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Evaluation of the rusle and disturbed wepp erosion models for predicting soil loss in the first year after wildfire in NW Spain

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ABSTRACT

Severe fire greatly increases soil erosion rates and overland-flow in forest land. Soil erosion prediction models are essential for estimating fire impacts and planning post-fire emergency responses. We evaluated the performance of a) the Revised Universal Soil Loss Equation (RUSLE), modified by inclusion of an alternative equation for the soil erodibility factor, and b) the Disturbed WEPP model, by comparing the soil loss predicted by the models and the soil loss measured in the first year after wildfire in 44 experimental field plots in NW Spain. The Disturbed WEPP has not previously been validated with field data for use in NW Spain; validation studies are also very scarce in other areas. We found that both models underestimated the erosion rates. The accuracy of the RUSLE model was low, even after inclusion of a modified soil erodibility factor accounting for high contents of soil organic matter. We conclude that neither model is suitable for predicting soil erosion in the first year after fire in NW Spain and suggest that soil burn severity should be given greater weighting in post-fire soil erosion modelling.

1. Introduction

The increased runoff and erosion caused by wildfire can lead to severe degradation of soil quality and productivity and also affect downstream surface water quality. Estimating the risk of soil erosion is essential for planning post-wildfire soil stabilization measures (Vega et al., 2013a), and soil erosion prediction models are important in this respect. The availability of operational tools for the rapid assessment of potential soil loss after wildfire is essential to enable prioritization of the responses (Robichaud and Ashmun, 2013). This is particularly important in NW Spain, a region where a large number of summer wildfires occur each year (MAPAMA, 2016) and where there is usually little time available for planning and implementing soil stabilization measures before the start of the rainy season. In NW Spain, between 73.0% and 98.6% of post-fire soil losses take place during the first year after wildfire (Fernández and Vega, 2014, 2016a; Fernández et al., 2011; Vega et al., 2015).

Empirical models such as RUSLE (Renard et al., 1997) are widely used for multiple purposes, including assessment of post-fire erosion risk (e.g. Miller et al., 2003; Myronidis et al., 2010; Rulli et al., 2013). However, the model performance has not been widely tested using field data, and existing studies have indicated that its performance for burned soils is questionable (Fernández and Vega, 2016b; Fernández et al., 2010; Larsen and MacDonald, 2007). One of the main reasons given for the poor performance is the method used to calculate the soil erodibility factor (Benito et al., 2009; Fernández and Vega, 2016b; Fernández et al., 2010; Larsen and MacDonald, 2007). The soil erodibility factor was originally formulated by Wischmeier et al. (1971) by combining a series of variables in a nomogram (silt and very fine sand content, clay content, organic matter content, an aggregation index and a permeability index). Rock fragment cover was later added thus providing the classic equation that is still widely used to calculate the soil erodibility factor (Wischmeier and Smith, 1978). However, amongst other limitations, the equation simplifies condition of soils that are rich in organic matter. To overcome this, a system of equations emulating the nomogram and that can be applied to a range of soil characteristics was proposed (Auerswald et al., 2014). However, the approach has not been yet validated for application to burned soils.

The WEPP model is a physically based model used to predict runoff and soil loss at hillslope and watershed scale (Nearing et al., 1989). It is one of the models most commonly used in the USA to predict post-fire soil loss (Miller et al., 2016). However, WEPP requires large amounts of input data, often rendering it difficult to apply. By contrast, the Disturbed WEPP model (Elliot, 2004), a web-based interface to the WEPP model, only has seven input variables (climate type, slope length, slope gradient, soil texture, proportion of rock fragments in the soil, percentage surface cover and land use information), and can be easily applied in operational settings. Assessment of the performance of

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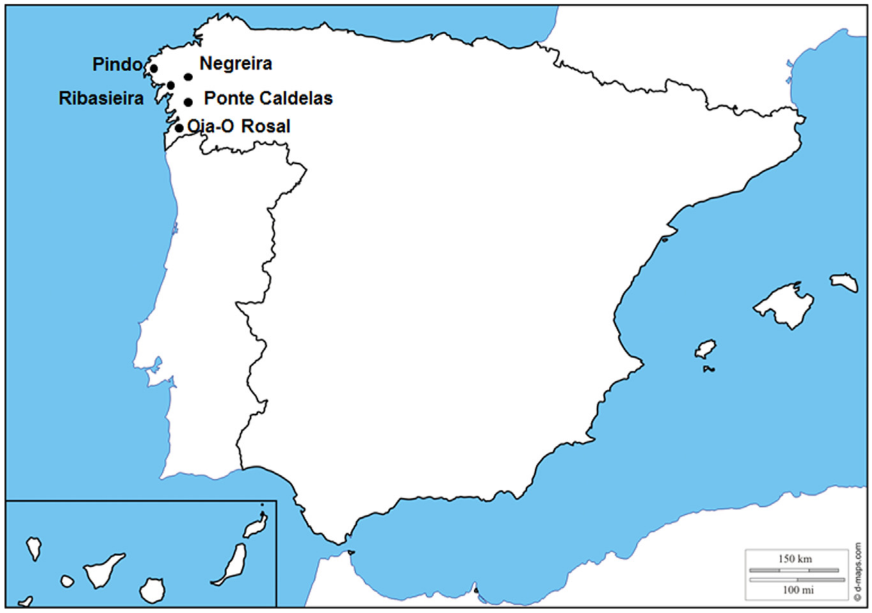

Fig. 1. Location of the study sites.

Disturbed WEPP in recently burned areas has shown a tendency for the model to underestimate soil loss, although the available information is very scarce (Larsen and MacDonald, 2007).

Evaluation of post-fire soil erosion models based on field measured data, at suitable scale, is considered essential to ensure that models can be applied as operational tools (Larsen and MacDonald, 2007). The main objective of this study was therefore to compare the performance of a version of the RUSLE, modified by inclusion of the alternative equation of the soil erodibility factor proposed by Auerswald et al. (2014), and that of the Disturbed WEPP for predicting soil erosion rates measured at hillslope scale over the first year after fire in five areas in NW Spain.

2. Materials and methods

2.1. Study sites and field measurements

The study was carried out in five areas affected by wildfires in the Atlantic coastal region of NW Spain (Fig. 1). A total of 44 plots were established in the areas immediately after fire in the summer of 2013. The burned areas mainly included pine stands (*Pinus pinaster* Ait.) affected by a crown fire and shrublands in which most of the aboveground portions had been combusted thus leaving only partially charred stalks. The experimental plots (80 m^2) were established in burned areas immediately after the wildfires. Sediment fences made from a geotextile fabric were erected in the downhill portion of the plots to enable periodic collection of sediment.

In each plot, soil burn severity was assessed along two transects parallel to the steepest slope of the hill by a modification of the soil burn severity index proposed by Vega et al. (2013b). The soil burn severity levels are summarized as follows: (1) Very low soil burn severity. Burnt litter (Oi) but limited duff (Oe + Oa) consumption; (2) Low burn severity. Oa layer totally charred and covering the mineral soil, possibly some ash; (3) Moderate soil burn severity. Forest floor (Oi + Oe + Oa layers) completely consumed (bare soil), but soil organic matter not consumed and surface soil intact; (4) High soil burn severity. Forest floor completely consumed. Soil organic matter in the Ah horizon consumed and soil structure altered within the upper 1 cm; (5) Very high soil burn severity. As 4 but at depth greater than1cm; and (6), as 4 /5 and colour altered (reddish).

Remnant vegetation cover and height were measured along the above- mentioned transects. Percentage cover by rock fragments larger than 2 cm was measured simultaneously at 10 randomly chosen points in the transects by counting such fragments within a 1 m^2 sampling quadrat. Sampling was carried out twice during the study period. In each plot, a composite sample from the surface soils (0–5 cm) was collected immediately after fire to determine the particle size distribution by means of the vacuum pipetting system (Gee and Bauder, 1986). The carbon content was determined in an element analyzer (LECO) and the value obtained was multiplied by 1.72 to determine the percentage of soil organic matter.

At each study site, the amount and intensity of rainfall were measured with tipping bucket rain gauges placed 1.20 m above ground level.Main site characteristics are compiled in Table 1.

2.2. RUSLE

RUSLE is a modified version of the Universal Soil Loss Equation developed by Wischmeier and Smith (1978) for predicting annual soil loss. In the present study, RUSLE was used to estimate soil loss during the first year after fire (Mg $ha^{-1} year^{-1}$). The model include five factors: rainfall erosivity (R in MJ mm ha^{-1} $h^{-1} year^{-1}$); soil erodibility (K in Mg ha^{-1} h ha^{-1} MJ^{-1} mm^{-1}); a non-dimensional topographic factor (LS); a cover-management factor (C); and a factor reflecting soil conservation practices (P).

Rainfall erosivity, R, was determined on the basis of rainfall data collected at each study site for all the events that occurred during the year of study. The temporal resolution of the rainfall data was 5 min. The values were calculated by applying the criteria proposed by Renard et al. (1997); i) rain showers of less than 12.5 mm were not included in the computation, unless 6.25 mm of rain fell in 15 min, ii) rainfall accumulation of less than 1.25 mm during a period of 6 h was used to divide a longer storm period into two storms.

The topographic factor LS was obtained according to the characteristics of the different plots by using the equation reported by Renard et al. (1997) for slope angles greater than 9%. Plot slope angle was measured in the field with a clinometer. A standard slope length of 20 m was considered.

The soil erodibility factor, K, was calculated using the equation proposed by Auerswald et al. (2014), which is better suited to soils with high soil organic matter contents and elevated stone ground covers than the original equation of Wischmeier and Smith (1978).