Geoderma 223–225 (2014) 79–87

Contents lists available at ScienceDirect

Geoderma

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

Testing the effects of straw mulching and herb seeding on soil erosion after fire in a gorse shrubland

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ARTICLE INFO

Article history:
Received 7 June 2013
Received in revised form 23 December 2013
Accepted 3 January 2014
Available online 18 February 2014

Keywords:
Sediment yield
Experimental fire
Antecedent soil moisture
Stabilization treatments
Mulching
Seeding

ABSTRACT

In this study, we compared the effectiveness of two treatments, straw mulching (2.3 Mg ha^{-1}) and seeding (4 g m^{-2}), in reducing soil erosion after an experimental fire in a gorse (*Ulex europaeus* L.) shrubland. Straw mulch provided an initial ground cover of 87%.

The maximum temperature reached at the mineral soil surface during fire was positively related to accumulated sediment yield during the first year after fire, supporting the importance of soil heating (or its surrogate, soil burn severity) to explain post-fire erosion.

The first year after fire only straw mulch application significantly reduced soil erosion relative to the untreated burned soils (89%). Seeding did not affect soil erosion after fire. The mean sediment yield after the seeding treatment (2.9 Mg ha^{-1}) was similar to that in the untreated burned plots (3.6 Mg ha^{-1}).

Maximum concentrated precipitation (rainfall > 20 mm accumulated in one or two consecutive days) and antecedent soil moisture were the variables most strongly associated with soil losses in the untreated burned and seeded soils. Rainfall intensity-related variables did not explain soil loss variability. In the mulched soil, only the maximum concentrated rainfall was related to sediment yield.

Total vegetation cover recovery was quite fast, particularly between 6 and 12 months after fire, and 70% of the ground was covered by the end of the study. However, when the erosion rate was maximal, the cover provided by vegetation was very low. Seeding contribution to total vegetation cover was small and we did not find any differences in the mean vegetation cover between treatments although mulching delayed vegetation cover recovery slightly.

In summary, seeding failed to reduce soil loss after fire, and the cover, directly in contact with the ground was a key factor in reducing post-fire soil loss.

1. Introduction

Shrublands cover more than 600,000 ha of land in Galicia (NW Spain), representing about 30% of uncultivated lands in the region (MMAMRyM, 2011). The shrubland is frequently affected by wildfires: in the period 2001–2010, more than 50% of the wildfires in Spain occurred in Galicia and about 70% of the wildland area burned in that region occurred in shrubland ecosystems (MMA, 2010).

Increased levels of runoff and erosion following fire have been measured in different shrubland-type ecosystems, including those in Galicia (Benavides-Solorio and MacDonald, 2001; Fernández et al., 2011; Johansen et al., 2001; Martin and Moody, 2001; Pierson et al., 2008), and diverse emergency post-fire soil stabilization treatments have been proposed, used and tested in field studies (Bautista et al., 1996; Groen and Woods, 2008; Robichaud et al., 2006, 2008a; Wagenbrenner et al., 2006). Among these treatments, grass seeding has been widely used for post-fire erosion control because it is relatively inexpensive and easy to apply. Some recent literature reviews (Beyers, 2009; Peppin et al., 2010) have concluded that post-fire grass seeding may be ineffective in increasing ground cover or reducing post-fire erosion rates, particularly in the first year after fire, when the risk of erosion is highest. However, research in Galicia has demonstrated the efficacy of sowing grass in reducing post-fire soil losses in a warm and rainy climate (Díaz-Raviña et al., 2012; Pinaya et al., 2000). Other treatments, such as mulching, are used to increase ground cover and have been proved suitable for reducing soil losses after wildfire in Galicia and elsewhere (Bautista et al., 1996; Fernández et al., 2011; Groen and Woods, 2008; Prats et al., 2012; Wagenbrenner et al., 2006) although studies that directly compare mulching and seeding in the same burned site are scarce (Bautista et al., 1996; Díaz-Raviña et al., 2012; Wagenbrenner et al., 2006).

It is well known that the changes in soil physical properties influencing erosion after fire depend on threshold temperatures in the soil (De Bano et al., 1998; Neary et al., 2005; Ubeda and Outeiro, 2009). However such information is generally obtained in laboratory-based soil heating studies, with limitations to be extrapolated to the real world, and the relationship between temperature reached at soil during real fires and subsequent soil erosion, both measured in field

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http://dx.doi.org/10.1016/j.geoderma.2014.01.014

studies, has less often been investigated (De Luis et al., 2003; Fernández et al., 2008, 2012a, 2012b; Gimeno et al., 2000; Vega et al., 2005).

The change in soil moisture brought about by fire and stabilization treatments can in turn affect water repellency, overland flow, and soil erosion, the latter particularly through the antecedent soil moisture. However, there is a gap of knowledge about this point (Prats et al., 2012; Smets et al., 2008).

Stabilization treatments can also have other ecological consequences. There is a growing concern that seeding may interfere with the natural recovery of vegetation (Beyers, 2004; Dodson and Peterson, 2009; Keeley, 2004). Previous studies have also suggested that mulching can both inhibit plant establishment and introduce exotic species (Beyers, 2009; Kruse et al., 2004) or enhance plant installation in dry sites (Bautista et al., 2009; Dodson and Peterson, 2010; Wagenbrenner et al., 2006). However, those studies were carried out under different climates than in Galicia.

Although the regional Forest Service has recently initiated a post-fire stabilization programme in Galicia, there is still a need for quantitative studies on the effectiveness of post-fire stabilization and the relationship between fire characteristics (particularly fire severity) and soil erosion, which would help managers to plan post-fire actions (Vega et al., 2013a).

The main aim of the present study was to compare the effectiveness of two different post-fire soil stabilization treatments (mulching and seeding) in reducing sediment yields relative to an untreated control after an experimental fire in a representative shrubland in Galicia under an oceanic climate. Further objectives were as follows: i) to determine if the above treatments affect the recovery of vegetation cover relative to a burned untreated control, and ii) to explore how sediment yields during the first year after fire can be affected by soil heating, rainfall characteristics and antecedent soil moisture.

2. Materials and methods

2.1. Study site

The study site was a mountain hillslope (42° 38′ 58″ N; 8° 29′ 31″ W; elevation 660 m a.s.l.), of mean slope 46% and NW orientation, in the municipality of A Estrada (Pontevedra, NW Spain). Prior to the experimental fire, the shrubland was dominated by gorse *Ulex europaeus* L. and *Pteridium aquilinum* (L.) Kuhn. *Ulex gallii* Planch., *Daboecia cantabrica* ((Huds.) K. Koch), and *Pseudoarrhenaterum longifolium* (Thore) Rouy were also present. Shrub cover was 100% with a mean height of 1.2 m.

The climate in the area is oceanic. The average rainfall is 1810 mm $year^{-1}$, with a dry period of one month in summer. The mean annual temperature is 12 °C. The soils, developed over a granite bedrock, are classified as Humic Cambisols (FAO, 1998) and have a sandy-loam texture and very low stoniness. Mean organic carbon content is 18% in the first 0.05 m depth. The mean soil depth in the study site is 0.50 m.

2.2. Experimental design and field measurements

2.2.1. Experimental fire

Fifteen experimental macro-plots (each 30 m × 10 m) were established with the longest dimension parallel to the maximum slope.

Shrub was cut six months before burning and laid over the soil to facilitate its consumption and favour heat transfer to the soil (Fig. 1). The pre-burn and post-burn shrub and litter fuel loading for each plot was estimated by cutting and weighing all the phytomass within five square subplots (1 m × 1 m) for each plot. The material removed was oven-dried and litter subsamples were combusted at 450 °C for 4 h to prevent contamination from mineral soil. Shrub and litter loads before and immediately after fire are shown in Table 1. The soil thermal regime during burning was monitored via a datalogger connected to thermocouples (chromel alumel K type; inconel sheath 1 mm diameter) positioned at five randomly selected points within each burning plot. At each point, three thermocouples were inserted to different depths: one in the litter and duff interface, one in the mineral soil surface and one at 2.5 cm below the mineral soil surface. Relative air humidity, temperature and wind velocity were measured continuously, at a height of 2 m, during burning, by an automatic meteorological station positioned near the plots. The plots were burned on 15 October 2009 by the use of the back-fire technique. Environmental and fire behaviour variables during the experimental fire are listed in Table 1. Changes in litter depth resulting from the burning were measured using metal pins placed flush with the litter surface at 1 m intervals along two parallel transects in each plot. Additional pins were located immediately beside each thermocouple. Immediately after fire, the emergent portion of the pins and the residual litter depth were measured to determine the absolute and relative changes in the litter thickness. The remaining soil organic layer cover in each experimental plot was also measured immediately after the experimental fire. The percentage of remaining charred soil organic cover was assessed in 20 cm × 20 cm quadrats placed at thirty systematically selected points along two transects parallel to the plot longest dimension in each plot. In addition, each quadrat was assigned to one of the five severity levels described in Vega et al. (2013b): 1. Burnt litter (Oi) but limited duff (Oe + Oa) consumption. 2. Oa layer totally charred and covering mineral soil, there may be ash. 3. Forest floor (Oi + Oe + Oa layers) completely consumed (bare soil) but soil organic matter not consumed and surface soil intact 4. Forest floor completely consumed and soil organic matter in Ah horizon also consumed, a thick layer of ash deposited and soil structure altered. 5. As 3 and colour altered (reddish).

2.2.2. Erosion-related measurements

Immediately after the fire and before any appreciable rainfall, plots were delimited by a geotextile fabric fixed to posts (Fig. 2). Uphill borders of the plots were trenched to avoid external inputs from runoff or erosion. Sediment fences, made from a geotextile fabric similar to that described by Robichaud and Brown (2002), were placed in the downhill portion of the plots and were used for periodic collection of sediment.

To study the effect of different post-fire soil stabilization treatments on sediment yield, three treatments were randomly assigned to the fifteen burned plots (Fig. 3): straw mulching, seeding, and control (untreated burned soil). Grass seed was sown manually, at a rate of 4 g m^{-2}, throughout the plot, as homogeneously as possible. The seed mixture comprised 35% *Lolium multiflorum*, 20% *Dactylis glomerata*, 10% *Festuca arundinacea*, 5% *Festuca rubra*, 5% *Agrostis tennuis* and 25% *Trifolium repens* (each % of total weight). Wheat straw mulch was applied uniformly by hand at a rate of 2.3 Mg ha^{-1}. Initial mulch cover was 87%. The treatments were carried out seven days after fire and before any appreciable rainfall.

Eroded soil was collected periodically during the first hydrological year following treatments (between October 2009 and September 2010). The soil was collected from the sediment fences one or two days after low pressure fronts causing precipitation had passed, to allow drainage of most of the water and facilitate the collection. These fronts originated rainfall during nine consecutive days, as an average, in the study-period. Samples of eroded soil were oven-dried (105 °C) for 24 h to determine dry sediment mass.

Rainfall amount and intensity were measured by two recording rain gauges positioned adjacent to the experimental site, 1.20 m above ground level. Total precipitation, maximum concentrated precipitation (accumulated precipitation > 20 mm concentrated in one or two consecutive days in the period between two sediment collection dates), maximum rainfall intensity in 10 (I_{10}) and 30 min (I_{30}), rainfall kinetic energy and the rainfall erosivity factor (Wischmeier and Smith, 1978) were determined for the periods of sediment collection. The rainfall erosivity factor for each rainfall event was calculated as the product of the maximum rainfall intensity in thirty minutes and the rainfall kinetic energy. These values were added for each period of sediment collection.