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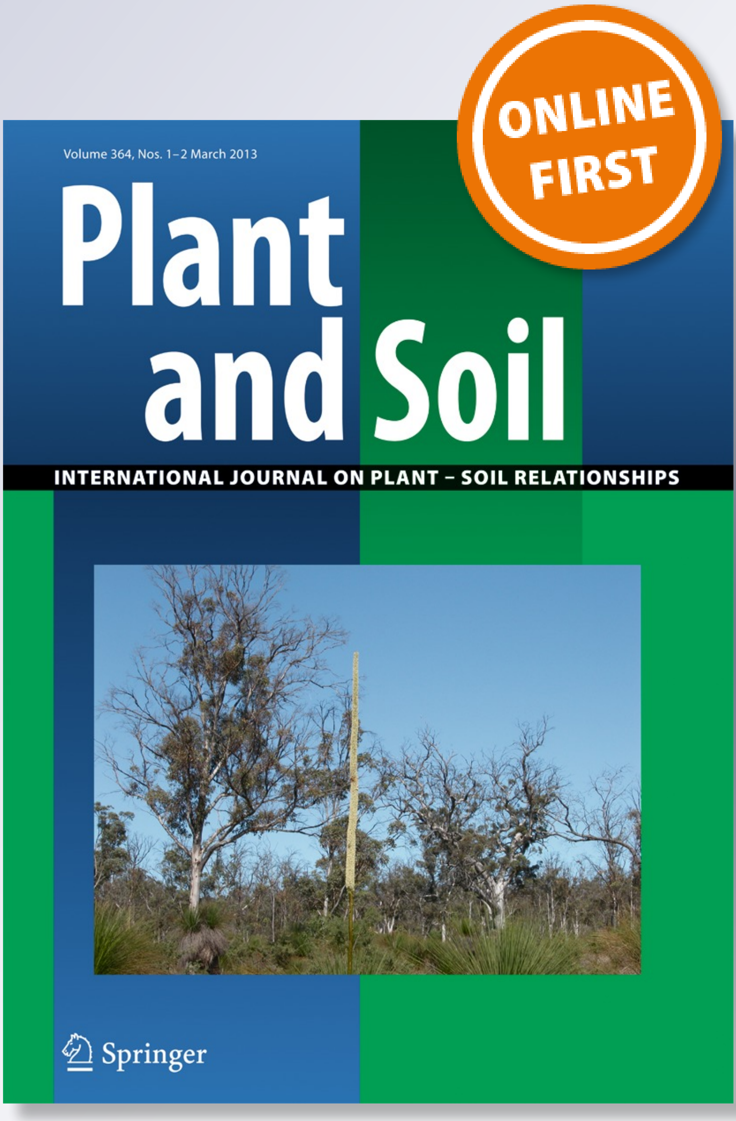

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REGULAR ARTICLE

Runoff and erosion from volcanic soils affected by fire: the case of *Austrocedrus chilensis* forests in Patagonia, Argentina

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Abstract
Aims We characterized the runoff and erosion from a volcanic soil in an *Austrocedrus chilensis* forest affected by a wildfire, and we evaluated the effects of a mitigation treatment.
Methods Rainfall simulations were performed in the unburned and burned forest, with and without vegetation cover, and under a mitigation treatment.
Results After the wildfire, the mean infiltration rate decreased from 100 mmh^{-1} in unburned soils to 51 and 64 mmh^{-1} in the burned with and without litter and vegetation cover, respectively. The fast establishment of bryophytes accelerated the recovery of soil stability. Sediment production was negligible in the control plots (4.4 gm^{-2}); meanwhile in the burned plots, it was 118.7 gm^{-2} and increased to 1026.1 g m^{-2} in the burned and bare plots. Total C and N losses in the control plots were negligible, while in the burned and bare plots the organic C and total N removed were 98.25 and 1.64 gm^2, respectively. The effect of mitigation treatment was efficient in reducing the runoff, but it did not affect the sediment production.
Conclusions These fertile volcanic soils promoted the recovery of vegetation in a short time after the wildfire, diminishing the risk of erosion.

Keywords *Austrocedrus chilensis* · Erosion · Mitigation · Volcanic soils · Wildfire

Introduction

Wildfires have been a serious problem in the Patagonian Andean region, causing significant environmental, social and economic impacts (Veblen et al. 1999; Kitzberger

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and Veblen 1999) *Austrocedrus chilensis* (D. Don) Pic. Sern and Bizarri is an endemic conifer of the north Patagonian forests and it is one of the most affected by fire (Loguercio et al. 1999). It is found between 37°07′ and 43°44′ S latitude (Pastorino et al. 2006), covering an area of 141,000 ha (Bran et al. 2002).

Wildfires, especially those of high severity, may increase runoff and accelerate erosion processes, producing serious impacts on surface water quality. The loss of vegetation and litter layers decreases rainfall interception leaving the soil exposed to the direct impact of raindrops, modifying soil properties related to infiltration and sediment transport (Neary et al. 1999; Robichaud 2000; Martin and Moody 2001).

The decrease in infiltration may also be favored by the formation of a water-repellent layer in the soil. Soil water repellency is a naturally occurring phenomenon that can be intensified by soil heating during fire (DeBano 1990). Thus, the loss of surface cover and the decrease in infiltration rates increase surface runoff and the risk of soil erosion, which can have an impact on soil and affect surface water quality (Neary et al. 1999).

The increase in sediment production depends on fire frequency, burn severity, weather, vegetation, geology and type of soil. In some regions, more than 60 % of the total sediment production over the long term is fire-related (Robichaud 2000). These increases can occur in the first year after the fire, depending on the intensity of rainfall and on the evolution of vegetation cover (DeBano et al. 1998).

Several studies have used simulated rainfall for exploring the impact of rainfall on the response of soil erosion (Rostagno 1989; Cerdà et al. 1997; Benavides-Solorio and MacDonald 2001; González-Pelayo et al. 2006). Sediment transport is strongly related to rainfall and runoff characteristics which determine the selectivity of the erosion process (Shakesby 2011). Sediment may contain a higher nutrient concentration than the original soil (Sharpley 1985). Thus, water erosion represents a selective process that may remove soil colloids and nutrients, reducing soil fertility and productivity (Sharpley 1985; Adema et al. 2001). However, few studies have assessed the impact of post-wildfire erosion on soil fertility, through quantification of soil nutrient depletion resulting from single or multiple fire cycles (Shakesby 2011).

Soil erodibility depends mainly on physical and chemical characteristics (i.e. nature and amount of soil aggregates, OM content and particle size distribution). Though the effect of fire on aggregate stability is complex (Mataix-Solera et al. 2011), in some cases the stability of the soil aggregates may be reduced by fire (Jordan et al. 2011), producing more easily eroded soils (Shakesby and Doerr 2006; Shakesby 2011). Clay dispersion ratio, a measure of aggregate stability, has been used to assess the erodibility of different soils (Middleton 1930; Kumar and Singh 2007; Singh and Khera 2008). Clay dispersion can also contribute to the formation of structural soil seals due to pore clogging by fine particles (Assouline 2004).

In the Patagonian Andean region, the predominant soils are derived from volcanic deposits. These soils, characterized by the presence of allophane, present a high capacity to stabilize OM, buffer pH and have a high water retention capacity (Irisarri and Mendía 1997). Thus, the impacts of wildfire on these soils could be expected to be less marked than in other soils. However, some studies in the burned native forests of Patagonia showed explicitly the nature of the changes in the physical and chemical properties of these soils (Alauzis et al. 2004; Kitzberger et al. 2005; Urretavizcaya 2010; La Manna and Barroetaveña 2011).

The effects of fire on the hydrological behavior of the soils can last for weeks or decades depending on burn severity, vegetation recovery rate and mitigation treatments. Implementation of mitigation treatments to control soil erosion can be necessary to either protect the soil itself or avoid the contamination of surface waters. The most frequent treatments include straw wattles, mulching that improves soil condition (i.e. increased moisture, OM) and contour log terraces that provide a barrier to runoff from heavy rainstorms (Wagenbrenner et al. 2006; Myronidis et al. 2010). In the Patagonian region, rehabilitation treatments are rarely implemented in burned areas. Varela et al. (2006) studied the post-fire soil rehabilitation with biosolids compost and they found that this application enhanced seedling abundance and species richness. However, little information is available about the rehabilitation of burned areas and the efficiency of different treatments to control soil erosion. The objectives of this study were: i) To characterize the runoff and erosion from a volcanic soil in an *A. chilensis* forest affected by wildfire, and ii) To evaluate the runoff and erosion after a mitigation treatment.

Materials and methods

Study area

The study area is located in the north-west of Patagonia, 15 km SW of Esquel city, near to Los Alerces National Park (42°58′40.36″S, 71°30′53.04″W) and to the north of the Terraplen Lagoon (Fig. 1). The study area was affected by an accidental fire that occurred on February 2008 and burned approximately 6,000 ha, mostly of *A. chilensis* forest (CIEFAP et al. 2008).

Average annual temperature in the study area is around 8 °C and annual precipitation is approximately 1,000 mm, concentrated mainly in autumn and winter (Cordon et al. 1993).

The relief is characterized by glacial topography. The Terraplen Lagoon is a shallow depression located in the terminal portion of a glacial valley. The Lagoon is bounded by deltaic sediments from different ages. The north limit of the Lagoon is comprised of mountains of volcanic rocks eroded by glaciers (Martinez personal communication).

This landscape is covered by volcanic ash, as is the rest of the Andean Patagonian region (Irisarri and Mendía 1997). Soils are derived from these volcanic deposits and are classified as Humic Udivitrands (Soil Survey Staff 1999).

The positive Fieldes's test suggested the presence of allophane, an amorphous silicate colloid (Fieldes and Perrot 1966). Allophanic soils present a high fertility and forest suitability as they have high OM contents, cation exchange capacity and water retention capacity and low bulk density (Irisarri and Mendia 1997). The vegetation is a forest dominated by *A. chilensis* in the canopy layer; the understory vegetation consists of *Maytenus chubutensis* (Speg.) Lourt., O'Don. et Sleum., *Lomatia hirsuta* (Lam.) Diels ex J.F. Macbr., *Schinus patagonicus* (Phil.) I. M. Johnst. and *Aristotelia maqui* (Mol.) Stuntz.

Rainfall simulations experiment in burned and bare soils

Infiltration and sediment production

In December 2009 (Fig. 2) a severely burned site and an unburned (control) forest patch, adjacent to the burned area, were selected on an 80 m steep slope in the northern rim of the Terraplen Lagoon. In the intermediate portion of the slope, approximately 30 m above the level of the Terraplen Lagoon we defined two 210 m^2 plots (14 m × 15 m), one in the burned and one in the unburned area, where the following treatments were considered: 1. The burned forest treatment (BF), that represents the condition of the burned area 22 months after the fire, when simulated rainfall was

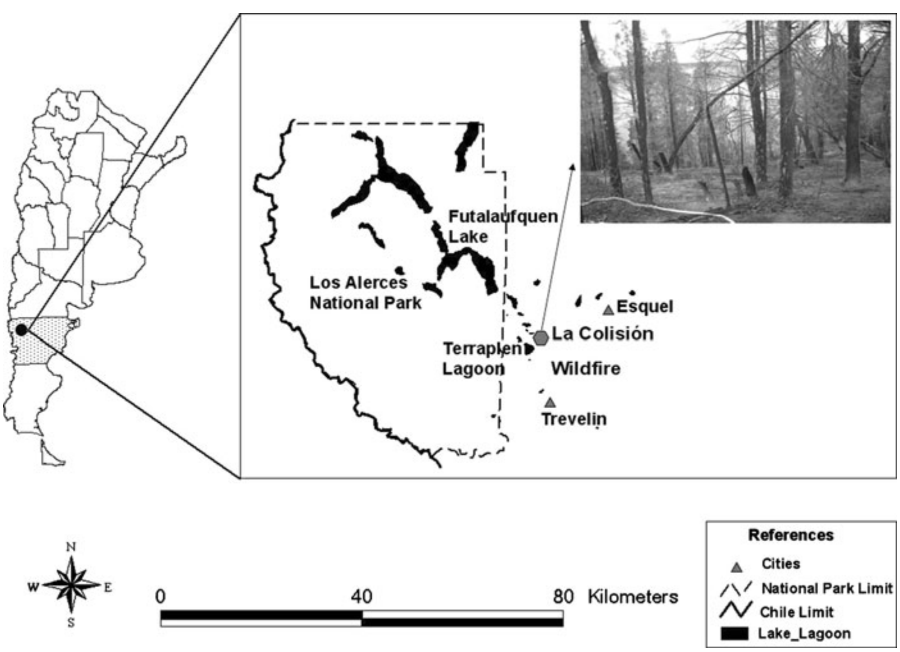


Fig. 1 Location of the study area (La Colisión Wildfire) in Chubut province, Argentina

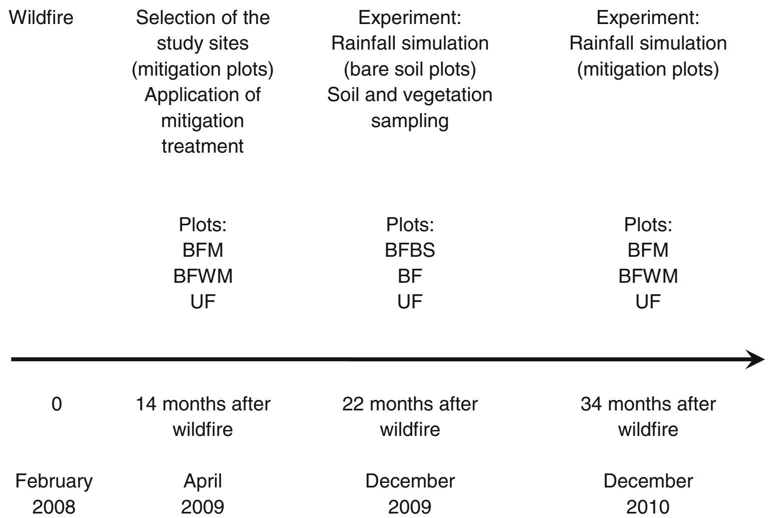


Fig. 2 Temporal scheme of sampling

applied and 2. The burned forest and bare soil (BFBS). This treatment was intended to represent the soil immediately after the fire. Thus, litter, vegetation cover, mainly bryophytes and herbaceous plants, were removed picking by hand, leaving the soil completely bare. 3. A third treatment or control that was defined is the unburned forest (UF).

In each treatment, four 1 m×2 m plots were randomly established. Each plot was delimited by a 10 cm high sheet metal frame, which was buried 5 cm into the soil. The major axes of the plots were oriented in the direction of the dominant slope. The slopes of the plots were determined by means of a level.

In each treatment, rain was applied using a rainfall simulator with a full cone nozzle, similar to the one described by Rostagno and Garayzar (1995) and runoff and sediment production were determined. The applied rainfall intensity was 100 $mm h^{-1}$ for 30 min; its kinetic energy was 18.6 $J m^{-2} mm^{-1}$ and represented 66 % of the kinetic energy of a natural rainfall with the same intensity (Rostagno and Garayzar 1995).

Runoff leaving the lower border of the plots was collected with a u-shaped collector provided with a cover to protect it from the rainfall and a tube that delivered the runoff to 5 L containers. Runoff was collected at 5 min intervals in separate containers and the volume was determined. Infiltration was defined as the difference between total water applied during a given time period and total runoff collected during the same period. Initiation of runoff was defined as the time when measurable runoff occurred from the plot.

Soil and vegetation sampling

Soil samples at the 0–5 cm depth were collected before applying the simulated rainfall on areas adjacent to each plot for determining moisture content (gravimetric method), OM by the loss on ignition method (Davies 1974), total nitrogen (TKN) following the Kjeldahl procedure (Bremner and Mulvaney 1982), texture by the pipette method (Day 1965) and bulk density by the core method (Blake 1965). Soil hydrophobicity was assessed at two depths (0–2.5 cm and 2.5–5 cm) using ethanol with different concentrations, following the methodology described by MacDonald and Huffman (2004); hydrophobicity was classified according to Doerr (1998). The clay dispersion (CD) ratio, a measure of aggregate stability and soil erodibility, was determined according to Middleton (1930) from the relation:

$$\mathrm{CD\%} = (\text{water dispersed clay}/\text{sodium hexamethaphosphate dispersed clay}) \times 100.$$

The water dispersed clay was determined by the pipette method after centrifuging a 20 g soil sample (< 2 mm) in 200 ml distilled water for 5 min at 1,000 rpm. The clay content of the sodium hexamethaphosphate dispersed samples was obtained from the textural analysis as mentioned above.

The litter and vegetation cover (including bryophytes, shrubs and herbaceous plants) were visually estimated by vertical projection in each sample plot before rainfall application.

Sediment production obtained from rainfall simulations was determined from the total runoff collected during each time interval. The sediment production in each plot was obtained by decantation (72 h) in each container. After discarding the overfloating we collected the sediments in 250-ml flask that were dried for 48 h at 60 °C and weighed. The sediments were analyzed for organic carbon (OC) and TKN following the procedures mentioned above. The enrichment ratio (ER) of OC and TKN were calculated by dividing the content of each constituent in the transported sediment by its content in the original soil material (Avnimelech and McHenry 1984). When the sediments are enriched with a given component, as compared to the contributing soils, the ER is greater than unity.

Rainfall simulations experiment in burned soils after mitigation treatment

In April 2009, 14 months after the fire, the mitigation treatment was applied in a severely burned site (Fig. 2). The site was located in the intermediate portion of the slope, in a topographic condition similar to the sampling site previously described.

A 14 m×15 m burned plot was installed, and it was divided into two 7 m×15 m subplots. The mitigation treatment was applied in one of the subplots. Before mitigation, the site presented no litter remaining post-fire, had 93 % of bare soil in the tree interspaces and 60 % tree canopy cover, with all the trees with damaged bark. The treatment consisted of the application of ~42 m^3 of branches and small trunks that covered ~90 % of the soil. Branches and fallen tree trunks chosen for the treatment had diameters lower than 10 cm and 35 cm, respectively. These materials remaining post-fire were easily available and transported, allowing the application of a feasible and at low cost mitigation treatment.

In December 2010, 34 months after the fire, simulated rainfall was applied to four 1 m×2 m plots per treatment to determine the infiltration and sediment production. The treatments were: burned forest with mitigation treatment (BFM), burned forest without mitigation treatment (BFWM) and the unburned forest (UF) as control. The rainfall simulation experiment and soil and vegetation sampling were carried out following the methods previously described.

Data analysis

Data were analyzed by analysis of variance (ANOVA). Assumptions of normality and homoscedasticity were tested using the Shapiro-Wilk and Levene's test, respectively. Alternative non-parametric tests were used (Kruskal and Wallis ANOVA) for variables that did not satisfy these assumptions. Analyses were carried out using InfoStat Statistical Package (Di Rienzo et al. 2010).

Results

Vegetation and soil

Vegetation and soil characteristics before the application of simulated rainfall are shown in Table 1. Twenty-two months after the fire, vegetation cover mainly consisted of bryophytes, with a low percentage of herbaceous and shrubby plants. The most abundant understory species were *Holcus lanatus L, Clarckia tenella* (Cav.) Lewis et. Lewis sbsp. tenella, *Vicia nigricans* Hook., Arn, *Galium aparine* L and *Muelenbeckia hastulata* (Sm.) Johnst. Litter was also an important cover factor in the burned area, although it represented approximately 25 % of the litter cover in the unburned forest. Litter in the BF treatment was represented mainly by a thin and discontinuous layer of recently fallen *A. chilensis* leaves as compared to the 5 cm thick layer in the unburned plots. In unburned plots the dominant shrub was *M. chubutensis*.

The slope inclination varied between 44 and 66 % with an average of 59 %. Bulk density was low in all plots and the lowest values were found in the UF plots. Soils had a sandy loam texture. Soil OM decreased in the burned soils with respect to the UF, from approximately 26 % in the BF to 60 % in the BFBS. The soil moisture was approximately 20 % lower in the burned than in the unburned soils.

The values of the CD ratio were low in the soils of the three treatments, being classified as non-erodible according to Middleton (1930). Hydrophobicity varied greatly, but tended to increase in the deeper layer (2.5–5 cm). The highest values were observed in BFBS plots, but even the control soil was hydrophobic.

Table 1 Means and standard errors of vegetation and edaphic characteristics for each treatment before the application of simulated rainfall

Characteristics	BFBS	BF	UF
Surface cover			
Vegetation (%)	0 (0)	51.25 (±12.24)	37.3 (±3.68)
Herbaceous (%)	0 (0)	6 (±1.96)	1.3 (±1.25)
Shrubby (%)	0 (0)	1 (±1)	35 (±4.56)
Bryophytes cover (%)	0 (0)	44.2 (±12.10)	1.0 (±0.58)
Litter cover (%)	0 (0)	25.75 (±6.10)	98.9 (±0.13)
Soil			
Slope (%)	60 (±0.95)	55 (±4.02)	63 (±1.46)
Bulk density (mgm-3)	0.55 (±0.08)	0.64 (±0.05)	0.40 (±0.03)
Organic matter (%)	9.37 (±1.86)	16.88 (±1.84)	22.91 (±3.24)
Moisture (%)	20.44 (±3.12)	20.82 (±2.18)	41.91 (±4.59)
Clay dispersion (%)	8.71 (±0.77)	8.33 (±1.23)	7.52 (±0.70)
Texture	Sandy loam	Sandy loam	Sandy loam
Hydrophobicity	Not hydrophobic to Extremely hydrophobic	Not hydrophobic to Moderately hydrophobic	Not hydrophobic to very strongly hydrophobic

BFBS burned forest and bare soil; *BF* represent the conditions 1 year after the fire; *UF* unburned forest

Runoff and sediment production

Runoff was an order of magnitude higher in the soils of the burned forest as compared to the unburned. Contrary to what was expected, mean runoff from the burned forest plots (BF) tended to be slightly higher than the runoff generated by the burned forest and bare soil plots (BFBS), although there were no significant differences between these treatments (Table 2).

The mean time to the initiation of runoff showed no significant differences. The values in UF and BF plots were similar, approximately 140 s; in the BFBS plots it was 90 s.

The infiltration rate in the unburned plots was very high, similar to the rainfall intensity during the 30 min period and the runoff from this treatment was very low (Fig. 3). At the end of simulated rainfall, the infiltration rate in the burned plots was lower than in the unburned plots, with a reduction of 35 % in the BFBS and 48 % in the BF. Infiltration rate from the burned plots declined during the first 10–15 min; afterward it was almost constant (Fig. 3).

Table 2 Means and standard errors of runoff and sediment production in soils affected by the wildfire. Different letters indicate significant differences ($p<0.05$)

Treatments	Runoff ($\mathrm{l m^{-2}}$)	Sediment production ($\mathrm{g m^{-2}}$)
BFBS	14.5 (±3.4)b	1026.1 (±268.68)c
BF	18.1 (±0.9)b	118.7 (±36.28)b
UF	1.18 (±0.6)a	4.4 (±2.51)a

BFBS burned forest and bare soil; *BF* represent the conditions 1 year after the fire; *UF* unburned forest

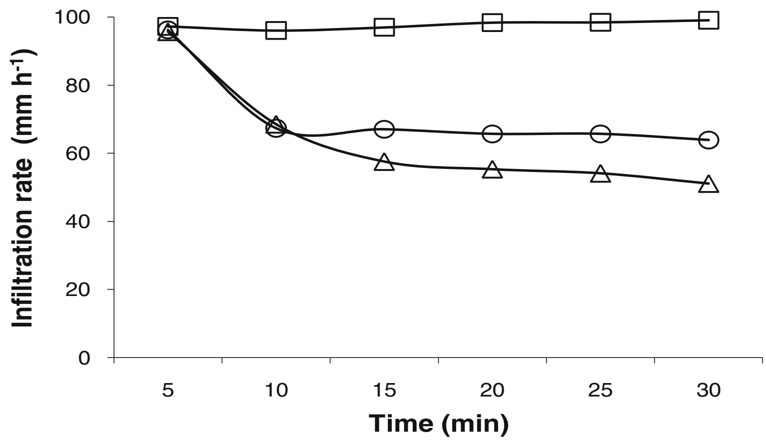
Although the soils would be considered as non-erodible according to the CD ratio, the rainfall simulation experiment showed that wildfire significantly affected the sediment production in *A. chilensis* forest soils (Table 2).

Sediment production was much higher in the burned than in the unburned plots, and BFBS showed the highest values. Mean sediment production from the BFBS and BF plots was 200 and 22 times greater than the sediment produced from the unburned plots (Table 2).

Enrichment ratio

The concentrations of OC and TKN in the soils and sediments and their ER for the different treatments are presented in Table 3. ER was above 1 in the control and BFBS plots. In the soils of the burned plots, OC decreased, although differences with the control (UF) were significant only for the BFBS. TKN content was higher in BF than in the BFBS, with no difference with respect to the control (UF) plots.

Fig. 3 Infiltration rates during the 30-min simulated rainfall for the different treatments. References: ○ BFBS (burned forest and bare soil), ▵ BF (burned forest), □ UF (control plots)



The sediments were significantly different only for TKN concentrations, with the lowest values recorded in the BF plots. In the sediments of the BFBS and BF plots, TKN varied between 0.7 and 1.9 gkg^{-1} while TKN in the UF plots varied between and 3.6 and 3.7 gkg^{-1}.

The ER of OC and TKN for the three treatments showed a different pattern. The lowest ER for TKN and OC was found in the BF treatment. In this treatment, the mean ERs for OC and TKN were lower than 1. For the BFBS and UF plots, the ER was higher than one, indicating a selective removal of OC and TKN in these treatments. Considering the total sediment production, BFBS treatment showed the highest loss of nutrients during the simulated rainfall (Table 3).

Effect of mitigation treatment

Vegetation characteristic

At the moment of the mitigation treatment application, 14 months after the fire (Fig. 2), the vegetation cover in the burned forest was 7 %, with more than 90 % of bare soil. Twenty months later, when the mitigation was evaluated, the vegetation cover had increased dramatically (Table 4). Rainfalls was, with respect to the historical average, 28 % higher for 2009 and 17 % lower for 2010 (data from the weather stations located in the Los Alerces National Park, Esquel). In 2009, a displacement of the typically fall-winter rainfall to spring was also recorded and could have favored the vegetation recovery.

The vegetation cover for the different treatments did not show significant differences. In the BFWM plots, vegetation cover was dominated by herbaceous plants (Table 4), although *M. hastulata*, a tall shrub, was present in the adjacent areas. The BFM plots were mainly covered by shrubs. The litter cover was significatively lower in the burned than in the unburned plots while bryophytes cover was greater in the burned plots (Table 4).

Runoff and sediment production in the mitigation treatments

The mitigation treatment had a significant effect on runoff production; the highest values were observed in BFWM plots. The runoff production from BFM and UF plots was 3 and 35 times lower than the runoff from BFWM plots (Table 5).

The infiltration rate increased with the application of a mitigation treatment reaching values closer to the UF plots (Fig. 4).

Sediment production was significantly lower in the UF than in the BFWM, and there were no significant differences between the burned treatments (Table 5). The mean sediment production from the BFWM and BFM plots were almost 8 and 4 times greater than UF plots. However, sediment production was low even in the BFWM, showing that vegetation recovery, 34 months after wildfire, promoted a significant decrease in the erosion process (Table 5).

Discussion

The impacts of fire on vegetation and its recovery

Wildfires are one of the most common disturbances in forest ecosystems and may have devastating effects on vegetation. In the *A. chilensis* forest, the studied

Table 3 Means and standard errors of organic carbon and total Kjeldahl nitrogen in the soils and sediments and their enrichment ratios and total losses. Different letters indicate significant differences ($p<0.05$)

Treatments	Soil		Sediments		Enrichment ratio		Total losses	
	OC gkg^{-1}	TKN gkg^{-1}	OC gkg^{-1}	TKN gkg^{-1}	OC gkg^{-1}	TKN gkg^{-1}	OC gm^{-2}	TKN gm^{-2}
BFBS	54.38 (±10.79)a	1.07 (±0.09)a	95.76 (±7.72)a	1.60 (±0.11)ab	1.76 (±0.38)b	1.49 (±0.17)ab	98.25 (±7.92)b	1.64 (±0.11)b
BF	97.89 (±10.68)ab	1.68 (±0.16)b	72.39 (±12.17)a	0.97 (±0.13)a	0.74 (±0.20)a	0.57 (±0.01)a	8.59 (±1.44)ab	0.12 (±0.02)ab
UF	132.88 (±18.78)b	1.33 (±0.13)ab	147.16 (±65.89)a	3.69 (±0.02)b	1.11 (±0.55)ab	2.76 (±0.07)b	0.65 (±0.29)a	0.02 (±0.01)a

BFBS burned forest and bare soil; *BF* represent the condition 1 year after the fire; *UF* unburned forest *OC* organic carbon; *TKN* total Kjeldahl nitrogen

summer wildfire caused a strong reduction in litter and vegetation cover in most of the burned area. Wildfire behavior is generally variable and can damage or completely remove aerial biomass depending on its intensity and severity (Cochrane 2009). Some species can be eliminated and others can appear where they had not been present before the fire (Granged et al. 2011).

The recovery of the vegetation cover during the first year was slow and herbaceous plants and bryophytes were the first colonizers. Both herbaceous plants and bryophytes represent transient cover, as their abundance is negligible in the unburned forest, but they played an important role in reducing soil erosion (Table 1). According to Ryömä and Laaka-Lindberg (2005), bryophytes are often the first species to colonize burned substrates, especially after intense fires, and play an important role in the first stages of succession, especially where recolonization by vascular plants is slow.

Resprouting plants are able to respond to many types of disturbances by quickly regaining aboveground biomass after a disturbance event such as fire (Bond and Midgley 2003). Plants with this reproduction mechanism may accelerate the recovery of plant and litter cover and afford protection against erosion after wildfires. *M. hastulata*, a resprouting tall shrub, was present in the initial successional stages of the open areas generated by fire. This shrub could become dominant and represent a long lasting successional stage in the burned areas. Post-fire colonization of this species in burned forests was also found in Chile (Quintanilla-Pérez 1996).

Changes in litter and soil properties

In the *A. chilensis* forest, the wildfire caused a strong reduction of soil OM (approximately 25 % and 60 % in the BF and BFBS, with respect to UF). The difference between the BFBS and BF could be due to the removal of the surface soil with the highest OM content when removing the litter and vegetation cover in BFBS.

Similar findings were reported for soils of a *Nothofagus sp*. forest under similar climatic, soil and fire conditions where OC in the burned soil decreased by 65 % at a depth of 0–10 cm (Alauzis et al. 2004). Jordan et al. (2011) found a decrease in soil OM in volcanic soils affected by high severity fires. Soil OM destruction

Table 4 Means and standard errors of vegetation cover for the plots with- and without- mitigation and the control before the application of simulated rainfall. Different letters indicate significant differences ($p<0.05$)

Surface cover	BFWM	BFM	UF
Vegetation (%)	71.5 (±8.88)a	89 (±2.48)a	79 (±4.95)a
Herbaceous (%)	39 (±5.10)b	24.5 (±10.85)ab	0 (0)a
Shrubby (%)	0 (0)a	40 (±23.45)ab	77.5 (±5.30)b
Bryophytes (%)	33 (±6.29)b	24.5 (±11.7)a	1.5 (±0.35)a
Litter (%)	9 (±4.20)a	9.25 (±1.49)a	98.5 (±0.35)b

BFWM burned forest without mitigation; *BFM* burned forest with mitigation; *UF* unburned forest

during burning occurred mainly in coarse aggregates where combustion can be more intense. It can be postulated that the main pool of soil OM lost by combustion must be the particulate OM (i.e. the fraction >0.05 mm).

In the control plots, litter cover was almost 100 % and was a 5 cm thick layer. In the burned plots, 22 months after the fire, litter cover was a thin layer of partially burned *A. chilensis* leaves. Litter cover and soil OM losses as a consequence of fire influenced soil properties and its resistance to erosion. Closely related to the decrease in soil OM was an almost 50 % increase in the bulk density of the 0–5 cm soil layer. In spite of these increases, bulk density maintained low values in the burned soils (Table 1). The high OM content, even in the burned soils, and the presence of allophane can explain their low bulk density (Irisarri and Mendía 1997).

Other characteristic of the soil closely related to soil OM content is the soil hydrophobicity (DeBano 2000). Twenty-two months after wildfire, the highest values of hydrophobicity were detected in the soils of the BFBS treatment, mainly below to the soil surface. The highest hydrophobicity found in the BFBS could be due to the removal of the surface soil, exposing a more hydrophobic zone. When a fire occurs, the litter and upper soil layers are exposed to very intense heating, particularly during an intense burn. After fire has passed, the continued heat movement downward through the soil can re-volatilize some of the hydrophobic substances. The final result is a water repellent layer below and parallel to the soil surface on the burned area (DeBano 2000).

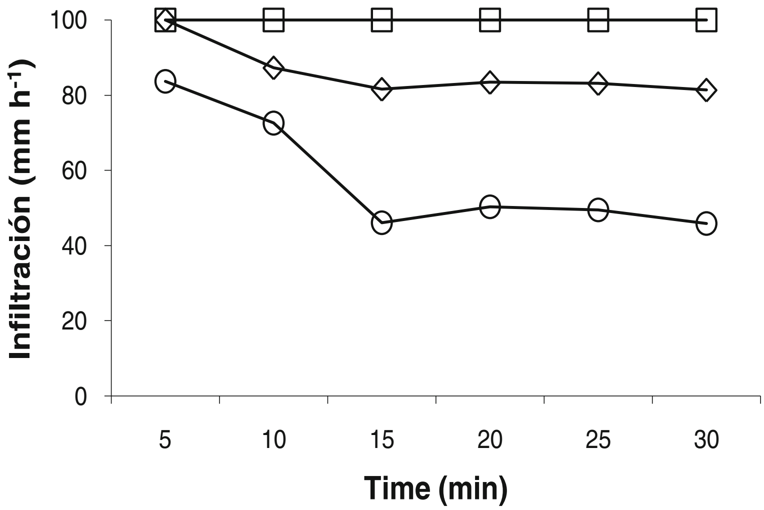
Table 5 Means and standard errors of runoff and sediment production of soils according mitigation treatments. Different letters indicate significant differences ($p<0.05$)

Treatments	Runoff production (lm^{-2})	Sediment production (gm^{-2})
BFWM	21.01 (±1.53)b	17.38 (±4.16)b
BFM	6.96 (±4.06)a	9.6 (±5.55)ab
UF	0.59 (±0.38)a	2.19 (±1.43)a

BFWM burned forest without mitigation; *BFM* burned forest with mitigation; *UF* unburned forest

More severe soil water repellency has usually been found under a deeper litter layer (Scott and Van Wyk 1990; Crockford et al. 1991). However, not all wildfires are able to generate an impermeable layer on the soil (Doerr et al. 1996) as they depend on the litter, fire intensity, period of intense heat, soil texture and soil moisture (DeBano 1981). On the other hand, the hydrophobicity in the studied soils was high even in unburned soils, which could be related to the intrinsic properties of volcanic ash as parent material (Jaramillo 2004) and the high levels of poorly humified OM (Benito et al. 2003). Contrary to what was expected, CD ratio was not affected by fire. The low clay dispersion ratio values found in the burned soils could be explained by their high OM content, which was above the OM threshold for being considered as erodible (Evans 1980). Soils with less than 2 % OC content can be considered erodible (Evans 1980) and soil erodibility decreases linearly with the increment in OM content, over a range of 0–10 % of OM (Voroney et al. 1981). The high contents of both, OM and allophanic alumino silicates that form clay-humic complexes, favors the stability of aggregates reducing the susceptibility of soil to erosion (Mussini et al. 1984; Wada 1985). The increase in aggregate stability could be explained, in part, as a consequence of water repellency (Arcenegui et al. 2008). Jordan et al. (2011) concluded that water repellent substances coating soil aggregates may impart stability since they delay water entering the aggregates.

Fig. 4 Infiltration rates during simulated rainfall events in soils from A. chilensis forest under different mitigation treatments in Patagonia, Argentina. References: ○ BFWM (burned forest without mitigation), ◊ BFM (burned forest with mitigation) □ UF (unburned forest)



Soil erosion

When fire consumes the litter layer and the surface soil OM, the porosity can be affected, hindering the process of water entry into the soil. Reductions in litter and vegetation cover leave the soil prone to raindrop impact and reduce the opportunities for rainfall storage, so that erosive overland flow tends to occur more readily (Shakesby and Doerr 2006).

In this study, the infiltration rate of burned soils was lower than unburned soils, which is consistent with several previous studies (DeBano 1990; Neary et al. 1999; Martin and Moody 2001; Shakesby 2011). The soils from UF plots showed a high infiltration capacity. This characteristic along with its high capacity to retain water, are typical of most volcanic soils due to the formation of allophane-humic complexes (Buol et al. 1991). Several studies have demonstrated that high post-fire runoff can be due to soil sealing (Martin and Moody 2001; Larsen et al. 2009). Sealing may reduce infiltration by action of raindrop impact on soil aggregates, dispersing the fine-grained particles.

Burned soils showed a different behavior in response to vegetation cover. In the BFBS treatment, the infiltration rate tended to be higher than in the BF. Although in the BF treatment bryophytes contributed to the stability and protection of soils, as has been found in other studies (Gaskin and Gardner 2001), they reduced the infiltration rate, acting as physical barrier that increased the runoff generation.

The biological soil crust could have reduced infiltration by occupying matrix pores, thereby creating a hydrophobic surface or through the swelling of the crusts (Eldridge 2001). In addition, it has been reported that porosity at the soil surface decreases as lichen/moss cover increases, especially once lichen and moss cover exceeds a critical threshold (Eldridge 2001; Belnap 2006). However, there are no reliable data on conditions under which pore clogging by crust organisms retards infiltration more or less than the formation and stabilization of pores increase infiltration (Belnap 2006).

Results on the effects of bryophytes have been conflicting, varying from place to place and ranging from strongly positive to strongly negative (Bowker et al. 2012). Some studies have demonstrated infiltration increasing on soils dominated by biological crusts when compared with crust-free soils; however other studies have found opposite results (Eldridge 2001, Belnap 2006). Warren (2001) suggested that overall, biological crusts decreased water infiltration in sandy soils (>66 % sand) and increased infiltration where clays exceeded 15 %.

Our study showed that sediment production was greater in burned soils than unburned soils. However, there was an important variation in sediment production according to the conditions of the burned plots. Sediment production from BFBS plots was an order of magnitude greater than from BF plots. This result agrees with other studies that showed that the rhizoids of mosses are woven into the surface soil to form a network that protect the soils (Warren 2001; Aguilar et al. 2004). Although bryophytes in the study site may account for a low proportion of the plant biomass, they played an important role as a pioneer in colonizing the bare soils, due to their capacity to stabilize the soil it and prevent soil erosion and nutrient losses.

Several authors postulated a great increase in erosion rates with increasing burn severity (Benavides-Solorio and MacDonald 2001; Robichaud et al. 2007). Indeed, our study showed that the mean sediment production (1026 gm^{-2}) from the burned and bare soil plots was very high, possibly enhanced by the steep slopes and the increase in sediment availability due to surface soil disturbance when removing litter and vegetation cover. The sediment production was even higher than rates recorded in studies with fires of high severity. For example, the mean sediment production recorded by Benavides-Solorio and MacDonald (2001) by applying a 80 mm h^{-1} simulated rainfall for 60 min on high severity burned plots, was 410 gm^{-2} in a site with means slope of 26 %. On the other hand, Robichaud and Waldrop (1994) measured sediment production of 139 gm^{-2} after a simulated rainfall of 100 mmh^{-1} for 30 min in a site with a high severity burn and mean slope of 30 %.

In spite of the low CD ratio of the Andisols, rainfall simulations showed that sediment production was high in both burned treatments, BF and BFBS. Studies in other Andisols have shown that sediment detachment is mainly caused by the impact of raindrop on large soil aggregates, with the subsequent runoff of smaller aggregates (Rodriguez-Rodriguez et al. 2002; Armas et al. 2004).

OC and N enrichment ratio

Despite the fact that the sediment production in the control plots was low, the ER of OC in these sediments only was slightly greater than 1. On the contrary, the high ER of TKN, 2.7, is evidence of a selective process of nutrient loss, which has been documented in other studies (Avnimelech and McHenry 1984; Sharpley 1985). Volcanic soils typically have high C/N ratios (Mussini et al. 1984) (Table 3), and this ratio was less for the eroded sediment.

The BFBS treatment, with the highest erosion rate, presented the highest sediment ER for OC and an intermediate level of TKN (Tables 2 and 3). Contrasting results have been found by several researchers who observed in laboratory and field experiments that ER of OC increases with decreasing soil loss (Young et al. 1986; Schiettecatte et al. 2008). Water erosion is a selective process that removes OM and nutrients in runoff reducing soil fertility and subsequently productivity (Sharpley 1985). Fire may further increase the total nutrient losses by increasing the sediment production. In fact, the total OC and TKN removed from BFBS plots were equivalent to 98.25 and 1.64 gm^2, respectively.

In the burned soils (BF) the ER was lower than 1, suggesting that bryophyte cover not only reduced sediment production but also prevented nutrient losses (Turetsky 2003). In fact, the total OC and TKN losses from the BF plots were 8.59 and 0.12 gm^{-2}, respectively.

Some conflicting results found in this study and the lack of previous work on ER in the volcanic soils of our study area, prevent an accurate understanding of the edaphic processes that determine the variations in ER for OC and N for the different treatments. This is an issue that should be explored in future works.

Mitigation treatment

Our study showed that the mitigation treatment reduced runoff and enhanced water entry into the soil reaching values closer to the UF plots. Several studies in postfire areas have showed the efficiency of mitigation treatments in the decrease of runoff (Robichaud et al. 2000; Wagenbrenner et al. 2006). Although in the mitigation treatment sediment production was 45 % lower than in the BFWM treatment, differences were not significant. Similar results were reported by Robichaud et al. (2008) in burned soil with contour-felled log treatment and the contour trench treatment. Our results could be associated with the high fertility of volcanic soils and the good climatic condition after the wildfire that promoted the recovery of vegetation cover in a relatively short time. Thirty-four months after wildfire, when the mitigation treatment was evaluated, the vegetation cover, composed mainly by bryophytes and herbaceous plants, was high enough as to minimize erosion rates, even in the burned soil without mitigation treatment.

Although the mitigation treatment, evaluated 20 months after its application, showed no significant effect on sediment production, the possibility of a positive effect within a short period after mitigation cannot be discarded. Our study showed that sediment production in burned and bare soils was very high. BFBS treatment tried to simulate the soil hydrological behavior immediately after the wildfire when litter and vegetation cover had been totally removed. Under these extreme conditions, and considering the steep slope and the high risk of surface water contamination, the application of mitigation treatments in burned forest could be justified, albeit in specific areas, in order to protect the surface water quality.

Conclusions

Simulated rainfall allowed us to estimate infiltration, runoff and sediment production after wildfire. The results showed a decrease in infiltration rate and an increase in sediment production in burned soils.

The increase of erosion rate in burned and bare soils showed the importance of bryophytes as first colonizers after wildfire. Vegetation recovery improved soil stability reducing the erosion rates.

Mitigation treatment had a significant effect in reducing runoff but it had little impact on sediment production compared to the unmitigated burned plot. Thirty-four months after wildfire, the sediment production was low even in burned soils without mitigation. The high fertility of the volcanic soils and good climatic conditions after wildfire promoted the recovery of vegetation in a relatively short time, reducing naturally the erosion risk in burned forest.

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