EARTH SURFACE PROCESSES AND LANDFORMS
Earth Surf. Process. Landforms **41**, 928–935 (2016)

Published online 29 December 2015 in Wiley Online Library
(wileyonlinelibrary.com) DOI: 10.1002/esp.3876

Modelling the effect of soil burn severity on soil erosion at hillslope scale in the first year following wildfire in NW Spain

Cristina Fernández* **and José A. Vega**
Centro de Investigación Forestal de Lourizán, Consellería do Medio Rural, Xunta de Galicia, Pontevedra, Spain

Received 10 April 2015; Revised 23 November 2015; Accepted 23 November 2015

*Correspondence to: Cristina Fernández, Centro de Investigación Forestal de Lourizán, Consellería do Medio Rural, Xunta de Galicia, PO Box 127, 36080 Pontevedra, Spain. E-mail: cffilgueira@gmail.com

ESPL
Earth Surface Processes and Landforms

ABSTRACT: Fire severity is recognized as a key factor in explaining post-fire soil erosion. However, the relationship between soil burn severity and soil loss has not been fully established until now. Sediment availability may also affect the extent of post-fire soil erosion. The objective of this study was to determine whether soil burn severity, estimated by an operational classification system based on visual indicators, can significantly explain soil loss in the first year after wildfire in shrubland and other areas affected by crown fires in northwest (NW) Spain. An additional aim was to establish indicators of sediment availability for use as explanatory variables for post-fire soil loss. For these purposes, we measured hillslope-scale sediment production rates and site characteristics during the first year after wildfire in 15 experimental sites using 65 plots. Sediment yields varied from 0.2 Mg ha^{-1} to 50.1 Mg ha^{-1} and soil burn severity ranged from low (1.8) to very high (4.5) in the study period. A model that included soil burn severity, annual precipitation and a land use factor (as a surrogate for sediment availability) as explanatory variables reasonably explained the erosion losses measured during the first year after fire. Model validation confirmed the usefulness of this empirical model. The proposed empirical model could be used by forest managers to help evaluate erosion risks and to plan post-fire stabilization activities. Copyright © 2015 John Wiley & Sons, Ltd.

KEYWORDS: wildfire; soil erosion; soil burn severity; land use; sediment availability

Introduction

Post-fire soil erosion is of major concern because of the potential effects on soil and water resources. High severity wildfire usually increases soil erosion rates through the destruction of vegetation and ground cover, which exposes the mineral soil to raindrop impact and alters the soil structure by affecting bulk density and total porosity, thus reducing infiltration and promoting overland flow (De Bano *et al.*, 1998; Shakesby and Doerr, 2006; Novara *et al.*, 2011; Novara *et al.*, 2013; Keesstra *et al.*, 2014; Tessler *et al.*, 2015). Moderate to high severity burning of soils reduces aggregate stability through combustion of soil organic matter binding agents (Mataix-Solera *et al.*, 2011; Pereira *et al.*, 2014). Alteration or creation of hydrophobic and a hyper-dry soil conditions during soil heating may also contribute to soil loss (De Bano, 1981; Keizer *et al.*, 2008; Bodí *et al.*, 2011; Moody and Ebel, 2012; León *et al.*, 2013; Williams *et al.*, 2014). Moreover, soil burn severity has a strong impact on soil biota, being also the key of soil recovery (Guénon *et al.*, 2013; Vega *et al.*, 2013b; Guénon and Gros, 2015; Hedo *et al.*, 2015; Wang *et al.*, 2015).

As a descriptor of the magnitude of the changes that occur in the soil, fire severity is recognized as a decisive factor in ecologically based fire management. Moreover, fire severity is a crucial factor controlling post-fire soil erosion rates (Benavides-Solorio and MacDonald, 2005; Vega *et al.*, 2005; Cawson *et al.*, 2013). In a recent meta-analysis based on rainfall simulations in burned soils, Vieira *et al.* (2015) highlighted the importance of soil burn severity in explaining post-fire soil loss and pointed out the inconsistencies between existing methods of fire severity classification. The type of ashes remaining after fire has been used as a soil burn severity indicator due to its relationship with soil physical properties (Bodí *et al.*, 2011; León *et al.*, 2013; Pereira *et al.*, 2013; Pereira *et al.*, 2014). Although some research has been devoted to the quantification of soil burn severity (Guerrero *et al.*, 2007; Arcenegui *et al.*, 2008; Neris *et al.*, 2014), consistent and objective quantification of soil burn severity remains elusive (Moody *et al.*, 2013; Morgan *et al.*, 2014), and direct links between fire severity proxies and soil erosion are not well established (Shakesby and Doerr, 2006; Moody *et al.*, 2013). Vega *et al.* (2013b) proposed a soil burn severity classification based on visual indicators that accurately reflected changes in some soil properties. This classification is currently being used as an operational tool to prioritize emergency stabilization of areas after fire in northwest (NW) Spain (Vega *et al.*, 2013a). Nonetheless, it is worthwhile exploring whether these simple visual indicators of fire severity are useful predictors of post-fire soil erosion in areas prone to high soil losses due to the combination of absence of vegetation cover (areas where vegetation cover has been eliminated such as burned shrubland or forest stands affected by crown fires), pronounced erodibility of burned soils, steep terrain and high levels of precipitation. In fact, the highest post-fire erosion rates in Spain occur in the NW (Cerdá, 1998; Lasanta and Cerdà, 2005; Vega *et al.*, 2005; Fernández *et al.*, 2011; Vega *et al.*, 2015).

Soil characteristics associated with past fire activity or/and agricultural and forestry management can also play a significant role in post-wildfire soil loss through increments in soil erodibility or as a result of sediment exhaustion. In areas such as NW Spain where there is a long history of intensive land use, this factor may be critical for explaining soil loss. Post-fire soil erosion may be also limited in shallow and stony soils (Shakesby, 2011). Indeed, Moody *et al.* (2013) highlighted the need to increase our understanding of the role played by soil depth, in combination with other variables, for explaining the hydrological response of burned soils.

Several existing models can be used to predict soil erosion. Empirically based models such as the universal soil loss equation (USLE) (Wischmeier and Smith, 1978) and its revised version RUSLE (Renard *et al.*, 1997) are useful for estimating soil erosion on an annual basis and are frequently used as operational tools (Miller *et al.*, 2003; Myronidis *et al.*, 2010; Rulli *et al.*, 2013). These are potentially feasible management tools as they require less field data than other more complex models. However, the performance of such models in relation to burned forest soils has yielded questionable results (Larsen and MacDonald, 2007; Fernández *et al.*, 2010). Physically-based models such as the water erosion prediction project (WEPP) (Nearing *et al.*, 1989) and the European soil erosion model (EUROSEM) (Morgan *et al.*, 1998) have increased our knowledge of the underlying mechanisms involved in the erosive processes. They can simulate the effects of vegetation and other soil properties on erosion in individual storms, but are often too demanding as regards inputs to be used as operational tools.

In summary, classification of soil burn severity is crucial for the quantification of post-fire erosion and for planning post-fire mitigation activities (Morgan *et al.*, 2014); however, the lag between theoretical development and application in current models has not been satisfactorily addressed. Indeed, the need to include fire severity in predictive erosion models is a current research priority worldwide (Moody *et al.*, 2013) along with an accurate quantification in the field (Pérez-Cabello *et al.*, 2012). From the perspective of land management, these models must be as simple as possible for use as operational tools for planning post-fire mitigation activities. In NW Spain, most soil erosion occurs during the first year after fire as a result of climatic conditions and the rapid regrowth of vegetation (Fernández *et al.*, 2011; Vega *et al.*, 2015). It is therefore essential to develop predictive tools that focus on this period.

The main objective of this study was to determine whether soil burn severity, estimated through an operational classification system based on visual indicators, is significantly related with soil loss in the first year after wildfire in shrubland and other areas affected by crown fires in NW Spain. An additional aim was to determine whether indicators of sediment availability can also be used as explanatory variables for post-fire soil losses.

Materials and Methods

Study sites and field measurements

The study was carried out in 10 burned areas in NW Spain (Figure 1) where 65 experimental plots were installed in 15 different study sites (Table I). The burned areas were pine (*Pinus pinaster* Ait.) and eucalypt (*Eucalyptus globulus* Labill.) stands affected by crown fires and shrubland areas in which most of the aboveground portions had been combusted and only partially charred stalks remained.

The bedrock is predominantly granite (Table I). Soils were poorly developed (< 0.75 m deep), as is typical in shrubland and forested areas, and most were of sandy-loam texture (Table I). The soils are classified as Alumi-umbric Leptosols or Alumi-umbric Regosols (FAO, 1998).

Experimental plots (each 20 m × 4 m) were installed in burned areas immediately after wildfire. Sediment fences, made from a geotextile fabric similar to that described by Robichaud and Brown (2002), were located in the downhill portions of the plots for periodic collection of sediment. The uphill edges of the plots were trenched to avoid external inputs from runoff or erosion.

Soil burn severity was assessed with the aid of a 30 cm × 30 cm quadrat, which was placed at 30 systematically selected points along two transects parallel to the longest dimension of each plot. The soil in each quadrat was classified using a modification of the soil burn severity index proposed by Vega *et al.* (2013b) and a numerical value was assigned as follows: 0, unburnt soil; 1, burnt litter layer (Oi layer) but limited duff (Oe + Oa layers) consumption; 2, Oa layer totally charred and covering mineral soil, possibly

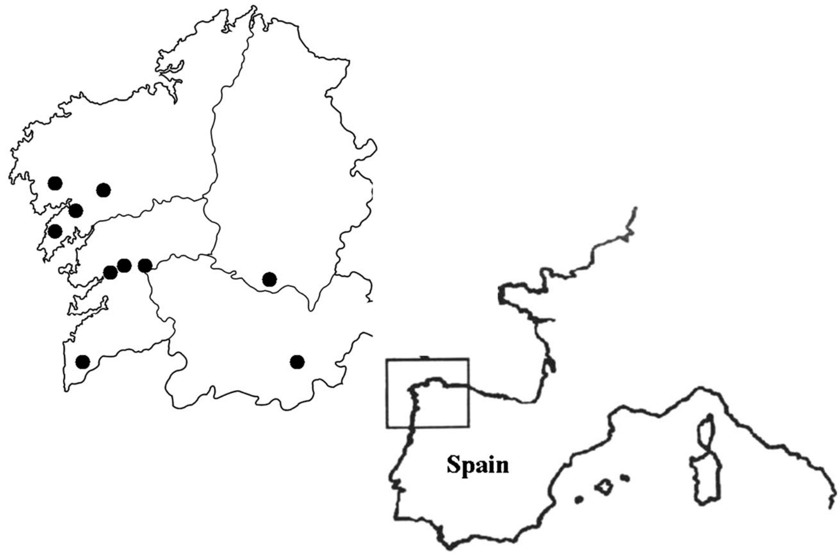

Figure 1. Location of study sites.

Table I. Main characteristics of the study sites

Wildfire (Date)	Area (ha)	Substrate/soil texture	Study site/number of plots	Vegetation type	Elevation (m)	Mean slope (%)	Soil depth (m)	Stoniness (%)	Mean soil burn severity	Land use factor
Ponte-Caldelas (2013)	707	Granite/sandy-loam	Rebordelo/5	Shrubland	480	36 (27–43)	0.40 (0.36–0.46)	3.3 (1.0–5.3)	3.8 (3.4–4.2)	3
		Granite/sandy-loam	Outeiro Vello/5	Shrubland	500	40 (39–45)	0.63 (0.59–0.69)	1.0 (0.3–2.5)	4.0 (3.5–4.4)	3
		Granite/sandy-loam	Costa Xandra/5	Shrubland	450	35 (29–39)	0.28 (0.26–0.33)	5.0 (4.0–6.0)	4.5 (4.4–4.7)	2
Oia-O Rosal (2013)	1824	Schist/sandy-loam	Cova Lobo/3	Pine stand	125	36 (30–48)	0.29 (0.28–0.30)	33.2 (30.2–37.2)	2.7 (2.6–2.8)	2
		Schist/sandy-loam	Villeiras/3	Pine stand	250	43 (42–45)	0.28 (0.26–0.30)	17.5 (2.3–26.7)	2.9 (2.8–3.0)	2
		Schist/sandy-loam	Verduriña/3	Pine stand	325	28 (25–34)	0.25 (0.22–0.28)	27.7 (15.4–44.5)	2.8 (2.7–2.9)	2
		Granite/sandy-loam	Acevedo/5	Pine stand/Shrubland	250	48 (42–53)	0.31 (0.21–0.50)	15.8 (2.6–33.6)	3.9 (3.7–4.2)	3
Carnota (2013)	2377	Granite/sandy-loam	Pindo/5	Pine stand	550	50 (48–52)	0.30 (0.26–0.33)	7.7 (3.0–10.0)	3.2 (3.1–3.5)	1
Negreira (2013)	672	Granite/loamy sand	Lueiro/5	Shrubland	400	46 (41–49)	0.34 (0.27–0.43)	7.9 (4.1–14.3)	3.9 (3.6–4.2)	3
Ribasieira (2013)	325	Granite/sandy-loam	Xufres/5	Shrubland	350	45 (43–48)	0.27 (0.23–0.31)	7.7 (5.7–10.0)	3.4 (3.1–3.7)	1–2
Saviñao (2012)	90	Schist/loamy	Estrumil/4	Shrubland	530	35 (30–39)	0.38 (0.35–0.43)	14.0 (9.4–16.8)	2.3 (2.2–2.4)	1
Muros (2007)	534	Granite/sandy-loam	Freixo/6	Pine stand/Shrubland	300	31 (28–35)	0.41 (0.36–0.45)	12.0 (5.2–15.5)	1.8 (1.3–2.2)	2
Pontevedra (2006)	125	Granite/sandy-loam	Coirego/2	Shrubland	225	31 (30–32)	0.42 (0.39–0.45)	15.5 (15.0–16.0)	2.0 (1.8–2.1)	2
Pontevedra (2006)	95	Granite/sandy-loam	Sino/4	Eucalypt stand	90	23 (20–27)	0.22 (0.20–0.25)	4.0 (3.5–4.5)	1.9 (1.8–2.2)	1
Verín (2003)	300	Schist/sandy-loam	Queirugas/5	Pine stand	550	33 (25–40)	0.50 (0.45–0.56)	3.0 (2.0–5.7)	2.0 (1.8–2.2)	2

[1]Note: The range of variation is shown in brackets.

some ash deposition; 3, soil organic layer (Oi + Oe + Oa) completely consumed (bare soil) but soil organic matter not consumed and surface soil intact, some ash deposition; 4, soil organic layer completely consumed, soil organic matter in the Ah horizon partially consumed and soil structure altered within a soil thickness less than 1 cm, noticeable ash deposition; 5, as 4, but with the thickness of affected soil equal to or more than 1 cm; and 6, as 4 or 5 and colour altered (reddish). Duff depth, when present, was measured at four points inside the 30 cm × 30 cm quadrats. The mean value of the soil burn severity indicator was computed for each plot as the average of the soil severity numerical levels observed weighted by the percentage presence of the above soil burn severity levels in the transects.

Remnant vegetation linear cover was measured along the earlier-mentioned transects. Percentage cover by rock fragments larger than 2 cm was measured simultaneously at 10 points in the transects with the aid of a 1 m^2 quadrat. Vegetation sampling was repeated every six months. Soil depth was measured with a metal bar at 20 randomly selected points close to each plot.

At each study site, the amount and intensity of rainfall were measured by a recording rain gauge positioned 1.20 m aboveground. The rainfall erosivity factor (Wischmeier and Smith, 1978) was also calculated (Table II).

Statistical analysis

A regression tree (Quiang, 2010) was used to explore the following as explanatory variables for annual soil erosion: soil burn severity, rainfall parameters (annual precipitation, rainfall erosivity factor, mean and maximum rainfall intensity, mean maximum rainfall intensity in 30 minutes and mean and maximum rainfall intensity in 10 minutes), plot characteristics (soil depth, stoniness and slope percentage), vegetation cover and a land use factor.

The land use factor was considered as a dummy variable, with three levels, in accordance with the Galician Forest Service:

i Young pine plantations (> six years). Previous agricultural lands, abandoned 30 years previously. Activities include soil ripping, grading and tilling. Shrubland areas burned in the last five years.
ii Shrubland areas not burned in the previous five years. Forest stands logged in the previous five years.
iii Pine or eucalypt stands (old growth and pole size trees). Well developed litter and duff layers were present in both cases. Shrubland areas not burned for more than 10 years and with a conspicuous soil organic layer.

Soil burn severity, annual precipitation and the land use factor were selected by the regression tree and a non-linear mixed effects model was used to explore their influence on annual soil erosion. Experimental site and plot within each site were considered as random factors in the model. The variance of the error was modelled as a power function of the independent variables.

All statistical analyses were carried out using the R statistical package (Core Team Development, 2014).

The model was validated using the cumulative values ($n = 32$) of soil losses measured in the first year following fire in previous studies carried out in NW Spain (Fernández *et al.*, 2011; Fernández and Vega, 2014; Vega *et al.*, 2014; Vega *et al.*, 2015; Fernández and Vega, 2016). The following criteria were used to test the accuracy and precision of estimates made with the model.

Table II. Mean rainfall parameters during the first year after fire

Wildfire	Precipitation (mm)/ precipitation in first six months (% annual)	Mean rainfall intensity (mm h^{-1})	Mean maximum rainfall intensity in 30 minutes (mm h^{-1})	Mean maximum rainfall intensity in 10 minutes (mm h^{-1})	R (MJ mm ha^{-1} h^{-1} $year^{-1}$)
Ponte-Caldelas	2165–3187/66–74	2.5–2.7	4.2–4.8	7.5–8.4	2978–5370
Oia-O Rosal	2224–2696/66–74	2.8–3.4	4.7–5.4	8.1–9.3	3256–5250
Carnota	1280/69	2.3	3.5	6.3	1255
Negreira	1866/59	1.9	3.9	6.5	2998
Ribasieira	2328/70	2.7	5.1	9.7	4210
Saviñao	900/72	2.0	2.9	4.7	420
Muros	2002/70	2.4	4.5	8.0	3126
Pontevedra (Coirego)	2408/77	3.0	5.0	8.5	3142
Pontevedra (Sino)	2315/75	3.3	5.6	9.0	4420
Verín	674/73	1.6	2.7	5.2	214

Coefficient of determination, $R^2 = 1 - \dfrac{\sum_{i=1}^{n} \dfrac{(O_i - P_i)^2}{n}}{\sum_{i=1}^{n} \dfrac{\left(O_i - \overline{O}\right)^2}{n-1}}$.

Coefficient of efficiency, $E = 1 - \dfrac{\sum_{i=1}^{n} (O_i - P_i)^2}{\sum_{i=1}^{n} \left(O_i - \overline{O}\right)^2}$.

Index of agreement, $d = 1 - \dfrac{\sum_{i=1}^{n}(O_i - P_i)^2}{\sum_{i=1}^{n}\left(|P_i - \overline{O}| + |O_i - \overline{O}|\right)^2}$.

The root mean squared error, $\mathrm{RMSE} = \sqrt{\dfrac{\sum_{i=1}^{n}(O_i - P_i)^2}{n}}$.

Mean absolute error, $\mathrm{MAE} = \frac{1}{n}\sum_{i=1}^{n}|O_i - P_i|$.

Model bias $= \frac{1}{n}\sum_{i=1}^{n}(O_i - P_i)$.

where O_i is the measured data, P_i is the predicted data, $\overline{O}$ is the mean value of measured data.

Results

Post-fire soil loss

Total precipitation was close to the median values for the area in only two of the study sites (Saviñao and Negreira); it was below the median in Verín and higher than the median in the other seven sites. On average, 70% of annual precipitation was collected in the first six months after fire (Table II). The values of the rainfall erosivity factor were close to the mean value for the area in most cases, except in Oia and Ponte-Caldelas, in which it was above the mean value, and in Pindo where it was below the mean value.

First year mean post-fire soil losses varied greatly, from the 0.2 Mg ha^{-1} for the Muros site to the 50.1 Mg ha^{-1} for the Ponte-Caldelas III site (Figure 2). The mean soil burn severity ranged from 1.8 to 4.5 (Table I), shrublands being the vegetation type with the highest average severity of burn. The erosion rates in the study sites were low when soil burn severity was lower than 3.0 and increased sharply when the soil burn severity index was 3.8 or higher (Figure 2).

Most of the study sites were included in class 2 for the land use factor, reflecting a past history of soil perturbation or forestry activities typical of the area.

The non-linear mixed regression analysis revealed that in the first year after fire, the best model to explain soil loss (Table III) included soil burn severity index, annual precipitation and the land use factor, as follows:

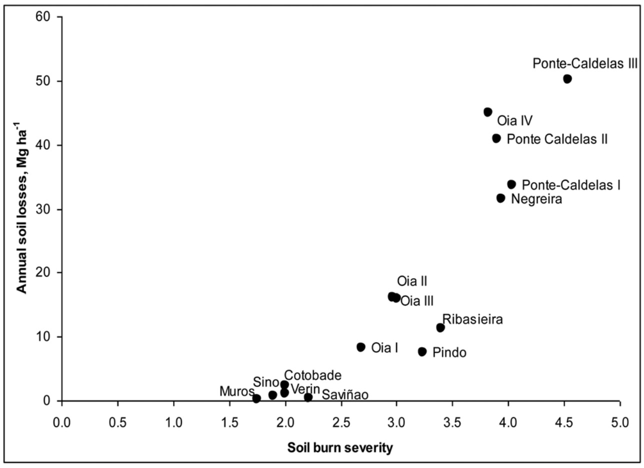

Figure 2. Mean value of the soil burn severity index and mean sediment yield for the first post-fire year in each study site.

Table III. Selection of the soil burn severity model using the Akaike Information Criterion (AIC)

Model	Variables	AIC
1	Soil burn severity	463.92
2	Soil burn severity + annual precipitation	439.47
3	Soil burn severity + annual precipitation + land use factor	389.71

$$\mathrm{SE} = a * e^{(b\ \mathrm{SBSI})} * P * \mathrm{LU}$$

residual standard error = 2.16.

where SE is the soil erosion in the first year after fire (in Mg ha^{-1} $yr.^{-1}$); SBSI is the soil burn severity index value; P the annual precipitation (in millimetres); LU the land use factor; a = 0.0004 (standard error = 0.0001); b = 0.7284 (standard error = 0.0585).

The results showed a reasonably good agreement between observed and predicted values (Figure 3). The model accuracy criteria are shown in Table IV. Validation statistics were similar to those obtained for model calibration (Table IV) although a tendency towards overestimation of the values was detected, particularly for the higher values of sediment yields (Figure 4).

Discussion

The proposed model significantly relates the degree of soil burn severity, estimated by visual indicators, to soil erosion during

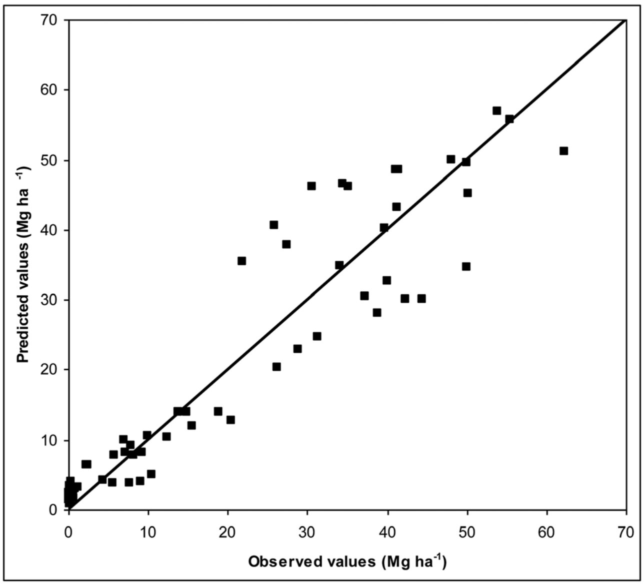

Figure 3. Predicted versus observed sediment production values for the model data set.

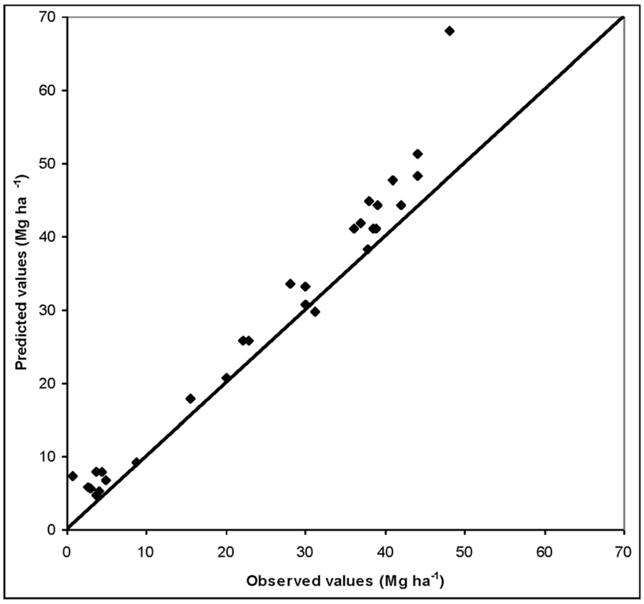

Figure 4. Predicted versus observed sediment production values for the validation data set.

the first year after wildfire in NW of Spain, in shrubland areas and forest stands burned by crown fire. The erosion risk after fire is particularly high in these areas as protection from needle cast or other vegetation residues (Cerdà and Doerr, 2008) is not expected to occur. Although in this study the areas burned at higher soil burn severity were mostly shrublands, a direct relationship between vegetation type and soil burn severity can not be established.

This is the first model that quantitatively relates soil burn severity and soil erosion by using the same soil burn severity indicator for all sites. In a meta-analysis of post-fire rainfall simulation experiments, Vieira *et al.* (2015) found that soil erodibility and soil burn severity were related. However, they could not use the same soil burn severity indicators because the data were obtained from different sources. These authors also pointed out the need to develop common indicators of soil burn severity. Notario del Pino and Ruiz-Gallardo (2015) found that fire severity was the main factor explaining soil erosion risk after fire in south-eastern Spain. However, their estimations of fire severity were based on the degree of vegetation consumption rather than the level of soil alteration immediately after fire. Moreover, the degree of soil erosion was estimated several years after fire.

Previous studies relating fire severity and soil erosion measured in field plots focused on the percentage of bare soil as a surrogate for soil burn severity (Benavides-Solorio and MacDonald, 2005; Vega *et al.*, 2005; Cawson *et al.*, 2013). This variable is indeed useful in semi-desertic areas or under scenarios of low precipitation and sparse vegetation, in areas with no well developed soil organic cover and with low soil organic matter content. However, in NW Spain the structure of the bare soil is often maintained because of the network of rootlets existing as a result of the continuous vegetation cover and the high carbon content of mineral soil (provided the soil organic matter content has not been notably affected). The visual indicators proposed here take into account the role of the remaining soil organic cover plus alterations in two soil properties directly related to the soil erosion potential (water repellency and soil aggregate stability). For soil burn severity levels higher than 4, marked decreases have been observed in soil aggregate stability (Vega *et al.*, 2015) and the occurrence of soil water repellency in the subsurface soil layers (Fernández *et al.*, 2013).

The low sediment yield observed in the plots in which the soil burn severity level was below 3 (indicating bare soil) confirms the protective role of the remaining organic cover in controlling soil erosion after fire (Benavides-Solorio and MacDonald, 2005; Vega *et al.*, 2005; Fernández *et al.*, 2008; Zavala *et al.*, 2009). The presence of soil organic layer can drastically reduce detachment of soil particles by splash erosion. The presence of this layer can also favour water infiltration into the mineral soil, thus reducing overland flow and also acting as an effective zone for detention of water (Martin *et al.*, 2011).

Despite the rough approach used to reflect previous land use as a surrogate for sediment availability, this factor was also found to play a significant role in explaining the soil losses in the first year after fire. Sediment availability has been poorly quantified until know and deserves detailed research attention, as pointed out by Moody *et al.* (2013). Although the qualitative classification used here must be further refined and tested, it seems to indicate that past land use limits sediment availability and affect the post-fire erosive response (Malvar *et al.*, 2015). This could be particularly relevant in NW Spain where a long history of intensive land use, increasing abandonment of land cultivation in recent decades and short fire return periods are all very common factors.

Although the experimental plots are representative of most forested and shrubland areas in the region, soil depth was not

Table IV. Performance of soil burn severity model in predicting overall erosion rates

	Coefficient of determination, R^2	Coefficient of efficiency, *E*	Index of agreement, *d*	Root mean square error (RMSE)	Mean absolute error (MAE)	Bias
Model development	0.89	0.89	0.97	6.16	4.43	0.04
Model validation	0.96	0.81	0.96	7.94	5.47	5.41

selected by the model. It is possible that the range of variation of this variable was too narrow for its influence on post-fire soil losses to be detected. The percentage of stoniness in most of the plots under study (Table I) is probably too low to have significant effects on soil erosion (Poesen and Lavee, 1994; Poesen *et al.*, 1994; Cerdan *et al.*, 2010), although moderate levels of stoniness were found in some sites.

Likewise, although the experimental plots cover a range of slopes of between 20 and 53%, the model did not indicate percentage slope as a determining factor in soil loss. Similar results have been reported by Benavides-Solorio and MacDonald (2005) and Notario del Pino and Ruiz-Gallardo (2015) in different climates. Moody and Martin (2009) used a large data set to analyse post-fire sediment yields in western United States and did not find a significant correlation between this factor and topographic slope. They suggested that sediment availability may be more important than slope in predicting post-fire sediment yields. Our findings seem to confirm this statement.

Vegetation cover did not have a significant effect on soil erosion. This is consistent with previous observations made in the first year after fire in NW Spain (Fernández and Vega, 2016). This may be explained by rainfall distribution as 70% of the annual precipitation accumulates during the autumn and winter after fire (Table II). Thus 81% of annual erosion occurred during the first six months after fire whereas vegetation cover only reached 25% at the end of the same period (Figure 5) and thus the contribution of vegetative cover to soil protection was probably negligible during the first months after fire. Under this scenario, the quantity and characteristics of ash remaining after fire is revealed as being crucial (Bodí *et al.*, 2014).

The model indicated that accumulated precipitation in the first year after fire was more important than other rainfall parameters for explaining soil loss. This confirms previous results in NW Spain in which precipitation proved a better predictor of post-fire soil erosion than rainfall intensity or rainfall erosivity (Fernández *et al.*, 2011; Vega *et al.*, 2014). Annual rainfall was above the background values for the area in most cases, and this may have led overestimation of the erosion values observed in the validation (Figure 4). However, this may be useful given that the objective of the study was to develop an operational tool for prioritizing soil stabilization treatments in areas where the soil erosion risk is high. This is a crucial issue, due to the risk of losing ecosystem sustainability as a result of severe post-fire erosion (Berendse *et al.*, 2015, Brevik *et al.*, 2015, Keesstra *et al.*, 2012).

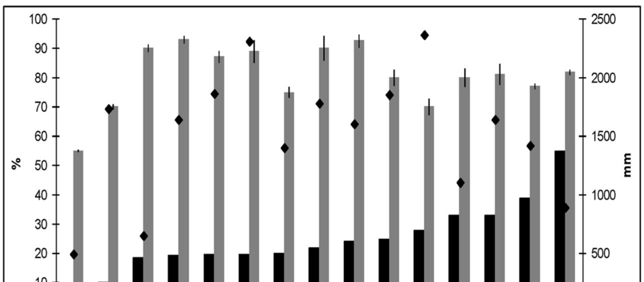

Figure 5. Mean value of vegetation cover six month after fire and the percentage of soil loss during the period relative to the annual soil losses in each study site. Vertical bars represent the standard errors.

Conclusions

The proposed empirical model revealed, for the first time, a significant relationship between soil burn severity and soil erosion during the first year after fire. However, this should be considered as a preliminary finding as accurate quantification of soil burn severity remains a challenge. Nonetheless, the proposed model is simple and provides a first-order estimate of post-fire sediment production in NW Spain and similar environments. It could be used by forest managers to evaluate erosion risks and for planning post-fire stabilization activities. As the predictors are readily available, the model can be used as an operational tool and easily incorporated in a geographic information system (GIS)-environment.

Acknowledgements—The study was funded by the National Institute of Agricultural Research of Spain (INIA) through project RTA2014-00011-C06-02, cofunded by FEDER and the Plan de Mejora e Innovación Forestal de Galicia (2010-2020) and INDITEX.

The authors are grateful to everyone who helped with fieldwork, particularly José Ramón González, Jesús Pardo, Emilia Puga and José Gómez. The authors sincerely acknowledge the anonymous reviewers and Editor for their constructive comments.

References

Arcenegui V, Guerrero C, Mataix-Solera J, Mataix-Beneyto J, Zornoza R, Morales J, Mayoral AM. 2008. The presence of ash as an interference factor in the estimation of the maximum temperature reached in burned soils using near-infrared spectroscopy (NIR). *Catena* **74**: 177–184.

Benavides-Solorio J, MacDonald LH. 2005. Measurement and prediction of post-fire erosion at the hillslope scale, Colorado Front Range. *International Journal of Wildland Fire* **14**: 457–474.

Berendse F, van Ruijven J, Jongejans E, Keesstra S. 2015. Loss of plant species diversity reduces soil erosion resistance. *Ecosystems* **18**: 881–888. DOI:10.1007/s10021-015-9869-6.

Bodí MB, Martin DA, Balfour VN, Santín C, Doerr SH, Pereira P, Cerdà A, Mataix-Solera J. 2014. Wildland fire ash: production, composition and eco-hydro-geomorphic effects. *Earth-Science Reviews* **130**: 103–127.

Bodí MB, Mataix-Solera J, Doerr SH, Cerdà A. 2011. The wettability of ash from burned vegetation and its relationship to Mediterranean plant species type, burn severity and total organic carbon content. *Geoderma* **160**: 599–607.

Brevik EC, Cerdà A, Mataix-Solera J, Pereg L, Quinton JN, Six J, Van Oost K. 2015. The interdisciplinary nature of SOIL. *The Soil* **1**: 117–129. DOI:10.5194/soil-1-117-2015.

Cawson JG, Sheridan GJ, Smith HG, Lane PNJ. 2013. Effects of fire severity and burn patchiness on hillslope-scale surface runoff, erosion and hydrologic connectivity in a prescribed burn. *Forest Ecology and Management* **310**: 219–233.

Cerdá A 1998. Postfire dynamics of erosional processes under Mediterranean climatic conditions. *Zeitschrift für Geomorphologie* **42**: 373–398.

Cerdà A, Doerr SH. 2008. The effect of ash and needle cover on surface runoff and erosion in the immediate post-fire period. *Catena* **74**: 256–263.

Cerdan O, Govers G, Le Bissonnais Y, Van Oost K, Poesen J, Saby N, Gobin A, Vacca A, Quinton J, Auerswald K, Klik A, Kwaad FJPM, Raclot D, Ionita I, Rejman J, Rousseva S, Muxart T, Roxo MJ, Dostal T. 2010. Rates and spatial variations of soil erosion in Europe: a study based on erosion plot data. *Geomorphology* **122**: 167–177.

Core Team Development R. 2014. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing: Vienna.

De Bano LF. 1981. Water Repellent Soils: A State of Art, General Technical Report PSW-46. USDA Forest Service: Washington, DC.

De Bano LF, Neary DG, Ffolliott PF. 1998. Fire's Effects on Ecosystems. John Wiley & Sons: New York.

Food and Agriculture Organization of the United Nations (FAO). 1998. Soil Map of the World. FAO-UNESCO: Rome.

Fernández C, Vega JA. 2014. Efficacy of bark strands and straw mulching after wildfire in NW Spain: effects on erosion control and vegetation recovery. *Ecological Engineering* **63**: 50–57.

Fernández C, Vega JA. 2016. Are erosion barriers and straw mulching effective for controlling soil erosion after a high severity wildfire in NW Spain?. *Ecological Engineering* **87**: 132–138.

Fernández C, Vega JA, Fontúrbel MT. 2013. Severidad del fuego y repelencia al agua en el suelo después de incendios forestales en Galicia. In 6° Congreso Forestal Español. Sociedad Española de Ciencias Forestales: Vitoria(Spain).

Fernández C, Vega JA, Fonturbel T, Jiménez E, Pérez JR. 2008. Immediate effects of prescribed burning, mastication and clearing on runoff, infiltration and erosion in a shrubland area in Galicia (NW Spain). *Land Degradation & Development* **19**: 502–515.

Fernández C, Vega JA, Jiménez E, Fonturbel T. 2011. Effectiveness of three post-fire treatments at reducing soil erosion in Galicia (NW Spain). *International Journal of Wildland Fire* **20**: 104–114.

Fernández C, Vega JA, Vieira DCS. 2010. Assessing soil erosion after fire and rehabilitation treatments in NW Spain: performance of RUSLE and Revised Morgan–Morgan–Finney models. *Land Degradation & Development* **21**: 74–87.

Guénon R, Gros R. 2015. Soil microbial functions after forest fires affected by the compost quality. *Land Degradation & Development*. DOI:10.1002/ldr.2393.

Guénon R, Vennetier M, Dupuy N, Roussos S, Pailler A, Gros R. 2013. Trends in recovery of Mediterranean soil chemical properties and microbial activities after infrequent and frequent wildfires. *Land Degradation & Development* **24**: 115–128. DOI:10.1002/ldr.1109.

Guerrero C, Mataix-Solera J, Arcenegui V, Mataix-Beneyto J, Gómez I. 2007. Near-infrared spectroscopy to estimate the maximum temperatures reached on burned soils. *Soil Science Society of America Journal* **71**: 1029–1037. DOI:10.2136/sssaj2006.0187.

Hedo J, Lucas-Borja ME, Wic C, Andrés-Abellán M, de Las Heras J. 2015. Soil microbiological properties and enzymatic activities of long-term post-fire recovery in dry and semiarid Aleppo pine (Pinus halepensis M.) forest stands. *Solid Earth* **6**: 243–252. DOI:10.5194/se-6-243-2015.

Keesstra SD, Geissen V, Mosse K, Piiranen S, Scudiero E, Leistra M, van Schaik L. 2012. Soil as a filter for groundwater quality. *Current Opinion in Environmental Sustainability* **4**: 507–516.

Keesstra SD, Maroulis J, Argaman E, Voogt A, Wittenberg L. 2014. Effects of controlled fire on hydrology and erosion under simulated rainfall. *Cuadernos de Investigación Geográfica* **40**: 269–293.

Keizer JJ, Doerr SH, Malvar MC, Prats SA, Ferreira RSV, Oñate MG, Coelho COA, Ferreira AJD. 2008. Temporal variation in topsoil water repellency in two recently burnt eucalypt stands in north-central Portugal. *Catena* **74**: 192–204.

Larsen IJ, MacDonald LH. 2007. Predicting postfire sediment yields at the hillslope scale: testing RUSLE and disturbed WEPP. *Water Resources Research* **43**: W11412. DOI:10.1029/2006wr005560.

Lasanta T, Cerdà A. 2005. Long-term erosional responses after fire in the central Spanish Pyrenees: 2. Solute release. *Catena* **60**: 81–100.

León J, Bodí MB, Cerdà A, Badía D. 2013. The contrasted response of ash to wetting: the effects of ash type, thickness and rainfall events. *Geoderma* **209–210**: 143–152.

Malvar MC, Prats SA, Keizer JJ. 2015. Runoff and inter-rill erosion affected by wildfire and pre-fire ploughing in eucalypt plantations of north-central Portugal. *Land Degradation & Development*. DOI:10.1002/ldr.2365.

Martin YE, Johnson EA, Gallaway JM, Chaikina O. 2011. Negligible soil erosion in a burned mountain watershed, Canadian Rockies: field and modelling investigations considering the role of duff. *Earth Surface Processes and Landforms* **36**: 2097–2113. DOI:10.1002/esp.2236.

Mataix-Solera J, Cerdà A, Arcenegui V, Jordán A, Zavala LM. 2011. Fire effects on soil aggregation: a review. *Earth-Science Reviews* **109**: 44–60.

Miller JD, Nyhan JW, Yool SR. 2003. Modeling potential erosion due to the Cerro Grande Fire with a GIS-based implementation of the Revised Universal Soil Loss Equation. *International Journal of Wildland Fire* **12**: 85–100.

Moody JA, Ebel BA. 2012. Hyper-dry conditions provide new insights into the cause of extreme floods after wildfire. *Catena* **93**: 58–63.

Moody JA, Martin DA. 2009. Synthesis of sediment yields after wildland fire in different rainfall regimes in the western United States. *International Journal of Wildland Fire* **18**: 96–115.

Moody JA, Shakesby RA, Robichaud PR, Cannon SH, Martin DA. 2013. Current research issues related to post-wildfire runoff and erosion processes. *Earth-Science Reviews* **122**: 10–37.

Morgan P, Keane RE, Dillon GK, Jain TB, Hudak AT, Karau EC, Sikkink PG, Holden ZA, Strand EK. 2014. Challenges of assessing fire and burn severity using field measures, remote sensing and modelling. *International Journal of Wildland Fire* **23**: 1045–1060.

Morgan RPC, Quinton JN, Smith RE, Govers G, Poesen JWA, Auerswald K, Chisci G, Torri D, Styczen ME. 1998. The European Soil Erosion Model (EUROSEM): a dynamic approach for predicting sediment transport from fields and small catchments. *Earth Surface Processes and Landforms* **23**: 527–544. DOI:10.1002/(sici)1096–9837(199806)23:6 < 527::aid-esp868 > 3.0.co;2–5.

Myronidis D, Emmanouloudis D, Mitsopoulos I, Riggos E. 2010. Soil erosion potential after fire and rehabilitation treatments in Greece. *Environmental Modeling and Assessment* **15**: 239–250. DOI:10.1007/s10666-009-9199-1.

Nearing MA, Foster GR, Lane LJ, Finkner SC. 1989. A process-based soil erosion model for USDA-water erosion prediction project technology. *American Society of Agricultural Engineers* **32**: 1587–1593.

Neris J, Doerr SH, Tejedor M, Jiménez C, Hernández-Moreno JM. 2014. Thermal analysis as a predictor for hydrological parameters of fire-affected soils. *Geoderma* **235–236**: 240–249.

Notario del Pino JS, Ruiz-Gallardo J-R. 2015. Modelling post-fire soil erosion hazard using ordinal logistic regression: a case study in south-eastern Spain. *Geomorphology* **232**: 117–124.

Novara A, Gristina L, Bodì MB, Cerdà A. 2011. The impact of fire on redistribution of soil organic matter on a Mediterranean hillslope under maquia vegetation type. *Land Degradation & Development* **22**: 530–536. DOI:10.1002/ldr.1027.

Novara A, Gristina L, Rühl J, Pasta S, D'Angelo G, La Mantia T, Pereira P. 2013. Grassland fire effect on soil organic carbon reservoirs in a semiarid environment. *Solid Earth* **4**: 381–385. DOI:10.5194/se-4-381-2013.

Pereira P, Cerdà A, Úbeda X, Mataix-Solera J, Martin D, Jordán A, Burguet M. 2013. Spatial models for monitoring the spatio-temporal evolution of ashes after fire – a case study of a burnt grassland in Lithuania. *Solid Earth* **4**: 153–165. DOI:10.5194/se-4-153-2013.

Pereira P, Úbeda X, Mataix-Solera J, Oliva M, Novara A. 2014. Short-term changes in soil Munsell colour value, organic matter content and soil water repellency after a spring grassland fire in Lithuania. *Solid Earth* **5**: 209–225. DOI:10.5194/se-5-209-2014.

Pérez-Cabello F, Cerdà A, de la Riva J, Echeverría MT, García-Martín A, Ibarra P, Lasanta T, Montorio R, Palacios V. 2012. Micro-scale post-fire surface cover changes monitored using high spatial resolution photography in a semiarid environment: a useful tool in the study of post-fire soil erosion processes. *Journal of Arid Environments* **76**: 88–96.

Poesen J, Lavee H. 1994. Rock fragments in top soils: significance and processes. *Catena* **23**: 1–28.

Poesen JW, Torri D, Bunte K. 1994. Effects of rock fragments on soil erosion by water at different spatial scales: a review. *Catena* **23**: 141–166.

Quiang SS. 2010. Environmental and Ecological Statistics with R. Chapman & Hall/CRC: Boca Raton, FL.

Renard KG, Foster GR, Weesies GA, Mc Cool DK, Yoder DC. 1997. Predicting Soil Erosion by Water: A Guide to Conservation Planning with the Revised Universal Soil Loss Equation (RUSLE). USDA, Natural Resources Conservation Service: Washington, DC.

Robichaud PR, Brown RE. 2002. Silt Fences: An Economical Technique for Measuring Hillslope Soil Erosion, General Technical Report RMRS-GTR-94. USDA Forest Service: Washington, DC.

Rulli MC, Offeddu L, Santini M. 2013. Modeling post-fire water erosion mitigation strategies. *Hydrology and Earth System Sciences* **17**: 2323–2337. DOI:10.5194/hess-17-2323-2013.

Shakesby RA. 2011. Post-wildfire soil erosion in the Mediterranean: review and future research directions. *Earth-Science Reviews* **105**: 71–100.

Shakesby RA, Doerr SH. 2006. Wildfire as a hydrological and geomorphological agent. *Earth-Science Reviews* **74**: 269–307.

Tessler N, Sapir Y, Wittenberg L, Greenbaum N. 2015. Recovery of Mediterranean vegetation after recurrent forest fires: insight from the 2010 forest fire on Mount Carmel, Israel. *Land Degradation & Development*. DOI:10.1002/ldr.2419.

Vega JA, Fernández C, Fontúrbel MT, González-Prieto SJ, Jiménez E. 2014. Testing the effects of straw mulching and herb seeding on soil erosion after fire in a gorse shrubland. *Geoderma* **223–225**: 79–87.

Vega JA, Fernández C, Fonturbel T. 2005. Throughfall, runoff and soil erosion after prescribed burning in gorse shrubland in Galicia (NW Spain). *Land Degradation & Development* **15**: 1–15.

Vega JA, Fernández C, Fonturbel T. 2015. Comparing the effectiveness of seeding and mulching + seeding in reducing soil erosion after a high severity fire in Galicia (NW Spain). *Ecological Engineering* **74**: 206–212.

Vega JA, Fontúrbel MT, Fernández C, Arellano A, Díaz-Raviña M, Carballas T, Martín A, González-Prieto S, Merino A, Benito E. 2013a. Acciones urgentes contra la erosión en áreas forestales quemadas. Guía para su planificación en Galicia Santiago de Compostela: Santiago de Compostela.

Vega JA, Fontúrbel MT, Merino A, Fernández C, Ferreiro A, Jiménez E. 2013b. Testing the ability of visual indicators of soil burn severity to reflect changes in soil chemical and microbial properties in pine forests and shrubland. *Plant and Soil* **369**: 73–91. DOI:10.1007/s11104-012-1532-9.

Vieira DCS, Fernández C, Vega JA, Keizer JJ. 2015. Does soil burn severity affect the post-fire runoff and interrill erosion response? A review based on meta-analysis of field rainfall simulation data. *Journal of Hydrology* **523**: 452–464.

Wang C, Wang G, Wang Y, Rafique R, Ma L, Hu L, Luo Y. 2015. Fire alters vegetation and soil microbial community in alpine meadow. *Land Degradation & Development*. DOI:10.1002/ldr.2367.

Williams CJ, Pierson FB, Al-Hamdan OZ, Kormos PR, Hardegree SP, Clark PE. 2014. Can wildfire serve as an ecohydrologic threshold-reversal mechanism on juniper-encroached shrublands. *Ecohydrology* **7**: 453–477. DOI:10.1002/eco.1364.

Wischmeier WH, Smith DD. 1978. Predicting Rainfall–Erosion Losses – a Guide to Conservation Planning, Agriculture Handbook N°537. USDA Forest Service: Washington, DC.

Zavala LM, Jordán A, Gil J, Bellinfante N, Pain C. 2009. Intact ash and charred litter reduces susceptibility to rain splash erosion post-wildfire. *Earth Surface Processes and Landforms* **34**: 1522–1532. DOI:10.1002/esp.1837.