Catena 163 (2018) 276–287

Contents lists available at ScienceDirect

Catena

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

Empirical models of annual post-fire erosion on mulched and unmulched hillslopes

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ARTICLE INFO

Keywords:
Hillslope erosion
High Park Fire
Wildfire
Bare soil
Erosion modeling
Mulch

ABSTRACT

Erosion is one of the primary land management concerns following wildfire. This study examines controls on post-fire hillslope-scale erosion for the 2012 High Park Fire in northern Colorado, develops simple empirical models for predicting post-fire sediment yields, and evaluates model performance on several nearby fires. From 2013 to 2015 we collected ground cover, rainfall, topographic, and sediment yield measurements from 29 convergent hillslopes; eight of these hillslopes had varying amounts of mulch applied to reduce erosion. From these data we examined correlations between annual sediment yield and three categories of predictor variables (ground cover, precipitation, and topography). Percent bare soil was the single largest control on sediment yield, followed by rainfall variables. Sediment yield generally decreased with flow path length, but the correlation was weak. The empirical models each predicted sediment yield with three variables: percent bare soil, one precipitation variable, and one topographic variable. The models had similar accuracy for the High Park Fire using varying combinations of precipitation and topographic variables (Nash-Sutcliffe coefficients 0.70–0.84). An empirical model predicting annual sediment yields as a function of percent bare soil, June–October precipitation, and the maximum flow path length had variable performance when applied to other fires in the same region, with predictions ranging from poor to good for individual fires and Nash-Sutcliffe coefficients of 0.26–0.32 for all fires combined. These tests show some promise for applying the empirical model to fires in the study region, but further model testing is needed to determine the range of conditions under which the model can be applied.

1. Introduction

Wildfires are increasing in frequency, extent, and severity in many regions throughout the world (Flannigan et al., 2009; Miller et al., 2009; Dennison et al., 2014). Elevated erosion after wildfire can impact downstream water quality, fill reservoirs, and damage aquatic habitat (Shakesby and Doerr, 2006; Goode et al., 2012), so land managers need to predict reliably which areas on the landscape have high post-fire erosion risk. Empirical regression models have been developed to predict post-fire erosion rates at individual research sites (Benavides-Solorio and MacDonald, 2005; Pietraszek, 2006), but these often use the particular variables collected in the study area and are not easily transferred to other fires. Process-based hillslope erosion models such as RUSLE (Renard et al., 1997), Disturbed WEPP (Elliott, 2004), and ERMiT (Robichaud et al., 2007) have shown variable performance for predicting annual post-fire erosion rates on individual hillslopes (Larsen and MacDonald, 2007; Fernández et al., 2010; Robichaud et al., 2016), so there is still a role for ongoing research on post-fire erosion prediction, particularly given the wide range of post-fire conditions.

Post-fire emergency response teams around the world have developed excellent tools for determining what parts of the landscape are most vulnerable to erosion after fire (Robichaud et al., 2007; Goodrich et al., 2005; Vafeidis et al., 2007; Van Eck et al., 2016), yet sediment yield remains one of the most difficult physical variables to predict accurately. Inaccuracies in sediment yield predictions can relate to the quality of input and sediment yield data, scale of application, and adequacy of model structure. Hillslope erosion models generally require inputs related to ground cover, soil erodibility, precipitation, and topography (Woolhiser et al., 1990; Renard et al., 1997; Elliott, 2004). Research projects on post-fire erosion have collected ground cover information through field surveys and derived soil properties from field samples (Benavides-Solorio and MacDonald, 2005; Pietraszek, 2006),

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https://doi.org/10.1016/j.catena.2017.12.029
Received 9 February 2017; Received in revised form 20 December 2017; Accepted 22 December 2017

but it is infeasible to collect such field data over entire watersheds. Soil properties are particularly difficult to represent accurately because they are heterogeneous across hillslopes (Russo et al., 1997), change as a result of burning and burn severity (Larsen and MacDonald, 2007), and can change over time due to surface armoring (e.g., Morris and Moses, 1987; Schaffrath, 2009). Where detailed field measurements of ground cover and soils are lacking, post-fire erosion model applications derive ground cover from land cover and burn severity maps and soil properties from soil survey maps (Vafeidis et al., 2007; Terranova et al., 2009). Some erosion model applications have developed look-up tables that use pre-fire land cover and burn severity to predict ground cover and changes in soil properties (Elliott, 2004; Canfield and Goodrich, 2005).

Rainfall intensity is a key control on hillslope erosion after burning (Spigel and Robichaud, 2007; Kampf et al., 2016), but many areas lack the fine resolution rainfall data needed to capture spatial and temporal storm patterns. Post-fire erosion models therefore use a variety of estimation strategies for quantifying the precipitation input. For example, applications of the Revised Universal Soil Loss Equation (RUSLE) typically use mean annual rainfall erosivity, which combines the kinetic energy of total rainfall with maximum storm intensity (Brown and Foster, 1987). The Water Erosion Prediction Project (WEPP) model generally uses stochastic rainfall intensity characteristics generated from daily long-term precipitation data (Flanagan and Livingston, 1995), or it can directly use rain gauge data. Similarly, the KINEROS2 erosion model uses rainfall hyetographs from either rain gauge measurements or design storms (Woolhiser et al., 1990).

Topographic variables such as hillslope length and slope may be the easiest to obtain after fires because of the widespread availability of digital elevation data, although these variables are affected by the scale of topographic data (Thompson et al., 2001). Commonly applied hillslope erosion models such as RUSLE and WEPP were developed and calibrated to data collected primarily from relatively small plots. The standard plots used to develop the Universal Soil Loss Equation were 22 m long (Renard et al., 2011), and many of the plots used to develop WEPP were 9–11 m long (Laflen et al., 1991). For post-fire applications hillslope erosion assessments are usually needed over much larger areas for which field measurements of erosion are more difficult to obtain.

The goal of this research is to develop and test simple empirical hillslope-scale erosion prediction models that use commonly measured post-fire variables. We develop the models based on field data collected in the area burned by the 2012 High Park Fire in north-central Colorado and test them using field data from five other fires in the region. Our specific objectives are to: (1) quantify post fire ground cover, rainfall, and annual sediment yields for unmulched and mulched hillslopes; (2) examine how ground cover, rainfall, and topographic variables relate to annual hillslope sediment yields; (3) develop empirical models to predict annual hillslope-scale sediment yields; and (4) test the performance of these models against measured sediment yields from other fires in the Colorado Front Range.

2. Study site

The High Park Fire burned in June 2012, and its perimeter encompassed about 350 km^2 of mostly forested land west of Fort Collins, Colorado (BAER, 2012) (Fig. 1). Our study sites were located within two ~15 km^2 watersheds, Skin Gulch and Hill Gulch, that each had about 65–70% of their area burned at high or moderate severity. These watersheds were selected because they have similar size, aspect and burn severity, and both drain directly to the Cache la Poudre River. Hill Gulch is the eastern watershed with elevations ranging from 1740 to 2380 m, and Skin Gulch is slightly larger and higher with an elevation range of 1890 to 2580 m. Prior to burning the vegetation at lower elevations was primarily ponderosa pine (*Pinus ponderosa*) with grasses and shrubs in drier south-facing and lower elevation areas. Higher elevations in the burned area had a denser mixed conifer forest (BAER, 2012). The primary soil within the watersheds is Redfeather sandy loam (BAER, 2012), and this is a shallow to moderately deep (40–100 cm), well-drained sandy loam formed on granitic bedrock; the taxonomic description is loamy-skeletal, mixed, superactive Lithic Glossocryalfs (Moreland, 1980; USDA NRCS, 1998). Soils are often shallow and rocky, with rock outcrops on many of the steeper hillslopes and ridgetops.

Average annual precipitation in the burned area ranges from approximately 440 to 600 mm (PRISM Climate Group), and precipitation falls as snow during the winter months. Summer rain events are typically spatially variable, high-intensity convective storms. The area also experiences occasional low-intensity frontal storms, particularly in the spring and fall (MacDonald and Stednick, 2003). Previous studies have shown that nearly all of the post-fire erosion in this region results from higher-intensity summer thunderstorms rather than snowmelt or lower intensity frontal storms (Benavides-Solorio and MacDonald, 2005; Wagenbrenner et al., 2015).

3. Methods

3.1. Field methods

After the fire we installed 29 sediment fences (similar to Robichaud and Brown, 2002) to capture post-fire erosion from convergent hillslopes. Fences were installed in areas with moderate to high burn severity in the central axes of convergent hillslopes. Locations were selected to represent a range of slope lengths, slope angles and contributing areas. Contributing areas to the fences were delineated in the field using a Juno Trimble handheld GPS with horizontal accuracy of < 5 m. These field delineations gave hillslope contributing areas ranging from 0.1–1.5 ha (Table 1). For nine hillslopes with particularly large contributing areas or long slope lengths, we installed two fences in succession to increase sediment storage capacity. We established twenty-one of the sediment fences in August–September 2012 and eight in May–June 2013. The measured hillslopes were in five clusters of 4–7 sites at different elevations of each of the study watersheds (Fig. 1; Table 1). Each hillslope ID begins with the name of the watershed (S for Skin Gulch; H for Hill Gulch) followed by the elevation zone (L for lower; M for middle; U for upper) and the hillslope number (Fig. 2; Table 1).

Each hillslope cluster was co-located with one or more Rainwise tipping bucket rain gauges with a 0.25 mm resolution and data loggers to record the time of each tip. Sediment fences were located 10–830 m away from the nearest rain gauge. To the extent possible, we checked fences for sediment after each rain storm and at the beginning (late October) and end (late April) of the winter snowmelt season from 2013 to 2015. During each site visit we manually removed trapped sediment and measured the field mass to the nearest 0.5 kg on a hanging scale. For each sediment measurement we collected a representative soil sample, dried it in the lab to determine the water content, and used the water content to convert the wet sediment mass to a dry mass. We estimate that the soil samples contained < 1% organic matter, although another study on severely burned slopes in a more humid climate measured 5–7% organic matter (Robichaud et al., 2006). The dry mass was divided by the contributing area to obtain sediment yield (*SY*) in Mg ha^{-1}.

Management agencies applied wood shred mulch at a planned rate of 6 Mg ha^{-1} to four of the studied hillslopes (HU1–4) in November 2012 (Fig. 2b) and straw mulch in June 2013 at a planned rate of 3–4 Mg ha^{-1}. The wood shreds had stubble lengths of 10–20 cm, < 3 cm diameter, and minimal fines (NRCS, 2013). Straw mulch was applied to two of the hillslopes that had wood shred mulch (HU3,4) and four additional hillslopes (HL5,6; SU1,2). Of the latter four sites, straw coverage was sparse and clumpy on HL5 and HL6 and dense and evenly distributed on SU1 and SU2 (Fig. 2a). Four hillslopes in HL (HL1–4) also had scattered mulch cover (≤ 5%) in June 2013 that had blown in from