($r^2 = 0{\cdot}67$; $n = 6$; $SE = 0{\cdot}52$; $P < 0{\cdot}10$).

DISCUSSION

In this study an annual throughfall value of 65 per cent of rainfall was measured in control plots, which is slightly higher than percentages of from 35 per cent to 59 per cent found in other studies in gorse shrublands (Aldridge,

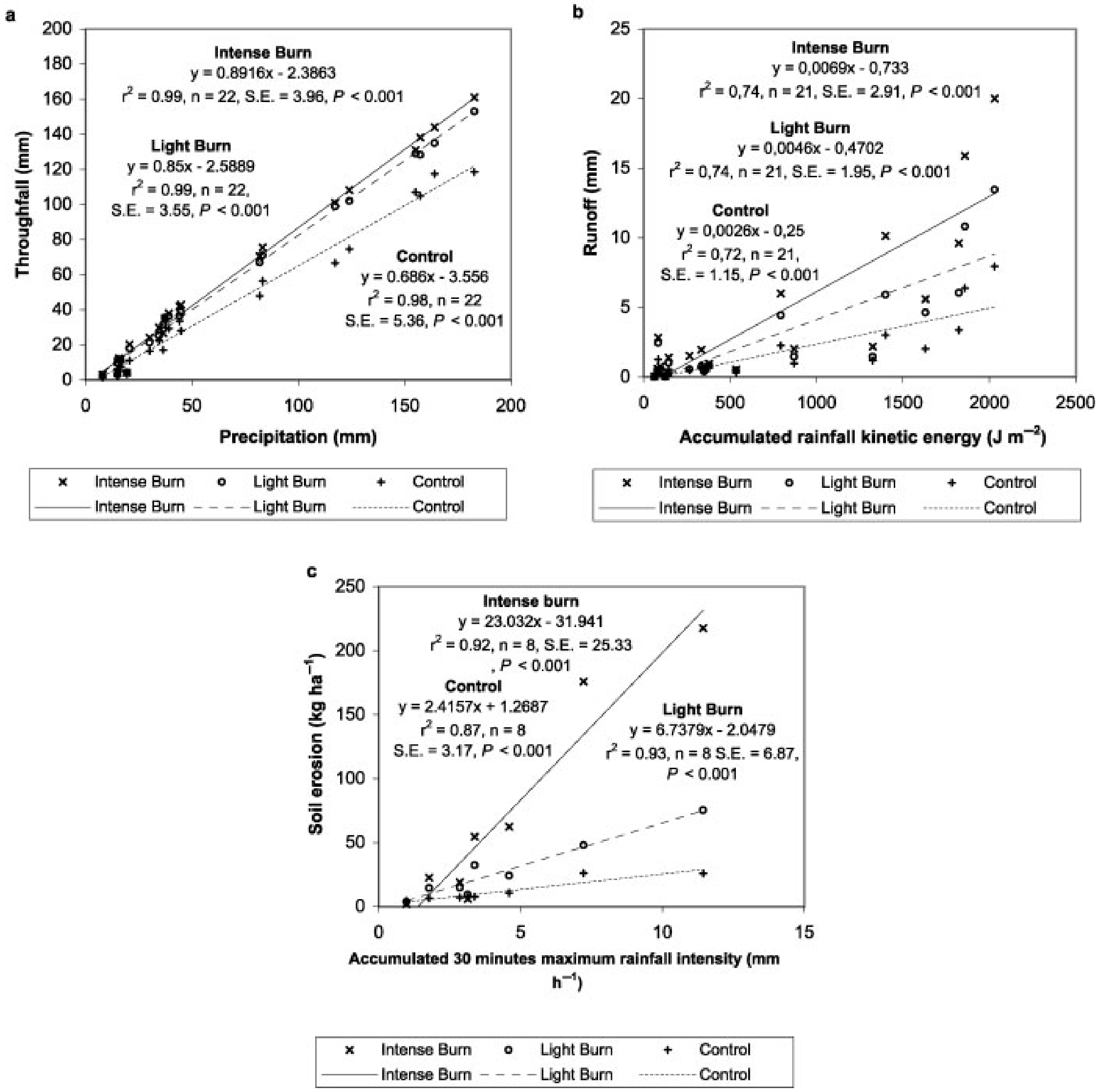

Figure 2. Statistical relationships between: (a) mean throughfall in each measurement period in burned treatments and unburned controls and precipitation; (b) mean runoff in each measurement period in burned treatments and unburned controls and accumulated rainfall kinetic energy; and (c) mean soil erosion in each measurement period in burned treatments and unburned controls and accumulated 30 minutes' maximum rainfall intensity. *SE* = standard error.

1968; Egunjobi, 1971; Calvo de Anta *et al*., 1979; Soto *et al*., 1993; Soto and Díaz-Fierros, 1997). This difference may be due to differences in accumulated biomass or in the degree of shrub overlayering.

Annual throughfall represented 87 per cent and 81 per cent of annual rainfall in the intensely burned and lightly burned plots, respectively. These figures were very similar to the percentages (88 per cent and 89 per cent) reported by Soto *et al*. (1993) and Soto and Díaz-Fierros (1997) after burning in a *Ulex europaeus* shrubland in Galicia. These accumulated values were very similar to the slope in the regression equations (Figure 2a) between throughfall and precipitation. The absence of significant differences between the slopes of regression lines for burning treatments, was probably due to the comparable shrub fuel consumption observed in both treatments. The negative intercept on the throughfall axis in those relationships approximately represents the canopy storage

Table VII. Predictive equations produced from stepwise regression for runoff (mm). Data for every sampling period during the first year after burns

Evaluation data	Regression equation	Adjusted coefficient of determination	Equation
Intense burn ($n = 21$)	Runoff = 5·80 + 0·008 rainfall kinetic energy ($J\,m^{-2}$) − 11·63 litter layer depth (cm) ($SE = 2{\cdot}54$)	0·79	I
Light burn ($n = 21$)	Runoff = 12·05 + 0·005 rainfall kinetic energy ($J\,m^{-2}$) − 8·89 litter layer depth (cm) ($SE = 1{\cdot}80$)	0·76	II
Control ($n = 21$)	Runoff = 42·56 + 0·003 rainfall kinetic energy ($J\,m^{-2}$) − 11·59 litter layer depth (cm) ($SE = 1{\cdot}06$)	0·75	III

All equations are significant at $p < 0{\cdot}05$.

capacity (Bruijnzeel and Wiersum, 1987). In our case, the measured values ranged from 2·4 mm and 2·6 mm for burned shrubland to 3·6 mm for the control. The lack of significant differences in the vegetation cover between light and intense burns could explain the similarities in the canopy storage capacities. Soto and Díaz-Fierros (1997) obtained similar relationships between throughfall and precipitation and values of the intercept coefficients of 2·5 mm in the unburned control, and between 1·2 mm and 1·3 mm in burned plots. The influence of differences in vegetation cover and recovery patterns might explain these minor differences.

Runoff was low in all the treatments. The ratios between runoff measured in burned and control plots were generally lower than those measured in other experiments, especially under a Mediterranean climate (Table VIII). The remaining litter depth after burning modulated the different responses of runoff to rainfall kinetic energy found for intense and light burns. This effect was even significant in control plots. The importance of the litter + duff depth on infiltration agrees with the observations of Morales *et al*. (2000) after prescribed burning in *Pinus. arizonica*.

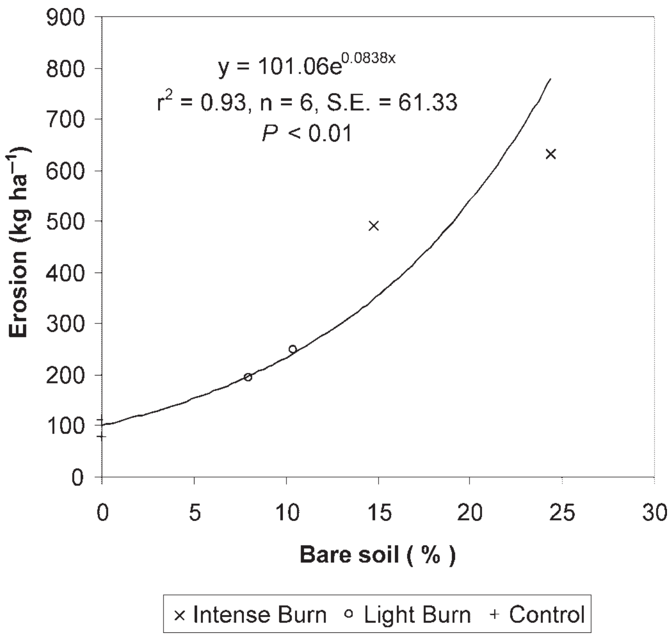

Figure 3. Statistical relationship between mean annual percentage of bare soil in burned treatments and unburned controls and soil erosion losses. SE = standard error.

Table VIII. Sediment yields and runoff from prescribed burn studies in shrublands and grasslands

Location	Dominant species	Soil texture	Rainfall (mm)	Burned condition	Sediment yield (kg ha^{-1})			Runoff (mm)			Slope (%)	Time lapse after fire	Reference
					Burned	Control	Burned/control	Burned	Control	Burned/control			
Arizona (USA)	*Q. turbinella* *C. montanus*	Coarse textured		Repeated burnings for 5 years	12170	0		—	—	—	30	5 years	Pase and Lindemuth (1971)
Nevada (USA)	Pinyon–juniper	Loamy, mixed	83·8 mm h^{-1} for 1 h	Prescribed shrub coppice recently burned	630–1096	397–813	1.6–0.74	—	—	—	5–8	1–2 months	Roundy *et al*. (1978)
Scotland (UK)	*Calluna vulgaris*	Iron-pan	760–890		1270–3850	0		—	—	—	5–31	18–24 months	Kinako and Gimigham (1981)
California (USA)	Chamise, manzanita	Coarse textured	559	Prescribed fires	32420	119	272	36	1	36	35–50	1 year	Wells (1981)
Texas (USA)	whitebrush	Sandy-loam	230 mm h^{-1} for 30 min.	Herbicide prescribed burn	615	730		—	—	—		10 months	Knight *et al*. (1983)
	Mesquite	Clay-loam		Herbicide prescribed burn	1160	1955		—	—	—			
				Prescribed burn	1300	1955							
Texas (USA)	Brownseed paspalum/little bluestem	Clay-pan	1220	Winter burns	255	340		21	29·2		3–6.2	15 months	Garza and Blackburn (1985)
				Spring burns	220	340		27	28				
	Post-oak/yaupon brownseed paspalum	Sandy-loam		Winter burns	380	340	1·1	15·1	24				
				Spring burns	285	340		21	24				
Coruña (Spain)	*Ulex europaeus*	Sandy-loam	1071	Light intensity	1974	592	3·3	64·3	37·5	1·7	30	1 year	Díaz-Fierros *et al*. (1990) Soto *et al*. (1993, 1994)
				Moderate intensity	1649		2·8	55·7		1·5			
Arizona (USA)	Introduced grass	Gravely loam	55 mm h^{-1} . for 45 min. + 110 mm h^{-1} for 15 min.	Low intensity	249	8	31	15·4	0·9	17	5–6	Same day	Emmerich and Cox (1994)
				Repeated burning after a year	534	34	16	18·7	2·1	9			
Arizona (USA)	Native grass	Gravely sandy-loam	55 mm h^{-1} for 45 min. + 110 mm h^{-1} for 15 min.	Low intensity	389	76	5	19·1	5·6	3	6–7	Same day	
				Repeated burning after a year	642	154	4	22·2	9·3	2			
Alicante (Spain)	*Brachypodium retusum*	Silty-loam	196·4	*P. halepensis* afforestation + terraces + low intensity fire	479·7			8·54			26	1 year	Sánchez *et al*. (1994)
				P. halepensis afforestation + linear ploughing + low intensity fire	250·5			6·71					
				Low intensity fire	896.3			19·59					
Málaga (Spain)	*Ulex parviflorus*	Gravely sandy-loam	92	Medium intensity fire	667	42	16				73	95 days	Carreira and Niell (1995)
				Clipping + fire	790		19						
Texas	Live oak	Silty-clay	203 mm h^{-1}	Prescribed low severity	4500	2	2250	—	—	—	4	Immediately after fire	Hester *et al*. (1997)
(USA)	juniper		for 50 min.		1926	34	57	—	—	—			
	bunchgrass				4463	300	15	—	—	—			
	shortgrass				5766	1299	4	—	—	—			

Continues

Table VIII. Continued.

Location	Dominant species	Soil texture	Rainfall (mm)	Burned condition	Sediment yield (kg ha^{-1})			Runoff (mm)			Slope (%)	Time lapse after fire	Reference
					Burned	Control	Burned/control	Burned	Control	Burned/control			
Alicante (Spain)	*Ulex parviflorus*	Loamy	466	Moderate to high severity	1540–2508						40–47	13 months	García-Cano *et al*. (2000)
Valencia (Spain)	*R. officinalis*	Sandy-loam	425	High intensity	4100	100	41	15·4	2·1	7·3	—	1·5 years	Gimeno *et al*. (2000)
	U. parviflorus			Moderate intensity	3300		33	12·1		5·8			
León (Spain)	*Erica australis*	Loamy-sand	180 mm h^{-1} for 5 min.	Low severity	103·2	11·8	8·7	5·1	1·4	3·6	10	1·5 years	Marcos *et al*. (2000)
Arizona (USA)	Bunchgrasses	Loamy	63·5 mm h^{-1} for 40 min.	Low intensity	—	—	—	13·8	9·3	1·5	1–3	3 months during 2 years	O'Dea and Guertin (2003)
			100 mm*		28 000	17 000	1·6	6·1	5·8	1·1			
			106 mm		22 000	18 000	1·2	15·1	14·8	1·0			
Alicante (Spain)	*Ulex parviflorus*	Loamy	2·6 mm min^{-1} for 105 min.	Prescribed high severity	8420	20	421	—	—	—	49	2 months	De Luis *et al*. (2003)
				Moderate severity	300	50	6						
Pontevedra (Spain)	*Ulex europaeus*	Sandy-loam	1412	Intense burn	560	96	5·8	89	35	2·5	30	1 year	This study
				Light burn	222		2·3	59		1·7			

*Sediments from simulation + natural rainfall of 100 mm.

In this study, maximum temperatures recorded at the mineral soil surface and 2 cm under the soil surface in both types of burn were too low to destroy the soil organic matter or fine root (Giovannini *et al*., 1990; De Bano *et al*., 1998). Consequently, changes in soil bulk density and porosity affecting runoff and erosion seemed unlikely. Neither did the formation of a hydrophobic layer seem likely, since higher temperatures than those measured in our study have been reported to result in its formation (De Bano, 1981; De Bano *et al*., 1998).

In our study, soil erosion in burned sites were in the lower range of soil losses reported in other experiments after gorse or other shrubland burnings (Table VIII). This would seem to suggest that prescribed burning in shrubs could be used with an acceptable level of soil disturbance. Nevertheless, the applicability of our results should be taken with caution because rainfall during the study year was low and there were no rainfall events of high erosive power.

The differences in soil losses observed between intense and light burns agrees with the results from De Luis *et al*. (2003), although our values are clearly lower. However, these differences have not been found in some other similar studies (Soto *et al*., 1994; Gimeno *et al*., 2000).

Maximum soil losses did not occur in the first months following burns (Figure 2c), when the percentage of bare soil was greatest and the vegetation cover was least, but during the following spring when the accumulated values for 30 minutes' maximum rainfall intensity were higher (see Table V). This variable showed the best correlation coefficient with erosion rates. Note that, in our case, the variable was the accumulated values of 30 minutes' maximum rainfall intensity for each rainfall event within each measurement period instead of the isolated values for each rainfall event. After wildfires, Moody and Martin (2001) found soil loss acceleration with values of 30 minutes' maximum rainfall intensity higher than $10\,\mathrm{mm\,h^{-1}}$. De Luis *et al.* (2003) also measured an increment of erosion with the highest rainfall intensities. In our case, 30 minutes' maximum rainfall intensity was always below $5\,\mathrm{mm\,h^{-1}}$ (see Table V) and it seems unlikely that those storms could exceed the infiltration capacity for these coarse-textured soils even after burning (Díaz-Fierros *et al*., 1990). In fact, a reduction in precipitation of 25 per cent in April–June compared to the previous period produced a reduction in runoff of about 50 per cent (see Figure 1b). This suggests that there was not a diminution in the infiltration capacity of these burned soils. Some other authors (Marcos *et al.*, 2000; Martin and Moody, 2001) have argued that the washing off of ash has caused these increments in runoff and erosion after fires. In our case, the more likely explanation is that during spring, soil moisture content was low due to vegetation flush and evapotranspiration, which could promote a temporary increment of soil detachability or a reduction in soil aggregates stability. This has been observed by other authors for summer storms (Garza and Blackburn, 1985; Bresson and Cadot, 1992; Stolte *et al*., 1997; Johansen *et al*., 2001; O'Dea and Guertin, 2003).

The observed correlations between soil losses and the percentage of bare soil agrees with the well-observed fact that soil organic cover remaining after fire provides adequate protection against soil erosion (e.g. Roundy *et al*., 1978; Emmerich and Cox, 1994; Hester *et al*., 1997; García-Cano *et al*., 2000; Marcos *et al*., 2000; Johansen *et al*., 2001; Pierson *et al*., 2001; De Luis *et al*., 2003). However, we do not know of any other previous experiments specifically quantifying the contribution of litter depth to soil erosion.

The correlations obtained, at the plot scale, showed the stronger influence of the remaining litter depth rather than the vegetation cover on soil losses. This suggests that, in the future, refined prescribed burning techniques should pay particular attention to the litter consumption during burns.

From the management point of view, runoff and soil erosion losses from the burned sites were low although different fire prescriptions did have significant impacts on runoff and sediment production. The prescription key variables controlling fire severity were the ignition technique and, most importantly, the moisture content of the litter just before ignition. It has been found (Vega *et al*., 2001) that this variable accounts for a high percentage of litter consumption during prescribed burning in Galician gorse shrub communities. The very strong influence of litter cover on soil erosion, indicates that in the operational use of prescribed burning in shrublands for fuel reduction, at least 80 per cent of soil should remain protected by a litter cover after the fire. The quick herbaceous and shrub recovery also had a positive effect on post-burn erosion and this also suggests a new burning application would be necessary for fire-hazard reduction in the short term. More research on these aspects is neccesary.

CONCLUSIONS

Prescribed burning of different intensity in gorse shrubland significantly increased (29 per cent) throughfall compared to the unburned control. However, throughfall during the first year after prescribed burning was similar in both burning treatments.

Fire intensity significantly affected annual runoff and erosion. At the plot level, annual runoff was strongly correlated with litter thickness and, to a lesser degree, with the percentage of bare soil and vegetation cover. The rainfall variable best correlated with runoff was rainfall kinetic energy. This influence was higher in the intensely burned shrubs than in lightly burned, followed by the control.

Soil losses during the first year after burning in the intense burn were 5·8-times higher than in the control and 2·5-times larger than in the light burn. Soil erosion losses after light burn was 2·3-times higher than in the control although these figures did not statistically differ. Accumulated maximum rainfall intensity in 30 minutes was the rainfall variable more correlated with soil erosion losses by measurement period. Percentage of bare soil explained 93 per cent of annual soil loss variability at the plot level. Shrub cover reduction seemed less critical for erosion than the remaining soil cover, explaining 67 per cent of its variability.

This study showed that differences in fire prescription can have large effects on runoff and sediment production. Litter moisture content seemed to be a critical variable to control litter consumption and subsequent runoff and erosion.

Apparently, from our results prescribed burning could be a feasible management tool both to reduce wildfire hazard and for soil conservation purposes, provided that the soil organic layer is not strongly reduced by burning. However, this conclusion must be taken with caution due to the dry conditions of the first post-fire year in this study. More research, including a broader range of conditions, is needed to clarify this point.

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