Universidade de Aveiro Departamento de Ambiente e Ordenamento
Ano 2015

Diana Catarina Simões Vieira

Estudo e modelação dos processos hidrológicos e erosivos em bacias hidrográficas ardidas

Understanding and modelling hydrological and soil erosion processes in burnt forest catchments

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Tese apresentada à Universidade de Aveiro para cumprimento dos requisitos necessários à obtenção do grau de Doutor em Ciências e Engenharia do Ambente, realizada sob a orientação científica do Doutor Jan Jacob Keizer, Investigador Auxiliar do CESAM no Departamento de Ambiente e Ordenamento da Universidade de Aveiro. A co-orientação foi realizada pelo Doutor João Pedro Nunes, bolseiro de pos-doc pela Fundação para a Ciência e Tecnologia de Portugal, e pela Doutora Cristina Fernández, investigadora pelo Centro de Investigación Forestal de Lourizán.

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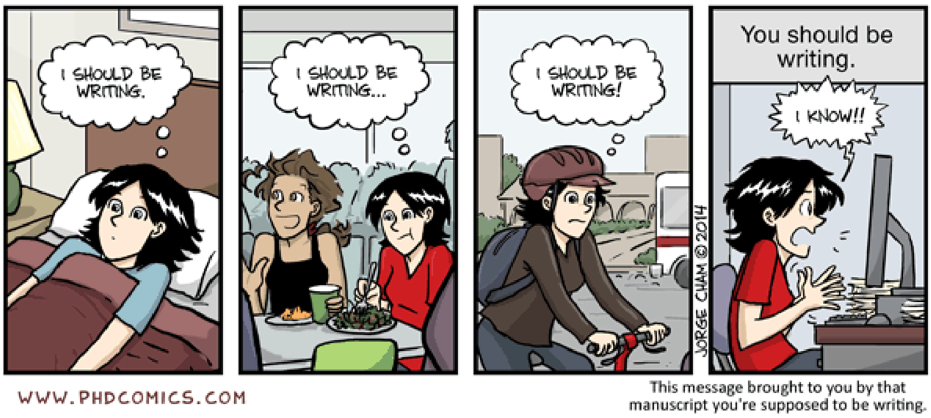
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palavras-chave fogos florestais, severidade de queima, escorrência, erosão, modelação

resumo

As implicações dos fogos florestais na escorrência superficial e erosão dos solos têm sido objeto de estudo desde há vários anos. Como tal, é do conhecimento geral, que os fogos tendem a aumentar a atividade hidrológica e geomorfológica em todo o mundo e também nas zonas mediterrânicas. A severidade da queima do solo tem sido utilizada para descrever o impacto dos fogos nos solos e reconhecida como um fator decisivo no controle das taxas de erosão pós-fogo. No entanto, não existe uma definição única do termo e a relação entre severidade de queima do solo com a resposta hidrológica e erosiva não é ainda totalmente conhecida. Por outro lado, escasseiam os estudos com registos de taxas de erosão pós-fogo durante um período de quatro anos, nenhum dentro desse período com registos de escorrência superficial pós-fogo. Menos estudos ainda, que retratem a resposta erosiva pós-fogo, mencionando práticas de gestão florestal anteriores ao mesmo. No caso da modelação de erosão dos solos, apesar dos modelos aplicados – a Equação Universal de Perdas do Solo Revista (RUSLE) e o modelo de Morgan-Morgan-Finney (MMF) – serem bem conhecidos, a informação referente à sua aplicabilidade para prever taxas de erosão em solos florestais após o fogo é bastante limitada. No caso da aplicabilidade destes modelos, considerando tratamentos de mitigação após incêndio, ainda menos informação existe.

O objetivo deste trabalho é o aprofundar do conhecimento relativo à resposta hidrológica e erosiva após incêndios florestais através do estudo dos efeitos da severidade de queima nos ecossistemas e das suas implicações na resposta hidrológica e erosiva em todo o mundo. Para este fim, testámos também o efeito de diferentes práticas de gestão florestal (não lavrado, lavrado no sentido do declive e lavrado segundo as curvas de nível), executadas previamente ao incêndio florestal, entre dois dos usos do solo mais comuns em Portugal: o pinheiro e o eucalipto. Testámos ainda a eficiência com que dois modelos, amplamente conhecidos (RUSLE e MMF revisto), conseguem prever, em duas severidades distintas e com tratamentos de reabilitação pós fogo, as taxas de erosão durante o ano que seguiu ao incêndio florestal. Com essa informação, que veio melhorar as estimativas, alterámos o modelo e verificámos a sua eficiência, tanto nas previsões de escorrência superficial como na erosão do solo em pós-fogo e em pós-fogo com tratamentos de reabilitação.

Essas alterações, que consistiam em (1) passar todos os *inputs* numa escala sazonal para incorporar as variações sazonais sentidas na formação de escorrência superficial e erosão do solo, e (2) inclusão do efeito hidrófobo do solo à água nas previsões da escorrência superficial. Adicionalmente, validar estas melhorias noutra área florestal independente no centro de Portugal para pinhal e eucaliptal, pós-fogo e pós-fogo com tratamentos de reabilitação.

A revisão e a meta-análise demonstraram que a ocorrência de um fogo florestal provoca alterações significativas na resposta hidrológica e erosiva. No entanto, este efeito só é significativamente diferente com o aumento da severidade da queima do solo para a erosão e não para a geração de escorrência superficial. Este estudo também aludiu a incoerência entre várias classificações de severidade de queima e propõe ainda uma classificação não ambígua.

No caso das parcelas de erosão com chuva natural, verificou-se que o uso do solo é um fator que afeta a geração de escorrência; em contrapartida, a gestão florestal afeta tanto a escorrência como a erosão do solo. O tempo decorrido desde o incêndio surge como fator de elevada importância entre locais não lavrados, relativamente às perdas de solo, e entre eucaliptais, relativamente à escorrência e erosão. Em todos os locais os coeficientes de escorrência aumentaram do primeiro para o quarto ano de estudo. Noutra nota, notou-se um decréscimo nas concentrações de sedimentos na escorrência durante o mesmo período. Foi explorada a discrepância entre este estudo e entre os modelos clássicos de recuperação pós-fogo; também o curto intervalo entre fogos e as constantes práticas de gestão florestal são vistas como as principais razões pela severa e continuada degradação dos solos.

O modelo de MMF revisto apresentou uma razoável acuidade nas previsões enquanto que, o RUSLE claramente sobrestimou as taxas de erosão observadas. Ambos os modelos demonstraram capacidades para serem usados como ferramentas operacionais para ajudarem gestores a determinar áreas de risco de erosão pós-fogo e a tomarem ações prioritárias. O Modelo MMF revisto permitiu determinar as taxas de erosão durante o primeiro ano, após o fogo, para os dois usos do solo estudados: o pinheiro e o eucalipto. Essas previsões melhoraram com a implementação da modelação sazonal e com a inclusão da hidrofobia do solo à água para as previsões de escorrência. Por fim, o modelo de MMF revisto provou ser capaz de providenciar um conjunto de critérios para ajudar à tomada de decisões por parte dos gestores relativamente à escorrência, erosão e tratamentos de mitigação em áreas recentemente ardidas. Este modelo sugere, segundo os resultados obtidos aquando da validação e calibração, uma elevada robustez e um potencial de ser aplicado a outras áreas.

keywords forest fires, burn severity, runoff, erosion, modelling

abstract

Forest fires implications in overland flow and soil erosion have been researched for several years. Therefore, is widely known that fires enhance hydrological and geomorphological activity worldwide as also in Mediterranean areas. Soil burn severity has been widely used to describe the impacts of fire on soils, and has being recognized as a decisive factor controlling post-fire erosion rates. However, there is no unique definition of the term and the relationship between soil burn severity and post-fire hydrological and erosion response has not yet been fully established. Few studies have assessed post-fire erosion over multiple years, and the authors are aware of none which assess runoff. Small amount of studies concerning pre-fire management practices were also found. In the case of soil erosion models, the Revised Universal Soil Loss Equation (RUSLE) and the revised Morgan–Morgan–Finney (MMF) are well-known models, but not much information is available as regards their suitability in predicting post-fire soil erosion in forest soils. The lack of information is even more pronounced as regards post-fire rehabilitation treatments.

The aim of the thesis was to perform an extensive research under the post fire hydrologic and erosive response subject. By understanding the effect of burn severity in ecosystems and its implications regarding post fire hydrological and erosive responses worldwide. Test the effect of different pre-fire land management practices (unplowed, downslope plowed and contour plowed) and time-since-fire, in the post fire hydrological and erosive response, between the two most common land uses in Portugal (pine and eucalypt). Assess the performance of two widely-known erosion models (RUSLE and Revised MMF), to predict soil erosion rates during first year following two wildfires of distinctive burn severity. Furthermore, to apply these two models considering different post-fire rehabilitation treatments in an area severely affected by fire. Improve model estimations of post-fire runoff and erosion rates in two different land uses (pine and eucalypt) using the revised MMF.

To assess these improvements by comparing estimations and measurements of runoff and erosion, in two recently burned sites, as also with their post fire rehabilitation treatments. Model modifications involved: (1) focusing on intra-annual changes in parameters to incorporate seasonal differences in runoff and erosion; and (2) inclusion of soil water repellency in runoff predictions. Additionally, validate these improvements with the application of the model to other pine and eucalypt sites in Central Portugal.

The review and meta-analysis showed that fire occurrence had a significant effect on the hydrological and erosive response. However, this effect was only significantly higher with increasing soil burn severity for inter-rill erosion, and not for runoff. This study furthermore highlighted the incoherencies between existing burn severity classifications, and proposed an unambiguous classification.

In the case of the erosion plots with natural rainfall, land use factor affected annual runoff while land management affected both annual runoff and erosion amounts significantly. Time-since-fire had an important effect in erosion amounts among unplowed sites, while for eucalypt sites time affected both annual runoff and erosion amounts. At all studied sites runoff coefficients increase over the four years of monitoring. In the other hand, sediment concentration in the runoff, recorded a decrease during the same period. Reasons for divergence from the classic post-fire recovery model were also explored. Short fire recurrence intervals and forest management practices are viewed as the main reasons for the observed severe and continuing soil degradation.

The revised MMF model presented reasonable accuracy in the predictions while the RUSLE clearly overestimated the observed erosion rates. After improvements: the revised model was able to predict first-year post-fire plot-scale runoff and erosion rates for both forest types, these predictions were improved both by the seasonal changes in the model parameters; and by considering the effect of soil water repellency on the runoff, individual seasonal predictions were considered accurate, and the inclusion of the soil water repellency in the model also improved the model at this base. The revised MMF model proved capable of providing a simple set of criteria for management decisions about runoff and erosion mitigation measures in burned areas. The erosion predictions at the validation sites attested both to the robustness of the model and of the calibration parameters, suggesting a potential wider application.

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I. Introduction

I.I Wildfire concerns worldwide

Wildfires have been considered as an important, if not the major, cause of hydrological and geomorphological change in fire-prone landscapes (Shakesby and Doerr, 2006). The direct effects from wildfires such as vegetation and litter cover removal together with soil physical and chemical alterations are usually described as primary observed changes from wildfires (Figure 1). These changes are followed by the indirect hydrological and geomorphological effects, such as reduced infiltration and increased sediment availability for transport. Ultimately, these effects will lead to an increase of overland flow generation and soil erosion (Figure 2) (Shakesby and Doerr, 2006).

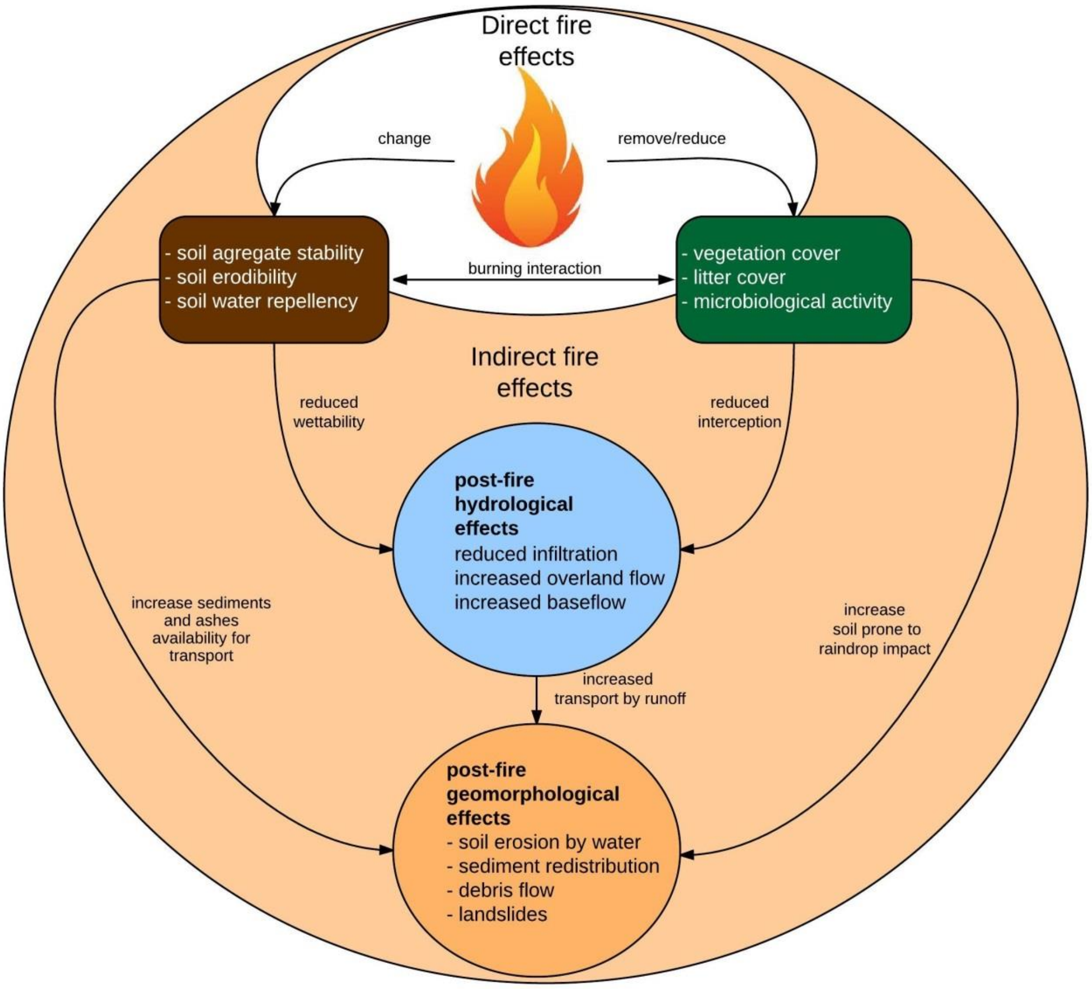


Figure 1 –Main direct fire effects on soil surface and aboveground litter and vegetation, and indirect effects to hydrological and geomorphological processes during the post-fire period.

The interest in this line of research started approximately in 1930's (e.g. Connaughton, 1935; Hendricks and Johnson, 1944) in the USA, and elevated dramatically with one of the most emblematic publication on fire and its impacts that became known as the "Rainbow Series." The series consisted of six publications, each with a different coloured cover, describing the effects of fire on soil (Wells et al., 1979), water (Tiedemann et al., 1979), air (Sandberg et al., 1979), flora (Lotan et al., 1981), fauna (Lyon et al., 1978), and fuels (Martin et al., 1979), providing a wealth of information and examples to advance understanding of basic concepts regarding fire effects in the United States and Canada (Neary et al., 2005). Several reviews dealing with post-wildfire hydrology and soil erosion worldwide have been published since then (e.g. Anderson et al., 1976; Swanson, 1981; Robichaud et al., 2000; Neary et al., 2005; Shakesby and Doerr, 2006; Moody et al., 2013), but few of them focused on the specific case of Mediterranean climate regions(Pausas et al., 2008; Shakesby, 2011). Publications related to post-fire studies in the Mediterranean initiated could only be found from the early 1980s onwards, following the beginning of the dramatic increase in fire activity (Moreno et al., 1998; Pausas, 2004). This rise in fire activity, resulting in 600,000 ha burnt annually by 50,000 ignitions by the end of the century (Lloret et al., 2009), allowed to view wildfire a natural phenomenon in regions with a Mediterranean-type climate (Naveh, 1990) and contributed to the increase of interest in this research line.

The increase in sediment losses following wildfire already has been reported by several authors (Swanson, 1981; Scott and Van Wyk, 1990; Robichaud and Brown, 1999; Moody and Martin, 2001; Meyer et al., 2001; Benavides-Solorio and MacDonald, 2005), and can be described by the window of disturbance model (Figure 2). This model represents a simplification of the sediment yield response to the "new" fire induced hydro-geomorphic regime. However, the contribution of each fire induced change to the post-fire hydrological and erosive response is still not fully understood. Although the amount of research about this subject increased dramatically in the last 20 years, some comparability difficulties between studies still arise. This was verified by Moody et al. (2013) when comparing existing studies that identified distinct key processes regarding post-fire erosion. Some studies presented sediment contribution by channel erosion as the main source of post-fire erosion, while others attributed their main source to hillslope erosion. Runoff generation could be also originated by infiltration-excess in some studies while in others saturation-excess overland flow was the dominant process. The reasons for these discrepancies were mostly attributed to differences in fire regimes, precipitation regimes, hydro-geomorphic regimes and post-fire response domains (Moody et al., 2013). Fire behaviour and recurrence, climate conditions (precipitation amounts) during the post-fire period, fire-induced changes (Figure 1) according to different burn severities, and ecosystems specific characteristics combinations, can difficult the comparability among studies.

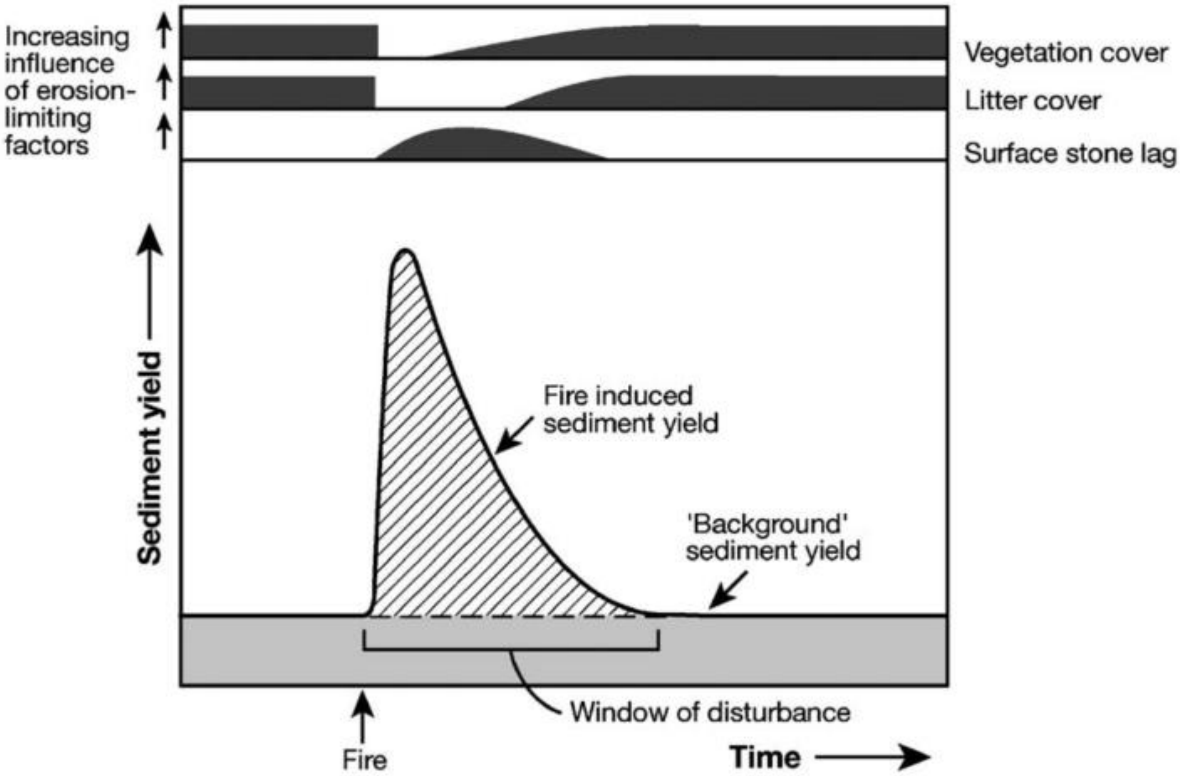


Figure 2 - The hypothetical decline in sediment yield after wildfire, and the role of three other factors (vegetation cover, litter cover, and stone lag development) in reducing erosion rates (Shakesby and Doerr 2006).

The number and severity of wildfires all over the world have become a major concern in recent decades (Moody et al., 2013). In the Portuguese case, on average, wildfires consume each year 100,000 ha in Portugal (Pereira et al., 2006a), from the 500,000 ha in southern Europe (San-Miguel and Camia, 2009) (Figure 3). Fire activity in Portugal is not expected to decline markedly in the foreseeable future, both because of the economic importance of the country's forestry activities using flammable species and the likely increase in the occurrence of meteorological conditions conducive to fire (Carvalho et al., 2010; Pereira et al., 2006b; Harding et al., 2009). Galicia (NW Spain) faces a similar problem as Portugal, since about 8,000 fires per year burned in the period 2001-2010 (MMA, 2010) (Figure 3). Additionally, wildfire frequency, severity and the size of burned areas are expected to increase under the probable future climate scenarios in NW Spain (Moreno, 2005; Carvalho et al., 2008; Good et al., 2008; Moreno, 2009; Vega et al., 2009).

Wildfire occurrence raises some concerns mostly due to the above mentioned direct and indirect effects (Figure 1), but also due to concerns of fire effects on carbon storage, water quality, and ecosystem disturbances. Additionally, there are also concerns related to population increase near wildfire-prone areas, so that post-wildfire enhanced runoff and erosion could result in catastrophic damage by destructive floods and debris flows (Neary and Gottfried, 2002; Pausas et al., 2008, Moody et al., 2013).

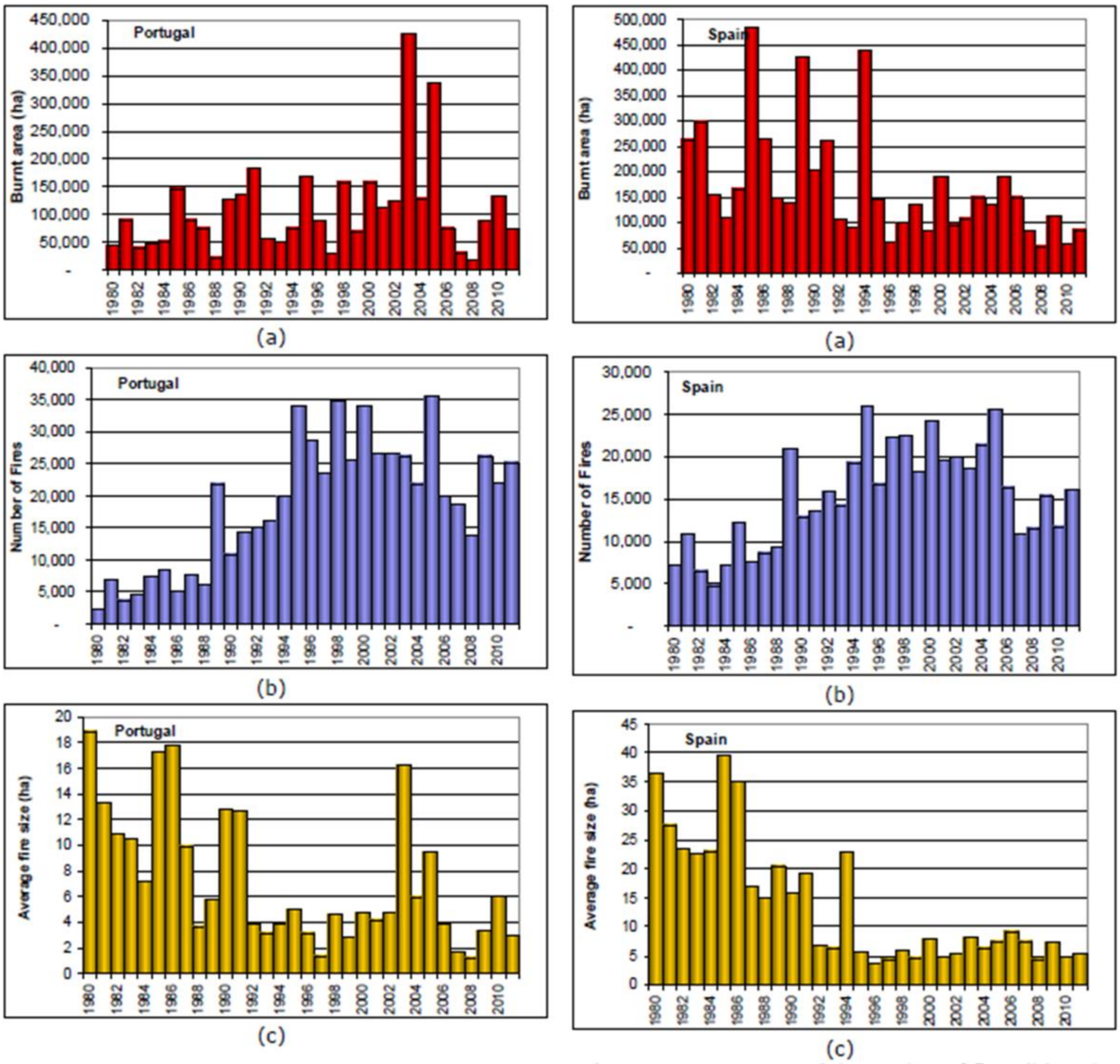


Figure 3 - Burned area (a), number of fires (b) and average fire size (c) in Portugal (left) and in Spain(right) between 1980 and 2010 (European Commission, 2011)

Human influence also has been viewed as one of the main drivers of the increase of the wildfire activity, through climate change (Scott et al., 2004; Harding et al., 2009), socio-economic changes and urban expansion (Pausas et al., 2008). Although the impacts of climate change on wildfire ignitions and behaviour have been researched for some time (Flannigan et al., 2000; Westerling et al., 2003; Bachelet et al., 2007; Littell et al., 2009; Moritz et al., 2010; Westerling et al., 2011), their implications for post-wildfire runoff and erosion response are only being explored recently (Pierce and Meyer, 2008; Moody and Martin, 2009; Goode et al., 2012). Human activities, such as rural depopulation in the Mediterranean basin have been also a factor for the increasing wildfire ignitions. Moreover, during the last half century an abandonment of traditional land management practices (Pardini et al., 2003), led to an increase of fuel load and fire prone forest cover (Shakesby, 2011).

The lack of land management due to land abandonment has been reported by Llovet (2005), as an additional cause of increased post fire soil losses. This author showed that recently abandoned fields (<15 years (Burnt abandoned), Figure 4) had a stronger vegetation recovery (70%) during first year after fire wildfire than long abandoned fields (>35 years (Burnt pine), Figure 4) (40%), leading to lower soil losses (Figure 4).

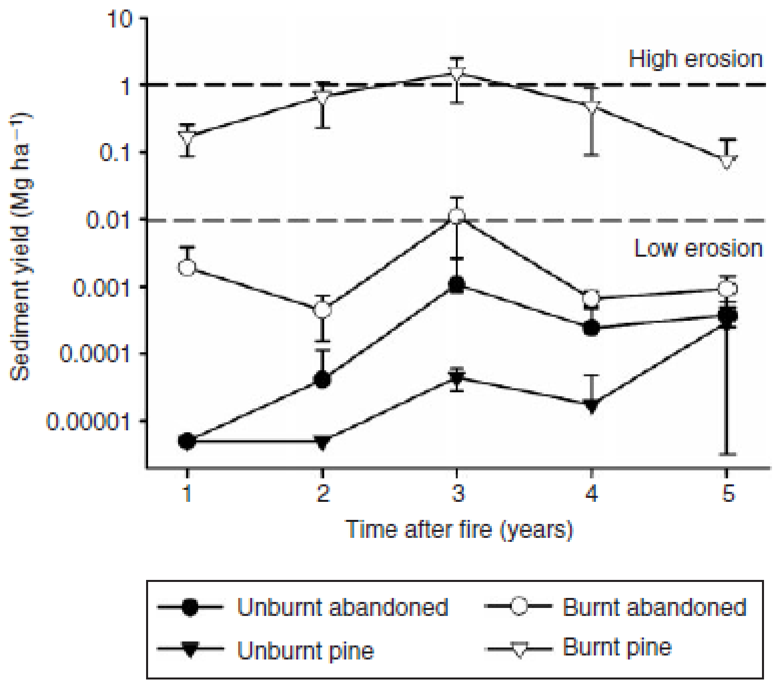


Figure 4 - Sediment yield on erosion plots located in burnt (open symbols) and unburnt (black symbols) of fields of Alicante (SE Spain, Llovet 2005, Pausas et al, 2008)

Other types of human influences in post-fire soil losses are the implementation of recent post-fire land management operations with heavy machinery (e.g. plowing; logging), that leads to forest soil disturbances. These post-fire forest management practices have been pointed out as an important cause for elevated soil erosion rates in some post-fire studies over the Mediterranean (Shakesby et al., 1994; Fernández et al., 2004; Shakesby, 2011; Martins et al., 2013). The impacts of these commonly used practices, together with an increase of fire frequency and recurrence at the same location can represents one of the biggest threats to Mediterranean soils.

According to Shakesby (2011), the Mediterranean basin can be considered as a 'global variant' in respect to the post-wildfire erosion, mostly due to the strong influence of the human activities, either through fire ignition, wildfire suppression and prescribed fire use or through land-use changes such as land abandonment and widespread introduction of highly flammable pine and eucalypt plantations (Moreira et al., 2009; Shakesby 2011). Thereby, should be considered that although post-fire soil losses in the Mediterranean are considered as low when compared to other regions, they are still very important. Sediment losses in the Mediterranean basin are generally low due to the frequent presence of shallow soils as a consequence of its history of intense disturbances. However these soil losses should be regarded as significant, not only because of the soil losses itself, but also because of the quality of the material that is being lost. Modest post-fire soil losses could be important for soil longevity in some areas, since organic matter and nutrient losses in solution or adsorbed onto eroded sediment particles (e.g. Kutiel and Naveh, 1987). Organic matter is mostly concentrated near the surface in Mediterranean soils, where it is particularly vulnerable to major losses when the protective vegetation and litter cover is depleted or removed by wildfire. Moreover, much of the post-fire nutrient content in the soils is in the form of ash (e.g. Ferreira et al., 2008).

I.II Burn severity and the distinctive runoff and erosion response

Burn severity can influence the magnitude and duration of post-fire effects. The term ‘burn severity’ was born out of the need to provide a description of how fire intensity affects ecosystems. This term has replaced ‘fire severity’ term, although the metric is very similar and is largely based on loss of organic matter in the soil and aboveground organic matter conversion to ash (Keeley, 2009). Is constrained to the loss of organic matter in or on the soil surface (NWCG, 2014), and also represents BAER assessments term ‘soil burn severity’ (Parsons, 2003). dependent of peak temperatures reached, duration of fire, and initial soil properties (soil type, soil moisture).

The existing burn severity classification methodologies can be divided by three groups:

- In situ measurements of one or more indexes providing a qualitative classification (e.g. Neary et al., 1999; Shakesby and Doerr, 2006; Keeley, 2009; Jain et al., 2012; Vega et al., 2013);

- Estimation of temperatures reached in soil and severity classification through calibration curves with Near Infrared (NIR) spectroscopy (Arcenegui et al., 2008; Guerrero et al., 2007; Maia et al, 2012);
- Satellite imagery to estimate the relative amount of damage attributed to an area of vegetation (Key and Benson, 2006; Hammill and Bradstock, 2006; Chafer, 2008).

The usage of various methodologies for the same term, or similar terms, can create comparability difficulties. For example, burn severity is dependent on fire intensity (measure of the time-averaged energy flux) but also has a similar metric as used in fire severity. Given the numerous definitions of fire and burn severity, it is important for all users to explain how they define and assess severity (Jain et al. 2008). Since expert judgement can lead to different fire severity interpretations, because observers focus on fire effects selected for a particular set of local objectives or outcomes Morgan et al., 2014).

Burn severity has been assessed for different purposes (Morgan et al., 2014), such as:

- mitigation of erosion potential , invasive species establishment and soil erosion risk (Fox et al., 2008; Clark and McKinley, 2011);
- post-fire vegetation recovery (Miller et al., 2003; Pausas et al., 2003; Beschta et al., 2004; Lentile et al., 2006);
- overall vegetation conditions (Bisson et al., 2008; Guay, 2011)

'Soil burn severity' has been used to predict the physical, chemical or biological effects (Jain et al., 2012), including water repellency (Lewis et al., 2006), erodibility (Pierson et al., 2001), nutrient availability (Belillas and Feller, 1998), and also have been used to describe the fire regime (Beukema and Kurz, 1998; Morgan et al., 2001; Barrett et al., 2006; Keane et al., 2006),

The knowledge of burn severity can be used by land managers to better predict the susceptibility of burnt areas to the post-fire occurrence of soil erosion and its implications in affecting the water quality of drinking water supply reservoirs (Lewis et al., 2006; Blake et al., 2006; Doerr et al., 2006; Shakesby et al., 2007; Chafer, 2008). It is well documented in the literature, that understanding the spatially heterogeneous distribution of fire severity and its impacts on soils is an important management tool for identifying areas that may become impacted by post-fire erosion (Benavides-Solorio and MacDonald, 2005; Mayor et al., 2007; Gimeno-Garcia et al., 2007, Chafer, 2008). Information of burn severity, allied to post-fire rehabilitation/mitigation treatments, and to a modelling tool (Robichaud et al., 2007), is the base for an emergency planning aiming at the prevention of flooding and increased soil losses, and at increasing vegetation recovery after fire (Robichaud et al., 2003).

Low intensity fires such as prescribed fires commonly present low burn severity because they are used only to reduce fuel accumulation and are applied during specific meteorological conditions, frequently associated to high soil moisture content to avoid elevated impacts. The changes to the soil produced by this type of fire are in most cases only transient. Severe fires such as summer wildfires, however, generally have several negative effects on soil (Certini, 2005). High soil burn severity is often associated with the decrease of ground cover, increased soil water repellency, decreased infiltration and increased erosion (DeBano et al., 1998; Robichaud, 2000; Benavides-Solorio and MacDonald, 2001; Pierson et al., 2001, Robichaud et al., 2007).

Despite the importance of the 'burn severity 'or 'soil burn severity' factor there is no unique use of the term, and the indicators by which burn severity is determined seem to vary among classification methodologies. Given the numerous definitions of fire and burn severity, it is important to clarify how severity is defined and assessed (Lentile et al., 2006; Jain et al., 2008, Morgan et al., 2014). Moreover, not every post-fire related study contemplates burn severity classification.

I.III Modelling post fire runoff and erosion rates

The effect of wildfires on runoff and erosion has created a strong demand for model-based tools to predict the post-fire hydrological and erosion response. Post-fire runoff-erosion modelling has been a research topic of various studies (Díaz-Fierros et al., 1987; Soto and Díaz-Fierros, 1998; Benavides-Solorio and MacDonald, 2005; Larsen and MacDonald, 2007; Robichaud et al., 2007; Moody et al., 2008; Fernández et al., 2010; Esteves et al., 2012; Vieira et al., 2014). Different models have been applied for this purpose, such as simple empirical models as the Universal Soil Loss Equation (USLE; Wischmeier and Smith, 1978) or the Revised Universal Soil Loss Equation (RUSLE; Renard et al., 1997) and the Revised Morgan–Morgan–Finney (MMF) model (Morgan, 2001). More complex and data-demanding models also have been used, such as process-based models, as the Water Erosion Prediction Project (WEPP; Nearing et al., 1989) and the Pan-European Soil Erosion Risk Assessment (PESERA; Kirkby et al., 2008).

The development of tools integrating several models with probabilistic approaches, such as Erosion Risk Management Tool (ERMiT; Robichaud et al., 2007) for the USA, elevated the potential for this research subject. Soil erosion models adapted for burnt areas provide promising alternative routes for assessing medium- to long-term impacts of this landscape-disturbing agent, providing a complement to small-scale and short-term field monitoring (Esteves et al., 2012).

As a very practical approach, the ERMiT tool has been developed for USA post-fire conditions (Robichaud et al., 2007). It allows to predict erosion risk in burnt forest areas, as well as to evaluate the effectiveness of applied treatments. ERMiT provides probabilistic estimates of single-storm post-fire hillslope erosion by incorporating variability in rainfall characteristics, soil burn severity, and soil characteristics into each prediction. ERMiT uses WEPP technology for runoff and erosion calculations. WEPP incorporates the processes of evapotranspiration, infiltration, runoff, soil detachment, sediment transport, and sediment deposition to predict runoff and erosion at the hillslope scale and simulates both inter-rill and rill erosion processes (Flanagan and Livingston, 1995). Through the ERMiT interface, stochastic weather files generated by CLImate GENerator (CLIGEN) (Nicks et al., 1995) are selected for use in WEPP.

The output of ERMiT enables forest managers to assess the impact of fire on site productivity and potential benefits of rehabilitation treatments (Larsen and MacDonald, 2007), and it can help them to formulate erosion mitigation treatment decisions based on the probability of the occurrence of high sediment yields (Robichaud et al., 2007). In Portugal and other Mediterranean regions, there is also a need for a similar tool to support post-fire management. However, the ERMiT tool and other models described above have been parameterized for specific circumstances that are not appropriate for Portuguese and Mediterranean conditions. Esteves et al. (2012), after applying PESERA to Portuguese post-fire conditions, highlighted that SWR, the influence of ash and the presence of a high stone content (frequently observed in burnt areas in Portugal and in the Mediterranean), are factors that need further consideration for locally-applied models. The authors also considered that these factors were the reasons why PESERA model overestimated post-fire soil erosion at both the plot and hillslope scale.

Many models also require a very detailed input dataset from the site, often unavailable in the Mediterranean, especially for post-fire conditions. Empirically-based models require less field data than other more complex models and might therefore be more feasible as a local management tool.

I.IV Objectives and thesis structure

The overall aim of this study was to further knowledge and understanding of the hydrological and erosion response in recently burnt forest areas, and of their modelling. The specific objectives were the following:

i. To determine the effect of burn severity in ecosystems and its implications regarding post fire hydrological and erosive responses worldwide, based on a critical review of existing studies using rainfall simulation experiments (RSE's) studies.

ii. To assess the effects of different pre-fire land management practices (unplowed, downslope plowed and contour plowed) and time-since-fire on the post fire hydrological and erosive response for the two of the most common land uses in Portugal (pine and eucalypt) (ICNF, 2013), and to provide new insights about the long term post fire response, by comparing runoff and erosion rates with known window of disturbance models (Figure 2).

iii. To determine the performance of two widely-known erosion models (RUSLE and Revised MMF) to predict soil erosion rates during the first year following two wildfires of distinctive burn severity, and to predict the efficiency of different post-fire rehabilitation treatments to reduce these erosion rates.

iv. To improve the Revised MMF model for two different land uses (pine and eucalypt) by incorporating seasonal variations, such as climate, soil moisture and vegetation recovery and by including soil water repellency. As well as, to assess model performance for the two published erosion mitigation studies carried out in north-central Portugal.

The organization of this thesis follows four main chapters. Chapters II and III, correspond to the publications in which these objectives were set: Afterwards, the thesis is followed by a chapter mentioning other contributions by the author (IV), and by general conclusions and future perspectives (IV).

II. Factors affecting post fire runoff and erosion

In this chapter, two extensive post fire datasets were analysed to identify key factors in post fire runoff and erosion: (i) rainfall simulation experiences dataset; (ii) Four years monitoring dataset of Colmeal study site.

The first database resulted in a review article in which results from different burn severities were compared by mean of Meta-analysis (*II.I Burn Severity*).

The second database explored the effects of land use, pre-fire land management and time-since-fire (*II.II Land cover, land management and time-since-fire*). A comparison was carried out between two land uses (pine and eucalypt), three pre-fire land management practices in eucalypt plantations (unplowed, downslope-plowed and contour-plowed), and the first four years after a wildfire. This comparison was done by mean of a two-way repeated ANOVA of annual runoff and erosion amounts at four study sites. This study also included auxiliary variables such as ground cover, soil water repellency, soil moisture, rainfall, rainfall intensity and rainfall erosivity.

III. Modelling post fire runoff and erosion

In this chapter, post fire runoff and erosion were predicted using two semi-empirical models.

The first approach (*III.I Performance of two erosion models after fire and rehabilitation treatments)* compared two different erosion model (RUSLE and revised MMF) estimations with the same dataset. This comparison was focused over soil erosion prediction efficiency of each model, between two study sites exposed to different burn severity wildfires. In this chapter a first approach to predict post fire rehabilitation treatments efficiency was also executed.

The second part is a follow-up from the previous publication. This chapter presents modelling results with the revised MMF model only, but focused in both runoff and erosion estimations (*III.II Improving runoff and erosion predictions in burnt forest using the revised Morgan-Morgan-Finney model*). In this case, some improvements were also implemented in the MMF model allowing the improvement of its prediction efficiency, over post fire and post fire with rehabilitation treatments cases. Model adjustments to the post fire case were based with the long term monitoring observations of Colmeal study site. Continuous observations of seasonal patterns of rainfall and soil water repellency inspired such model improvements.

Model validation was also performed at two independent sites, subjected to a wildfire in 1991 and 1991.

The four above-mentioned publications can also be organized in terms of time-since-fire and space scale, as shown in Figure 5.

Chapter II.I has a dataset of several rainfall simulation experiments, representing processes at micro-plot scale. At this scale erosion is controlled largely by the stability of the soil aggregates, and its breakdown is largely a result of raindrop impact, the frequency and erosivity of individual rainstorms (Morgan, 2005). And although these experiments were performed between the period, immediately after fire and 7 years after fire, each experiment represents a point-in-time measurement. The following chapter (II.II) still represent processes at micro-plot scale; however, the dataset contains continuous measurements in time during four years. Both modelling applications in chapter III were done at plot scale during 1 year. At plot scale, erosion is controlled by the processes that generate surface runoff, and usually include the infiltration characteristics of the soil and changes in the surface micro topography related to surface roughness (Morgan, 2005).

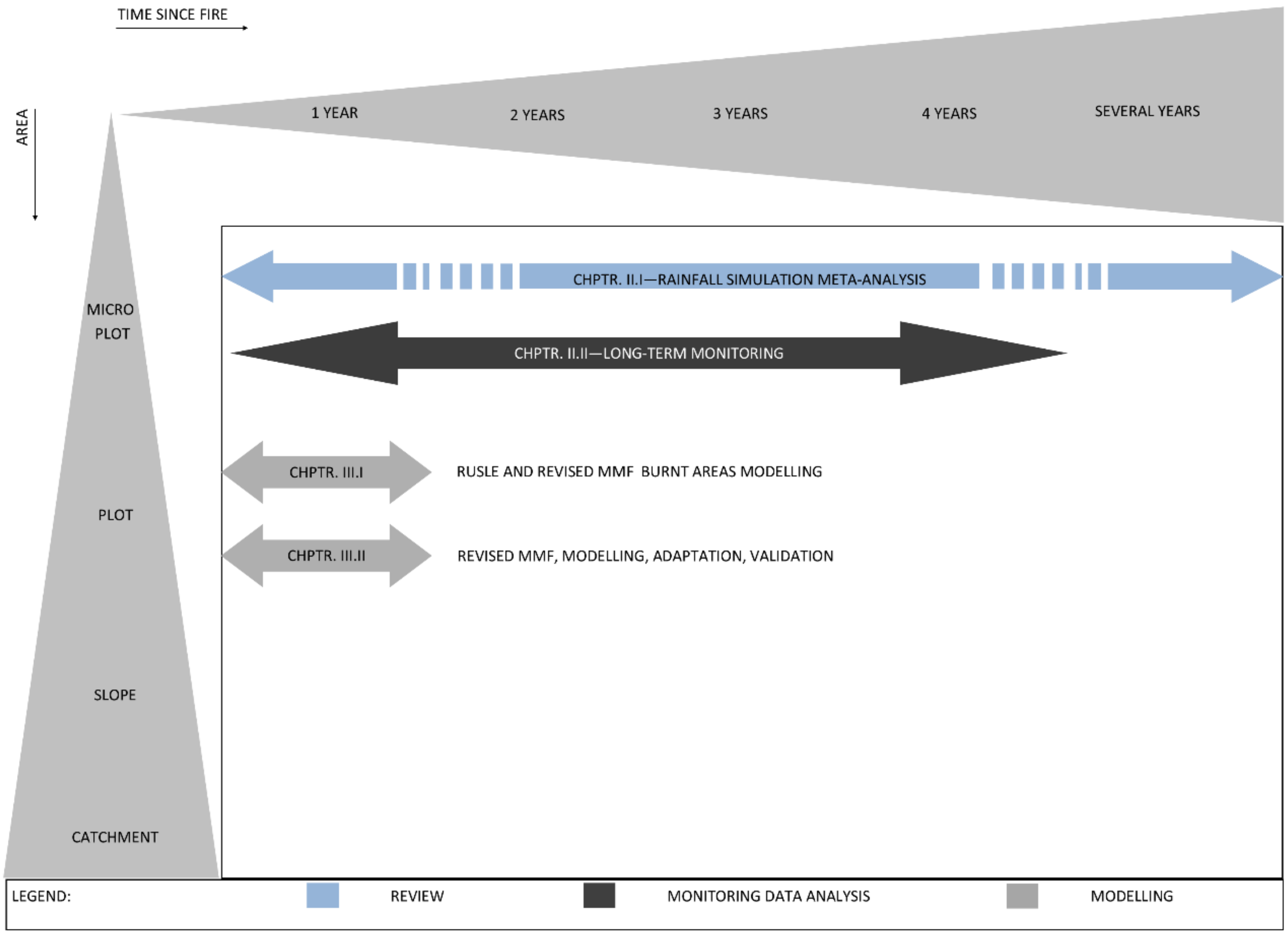
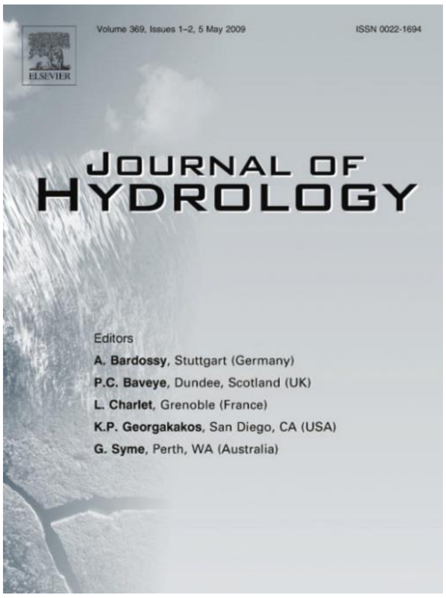


Figure 5 - Temporal and spatial scale for each publication/chapter of the thesis.

II. Factors affecting post fire runoff and erosion

II.I Burn severity



Does soil burn severity affect the post-fire runoff and interrill erosion response? A review based on meta-analysis of field rainfall simulation data.

Vieira, D.C.S, Fernández, C., Vega, J.A., Keizer J.J. (2015).
Journal of Hydrology 523, 452-464.

Soil burn severity has been widely used to describe the impacts of fire on soils and is increasingly being recognized as a decisive factor controlling post-fire erosion rates. However, there is no unique definition of the term and the relationship between soil burn severity and post-fire hydrological and erosion response has not yet been fully established.

The objective of this work was to review the existing literature on the role of soil burn severity on post-fire runoff and erosion ratios. To this end, a meta-analysis was carried out of the runoff and inter-rill erosion data from field rainfall simulation experiments (RSE's) that compared burnt and unburnt conditions. In this study, 109 individual observations were analysed that covered a wide geographical range, various types of land cover (forest, shrubland, and grassland) and two types of fire types (wildfire and prescribed fire). The effect size of the post-fire runoff and erosion response was determined, for four key factors: i) soil burn severity; ii) time-since-fire; iii) rainfall intensity; and iv) bare soil cover.

Statistical meta-analysis showed that fire occurrence had a significant effect on the hydrological and erosive response. However, this effect was only significantly higher with increasing soil burn severity for inter-rill erosion, and not for runoff. This study furthermore highlighted the incoherencies between existing burn severity classifications, and proposed an unambiguous classification.

Introduction

Wildfire has been a natural disturbance factor in most forest ecosystems since late Devonian times (Schmidt and Noack, 2000). However, mankind's key role in present-day fire regimes, especially through changes in land cover/use, has turned wildfires into an environmental problem in various countries over the last five decades (Cerdà and Mataix-Solera, 2009). Amongst other environmental impacts, wildfire is widely regarded as the principal agent of soil erosion and land degradation in woodlands and shrublands (DeBano et al., 2005; Shakesby and Doerr, 2006; Shakesby, 2011). Fire-enhanced runoff and erosion are commonly attributed to the (partial or total) removal of the protective soil cover by vegetation and litter, in combination with heating-induced changes in soil properties such as aggregate stability and water repellency (Neary et al., 1999; Mataix-Solera and Guerrero, 2007; Úbeda and Outeiro, 2009; Varela et al., 2010). These changes can markedly enhance runoff and associated transport processes during the so-called window of disturbance, both at the hillslope and catchment scale (Prosser and Williams, 1998; Shakesby and Doerr, 2006; Shakesby, 2011; Prats et al., 2013).

Soil burn severity is a commonly-used term to describe the heating-induced alterations to soil properties caused by fire. However, there is not a single and unique definition of this term (Keeley, 2009). Besides as an indicator of the direct impacts of fire on soil properties, soil burn severity is often used as an indicator of the fire's indirect impacts on the hydrological and erosion response of recently burnt areas. Most of the soil burn indices that have been proposed are based on the amount of surface litter layer consumed, the visually-observable changes of the mineral soil surface, the amount and colour of the deposited ashes, and/or the amount of charcoal present after the combustion of the aboveground biomass (Ryan and Noste, 1985; Neary et al., 2005; Shakesby and Doerr, 2006; Jain et al. 2012; Vega et al., 2013a). An indicator that is more closely linked to the soil heating regime itself is that of the maximum soil temperatures reached (MTR) estimated from Near Infrared (NIR) spectroscopy of laboratory-heated and field soil samples (Arcenegui et al., 2008; Guerrero et al., 2007; Maia et al., 2012). The use of distinct burn-severity classifications will hamper a direct comparison of the results obtained by different studies.

Wildfires often produce mosaics of areas with different soil burn severities (Robichaud et al., 2000; Maia et al., 2012; Vega et al., 2013a). These spatial patterns in burn severity are often important in identifying areas with a high risk of post-fire erosion (Benavides-Solorio and MacDonald, 2005; Mayor et al., 2007; Gimeno-Garcia et al., 2007, Chafer, 2008). In fact, mapping of soil burn severity has become an integral part of operational procedures in the USA to assess the risks of soil erosion and off-site impacts on downstream “values-at-risk” and, thereby, to prioritize emergency measures for reducing these risks (e.g. USDA, 1995; Parsons et al 2010; Vega et al 2013b). In the USA, a high soil burn severity is often associated with a decrease in ground cover, an increase in soil water repellency, and a decrease in infiltration (DeBano et al., 1998; Robichaud, 2000; Benavides-Solorio and MacDonald, 2001; Pierson et al 2001; Robichaud et al., 2007). Even so, in other parts of the world the relationship between soil burn severity and post-fire erosion has not been extensively studied and remains to be fully quantified (Shakesby, 2011). The reviews on wildfire effects in the Mediterranean Basin by Pausas et al. (2008) and Shakesby (2011) did not include a comprehensive analysis of the role of burn severity. To gain further insight in this relationship between burn severity and post-fire erosion, the present study aimed to compile the existing rainfall simulation data from field studies across the world and to analyse them quantitatively by means of meta-analysis.

Field rainfall simulations experiments (RSE’s) have been widely used to study the effects of fire on runoff and sediment losses (e.g. Emmerich and Cox, 1992; Robichaud, 2000; De Luis et al., 2003; Ferreira et al., 2005; Fernández et al., 2008; Groen and Woods, 2008). Nevertheless, the most comprehensive recent reviews on post-fire erosion (Shakesby and Doerr, 2006; Shakesby, 2011; Moody et al., 2013) have not included these RSE studies. The last review that did include them was that of Rulli et al. (2006). While the limitations of RSE’s as a research tool are well-established , RSE’s do permit a direct comparison of the hydrological and inter-rill erosion response of different study sites, distinct soil conservation measures and different times-since-fire (e.g. Cerdà, 1998; Nunes et al., 2009; Johansen et al., 2001a; Malvar et al., 2011; Fernández et al., 2012).

The specific objectives of the present meta-analysis were to quantify and critically review the association of differences in runoff and erosion between burnt and unburnt RSE plots with four potential key explanatory factors: (i) soil burn severity, irrespective of the underlying severity indicator ; (ii) time-since-fire; (iii) simulated rainfall intensity; (iv) bare soil cover. This included an assessment of the robustness of these associations by means of resampling (jack-knife) as well as an analysis of the main limitations of the present dataset and of possible manner to overcome these limitations in future datasets.

Materials and Methods

Origin of data

Science Direct and Google Scholar were searched exhaustively for publications in peer-reviewed journals that addressed the effects of fire on runoff and erosion by comparing the results of field rainfall simulation experiments (RSE's) carried out in recently burned vs. long-unburned areas that were otherwise comparable. From these publications, the present study selected those that provided the following elements:

- the applied rainfall intensity and duration;
- runoff and erosion figures that allowed computing runoff coefficients (%) and specific erosion rates ($Mg.ha^{-1}.mm_{rain}^{-1}$);
- the timing of the RSE's relative to the occurrence of the fire;
- the type of fire (wildfire or prescribed fire) and its soil burn severity, either simply stated as an expert opinion or sustained by observations on one or more indicators of soil burn severity.

In total, 20 publications were included in the present meta-analysis (Table 1). Some publications, however, were only included after the authors had provided additional information or clarified specific doubts (such requests for further information were restricted to the studies published after 2000). The study by Cerdà and Doerr (2005) was an exceptional case in that it was included in spite of not involving a direct comparison between burnt and unburnt plots. This was done because the study included RSE's that were carried out 11 years after a wildfire, i.e. under long-unburnt conditions closely approximating control conditions.

Data compilation, revision and gap-filling

A varied number of observations (1 to 26) were obtained among the 20 selected publications, providing a total of 109 observations (Table 1). Each observation is defined as a comparison of runoff and erosion ratios, obtained by concomitant RSE's on recently burnt vs. long-unburnt plots. The identification of one or more observations in a specific study reflected the number of independent factors included in the study's experimental set-up, and, thus, aimed at minimizing within-group variation in the observations. For example, the study by Kutiel et al. (1995) compared runoff and erosion on burnt vs. unburnt RSE plots on north-facing slopes as well as on south-facing slopes, so that two, exposition-specific observations were included in the meta-analysis.

Table 1 – References used for Meta-analysis, and main conditions of performed rainfall simulations.

Reference	Vegetation	Location	Burn severity			Time since fire	Simulator type	RSE Intensity (duration)	Nº obs.
			classes	method[1]	indicators[2]				
Benavides and MacDonald (2001)	Ponderosa and Lodgepole Pine	Colorado Front Range, USA	low, moderate, high	Wells et al. (1979)	organic layer; litter; duff; mineral soil	Immediately after fire	Meyer and Harmon (1979)	66-94 mm/h (60min)	6
Cerdà and Doerr (2005)	Trees, herbs, shrubs, dwarf shrubs.	Serra Grossa Range,Valencia, Spain	high	Bentley and Fenner (1958)	ash	1,2,3,6,8 and 11 years after fire	Cerdà et al. (1997)	55 mm/h (60min)	10
De Luis et al. (2003)	Shrubs	Onil, SE Spain	low, moderate, high	expert judgment	temperatures; ash; woody and litter debris	Immediately after fire	De Luis et al. (2003)	156 mm/h (105min)	9
Dobrowolski et al. (1992)	Longleaf pine	Lousiana, USA	high	estimated	temperatures; rel. humidity; bare soil; vegetation; litter; fire type	Immediately after fire, One year	Meyer and Harmon (1979)	126 mm/h (45 min)	26
Emmerich and Cox (1992)	Grass (introduced/ native)	SE Arizona, USA	low	expert judgment	biomass; litter; plant crowns	Immediately after fire	Swanson (1965)	55 and 110 mm/h (45 and 15min)	1
Emmerich and Cox (1994)	Grass (introduced/ native)	SE Arizona, USA	low	estimated	temperatures; rel. humidity; bare soil; vegetation; litter; fire type	Immediately after fire	Swanson (1965)	55 and 110 mm/h (45 and 15min)	8
Fernández et al. (2006)	Pinus pinaster + shrubs	Orense, N.W. Spain	moderate	request	organic cover; ash; soil structure	Immediately after fire	Wilcox et al. (1986)	120 mm/h (30min)	1
Fernández et al. (2008)	Shrubs	Pontevedra, NW Spain	low	request	organic cover; ash; soil structure	Immediately after fire	Wilcox et al. (1986)	67 mm/h (30 min)	1
Fernández et al. (2012)	Shrubs	Orense and Santander, N Spain	low	request	organic cover; ash; soil structure	Immediately after fire	Wilcox et al. (1986)	67 mm/h (30 min)	2
Hester et al. (1997)	Oak, juniper, bunchgrass and shortgrass	Texas, USA	high	expert judgment	na	Immediately after fire	Blackburn et al. (1974)	203 mm/h (50 min)	1
Johansen et al. (2001a)	Ponderosa Pine	NW Santa Fe, New Mexico, USA	high	BAER (Robichaud et al 2000)	na	Immediately after fire	Swanson (1965)	60 mm/h (60+30+30min)	4
Johansen et al. (2001b)	Desert grass	Carlsbad, New Mexico and Westminster, Colorado; USA	low	estimated	temperatures; rel. humidity; bare soil; vegetation; litter; fire type	Immediately after fire	Swanson (1965)	60 mm/h (60+30+30min)	2
Knight et al. (1983)	Shrubs	Texas, USA	low	estimated	temperatures; rel. humidity; bare soil; vegetation; litter; fire type	Immediately after fire	Blackburn et al. (1974)	203 mm/h (30 min)	1
Kutiel et al. (1995)	Pine, oak, shrub	NW Israel	low	Sampson (1944)	temperatures	Immediately, 2 weeks and 1 year after fire	Morin et al. (1967)	30mm/h (60min)	6

Reference	Vegetation	Location	Burn severity classes	Burn severity method(1)	Burn severity indicators(2)	Time since fire	Simulator type	RSE Intensity (duration)	Nº obs.
Pierson et al. (2001)	Sagebrush	Denio, Nevada, USA	high	expert judgment	ground cover; canopy	Immediately and 1 year after fire	Meyer and Harmon (1979)	85 mm/h (60min)	4
Pierson et al. (2002)	Sagebrush	Idaho, USA	low, moderate, high	BAER (Robichaud et al 2000)	litter; duff and woody debris; ash color; mineral soil	1 year	Meyer and Harmon (1979)	67 mm/h (10-30min)	8
Pierson et al. (2008)	Sagebrush	Denio, Nevada, USA	high	expert judgment	ground cover; canopy	Immediately, 1 and 2 years after fire	Meyer and Harmon (1979)	85 mm/h (60 min)	2
Robichaud (2000)	Ponderosa and Lodgepole Pine	Western Montana and central Idaho, USA	low, moderate, high	expert judgment	ground cover; duff thickness	Immediately after fire	USDA-Forest Service oscillating nozzle rainfall simulator.	94 mm/h (30 min)	6
Sheridan et al. (2007)	Eucalypts	NE Victoria, Australia	moderate, high	expert judgment	crown burnt/scorch levels	Immediately, 0.5, 1, 1.5, 2, 2.5 and 3 years after fire	Bubenzer and Meyer (1965)	100 mm/h (30min)	7
Zavala et al. (2009)	Shrub	Cádiz, SW Spain	low	expert judgment	ashes; charred litter; soil	Immediately after fire	Navas et al. (1990)	56.5 mm/h (30 min)	4

(1) Method cited by authors to justify burn severity classification;
(2) Burn severity indicators taken into account for burn severity classification given by the authors in the articles, when those parameters were not described filled with “na”;

Data on the simulated rainfall (amount, intensity and duration) and on the area and slope angle of the RSE plots were available for (almost) all 109 observations (Table 2, Figure 6, Figure 7). The height from which the simulated rainfall was falling was only specified for 30% of the observations. Data availability on soil properties was highly variable. While soil texture class was mentioned for 94% of the observations, the fractions of clay, silt and sand were referred for less than half (44%). Also potential key explanatory variables such as soil moisture content and bulk density were missing for a large part of the observations (45% and 62%, respectively). Data availability on soil cover reflected the importance that is commonly attributed to bare soil and vegetation cover, being referred for 77% and 70% of the observations, respectively. Other cover variables such as litter and stone/rock cover were available for less than a third of the observations.

Table 2 – List of rainfall simulation variables and auxiliary parameters. The availability of each variable relatively to the total number of runs (n=109) in percent (%-available) and the fraction of estimates within the entire data set (%-flagged) are given.

Variable	Description	Units	Min	Max	%-available	%-flagged
Rainfall	Total rainfall amount applied	mm	24	273	100	0
Rainfall Intensity	Rainfall intensity	$mm.h^{-1}$	34	203	100	0
Rainfall duration	Rainfall duration	min	30	120	100	0
Plot Area	Area of the RSE plot	m^2	0.13	32	100	0
Slope angle	Slope angle of the RSE plot	%	4	51	99	0
Simulator height	Height from which rainfall	m	2	3.5	30	0
Runoff	Runoff coefficient over the entire RSE	%	2	68	100	0
Erosion	Specific erosion rate per mm of rainfall over the entire RSE	$Mg.ha^{-1}\ mm_{rain}^{-1}$	<0.01	0.11	100	0
Antecedent soil moisture	Soil moisture content immediately before rainfall simulation	%	0	35	55	0
Bulk density	Air-dry bulk density	$kg.m^{-3}$	0.51	1.39	38	0
Texture class	Texture classification	-	-	-	94	0
Clay	Clay fraction (<2 µm)	%	5	91	44	0
Silt	Silt fraction (2–63 µm)	%	4	47	44	0
Sand	Sand fraction (63–2000 µm)	%	5	83	44	0
Cover	Total protective surface cover	%	0	100	28	0
Bare soil	Cover of bare soil	%	1	100	77	0
Vegetation	Cover by vegetation	%	2	100	70	0
Stone	Cover by stones/rock	%	0	2	7	0
Ash	Cover by ash	%	26	64	16	0
Litter	Cover by litter	%	37	27	22	0
Time since Fire	Time occurred between fire and rainfall simulation	months	0	84	100	0
Burn Severity	Classification describing the heating-induced alterations to soil	-	-	-	68	32*

(*) Corresponds to 4 studies (Dobrowolski et al. (1992), Emmerich-Cox (1994), Johansen et al. (2001b) and Knight et al. (1983)

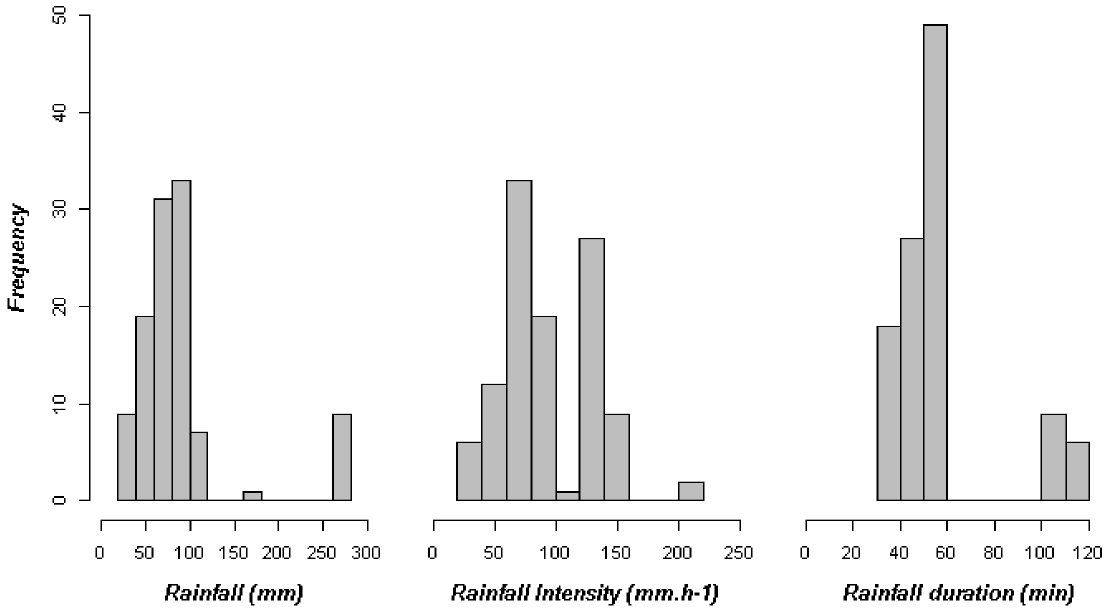


Figure 6 – Histograms of rainfall properties for 109 runs; total rainfall amount in the left; mean rainfall intensity in the middle; and rainfall duration on the right.

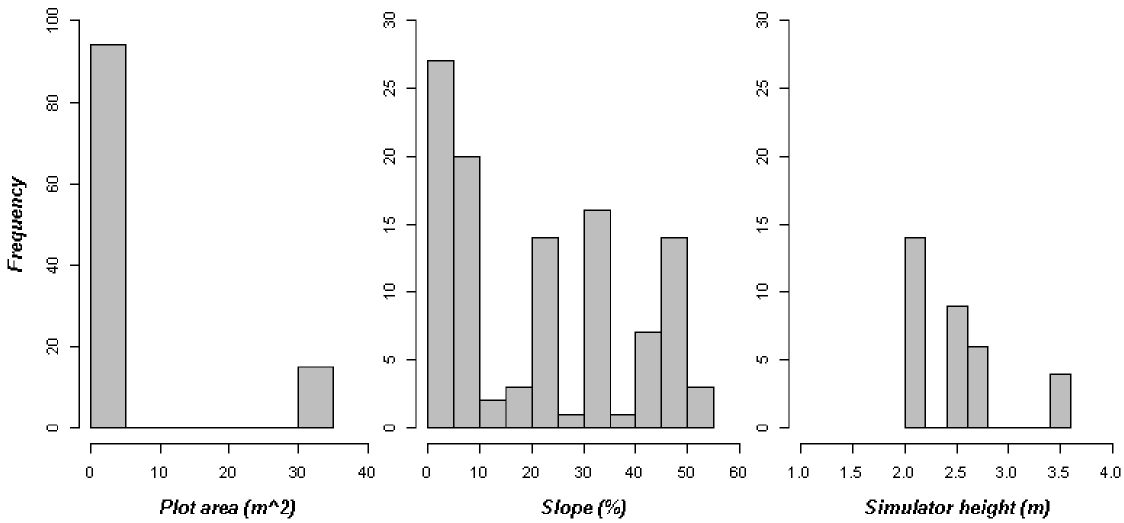


Figure 7 – Histograms of rainfall simulation experiments characteristics; plot area in the left (n=109); plot slope in the middle (n=108); and simulator height on the right (n=30).

While soil burn severity was crucial to the aims of this meta-analysis, it was not explicitly referred in 7 out of the 20 publications (Figure 8). In three of these cases (Fernández et al., 2006, 2008, 2012), the authors of the publications provided this information, upon a specific request. In the remaining four cases (Dobrowolski et al., 1992; Emmerich-Cox, 1994; Johansen et al., 2001b; Knight et al., 1983), the present authors felt confident to estimate soil burn severity following the classification by Vega et al. (2013a), based on the description of the fire and of post-fire conditions in the article.

All four studies concerned prescribed fire and, more specifically, headfires under temperature and relative humidity conditions guaranteeing the - intended - low soil burn severity. This low severity was also confirmed by studies' data on bare soil and remaining vegetation-litter cover immediately after the fire. and on the degree of litter consumption Furthermore, the bulk of the other publications on prescribed fire included in the dataset reported low severity burning, with the two exceptions (De Luis et al., 2003; Robichaud, 2000) having used prescribed fire with the explicit aim to study the effects of different burn severities.

From the 13 publications that did explicitly refer soil burn severity, more than half involved expert judgement as opposed to being based on published data on one more of severity indicators (Table 1, Figure 8). The remaining five publications specifically referred the method that was used to classify soil burnt severity, i.e. the methods of Sampson (1944), Bentley and Fenner (1958), Wells et al., (1979), and Robichaud et al., (2000). These publications also provided information on the underlying burn severity indicator(s). The most referred severity indicators were, either the quantification of the remaining protective soil cover (litter, vegetation, duff, organic layer, canopy) or the absence of protection (bare soil). Followed by, vegetation consumption indicators (duff thickness, debris, scorch levels) and fire temperatures (prescribed fire studies only) and ashes (presence and colour).

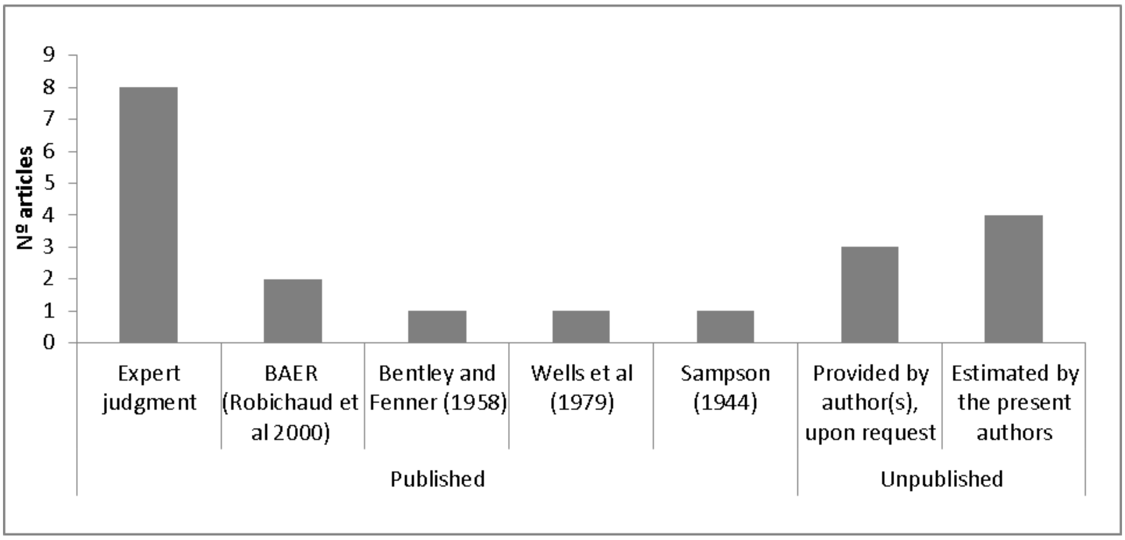


Figure 8 – Number of studies per burn severity classification sources in the meta-analysis database.

Statistical data analysis

The effect of fire and, in particular, soil burn severity on the runoff and erosion response in the present dataset was analysed by means of meta-analysis (Cooper et al., 2009). The same was done regarding the effects of time-since-fire and applied rainfall intensity. To this end, all three variables were divided into 3 or 4 classes on an interval scale. Following the terminology used throughout the 20 publications included in this meta-analysis, soil burn severity was divided into low, moderate and high. Time-since-fire and rainfall intensity, however, were divided into arbitrary classes. These classes were respectively: (i) < 0.5; 0.5 - 1.5; 1.5 – 3; > 3 years after the fire; and (ii) 30 - 60, 60 - 100, 100 - 150, and > 150 mm h^{-1}.

For each of the 109 observations, the average runoff coefficient (%) and the average specific erosion rate (Mg ha^{-1} mm_{rain}^{-1}) of the multiple RSE's that were carried out on recently burnt plots or on long-unburnt plots were compiled or computed. Also the corresponding standard deviations were compiled or calculated, based on the measures of variation provided in the publication or by the authors upon specific request.

The meta-analyses tested the effects of soil burn severity, time-since-fire and rainfall intensity by means of fixed effects models. The logarithmic response ratio was used as test metric, since it is widely considered to be the most appropriate metric for meta-analysis of ecological data (e.g. Wan et al., 2001; Kopper et al., 2009; Kalies et al., 2010 Abalos et al., 2014). This ratio expresses the proportional difference in a response variable between a "treatment" and a reference:

$$\ln R = \ln\left(\frac{\overline{x^E}}{\overline{x^C}}\right) \ln R = \ln\left(\frac{\overline{x^E}}{\overline{x^C}}\right)$$

, where $\overline{x^E}$ stands for the average of the response variable for treatment; and $\overline{x^C}$ for the average of the response variable for the reference. Mean effect size was calculated with bias-corrected 95% intervals.

In the present meta-analysis, the treatment data corresponded to the results from the recently burnt RSE plots, either from a specific soil burn severity class or from all severity classes together (overall fire effect), while the reference data corresponded to the results from the long-unburnt RSE plots. The response variables analysed here were runoff coefficient (%) and specific erosion rate (Mg ha^{-1} mm_{rain}^{-1}). The standard deviations in these response variables were used as weighting factors of the individual observations, also called as moderator variables. This allowed estimating the weighted least squares relationship between the moderator variables and the true effects (Viechtbauer, 2010).

The so-called effect size was determined for the three classes of soil burn severity separately as well as together (in the latter case being referred to as overall effect). The same applied for time-since-fire and applied rainfall intensity. Mean effect sizes were considered to be significantly different from zero if the 95% confidence interval did not overlap zero, and significantly different from one another if their 95% confidence intervals were not overlapped. A positive effect size then meant that, for example, a high soil burn severity significantly enhanced the average runoff coefficient or the average specific erosion rate compared to long-unburnt conditions, with the mean effect size expressing the extent of this fire effect on a logarithmic scale (see e.g. Kalies et al., 2010).

A test for residual heterogeneity was carried out as integral part of the applied meta-analysis function considering that the applied model includes moderator variables. The applied test is the Cochran's QE-test (Cochran, 1954) for residual heterogeneity. This test assesses whether the variability in the observed effect sizes or outcomes that is not accounted for by the model's moderator variables, is larger than would be expected based on sampling variability alone (Viechtbauer, 2014).

The robustness of the meta-analysis results of the effects of fire severity, rainfall intensity and time-since-fire was assessed by means of a Jackknife procedure (Sokal and Rohlf, 1981). This was done in view of the heterogeneity of the data set as described in the next section and, in particular, to asses if individual observations unduly influenced effect size. The procedure involved repeating the meta-analysis while ignoring one of the observations at a time. All analyses were carried out using the R statistical package (R Development Core Team, 2012).

Description of the dataset

The 20 studies included in the present analysis involved a broad geographical range, covering four continents, but also revealed a strong geographical bias, since the bulk of the studies were carried out in the USA (60 %) and Spain (30 %) (Table 1). The 109 observations revealed a similar predominance of the USA (63 %) and Spain (25 %). In terms of land-cover type, the 109 observations concerned noticeably more forest stands (62 %) than shrublands (28 %) or grasslands (10 %).

A larger fraction of the 109 observations concerned the effects of prescribed fires as opposed to the effects of wildfires (58 vs. 42 %; Figure 9). The soil burn severity of these fires was mostly low (48 % of the observations), while it was moderate and high in roughly similar proportions (30 and 22 % of the observations, respectively). The bulk of the RSE's included in this analysis (58 % of the observations) were carried out during the first six months after the fire (Figure 9). The remaining observations decreased markedly with the three subsequent time-since-fire classes. Rainfall intensities within first two classes (30-60 and 60-100 mm h^{-1}) had similar number of observations (27%). The class with most of the observations (37%) was 100-150 mm h^{-1}, while the least represented was <150 mm h^{-1} with only 10% of the cases.

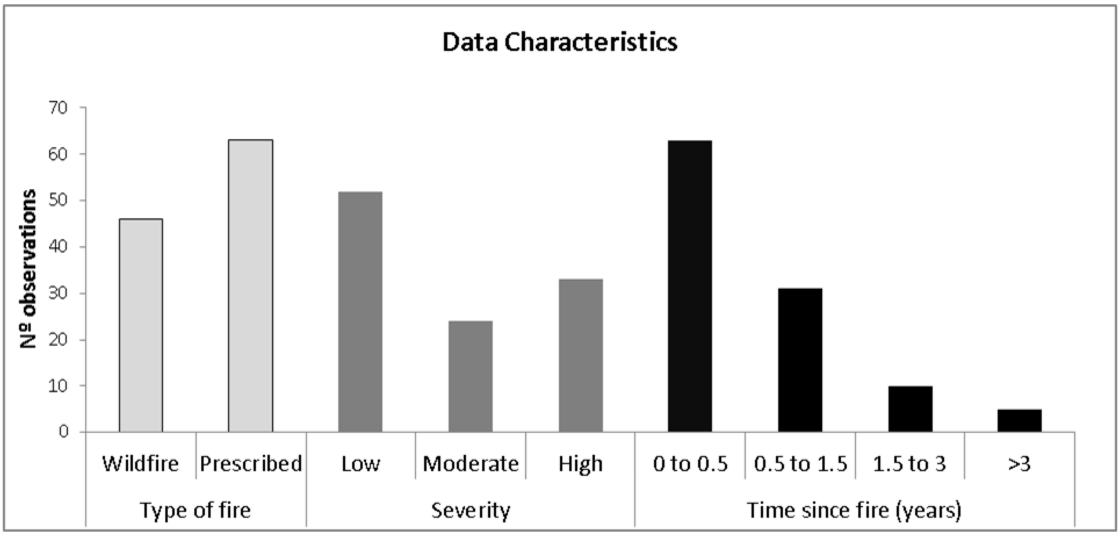


Figure 9 – Number of observations from the meta-analysis database classified by type of fire, soil burn severity level and time since fire.

Results

The effects of fire occurrence and the role of soil burn severity

The occurrence of fire by itself had a significant effect on the hydrological response, increasing the runoff coefficient compared to unburnt conditions (Figure 10). The same was true for each of the three soil burn severity classes (Figure 10). All four corresponding QE-test values were highly significant ($p < 0.001$). The runoff effect size, however, did not differ significantly between the three soil burn severity classes. Nonetheless, there was some suggestion that high burn severity had a lesser impact on overland flow generation than low as well as moderate burn severity, as there was very little overlap between the effect size of high severity and the effects sizes of the other severity classes.

Fire per se also enhanced the erosion response in a significant manner but to a greater extent than it increased the runoff response, as indicated by the markedly larger overall effect size (1.73 vs. 0.84; Figure 10). There was again significant between-group heterogeneity ($P < 0.001$) for severity classes. Unlike the runoff response ratio, however, the erosion response ratio differed significantly between all three severity classes. Furthermore, it increased with increasing burn severity and, most conspicuously, from the low to the moderate soil burn severity.

The effects of time-since-fire

The average effect size of the runoff response ratio decreased monotonically with the four classes of increasing time-since-fire (Figure 11). Even so, fire only had an unequivocal significant effect on overland flow generation in the case of the two initial time-since-fire classes, and this effect was significantly stronger immediately after fire than between 0.5 and 1.5 years after fire. The runoff-enhancing effect of fire also appeared to be significant between 1.5 and 3 year after fire but this hypothesis was rejected as the QE value was not statistically significant ($p = 0.07$). More than 3 years after fire, the effect size was not significant and also was not correspond to a significant QE value ($p = 0.21$).

The effect size of the erosion response appeared to be significant for all four time-since-fire classes (Figure 11). In the case of the final class, however, this erosion-enhancing effect was questionable because the corresponding QE-test of heterogeneity yielded a p-value that was clearly non-significant (0.24). The three remaining classes did not suggest an obvious pattern in erosion effect size with time-since-fire, unlike was the case for the runoff response ratio. Even so, the erosion response ratio did agree with the runoff response ratio in that effect size was significantly higher immediately after fire than between 0.5 and 1.5 years after fire.

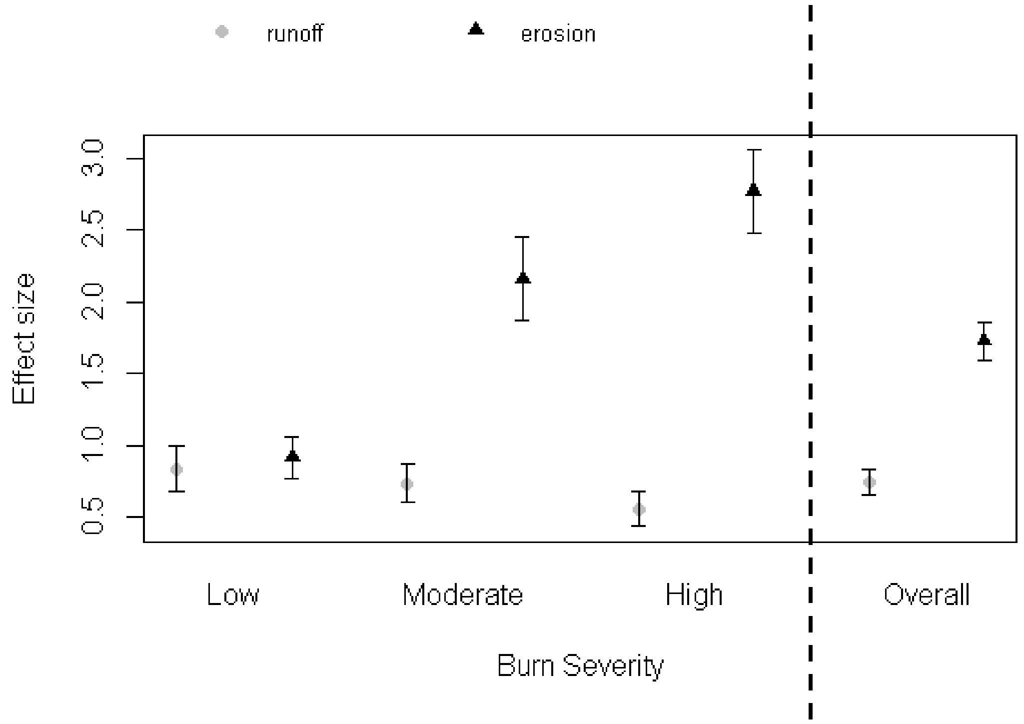


Figure 10 – Effect size for runoff coefficients and specific erosion rates, for fire occurrence (“overall”) (n=109) and for the three classes of soil burn severity: Low (n=52); Moderate (n=24); and High (n=33). Confidence intervals (95%) that do not cross the zero y-axes are statistically significant.

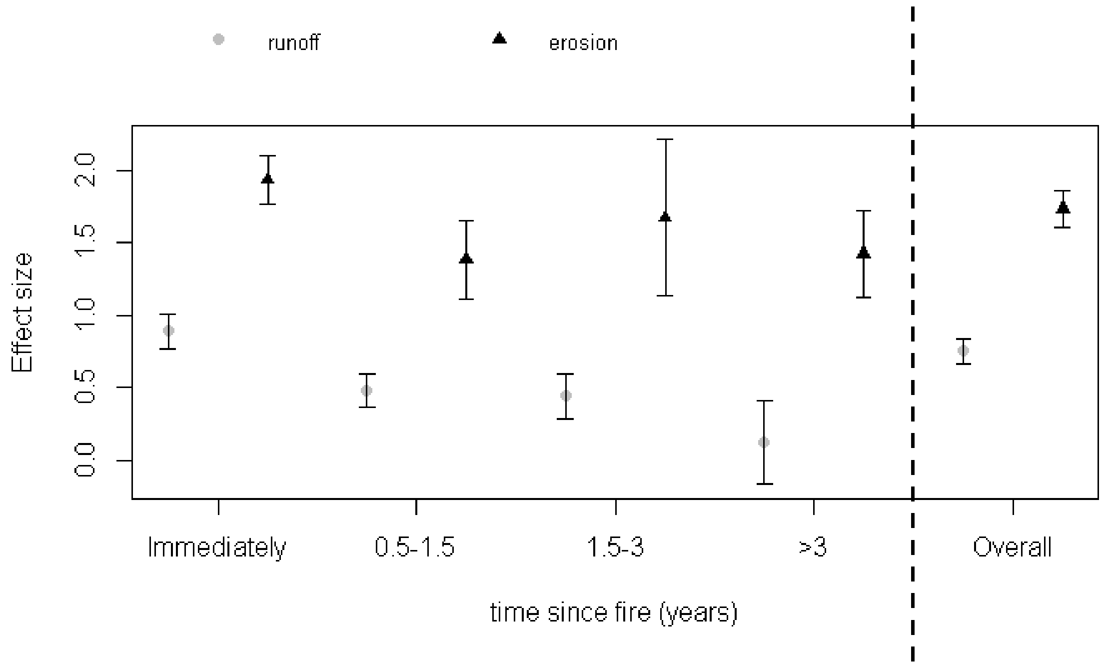


Figure 11 – Effect size for runoff coefficients and specific erosion rates, for fire occurrence (“overall”) (n=109) and for the four classes of time since fire: Immediately (n=63); 0.5-1.5 years (n=30); 1.5-3 years (n=9); and >3 years (n=7). Confidence intervals (95%) that do not cross the zero y-axes are statistically significant.

The effects of rainfall intensity

All four classes of applied rainfall intensity revealed a significant effect size for overland flow generation (Figure 12). Also all four corresponding QE values were highly significant (p's < 0.01). There was some suggestion that the average effect size increased with increasing rainfall intensity but this tendency was limited to the rainfall intensity classes that exceeded 60 mm h^{-1}. The effect size of the lowest rainfall intensity class (30-60 mm h^{-1}) was, on average, intermediate between those of the two highest classes (100-1500 and >150 mm h^{-1}) and also strongly overlapped with them.

The four rainfall intensity classes equally revealed a significant fire effect on specific erosion rates (Figure 12) and highly significant QE-test values (p's < 0.01). However, there was no straightforward relationship of effect size with rainfall intensity, except that the effect size of the highest rainfall intensity class clearly contrasted with those of the other three classes.

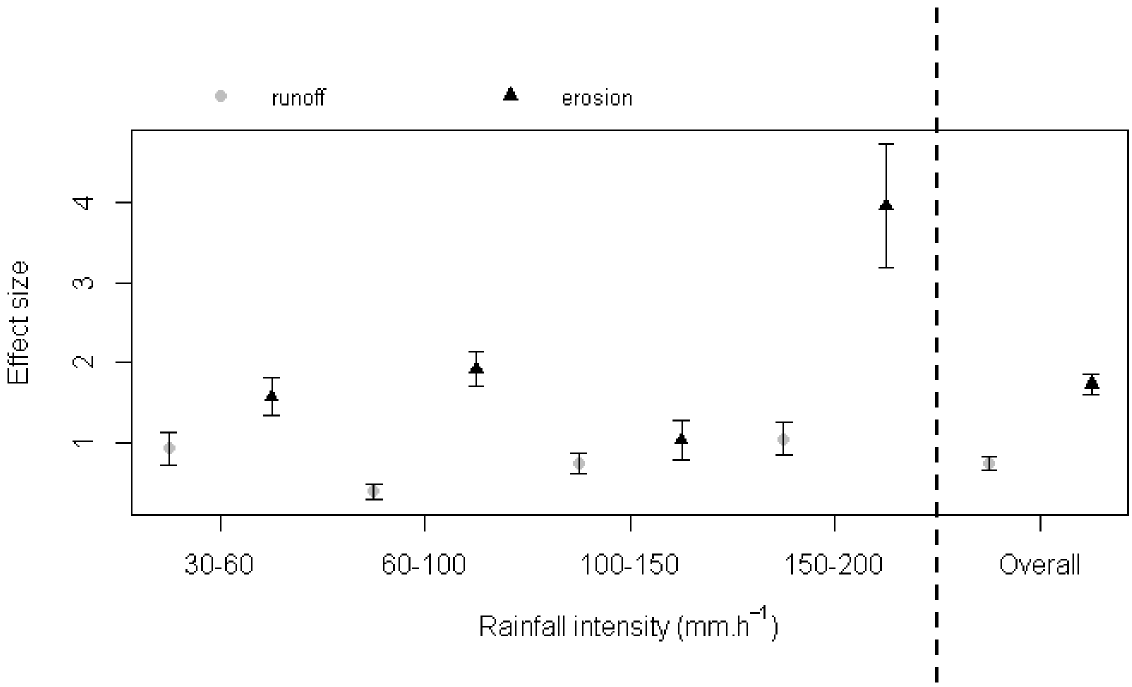


Figure 12 – Effect size for runoff coefficients and specific erosion rates, for fire occurrence ("overall") (n=109) and for the four classes of rainfall intensity: 30-60 mm h^{-1} (n=29); 60-100 mm h^{-1} (n=29); 100-150 mm h^{-1} (n=40); and 150-200 mm h^{-1} (n=11). Confidence intervals (95%) that do not cross the zero y-axes are statistically significant.

Robustness of the meta-analysis results

Generally, some differences between control and burned plots (besides the burning) will always exist, especially in the wildfire cases with plots outside the burning area, and less in the case of many prescribed fires, because RSE's were made in the exact same plot (control plots = pre-burn plots) leading to a smaller error. Nevertheless, the coefficients of determination (R^2) between the effect size for runoff and erosion and all the listed variables (Table 2) were calculated, and its result remained well below 0.3. Also, the meta-analysis results obtained through a Jackknife procedure revealed relatively minor variations in average effect size (Figure 13). Furthermore, the Jackknife results did not alter the statistical inferences on the role of fire occurrence or the different classes of soil burn severity, time-since-fire and rainfall intensity. Thus, the current data set appeared to be sufficiently robust against unduly impacts of individual observations, in spite the number of 109 observations was somewhat limited and the 109 observations included considerable variation in potential key factors such as vegetation type or slope angle.

Runoff ratios effect size, were inverted for low and moderate burn severity classes in two occasions (2%), while the high burn severity group never intercepted another classification (Figure 13). However, standard error was always overlapped for all the classes and due to that they never differ significantly. In the case of the erosion, effect size variation for each burnt severity class never intercepted another class, and the standard error between low and moderate burn severity never overlapped. Also, high and moderate severity classes were significantly different for 74% of the runs (Figure 13).

Runoff effect size for RSE's made immediately after fire were significantly higher, with no overlapping or interception from de other groups. Between 0.5-1.5 and 1.5-3 years few variation occurred (standard deviation, s.d. = 0.01), but no significant differences could be observed. In the case of the >3 years class a higher variation of results was observed (s.d.=0.06) due to 2 observations from 2 different studies. For erosion results however, this classification seem to show the same results as the meta-analysis with the entire dataset.

In the case of the rainfall intensity groups, few variations were observed within each class regardless of the removed observation in the case of runoff (s.d. varied between 0.01-0.02) and in the erosion this variation was slightly higher (s.d. between 0.02 and 0.06), however none of the effect sizes for runoff and erosion switched positions from the initial result (Figure 13).

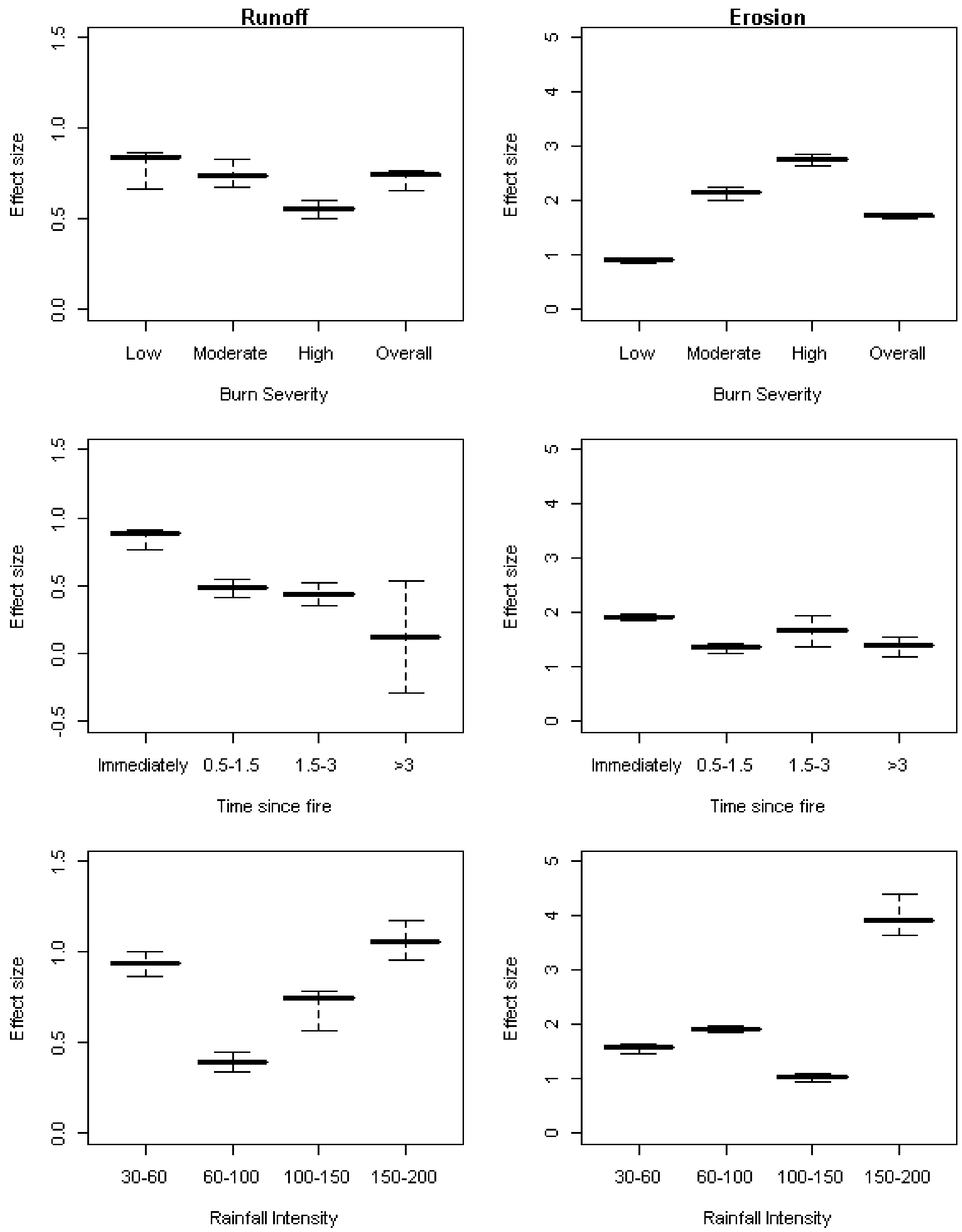


Figure 13 – Average effect size of runoff coefficients (left) and specific erosion rates (right) for fire occurrence and the various classes of soil burn severity (up), time-since-fire (middle) and rainfall intensity (bottom) for the 108 realizations analysed through a Jackknife procedure. Error bars indicate max and min.

Discussion

This meta-analysis study is the first quantitative synthesis of the literature in scientific journals on the role of soil burnt severity in the runoff and erosion response to simulated rainfall under field conditions. The number of encountered publications is surprisingly low compared to the number of studies included in a similar meta-analysis for arable soils (Fiener et al., 2011). Shakesby (2011) noted the same in relation to erosion studies using runoff plots.

The role of fire and soil burn severity

The present meta-analysis provided statistical evidence for the long-standing notion that the occurrence of fire tends to lead to an increase in overland flow generation and associated sediment losses (e.g. DeBano et al., 2005; Shakesby and Doerr 2006; Pausas et al., 2008; Shakesby, 2011). The present results further indicated that this fire effect is clearly less pronounced for the hydrological than erosion response. This enhanced impact on erosion rates as compared to runoff rates could be explained by heating-induced changes in soil properties that determine soil erodibility, such as aggregate stability and organic matter content (Varela et al., 2010; Mataix-Solera et al., 2011). However, also the consumption of the ground-covering vegetation and/or the litter layer could play a role by reducing the resistance to flow and, hence, increasing the erosive power of the runoff (Shakesby and Doerr, 2006).

This meta-analysis also provided statistical evidence for the increasingly widespread opinion that the geomorphological implications of fires depend strongly on their severity (Benavides-Solorio and MacDonald, 2005; Vega et al., 2005; Shakesby and Doerr 2006; Pausas et al., 2008; Fernández et al 2010; Shakesby 2011). The role of soil burn severity was found here to be statistically significant in the case of soil erosion but not in the case of overland flow. This was in line with the above-mentioned finding that fire per se had a more pronounced effect on soil erosion than on runoff. Furthermore, the fire-induced enhancement of the erosion response differed in a plausible manner among the three soil burn severity classes, as increasing fire severity can be expected to aggravate not only the heating-induced changes in key soil properties (Varela et al., 2010; Mataix-Solera et al., 2011) but also the consumption of the ground-covering vegetation and litter layer (Shakesby and Doerr, 2006). These findings sustain a generalization of the results obtained by several of the individual studies included in the present dataset. Benavides-Solorio and MacDonald (2001) found much smaller differences in overland flow than in sediment yield between areas burnt at low, medium and high severity. However, contrasting results have been reported as well, Vega et al. (2005) found that both lightly and intensely burnt plots produced significantly more runoff than unburnt plots during the first year after a prescribed fire, while only the intensely burnt plots produced significantly more erosion than the unburnt plots.

The high heterogeneity of the data highlights the use of different methods to estimate soil burn severity

The meta-analysis of the runoff data revealed as main pattern among the three soil burn severity classes a tendency towards a smaller increase following burning at high severity than at low or moderate severity. This tendency could be explained by the - widely accepted - role of soil water repellency in post-fire overland flow generation (e.g. Crockford et al., 1991; Burch et al., 1989; Shakesby et al., 1993; Doerr et al., 1998; Scott, 2000), in combination with the non-linear relationship of changes in soil water repellency with changes in soil heating that is well-established by laboratory studies (e.g. DeBano, 2000; Doerr and Moody, 2004; Varela et al, 2005). In a field study, Doerr et al. (2006) found that burning at high severity typically destroyed the pre-existing water repellency, rendering the topsoil wettable at the majority of sampling points and, thus, less susceptible to the generation of overland flow. Unfortunately, it was impossible to analyse the role of soil water repellency in the present runoff results, as the bulk of the studies included in the dataset did not comprise detailed information on the levels of soil water repellency prior to the start of the RSE's.

In many of the studies included in the dataset analysed here, fire-induced increases in bare soil cover were referred as (one of the) cause(s) of higher runoff and/or erosion rates in recently burnt plots than in long-unburnt plots. Therefore, an additional analysis was carried out using bare soil cover as indicator of soil burn severity as follows: low severity – bare soil cover < 30 % (based on MacDonald and Larsen (2009)); medium severity – 30 % ≤ bare soil cover ≤ 60 %; high severity – bare soil cover > 60 % (based on Johansen et al. (2001a). This additional analysis, however, did not produce "better results" than those shown in **Figure 10**. In terms of runoff effect size, the additional analysis did in fact reveal statistically significant differences between the three severity classes but these differences involved a significantly greater effect of moderate than low/high severity and, as such, were more difficult to comprehend than the contrast between high and low/moderate severity in **Figure 10**. In terms of erosion effect size, the additional analysis equally suggested an increase in effect size with increasing burn severity but the difference between moderate and high severity was not statistically significant. Possibly, bare soil cover would have been a more informative proxy of hydrological and erosion effects, if the meta-analysis were limited to the immediate post-fire period and, hence, the confounding role of time-of-vegetation recovery were minimized.

The role of time-since-fire

An examination of the data used to develop the meta database revealed that in 85 % of the studies examined were evaluated (i.e., monitored) for less than 1.5 years, strongly biasing the evaluation of hydrological responses to short-term fire effects (Figure 11). Other authors (Moody and Martin, 2000; Moody et al., 2013) have also noted the inadequacy of short-term evaluations when evaluating hydrological responses to fire, particularly when delayed erosion occurs (Cerdà., 1998; Larsen, et al., 2009). The fact that led to the impossibility to separate runoff and erosion ratios per different periods since fire can be related site variability and burn severity in the different studies (Figure 11). Several authors referred that the responses of burned areas are transient, often lasting less than seven years, depending on various aspects, as the speed of vegetation recovery, post-wildfire weather conditions, sediment availability and morphology (Rowe et al., 1954; Cerdà, 1998; Moody and Martin, 2001; Gartner et al., 2004; Shakesby et al., 2007; Sheridan et al., 2007; Cannon et al., 2010; Moody et al 2013) and also as MacDonald and Larsen (2009) mentioned different fire severities led to different recovering periods until the background levels.

The role of rainfall intensity

Regarding rainfall intensities, the meta-analysis provided the separation of only one group corresponding to intensities between 60 and 100 mm h^{-1} (Figure 12), as the one that provided the lowest effect size in runoff ratio, and intensities higher than 150 mm h^{-1} that provided the higher effect size for normalized erosion (Figure 12). The overall observation is that rainfall intensity groups might not result in different runoff ratios and normalized erosion responses, verified by the overlapped confidence intervals in most groups (Figure 12). This was also observed by Malvar et al. (2011), when different rainfall intensities from RSE's in post-fire areas were compared. Runoff coefficients from RSE's with 45 vs 80 mm h^{-1} resulted in 54% vs 55% runoff coefficient for one site and 34% vs 38% for another. In the case of the erosion rates the first site increased erosion with intensity 0.094 vs 0.192 g m^{-2} mm_{rain}^{-1}, while in the other, similar ratios were obtained 0.051 vs 0.052 g m^{-2} mm_{rain}^{-1}. The fact that this comparison corresponds to normalized data (divided by total applied rainfall (mm)), means the possible effect of rainfall intensity might be already included due to the inclusion of the rainfall amounts. Thereby, the fact that few intensity classes separations were possible to be made, plus no significant relationship between intensity versus runoff and erosion were achieved, might indicate that rainfall intensity may not be significant variable as their differences are already included in data in the form of rainfall amounts for comparability purposes. However, the extrapolation of these observations into natural rainfall events might not be as straightforward as observed in this study.

Limitations of the dataset and guidelines for future studies

The observed data variability can be attributed to several database weaknesses to represent the study subject (Table 3). These weaknesses are mostly associated to lack of data of interest and lack of representativeness by a disproportional distribution of the cases according to the following categories:

Environmental - Most of the studied cases concern USA (60%) and Spain (30%) and then two other reports in only one location as is the case of Israel and Australia (Table 1). Other regions across a broad range of soil types and environmental conditions have been scarcely studied under the parameters in which concerns this study. This might be related to national priorities in terms of fire related research, adding to the fact that burn severity classification only has been widely used in the last decade (Keeley 2009), although one of the first metrics for fire severity was reported approximately 30 years ago by Ryan and Noste (1985).

Land-use/ Vegetation cover– Most of the analysed cases concern pine forest (38%) and shrubland (26%), followed by eucalypt (10%) and grasslands (10%) as the site dominant vegetation. A better balance and more number of cases between the affected vegetation could provide an improved overview about the effect-size of runoff and erosion after fire for each land-use type. More resource evaluations should be included to arrive at an overall burn severity rating for a particular burned area.

Dataset size - The number of observations is reduced considering the amount of dependent variables of this dataset (Table 1). The leading cause is the fulfilling of these meta-analysis main requirements, which are the existence of a burn severity classification and control data for each study. The lack of such info is justified by the scope of each original study, and also because it could imply several difficulties for field site measurements (e.g. control plots). Nevertheless, we propose for future post-fire studies the inclusion of that information. Even if the study objective might not consider different burn severities or the existence of control plots, their results would improve post-fire data comparability towards a global scientific context. It is as valid as soil texture, vegetation type, climate or amount of rainfall as indicator of comparability.

Non-uniform burn severity classification –From all the possible studies to be included in this meta-analysis only 51% (20 studies out of 39) presented a burn severity classification (or the enough information to be estimated), and from these, only 50% (10 studies out of 20) described a method and factors in which the classification was based. Also, most of the burn severity classification methodology in these studies was made by the author's expert judgment, representing 40% of all the meta-analysis (Figure 8).

The parameters that are more often used for burn severity classifications are post-fire descriptions of soil, ashes, vegetation, duff, litter and also fire temperatures (prescribed/experimental fires). However, frequently only one parameter is used, e.g. ashes for Bentley and Fenner (1958) and rarely all these parameters are used in a single classification, e.g. BAER classification (Robichaud et al 2000; Parsons et al., 2010). It is widely accepted that soil burn severity is low if the protective soil cover by vegetation and litter immediately after fire is higher than 70 %, or bare soil lower than 30% (MacDonald and Larsen, 2009), and that it is high if the bare soil cover is high, say more than exceeds 60 % (Johansen et al., 2001a); however, moderate burnt severity is less well-define in literature, possibly contributing to the lower number of observations in the data set with moderate severity than with low and high severity.

Should be highlighted that the concept of 'burn severity' and the criteria to classify it have been changing through time since the first severity indicators were published (Kelley 2009). Namely, the studies included in the present analysis covered a long period, ranging from 1983 to 2009. Likewise, the classifications applied in these studies were published between 1944 and 2000.

It's highly relevant to present the burn severity classification in any sort of study related with post-fire context. Allied to that, a uniform global classification is still needed, since burn severity classification depends of the combination of several indicators. Ideally, a sum of indicators that already are presented in the existing burn severity classification methodologies, with their respective impact on soil and hydrological response in a quantitative matter would be the answer for a more well defined methodology and accurate classification. Additionally, these severity indicators should take into account the pre-fire conditions, either by determination of these indices in site before the prescribed fire or by their estimation in adjacent unburnt areas in the case of the wildfires. This would allow the determination of the real fire impact in relation to the pre-fire conditions, whereas burn severity classification would represent a measurement of change between pre and post-fire. Some authors (Moody et al 2013) state that the knowledge of the relations between soil properties and burn severity metrics are the highest priority for future post-fire runoff and erosion research.

Table 3 - Main weaknesses and guidelines for future studies, in the current scientific evidence on the relationship between runoff and erosion after fire and fire severity classes (see for more detailed discussion chapter 4.2).

Weaknesses		Guidelines for future studies
Categories	Description	
Environmental	Studied reports concern limited number of situations, climate and soil types.	Authors should include control plots in their research, even if is not the objective of their study. Their results would improve post-fire data comparability towards a global scientific context. Burn severity classification should be present in all the studies related with post-fire context.
Land Use/ Vegetation	Few vegetation and land uses types were analysed. Numbers of cases are not uniformly distributed according to their type.	
Numbers	The number of observations is low considering the number of variables involved	
Non-uniform severity classification	Usage of different classifications amongst the several studies, and the fact that they not always based in common fire severity factors.	Any sort of study related with post-fire context, should present the fire severity classification and the criteria in which the burn severity is based. Control data could provide the level of change due to burn severity.
Land Management	Data lacking on soil management (e.g. tillage and rotation). Information about the use of heavy machinery, immediately after fire or just as regular use for site management is very scarce.	The reference to several parameters, that reflect antecedent conditions, site characteristics and experiment conditions, would allow improving this meta-analysis in the determination of other variables in which data can be influenced besides the fire itself.
Soil Characteristics	Texture, Organic matter content, bulk density, soil moisture before/after rainfall simulation, soil moisture at field capacity, bare soil cover, are not always available for all the case studies	
Study design	Rainfall amount/intensity, plot size, drop fall height, nozzle number variations, might influence the runoff and erosion rates, and thereby might be contradictory to the fire severity classifications in spite of the usage of normalized data.	
Time since fire	Might influence the runoff and erosion and contradictory to the fire severity classifications.	Long-term monitoring is advisable towards an adequate evaluation of burn severity impact

Land Management – Would be important to refer also the existence of post-fire interventions (e.g. salvage logging using heavy machinery, erosion mitigation), also in the case of forests for commercial purposes information on soil management (e.g. tillage and rotation), to clarify the possible background factors that might influence the rainfall simulation results.

Soil Characteristics – This analysis was made transversally to the soil characteristics, and thereby should include variability in the runoff and erosion data as a reflection of their differences (e.g. permeability, soil moisture field capacity, erodibility).

Study design /Rainfall simulation auxiliary variables – The conditions in which the rainfall simulation took place. Plot size, slope and nozzle high in which the artificial rain was applied can influence the results (Smets et al., 2008). In this study the wide range of conditions in which the RSE's were performed, regarding rainfall properties (Figure 6) or field and equipment conditions (Figure 7), cause a correspondingly large range in the runoff and erosive response. However, the influence of this wide range of methodologies does not have major implications in the effect size result, since control and burned plots are submitted to similar RSE's conditions.

The reference to parameters such as bare soil, vegetation cover, soil moisture and soil water repellence before and after the simulation, number of replicates or standard deviation, could also improve this meta-analysis in the sense of determine the influence of several other variables in which data can be influenced besides the fire itself.

Time since fire – Most of the study cases were included in the first 1.5 years after the fire (86%), however there are cases of rainfall simulations performed after that period in which the severity classification can be compromised. More data concerning long-term monitoring would also benefit such study as the one performed here, plus would allow the evaluation of the total impact of fire in the monitored areas, especially for different fire severities.

Representativeness and maintenance of this meta-analysis

The meta-database assembled for analysing the hydrological responses on wildland areas exposed to different soil burn severities during prescribed and wildfires was limited to a few vegetation types and climatic regimes. All studies complied were obtained from publications where rainfall simulators were used on small plots. The database generated for the current analysis is available to other fire researchers where it hopefully will be used by future fire researchers and managers to improve the understanding of soil burn severity effects on post-fire hydrological responses and lead to the improvement of the current fire severity classification systems.

This meta-analysis is expected to be maintained and improved during the next years. Since the latest reviews (Shakesby 2011, Moody et al 2013) have highlighted a possible increase of wildfire occurrence, linked with increased human activity (Wittenberg and Malkinson 2009) and with climate change (Flannigan et al., 2000; Westerling et al., 2003; Bachelet et al., 2007; Moritz et al., 2010; Westerling et al., 2011). And by the expected increase of prescribed burning use as mitigation strategy to avoid highly severe fire occurrence as proposed by Shakesby (2011). In this way is expected that post-fire hydrological response studies will prevail for some time, especially with such widely used method as the RSE's. Followed by that, is expected also that an extension of this dataset would produce a more representative meta-analysis of the current subject

All existing data should be made available as much as possible in a transparent way, with full disclosure of data, statistics and funding. In all cases, it is advised that studies are reported according to the guidelines described above. The database used for this meta-analysis is publicly available to download at (Background dataset for online publication).

Conclusions

The review, database compilation and meta-analysis on post-fire rainfall simulation studies, allowed to identify some important issues regarding burn severity. Statistical meta-analysis showed that fire occurrence had a significant effect on the hydrological response, increasing the runoff coefficient in comparison to unburnt conditions for each of the burn severity classes. The runoff effect size, however, did not differ significantly between the three soil burn severities. Meta-analysis also revealed a significantly enhanced post-fire erosion response ratio, and this enhanced response differed significantly between all three severity classes. Verifying an increasing erosion rate, with increasing burn severity.

The analysis of the runoff and erosion ratios according to rainfall intensities and time-since-fire classes resulted in overlapped confidence intervals in most groups. Further analysis with other parameters couldn’t be performed without reducing the number of observations. Nevertheless, a meta-analysis focused in bare soil cover classes as a substitute for burn severity was performed, but no improvement were obtained.

Dataset evaluation revealed several weaknesses, often associated to limited data availability in the research articles. The absence of burnt severity classification and control RSE’s limited greatly the number of observations for this review. While the limited availability of RSE’s auxiliary variables limited further explanatory analysis.

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References

Abalos, D., Jeffery, S., Sanz-Cobena, A., Guardia, G., Vallejo, A. 2014. Meta-analysis of the effect of urease and nitrification inhibitors on crop productivity and nitrogen use efficiency. Agriculture, Ecosystems and Environment. 189, 136-144.

Arcenegui, V., Guerrero, C., Mataix-Solera, J., Mataix-Beneyto, J., Zornoza, R., Morales, J., Mayoral, A.M., 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.

Bachelet, D., Lenihan, J.M., Nielson, R.P., 2007. Wildfires and Global Climate Change. The Importance of Climate Change for Future Wildfire Scenarios in the Western United States, Excerpted From Ebi, Kristie, et al., Regional Impacts of Climate Change, Four Case Studies in the United States, Pew Center on Global Climate Change (20 pp. Available at http://www.pewclimate.org/docUploads/Regional-Impacts-West.pdf, last accessed 17 July 2013).

Benavides-Solorio, J., MacDonald, L.H., 2001. Post–fire runoff and erosion from simulated rainfall on small plot, Colorado Front Range. Hydrological Processes. 15, 2931–2952.

Benavides-Solorio, J.D.D., MacDonald, L.H., 2005. Measurement and prediction of post-fire erosion at the hillslope. International Journal of Wildland Fire 14, 457–474.

Bentley, J.R., Fenner, R.L., 1958. Soil temperatures during burning related to postfire seedbeds on woodland range. Journal of Forestry. 56, 737-740.

Blackburn, W.H., Meeuwig, R.O., Skau, C.M., 1974. A mobile infiltrometer for use on rangeland. Journal of Range Management. 27: 322-323.

Blake, W.H., Doerr, S.H., Shakesby, R.A., Wallbrink, P.J., Humphreys, G.S., Chafer, C.J., 2006. Tracing eroded soil in a burnt water supply catchment, Sydney, Australia: linking magnetic enhancement to soil water repellency. In: Owens, P.N. Collins, A.J. (Eds.), Soil erosion and sediment redistribution in river catchment. CAB International, pp. 62–69.

Brock, J.H., DeBano, L.F., 1990. Wettability of an Arizona chaparral soil influenced by prescribed burning. In: Proceedings of a symposium on effects of fire of Southwestern natural resources (Krammes, J.S., technical coordinator). November 15–17, 1988. Tucson, AZ. USDA Forest Service General Technical Report RM-191, pp. 206–209.

Bubenzer, G.D., Meyer, L.D., 1965. Simulation of rainfall and soils for laboratory research. Transactions of the ASAE. 17, 1033– 1037.

Burch, G.J., Moore, I.D., Burns, J., 1989. Soil hydrophobic effects on infiltration and catchment runoff. Hydrological Processes. 3, 211–222.

Cannon, S.H., Gartner, J.E., Rupert, M.G., Michael, J.A., Rea, A.H., Parrett, C., 2010. Predicting the probability and volume of post-wildfire debris flows in the intermountain west, USA. Geological Society of America Bulletin. 122, 127–144.

Cerdà, A., 1998. Changes in overland flow and infiltration after a rangeland fire in a Mediterranean scrubland. Hydrological Processes. 12, 1031–1042.

Cerdà, A, Ibáñez, S., Calvo, A., 1997. Design and operation of a small and portable rainfall simulator for rugged terrain. Soil Technology. 11, 163-170.

Cerdà, A., Doerr, S.H., 2005. Influence of vegetation recovery on soil hydrology and erodibility following fire: an 11-year investigation. International Journal of Wildland Fire 14, 423–437.

Cerdà, A., Mataix-Solera, J., 2009. Efectos de los incendios forestales sobre los suelos en España. El estado de la cuestión visto por los científicos españoles. Cátedra de Divulgación de la Ciencia-Fuegored, Valencia. (529 pp.).

Chafer, C.J., 2008. A comparison of fire severity measures: An Australian example and implications for predicting major areas of soil erosion. Catena. 74, 235–245.

Cochran, W.G., 1954. The combination of estimates from different experiments. Biometrics, 10, 101–129.

Cooper, H.C., Hedges, L.V., Valentine, J.C., 2009. The handbook of research synthesis and meta-analysis, second Ed. New York, Russell Sage Foundation.

Crockford, S., Topadalis, S., Richardson, D.P., 1991. Water repellency in a dry sclerophyll forest—measurements and processes. Hydrological Processes. 5, 405–420.

DeBano, L.F., 2000. The role of fire and soil heating on water repellency in wildland environments: A review. Journal of Hydrology. 231–232, 195–206.

DeBano, L.F., Neary, D.G., Ffolliott, P.F., 1998. Fire's Effects on Ecosystems. John Wiley & Sons, New York. 159–196.

DeBano, L.F., Neary, D.G., Ffolliott, P.F., 2005. Soil physical properties. In: Neary, D.G. Ryan, K.C. DeBano, L.F. (Eds.), Wildland Fire in Ecosystems. Effects of Fire on Soil and Water. General Technical Report RMRS-GTR-42, Volume 4. USDA Forest Service, Rocky Mountain Research Station, pp. 29–52.

De Luis, M., González-Hidalgo, J.C., Raventós, J., 2003. Effects of fire and torrential rainfall on erosion in a Mediterranean gorse community. Land Degradation and Development. 14, 203–213.

Doerr, S.H., Shakesby, R.A., Walsh, R.P.D., 1998. Spatial variability of soil hydrophobicity in fire-prone eucalyptus and pine forests, Portugal. Soil Science. 163, 313–324.

Doerr, S.H., Shakesby, R.A., Walsh, R.P.D., 2000. Soil water repellency: its characteristics, causes and hydro-geomorphological consequences. Earth Science Reviews. 51, 33–65.

Doerr, S.H, Moody, J., 2004. Hydrological impacts of soil water repellency: on spatial and temporal uncertainties. Hydrological Processes 18: 829– 832.

Doerr, S.H., Shakesby, R.A., Blake, W.H., Chafer, C.J., Humphreys, G.S., Wallbrink, P.J., 2006. Effects of differing wildfire severities on soil wettability and implications for hydrological responses. Journal of Hydrology. 319, 295–311.

Dobrowolsky, J.P., Blackburn, W.H., Pearson, H.A., 1992. Changes to infiltration and interrill erosion from Long-term prescribed burning in Louisiana. Water Resources Bulletin. 28(2), 287-298.

Emmerich, WE, Cox, JR., 1992. Hydrologic characteristics immediately after seasonal burning on introduced and native grasslands. Journal of Range Management. 45: 476–479.

Emmerich, WE, Cox, JR., 1994. Changes in surface runoff and sediment production after repeated rangeland burns. Soil Science Society of America Journal. 58: 199–203.

Fernández, C., Vega, J.A., Fontúrbel, T., Pérez-Gorostiaga, P., Jiménez, E., Pérez, J.R., 2006. Runoff and soil erosion after rainfall simulations in burned soils. Proceedings of V International Conference on Forest Fire Research. D.X. Viegas (Ed.). Forest Ecology and Management. 234. Supplement 1. Rotterdam: Millpress Science Publishers. CD-ROM.

Fernández, C., Vega J.A., Fontúrbel, M.T., Jiménez, E., Pérez, J.R., 2008. Immediate effects of prescribed burning, chopping and clearing on runoff, infiltration and erosion in a shrubland area in Galicia (NW Spain). Land Degradation & Development. 19, 502–515.

Fernández, C., Vega, J.A., Fontúrbel, M.T., 2012. The effects of fuel reduction treatments on runoff, infiltration and erosion in two shrubland areas in the north of Spain. Journal of Environmental Management. 105, 96-102.

Fernández, C., Vega, J.A., Vieira, D.C.S., 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, 774-87.

Ferreira, A.J.D., Coelho, C.O.A., Boulet, A.K., Leighton-Boyce, G., Keizer, J.J., Ritsema, C.J., 2005. Influence of burning intensity on water repellence and hydrological processes at forest and shrub sites in Portugal. Australian Journal of Soil Research. 43(3), 327-336.

Fiener, P., Seibert, S.P., Auerswald, K., 2011. A compilation and meta-analysis of rainfall simulation data on arable soils. Journal of Hydrology. 409(1–2), 395-406.

Flannigan, M.D., Stocks, B.J., Wotton, B.M., 2000. Climate change and forest fires. Science of the Total Environment. 262, 221–229.

Gartner, J.E., Bigio, E.R., Cannon, S.H., 2004. Compilation of post wildfire runoff-event data from the western United States. (Online USGS Open-File Report 2004–1085, http://pubs.usgs.gov/of/2004/1085 [accessed: 17 July 2013]).

Gimeno-Garcia, E., Andrea, V., Rubio, J.L., 2007. Influence of vegetation recovery on water erosion at short and medium-term after experimental fires in a Mediterranean shrubland. Catena. 69, 150–160.

Groen, A.H., Woods, S.W., 2008. Effectiveness of aerial seeding and straw mulch for reducing post-wildfire erosion, north-western Montana, USA. International Journal of Wildland Fire. 17, 559–571.

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 (3), 1029–1037.

Gurevitch, J, Hedges, L.V., 1993. Meta-analysis: combining the results of independent experiments. In: Scheiner S. M. and Gurevitch J. (Eds.). The design and analysis of ecological experiments. Oxford University Press. New York, 378-398.

Hester, J.W., Thurow, T.L., Taylor, C.A., 1997. Hydrologic characteristics of vegetation types as affected by prescribed burning. Journal of Range Management. 50,199-204.

Huffman, E.L., MacDonald, L.H., Stednick, J.D., 2001. Strength and persistence of fire-induced soil hydrophobicity under ponderosa and lodgepole pine, Colorado Front Range. Hydrological Processes. 15, 2877– 2892.

Jain, T.B.; Pilliod, D.S.; Graham, R.T.; Lentile, L.B.; Sandquist, J.E. 2012. Index for characterizing post-fire soil environments in temperate coniferous forests. Forests. 3: 445-466.

Johansen, P.J., Hakonson, T.E., Breshears, D.D., 2001a. Post fire runoff and erosion from rainfall simulation: contrasting forests with shrublands and grasslands. Hydrological Processes. 15, 2953–2965.

Johansen, M.P., Hakonson, T.E., Whicker, F.W., Simanton, R.J., Stone, J.J., 2001b. Hydrologic response and radionuclide transport following fire at semiarid sites. Journal of Environmental Quality. 30, 2010-2017.

Johnson DW, Curtis PS. 2001. Effects of forest management on soil C and N storage: a meta-analysis. Forest Ecology and Management 140, 227-238.

Kalies, E.L., Chambers, C.L., Covington, W.W., 2010. Wildlife responses to thinning and burning treatments in southwestern conifer forest: A meta-analysis. Forest Ecology and Management. 259, 333-342.

Keeley, J.E., 2009. Fire intensity, fire severity and burn severity: a brief review and suggested usage. International Journal of Wildland Fire. 18, 116–126.

Knight, R.W., Blackburn, W.H., Scifres, C.J., 1983. Infiltration rates and sediment production following herbicide/fire brush treatments. Journal of Range Management. 36 (2), 154-157.

Knorr, M., Frey, S.D., Curtis, P.S., 2005. Nitrogen additions and litter decomposition: A meta-analysis. Ecology. 86, 3252-3257.

Kopper, K.E., McKenzie, D., Peterson, D.L., 2009. The evaluation of Meta-analysis techniques for quantifying prescribed fire effects on fuel loadings. USDA For. Serv. Res. Pap. PNW-RP-582. 24 pp.

Kutiel, P., Lavee, H., Segev, M., Benyamini, Y., 1995. The effect of fire-induced surface heterogeneity on rainfall-runoff-erosion relationships in an eastern Mediterranean ecosystem, Israel. Catena. 25, 77–87.

Larsen, I.J., MacDonald, L.H., Brown, E., Rough, D., Welsh, M.J., Pietraszek, J.H., Libohova, Z., Schaffrath, K., 2009. Causes of post-fire runoff and erosion: the roles of soil water repellency, surface cover, and soil sealing. Soil Science Society of America Journal. 73(4), 1393-1407.

MacDonald, L.H., Huffmann, E.L., 2004. Post-fire soil water repellency: Persistence and soil moisture thresholds. Soil Science Society of America Journal. 68, 1729–1734.

MacDonald, L.H., Larsen, I.J., 2009. Effects of forest fires and post-fire rehabilitation: a Colorado Case study. In: Fire Effects on Soils and Restoration Strategies, edited by A. Cerda and P.R. Robichaud. Science Publishers, Enfield, NH, pp. 423-452.

Maia, P., Pausas, J.G., Arcenegui, V., Guerrero, C., Pérez-Bejarano, A., Mataix-Solera, J., Varela, M.E.T., Fernandes, I., Pedrosa, E.T., Keizer, J.J., 2012. Wildfire effects on the soil seed bank of a maritime pine stand — The importance of fire severity. Geoderma. 191 (2012) 80–88.

Malvar, M.C., Prats, S.A., Nunes, J.P., Keizer, J.J., 2011. Post-fire overland flow generation and inter-rill erosion under simulated rainfall in two eucalypt stands in north-central Portugal. Environmental Research. 111, 222-236.

Martin, D.A., Moody, J.A., 2001. Comparison of soil infiltration rates in burned and unburned mountainous watersheds. Hydrological Processes. 15, 2893–2903.

Mataix-Solera, J., Doerr, S.H., 2004. Hydrophobicity and aggregate stability in calcareous topsoil from fire affected pine forests in southeastern Spain. Geoderma. 118, 77–88.

Mataix-Solera, J., Guerrero, C., 2007. Efectos de los incendios forestales en las propriedades edáficas. In: Mataix-Solera, J. (Ed.), Incendios Forestales, Suelos y Erosión Hídrica. Caja Mediterráneo CEMACAM Font Roja-Alcoi, Alicante, pp. 5–40.

Mataix-Solera, J., Cerdà, A., Arcenegui, V., Jordán, A., Zavala, L.M., 2011. Fire effects on soil aggregation: A review. Earth-Science Reviews. 109(1–2), 44-60.

Mayor, A.G., Bautista, S., Llovet, J., Bellot, J., 2007. Post-fire hydrological and erosional responses of a Mediterranean landscape: seven years of catchment scale dynamics. Catena. 71, 68–75.

Meyer, L.D., Harmon, W.C., 1979. Multiple-intensity rainfall simulator for erosion research on row side slopes. Transactions of the American Society of Agricultural Engineers. 22: 100–103.

Moody, J.A., Martin, D.A., 2001. Initial hydrologic and geomorphic response following a wildfire in the Colorado Front Range. Earth Surface Processes and Landforms. 26, 1049–1070.

Moody, J.A., Shakesby, R.A., Robichaud, P.R., Cannon, S.H., Martin, D.A., 2013. Current research issues related to post-wildfire runoff and erosion processes. Earth-Science Reviews. 122, 10–37.

Morin, J., Goldberg, D., Seginer, I., 1967. A rainfall simulator with rotating disk. Trans. ASAE, 10, 1085-1093.

Moritz, M.A., Moody, T.J., Krawchuk, M.A., Hughes, M., Hall, A., 2010. Spatial variation in extreme winds predicts large wildfire locations in chaparral ecosystems. Geophysical Research Letters. 37, L04801.

Navas, A., Alberto, F., Machín, J., Galán, A., 1990. Design and operation of a rainfall simulator for field studies on runoff and soil erosion. Soil Technology. 3: 385–397.

Neary, D.G., Klopatek, C.C., DeBano, L.F., Ffolliott, P.F., 1999. Fire effects on belowground sustainability: a review and synthesis. Forest Ecology and Management. 122, 51–71.

Neary, D.G., Ryan, K.C., DeBano, L.F., 2005. Wildland fire in ecosystems. Effects of fire on soil and water. USDA Forest Service, Rocky Mountain Research Station. General Technical Report RMRS-GTR-42-vol 4, Ogden, UT.

Nunes, J.P., Seixas, J., Keizer, J.J., Ferreira, A.J.D., 2009. Sensitivity of runoff and soil erosion to climate change in two Mediterranean watersheds. Part I: model parameterization and evaluation. Hydrological Processes. 23 (8), 1202-1211.

Parsons, A., Robichaud, P., Lewis, S., Napper, C., Clark, J., 2010. Field guide for mapping post-fire soil burn severity. USDA For Serv Gen Tech Rep RMRS-GTR-243.

Pausas, J.G., Llovet, J., Rodrigo, A., Vallejo, R., 2008. Are wildfires a disaster in the Mediterranean basin?-A review. International Journal of Wildland Fire. 17:713-723.

Pierson, F.B., Robichaud, P.R., Spaeth, K.E., 2001. Spatial and temporal effects of wildfire on the hydrology of a steep rangeland watershed. Hydrological Processes. 15, 2905–2916.

Pierson, F.B., Carlson, D.H., Spaeth, K.E., 2002. Hydrologic impacts of fire on steep intermountain sagebrush–steppe rangeland. International Journal of Wildland Fire. 11, 145–151.

Pierson, F.B., Robichaud, P.R., Moffet, C.A., Spaeth, K.E., Hardegree, S.P., Clark, P.E., Williams, C.J., 2008. Fire effects on rangeland hydrology and erosion in a steep sagebrush-dominated landscape. Hydrological Processes. 22(16), 2916-2929.

Prats, S.A., Malvar, M.C., Vieira, D.C.S, MacDonald, L.H., Keizer, J.J., 2013. Effectiveness of hydro-mulching to reduce runoff and erosion in a recently burnt pine plantation in central Portugal. Land Degradation & Development. (DOI: Link: 10.1002/ldr.2236)

Prosser, I.P., Williams, L., 1998. The effect of wildfire on runoff and erosion in native Eucalyptus forest. Hydrological Processes. 12, 251–265.

R Development Core Team, 2012. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. ISBN 3-900051-07-0, URL http://www.R-project.org/.

Reeder, C.J., Jurgensen, M.F., 1979. Fire-induced water repellency in forest soils of upper Michigan. Canadian Journal of Forest Research. 9, 369–373.

Robichaud, P.R., 2000. Fire Effects on infiltration rates prescribed fire in Northern Rocky Mountain forests, USA. Journal of Hydrology. 231–232, 220–229.

Robichaud, P.R.; Hungerford, R., 2000. Water repellency by laboratory burning of four northern Rocky Mountain forest soils. Journal of Hydrology. 231–232: 207–219.

Robichaud, P.R.; Beyers, J.L.; Neary, D.G., 2000. Evaluating the effectiveness of post fire rehabilitation treatments. Gen. Tech. Rep. RMRS-GTR-63. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 85 p.

Robichaud, P.R., Elliot, W.J., Pierson, F.B., Hall, D.E. Moffet, C.A., Ashmun, L.E., 2007. Predicting post fire erosion and mitigation effectiveness with a web-based probabilistic erosion model. Catena. 71, 229–241.

Rowe, P.B., Countryman, C.M., Storey, H.C., 1954. Hydrologic Analysis Used to Determine Effects of Fire on Peak Discharge and Erosion Rates in Southern California Watersheds. USDA Forest Service, California Forest and Range Experimental Station (49 pp.).

Rulli, M.C., Bozzi, S., Spada, M., Bocchiola, D., Rosso, R., 2006. Rainfall simulations on a fire disturbed Mediterranean area. Journal of Hydrology 327, 323–338.

Ryan, K.C., Noste, N.V., 1985. Evaluating prescribed fires In: Lotan JE, Kilgore BM, Fischer WC Mutch RW (Tech Coord). Workshop of Wilderness Fire. USDA Forest Service Res Sta, Proc Gen Tech Rep INT-182, pp 230–238.

Sampson, A.W., 1944. Plant succession on burned chaparral lands in northern California. Univ. Calif. Coll.Agric. Exp. Stn. Bull. 685, 144 pp.

Schmidt, M.W., Noack, A.G., 2000. Black carbon in soils and sediments: analysis, distribution, implications and current challenges. Global Biochemical Cycles. 14, 777–793.

Scott, D.F., 2000. Soil wettability in forested catchments in South Africa: as measured in different methods and as affected by vegetation cover and soil characteristics. Journal of Hydrology. 231–232, 87–104.

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

Shakesby, R.A., Coelho, C.O.A., Ferreira, A.D., Terry, J.P., Walsh, R.P.D., 1993. Wildfire impacts on soil erosion and hydrology in wet Mediterranean forest, Portugal. International Journal of Wildland Fire. 3, 95–110.

Shakesby, R.A., Doerr, S.H., 2006. Wildfire as a hydrological and geomorphological agent. Earth Science Reviews. 74, 269–307.

Shakesby, R.A., Wallbrink, P.J., Doerr, S.H., English, P.M., Chafer, C.J., Humphreys, G.S., Blake, W.H., Tomkins, K.M., 2007. Distinctiveness of wildfire effects on soil erosion in south-east Australian eucalypt forests assessed in a global context. Forest Ecology and Management. 238, 347–364.

Sheridan, G.J., Lane, P.N.J., Noske, P.J., 2007. Quantification of hillslope runoff and erosion processes before and after wildfire in a wet Eucalyptus forest. Journal of Hydrology. 343, 12–28.

Smets, T., Poesen, J., Knapen, A., 2008. Spatial scale effects on the effectiveness of organic mulches in reducing soil erosion by water. Earth Science reviews. 89, 1-12.

Sokal, R.R., Rohlf, F.J, 1981. Biometry. Second Edition. W.H. Freeman and Company, New York.

Swanson, F.J., 1981. Fire and geomorphic processes. In: Mooney, H.A. Bonnicksen, T.M. Christiansen, N.L. Lotan, J.E. Reiners, W.A. (Eds.), United States Department of Agriculture, Forest Service, General Technical Report WO-26. United States Government Planning Office, Washington DC, pp. 401–421.

Swanson, N.P., 1965. Rotating-boom rainfall simulator. Trans. ASAE 8:71–72.

Úbeda, X., Outeiro, L., 2009. Physical and chemical effects of fire on soil. In: A. Cerdà, P.R. Robichaud (Eds.), Fire Effects on Soils and Restoration Strategies. Science Publishers, Enfield, NH, USA, pp. 105-132.

USDA Forest Service, 1995. Burned area rehabilitation handbook. Forest Service Handbook 2509.13, Amendment nº 2509-1395.7. Washington D.C..

USDA (United Stated Department of Agriculture), 2000. Fire burn intensity classification factsheet, Natural resources conservation service 2000. (http://www.co.nrcs.usda.gov/technical/eng/BURNINTENSITYfactsheet.pdf.).

Varela M.E., Benito E., de Blas E., 2005. Impact of wildfires on surface water repellency in soils of northwest Spain. Hydrological Processes. 19, 3649-3657.

Varela, M.E., Benito, E., Keizer J.J., 2010. Effects of wildfire and laboratory heating on soil aggregate stability of pine forest in Galicia: the role of lithology, soil organic matter content and water repellency. Catena. 83, 127-134

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

Vega, J.A., Fontúrbel, T., Merino, A., Fernández, C., Ferreiro, A., Jiménez, E., 2013a. 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.

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

Viechtbauer, W. 2010. Conducting meta-analysis in R with the metafor package. Journal of Statistical Software.36 (3), 1-48.

Viechtbauer, W. 2014. The metafor package: A meta-analysis package for R. http://www.metafor-project.org/doku.php

Wan, S, Hui, D, Luo, Y., 2001. Fire effects on nitrogen pools and dynamics in terrestrial ecosystems: a meta-analysis. Ecological. Applications. 11, 1349-1365.

Wells, C.G., Campbell, R.E., DeBano, L.F., Lewis, C.E., Fredriksen, R.L., Franklin, E.C., Froelich, R.C., Dunn, P.H., 1979. Effects of fire on soil. A state-of-knowledge review. Gen. Tech. Rep. WO-7. Forest Serv., US Department of Agriculture: Washington, DC; 34.

Westerling, A.L., Gerchunov, A., Brown, T.J., Cayan, D.R., Dettinger, M.D., 2003. Climate and wildfire in the western United States. Bulletin of the American Meteorological Society 84 (5), 595–604.

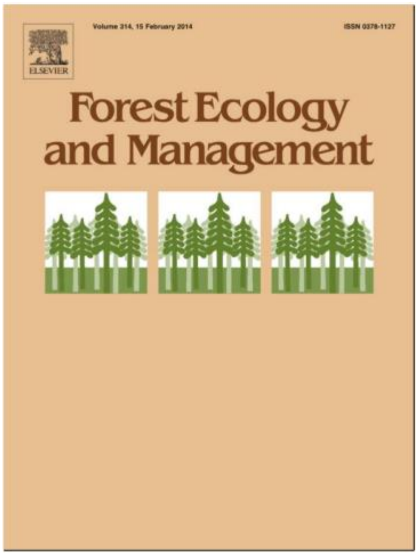
Westerling, A.L., Turner, M.G., Smithwick, E.A.H., Romme, W.H., Ryan, M.G., 2011. Continued warming could transform Greater Yellowstone fire regimes by mid-21st century. Proceedings of the National Academy of Sciences. 108 (32), 13165–13170.

Wilcox, B.P., Wood, M.K., Tromble, J.T., Ward, T.J., 1986. A hand-portable single nozzle rainfall simulator designed for use on steep slopes. Journal of Range Management. 39: 75–377.

Wittenberg, L., Malkinson, D., 2009. Spatio-temporal perspectives of forest fire regimes in a maturing Mediterranean mixed pine landscape. European Journal of Forest Research. 128, 297–304.

Zavala, L.M., 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.

II.II Land cover, land management and time since fire



submitted

Annual overland flow and interrill erosion in contrasting forest plantations during the first four years after a wildfire

Vieira D.C.S., Malvar M.C., Fernández C., Keizer J.J. (2015).
Forest Ecology and Management (submitted).

The impact of forest fire on overland flow and soil erosion has been assessed by many studies, and their effect on fires on hydrological and geomorphological activity both globally and in the Mediterranean region has been well established. However, few studies have assessed post-fire erosion over multiple years, and the authors are aware of none which assess runoff at the plot scale. In addition, few studies are available which consider the effect of pre-fire management practices.

This study evaluates annual overland flow and erosion rates during four years after a wildfire between sites with different land uses (pine and eucalypt) and different pre-fire land management techniques (unplowed, eucalypt downslope plowing, and eucalypt contour plowing). After the four years, runoff coefficients in the unplowed pine site (34%) were higher than in the unplowed eucalypt site (14%). Runoff coefficients at the downslope plowed eucalypt site (26%) and contour plowed eucalypt (37%) were also higher than in the unplowed one. Median sediment losses over the four years followed runoff differences, with 0.39 (Mg.ha^{-1}.year^{-1}) in the pine, 0.11 (Mg.ha^{-1}.year^{-1}) in the eucalypt unplowed, 0.47 (Mg.ha^{-1}.year^{-1}) in the eucalypt downslope plowed, and 0.83 (Mg.ha^{-1}.year^{-1}) in the eucalypt contour plowed. The type of land use affected annual runoff, while land management affected both annual runoff and erosion amounts significantly. Time since fire had an important effect on erosion amounts among unplowed sites, while for eucalypt sites time affected both annual runoff and erosion amounts. Annual runoff and erosion followed rainfall patterns during the four years of monitoring. At all studied sites, the runoff coefficients increased over the four years of monitoring. On the other hand, the sediment concentration in the runoff showed a decrease during the same period. The reasons for this divergence from the classic post-fire recovery model are also explored. The severe soil degradation of this study site is primarily attributed to the interval of fire recurrence and forest management practices during the pre-fire period..

Introduction

Frequent wildfire occurrence is a natural phenomenon in regions with a Mediterranean-type climate (Naveh 1990). However, the current rate of occurrence in southern Europe is unprecedented when compared to the natural cycle, and strongly reflects human activity both through direct ignition (Veléz, 2009) and through land use changes, such as land abandonment and the widespread introduction of highly flammable pine and eucalypt plantations (Moreira et al., 2009; Shakesby, 2011). On average, wildfires consume 500,000 ha/yr. in southern Europe (San-Miguel and Camia 2009), 100,000 ha of which typically occurs in only in Portugal (Pereira et al., 2006a). Fire activity in Portugal is not expected to decline markedly in the foreseeable future, both because of the use of highly flammable tree species in the economically important forestry sector, and the likely increase in the occurrence of meteorological conditions conducive to fire (Carvalho et al., 2010; Pereira et al., 2006b; Harding et al., 2009).

The main hydrologic consequence of wildfire is an increase in runoff and erosion. This effect is commonly attributed to the partial or complete combustion of vegetation and litter, together with soil properties alterations associated to the impact of high temperatures in the soil. These change include a reduction in aggregate stability (e.g. Varela et al., 2010, Mataix-Solera et al., 2011) and an increase in soil water repellency (SWR; e.g. Scott et al., 1998), a phenomenon which is widely reported in burned forest soils (e.g. Wells 1981, Vega and Díaz-Fierros, 1987; Prosser, 1990; Walsh et al., 1994; Keizer et al., 2008). These increases in runoff and erosion following wildfire have been observed for the two principal forest types in north-central Portugal, i.e. maritime pine and eucalypt plantations (e.g. Coelho et al., 2004; Ferreira et al., 1997; Shakesby et al., 1994).

Another important factor in runoff and erosion in Portuguese woodlands are post-fire management practices, such as plowing, terracing, clearcutting, and logging (Terry, 1996, Fernández et al., 2004, 2007; Martins et al., 2013; Shakesby et al., 1993, 2002). Due to socio-economic drivers, the occurrence of these post-fire practices is becoming increasingly common. Post-fire practices involving ground preparations or heavy machinery can have significant impacts on topsoils, which can lead to similar or even higher hydrological effects compared to the fire (Fernández et al 2007, Martins et al 2013, Shakesby et al., 2002). Over the long-term, these practices are seem to be executed between fire occurrences in the same site, which can magnify the negative effects of the fires. Limited research has been made examining the effect of pre-fire management practices on post-fire impacts (Malvar et al 2011; 2015). However, Shakesby (2011) has highlighted that the significance of post-fire erosion in Mediterranean fire-prone areas can only be accurately assessed if associated land management changes and disturbance effects are also considered.

Emmerich and Cox (1992) emphasized the low post-fire erosion rates reported in the Mediterranean, when compared to post-fire erosion rates reported elsewhere, and when compared to other forms of disturbances in unburned areas, such as tillage (García-Ruiz et al., 2015). These low post-wildfire erosion rates in the Mediterranean have been justified by several authors by the presence of shallow soils and high stone content (Cerdan et al., 2010). Due to the appearance of a stone armour, protecting the sediments availability of transport (Shakesby, 2011), providing a higher surface roughness thus limiting post-fire erosion (Kutiel et al., 1995), and due to the possible impact on water percolation through a water-repellent soils (Urbanek and Shakesby, 2009). These thin soils with high stone content are a consequence of land degradation, caused by the long history of anthropogenic impacts in the Mediterranean, due to activities such as deforestation, intensive cultivation, land misuse, and rural abandonment (Dunjó et al., 2004). This degradation is particularly problematic, given the typically low soil formation rates in Mediterranean soils (Cerdà, 2001; López-Bermúdez, 2002).

There have been relatively few studies monitoring erosion continuously for several years following wildfire, and therefore less is known about erosion rates after the first year post-fire (Benavides-Solorio and MacDonald, 2001). Those studies which have been conducted suggest that post-fire erosion during the window of disturbance takes the form of a peak lasting 1–2 years, followed by a decline of varying degrees returning back to pre-fire conditions (e.g. Helvey, 1980; Robichaud and Waldrop, 1994; Inbar et al., 1998).

The aim of this study is to compare long-term overland-flow and sediment losses within two distinctive representative Portuguese forest land use types, and between three different land management practices. The specific objectives of the study focus on the assessment of several factors that might influence post-fire hydrologic and erosive response: (i) the effect of land use among unplowed pine and eucalypt; (2) the effect of several pre-fire land management operations within unplowed, downslope plowed, and contour-plowed; and (3) the influence of time-since-fire during the four years of monitoring period, for both unplowed and eucalypt sites. Furthermore, these four years of observation will also be compared with the existing window of disturbance models (Prosser and Williams, 1998; Wittenberg and Inbar, 2009).

Materials and methods

Study area and sites

This study was carried out in an entirely burned catchment of approximately 10 ha near Colmeal village (40º08’46’’N, 7º59’50’’W), in the municipality of Góis, central Portugal (Figure 14a). The catchment burned on August 27th 2008. Burn severity was classified through the description of several indicators along a transect from the bottom to the top of the studied slopes. These indicators included: tree canopy consumption of 10 randomly selected trees (partial vs. total), degree of the litter consumption (partial vs. total), and ash colour (black vs. grey), which were compared to Hungerford (1996) and DeBano et al. (1998) classification methodologies. The selected indicators show that there was a moderate burn severity, which means a total consumption of the litter layer, charred or consumed duff, charred bigger woody debris, and unchanged mineral soil structure. The diameter of the 3 thinnest remaining twigs for each described tree was also used to calculate the ‘Twig Measurement Index’ (TMS). TMS can vary between 0 for unburned, to 1 for severe burn. The result agreed with the previous classification of moderate, resulting in a value of 0.5.

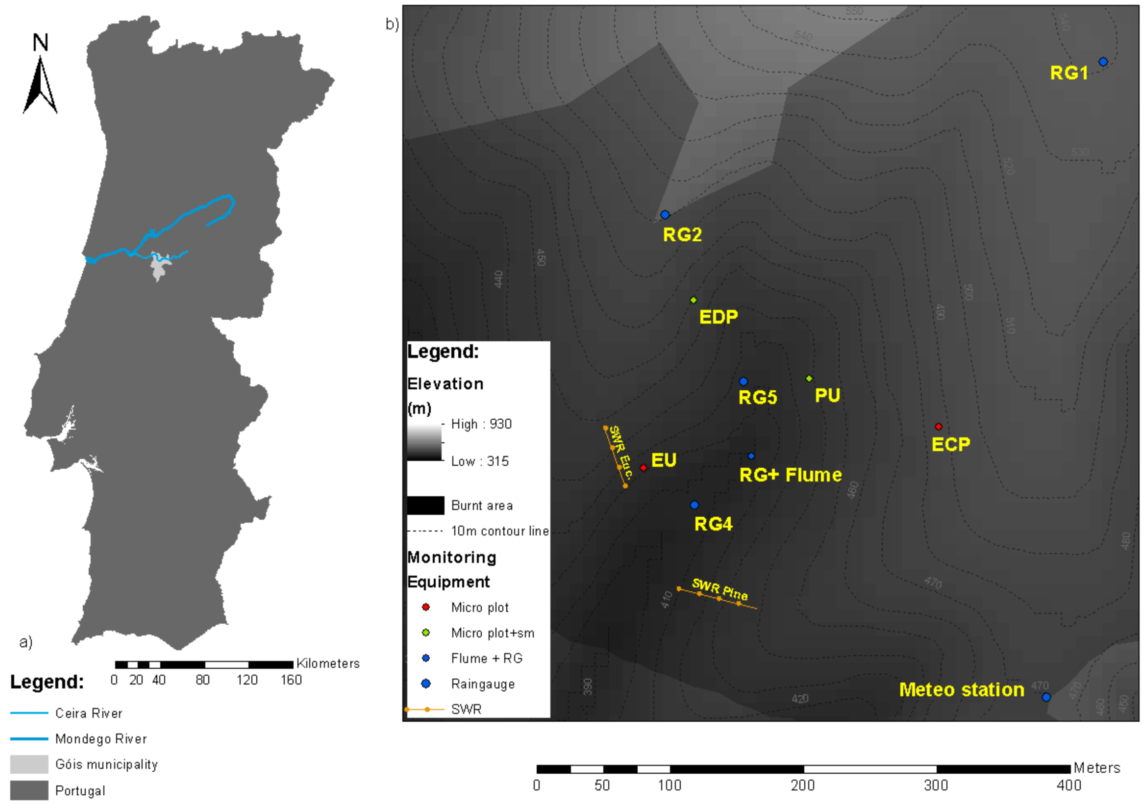


Figure 14 –Study area and sites: (a) location; (b) detailed topographic map of the burned catchment with specific locations of each study site and equipment. Note in (b): burned area in dark grey, red dots for micro plot locations, green dots for micro plots with soil moisture sensors; blue dots for rainfall gauges locations and other associated equipment (labelled); orange lines for soil water repellency (SWR) transects.

The climate of the study area is characterized as humid meso-thermal, with a prolonged dry and warm summer. The mean annual temperature is around 12 oC, whereas the mean annual precipitation amounts to about 1140 mm at the nearest climate station of Góis (SNIRH, 2012).

The geology of the area is composed of pre-Ordovician schists and greywackes (Ferreira, 1978; Pimentel, 1994), which have given rise to shallow soils that typically correspond to Humic Cambisoils (Cardoso et al., 1971, 1973).

The study site experienced a previous fire in 1990 (ICNF, 2014) and underwent several land management operations prior to the beginning of this study. Evidence of plowing operations immediately after the 1990 wildfire was found throughout the catchment. During the 18 years that separate these 2 wildfires, some eucalypt plantations were also continuously managed until the second fire, while other areas appeared to have been abandoned. This is evidenced by the eucalypt rotation cycle found at the beginning of this monitoring period. Some plantations were found to be in their 1st rotation cycle, implying another plowing during this interval to plant the new eucalypts, while other were found to be in their third rotation, and no other signs of management were found.

At the time of the fire, the burned area comprised 4 main land units, i.e. combinations of forest type - maritime pine and (predominantly) eucalypt plantations - and land management operations (unplowed, downslope plowed, and contour-plowed). Two months after the fire, some land management operations were implemented over the study area, but only a small portion of the studied catchment was affected by terracing, a new type of land management operation in this site. However, this study concerned only 4 types of land units: Eucalypt Unplowed (EU), Pine Unplowed (PU), Eucalypt Downslope Plowed (EDP), and Eucalypt Contour Plowed (ECP). For each of these land units, one study site was selected (Figure 14b). After that, the contour-plowed site was logged during the first four months after fire, while the other three study sites were not subjected to any intervention. Burn severity among sites was uniform and classified as moderate.

After the selection and monitoring of these four study sites, further soil characterization was performed through a destructive description of the plots (Table 4). This description allowed for an estimation of the timeline of events since 1990 for all the studied sites (Figure 15). This also allowed for verification of some inconsistencies in the initial study site classification. Although the EU site was initially classified as unplowed, as no ridges or furrows were observed at the soil surface, the profile description showed this location must have been plowed approximately 18 years ago.

Table 4 - General study site characteristics, standard deviation between brackets.

	PU	EU	EDP	ECP
Land cover	Pine	Eucalypt	Eucalypt	Eucalypt
Pre-fire land practices	Unplowed	Unplowed	Downslope plowing	Contour plowing
Plot angle (º) *(n=4)*	23 (2.9)	27 (5.5)	30 (4.5)	24 (6.6)
Downslope random Roughness *(n=4)*	0.71 (0.38)	1.12 (0.03)	1.53 (1.07)	2.00 (0.77)
O horizon	-	0-4	0-3	0-3
A horizon	0-7	A1 - 0-10 A2 – 10-22*	A1 - 0-14 A2 – 14-56*	A1 - 0-15 A2 – 15-29*
B horizon	-	-	-	29-36
C horizon	7-15	22-**60**	56-**60**	36-60
R horizon	**15**	-	-	60-**63**
Soil type classification (WRB, 2006)	Lithic leptosol	Haplic umbrisol & Umbric regosol	Humic cambisol & Haplic umbrisol	Humic cambisol & Haplic umbrisol
Soil texture class *(3-10 cm) (n=5)*	Sandy Loam			
Bulk density (g/cm3) *(3-10 cm) (n=8)*	1.00 (0.12)	0.83 (0.10)	1.05 (0.16)	0.85 (0.24)
Stone content (%) *(3-10 cm) (n=8)*	40 (11)	42 (6)	40 (11)	46 (12)

(*)- Presence of ashes in the profile.
Standard deviation in brackets.
Values in bold are the limit of the profile depth.

The fact that buried ashes were found in all A horizons of the eucalypt sites proved that the plowing occurred immediately after the 1990 wildfire. It was also concluded that EU and EDP were last plowed on this occasion, and that the depth of disturbed soil (depth of plowing) allowed assuming different plowing techniques were used between those sites (22 vs 56 cm, respectively). Although the EU classification was not entirely correct, the classification was kept the same because: i) the surface roughness was much smaller than other sites with different land management; ii) the depth of plowing was much smaller than the EDP site; iii) although it was plowed at the same time as EDP, 18 years was enough time to remove the effects of plowing, while in EDP the micro-topography variance was still visible.

The soil texture of the A horizon revealed a sandy loam texture in all sites, due to the elevated percentage of coarser material (sand >70%) in comparison to the fines (silt and clay). The stone content in the A horizon was approximately 40% in all plots.

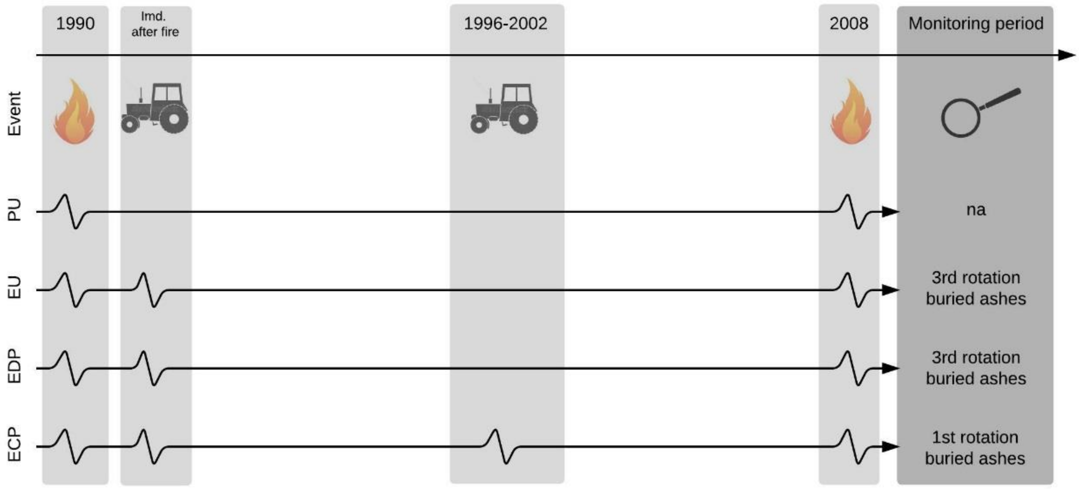


Figure 15 – History of events that took place in the study site before the monitoring period. Affected plots by those specific events and general evidences found during the monitoring period (rotation cycle o

Field experimental design and data collection

This study used a set of 16 experimental plots (0.25 and 0.50 m2) equally distributed between the 4 land units described above. In each land unit, 4 plots were installed (2 of each size) at the base of the slope, and their relative position was attributed randomly. In the downslope plowed eucalypt site however, 2 plots were installed on the ridges, and another 2 in the furrows (both sizes), to capture a representative response of the land unit considering such micro-topographic variation. These plots were installed by 25 September 2008, and then monitored for four years till 1 October 2012. It rained 27.7 mm between the time of the wildfire and plot installation (according to the nearest meteo station in Góis; SNIRH, 2012), however, no evidences of major overland flow response was found.

Monitoring involved field trips at 1-weekly (first two years), to 2 -weekly (third year), and monthly (fourth year) intervals, also depending on rainfall. These plots were connected to 30 or 70 L tanks, the runoff volumes were determined by measuring the runoff collected in those tanks, and erosion was determined by gathering a runoff sample with a minimum volume of 250 mL from each tank for sediment concentration analysis. Sediment concentrations in the runoff samples were determined in the laboratory by filtration with a 330 mm VWR filter and oven dried at 105ºC for 24 to 48 hours. For the entire period of study, a total of 1663 samples were processed.

The catchments was further instrumented with 4 tipping-bucket rainfall gauges (Pronamic Professional Rain Gauge, with 0.2 mm resolution) linked to an ONSET Hobo

Event Logger Automatic, as well as with 5 storage gauges for the purpose of validating the automatic data. The automatic rainfall data were used to compute maximum rainfall intensity over 30 min (I30, mm h^{-1}) as well as rainfall erosivity (R, MJ mm ha^{-1} $year^{-1}$) in accordance with the Universal Soil Loss Equation (Wischmeier and Smith, 1978) and using the rainfall kinetic energy equation of Coutinho and Tomás (1995), which is considered to be suitable for the western part of the Mediterranean Basin.

Soil moisture sensors (DECAGON EC-5) were installed by 30 October 2008 at two of the study sites, at PU (4 sensors) at the bottom of the catchment and at EDP (4 sensors). These data were recorded with DECAGON ECH20 data loggers with a 5 to 15 minutes time step at 3-5 cm depth.

Soil water repellence (SWR) was monitored on a monthly basis using the 'Molarity of Ethanol Droplet' test (MED) (King, 1981; Doerr, 1998), in a representative pine (SWR Pine) and eucalypt (SWR Eucalypt) stand inside the catchment (Figure 14b). SWR was measured at the edge of the catchment, and not near the plots, to minimize the disturbance of these measurements on the surrounding plots. The molarity ethanol drop test was slightly modified in accordance with findings from prior studies in this region (e.g., Keizer et al., 2005a, 2005b, 2008). In this study, three drops of pure water were applied to the soil surface, and if two of the three drops did not infiltrate within 5 seconds, then three drops with successively higher ethanol concentrations were applied until two of the three drops infiltrated within 5 seconds. The nine ethanol concentrations used were 0, 1, 3, 5, 8•5, 13, 18, 24, and 36%, corresponding to class 0 to 9 respectively (Santos et al 2013). Measurements were performed each month in each site over a five point transect, at the surface and at 5 cm depth (n=30). In the data analysis, the frequency of each class was calculated over the total measurements from that period, which was called SWR frequency. As a reference to these SWR measurements, average soil moisture from the week before was determined (antecedent soil moisture) through the previously described soil moisture sensor data.

Ground-level soil cover (GC,%) was also monitored in the plots at a monthly frequency, through pictures and descriptions over a 5 x 5 cm grid. GC data was obtained by determining the variables: rock; stones; vegetation, litter, ashes and carbon, and bare soil, for all the grid intersections inside the plot.

Soil profiling (n=16), downslope roughness (n=16), and surface soil sampling (n=20) was performed in each plot by end of the monitoring period to determine soil depth, soil texture, bulk density, and stone content of the surface soil (3-10cm) (Table 4).

Statistical analysis

Simple linear regression was used to examine the influence of a set of variables on annual runoff and erosion figures. Annual rainfall and rainfall erosivity (R), annual frequency of high and very high (class 6-9) SWR, and end-of-year ground cover variables (GC), percentage of litter + vegetation, stones + rocks, bare soil and ashes, were used as independent variables.

The effects of land use, land management, and time since fire on runoff and erosion response were assessed by 2-way repeated measures ANOVA against 2 groups of data. The first group consisted of unplowed sites, pine (PU), and eucalypt (EU) unplowed. The second group consisted of eucalypt sites, eucalypt unplowed (EU), eucalypt downslope plowed (EDP), and eucalypt contour plowed (ECP). The SAS 9.2 software package (SAS Institute, Inc., 2008) was used to carry out the data analysis. When either the overall effect or the interaction was significant, a simple main effects and post-hoc LS-Means adjusted Tukey's test were used to assess site variability on runoff and erosion values within years and vice versa (Littel et al., 2006). The variance-covariance structure for the dependent variables was selected as first-order auto-regressive, according to the smallest -2 restricted log likelihood (Littel et al., 2006). The significance level for the statistical tests was 0.01. The original values of runoff (mm), and runoff coefficient (%) did not violate the assumption of normality of the residuals, but the soil loss ($g\ m^{-2}$) and specific soil loss ($g\ m^{-2}\ mm^{-1}$ runoff) did, and were therefore square root and four root transformed, respectively.

Results

Ground cover

By December 2008 no appreciable ground level vegetation was observed in any of the study sites, despite 4 months having passed since the wildfire (Figure 16). At the same time, a clear difference was found in mean litter cover between the different land uses (70% pine vs 35% eucalypt), and between unplowed and plowed sites (35% unplowed vs 20% plowed). Marked differences in stone cover were also found between land uses (7% pine vs 50% eucalypt), while between different land management types the cover was approximately the same (50%). Both unplowed sites had an average bare soil cover under 10%, while the cover in the plowed sites was slightly higher (15%).

During the subsequent years, the pine site had a steady increase in average vegetation cover up to 40%, while the eucalypt site recovery never reached 10% (Figure 16a). Although vegetation cover in the pine site increased steadily, the litter cover also declined by approximately the same amount (37%) (Figure 16b). In the eucalypt site however, litter cover slightly increased during the same period. While the pine site had higher vegetation and litter cover during the monitoring period, the eucalypt site had much higher stone cover. Mean stone cover in the eucalypt site was approximately 68% by the end of the first year and 50% during the remaining years. These values were much higher than the pine site stone cover, which had a maximum of 23% (Figure 16c). The mean coverage of bare soil was very low in both locations, and decreased during the study period down to 2% in both sites (Figure 16d).

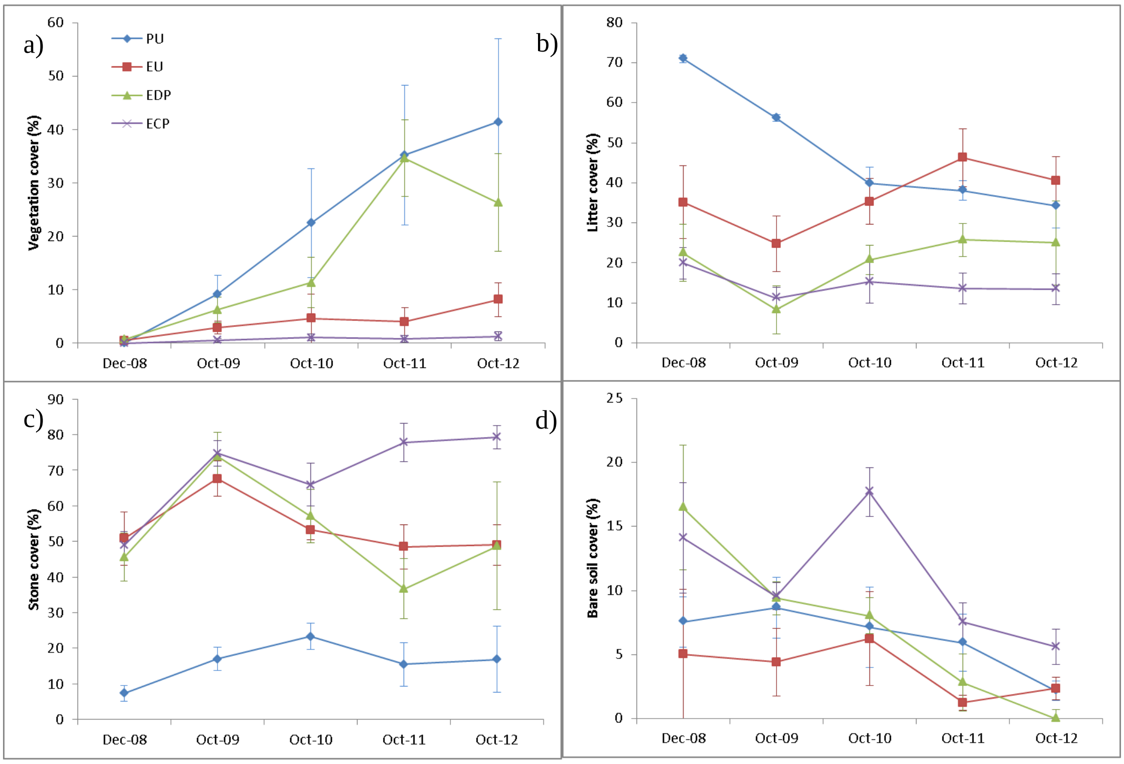


Figure 16 - Average ground cover for (a) vegetation, (b) litter, (c) stones and (d) bare soil for each studied site (n=4). Standard error indicated by error bars.

Mean vegetation cover showed variability between sites with different land management. While the contour-plowed site had almost no vegetation recovery, even less than unplowed site, the downslope plowed site had increasing vegetation cover, until October 2011 (Figure 16a At this time, average vegetation cover reached its maximum of 35%. During the four years, the plowed sites always had less litter cover than the unplowed eucalypt site (Figure 16b). Both plowed locations had a similar mean litter cover until October 2010, after which EDP stabilized at 25% and ECP at 13%. After October 2010, the mean stone cover also showed some variation between the eucalypt sites (Figure 16c). The contour-plowed site had an increase in mean stone cover up to 79%, while unplowed and downslope plowed sites had a decrease to 50%. Bare soil cover decreased from the first to the fourth monitoring year in all eucalypt sites (Figure 16d). The plowed sites had higher mean bare soil cover than the unplowed site, and the maximum value (18%) was in the ECP site in October 2010.

By December 2008, ash cover was approximately 10% at the unplowed sites and 15% at the plowed sites, and had almost entirely disappeared by the end of the first year. Rock outcrop was only observed at the pine site where the soils were most shallow.

Annual rainfall and soil water repellency

Annual rainfall amounts varied substantially during the first four hydrological years after the wildfire (Table 5). When compared to the long-term mean annual rainfall of 1143 mm at the nearest climate station of Góis (SNIRH, 2012), the first year was average (1095 mm), while the second and third year were relatively wet (+13 and +15 %). By contrast, the fourth year was markedly dry, with 34 % less rainfall. The above average rainfall observed in the second and third year were due to higher than average rainfall amounts in autumn. The second and third year autumn rainfall corresponded approximately to 2 and 3 times the amount which occurred in the first year during the same season. In the fourth year however, the low annual amount was due to an almost complete lack of winter rainfall. Annual rainfall erosivity closely followed these annual rainfall patterns, whereas the third and the fourth year were represented by the maximum and the minimum rainfall erosivity value, respectively.

During the first year, both the pine and eucalypt sites showed a higher frequency of high and severe soil water repellency ($SWR_{pine\ (6\text{-}9)}$=54% and $SWR_{euc.(6\text{-}9)}$=52%) in comparison to the remaining years (Table 5). From the second year onwards, the eucalypt site had an increase in repellent conditions until the fourth year. By contrast the pine site stabilized at 29% from the 3rd to the 4th year.

Table 5 – Annual figures regarding rainfall amounts (mm) and rainfall erosivity (MJ mm ha^{-1} year^{-1} h^{-1}) for the entire study area, and Soil Water Repellency frequency (%) from pine and eucalypt site.

		year 1	year 2	year 3	year 4
Rainfall (mm)		1095	1295	1534	833
Rainfall erosivity (RE) (MJ mm ha^{-1} year^{-1} h^{-1})		2168	3938	5161	1679
SWR freq. $_{class\ 6\text{-}9}$ (%) *(n=360)*	pine	54	24	29	29
	eucalypt	53	27	33	35

Overall and annual overland flow amounts

Runoff amounts at all four study sites for the period 2008-2012 are presented in Table 6. These totals show the marked difference between land use and pre-fire land management practices. This can be observed at pine site (PU), which had 77% more runoff than the eucalypt site (EU), and the results of the variance analysis among unplowed sites (Table 7) show that annual runoff amounts and coefficients differ significantly between pine and eucalypt. In the comparison between land management (Table 6), the contour-plowed site (ECP) had 52% more runoff than the downslope plowed eucalypt site (EDP), and the unplowed eucalypt site (EU) generated substantially less overland flow in comparison to the plowed sites (55% and 31% less, than ECP and EDP respectively). This was equally suggested by the ANOVA analysis among eucalypt sites with different land managements. Indicating that land management was an important factor justifying runoff amounts and runoff coefficients differences (Table 8).

Table 6 – Total rainfall (mm), runoff (mm), runoff coefficient (%), mean sediment losses ($Mg.ha^{-1}$ $year^{-1}$), Total sediment losses ($Mg\ ha^{-1}$ $4years^{-1}$) and mean sediment concentration in the runoff ($g.m^{-2}.mm_{Runoff}^{-1}$), for the entire study period (bottom table).

sites	Total rainfall (mm)	Total runoff (mm)	Runoff coef. (%)	Mean sed. losses ($Mg\ ha^{-1}\ year^{-1}$)	Total sed. losses ($Mg\ ha^{-1}\ 4years^{-1}$)	Mean Sed. conc. ($g\ m^{-2}\ mm_{Runoff}^{-1}$)
PU	4756	1532	32	0.43	1.72	0.47
EU		865	18	0.31	1.26	0.49
EDP		1257	26	0.52	2.09	0.87
ECP		1915	40	0.98	3.94	0.89

Contrary to the original hypothesis, annual overland flow did not reveal a marked decrease with time-since-fire. In fact, runoff amounts increased from the first to the third year, and then decreased until the fourth year. Exhibiting similar pattern to rainfall (Figure 17a, Figure 18a.), at unplowed pine (PU), unplowed eucalypt (EU), and at contour-plowed eucalypt site (ECP). The downslope plowed site (EDP) had a constant increase in runoff amount from the first to the fourth year, the runoff coefficients however showed another pattern (Figure 17b, Figure 18b). The maximum observed runoff coefficient in the pine site (PU) occurred during the first year. But after dropping to the minimum value in the second year, there was a constant increase in runoff coefficient from this point until the fourth year. At the same time, runoff coefficient in all eucalypt sites (EU, ECP, EDP) constantly increased from the first year until the last. Between the unplowed sites, the variance analysis suggested that neither time-since-fire, nor time combined with land use characteristics influence overland flow generation at pine and unplowed eucalypt. Among the eucalypt sites however, time-since-fire has a strong influence on runoff amounts and especially in the runoff coefficient figures, as observed by their different F value (12.2 vs 18.6). This is easily observed in the runoff totals (Figure 18a) which show a similar time-since-fire effect (increase) among all the sites until the third year, while in runoff coefficients (Figure 18b) this occurs for all sites during the 4 years.

Overall and annual erosion rates

Total erosion amounts show higher sediment losses at the unplowed pine site (37% more) than the unplowed eucalypt site (Table 6). The observed difference between land uses are possibly caused by the substantially higher overland flow generated in pine site (+77%) when compared to the eucalypt site, since mean sediment concentration in runoff for both locations are similar. This is also shown by the comparative analysis of annual erosion figures at both unplowed sites through ANOVA (Table 7), indicating that land use was not a major factor controlling erosion amounts (F=2.09), and was even less of a factor in sediment concentrations in runoff (F=0.03). Overall erosion amounts at the eucalypt sites, show a production of 89% more erosion at the contour plowed site (ECP) when compared to the downslope plowed one (EDP), which in turn produced 66% more erosion than the eucalypt unplowed site (EU) (Table 7). In this case however, the mean sediment concentration in runoff was almost double for plowed sites when compared to the unplowed one. Moreover, land management characteristics seemed to be the dominant factor explaining significant differences in erosion amounts and sediment concentration in runoff results among the eucalypt sites (Table 8). The effect of management, however, was more noticeable in erosion amounts (F=9.85) than in sediment concentrations (F=5.16).

The annual erosion figures among unplowed sites for both pine and eucalypt sites had their maximum erosion amounts during the third year (62 vs 54 g m^{-2} y^{-1}, respectively) (Figure 17c). However, contrary to the eucalypt site, the erosion pattern at the pine site was not always corresponding to rainfall amounts. In this site, a decrease in mean erosion amounts (-22%) occurred from the first year to the second, during a 13% rainfall increase. Consequently, no significant differences were found between years regarding annual erosion amounts. Between eucalypt sites with different managements, both unplowed (EU) and downslope plowed (EDP) sites had erosion patterns similar to the rainfall pattern, with an increase from the first to the third year after fire (maximum of 54 and 82 g m^{-2} y^{-1}, respectively), followed by a decrease until the fourth year (Figure

18c). The contour plowed eucalypt site (ECP) behaved differently, reaching its maximum in the second year (137 g m^{-2} y^{-1}), and had a decrease only from the third (132 g m^{-2} y^{-1}) to the fourth year (71 g m^{-2} y^{-1}). Despite the different temporal patterns among these sites, time since fire seemed to influence annual erosion amounts between eucalypt sites (F=2.97).

Annual sediment concentration shows a clear decrease from the first to the fourth year in all the study sites (Figure 17d, Figure 18d). These sediment concentration reductions were approximately 53% at the unplowed pine site (PU), and 65% in the unplowed eucalypt site (EU), indicating a possible recovery pattern. This was also shown by the variance analysis, whereas time-since-fire factor seemed to perform an important role between annual sediment concentrations in runoff (F=8.08), and even combined with land use factor, but only with a minor magnitude (F=4.09). Between different land management practices, the highest decrease was observed within the downslope plowed site (EDP) with a 79% reduction, and the least at the contour-plowed eucalypt site (ECP) at 49%. Again, time-since-fire seemed to be an important factor explaining differences in annual sediment concentration in runoff results (Table 8).

Table 7 - 2-way repeated measures ANOVA results (F-value) with plot-wise annual values by site (n=4), to determine land use (pine, eucalypt) and time since fire (4 years; N=32) effect over the studied variables. The underlined and bold F values are statistically significant at α ≤ 0.05.

Source of variation	D.f	Runoff (mm)	Runoff coefficient (%)	Sed. rate (g m^{-2})	Sed. Conc. runoff (g m^{-2} mm_{Runoff}^{-1})
Land use	1,24	<u>**8.68**</u>	<u>**9.10**</u>	3.34	0.03
Time-since-fire	3,24	2.09	1.38	2.13	<u>**8.08**</u>
Land use*time	3,24	1.77	1.83	0.55	<u>**4.09**</u>

Df, degrees of freedom; num, numerator; den, denominator.

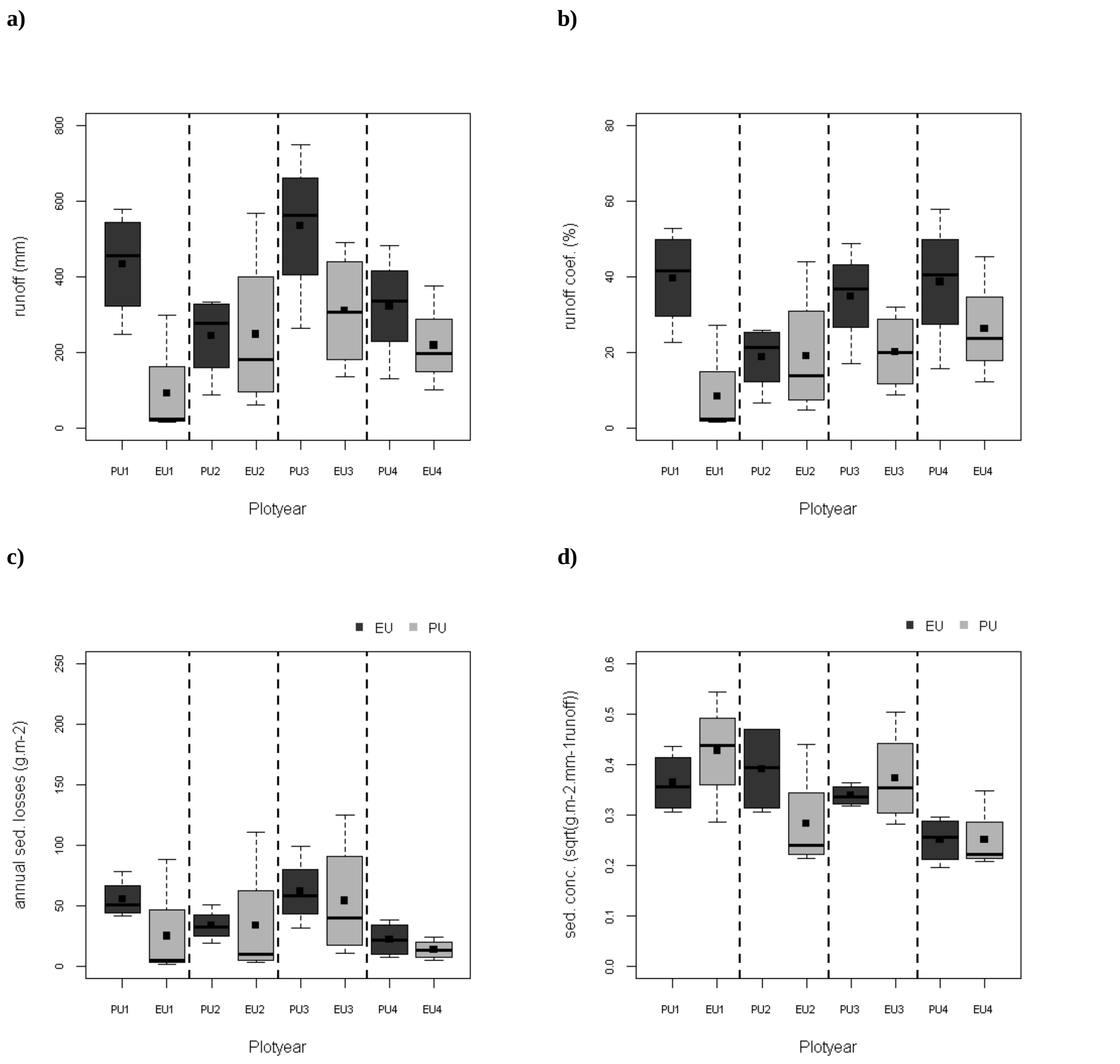


Figure 17 – Mean annual runoff and erosion of 4 years following the wildfire for PU and EU: (a) runoff (mm), (b) runoff coefficient (%), (c) sediment losses (Mg ha^{-1} y^{-1}) and (d) sediment concentration in the runoff (g m^{-2} mm_{Runoff}^{-1}). Note maximum and minimums in the error bars, boxplot line represent median value, filled squares mean values.

Table 8 - 2-way repeated measures ANOVA results (F-value) with plot-wise annual values by site (n=4), to determine land management (unplowed, contour plowed, down-slope plowed) and time since fire (4 years; N= 48) effect over the studied variables. The underlined and bold F values are statistically significant at α ≤ 0.05.

Source of variation	D.f	Runoff (mm)	Runoff coefficient (%)	Sed. rate (g m^{-2})	Sed. Conc. runoff (g m^{-2} mm_{Runoff}^{-1})
Management	2,36	**14.52**	**15.65**	**9.85**	**5.16**
Time-since-fire	3,36	**12.24**	**18.64**	**2.97**	**8.64**
Management*time	6,36	1.4	1.75	0.45	0.81

Df, degrees of freedom; num, numerator; den, denominator.

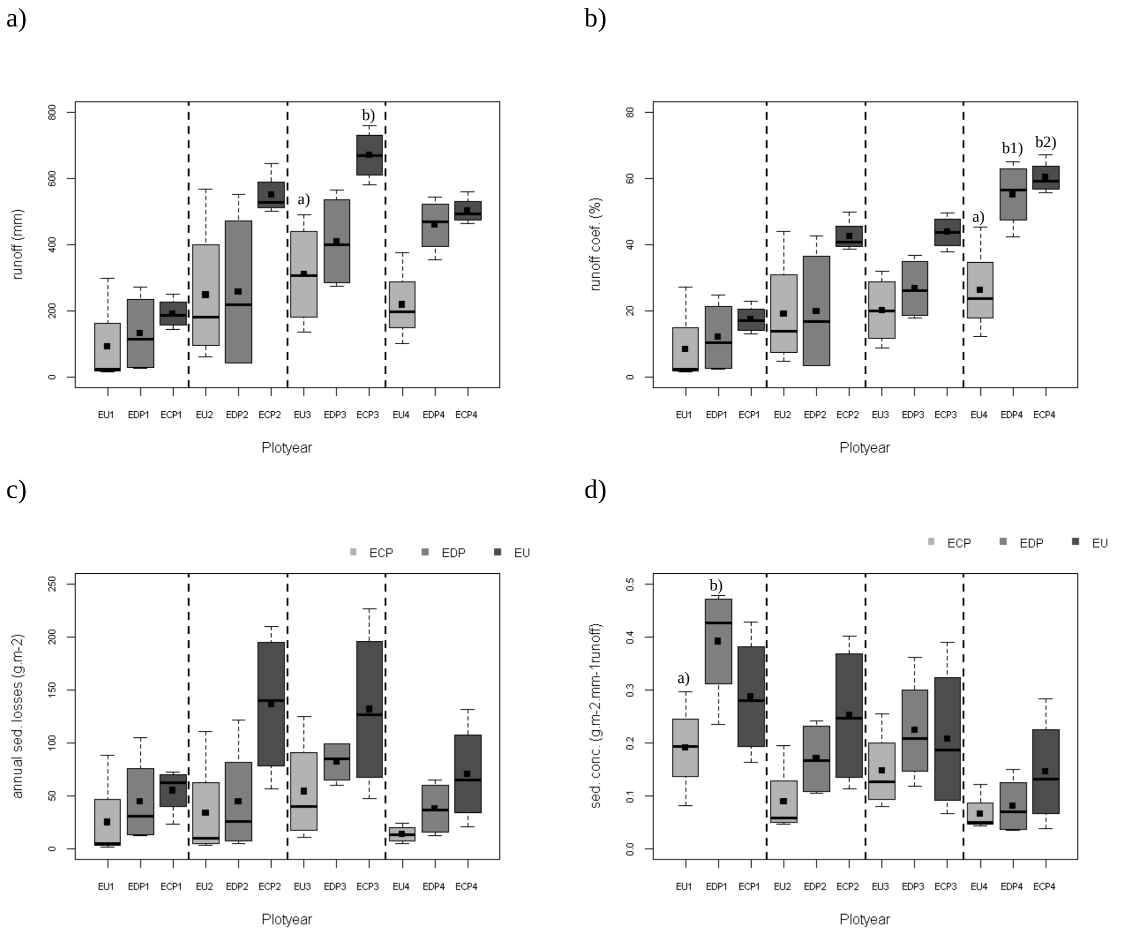


Figure 18 –Mean annual runoff and erosion of 4 years following the wildfire for EU, EDP and ECP: a) runoff (mm), b) runoff coefficient (%), c) sediment losses (Mg ha^{-1} y^{-1}) and d) sediment concentration in the runoff (g m^{-2}.mm_{Runoff}^{-1}). Note maximum and minimums in the error bars, boxplot line represent median value, filled squares mean values. Different letters within a post-fire year represent least mean squares significances (p<0.05) on runoff/erosion response between sites for that year.

Potential explanatory variables

The correlations between post-fire annual runoff and erosion with a set of independent variables are depicted in **Table 9**. Among the unplowed sites, only erosion response was significantly related with any of the independent variables. For both pine and eucalypt sites, bare soil cover was positively related with annual erosion amounts. This relationship was much stronger in the pine than in the eucalypt site. In the case of the pine site only, bare soil was followed by the variables litter+vegetation and rainfall amounts. In the case of the land managed eucalypt sites (EDP and ECP), a correlation was found with runoff and erosive response. Nevertheless, some relationships were contradictory to the expected (theoretical) relationship. This was the case with the litter+vegetation variable, which had a positive relationship with runoff amounts, coefficients, and sediment concentration at EDP, and SWR which had a negative relationship with runoff and runoff coefficient at the ECP site.

The contour plowed site was the only location with annual runoff and erosion amounts that could be partially explained by the rainfall pattern.

Table 9 – Linear model correlations between site variables (n=16). For each site dependent variables: annual runoff (runoff, mm), annual runoff coefficient (runoff coef., %) and annual sediment concentration in runoff (sed. Conc., g m^{-2} mm_{Runoff}^{-1}); were correlated with the independent variables: end-of-year ground cover (litter + vegetation, bare soil, stones, %), annual rainfall (mm), annual rainfall erosivity (R , MJ mm ha^{-1} $year^{-1}$) and soil water repellency frequency (SWR, %). Only linear models with $p<0.05$ are presented.

Site	dependent variables	independent variables	sign	s.e.	R^2	Sig
PU	erosion	litter + vegetation	-	0.27	0.31	*
		bare soil	+	0.93	**0.51**	**
		rainfall	+	0.02	0.25	*
EU	erosion	bare soil	+	2.03	0.26	*
EDP	runoff	litter + vegetation	+	1.59	**0.44**	**
		stones	-	1.99	0.36	*
	runoff coef.	litter + vegetation	+	0.17	0.37	*
		bare soil	-	1.02	0.26	*
		stones	-	0.22	0.28	*
	sed.conc.	litter + vegetation	+	0.00	0.37	*
		bare soil	+	0.01	0.28	*
		stones	+	0.00	0.34	*
ECP	runoff	R	+	0.03	**0.44**	**
		SWR	-	2.73	**0.73**	***
	runoff coef.	SWR	-	0.32	**0.51**	**
	erosion	R	+	0.01	0.26	*

*p<0.05
**p<0.01
***p<0.001

Discussion

The role of land use and land management practices

Previous studies in north-central Portugal have examined the importance of land-use on hydrological and erosive response after wildfire, by comparing post-fire runoff and erosion at pine and eucalypt sites (Prats et al., 2012; Martins et al., 2013). In the beginning of these studies (first 3 months in Prats et al. (2012); first 7 months in Martins et al. (2013)), few differences were found between pine and eucalypt responses to the same rainfall events. Only later, and during one year, Prats et al. (2012) found substantially higher runoff and erosion amounts at the eucalypt site (466 mm and 5.41 Mg ha^{-1}) when compared to the pine site (93 mm and 0.32 Mg ha^{-1}). However, the bigger plot size in Prats et al (2012), and the higher burn severity in the eucalypt site than in the pine site may limit the comparison with the present study. Nevertheless, both studies contrast with the findings of the current study, where the pine site had substantially higher runoff (1532 vs 865 mm) and erosion amounts (1.72 vs 1.26 Mg ha^{-1} $4years^{-1}$) than eucalypt site.

Land use was found in this study to have a significant influence over runoff variability. However, the differences between the studied unplowed pine and eucalypt sites might not be restricted to the type of land cover found in these locations. The higher runoff amounts observed in the unplowed pine site may be due to the low water storage capacity of this site, as a consequence of the shallow soil depth. In addition, the unplowed eucalypt site can promote higher infiltration than the pine site due to the higher surface roughness (Burwell and Larson, 1969) and elevated stone cover (Zavala et al., 2010). The low runoff amounts in the eucalypt site, combined with the elevated stone cover that protects soil particle detachment (Shakesby, 2011) in this location, possibly led to a smaller transport capacity from runoff and, by consequence, less erosion amounts in comparison to the pine site.

The occurrence of high SWR conditions presented similar frequencies between land uses at annual scale. However, several other studies contradict these observations, which show substantial differences among burned (Prats, et al 2012) and unburned (Santos et al., 2013) pine and eucalypt stands at smaller temporal scales (weekly to monthly intervals).

Little research has been conducted which assesses the influence of pre-fire management practices over runoff and erosion in burned areas (Malvar et al., 2011, 2013, 2015). These authors found that either by rainfall simulation campaigns or under natural rainfall, pre-fire plowed sites showed less runoff and erosion when compared to unplowed sites. The results of the present study suggest that pre-fire land management practices, either by plowing or by the absence of soil interventions, strongly influences post fire runoff and erosion response. But in this study, plowing led to an increase of runoff and erosion when compared to the unplowed site, contrary to the observations of Malvar et al. (2015) and conflicting with the objective of plowing (i.e. soil conservation).

When making these comparisons, the time-since-plowing and the degree of impact of the plowing technique variables should also be considered. It is known that after plowing an increase of runoff and erosion rates is usually observed (Shakesby et al., 1994, Walsh et al., 1995). The present study however used as comparison several sites at different stages of post-plowing recovery. Sediment exhaustion in the unplowed eucalypt site due to prior disturbances (plowed 18 years before, impacts only visible until first 22 cm soil depth) might explain the extremely low hydrologic and erosive response. The downslope plowed site on the other hand showed intermediate values of runoff and erosion amounts, between the unplowed and the contour-plowed (more recently plowed) sites, due to a deeper-in-soil intervention (evidences of disturbances until 56cm soil depth).

The results of this study also contrast with the assumptions that contour plowing techniques are a good management practice for preventing soil losses, especially when compared to downslope plowing (Morgan, 2005), which is widely known for is enhanced impact on runoff and erosion (Shakesby et al., 1996, 2002). The scale of this study however, might limit this impact due to the lack of representativeness of the plowing effect at such small scale (micro-plot). Whereas in this study the downslope plowed results originated from two plots on the ridges and two in the furrows, and very likely did not capture the entire micro-topographic effect. Nevertheless, this scale might be suitable for determining plowing impact on soils and sediment availability for erosion.

The role of time-since fire and its implications for Window of disturbance models

Several researchers (Shakesby and Doerr, 2006, Shakesby, 2011) have revealed that the first post-fire rainstorms usually cause enhanced overland flow and erosion. The reasons for this are usually attributed to the vulnerable soil conditions caused by fire, due to the reduced infiltration capacity together with a lack of vegetation and litter cover. After that period, there is a decrease in the hydrological response corresponding with vegetation recovery (Prosser and Williams, 1998). In this sense, time-since-fire was expected to be an important factor, representing post-fire recovery through a decrease of erosion with time. In fact, the analysis of variance showed that time-since-fire was an important factor controlling sediment concentration at unplowed sites, and also for runoff and erosion at eucalypt sites. Annual runoff coefficients and sediment concentration in both studied cases (unplowed and eucalypt) show a clear change with time. However, they show opposite patterns, runoff coefficient increased with time-since-fire, while sediment concentration shows a decrease during the same period.

The rainfall increase from the first to the third year of study can explain the runoff coefficient increase until the third year. However, the reduction of rainfall amounts in the last year was contradicted by another increase in runoff coefficients in all sites. Some previous studies have indicated that similar rainfall events may produce different hydrological responses within wet and dry soil moisture conditions (Ferreira et al., 2000, Keizer et al., 2005b). Runoff generation might occur either by saturation excess overland flow in the case of wet conditions, or by infiltration excess in the case of dry conditions due to limited infiltration capacity, often attributed to SWR (Walsh et al., 1994). This might explain the elevated runoff coefficients verified in the fourth year when compared to the remaining years. Because the lack of rainfall in the wet season (winter) led to an increase in the contribution of runoff generation by infiltration excess (typically observed at autumn and spring) in comparison to the contribution by saturation overland flow, which may have increased the runoff coefficients. Unfortunately, the annual SWR pattern does not provide enough detail about the formation of high or severe repellent conditions among dry periods.

Soil protection by vegetation and litter (Prats et al, 2014) or stones (Shakesby and Doerr, 2006) might play a significant role in erosive processes. However, vegetation growth in this area seems to have had a slow recovery within the four years of study, when compared to other burned areas for the same period (Wittenberg and Inbar, 2009), or even less time (Fernández and Vega, 2014). Similar results (vegetation<50%) were obtained by Mayor et al. (2007) within a similar period after fire at E Spain, in a dry–subhumid Mediterranean climate. Which was attributed to the presence of several dry periods that delayed plant regrowth and increased the length of the critical period for elevated post-fire erosion risk in the area. Additionally, the elevated stone cover that was present since the beginning of this study, together with the reduced bare soil cover, might be an important factor explaining low post-fire soil losses. This is primarily due to the protective effect of stones for erosion (Shakesby, 2011), and secondly because bare soil cover in all plots never reached critical levels. Bare soil cover was always under 30%, which according to MacDonald and Larsen (2009) is the limit of bare soil in which the sediment yields drop to near background levels. These explanatory variables, however, did not contribute in the same way for all the study sites, as shown by the linear model correlations. But its simplification into annual values might underestimate its importance. In this case, even a slight protection increase from vegetation, litter, or stones may have substantially reduced sediment losses, as indicated by the sediment concentration in runoff reduction.

Past interventions and fire recurrence in this location may also have limited the recovery of this study area. Fire recurrence might increase “baseflow” sediment yield during the window of disturbance (Wittenberg and Inbar, 2009), and soil fertility might be compromised due to management practices (Shakesby, 2011). This could arguably be considered as evidence that this long-term analysis may not have been sufficiently long to observe the hydrological response decrease, until background levels as described in the window of disturbance model (Prosser and Williams, 1998). Or that past disturbances (wildfires and others) have effectively enlarged the window of disturbance in this specific place, and a model such as Wittenberg and Inbar (2009) might be more realistic. The sediment concentrations in the runoff pattern might be comparable to the above cited window of disturbance behaviour, indicating a possible recovery of the site. However, this decrease might also be a reflection of sediment exhaustion in the bounded plots, as observed in Malvar et al. (2015) study.

As highlighted by Shakesby (2011), few studies have reported post-fire runoff and erosion rates for periods longer than 2 years for plot scale within the Mediterranean. Only 3 post-fire erosion studies were found which had 4 years of monitoring at the plot scale under natural rainfall conditions (Lavabre and Martin, 1997; Butorac, 2009; Wittenberg and Inbar, 2009) (Figure 19). However, the plot size used in these studies were considerably larger (75-200 m2) than those used in the current study (0.25-0.5 m2). Nevertheless, all of these studies showed a decrease in erosion rates from the first to the fourth year, this decline occurred in the second year for Butorac (2009) and Wittenberg and Inbar (2009), while Lavabre and Martin (1997) only presented evidence of recovery in the fourth year. Although the erosion records in the unplowed sites (PU and EU) are lower than the Lavabre and Martin (1997) (Figure 19) case and always inferior to the tolerable rate of soil losses (Verheijen et al., 2009), the inter-annual response is similar.

Post-fire erosion rates in micro-plots in Portugal were also compared with a two year (Malvar et al., 2015) and three year (Prats et al, 2013) study. Results show that soil losses of this specific study are much lower when compared to others in North-Central Portugal at the same spatial scale.

Soil losses in sites subjected to pre-fire (Malvar et al,. 2015) and post-fire (Martins et al., 2013) land management practices at the micro-plot scale were also included in this comparison (Figure 19). Results from Malvar et al., (2015) go against the findings of the present study. Pre-fire plowing not only resulted in less erosion than the unplowed plots, it also decreased the rates in the plowed site, while at the same time the erosion rates in the unplowed site increased. Post-fire terracing, as studied by Martins et al. (2013), show an extreme increase of the erosion rates in a small period of time for eucalypt and pine sites.

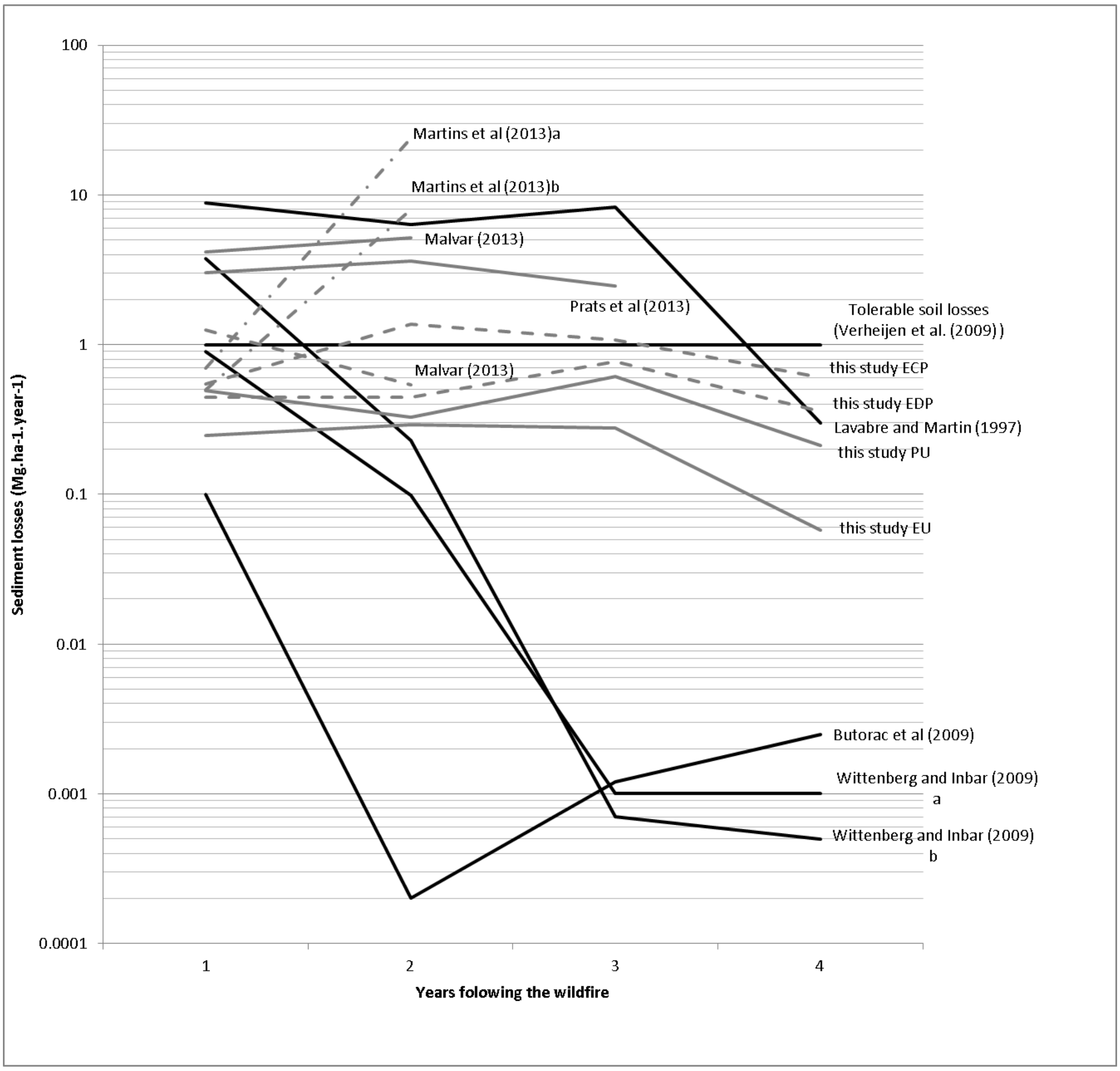


Figure 19 – Comparison of soil losses after wildfire with: (a) Long-term field studies (black lines); (b) Micro-plot scale measurements in Portugal (grey lines); (c) pre-fire plowing (dash line-); (d) post-fire plowing (dash-dot line).

Looking into the significance of the observed erosion rates, considering all the changes that these sites have been subjected to, before the most recent fire and other changes that we are unaware of prior to the 1990's fire. The sustainability of these soils seems to be compromised, and most likely this system will never recover back to the background runoff and erosion levels. It's true that sediment losses in the unplowed sites were always under the tolerable soil loss rate (Verheijen et al., 2009), but these very shallows soils, especially at the pine site with 7cm maximum soil depth, there is very little soil remaining to erode. In the case of the plowed sites, this limit of tolerable soil losses was reached, indicating higher sediment availability for transport. Nevertheless the most recent plowing seems to still have an arguably more significant impact after the wildfire.

Conclusions

The main conclusions of this study regarding overland flow and interrill erosion during four years after a wildfire in contrasting forest plantations are:

- Land use differences might be responsible for runoff generation variability, as the pine site consistently produced higher runoff amounts and coefficients than the eucalypt site.
- Different plowing techniques and time since implementation may be an important factor for both runoff and erosive processes. The plowed sites showed higher runoff and erosive response than the unplowed site. However, this comparison between plowed and unplowed sites might be influenced by the highly degraded soil conditions that the unplowed soils have already been subjected to, as well as the time since plowing and the type of technique used.
- The sediment concentration results highlighted a possible decrease in post-fire erosion response among all study sites. However, annual runoff, runoff coefficients, and erosion amounts do not seem to be attenuated with time since fire, most likely due to the reduced ground vegetation recovery. Thus it wasn’t possible to observe similar behaviour to the classic window of disturbance model; or the window of disturbance was possibly enlarged due to past disturbances in this specific location.
- The recurrence of fires, together with several forest interventions in this location, might compromise the sustainability of these soils. And due to that, even the low erosion rates found in this study represent a threat to such degraded soils.

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References

Benavides-Solorio, J., MacDonald, L.H., 2001. Post-fire runoff and erosion from simulated rainfall on small plots, Colorado Front Range. Hydrological Processes 15, 2931–2952.

Burwell, R.E., Larson, W.E., 1969. Infiltration as influenced by tillage-induced random roughness and pore space. Soil Sci. Soc. Am. Proc., 33: 449-452.

Butorac, L., Topic, V., Jelic, G., 2009. Surface runoff and soil loss in burnt stands of Aleppo pine (Pinus halepensis Mill.) growing on colluvial soils. Sumarski List 133, 165–174 (English summary).

Cardoso, J.C., Bessa, M.T., Marado, M.B., 1971. Carta dos solos de Portugal (1:1.000.000). Serviço de Reconhecimento e de Ordenamento Agrário, Secretaria de Estado da Agricultura, Lisboa, Portugal.

Cardoso, J.C., Bessa, M.T., Marado, M.B., 1973. Carta dos solos de Portugal (1:1.000.000). Agron Lusit 33: 461–602.

Carvalho, A., Flannigan, M., Logan, K., Gowman, L., Miranda, A.I., Borrego, C., 2010. The impact of spatial resolution on area burned and fire occurrence projections in Portugal under climate change. Climatic Change 98: 177–197

Cerdà, A., 2001. Erosión Hidrica del Suelo en Territorio Valenciano. El Estado de la Cuestión a través de la Revisión Bibliográfica. Geoforma Ediciones, Logroño.

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, F.J.P.M., Raclot, D., Ionita, I., Rejman, J., Rousseau, S., Muxart, T., Roxo, M.J., Dostal, T., 2010. Rates and spatial variations of soil erosion in Europe: a study based on erosion plot data. Geomorhology 122, 167–177.

Coelho, C.O.A., Ferreira, A.J.D., Boulet, A.K., Keizer, J.J., 2004. Overland flow generation processes, erosion yields and solute loss following different intensity fires. Quarterly Journal of Engineering Geology and Hydrogeology 37, 233–240.

Coutinho, M.A., Tomás, P., 1995. Characterisation of raindrop size distributions at the Vale Formoso Experimental Erosion Centre. Catena 25: 187–197.

DeBano, L.F., Neary, D.G., Ffolliott, P.F., 1998. Fire's Effects on Ecosystems. Wiley, New York.

Doerr, S.H., 1998. On standardizing the "Water Drop Penetration Time" and the "Molarity Ethanol Droplet" techniques to classify soil hidrophobicity: a case of study using medium textured soils. Earth Surface Processes and Landforms 23: 663–668.

Dunjó, G., Pardini, G., Gispert, M., 2004. The role of land use-land cover on runoff generation and sediment yield at a microplot scale, in a small Mediterranean catchment. Journal of Arid Environments 57, 99–116.

Emmerich, W.E., Cox, J.R., 1992. Hydrologic characteristics immediately after seasonal burning on introduced and native grasslands. Journal of Range Management. 45: 476–479.

Fernández, C., Vega, J.A., Gras, J.M., Fonturbel, T., Cuiñas, P., Dambrine, E., Alonso, M., 2004. Soil erosion after Eucalyptus globulus clearcutting: differences between logging slash disposal treatments. Forest Ecology and Management 195: 85–95. doi:10.1016/J.FORECO.2004.02.052

Fernández, C., Vega, J.A., Fonturbel, T., Pérez-Gorostiaga, P., Jiménez, E., Madrigal, J., 2007. Effects of wildfire, salvage logging and slash treatments on soil degradation. Land Degradation and Development 18, 591–607.

Fernández, C., Vega, J.A., 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.

Ferreira, de B.A., 1978. Planaltos e montanhas do norte da Beira - estudo de geomorfologia. Centro de Estudos Geográficos, Lisboa

Ferreira, A.J.D., Coelho, C.O.A., Shakesby, R.A., Walsh, R.P.D., 1997. Sediment and solute yield in forest ecosystems affected by fire and rip-ploughing techniques, central Portugal: a plot and catchment analysis approach. Physics and Chemistry of the Earth 22, 309–314.

Ferreira, A.J.D., Coelho, C.O.A., Walsh, R.P.D., Shakesby, R.A., Ceballos, A., Doerr, S.H., 2000. Hydrological implications of soil water repellency in Eucalyptus globulus forests, north-central Portugal. Journal of Hydrology 231–232, 165–177.

García-Ruiz, J.M., Beguería, S., Nadal-Romero, E., González-Hidalgo, J.C., Lana-Renault, N., Sanjuán, Y., 2015. A Meta-Analysis of soil erosion rates across the world, Geomorphology, Available online 17 March 2015, ISSN 0169-555X, http://dx.doi.org/10.1016/j.geomorph.2015.03.008.

Harding, A., Palutikof, J., Holt, T., 2009. The climate system. In: Woodward, J. (Ed.), The Physical Geography of the Mediterranean. Oxford University Press, Oxford, pp. 69–88.

Helvey, J.D., 1980. Effects of a north central Washington wildfire on runoff and sediment production. Water Resources Bulletin 16, 627–634.

Hungerford, R.D., 1996. Soils-fire in ecosystem management notes:unit II-I. Marana, AZ: U.S. Department of Agriculture, Forest Service, National Advanced Resource Technology Center.

ICNF, 2014. http://www.icnf.pt/portal/florestas/dfci/inc/info-geo consulted 9 January 2014

Inbar, M., Tamir, M., Wittenberg, L., 1998. Runoff and erosion processes after a forest fire in Mount Carmel, a Mediterranean area. Geomorphology 24, 17–33.

Keizer, J.J., Coelho, C.O.A., Matias, M.J.S., Domingues, C.S.P., Ferreira, A.J.D., 2005a. Soil water repellency under dry and wet antecedent weather conditions for selected land-cover types in the coastal zone of central Portugal. Aust. J. Soil Res. 43 (3), 297–308.

Keizer, J.J., Coelho, C.O.A., Shakesby, R.A., Domingues, C.S.P., Malvar, M.C., Perez, I.M.B., Matias, M.J.S., Ferreira, A.J.D., 2005b. The role of soil water repellency in overland flow generation in pine and eucalypt forest stands in coastal Portugal. Australian Journal of Soil Research 43(3), 337-350

Keizer, J.J., Doerr, S.H., Malvar, M.C., Prats, S.A., Ferreira, R.S.V., Oñate, M.G., Coelho, C.O.A., Ferreira, A.J.D., 2008. Temporal variation in topsoil water repellency in two recently burnt eucalypt stands in north-central Portugal. Catena 74: 192-204.

King, P.M., 1981. Comparison of methods for measuring severity of water repellence of sandy soils and assessment of some factors that affect its measurement. Australian Journal of Soil Research 19: 275–285.

Kutiel, P., Lavee, H., Segev, M., Benyamini, Y., 1995. The effect of fire-induced surface heterogeneity on rainfall-runoff-erosion relationships in an eastern Mediterranean ecosystem, Israel. Catena. 25, 77–87.

Lavabre, J., Martin, C., 1997. Impact d'un incendie de forêt sur l'hydrologie et l'érosion hydrique d'un petit bassin versant méditerranéan. Human Impact on Erosion and Sedimentation (Proceedings of Rabat Symposium S6, April 1997): IAHS Publication, no. 245, pp. 39–47.

Littell, R.C., Milliken, G.A., Stroup, W.W., Wolfinger, R.D., Schabenberger, O., 2006. SAS for mixed models, 2nd ed. SAS Institute Inc.: Cary, NC.

López-Bermúdez, F., 2002. Erosión y Desertificación. Nivola, Madrid.

MacDonald, L.H., Larsen, I.J., 2009. Effects of forest fires and post-fire rehabilitation: a Colorado Case study. In: Fire Effects on Soils and Restoration Strategies, edited by A. Cerda and P.R. Robichaud. Science Publishers, Enfield, NH, pp. 423-452.

Malvar, M.C., Prats, S.A., Nunes, J.P., Keizer, J.J., 2011. Post-fire overland flow generation and inter-rill erosion under simulated rainfall in two eucalypt stands in north-central Portugal. Environmental Research 111, 222–236.

Malvar, M.C., Martins, M.A.S., Nunes, J.P., Robichaud, P.R., Keizer, J.J., 2013. Assessing the role of pre-fire ground preparation operations and soil water repellency in post-fire runoff and inter-rill erosion by repeated rainfall simulation experiments in Portuguese eucalypt plantations. Catena 108, 69–83

Malvar, M.C., Prats, S.A., Keizer, J.J., 2015. Runoff and inter-rill erosion affected by wildfire and pre-fire ploughing in eucalypt plantations of North-Central Portugal. Land Degrad. Develop. (2015). DOI: 10.1002/ldr.2365

Martins, M.A.S., Machado, A. I., Serpa, D., Prats, S.A., Faria, S.R., Varela, M.E.T., Gonzalez-Pelayo O., Keizer, J.J., 2013. Runoff and inter-rill erosion in a Maritime Pine and a eucalypt plantation following wildfire and terracing in north-central Portugal. Journal of Hydrology and Hydromechanics 61, 4, 261-269

Mataix-Solera, J., Cerdà, A., Arcenegui, V., Jordán, A., Zavala, L.M., 2011. Fire effects on soil aggregation: a review. Earth Sci. Rev. 109 (1–2), 44–60.

Mayor, A.G., Bautista, S., Llovet, J., Bellot, J., 2007. Post-fire hydrological and erosional responses of a Mediterranean landscape: seven years of catchment-scale dynamics. Catena 71, 68–75.

Morgan, R.P.C., 2005. Soil Erosion and Conservation. Blackwell, Oxford.

Moreira, F., Vaz, P.J.G., Catry, F.X., Silva, J.S., 2009. Regional variations in wildfire susceptibility of land cover types in Portugal: implications for landscape management to minimize fire hazard. International Journal of Wildland Fire 18: 563-574.

Naveh, Z., 1990. Fire in the Mediterranean – a landscape ecological perspective. In: Goldammer, J.G. and Jenkins, M.J. (eds.), Fire in ecosystem dynamics. Pp. 1-20. SPB Academic Publishing, The Hague, Netherlands

Pereira, J., Carreira, J., Silva, J., Vasconcelos, M., 2006a Alguns conceitos básicos sobre os fogos rurais em Portugal. In: Pereira J.S., Pereira J.M.C. et al. (eds.), Incêndios florestais em Portugal – caracterização, impactes e prevenção, pg 133-16, ISAPress, Lisbon.

Pereira, J., Correia, A.V., Correia, A.C., Ferreira, M., Onofre, N., Freitas, H., Godinho, F., 2006b. Florestas e biodiversidade. In: Santos, F., Miranda, P. (Eds.), Alterações climáticas em Portugal—cenários, impactos e medidas de adaptação (Projecto SIAM II). Gradiva, Lisbon, pp. 301–343.

Pimentel, N.L., 1994. As formas do relevo e a sua origem. In: Brito RS (ed) Portugal Perfil Geogra´fico. Editorial Estampa, Lisboa, pp 29–50.

Prats, S.A., MacDonald, L.H., Monteiro, M., Ferreira, A.J.D., Coelho, C.O.A., Keizer, J.J., 2012. Effectiveness of forest residue mulching in reducing post-fire runoff and erosion in a pine and a eucalypt plantation in north-central Portugal, Geoderma 191: 115-124.

Prats, S.A., Malvar, M.C., Vieira, D.C.S, MacDonald, L.H., Keizer, J.J., 2013. Effectiveness of hydro-mulching to reduce runoff and erosion in a recently burnt pine plantation in central Portugal. Land Degradation & Development. (DOI: Link: 10.1002/ldr.2236)

Prats, S.A., Martins, M.A.S., Malvar, M.C., Ben-Hur, M., Keizer, J.J. 2014. Polyacrylamide application versus forest residue mulching for reducing post-fire runoff and soil erosion. Science of the Total Environment 468–469: 464–474

Prosser, I.P., 1990. Fire, humans and denudation at Wangrah Creek, Southern Tablelands, N.S.W. Australian Geographical Studies 28: 77–95.

Prosser, I.P., Williams, L., 1998. The effect of wildfire on runoff and erosion in native Eucalyptus forest. Hydrological Processes. 12, 251–265.

Robichaud, P.R., Waldrop, T.A., 1994. A comparison of surface runoff and sediment yields from a low- and high-severity site preparation burns. Water Resources Bulletin 30, 27– 34.

San-Miguel, J., Camia, A., 2009. Forest fires at a glance: facts, figures and trends in the EU. In: Birot, Y. (ed.), Living with wildfires: what can science tell us, Chp. 1, pp 11-19. European Forest Institute, Discussion Paper 15.

Santos, J.M., Verheijen, F.G.A., Wahren F.T., Wahren A., Gosch L., Bernard-Jannin L., Rial-Rivas M.E., Keizer J.J., Nunes, J.P., 2013. Soil water repellency dynamics in Pine and Eucalypt plantations in Portugal – a high resolution time series. Land Degradation & Development. DOI: 10.1002/ldr.2251

Scott, D.F., Versfeld, D.B., Lesch, W., 1998. Erosion and sediment yield in relation to afforestation and fire in the mountains of the Western Cape Province, South Africa. South African Geographical Journal. 80: 52-59.

Shakesby, R.A., 2011. Post-wildfire soil erosion in the Mediterranean: Review and future research directions. Earth-Science Reviews 105: 71–100.

Shakesby, R.A., Doerr, S.H., 2006. Wildfire as a hydrological and geomorphological agent. Earth-Science Reviews 74: 269–307.

Shakesby, R.A., Coelho, C.O.A., Ferreira, A.D., Terry, J.P., Walsh, R.P.D., 1993. Wildfire impacts on soil erosion and hydrology in wet Mediterranean forest, Portugal. International Journal of Wildland Fire 3: 95–110.

Shakesby, R.A., Coelho, C.de O.A., Ferreira, A.D., Terry, J.P., Walsh, R.P.D., 1994. Fire, post-burn land management practice and soil erosion response curves in eucalyptus and pine forests, north-central Portugal. In: Sala, M., Rubio, J.L. (Eds.), Soil Erosion as a Consequence of Forest Fires. Geoforma, Logroño, Spain, pp. 111–132.

Shakesby, R.A., Boakes, D.J., Coelho, C.O.A., Gonçalves, A.J.B., Walsh, R.P.D., 1996. Limiting the soil degradational impacts of wildfire in pine and eucalypt forests in Portugal. Applied Geography 16: 337–356.

Shakesby, R.A., Coelho, C.O.A., Ferreira, A.J.D., Walsh, R.P.D., 2002. Ground-level changes after wildfire and ploughing in eucalyptus and pine forests, Portugal: implications for soil micro-topographical development and soil longevity. Land Degradation & Development 13, 111 –127.

SNIRH, 2012. Sistema Nacional de Informação de Recursos Hídricos. http://snirh.pt/. Accessed 1 March 2012

Terry, J.P., 1996. Erosion pavement formation and slope process interactions in commercial forest plantations, northern Portugal. Zeitschrift für Geomorphologie 40: 97–115.

Urbanek, E., Shakesby, R.A., 2009. The impact of stone content on water flow in water-repellent sand. European Journal of Soil Science 60, 412–419.

Varela, M.E., Benito, E., Keizer, J.J., 2010. Effects of wildfire and laboratory heating on soil aggregate stability of pine forest in Galicia: the role of lithology, soil organic matter content and water repellency. Catena 83, 127-134

Vega, J.A., Díaz-Fierros, F., 1987. Wildfire effects on soil erosion. Ecologia Mediterranea 13 (4): 119–125.

Veléz, R., 2009. The causing factors: a focus on economic and social driving forces. In: Birot, Y. (ed.), Living with wildfires: what can science tell us? pp 21-26. European Forest Institute, Discussion Paper 15.

Verheijen, F.G.A., Jones, R.J.A., Rickson, R.J., Smith, C.J., 2009. Tolerable versus actual soil erosion rates in Europe. Earth-Science Reviews 94: 23–38.

Walsh, R.P.D., Boakes, D.J., Coelho, C.O.A., Gonçalves, A.J.B., Shakesby, R.A., Thomas, A.D., 1994. Impact of fire-induced water repellency and post-fire forest litter on overland flow in northern and central Portugal. In: Proceedings of the Second International Conference on Forest Fire Research, November 1994, Coimbra, Portugal, Volume II: 1149–1159.

Walsh, R.P.D., Boakes, D.J., Coelho, C.O.A., Ferreira, A.D., Shakesby, R.A., Thomas, A.D., 1995. Post-fire land use management and runoff responses to rainstorms in Portugal. In: McGregor, D.F.M., Thompson, D. (eds), Geomorphology and land management in a changing environment. Wiley, Chichester, pp. 283-308.

Wells, W.G., 1981. Some effects of brushfires on erosion processes in coastal southern California. Erosion and Sediment Transport in Pacific Rim Steeplands, International Association of Hydrological Sciences Publication No. 132. IAHS Press, Institute of Hydrology, Wallingford, UK, pp. 305– 342.

Wischmeier, W.H., Smith, D.D., 1978. Predicting rainfall-erosion losses- a guide to conservation planning. Agriculture Handbook. No. 537. USDA. Washington, D.C; 58.

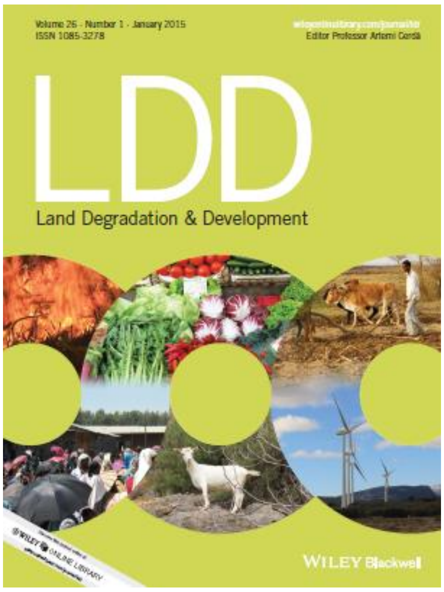
Wittenberg, L., Inbar, M., 2009. The role of fire disturbance on runoff and erosion processes — a long-term approach, Mt. Carmel case study, Israel. Geographical Research 47, 46–56.

WRF, 2006.World Reference Base (FAO). Rome.

Zavala, L.M., Jordán, A., Bellinfante, N., Gil, J., 2010. Relationships between rock fragment cover and soil hydrological response in a Mediterranean environment. Soil Science and Plant Nutrition 56, 95–104.

III. Modelling post fire runoff and erosion

III.I Performance of two erosion models after fire and rehabilitation treatments



Assessing soil erosion after fire and rehabilitation treatments in NW Spain: performance of RUSLE and revised Morgan–Morgan–Finney models.

Fernández, C., Vega, J.A., Vieira, D.C.S. (2010).
Land Degradation and Development. 21, 774–787.

Although the Revised Universal Soil Loss Equation (RUSLE) and the revised Morgan–Morgan–Finney (MMF) are well-known models, not much information is available as regards their suitability in predicting post-fire soil erosion in forest soils. The lack of information is even more pronounced as regards post-fire rehabilitation treatments.

This study compared the soil erosion predicted by the RUSLE and the revised MMF model with the observed values of soil losses, for the first year following fire, in two burned areas in NW of Spain with different levels of fire severity. The applicability of both models to estimate soil losses after three rehabilitation treatments applied in a severely burned area was also tested.

The MMF model presented reasonable accuracy in the predictions while the RUSLE clearly overestimated the observed erosion rates. When the R and C factors obtained by the RUSLE formulation were multiplied by 0·7 and 0.865, respectively, the efficiency of the equation improved.

Both models showed their capability to be used as operational tools to help managers to determine action priorities in areas of high risk of degradation by erosion after fire.

Introduction

Post-fire erosion is a major concern to society because of the potential effects on soil and water resources. Increases in soil erosion rates are frequently observed following wildfire (e.g. Megahan and Molitor, 1975; Campbell et al., 1977; San Roque et al., 1985; Shakesby et al., 1993; Scott et al., 1998; Robichaud and Brown, 2000; Johansen et al., 2001; Martin and Moody, 2001; Meyer et al., 2001; Benavides-Solorio and MacDonald, 2005; Shakesby and Doerr, 2006). Fire severity, as a descriptor of the magnitude of the changes occurred in the soil, has been recognized as a decisive factor controlling those post-fire soil erosion rates (e.g. Benavides-Solorio and MacDonald, 2005; Vega et al., 2005).

Most of these studies have emphasized the reduction or elimination of vegetation cover and ground cover as the main factors explaining the increased soil losses. Soil cover increases infiltration, maintains high soil porosity, prevents soil sealing and increases surface roughness, reducing thus soil erosion (De Bano et al., 1998; Larsen et al., 2009). Fire can also alter the soil structure, by affecting bulk density and total porosity, thus reducing infiltration and promoting overland flow (e.g. De Bano et al., 1998; Neary et al., 2005). Fire-induced hydrophobicity (De Bano, 1981; De Bano et al., 1998; Robichaud, 2000; Huffman et al., 2001; Keizer et al., 2008a) may also contribute to increased soil losses. The effect of fire on soil water repellency depends primarily on the amount and type of litter consumed, the duration and amount of soil heating, and the amount of oxygen available during burning (De Bano et al., 1998; Doerr et al., 2009).

Various models already exist that predict soil erosion for a great variety of crop characteristics. Models such as WEPP (Nearing et al., 1989) and EUROSEM (Morgan et al., 1998) can simulate the effects of vegetation on erosion in individual storms, but are often too complex to be used as operational tools. Simpler, empirically based models such as the revised Morgan–Morgan–Finney (MMF) (Morgan, 2001), USLE (Wischmeier and Smith, 1978) or its revised version Revised Universal Soil Loss Equation (RUSLE) (Renard et al., 1997) may be useful for estimating soil erosion on an annual basis (De Roo, 1996; Tiwari et al., 2000; Morgan, 2001; Morgan and Duzant, 2008). They require less field data than other more complex models and are therefore more feasible as management tools. The USLE model predictions have shown relatively good agreement with other soil erosion estimation data after fire in Galicia (Díaz-Fierros et al., 1987). Acceptable results were also obtained using WEPP and Disturbed WEPP to predict particular soil erosion episodes after fire in Galicia (Soto and Díaz-Fierros, 1998) and the United States (Larsen and MacDonald, 2007). Likewise, the MMF model has performed reasonably well to estimate soil losses in burnt areas in Portugal (Keizer et al., 2008b; Vieira, 2008). However, most of the validation studies of RUSLE and MMF models have been made on agricultural soils (e.g. Shrestha, 1997; Tiwari et al., 2000; Morgan, 2001; Vigiak et al., 2005; López-Vicente et al., 2008; Morgan and Duzant, 2008) and there is a lack of information on the performance of such models in forest soils and, particularly after fire (Dissmeyer and Foster, 1984; Larsen and MacDonald, 2007). Moreover, the validation of soil erosion models after post-fire rehabilitation treatments is particularly scarce all over the world (Robichaud et al., 2007).

Over the last 11 years, there have been about 9000 fires per year in Galicia, representing 47 per cent of forest fires in Spain (Ministerio Medio Ambiente, 2006). Increases in wildfire frequency and burned area are commonly expected under the probable future climate scenarios for the Mediterranean region countries (Moreno, 2005; Carvalho et al., 2008; Good et al., 2008; Moreno, 2009) and also in NW Spain (Vega et al., 2009).

Post-fire soil erosion rates have been assessed in different situations in Galicia, NW Spain (Díaz-Fierros et al., 1987; Vega and Díaz-Fierros, 1987; Díaz-Fierros et al., 1990; Soto et al., 1994; Vega et al., 2005; Fernández et al., 2007, 2008). Operationally useful tools providing reasonable accurate predictions of post-fire sediment yields are needed to guide management decisions to mitigate post-fire soil loss and land degradation and for post-fire rehabilitation planning.

The objective of this study was to assess the performance of the RUSLE and MMF models to predict first-year soil erosion following two wildfires of distinctive severity and after the application of different post-fire rehabilitation treatments in an area affected by a high-severity fire.

Materials and methods

Study Sites

The study was carried out in two burned areas with distinct levels of fire severity in Galicia (NW Spain): Verín (418 57’ 10’’ N; 78 23’ 30’’ W; 550 m a.s.l.) and Soutelo (428 30’ 31’’ N; 88 17’ 17’’ W; 800 m a.s.l.). The main characteristics of the areas are summarized in Table 10.

Table 10 - General characteristics of study sites

	Verín	Soutelo
Location	Ourense province	Pontevedra province
Wildfire date	Summer 2003	Summer 2006
Fire Severity	Moderate soil burn severity =1.0	Severe soil burn severity = 2.7
Dominant vegetation	Pinus pinaster stand	Ulex europaeus shrubland
Climate	Mediterranean	Oceanic
Mean air temperature (ºC)	12	11
Mean annual precipitation (mm)	800	1500
Mean rainfall erosivity (MJ mm h^{-1} ha^{-1} y^{-1})	1000	3000
Soil	Alumi-umbric Regosol	Alumi-umbric Regosol
Substrate	Schist	Schist

Data Collection and Field Measurements

This study used a set of plots initially installed for quantifying soil erosion after wildfire (Verín) and to assess the effect of different soil rehabilitation treatments on soil erosion (Soutelo).

Fourteen and sixteen experimental plots (50 x 10 m^2 each) with their longest dimension along the maximum slope, were installed in Verín and Soutelo, respectively, just after wildfire and before any appreciable rainfall. The plots were delimited by a geotextile fabric fixed to posts. 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 located at the downhill portion of the plots and were used for periodic collection of sediment.

In the Soutelo experimental site to study the effect of different soil rehabilitation treatments on erosion control, three different treatments were assigned at random: straw mulch, wood chip mulch, cut shrub barriers and a control (untreated burned soils). Wheat straw and wood chips were spread manually at a rate of 2·5 and 4 Mg ha^{-1}, respectively. Four barriers made from shrubs cut in an unburned adjacent area were located along the longest dimension of each plot, spaced at regular intervals of 10 m. The barriers were 10 m long, 0·5 m wide and 0·7 m high. Immediately after application of the treatment, the mean soil cover was 80 per cent in the straw mulched plots and 45 per cent in the wood chip mulched plots.

At each study site, amount and intensity of rainfall were measured by two recording rain gauges positioned at 1·20 m above ground, adjacent to the experimental site.

A few days after the wildfire, the percentage of soil organic cover was visually estimated by use of a 20 cm x 20 cm quadrat at 20 systematically selected points along two transects parallel to the plot longest dimension in each plot. Reference quadrats, corresponding to 1, 5, 10, 15, 20, 25 and 50 per cent cover of a 20 x 20 cm^2 quadrat, were prepared on paper to calibrate visual estimates of cover. In addition, each quadrat was assigned to one of the levels of a soil severity index with a modified version of the classification from Ryan and Noste (1983). Four degrees of fire severity were distinguished (Vega et al., 2008): (1) Burnt litter (Oi) but limited duff (Oe + Oa) consumption. (2) Forest floor (Oi + Oe + Oa layers) completely consumed (bare soil) but soil organic matter not consumed and surface soil intact. (3) Forest floor completely consumed and soil organic matter in Ah horizon also consumed, a thick layer of ash deposited and soil structure altered. (4) As (3) and colour altered (reddish). A mean value of these scores was used to assess the impact of fire on soil in each burnt plot.

A few days after fire, the percentage of ground cover by plants established from seeds or resprouting after fire was estimated visually, in a 70 x 70 cm^2 quadrat, at 20 systematically selected points in each plot. Measurements of vegetation height were also made. Sampling was repeated every 3 months in each experimental plot.

Immediately after fire, soil shear strength (0–5 cm) was measured with a vane tester (Eijkelkamp) at 20 points in each experimental plot. Measurements were made quarterly during the study period.

Samples of surface mineral soil (0–10 cm) were taken at 15 systematically chosen points within each plot to determine moisture content by gravimetry (oven-dried for 24 h at 1058C). The samples were taken at monthly intervals during the period of study.

Soil bulk density was determined immediately after fire in both study areas. In Soutelo, the measurements were repeated quarterly. A metal cylinder of 15 cm diameter was inserted into the upper 5 cm layer of mineral soil and bulk density was calculated by dividing the oven-dried soil mass by the volume of the soil core (free of gravel).

Soil depth was measured with a metal stick at 20 randomly selected points inside each plot. Further details about the study sites are available in Fernández et al. (2007) and Fernández et al. (2011).

Application of RUSLE Model

Application of this model (Renard et al., 1997) was based on the procedure described by Wischmeier and Smith (1978) to estimate soil losses, A (Mg ha^{-1} y^{-1}), which consists of the product of five factors, rainfall erosivity, R (MJ mm h^{-1} ha^{-1} y^{-1}), soil erodibility K (Mg h MJ^{-1} mm^{-1}), and the non-dimensional topographic factor (LS), crop factor (C) and soil conservation practices factor (P):

$$A = R \times K \times L \times S \times C \times P$$

Determination of the R factor was initially based on rainfall data for all the events that occurred in both study areas during the year of study. The topographic factor was obtained according to the characteristics of the different plots.

The soil erodibility, K, was calculated by use of the equation proposed by Wischmeier and Smith (1978) because in both areas the percentage of organic matter was higher than 4 per cent (Renard et al., 1997).

The C factor was calculated according to the following equation:

$$C = PLU \times CC \times SC \times SR \times SM$$

Where PLU is the prior land use subfactor, CC is the canopy cover subfactor, SC is the surface cover subfactor, SR is the surface roughness subfactor and SM is the soil moisture subfactor (Renard et al., 1997).

The PLU subfactor is computed from a soil reconsolidation factor, the mass of roots and the mass of buried residue (Renard et al., 1997). A value of 0·45 was assigned to the reconsolidation factor as proposed by Dissmeyer and Foster (1981) for forest soils; the mass of buried residue was assumed to be zero and the mass of roots was obtained according to Achat et al. (2008) for Pinus pinaster and Soto and Díaz-Fierros (1998) for Ulex europaeus.

The CC subfactor was calculated from percent canopy cover and fall height obtained from vegetation surveys in the field.

We used the values proposed by Larsen and MacDonald (2007) to calculate the SC subfactor: a value for the unitless coefficient that indicates the effectiveness of surface cover in reducing erosion (b) of 0·05 as rilling is the dominant process, percent of surface cover (Sp) as the mean of spring and autumn cover in each plot and for roughness of an untilled surface (Ru), a value of 1·52 cm in the severely burned plots and 2·54 cm in the moderately severely burned plots. The SR subfactor was calculated using the same Ru values.

Since the SM subfactor has not been calibrated yet for burned forest soils (González-Bonorino and Osterkamp, 2004), a value of 1·0 was used following Larsen and MacDonald (2007).

Variation in the C and R factors throughout the period of study in both areas is shown in Figure 20. The mean C factor was obtained according to the distribution of rainfall erosivity in each study area.

The maximum value of the P factor was 1 for the plots in which no conservation practices were applied. For the plots in which rehabilitation treatments were carried out, this value changed according to the effectiveness of treatments determined (Fernández et al., 2011) in terms of the ratio between annual soil losses measured in treated and untreated plots (0·343 straw mulch; 0·943 wood chip mulch and 0·857 cut shrub barriers).

The input parameters for the RUSLE model are listed in **Table 11**.

Table 11 - Input parameters for RUSLE model in both study sites

Factor	Parameter	Verín Moderate fire	Soutelo Severe fire
Rainfall erosivity	R (MJ mm h^{-1} ha^{-1} y^{-1})	224 (0·01)	2547 (0·02)
Soil erodibility	K (Mg ha^{-1} MJ^{-1} mm^{-1})	0·015 (0·001)	0·017 (0·001)
Topographic factor	LS	6·37 (0·24)	8·70 (0·10)
Crop factor	C	0·002 (0·0001)	0·249 (0·001)
Soil conservation practices	P	1	1

Standard errors are given in parentheses

Application of Revised Morgan–Morgan–Finney Model (MMF)

The revised MMF model (Morgan, 2001) used the concepts by Meyer and Wischmeier (1969) and Kirkby (1976). This model separates the soil erosion process in two phases: the water phase and the sediment phase. The water phase determines the energy of rainfall available for soil particles detachment from the soil and the volume of runoff. In the erosion phase, rates of soil particle detachment by rainfall and runoff are determined along with the transport capacity of runoff. Predictions of total particle detachment and transport capacity are compared and erosion rate is equated to the lower of the two rates.

The input parameters in the model are grouped in four factors. The rainfall factor includes annual rainfall (R), rainfall per rainy days (Rn) and the typical value for intensity of erosive rain (I). The soil factor includes, soil moisture at field capacity (MS), bulk density of the top soil layer (BD), hydrological depth of soil (EHD), soil detachability index (K) and cohesion of the surface soil (COH) parameters. The landform factor includes only slope steepness (S). The land cover factor includes rainfall interception (A), actual evapotranspiration (Et), potential evapotranspiration (E0) and crop cover management factor (C), canopy cover (CC), ground cover (GC) and vegetation cover to the ground surface (PH) parameters.

Rainfall parameters (R, Rn and I) were obtained from the recording rain gauges installed in each study site. The rainfall kinetic energy equations used were those proposed by Coutinho and Tomás (1995) in Verín, and by Marshall and Palmer (1948) in Soutelo.

Soil moisture, bulk density, hydrological depth of soil and cohesion of the surface soil parameters were measured in both areas during the year of study as explained before. The detachability index (K) was obtained according to the soil texture (Morgan, 2001).

The rainfall interception (A) was computed according to previous studies made in Galicia for pine stands (Gras, 1993) and shrublands (Vega et al., 2005). The potential and actual evapotranspiration were estimated by the methods proposed by Thornthwaite (1948) and Turc (1955), respectively. The C factor of MMF is the product of the C and P factors from the USLE equation (Wischmeier and Smith, 1978), and in the application of this model the same values as obtained from the RUSLE model were applied. Canopy cover (CC), ground cover (GC) and vegetation cover to the ground surface (PH) parameters were measured in both areas during the year of study as explained before. The model inputs are listed in Table 12.

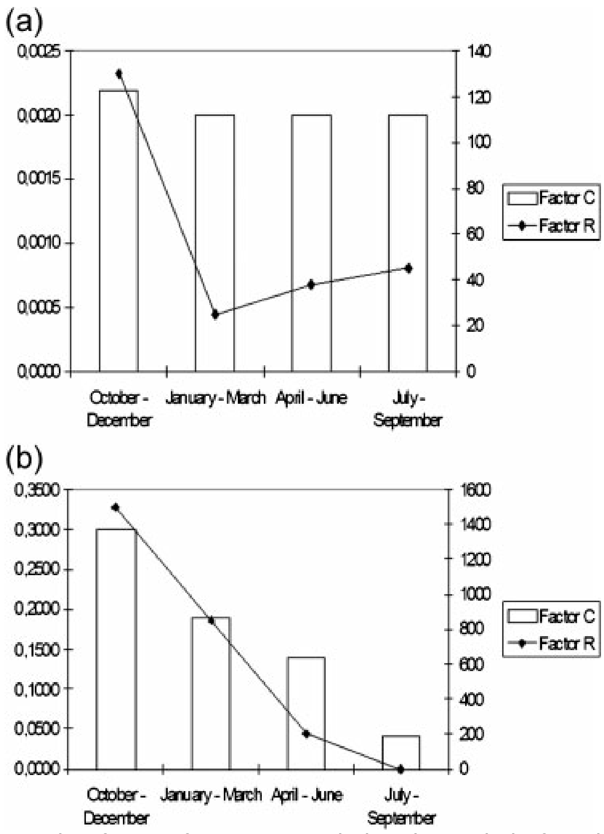


Figure 20 - Variation in R and C factors from RUSLE during the period of study in both study areas. (a, Verín; b, Soutelo).

Table 12 - Input parameters for MMF model in both study sites

Factor	Parameter	Verín Moderate fire	Soutelo Severe fire
Rainfall	R (mm y^{-1})	640·4 (0·2)	1554·9 (0·5)
	Rn (mm raining day^{-1})	4·5 (0·2)	15·5 (0·5)
	I (mm h^{-1})	18	30
Soil	MS (%)	27 (0·02)	25 (0·01)
	BD (g cm^{-3})	0·59 (0·02)	0·69 (0·01)
	EHD (m)	0·266 (0·02)	0·270 (0·03)
	K (g J-1)	0·5 (0·01)	0·5 (0·01)
	COH (kPa)	26 (0·8)	33 (2·5)
Landform	S (º)	16·2 (0·7)	22·2 (0·2)
Land cover	A	0·20	0·13
	Et/E0	0·56	0·75
	C	0·002 (0·0001)	0·249 (0·001)
	CC (%)	34 (0·5)	0 (0·0)
	GC (%)	100 (0·01)	1 (0·01)
	PH (m)	13·1 (0·20)	0·6 (0·01)

Standard errors are given in parentheses

Statistical Analysis

Predicted annual soil losses values were evaluated by

Coefficient of efficiency (Nash and Sutcliffe, 1970), Ef, a descriptor of the predictive accuracy of model outputs. Ef can range from -1 to 1. A negative value indicates that the mean observed value is a better predictor than the model, a value of 0·0 indicates that the mean observed value is as accurate a predictor as the model and an efficiency of 1 corresponds to a perfect match of predicted to the observed data. The closer the Ef is to 1, the more accurate the model is.

The root mean squared errors, RMSE, measures the average magnitude of error between observed and forecasted values.

The Wilcoxon rank sum method for the difference between forecasted and observed sediment losses. It is a non- parametric test for assessing if two independent samples come from the same distribution.

Results

Soil losses after moderate and severe fires

RUSLE

The results showed that the model overestimated erosion rates by one order of magnitude, particularly in the severe fire, and whereas the mean measured value of annual soil losses in Soutelo was 3·5 kg m^{-2}, those predicted by RUSLE were 9·2 kg m^{-2} (Figure 21). In Verín, the corresponding values were 0·003 and 0·005 kg m^{-2}, respectively.

The validation statistics for the RUSLE are shown in Table 13. The negative value of the efficiency index indicates that the mean of observed values is a better predictor than the model.

MMF

When the MMF model is applied according to the procedure described by Morgan (2001), all the results depend on the annual transport capacity of runoff. The MMF model tended to underestimate soil erosion rates (Figure 21). The mean predicted value of annual soil losses in Soutelo was 2·6 kg m^{-2} versus 3·5 kg m^{-2} observed and in Verín, 0·0001 kg m^{-2} versus 0·003 kg m^{-2}. However, the validation statistics were better than those obtained with the RUSLE model (Table 13) and annual values of predicted and measured soil losses did not differ according to the Wilcoxon test.

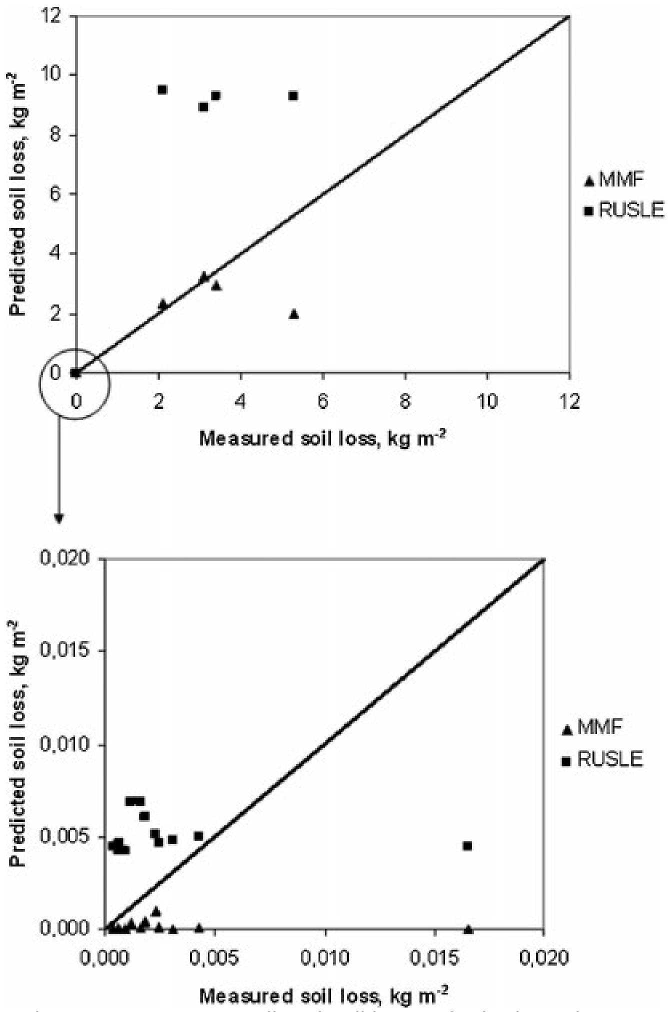


Figure 21 - Measured and RUSLE or MMF-predicted soil losses for both study areas.

Table 13 - Validation statistics for the RUSLE and MMF modelling for both study areas

	RUSLE	MMF
Ef	-2·208	0·736
RMSE (kg m^{-2})	3·146	0·902
Wilcoxon test—p-value	0·000	0·913

Soil Losses after Post-fire Erosion Control Treatments

RUSLE

The application of the RUSLE model to the different treatments applied for erosion control was based on the same inputs that were used for the severe fire in Soutelo (Table 11) with the exception of the P factor, which was different in the treatments: 0·343 for straw mulch, 0·857 for cut shrub barriers and 0·943 for wood chip mulch.

The results showed that the RUSLE model overestimate the soil losses when compared with the measured values (Figure 22). The validation statistics obtained to test the efficacy of RUSLE to predict soil erosion were also very poor (Table 14).

MMF

As in RUSLE, the application of the MMF model to the different treatments used for erosion control was based on the same inputs used for the severe fire in the Soutelo site (Table 12), with the exception of the P factor, which varied in the different treatments.

The poor agreement between observed and predicted values can be observed in Figure 22. The MMF presented a comparatively better efficiency index that RUSLE (Table 14). No differences between predicted and observed valves of soil losses were found (Table 14).

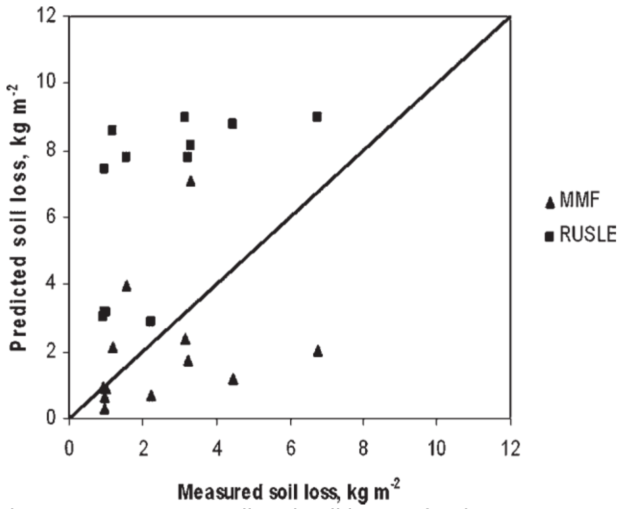


Figure 22 - Measured and RUSLE or MMF-predicted soil losses for the treatments applied

Table 14 - Validation statistics for the RUSLE and MMF modelling for the treatments applied

	RUSLE	MMF
Ef	-6·009	-0·687
RMSE (kg m^{-2})	1·914	2·457
Wilcoxon test—p-value	0·041	0·347

Discussion

The reasonably good predictions of post-fire soil losses achieved with MMF is consistent with those previously observed in burned areas in Portugal (Keizer et al., 2008b; Vieira, 2008). The poorer results obtained with RUSLE are similar to those reported by Larsen and MacDonald (2007), who also observed negative efficiency indexes when predicting sediment yields the first year after fires of different levels of fire severity in Colorado (USA) with RUSLE. Better results were obtained by Díaz-Fierros et al. (1987) with the application of USLE, although the different methodology used to measure soil losses in the field do not allow direct comparison with data obtained in the present study. Soto and Díaz-Fierros (1998) obtained efficiency indexes of 0·6 and 0·03 after prescribed burning and wildfire, respectively, in shrublands in NW Spain, with the WEPP model.

The results presented here correspond to the first year after fire and this may limit the accuracy of the predictions as it has been shown that models are better for predicting average conditions than soil losses for particular years (Larsen and MacDonald, 2007).

There is no data available from rehabilitation studies of burned areas for comparing the accuracy of prediction achieved by the models in the plots to which rehabilitation treatments were applied.

Although there are a considerable number of studies testing RUSLE, the available information on burned soils is particularly scarce. The overestimation of soil losses predicted by RUSLE, particularly in the severe fire, contrasts with the findings of Larsen and MacDonald (2007).

One of the possible reasons for the overestimates may be the use of an inadequate kinetic energy equation of rainfall for this climate, although its original formulation seems to be appropriate under oceanic influence climates (Van Dijk et al., 2002). Larsen and MacDonald (2007) suggest the incorporation of a rainfall erosivity threshold and a nonlinear relationship between rainfall erosivity and soil losses to improve the ability of RUSLE to predict post- fire soil erosion. However, in their case, convective storms were the dominant type of rainfall events.

In the present study, an alternative estimation of R according to the formulation proposed by Roose (1975) and Morgan (1995) for tropical areas, which involves multiplying the annual rainfall by 0·865, would result in a lower R value and increased the efficiency index from -2·208 (Table 13) to 0·690 and the RMSE decreased to 0·977 kg m^{-2}. This suggests that R calculated by the Wischmeier and Smith (1978) equation would overestimate the rainfall erosivity effect in this area.

The primary effects of burning are to alter the soil and surface cover, so this may induce noticeable changes in the K and C factors. The model estimations suggest that the K and C factors do not adequately describe soil modifications after fire.

The K factor is based on soil texture, soil organic matter, permeability class and soil structure. The decline in infiltration caused by increased post-fire soil water repellency is often considered as the primary cause of the increase in runoff after burning (e.g. DeBano, 2000; Shakesby and Doerr, 2006), although soil water repellency is not explicitly considered in the RUSLE model and was not measured in this study. Miller et al. (2003) suggested changing the permeability class chosen in the initial calculations to very slow, to take into account the effect of post- fire soil water repellency in the K factor. Moreover, very severe fires may also reduce the structural stability of the soil and increase the soil erodibility (Soto et al., 1991; Cerdà et al., 1995; Andreu et al., 2001; García-Corona et al., 2004; Mataix-Solera and Doerr, 2004). However, the opposite relationship is assumed in the quantitative effect of the structure classes on the K factor. As a result, a decrease in aggregate stability after fire decreases rather than increases the K factor. Larsen and MacDonald (2007) suggest that the current algorithms for calculating K values are not consistent with the understanding of erosion processes after fire and propose that a reformulation would be required to achieve more precise predictions. However, in the present case, the proposed modifications would produce an increase in the RUSLE predictions. The influence of the reduction of the soil organic matter content on soil erodibility after fire is not clear in these soils, because of the observed high content even after very severe fire and may partially explain the overestimation observed in the present study.

The cover-management factor (C) is one of the most important variables because soil organic cover is a major determining factor as regards post-fire sediment yields (e.g. Pierson et al., 2001; Pannkuk and Robichaud, 2003; Benavides-Solorio and MacDonald, 2005; Vega et al., 2005, Wagenbrenner et al., 2006; Fernández et al., 2007, 2008). The values of C obtained here appear to contribute to an overestimation of soil erosion losses in the high- severity area. The problem is that data on soil consolidation over time, soil root mass over time, drop fall height and surface roughness are approximations, because of the absence of detailed field data for an accurate calculation of this factor. In the absence of such data, it is not possible to assess the validity of the relationships used to calculate the C factor (González-Bonorino and Osterkamp, 2004; Larsen and MacDonald, 2007).

As stated before with the K factor, the high soil organic matter content of these soils could affect the computation of the C factor. Dissmeyer and Foster (1981) proposed a correction in the C factor for soils with high soil organic matter content that consists in multiplying the previously computed value of C by 0·7. If we use this correction factor, the C values would be 0·002 and 0·17 for the moderately and severely burned areas, respectively. Taking into account the above modifications in the C and R factors (Figure 23), the efficiency index increased to 0·872 and the RMSE decreased to 0·628 kg m^{-2}.

Unexpectedly, although the MMF model was not developed for burned soils, the Ef index obtained suggests the suitability of this model for predicting soil erosion after a fire. The discrepancies between observed and predicted data may be related to the fact that estimated values of evapotranspiration were used and there was no vegetative cover during some months. It is uncertain how these estimations could affect the soil moisture storage capacity in these burned soils and, thus the model predictions.

As stated by Morgan (2001), the hydrological depth of soil is a controversial parameter, and although in the present case the values used were based on field measurements, there remain uncertainties as regards the real value. Better knowledge of these parameters would probably produce more accurate estimations of soil erosion.

As regards as soil losses after post-fire erosion control treatments predictions, there are several possible reasons for the poor results obtained. For example, the values assigned to the P factor. As pointed out by Miller et al. (2003), P factor values are usually unreliable because of the lack of validation of the effectiveness of post-fire rehabilitation treatments. However, in the present study, we chose the values according to the respective efficacy values for the soil rehabilitation treatments measured in a field experiment (Fernández et al., 2011). The value of P for cut shrub barriers is consistent with that proposed by Miller et al. (2003) and with the results of some field studies on the effectiveness of rehabilitation treatments after fire (Wagenbrenner et al., 2006; Robichaud et al., 2008). A reduction in factor LS, taking into account the distance between barriers along the slope did not improve predictions.

The proposed modifications of R and C factors in the RUSLE substantially improved the predictions ($Ef = 0{\cdot}333$).

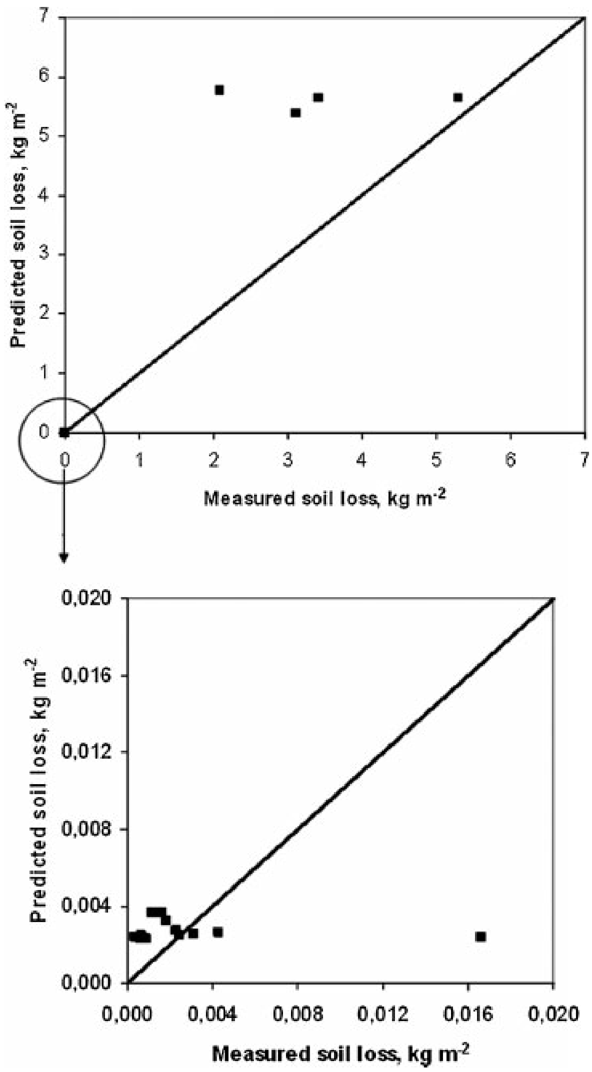


Figure 23 - Measured and RUSLE-predicted soil losses for both study areas after the modification of the R and C factors.

Conclusions

Post-fire soil losses predicted by the RUSLE and Morgan–Finney models were compared in two burned areas with different levels of fire severity in NW Spain. An acceptable efficiency index was only obtained with the MMF model although it slightly underestimates post-fire soil losses.

RUSLE model predictions overestimated actual annual soil losses. RUSLE K factor did not allow to reflect the changes on soil permeability and structure after fire. A correction of C factor to take into account the high organic matter content of the studied soils and a modification of the R factor could improve the applicability of RUSLE on similar burned soils as those under study.

The differences between observed and predicted values with MMF may be caused by using estimated values for evapotranspiration and how they affect the soil moisture storage capacity. More research on this aspect is needed.

No accurate prediction of soil erosion after soil rehabilitation was achieved with the models tested. The role played by the C and P factors was not fully established and may have led to the poor results.

Despite their limitations, both models were able to clearly distinguish situations of high and low post-fire erosion risk. This shows the applicability of both models to be used as operational tools in terms of prioritizing management areas.

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References

Achat, D.L., Bakker, M.R., Trichet, P., 2008. Rooting patterns and fine root biomass of Pinus pinaster assessed by trench wall and core methods. Journal of Forest Research 13: 165–175.

Andreu, V., Imeson, A.C., Rubio, J.L., 2001. Temporal changes in soil aggregates and water erosion after a wildfire in a Mediterranean pine forest. Catena 44: 69–84.

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

Campbell, R.E., Baker, M.B., Ffolliott, P.F., Larson, F.R., Avery, C.C., 1977. Wildfire effects on a ponderosa pine ecosystem: An Arizona case study. USDA Forest

Service. Rocky Mountain Forest and Range Experiment Station Research Paper. RM 191.

Carvalho, A., Flannigan, M., Logan, K., Miranda, A., Borrego, C., 2008. Fire activity in Portugal and its relationship to weather and the Canadian Fire Weather Index System. International Journal of Wildland Fire 17: 328–338.

Cerdà, A., Imeson, A.C., Calvo, A., 1995. Fire and aspect induced differences on soil erodibility and hydrology of soils at La Costera, Valencia, Southeast Spain. Catena 24: 289–304.

Coutinho, M.A., Tomás, P., 1995. Characterisation of raindrop size distributions at the Vale Formoso Experimental Erosion Centre. Catena 25: 187– 197.

De Bano, L.F., 1981. Water repellent soils: A state of art. USDA Forest Service General Technical Report, PSW-46.

DeBano, L.F., 2000. The role of fire and soil heating on water repellency in wildland environments: A review. Journal of Hydrology 231–232: 195– 206.

De Bano, L.F., Neary, D.G., Ffolliott, P.F., 1998. Fire’s Effects on Ecosystems. John Wiley and Sons: New York.

De Roo, A.P.J., 1996. Validation problems of hydrologic and soil-erosion catchment models: Examples from a Dutch soil erosion project. In Advances in Hillslope Processes, Anderson MG, Brooks SM (Eds). Wiley: Chichester; 669–683.

Díaz-Fierros, F., Benito, E., Pérez, R., 1987. Evaluation of the U.S.L.E. for prediction of erosion in burnt forest areas in Galicia (NW Spain). Catena 14: 189–199.

Díaz-Fierros, F., Benito, E., Vega, J.A., Castelão, A., Soto, B., Pérez, R., Taboada, T., 1990. Solute loss and soil erosion in burnt soils from Galicia (NW Spain). In Fire and Ecosystem Dynamics, Goldammer JG, Jenkins y MJ (Eds). SPB Academic Publishing: The Hague; 103–116.

Dissmeyer, G.E., Foster, G.R., 1981. Estimating the cover management factor in the USLE for forest conditions. Journal of Soil Water Conservation 36: 235–240.

Dissmeyer, G.E., Foster, G.R., 1984. A guide for predicting sheet and rill erosion on forest land, Technical Publication R8-TP 6.

Doerr, S.H., Shakesby, R.A., MacDonald, L.H., 2009. Soil water repellency: A key factor in post-fire erosion. In Restoration Strategies after Forest Fires, Cerdà, A., Robichaud P R (Eds). Science Publishers, Inc.: Enfield, NH.

Fernández, C., Vega, J.A., Fontúrbel, M.T., Jiménez, E., Pérez, J.R., 2008. Immediate effects of prescribed burning, chopping and clearing on runoff, infiltration and erosion in a shrubland area in Galicia (NW Spain). Land Degradation and Development 19: 502–515.

Fernández, C., Vega, J.A., Fontúrbel, M.T., Pérez-Gorostiaga, P., Jiménez, E., Madrigal, J., 2007. Effects of wildfire, salvage logging and slash treatments on soil degradation. Land Degradation and Development 18: 591–607.

Fernández, C., Vega, J.A., Jiménez, E., Fontúrbel, M.T., 2011. Effectiveness of different post-fire erosion control treatments in Galicia (NW Spain). International Journal of Wildland Fire.

García-Corona, R., Benito, E., de Blas, E., Varela, M.E., 2004. Effects of heating on some soil physical properties related to its hydrological behaviour in two north-western Spanish soils. International Journal of Wildland Fire 13: 195–199.

González-Bonorino, G., Osterkamp, W.R., 2004. Applying RUSLE 2.0 on burned forest lands: An appraisal. Journal of Soil and Water Conservation 59: 36–42.

Good, P., Moriondo, M., Giannakopoulos, C., Bindi, M., 2008. The meteorological conditions associated with extreme fire risk in Italy and Greece: Relevance to climate model studies. International Journal of Wildland Fire 17: 155–165.

Gras, J.M., 1993. Investigación sobre las relaciones hídricas de las plantaciones de E. globulus en Galicia. PhD. Dissertation, Universidad Politécnica de Madrid.

Huffman, E.L., MacDonald, L.H., Stednick, J.D., 2001. Strength and persistence of fire-induced soil hydrophobicity under ponderosa and lodgepole pine, Colorado Front Range. Hydrological Processes 15: 2877–2892.

Johansen, M.P., Hakonson, T.E., Breshears, D.D., 2001. Post-fire runoff and erosion from rainfall simulation: Contrasting forests with shrublands and grasslands. Hydrological Processes 15: 2953–2965.

Keizer, J.J., Doerr, S.H., Malvar, M.C., Prats, S.A., Ferreira, R.S.V., Oñate, M.G., Coelho, C.O.A., Ferreira, A.J.D., 2008a. Temporal variation in topsoil water repellency in two recently burnt eucalypt stands in north-central Portugal. Catena 74: 192–204.

Keizer, J.J., Nunes, J.P., Palacios, E., Beekman, W., Malvar, M.C., 2008b. Soil erosion modeling for two recently burnt eucalypt slopes: Comparing MEFIDIS, MMF and USLE. ISCO2008, 15th Conference of the International Soil Conservation Organization, Budapest, Hungary, 18–23 May.

Kirkby, M.J., 1976. Hydrological slope models: The influence of climate. In Geomorphology and Climate, Derbyshire E (Ed.). Wiley: London; 247–267.

Larsen, I.J., MacDonald, L.H., 2007. Predicting post-fire sediment yields at the hillslope scale: Testing RUSLE and Disturbed WEPP. Water Resources Research 43: W11412.

Larsen, I.J., MacDonald, L.H., Brown, E., Rough, D., Welsh, M.J., Pietraszek, J.H., Libohova, Z., Benavides-Solorio, J., Schaffrath, K., 2009. Causes of post- fire runoff and erosion: Water repellency, cover, or soil sealing? Soil Science Society American Journal 73: 1393–1407.

López-Vicente, M., Navas, A., Machín, J., 2008. Modelling soil detachment rates in rainfed agrosystems in the south-central Pyrenees. Agricultural Water Management 95: 1079–1089.

Marshall, J.S., Palmer, W.M., 1948. Relation of rain drop size to intensity. Journal of Meteorology 5: 165–166.

Martin, D., Moody, J.A., 2001. Comparison of soil infiltration rates in burned and unburned mountainous watersheds. Hydrological Processes 15: 2893–2903.

Mataix-Solera, J., Doerr, S.H., 2004. Hydrophobicity and aggregate stability in calcareous topsoils from fire-affected pine forests in southeastern Spain. Geoderma 118: 77–88.

Megahan, W.F., Molitor, A.C., 1975. Erosional effects of wildfire and logging in Idaho. Water Management Symposium. Logan, UT; 423–444. Meyer LD, Wischmeier WH. 1969. Mathematical simulation of the process of soil erosion by water.

Transactions of the America Society of Agricultural Engineers 12: 754–758 and 762.

Meyer, G.A., Pierce, J.L., Wood, S.H., Jull, A.J.T., 2001. Fire, storms, and erosional events in the Idaho Batholith. Hydrological Processes 15: 3025– 3038.

Miller, J.D., Nyhan, J.W., Yool, S.R., 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.

Ministerio de Medio Ambiente, 2006. Los incendios forestales en España: Decenio 1996–2005. Madrid.

Moreno, J.M., 2005. Impacto sobre los riesgos naturales de origen climático. Riesgo de incendios forestales. In Moreno JM (coord.), Evaluación preliminar de los impactos en España por efecto del cambio climático. Ministerio de Medio Ambiente (Madrid); 1–64.

Moreno, J.M., 2009. Impacts on potential wildfire risk due to changes in climate. In Birot Y (ed.). Living with Wildfires: What Science Can Tell Us. A Contribution to the Science-Policy Dialogue, EFI Discussion Paper 15; European Forest Institute. Joensuu.

Morgan, R.P.C., 1995. Soil Erosion and Conservation, 2nd ed. Longman Group Limited, London.

Morgan, R.P.C., 2001. A simple approach to soil loss prediction: A revised Morgan–Morgan–Finney model. Catena 44: 305–322.

Morgan, R.P.C., Quinton, J.N., Smith, R.E., Govers, G., Poesen, J.W.A., Auerswald, K., Chisci, G., Torri, D., Styzcen, M.E., 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.

Morgan, R.P.C., Duzant, J.H., 2008. Modified MMF (Morgan–Morgan–Finney) model for evaluating effects of crops and vegetation cover on soil erosion. Earth Surface Processes and Landforms 32: 90–106.

Nash, J.E., Sutcliffe, J.V., 1970. River flow forecasting through conceptual models. Part 1: A discussion of principles. Journal of Hydrology 10: 282– 290.

Nearing, M.A., Foster, G.R., Lane, L.J., Finkner, S.C., 1989. A process-based soil erosion model for USDA-water erosion prediction project technology. American Society of Agricultural Engineers 32: 1587–1593.

Neary, D.G., Ryan, K.C., De Bano, L.F., 2005. Wildland fire in ecosystems. Effects of fire on soil and water. General Technical Report. RMRS-GTR- 42-vol 4. USDA Forest Service, Rocky Mountain Research Station: Ogden, UT.

Pannuk, C.D., Robichaud, P.R., 2003. Effectiveness of needle cast at reducing erosion after forest fires. Water Resources Research 39: 1333–1342. Pierson FB, Robichaud PR, Spaeth KE. 2001. Spatial and temporal effects of wildfire on the hydrology of a steep rangeland watershed. Hydrological Processes 15: 2953–2965.

Renard, K.G., Foster, G.R., Weesies, G.A., Mc Cool, D.K., Yoder, D.C., (co-ordinators). 1997. Predicting soil erosion by water: A guide to conservation planning with the revised universal soil loss equation (RUSLE). Agriculture Handbook n- 703. USDA, Natural Resources Conservation Service: Washington, DC.

Robichaud, P.R., 2000. Fire effects on infiltration rates after prescribed fire in Northern Rocky Mountain forests, USA. Journal of Hydrology 231– 232: 220–229.

Robichaud, P.R., Brown, R.E., 2000. What happened after the smoke cleared: Onsite erosion rates after a wildfire in eastern Oregon. In Proceedings Wildland Hydrology Conference 1999, Olsen DS, Potyondy JP (Eds). American Resources Association: Hernon, VA; 419–426.

Robichaud, P.R., Brown, R.E., 2002. Silt fences: An economical technique for measuring hillslope soil erosion. General Technical Report RMRS- GTR-94, USDA Forest Service.

Robichaud, P.R., Elliot, W.J., Pierson, F.B., Hall, D.E., Moffet, C.A., 2007. Predicting postfire erosion and mitigation effectiveness with a web-based probabilistic erosion model. Catena 71: 229–241.

Robichaud, P.R., Pierson, F.B., Brown, R.E., Wagenbrenner, J.W., 2008. Measuring effectiveness of three postfire hillslope erosion barrier treatments, western Montana, USA. Hydrological Processes 22: 159–170.

Roose, E.J., 1975. Erosion et ruissellement en Afrique de l'ouest: vingt années de mesures en petites parcelles experimentales. Cyclo. ORSTOM: Adiopodoumé, Ivory Coast.

Ryan, K.C., Noste, N.V., 1983. Evaluating prescribed fires. In Lotan JE, Kilgore BM, Fischer WC, Mutch RW (Tech. Coord.), Symposium and Workshop of Wilderness Fire. Proc. General Technical Report. INT-182, USDA Forest Service Res. Sta.; 230–238.

San Roque, P., Rubio, J.L., Mansanet, J., 1985. Efectos de los incendios forestales en las propiedades del suelo, en la composición florística y en la erosión hídrica de zonas forestales de Valencia (España). Revue d'Ecologie et Biologie du Sol 22: 131–147.

Scott, D.F., Versfeld, D.B., Lesch, W., 1998. Erosion and sediment yield in relation to afforestation and fire in the mountains of the Western Cape Province, South Africa. South African Geographical Journal 80: 52–59.

Shakesby, R.A., Coelho, C.O.A., Ferreira, A.D., Terry, J.P., Walsh, R.P.D., 1993. Wildfire impacts on soil erosion and hydrology in wet Mediterranean forest, Portugal. International Journal of Wildland Fire 3: 95–110.

Shakesby, R.A., Doerr, S.H., 2006. Wildfire as a hydrological and geomorphological agent. Earth-Science Reviews 74: 269–307.

Shrestha, D.P., 1997. Assessment of soil erosion in the Nepalese Himalaya: A case study in Likhu Khola Valley, Middle Mountain Region. Land Husbandry 2: 59–80.

Soto, B., Basanta, R., Benito, E., Pérez, R., Díaz-Fierros, F., 1994. Runoff and erosion from burnt soils in Northwest Spain. In Soil Erosion as a Consequence of Forest Fires, Sala M, Rubio JL (Eds). Geoforma: Logroño, Spain; 91–98.

Soto, B., Benito, E., Díaz-Fierros, F., 1991. Heat-induced degradation processes in forest soils. International Journal of Wildland Fire 1: 147–152.

Soto, B., Díaz-Fierros, F., 1998. Runoff and soil erosion from areas of burnt scrub: Comparison of experimental results with those predicted by the WEPP model. Catena 31: 257–270.

Thornthwaite, C.W., 1948. An approach toward a rational classification of climate. Geographical Review 38: 55–94.

Tiwari, A.K., Risse, L.M., Nearing, M.A., 2000. Evaluation of WEPP and its comparison with USLE and RUSLE. Transactions of the American Society of Agricultural Engineers 43: 1129–1135.

Turc, L., 1955. Le bilan d' eau des sols. Relations entre les precipitations, l' evaporation, et l' ecoulement. Annales Agronomiques 6: 5–131.

Van Dijk, A.I.J.M., Bruijnzeel, L.A., Rosewell, C.J., 2002. Rainfall intensity-kinetic energy relationships: A critical literature appraisal. Journal of Hydrology 261: 1–23.

Vega, J.A., Díaz-Fierros, F., 1987. Wildfire effects on soil erosion. Ecologia Mediterranea 13: 175–181.

Vega, J.A., Fernández, C., Fontúrbel, T., 2005. Throughfall, runoff and soil erosion alter prescribed burning in gorse shrubland in Galicia (NW Spain). Land Degradation & Development 15: 1–15.

Vega, J.A., Fernández, C., Jiménez, E., Ruiz, A.D., 2009. Evidencias de cambio climático en Galicia através de la tendencia de los índices de peligro de incendios forestales. In Evidencias e impactos del cambio climático en Galicia. Xunta de Galicia, Santiago de Compostela; 173–194.

Vega, J.A., Fernández, C., Pérez-Gorostiaga, P., Fontúrbel, M.T., 2008. The influence of fire severity, serotiny, and post-fire management on Pinus pinaster Ait. recruitment in three burnt areas in Galicia (NW Spain). Forest Ecology and Management 256: 1596–1603.

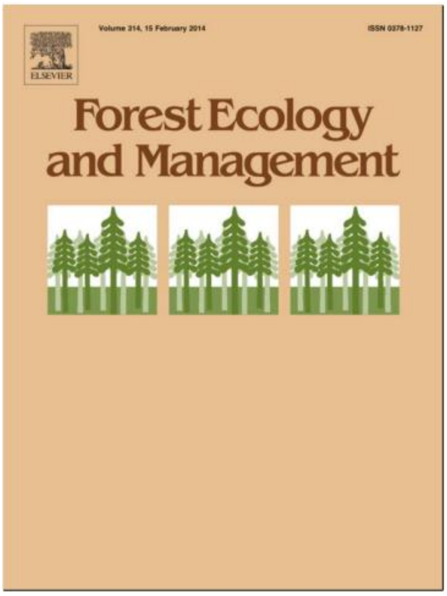
Vieira, D.C.S., 2008. Cartografia de risco de erosão, à escala de encosta, numa área recentemente ardida na região Centro. M.Sc. Thesis, University of Aveiro: Portugal.

Vigiak, O., Okoba, B.O., Sterk, G., Groenenberg, S., 2005. Modelling catchment-scale erosion patterns in the East African Highlands. Earth Surface Processes and Landforms 30: 183–196.

Wagenbrenner, J.W., MacDonald, L.H., Rough, D., 2006. Effectiveness of three post-fire rehabilitation treatments in the Colorado Front Range. Hydrological Processes 20: 2989–3006.

Wischmeier, W.H., Smith, D.D., 1978. Predicting rainfall-erosion losses- a guide to conservation planning. Agriculture Handbook N-537. USDA: Washington, DC; 58.

III.II Improving runoff and erosion predictions in burnt forest using the revised Morgan-Morgan-Finney model



Modelling runoff and erosion, and their mitigation, in burned Portuguese forest using the revised Morgan–Morgan–Finney model.

Vieira D.C.S., Prats S.A., Nunes J.P., Shakesby R.A., Coelho C.O.A., Keizer J.J. (2014) Forest Ecology And Management. 314, 150-165.

The revised Morgan–Morgan–Finney (MMF) model was used as a modelling approach, which has performed reasonably well to estimate soil losses for burned areas in humid Mediterranean forests in Portugal, and NW Spain. Simple model enhancement approaches are applied to recently burned pine and eucalypt forested areas in north-central Portugal and to subsequent post-wildfire rehabilitation treatments. Model enhancement is validated by applying it to another similar burned area to evaluate model calibration robustness and wider applicability. Model modifications involved: (1) focusing on intra-annual changes in parameters to incorporate seasonal differences in runoff and erosion; and (2) inclusion of soil water repellency in runoff predictions. The main results were that following wildfire and mulching in the plantations: (1) the revised model was able to predict first-year post-fire plot-scale runoff and erosion rates (NS_{Runoff} = 0.54 and $NS_{Erosion}$ = 0.55) for both forest types, and (2) first year predictions were improved both by the seasonal changes in the model parameters (NS_{Runoff} = 0.70 and $NS_{Erosion}$ = 0.83); and by considering the effect of soil water repellency on the runoff (NS_{Runoff} = 0.81 and $NS_{Erosion}$ = 0.89), (3) the individual seasonal predictions were considered accurate (NS_{Runoff} = 0.53 and $NS_{Erosion}$ = 0.71), and the inclusion of the soil water repellency in the model also improved the model at this base (NS_{Runoff} = 0.72 and $NS_{Erosion}$ = 0.74). The revised MMF model proved capable of providing a simple set of criteria for management decisions about runoff and erosion mitigation measures in burned areas. The erosion predictions at the validation sites attested both to the robustness of the model and of the calibration parameters, suggesting a potential wider application.

Introduction

Wildfires are a natural phenomenon in regions with a Mediterranean-type climate (Naveh, 1990). However, the present-day widespread occurrence of fires in southern Europe is unprecedented and strongly reflects human activity, not only directly through ignition (Veléz, 2009) but also indirectly through land/use changes such as land abandonment and widespread introduction of highly flammable pine and eucalypt plantations (Moreira et al., 2009; Shakesby, 2011). On average, wildfires consume each year 500,000 ha in southern Europe (San-Miguel and Camia, 2009), 100,000 ha of which in Portugal (Pereira et al., 2006a).Wildfire occurrence in Portugal is also not expected to decline markedly in the foreseeable future, both because of the economic importance of the country's forestry activities using flammable species and of the likely increase in meteorological conditions conducive to wildfires (Carvalho et al., 2010; Pereira et al., 2006b; Harding et al., 2009).

Wildfires are widely regarded as an important cause of increased runoff and soil erosion, and hence, land degradation in Mediterranean forests and woodlands, even though there remains considerable uncertainty about the long-term and landscape-scale impacts (e.g. Cerdà and Doerr, 2007; Shakesby and Doerr, 2006; Shakesby, 2011).This also applies to Portugal, where the degradational effects of post-fire land-use practices have equally been highlighted (Shakesby et al., 1993, 1996; Walsh et al., 1992, 1995; Ferreira et al., 2005, 2008; Malvar et al., 2011, 2013; Martins et al., 2013; Prats et al., 2012,2013).Fire-enhanced runoff and erosion are commonly attributed to the (partial) removal of the protective soil cover of vegetation and litter, in combination with heating-induced changes in soil properties such as aggregate stability (e.g. Varela et al., 2010; Mataix-Solera et al., 2011) and soil water repellency (SWR)(e.g.Scottetal.,1998).SWR is widely reported in burned forest soils (e.g. Wells, 1981; Vega and Díaz-Fierros, 1987; Prosser, 1990;Walsh et al., 1994;Keizer et al., 2008a) but is also commonly found in unburned soils (e.g. Imeson et al., 1992; Arcenegui et al., 2007; Martínez-Zavala and Jordán-López, 2009; Jordán et al., 2010; Keizer et al., 2005a). Although SWR can be induced and enhanced by wildfire (DeBano, 2000; Doerr et al., 2000; Doerr and Moody, 2004), the principal consequence of fire seems to be that SWR becomes geomorphologically 'activated' (Doerr et al., 1996; Doerr, 1998; Shakesby et al., 2000; Keizer et al., 2005b).

Many authors have investigated the relationships between SWR and soil moisture content and/or antecedent rainfall and overland flow response (Doerr and Thomas,2000;Doerr et al.,2003;Ferreira et al., 2005; Keizer et al., 2005a; Malvar et al., 2011; Santos et al., 2013). Apparently, the predominant runoff generating process can shift from saturation-excess to Hortonian overland flow when pre-storm soil conditions change from moist and wettable to dry and repellent (Doerr et al., 2003). In water repellent soils, the common assumption that infiltration capacity is inversely related to soil moisture content does not apply. Depending on the degree of water repellency, infiltration capacity is reduced for soil moisture contents below a critical threshold (Dekker and Ritsema, 1996) and often increases as soils become wet (Burch et al., 1989; Imeson et al., 1992; Doerr et al., 2003).

The effect of wildfires of increasing runoff and erosion has created a strong demand for a model-based tool for post-fire sediment loss prediction. Post-fire erosion prediction has been a research target by a number of authors (Benavides-Solorio and MacDonald, 2005; Díaz-Fierros et al., 1987; Fernández et al., 2010a; Larsen and MacDonald, 2007; Moody et al., 2008; Soto and Díaz-Fierros, 1998), and, in the case of Portugal, by the EROSFIRE-I and -II projects (Keizer et al., 2008b; Vieira et al., 2010). A variety of erosion models originally developed for agricultural areas have been applied to burnt areas. They range from simple empirical models such as the Universal Soil Loss Equation (USLE; Wischmeier and Smith, 1978) and the Revised Universal Soil Loss Equation (RUSLE; Renard et al., 1997), to semi-empirical models such as the revised Morgan–Morgan–Finney (MMF) model (Morgan, 2001) and the WEPP-based Erosion Risk Management Tool (ERMiT; Robichaud et al., 2007), and to process-based models such as the Water Erosion Prediction Project (WEPP; Nearing et al., 1989) and the Pan European Soil Erosion Risk Assessment (PESERA; Kirkby et al., 2008). Besides for evaluating post-fire erosion risk, soil erosion models have elevated potential for assessing the medium- to long-term impacts of fire as a landscape-disturbance and soil degradation agent, providing a welcome complement to the field studies that typically involve monitoring at small spatial scales and over short periods (Esteves et al., 2012; Shakesby, 2011).

The ERMiT tool deserves special mention as it has been developed as an operational tool for decision support in post-fire land management in (parts of) the USA (Robichaud et al., 2007). It allows predicting erosion risk during the early stages of the window-of-disturbance and, at the same time, the reduction of this risk by selected erosion control measures. This complementary information enables forest managers to evaluate the impact of fire on site productivity and the potential benefits of rehabilitation treatments (Larsen and MacDonald, 2007), and helps to formulate scenarios of erosion mitigation treatments to reduce the probabilities of high sediment yields (Robichaud et al., 2007). The ERMiT tool, however, has not been tested for post-fire conditions in

Portuguese or the Mediterranean in general. The need for testing and, in many cases adjusting existing models to local conditions is generally accepted (Shakesby, 2011). For example, Esteves et al. (2012) applied PESERA to post-fire conditions in central Portugal, and recommended that future applications would highlight factors such as SWR, the (temporary) presence of an ash layer and stone content (which is often high in the mountain soils in north-central Portugal).

The authors have been focusing their post-fire erosion modelling efforts on the revised MMF model (Morgan, 2001), as a relevant development compared to (R)USLE while maintaining much of (R)USLE’s ease-of-application, especially in comparison to process-based models with their elevated model input requirements. Furthermore, the revised MMF model has shown considerable promise for predicting soil losses in recently burnt woodlands in the humid Mediterranean climate region of the Western Iberian Peninsula (Fernández et al., 2010a; Vieira et al., 2010). It is a semi-empirical model that was originally developed for predicting annual soil loss from field-sized areas on hillslopes (Morgan, 2001). While MMF inherited many concepts of USLE (Wischmeier and Smith, 1978), its conceptualisation aimed at improving USLE physical basis by separating the soil erosion process into a water phase and a sediment phase (Figure 24). The water phase determines the energy of the rainfall available to detach soil particles from the soil mass as well as the volume of runoff; the sediment phase determines the rates of soil particle detachment by rain splash and runoff as well as the transporting capacity of the runoff volume. Runoff in MMF is estimated based on the method proposed by Kirkby (1976) which assumes that runoff occurs when the daily rainfall exceeds the soil moisture storage capacity and that daily rainfall amounts approximate an exponential frequency distribution (Morgan, 2001, 2005). The transport capacity of this runoff is then determined through a simplification of the scheme described by Meyer and Wischmeier (1969). MMF can easily accommodate soil conservation practices in its different phases. For example, agronomic measures can be simulated through the changes they produce in evapotranspiration, interception and crop management, which, in turn, affect the volume of runoff, the rate of detachment and the transport capacity, respectively (Morgan, 2005).

The overall aim of this study was to apply the revised MMF, testing simple enhancements of the model for recently burned pine and eucalypt forest in north-central Portugal. These model enhancements involved: (1) implementing seasonal changes in model parameters, in order to accommodate seasonal patterns in runoff and erosion as had been measured in the field trail; and (2) incorporating the role of SWR in overland flow generation, taking into account the findings of various post-fire hydrological/erosion studies in (north-) central Portugal (Walsh et al., 1994, 1995; Ferreira et al., 2005, 2008;

Esteves et al., 2012; Prats et al., 2012; Malvar et al., 2011, 2013). Worth stressing is that SWR has rarely (if ever) been incorporated explicitly in the modelling of post-fire runoff and erosion. These model enhancements were applied to two independent data sets collected by Shakesby et al. (1996) and Prats et al. (2012) at comparable sites at nearby locations but burnt and studied more than two decades apart. The data set of Prats et al. (2012) was used to calibrate the enhanced model, whilst the data set of Shakesby et al. (1996) was then used to validate it. The enhanced MMF model was evaluated to predict runoff and erosion following fire as well as following the application of mulching, a post-fire emergency treatment that both studies found to be highly effective.

Materials and methods

Study areas and sites

The two study areas in north-central Portugal where Shakesby et al. (1996) and Prats et al. (2012) collected their data sets were located near the villages of Falgarosa (40º 32'N, 8º 22' W) and Lourizela (40º 38' N, 8º 19' W), in the Águeda municipality, and near the village of Pessegueiro do Vouga 40º 43' 05''N; 8º 21' 15'' W), in the Sever do Vouga municipality, respectively. The former, validation data set concerned two sites covered by a pine plantation and a eucalypt plantation that burnt in 1991 and 1992, respectively; the latter, calibration data set concerned two nearby sites planted with pine and eucalypt that both burnt in 2007,in a single wildfire. A characterisation of wildfire severity at the four sites is given in Table 15.

Field experimental design and data collection

Calibration sites

Prats et al. (2012) used erosion plots to study the effectiveness of forest residue mulching in reducing post-fire runoff and erosion in their pine as well as a eucalypt site. In total, 12 experimental plots (8 m long x 2 m wide) were installed immediately after the wildfire and before any appreciable rain, 8 of which at the eucalypt site and 4 at the pine site. The plots were left untreated during most of the autumn (pre-treatment period) in order to assess the variability in erosion rates between the individual plots. However, the present study focused on the post-treatment period or, more specifically, the first year following mulch application. Mulch was applied to four randomly selected eucalypt plots and two randomly selected pine plots. This was done in December 2007, applying chipped eucalypt bark to the eucalypt plots at a rate of 0.87kg m^{-2} and, achieving a ground cover of 67%, while applying eucalypt logging residue to the pine plots at a rate of 1.75 kg m^{-2} and achieving a ground cover of 76%. Worth stressing is that the untreated pine plots had a markedly higher ground cover than the untreated eucalypt plots (60 vs. <10%) and, at the same time, a similar ground cover as the treated pine plots (Table 15).

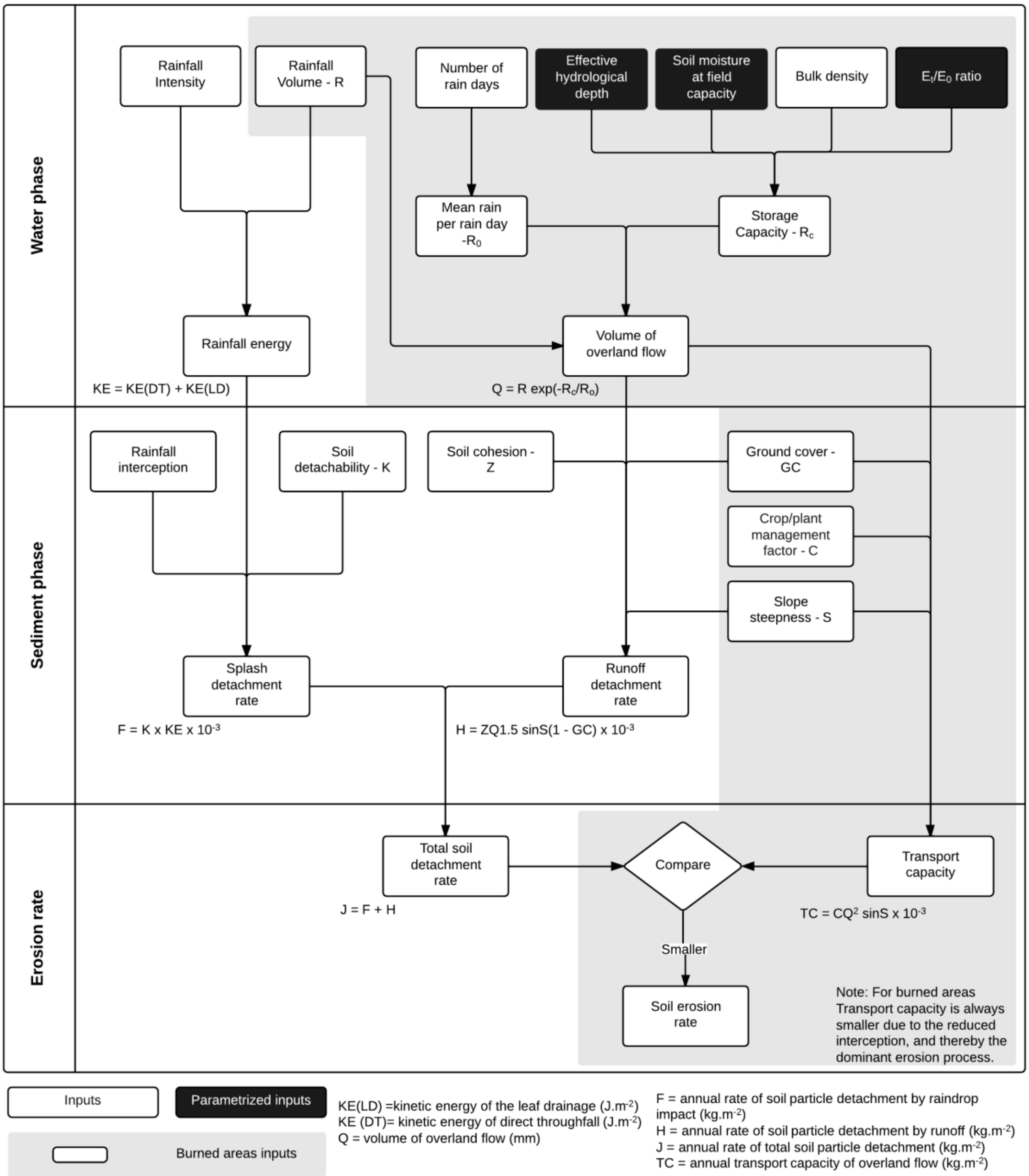


Figure 24 - Simplified flow chart of the revised Morgan–Morgan–Finney model, showing the key equations for the different model phases (adapted from Morgan, 2005).The boxes in black indicate the parameterized model inputs, whereas the grey area indicates model inputs considered when applied to post-fire conditions.

Table 15 – Main characteristics of the study sites

	Calibration sites		Validation sites	
	Eucalypt	Pine	Eucalypt	Pine
Source of measurement data	Prats et al. (2012)		Shakesby et al. (1996)	
Forest plantation type	Eucalyptus globulus Labill.	Pinus pinaster Ait.	Eucalyptus globulus Labill.	Pinus pinaster Ait.
Wildfire characteristics				
Date	August 2007	August 2007	August 1992	July 1991
Fire severity	Moderate	Low	Severe burn Understorey	Severe burn Understorey
Consumption of tree canopy	Total	Partial	Partial	Partial
Post-fire litter cover (%)	< 10	60	90	8
(Top-)soil properties				
Soil type (FAO, 1988)	Umbric Leptosol (25-30 cm depth)		Umbric Leptosol (≤ 40cm depth)	
Bulk density (0 –15 cm)(g m-3)	0.95 - 1.26	1.17 - 1.35	0.85 - 1.29	0.73 - 99
Texture class (0 –15 cm)	Sandy loam		Silt loam	

Rainfall was monitored continuously using two tipping-bucket rain gauges, and weekly using seven standard gauges. Runoff was monitored continuously using tipping-bucket devices, from where it was directed to storage tanks and measured weekly as a check. Eroded sediments were collected with modified Gerlach traps (Gerlach, 1967), whilst suspended sediment losses were determined using runoff samples collected from the storage tanks at weekly intervals. Ground cover on the plots was determined every 15 days from October 2007 to June 2008, and again at the end of the study period. This was done using a 100 cm2 quadrat, and identifying four cover categories: stones, bare soil (including ash), litter and vegetation. Soil moisture was measured weekly using permanent access tubes, into which a TDR-type Delta-T® PR2-probe was inserted to carry out readings at different soil depths (0–10 cm); no soil moisture data could be collected in the pine control plots due to a malfunction. SWR was measured weekly at three different soil depths (0–5 cm, 5–10 and 10–20cm), using the Molarity of Ethanol Droplet (MED) test. The methods, materials and data collection are described in more detail in Prats et al. (2012).

Validation sites

Shakesby et al. (1996) used erosion plots of the same dimensions and materials as Prats et al. (2012) but assessed the effectiveness of a variety of post-fire land management strategies: application of different quantities of logging litter, removal of pine needles (in order to assess their protective effect), rip-ploughing (deep ploughing achieved with long steel tines dragged behind heavy plant machinery used to clear old eucalypt or pine stumps in preparation for planting eucalypt seedlings), and minimum tillage prior to planting eucalypt seedlings. In total, 13 plots were installed, of which 6 at the eucalypt site and 7 at the pine site. The plots were installed immediately after the wildfires but were only treated one year (eucalypt site) and two years (pine sites) afterwards, in July 1994. The present study is limited to the first year of the post-treatment monitoring period. The treatments with logging litter were defined based on the collection and weighing of the waste slash that was left on the ground in 100 m^2 squares after the felling and removal of the trees at an unburned eucalypt stand as well as an unburned pine stand. Three application rates were selected that corresponded to 100% ('high'), 50% ('medium') and 10% ('low') of the mean weight per unit area of eucalypt/pine waste slash. The pine needle treatment involved a negative treatment, i.e. the removal of the pine needle cast that had fallen on the pine plot after the fire; in addition, all vegetation was clipped at the soil surface and then removed ('bare soil').

At the eucalypt site, the following treatments were applied: high litter (1 plot), medium litter (1 plot), low litter (2 plots) and untreated (2 plots). At the pine site, the litter treatments comprised only two waste slash categories (because the amount of waste was relatively low): high litter (100%, 2 plots) and medium litter (50%, 1 plot). The treatments further included: bare soil following needle removal and vegetation clipping (1 plot), minimum tillage and seedling planting (1 plot), and untreated (2 plots).

Rainfall was measured using standard rain gauges installed at the plot locations. Daily and hourly rainfall intensities were extrapolated from continuous records from tipping-bucket gauges installed at Falgueirinho (altitude, 460 m) and Castanheira (200 m). The plots themselves were all linked to a sediment trap, tipping bucket flow recorder and a series of large collecting tanks (Shakesby et al., 1991; Walsh et al., 1995). Eroded sediments were collected in these traps, while suspended sediment losses were determined using runoff samples collected from the tanks. Overland flow, however, was neither measured continuously nor for the entire sampling period, so that MMF model assessment was limited to erosion rates The vegetation cover in the plots was estimated in June 1993 and July 1994, using mosaics constructed with vertical photographs (taken from a 3-m-high movable frame) and directly in the field (details in Coelho et al., 1995).

Application of revised Morgan–Morgan–Finney model

The revised MMF model (hereafter referred to as MMF) was first applied to the untreated and treated, eucalypt and pine plots at the calibration sites studied by Prats et al. (2012). The plots will hereafter be designated as EC (eucalypt control), ET (eucalypt treated), PC (pine control), PT (pine treated) and (Figure 25). MMF was applied at two time scales: the entire 1-year post-treatment period from December 2007 to December 2008 and the individual seasons, hereafter designated as FP (full-period modelling) and SM (seasonal modelling).

Table 16 gives an overview of the model input parameters as well as of their values for the FP modelling and the methodology used to arrive at these values. From the 16 model inputparameters, 9 were available from Prats et al. (2012), whereas the remaining 7 had to be estimated from literature. The parameters can be divided in four different datasets (Figure 25):

- a universal dataset, comprising the parameters that have the same values for all 12 plots, i.e. R, Rn, I, EHD, COH and A;
- a land-use dataset, comprising the parameters with distinct values for the eucalypt plots and for the pine plots, i.e. BD, K, Et/E0, C, CC and PH;
- a treatment dataset, comprising the MS parameter and reflecting the observed differences in soil moisture contents on the mulched vs. control plots;
- a plot-specific dataset, comprising the parameters with differed values for each of the individual plots, i.e. S, P and GC.

The full-period (FP) modelling approach involved the following assumptions: Et/E0 and GC could be represented by their mean values over the full post-treatment period, and MS could be approximated by the maximum soil moisture content recorded during the 1-year study period. In the case of the pine site, MS could only be estimated directly from measurement data for the mulched plots, due to the above-mentioned malfunctioning of soil moisture sensors in the control plots. however, it was decided to use the same MS values in both plots Since the control pine plots presented a similar ground cover as well as a similar overland flow responses as the treated pine plots (Prats et al., 2012), MS was assumed to be the same at the control than untreated pine plots.

Table 16 – Model input values (values or range of values) used in the full period MMF application (FP) for the control and treated plots at the eucalypt and pine calibration sites.

Factor	Parameter	Eucalypt		Pine		Methodology applied to input determination
		control	treated	control	treated	
Rainfall	R (mm yr^{-1})	1684				Recorded at the study sites by Prats et al. (2012); the rainfall kinetic energy was then calculated following the procedure outlined by Coutinho and Tomás (1995).
	Rn (mm_{rain} day^{-1})	9.8				
	I (mm h-1)	30				
Soil	MS (%)	20	28	35		Derived from measurements by Prats et al. (2012).
	BD (g cm-3)	1.1		1.2		
	EHD (m)	0.09				Estimated following Morgan (2001), considering “bare soil without a surface crust”, as the best approximation to a burned bare soil.
	K (g J-1)	0.7		0.5		Estimated on base of the measured soil texture class, following Morgan (2001).
	COH (kPa)	2				
Landform	S (º)	22-26	22-29	22-22	24-28	Measured in the field by Prats et al. (2012).
Land cover	A	0.1				Based on the findings of Ferreira (1996) for newly burnt eucalypt stands in north-central Portugal.
	Et/E0	0.7		0.6		Computed with the Thornthwaite-Mather (1957) method, using temperature and climate data from the nearest weather station at Castelo-Burgães (SNIRH, 2011) and using the measured rather than calculated soil moisture values to estimate Et.
	C(a)	0.02		0.005		Estimated using the RUSLE methodology (Renard et al. 1997), as was done by Larsen and MacDonald (2007) and Fernández et al. (2010), and taking into account estimates of soil biomass, surface roughness and ground cover.
	P(a)	0.6-0.9	0.3-0.5	0.3-0.4	0.2-0.3	Calculated from the measured cover values using the equation P = 1 - (GC/100), as justified by the negative relationship between soil organic matter cover and cumulative soil losses reported by Fernández et al. (2004).
	CC (%)	1		5		Measured in the field by Prats et al. (2012).
	GC (%)	6-40	54-69	56-66	74-83	
	PH (m)	0		0		

[a] In the original input table (Morgan, 2001), the C factor represented a combination of C and P parameters, which are presented separately here.

Seasonal modelling (SM) involved estimating the seasonal (winter, spring, summer, autumn) variations in the following four input parameters: soil moisture at field capacity (MS) (Figure 26a); evapotranspiration (Et/E0) (Figure 26b); land cover (GC, C) (Figure 26c); soil effective hydrological depth (EHD) (Figure 26d). The seasonal changes in EHD were meant to represent the observed changes in ground cover (GC), reflecting post-fire vegetation and litter recovery, on the one hand, and, on the other, the decomposition and erosion of the applied mulch at the treated plots. To this end, EHD was estimated as a linear function of GC, such that the EHD for a GC of 0% corresponded to that of shallow soils on steep slopes (EHD =0.05 m; Morgan, 2001) and the EHT for a GC of 100% to that of a mature forest (EHD =0.20 m; Morgan, 2001).

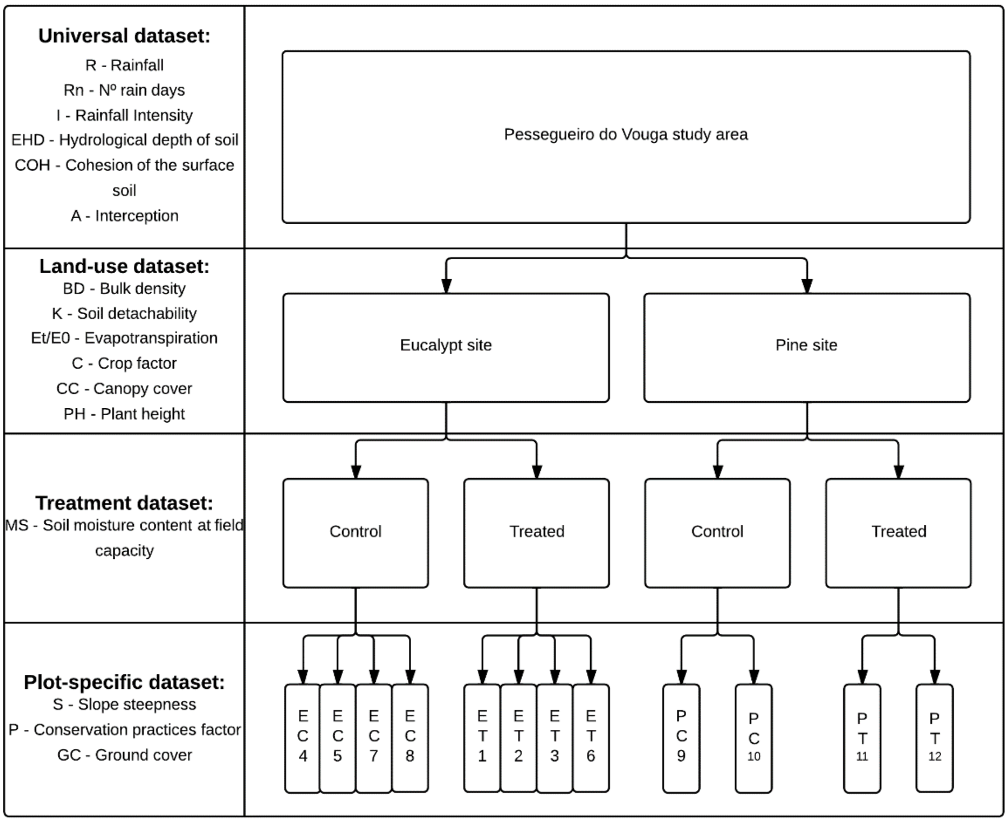


Figure 25 - Overview of the model input datasets used in applying MMF to the plots of Prats et al. (2012). For more details on model, please see Table 16 and/or Morgan (2001); plots are numbered according to their position on the study slope (left to right).

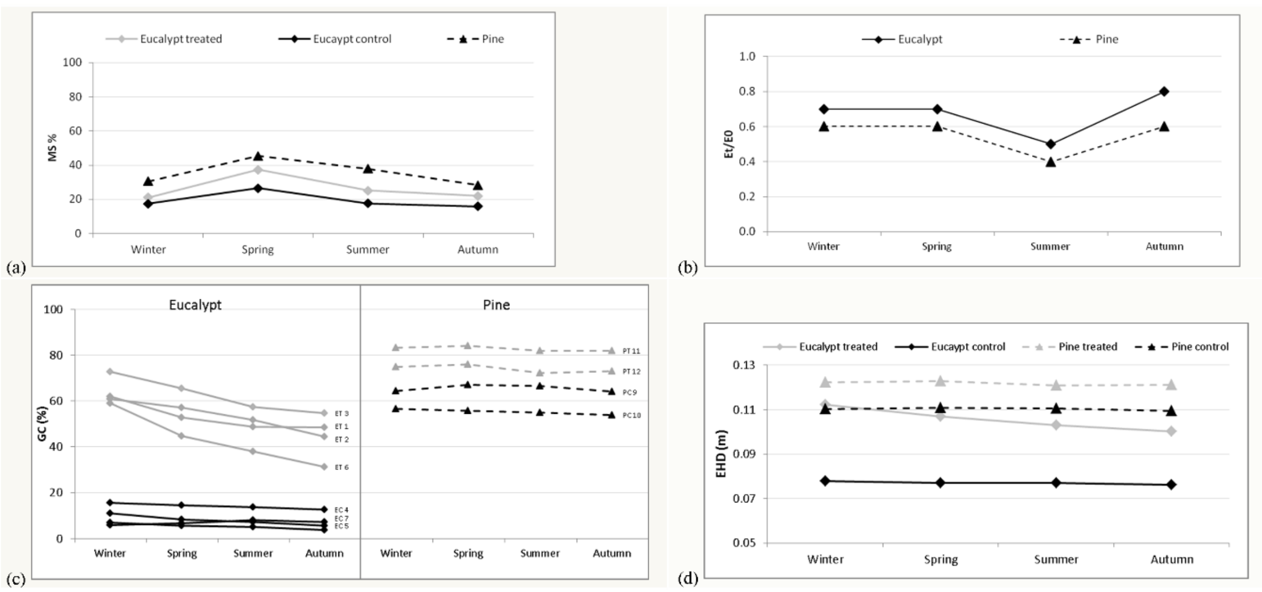


Figure 26 - Seasonal variations in four MMF parameters at the eucalypt and pine calibration sites: (a) soil moisture content at field capacity (MS, %); (b) evapotranspiration (Et/E0); (c) ground cover (GC, %) and (d) hydrological depth of soil (EHD, m).

The hydrological effects of SWR in MMF could only be simulated through calibration of soil storage capacity (Rc, Figure 24), decreasing it with increasing level of SWR. Soil storage capacity is determined by four parameters: Field Capacity (MS), Effective Hydrological Depth (EHD), Bulk Density (BD) and the evapotranspiration ratio (Et/E0). Given this limited choice of adjustable parameters and given the fact that EHD already needed to be calibrated for the seasonal modelling, MS seemed the most appropriate parameter for mimicking the effects of SWR. To this end, the ‘‘SM-SWR’’ modelling approach consisted of multiplying the MS value with a scalar that decreased with increasing severity SWR, ranging from 0.8 under extremely repellent conditions to 1.1 under wettable conditions (Table 17). The exact values of these multipliers were determined to be those that provided the best model performance. In the model runs, the scaling of MS is achieved by inferring the severity classes of SWR from the inter-annual variations in soil moisture, since the original version (Morgan, 2001) at annual scale is intended to represent both repellent and non-repellent conditions.

Table 17 –Multiplication factors used to parameterize soil moisture at field capacity (MS) in the SM-SWR modelling approach for different repellency severity classes.

Repellency severity class (Adapted from Keizer et al., 2008b)	Ethanol class	MS multiplication factor
Extreme	8	0.8
Very strong	7 – 6	0.9
Slight to strong	3-5	1
None	0-2	1.1

After application to the calibration sites, MMF was applied to the validation sites studied by Shakesby et al. (1996). This was done using a similar procedure as described above and selecting values for the calibration parameters as close as possible to those used for the data set of Prats et al. (2012), in the following manner:

- when measurements were available, parameter values were calculated directly, i.e. for of R, Rn and the plot-specific dataset (S, P, GC);
- from the unmeasured parameters, MS, BD, K and COH were inferred from Morgan (2001), based on the soil texture data available for the validation site;
- for the remaining unmeasured parameters (I, EHD, A, Et/E0, C, CC, PH), the same values were used as for the calibration sites, differentiating between eucalypt and pine plots; the values of the C factor, however, were estimated anew for the plots whose treatment that had not been

tested by Prats et al. (2012), i.e. minimum tillage and removal of pine needle cast;

- the seasonal variation in MS and EHD were calculated in the same way as for the calibration sites, but the seasonal patterns in Et/E0 and SWR were assumed to be the same as at the calibration sites, in view of the lack of soil moisture data for the plots studied by Shakesby et al. (1996).

Model assessment

Model performance was evaluated using two commonly-used assessment indicators (e.g. Moriasi et al., 2007): the coefficient of efficiency (NS: Nash and Sutcliffe, 1970) and the root mean squared error of the observed vs. predicted values (RMSE). The NS is a descriptor of the predictive accuracy of model outputs. It can range from -∞ to 1. A negative value indicates that the mean observed value is a better predictor than the model output. A value of 0 indicates that the mean observed value is as accurate a predictor as the model output, and an efficiency of 1 corresponds to a perfect match of the predicted with the observed values. NS values greater than 0.5 are widely considered to indicate satisfactory model performance (Quinton, 1997), while values exceeding 0.7 should not be expected (Nearing, 1998; Morgan, 2001).

Results

Seasonal calibrated parameters

Before presenting the modelling results themselves, further insight in the values attributed to the parameters varying seasonally was deemed helpful for a better grasp of the differences in model output between the full-period (FP) and seasonal (SM and SMSWR) modelling approaches. Each of these seasonal parameters influences how runoff varies with the various seasons. Following calibration, evapotranspiration (Et/E0) revealed marked seasonal differences (Figure 26b). These differences produced a reduction in water available for runoff generation during the dry seasons and an increase during the wet seasons. The Et/E0 parameter also contributed to runoff differences between the pine and eucalypt plots.

The ground cover parameter (GC, %) allowed accommodating the temporal patterns in cover and to do so for each plot individually (Figure 26c). By contrast, the FP approach involved using the mean ground cover over the entire 1-year study period, thereby ignoring the sometimes substantial changes in, for example, mulch cover (through decomposition and erosion) or in litter cover (due to pine needle cast). The seasonal variations in GC were used to produce changes in EHD with time (Figure 26d), avoiding the use of a single reference value (in particular, ‘‘bare soil without surface crust’’ from Morgan (2001)) for the entire post-treatment period.

Soil moisture at field capacity (MS, %) was estimated in the same manner for the individual seasons as it was for the full post-treatment period. Following calibration of MS to mimic SWR patters, MS values revealed a clear increase from winter to their maximum in spring and again a decrease until autumn (Figure 26a). Furthermore, MS values provided a distinction between pine and eucalypt plots and, at the same time, between control and treated eucalypt plots.

Annual runoff rates at the calibration sites

Overall, the application of MMF enabled the distinction between the three contrasting magnitudes of annual runoff volumes as produced by the pine plots, the treated eucalypt plots and the untreated eucalypt plots. Predictions with acceptable accuracy were obtained with the FP as well as the two seasonal modelling approaches.

The field measurements by Prats et al. (2012) showed a marked difference in annual runoff produced by the untreated eucalypt plots and the untreated pine plots, with consistently higher runoff amounts at the eucalypt (EC, 466 mm) than pine site (PC, 93 mm). Treatment effectiveness was high at the eucalypt site but inexistent at the pine site, with a decrease in average runoff of 43 % as opposed to an increase of 10%. The observed difference in average annual runoff between the mulched and untreated plots was predicted by all three modelling approaches in the case of the eucalypt plantation, but not in the case of the pine plantation. The seasonal approaches predicted less instead of more runoff for the treated (PT) and untreated (PC) pine plots, whereas the FP approach gave identical runoff estimates with and without mulching. The annual runoff predictions for the individual plots were plotted against the observed values in Figure 27a. The three modelling approaches captured well the three observed levels of runoff generation (high = eucalypt untreated vs. medium= eucalypt treated vs. low =pine treated and untreated) but did not provide accurate estimates for all of the individual plots. Plot-specific runoff differences were not represented in the FP modelling approach, providing the same prediction for each plot at the same site and with the same treatment. They were, however, represented in both seasonal modelling approaches, through the seasonal parameters (see Section 3.1). The spatial variation in overland flow was predicted more accurately for the untreated (EC) than treated (ET) eucalypt plots, especially because one of the treated plots produced noticeably more runoff than predicted. At the same time, however, this plot produced clearly more runoff than the other ET plots.

In general, the original FP modelling approach performed reasonably well in predicting annual runoff, with a NS index of 0.54 and a RMSE of 121mm (Table 18). Both seasonal approaches clearly improved model performance. The seasonal component in the SM approach increased the NS index to 0.70 and reduced the RMSE by 20 %, whereas the inclusion of soil water repellency in the SM-SWR approach improved the NS index to 0.81 and decreased the RMSE by a further 20%.

Table 18 – Average amounts of annual runoff and erosion of the control and mulched plots at the eucalypt and pine calibration sites as measured by Prats et al. (2012) and as predicted using the FP, SM and SM-SWR modelling approaches. Model performance was assessed by means of the NS index and the RMSE.

			Predicted		
		Measured	Full period (FP)	Seasonal (SM)	Seasonal-SWR. (SM-SWR)
		Runoff (mm)			
Eucalypt site	Control	466	335	406	454
	Treated	267	185	253	297
Pine site	Control	93	120	117	124
	Treated	103	120	89	95
		Mean erosion (kg m^{-2})			
Eucalypt site	Control	0.541	0.799	0.411	0.629
	Treated	0.074	0.122	0.026	0.049
Pine site	Control	0.032	0.011	0.003	0.004
	Treated	0.037	0.007	0.001	0.001
		Model performance			
NS	Runoff	-	0.54	0.70	0.81
	Erosion	-	0.55	0.83	0.89
RMSE	Runoff (mm)	-	121	98	78
	Erosion (kg m^{-2})	-	0.18	0.11	0.09

Annual erosion rates at the calibration sites

The overall model performance regarding annual soil loss estimations, allowed differentiation between the three main levels of observed erosion rates, corresponding, in increasing order, with the pine plots, and the treated and the untreated eucalypt plots, respectively. Predictions of annual soil losses had higher accuracy than predictions of annual runoff volumes.

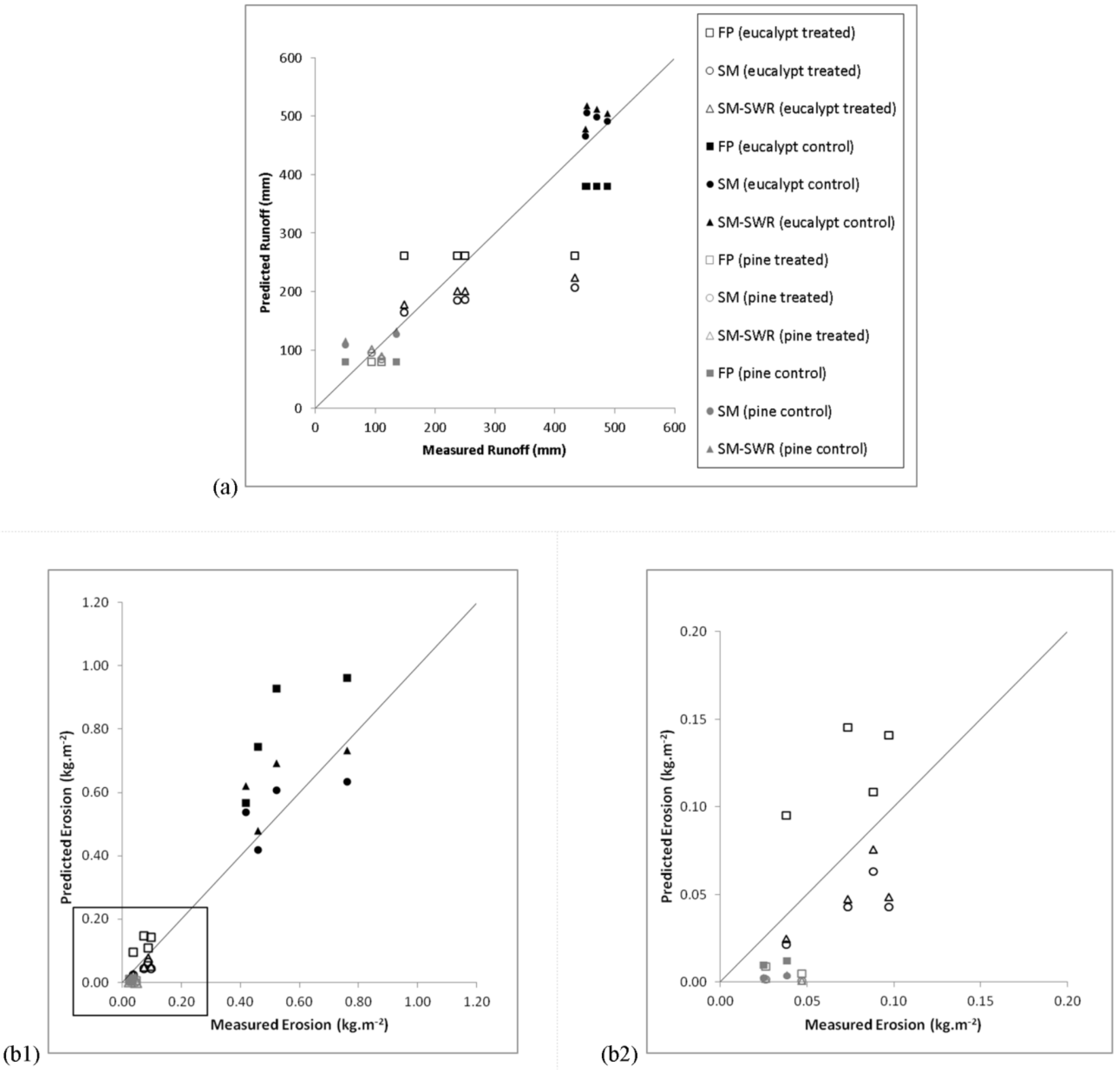


Figure 27 - Scatter plots of the measures vs. predicted annual runoff (a) and erosion (b1 and, zoomed-in, b2) values for the control and treated plots at the eucalypt and pine calibration sites, as measured by Prats et al. (2012) and predicted using the three modelling approaches.

Annual soil losses, like annual runoff amounts, revealed marked differences between the eucalypt and the pine site with a higher loss in the eucalypt (EC, 0.54kg m^{-2}) than in the pine (PC, 0.032 kg m^{-2}) plots. Treatment effectiveness in soil losses was improved in the eucalypt plots and worsted in the pine, comparably to the obtained runoff amounts. A decrease of 86% in erosion following mulching was observed for the eucalypt study site, while the control plots at the pine site produced 16% less erosion than the treated ones. The mulch effect was predicted by all the three approaches as effective either in eucalypt and pine sites, although the pine plot observations resulted in increased soil losses in the PT.

When individual plot predictions were plotted against the observed erosion results (Figure 27b1 and Figure 26b2), the same 3 levels were verified for erosion in similarity to the runoff (high, medium and low). In the case of the eucalypt site, observed soil losses for treated and control conditions were represented by all modelling approaches, improving overall model accuracy when compared to runoff predictions. However, in the pine site, this differentiation was not possible; due to a systematic underestimation of the observed soil losses for all the approaches, plus the prediction of less erosion in the PT instead of more in comparison to the PC, resulting in poor NS efficiencies for this specific land use.

The systematic overestimation of soil losses by the FP modelling approach was improved by introducing the seasonal component (SM), then it performed more accurately for the untreated than treated eucalypt plots. The second adaptation (SM-SWR) improved the predictions for all eucalypt plots, although the predictions for the treated plots continued to underestimate the observed values.

In general, the FP modelling approach can be considered accurate when predicting soil losses on a year basis, with a NS index of 0.55 and a RMSE of 0.18 kg m^{-2}. The seasonal approaches were able to improve model performance to a NS index of 0.83 and reduced RMSE by 40%, and, when incorporating SWR (SM-SWR), to further improve the NS index to 0.89 and further reduce the RMSE by 20%.

Seasonal runoff rates at the calibration sites

The seasonal pattern of mean observed runoff values was similar in all the sites, although the different levels of low (PC and PT), medium (ET) and high (EC) runoff amounts found. The observations reveal higher runoff amounts in winter and autumn, separated by a decrease in spring and the minimum in summer (Figure 28). These results justify by themselves the need of a seasonal modelling, since there was a pronounced variation of rainfall through the year (Prats et al. 2012), and the estimated parameters such as MS, EHD and Et/E0 reveal variations between seasons.

The eucalypt site presented the highest overland flow from this study (EC), with a runoff coefficient of 36% most of the year, with the exception of spring with 19%. While the pine site (PC) presentedrunoffamountsaround10%inthe winter, decaying dramatically to 2% in spring, increasing to 8%, followed by 7%, for summer and autumn periods. Prats et al. (2012) attributed these results primarily to the total rainfall amounts and secondly to the frequency of extreme repellence

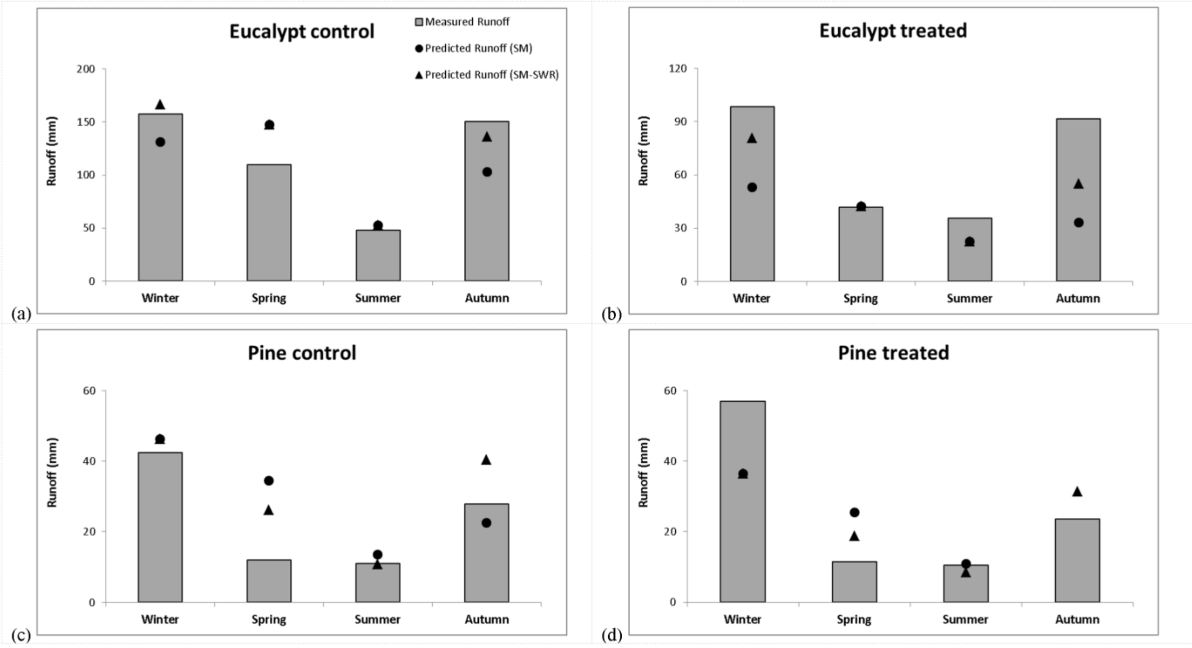


Figure 28 - Mean seasonal runoff amounts of the control and treated plots at the eucalypt and pine calibration sites, as measured by Prats et al. (2012) and predicted using the two seasonal modelling approaches (SM and SM-SWR). The legend of the symbols is given in graph (a); note the different scales for the pine and eucalypt plots as well as for the control than treated eucalypt plots.

Figure 28 compares the seasonal patterns of measured and predicted runoff. In the case of the seasonal approach (SM), the control eucalypt plots runoff amounts are underestimated in winter and autumn, then the predictions are overestimated mostly in spring and slightly in summer (Figure 28a). In the treated ones, by the other hand, present an underestimation of the runoff predictions for most of the year (Figure 28b). In the case of the control pine plots, runoff was overestimated from winter to summer and underestimated autumn (Figure 28c). While the treated ones the behaviour is approximated with the exception of the underestimated runoff amounts for the winter period (Figure 28d). The inclusion of SWR in the model (SM-SWR) caused the greatest improvements in winter and autumn for both eucalypt plots. For the pine runoff predictions (Figure 27c and d), these improvements are not so noticeable since the model predictions were improved in spring but became worse in autumn for both control and treated plots. The inclusion of SWR improved the model runoff predictions, as indicated by the NS index for all the plots increasing from 0.53 to 0.72, and the RMSE was reduced 40%. This improvement is also applied when model performance is evaluated for each land use and treatment, where all NS and RMSEs underwent improvement (Table 19). This was particularly true for the pine plots (and especially for the control), where model performance was raised from being poor to acceptable.

Seasonal erosion rates at the calibration sites

Following the same trend as the observed runoff, there are three levels of soil losses amounts, low for pine plots, medium for treated eucalypt plots and high for eucalypt control. However the soil losses behaviour does not correspond totally to the observed runoff, and also is not similar between site (eucalypt/pine) and treatment (control/treated) (Figure 29). From the modelling point of view the results in soil losses predictions differences depend mostly in predicted runoff amounts for these periods, plus the changes in ground cover (GC) as a protection agent.

The eucalypt site presented the highest soil losses means from this study (EC), with elevated erosion amounts in winter, spring and autumn (seasons with runoff coefficient around 36%). The minimum was verified in summer (0.03kg m-2) and the maximum in spring (0.21kg m-2) (Figure 29a). In the last case, the season not only had an elevated runoff coefficient, as also presented the highest rainfall events from the study period (Prats et al., 2012). In the case of the ET plots, some similarities in the erosion observations were found, more soil losses in autumn relatively to winter season, same minimum in summer. And the difference is basically the inexistence of the extreme soil losses verified in the spring season when compared to the untreated plots.

In the case of the pine site (Figure 29c), control plots present its maximum in autumn, and the minimum in spring, while the others presented intermediate values, but always with reduced soil losses under 0.012 kgm^{-2}. Similar figures were observed in the treated pine site in all the seasons (Figure 29d) with the exception of the winter season, the observed maximum, presenting a slightly increased amount of 0.015 kg m^{-2}.

The seasonal modelling approach (SM) when applied to eucalypt plots, either from control or treated ones, result in a systematic underestimation of soil losses amounts, although their differences in the magnitudes (high and medium). The EC plots present a more similar trend to the observed values in accordance to the obtained NS indexes, than the ET ones. In the pine sites either for control or treated, the soil losses amounts estimations are much reduced, and no trend is possible to be observed for comparison.

The inclusion of SWR in the revised MMF (SM-SWR), did not substantially improve the soil losses predictions, in contrast to the runoff amounts. This approach led to a slight increase of the soil losses in periods that had higher frequency of soil water repellency, resulting in some overestimation in the eucalypt control plots (Figure 29a) for winter and spring. In the overall evaluation, regarding soil losses predictions by land use, only the eucalypt site presented an accurate estimation (Table 19).

Table 19 – Model performance for seasonal runoff and erosion predictions at the calibration sites using the SM and SM-SWR modelling approaches. Model performance was assessed by means of the NS index and the RMSE.

		Seasonal (SM)		Seasonal-SWR (SM-SWR)	
			NS		
			Runoff predictions		
All plots		0.53		0.72	
Eucalypt site	Control	0.66	0.31	0.78	0.57
	Treated		0.44		0.55
Pine site	Control	0.09	-0.26	0.5	0.33
	Treated		0.64		0.69
			Erosion predictions		
All plots		0.71		0.74	
Eucalypt site	Control	0.64	0.41	0.66	0.45
	Treated		-3.16		-2.9
Pine site	Control	-0.69	-0.83	0.05	0.13
	Treated		-2.5		-1.08
			RMSE		
			Runoff predictions (mm)		
All plots		115		71	
Eucalypt site	Control	56	54	44	43
	Treated		14		13
Pine site	Control	59	58	44	42
	Treated		12		12
			Erosion predictions (kg m^{-2})		
All plots		0.15		0.09	
Eucalypt site	Control	0.09	0.09	0.09	0.09
	Treated		0.01		0.01
Pine site	Control	0.02	0.02	0.02	0.01
	Treated		0.01		0.01

In this site, the improvement was verified in both treatments, but the final NS index result showed that ET predictions were not improved enough to be considered accurate. The soil losses predictions are still far from accurate in the pine plots when NS index is used for comparison, however, is the RMSE shows the reduced error in sediment losses amounts. Overall, SM modelling approach can be considered accurate when predicting soil losses seasonally, with a NS index of 0.71 and a RMSE of 0.15kg m^{-2}. The inclusion of soil water repellency (SM-SWR) improved this performance to a NS index of 0.74 and a reduction of the RMSE by 40%.

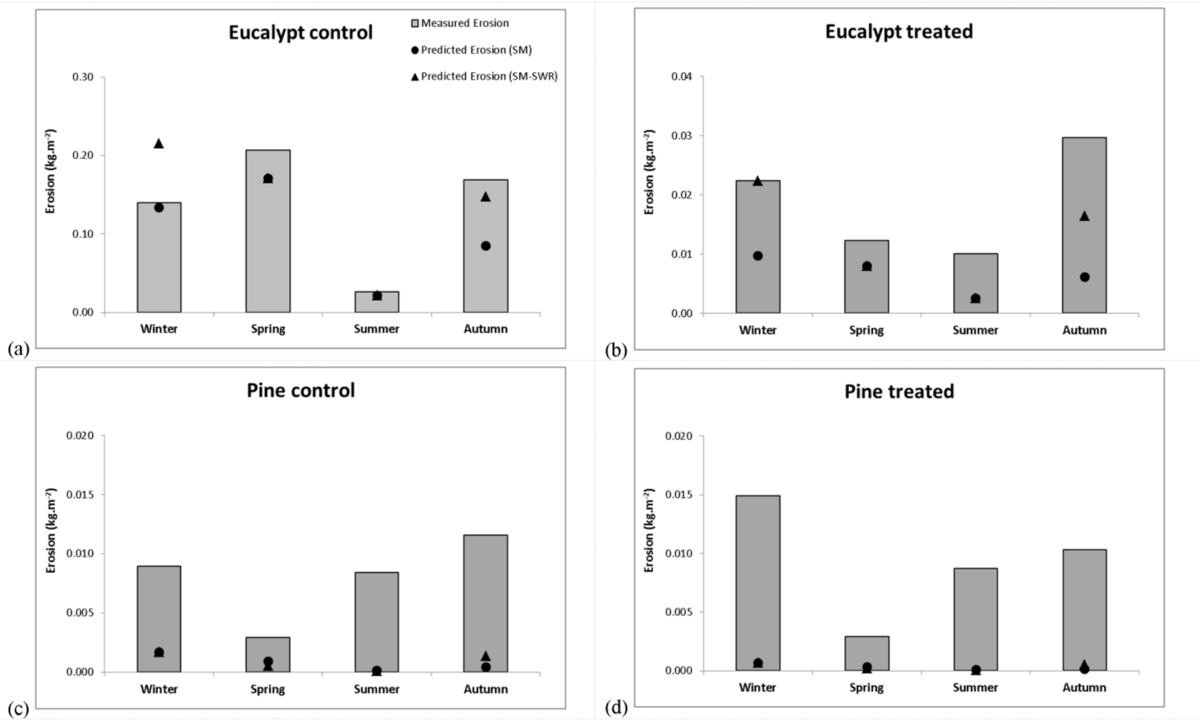


Figure 29 - Mean seasonal erosion rates of the control and treated plots at the eucalypt and pine calibration sites, as measured by Prats et al. (2012) and predicted using the two seasonal modelling approaches (SM and SM-SWR). The legend of the symbols is given in graph (a); note the different scales for the pine and eucalypt plots as well as for the control than treated eucalypt plots.

Annual and seasonal erosion rates at the validation sites

There is a marked difference in runoff and erosion amounts between the untreated plots for pine and eucalypt sites in the model validation results, which agrees with the field results for the validation site (Shakesby et al., 1996), as well as with the conclusions from the calibration test site (Prats et al., 2012). This difference between treated and untreated plots was identified in the model for the different modifications when annual soil losses were compared (Figure 30a). As far as the litter cover differences were explored, three soil losses amounts levels can be highlighted: high-medium litter (Figure 30b), low litter, and bare soil; these differences were also recognised by the model predictions (Figure 30a).

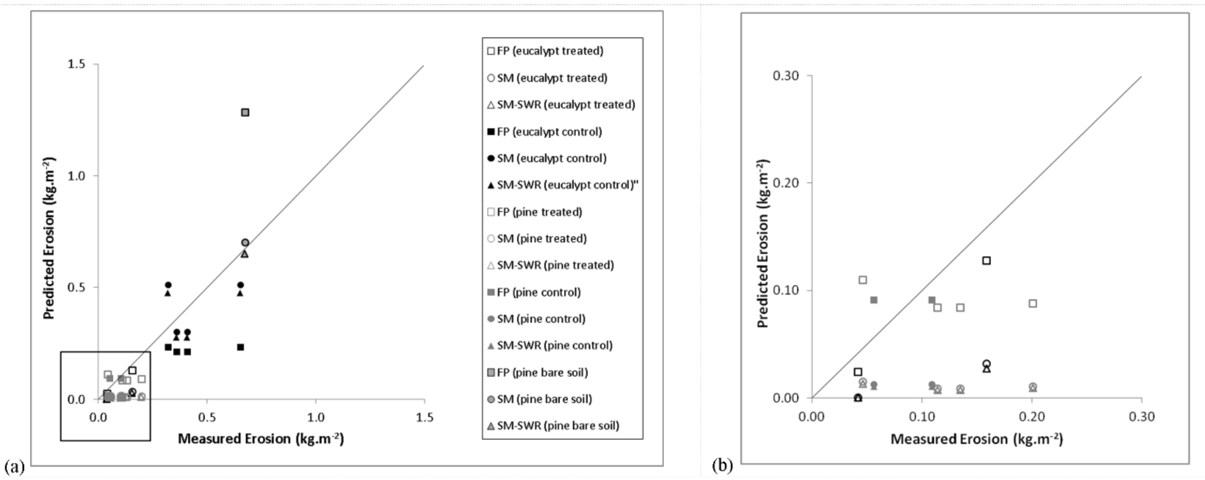


Figure 30 - Scatter plots of the measured vs. predicted annual erosion values (a and, zoomed-in, b) for the control and treated plots at the eucalypt and pine validation sites, as measured by Shakesby et al. (1996) and as predicted using the three modelling approaches (FP, SM and SM-SWR). Note the different scales for the pine and eucalypt plots as well as for the control than treated eucalypt plots.

Table 20 shows model efficiency as regards erosion rate predictions. FP modelling performed poorly compared with the seasonal modelling approaches (SM and SM-SWR). Negative NS efficiencies were obtained for the first, and similar efficiencies were obtained for seasonal modelling in comparison to the calibration results.

Table 20 – Model performance in predicting overall erosion rates at the calibration and validation sites using the three different modelling approaches. Model performance was assessed by means of the NS index and the RMSE.

Model efficiency for erosion predictions		Full Period (FP)	Seasonal (SM)	Seasonal-SWR (SM-SWR)
			NS	
Calibration	All plots	0.55	0.83	0.89
	Eucalypt site	0.38	0.70	0.82
	Pine site	-7.96	-9.58	-10.00
Validation	All plots	-0.08	0.81	0.84
	Eucalypt site	0.50	0.67	0.69
	Pine site	-0.20	0.90	0.93
			RMSE (kg m^{-2})	
Calibration	All plots	0.18	0.11	0.09
	Eucalypt site	0.18	0.12	0.09
	Pine site	0.02	0.02	0.02
Validation	All plots	0.22	0.11	0.12
	Eucalypt site	0.21	0.12	0.13
	Pine site	0.24	0.10	0.11

The model performed more accurately for pine than for eucalypt regarding the validation site, this situation was reversed in the calibration plots. This change occurred due to the very low erosion rates recorded on the pine plots (Prats et al., 2012), which required a prediction detail beyond model capacity.

In the validation pine plots (Shakesby et al., 1996), the presence of a ‘‘bare soil’’ treatment with high measured erosion rates provided a better test of model performance. Excluding this treatment, however, the model also performed poorly in distinguishing between control and treated pine plots, also due to the very low observed soil losses amounts, as can be seen in Figure 30b.

Overall, the original FP modelling approach performed poorly predicting erosion amounts, with a NS index of -0.08, and a RMSE of 0.22kg m^{-2} (Table 20). Seasonal approaches improved substantially the model performance. The seasonal component in the SM approach increased the NS index to 0.81 and reduced the RMSE by 50%, whereas the inclusion of soil water repellency in the SMSWR approach improved the NS index to 0.84 although increased the RMSE by 8%.

Between calibration and validation model applications, soil loss predictions enabled differentiation between the three main observed erosion groups for all the model approaches. However, only SM and SM-SWR in the validation site provided accuracy comparable to that obtained for the calibration plots.

Discussion

Model evaluation and comparison with other studies

Table 21 compares model accuracy for runoff and erosion predictions between this study and other approaches applied to burned areas. There have been few studies that have tested model accuracy for runoff predictions in recently burned forests and none was found for periods shorter than one year. The obtained efficiency for the annual runoff amounts predictions in this study, for full period modelling (FP, NS= 0.54), seasonal modelling (SM, NS= 0.70), and seasonal modelling with repellency calibration (SM-SWR NS = 0.81) are consistent with those of Soto and Díaz-Fierros (1998) (years 1–4) for prescribed burning and wildfire calculated using the WEPP model (Table 21). However, this comparison is limited, since high repellency periods were excluded by the authors while modelling with WEPP, and the period of study is much longer.

The accurate predictions of sediment losses achieved with the revised MMF in this study, for the FP, SM and SM-SWR modelling approaches (NS =0.55, NS = 0.83 and NS = 0.89, respectively) were consistent with those previously observed in burned areas in Portugal (Vieira et al., 2010) and NW of Spain (Fernández et al., 2010a). Model accuracy also agree with the results by Larsen and MacDonald (2007) for periods of 1–10 years after wildfire (Table 21); they were, however, better than the results for the first year after fire, for which the authors found negative efficiency indices for sediment yield when applying the standard and modified versions of RUSLE and disturbed WEPP. Also, model accuracy was better that one obtained by Soto and Díaz-Fierros (1998). It is important to mention that the prediction efficiencies with MMF achieved in the present study represent improvement over the previous MMF application by Fernández et al. (2010a). Finally, the results for accuracy from the present study were also in line with those generally obtained when applying the MMF model (as reported by Morgan (2001)).

Overall, comparison between the results obtained in the present study and those listed in Table 21 indicates that the accuracy in modelling annual runoff and erosion was similar to, or better than, that generally achieved with similar approaches for burned areas.

Table 21 – Overview of prior studies modeling erosion with MMF and/or modeling post-fire erosion

Study	Location	Land Use	Model	Burning conditions	Untreated/ treated	NS_{Runoff}	$NS_{Erosion}$	RMSE (mm)	RMSE (kg m^{-2})
Fernández et al. (2010)	NW of Spain	Forest (pine) and shrubland	RUSLE	Wildfire	Untreated	-	0.87[a]	-	0.63[a]
					Treated	-	0.33[a]	-	1.55[a]
			MMF	Wildfire	Untreated	-	0.74[a]	-	0.90[a]
					Treated	-	-0.59[a]	-	2.38[a]
Larsen and MacDonald (2007)	Colorado Front Range, USA	Forest (pine)	RUSLE	Wildfire	Untreated	-	0.52[b]	-	0.36[b]
			Modified RUSLE	Wildfire	Untreated	-	0.31[b]	-	0.43[b]
			Disturbed WEPP	Wildfire	Untreated	-	0.53[b]	-	0.35[b]
			Modified Disturbed WEPP	Wildfire	Untreated	-	0.65[b]	-	0.30[b]
Morgan (2001)	Various	Agriculture	MMF	Unburned	n.a.	0.58	0.65	-	-
					n.a.	0.94	0.84	-	-
Soto and Díaz-Fierros (1998)	NW of Spain	Shrubland	WEPP	Prescribed fire	Untreated	0.34[d]	0.61[d]	-	-
				Wildfire	Untreated	0.84[d]	0.03[d]	-	-

[a] Best achieved Nash Sutcliffe efficiency index and RMSE after RUSLE and revised MMF model modification.
[b] Statistics of grouped hillslope data, periods of 1 to 10 years after wildfire by Larsen and MacDonald (2007).
[c] From 67 sites data set within Foulam and Ødum in Denmark, El Ardal in Spain, Spata in Greece and Pakhribas in Nepal.
[d] Statistics were calculated from data provided in Soto and Díaz-Fierros (1998).

Runoff and erosion annual predictions were acceptable for all the proposed approaches. No studies were found that included seasonal runoff and erosion predictions within a burned area. In this study, model efficiency for seasonal predictions of runoff and erosion, was lower than that for annual predictions in both approaches (SM:

NS_{Runoff} = 0.53, $NS_{Erosion}$ = 0.71; SM-SWR: NS_{Runoff} =0.71, $NS_{Erosion}$ =0.74); however, these values can also be considered as acceptable. Furthermore, Morgan (2005) argued that it was necessary to know whether an erosion prediction has been achieved without having to compromise runoff figures, and that validation should also be made to all of its constituent models. Therefore, the accurate NS erosion indices shown in Table 21 do not necessarily mean that the model itself is valid unless runoff predictions can also be considered accurate. In the present study, NS runoff predictions were lower than measured values but still acceptable, providing a further indication of the validity of the model structure.

In general terms, the present application of the MMF model, with its different approaches, has allowed the identification of different hydrologic responses from the two studied land uses. In the case of the runoff predictions, this was mainly due to differences in the soil field capacity (MS parameter), the lower MS value for the eucalypt site leading to lower infiltration capacity, higher sensitivity to changes in SWR and, consequently, higher runoff rates than in the pine site. It should be noted that MS was estimated from in situ soil moisture measurements, and therefore might have been affected by measurement difficulties. Another important factor that allowed distinction between land uses and treatments was the EHD variation according to ground cover in the SM and SM-SWR approaches, allowing a higher saturation capacity in the plots with higher ground cover, where runoff rates were reduced. Despite their dependence on runoff amounts, the greater ground cover at the pine than at the eucalypt site led to lower predicted erosion rate predictions; this difference can be attributed to the thicker litter layer in the pine site, in agreement with the results and data from validation study site taken from Shakesby et al. (1996). The ability of MMF to distinguish erosion rates in different land use and fire conditions was noted by Fernández et al. (2010a), for a fire of moderate severity in a Pinus pinaster stand and one of high severity in Ulex europaeus shrubland.

For the post-fire rehabilitation treatments modelled with the MMF, Fernández et al. (2010a) found erosion predictions of low accuracy and the runoff outputs were not evaluated. In the present study, the accuracy of predictions for all modelling approaches, although more efficient, was also limited for two main reasons. First, the number of evaluated treated/control plots was comparatively small, and, second, there were limitations as regards the soil moisture and SWR data available for model calibrations. In particular, SWR data were only collected for untreated plots, and it is reasonable to assume that treatments might have led to lower repellency values (Prats et al., 2012). A better estimate of the impacts of post-fire rehabilitation treatments on repellency would be necessary to improve MMF runoff predictions for the treated plots; quite possibly, this would also improve runoff results from other models used for predicting the impact of rehabilitation treatments.

Sources of error

Prediction errors can usually be attributed to errors in the model, the input data, and the data used for validation (Nearing et al., 1999). In this study, one important source of error might stem from the revised MMF model being an empirical one developed from hillslope-scale data and validated by the developers using mostly smaller-scale erosion plot data derived from agricultural fields (Morgan et al., 1984; Morgan, 2001). The relationships between the hydrologic response at plot- and hillslope- or even catchment-scales are not well known. Typically, small-scale measurements are often compared, even though there is an approximately inverse relationship between erosion amounts and the scale of measurement (Shakesby and Doerr, 2006) and therefore the model is probably overestimating hillslope-scale erosion in post-fire situations. The empirical equations of MMF might, therefore, not be expected to represent sufficiently well key processes in burned areas. However, the objective of this work was to test, adapt and validate the MMF model with appropriate modifications for burned areas. The accurate predictions achieved with the model at both the calibration and, more importantly, the validation site, together with successful application of the model to other burned sites (Vieira et al., 2010; Fernández et al., 2010a) indicate that the model structure is valid for these situations and also provide a robust calibration for burned areas.

Two additional sources of errors were analysed: measurement limitations, related to values that were obtained in the site and used as model inputs or for calibration and validation; and modelling errors, related to difficulties in estimating inputs and model modifications.

Measurement errors

Uncertainties in rainfall, surface cover, and sediment yields are the most important potential sources of measurement errors (Pietraszek, 2006; Larsen and MacDonald, 2007). A good agreement was found between the automatic tipping-bucket rainfall gauges installed at the calibration site, and the seven standard rainfall gauges and rainfall data from the nearest long-term climate station (Castelo Burgães; SNIRH, 2012). The rainfall figures can, therefore, be regarded as accurate, with negligible differences between eucalypt and pine. The same is true of the accuracy of ground cover measurements, because they were always carried out frequently by the same observer, thereby reducing possible errors.

As regards sediment losses, weekly monitoring, together with the high capacity of runoff collection tanks, provided an overview of the hydrological response of each plot. The collected runoff in the tanks was also cross-checked with the tipping-bucket measurements in order to minimise the possibility of runoff losses not accounted for. Given that both suspended sediments collected in the runoff tanks and the trapped coarse material were measured, it seems reasonable to suggest that the erosion measurements were accurate, within the limitations of a plot-based set-up (e.g. Boix-Fayos et al., 2006).

Since the repellency measurements were conducted on an untreated transects parallel to the plots, their representativeness might be less for the control than for the treated plots. Because repellency measurement involves soil disturbance, measurements were made outside the plots. In an attempt to control for the known high spatial variability in this phenomenon (Leighton-Boyce, 2002), a large number of individual measurements were conducted (1179 measurements), at several depths and at five equidistant points down the slope to improve the representativeness of the results.

Modelling errors

As discussed above, the revised MMF model might still need to be adapted to specific conditions for each burned area, especially for primary effects such as fire-induced changes in the soil and surface cover. The inclusion of these changes in the model inputs led to some limitations, which might have been only partially solved by the calibration applied in this study.

One source of error is the effective hydrological depth of soil (EHD). In this study, guide values from Morgan (2001) were used, but they might not be very representative of the study sites where soils are very thin, especially for the applied treatments. The variation in EHD with ground cover for the SM and SM-SWR approaches seems, however, to have partially addressed this issue.

For the actual and potential evapotranspiration (Et and E0) estimates during the study period, uncertainties in evapotranspiration behaviour in a burned site were circumvented by using measured soil moisture data from the study sites to drive the soil water balance calculations. The calculation of Et using measured data could include a mixture of observation and parameterization, but we believe that this was not important since water balance calculations without measured soil moisture led to an estimation of similar seasonal Et patterns. In fact, the seasonal patterns of the Et/E0 ratio estimated in the calibration site could be representative, in an approximate way, of what happens in other burned areas during a typical year, as indicated by the acceptable MMF model performance for the validation site using Et/E0 patterns calculated for the calibration site.

The C factor estimation methodology used in this work, based on RUSLE (Renard et al., 1997), takes into consideration disturbed soils; however, it is not possible to assess how burning affects each of the subfactors used in the RUSLE methodology, or the validity of the relationships used to calculate the C factor (González-Bonorino and Osterkamp, 2004). These problems arise because most studies of post-fire sediment yields do not incorporate detailed measurements of C subfactors overtime (Larsen and MacDonald, 2007),even though there are many studies that have tested the RUSLE model.

Another concern is that the MMF model estimates erosion from whichever is the lower of estimated soil detachment and transport capacity. The sites in this study, and others, have a much reduced canopy cover because of the fire; the resulting high values of kinetic energy in rainfall lead to high estimates of detachment. Consequently, all erosion quantities were dictated by the simulated transport capacity. Since the K and COH factors are used in soil particle detachment determination and not in the transport capacity calculations (Meyer and Wischmeier, 1969; Morgan, 2005), it is impossible to include fire-induced soil changes over time (such as texture, soil organic matter, permeability class and soil structure) in the erosion predictions.

There is also the problem of changes after the initial impact of the wildfire on the soil and ground surface, such as vegetation recovery, repellency patterns, litter cover changes (natural or applied) and soil moisture changes. Owing to the lumped nature of MMF (Figure 24), these changes could not be translated into a single input value. In addition, the inverse relationship between soil moisture and SWR is not reproduced by the C factor formulation (Larsen and MacDonald, 2007). The seasonal modelling approaches adopted in the present study provide a possible way of avoiding this problem: in these approaches, the C factor is calculated to represent the initial soil conditions taking into account fire severity, while the litter cover changes are represented in the GC and P factors, and the SWR-soil moisture relationship are represented in the MS factor. This conflicts with Larsen and MacDonald's (2007) proposal to include the K factor in the soil water repellency-soil moisture relationship in RUSLE, arguing that it is a soil-related issue. However, this approach was not considered in the present study since (as stated above) the K factor is not relevant to the final erosion results of the MMF model.

Although adding the SWR effect in the MS input was considered the best option for the revised MMF model when applied to burned areas, some assumptions might have associated errors. Error propagation occurred by assuming that the SWR was the same for the treated as for the control plots, but it was not measured. In the control plots, SWR variations might have been more pronounced than in the treated plots as a result of higher soil moisture variations and therefore more SWR variability than in the treated plots, where the applied mulch maintained moist conditions for longer.

The presence of both ash at the soil surface and stones in large concentrations at the surface and within soils in burned forested areas of the Iberian Peninsula have been highlighted by different researchers (e.g. Cerdà and Doerr, 2008; Esteves et al., 2012). The effect of the former was not considered very important in the present study since most of it was removed during early storms during the pre-treatment period. The stone content influence could not be included because it was not contemplated in the model formulations, and also the impact of stones in the post-fire response is not fully known. Several authors have referred to elevated stone content reducing erosion by promoting structural stability thereby forming a protective surface stone lag (Shakesby, 2011), and also reducing the effect of soil water repellency by providing preferential flow routes through the soil (Urbanek and Shakesby, 2009). Thus both this factor and ash (particularly where it reaches considerable thicknesses) need to be considered in future model development as regards their different effects on infiltration and on sediment availability.

Finally, the adjustment of the land use dataset for the MS input into three groups (ET, ET and PT&C), carried out because of measurement limitations, led to the prediction of runoff rates which varied little for plots within each of these groups. Even though the plot-specific dataset contributed to erosion predictions, it was not possible for the model to explain the hydrological variability between individual plots belonging to each of these groups.

This problem with predicting variability between plots highlights a more general limitation in runoff and erosion modelling. Models intended for application to unmonitored slopes must use site-averaged parameters that can be estimated in more general terms, but which might not capture the smaller-scale variations in plot conditions and key processes such as infiltration (Beven, 2000).

The revised Morgan–Morgan–Finney model as a management tool

The main objective of this study has been to evaluate the revised MMF model as a management tool for burned areas, in order to predict not only areas with higher erosion risk but also the efficiency of post-fire rehabilitation treatments. Despite the advances made in recent years in developing physically-based models, a simple empirical model is often more successful at predicting soil erosion and usually easier to use (De Roo, 1996). In simple models, the data input is not so demanding and therefore easier to obtain; in the case of MMF, De Roo presented some input ranges for cases where site determinations could not be made. However, most of these inputs were not calibrated for burned areas, and (as the present study has shown) some modifications have to be made.

In particular, the present study indicates that the most likely constraint to the MMF, when applied to an unmonitored site, is that of SWR; first, because there is still a lack of knowledge associated with its occurrence, pattern and time of residence and erosional impact in a burned area (Shakesby et al., 2000; Shakesby, 2011); and, second, because the calibration presented here is specific to the present dataset and to the SWR methodology used, which does not necessarily mean it will be suitable elsewhere. However, after model calibration and application to the study sites, it was possible to derive accurate estimates of erosion similar to those obtained at the calibration plots, and therefore to consider the model as being robustly calibrated for this parameter. Limitations in repellency data were circumvented, for the validation site, by linking repellency to soil moisture, and therefore to seasonal rainfall (see Keizer et al., 2008a, for an example of annual repellency patterns in a similar burned area). However, and as noted before, there are possible impacts of treatment on repellency, which have not been studied in detail.

For the rehabilitation treatment predictions, the model performed accurately for the pine and eucalypt stands, but it still needs to be evaluated for other types of cover. Also, the model responded well to the simulation of the treatments with their different mulching application rates and therefore ground cover percentages. It was also able to simulate post-treatment losses of mulch by overland flow and the removal of pine needles. This good response means that the revised model is capable of providing an accurate indication of the amount of treatment required to protect a given soil, and therefore a means of determining the cost-effectiveness of any treatment. Opening with this, a new research opportunity under the subject of modelling post-fire management practices, either by mulch application (Fernández et al., 2010a,b; Prats et al., 2012,

2013) or other common techniques such as logging or clearcutting (Fernández et al., 2004, 2007).

Conclusions

The main conclusions of this application of the revised MMF model to post-wildfire treatments in pine and eucalypt forests in maritime north central Portugal, following wildfire and forest residue mulching are as follows.

(1) The revised MMF model was able to predict plot-scale runoff and erosion rates of the year following the wildfire with acceptable accuracy (NS> 0.5) in both forest land uses.

(2) The runoff and erosion predictions were improved by addressing seasonal changes in model parameters (NS > 0.7), and by incorporating soil water repellency into the runoff predictions (NS > 0.8).

(3) For the eucalypt plots, individual seasonal predictions for all the plots were less accurate than the full period ones, but were still satisfactory (NS> 0.6), with more efficient runoff and erosion predictions for the eucalypt control plots than for treated plots.

(4) For the pine site, seasonal predictions of runoff and erosion were poor, but it was impossible to adjust the model due to the small number of plots, data limitations, a marked variability in the hydrological and erosion response between plots, and the small amounts of erosion recorded.

(5) It has been shown that the revised MMF model can easily provide a set of simple criteria for management decisions for runoff and erosion in burned areas. The successful predictions of runoff and erosion at the validation site attest to its applicability to other eucalypt and pine sites in Portugal, and suggest that it may well have wider applicability to post-fire conditions for other vegetation types elsewhere in the Mediterranean.

(6) The modifications made to the MMF model have improved its capacity to determine the efficiency of soil rehabilitation treatments in preventing post-wildfire runoff and erosion, but the approach still requires improvement.

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References

Arcenegui, V., Mataix-Solera, J., Guerrero, R., Zornoza, R., Mayoral, A.M., Morales, J., 2007. Factors controlling the water repellency induced by fire in calcareous Mediterranean forest soils. Eur. J. Soil. Sci. 58, 1254–1259.

Benavides-Solorio, J., MacDonald, L.H., 2005. Measurement and prediction of postfire erosion at hillslope scale, Colorado Front Range. Int. J. Wildland Fire 14 (4), 457–474.

Beven, K.J., 2000. Rainfall-Runoff Modeling: The Primer. John Wiley, Chichester, UK.

Boix-Fayos, C., Martínez-Mena, M., Calvo-Cases, A., Arnau-Rosalén, E., Albaladejo, J., Castillo, V., 2006. Causes and underlying processes of measurement variability in field erosion plots in Mediterranean conditions. Earth Surf. Proc. Land. 32, 85–101.

Burch, G.J., Moore, I.D., Burns, J., 1989. Soil hydrophobic effects on infiltration and catchment runoff. Hydrol. Process. 3, 211–222.

Carvalho, A., Flannigan, M., Logan, K., Gowman, L., Miranda, A.I., Borrego, C., 2010. The impact of spatial resolution on area burned and fire occurrence projections in Portugal under climate change. Climatic Change 98, 177–197.

Cerdà, A., Doerr, S.H., 2007. Soil wettability, runoff and erodibility of major dry Mediterranean land use types on calcareous soils. Hydrol. Process. 21, 2325–2336.

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

Coelho, C.O.A., Shakesby, R.A., Gonzales del Tánago, M., Ternan, L., Walsh, R.P.D., Williams, A.G., 1995. IBERLIM: Land Management and Erosion Limitation in the Iberian Peninsula Final Report to the EC in fulfilment of Project EV5V-0041 'Land Management Practice and Erosion Limitation in Contrasting Wildfire and Gullied Locations in the Iberian Peninsula'.

Coutinho, M.A., Tomás, P., 1995. Characterisation of raindrop size distributions at the Vale Formoso Experimental Erosion Centre. Catena 25, 187–197.

DeBano, L.F., 2000. The role of fire and soil heating on water repellency in wildland environments: a review. J. Hydrol. 231–232, 195–206.

De Roo, A.P.J., 1996. Validation problems of hydrologic and soil-erosion catchment models: examples from a Dutch soil erosion project. In: Anderson, M.G., Brooks, S.M. (Eds.), Advances in Hillslope Processes. Wiley, Chichester, pp. 669–683.

Dekker, L.W., Ritsema, C.J., 1996. Variation in water content and wetting patterns in Dutch water repellent peaty clay and clayey peat soils. Catena 28, 89–105.

Díaz-Fierros, F., Benito, E., Pérez, R., 1987. Evaluation of the U.S.L.E. for prediction of erosion in burnt forest areas in Galicia (NW Spain). Catena 14, 189–199.

Doerr, S.H., 1998. On standardizing the ''Water Drop Penetration Time'' and the ''Molarity Ethanol Droplet'' techniques to classify soil hidrophobicity: a case of study using medium textured soils. Earth Surf. Proc. Land. 23, 663–668.

Doerr, S.H., Moody, J., 2004. Hydrological impacts of soil water repellency: on spatial and temporal uncertainties. Hydrol. Process. 18, 829–832.

Doerr, S.H., Thomas, A.D., 2000. The role of soil moisture in controlling water repellency: new evidence from forest soils in Portugal. J. Hydrol. 231–232, 134–147.

Doerr, S.H., Shakesby, R.A., Walsh, R.P.D., 1996. Soil water repellency variations with depth and particle size fraction in burned and unburned Eucalyptus globulus and Pinus pinaster forest terrain in the Águeda basin, Portugal. Catena 27, 25– 47.

Doerr, S.H., Shakesby, R.A., Walsh, R.P.D., 2000. Soil water repellency, its characteristics, causes and hydro-geomorphological consequences. Earth Sci. Rev. 51, 33–65.

Doerr, S.H., Ferreira, A.J.D., Walsh, R.P.D., Shakesby, R.A., Leighton-Boyce, G., Coelho, C.O.A., 2003. Soil water repellency as a potential parameter in rainfall-runoff modelling: experimental evidence at point to catchment scales from Portugal. Hydrol. Process. 17, 363–377.

Esteves, T.C.J., Kirkby, M.J., Shakesby, R.A., Ferreira, A.J.D., Soares, J.A.A., Irvine, B.J., Ferreira, C.S.S., Coelho, C.O.A., Bento, C.P.M., Carreiras, M.A., 2012. Mitigating land degradation caused by wildfire: application of the PESERA model to fire affected sites in central Portugal. Geoderma. http://dx.doi.org/10.1016/j.geoderma.2012.01.001.

FAO, PNUMA, UNESCO, 1980. Metodologia Provisional para la Evaluación de la Degradación de los Suelos. Rome.

Fernández, C., Vega, J.A., Gras, J.M., Fontúrbel, T., Cuiñas, P., Dambrine, E., Alonso, M., 2004. Soil erosion after Eucalyptus globulus clearcutting: differences between logging slash disposal treatments. For. Ecol. Manage. 195, 85–95. http://dx.doi.org/10.1016/J.FORECO.2004.02.052.

Fernández, C., Vega, J.A., Fontúrbel, M.T., Pérez-Gorostiaga, P., Jiménez, E., Madrigal, J., 2007. Effects of wildfire, salvage logging and slash treatments on soil degradation. Land Degrad. Dev. 18, 591–607.

Fernández, C., Vega, J.A., Vieira, D.C.S., 2010a. Assessing soil erosion after fire and rehabilitation treatments in NW Spain: performance of RUSLE and revised Morgan–Morgan–Finney models. Land Degrad. Dev. 21, 774–787.

Fernández, C., Vega, J.A., Jiménez, E., Vieira, D.C.S., Merino, A., Ferreiro, A., Fontúrbel, T., 2010b. Seeding and mulching + seeding effects on post-fire runoff, soil erosion and species diversity in Galicia (NW Spain). Land Degrad. Dev. 23, 150– 156.

Ferreira, A.J.D., 1996. Processos hidrológicos e hidroquímicos em povoamentos de Eucalyptus globulus labill e Pinus pinaster Aiton. Universidade de Aveiro, Tese de Doutoramento.

Ferreira, A.J.D., Coelho, C.O.A., Boulet, A.K., Leighton-Boyce, G., Keizer, J.J., Ritsema, C.J., 2005. Influence of burning intensity on water repellence and hydrological processes at forest sites in Portugal. Aust. J. Soil Res. 43 (3), 327–336.

Ferreira, A.J.D., Coelho, C.O.A., Ritsema, C.J., Boulet, A.K., Keizer, J.J., 2008. Soil and water degradation processes in burned areas: lessons learned from a nested approach. Catena 74, 273–285.

Gerlach, T., 1967. Hillslope troughs for measuring sediment movement. Revue geomorphologie dynamique 17, 173–174.

González-Bonorino, G., Osterkamp, W.R., 2004. Applying RUSLE2.0 on burned forest lands: an appraisal. J. Soil Water Conserv. 59, 36–42.

Harding, A., Palutikof, J., Holt, T., 2009. The climate system. In: Woodward, J. (Ed.), The Physical Geography of the Mediterranean. Oxford University Press, Oxford, pp. 69–88.

Imeson, A.C., Verstraten, J.M., Van Mulligen, E.J., Sevink, J., 1992. The effects of fire and water repellency on infiltration and runoff under Mediterranean type forest. Catena 19, 345–361.

Jordán, A., Gonzalez, F.A., Zavala, L.M., 2010. Re-establishment of soil water repellency after destruction by intense burning in a Mediterranean heathland (SW Spain). Hydrol. Process. 24, 736–748.

Keizer, J.J., Coelho, C.O.A., Matias, M.J.S., Domingues, C.S.P., Ferreira, A.J.D., 2005a. Soil water repellency under dry and wet antecedent weather conditions for selected land-cover types in the coastal zone of central Portugal. Aust. J. Soil Res. 43 (3), 297–308.

Keizer, J.J., Coelho, C.O.A., Shakesby, R.A., Domingues, C.S.P., Malvar, M.C., Perez, I.M.B., Matias, M.J.S., Ferreira, A.J.D., 2005b. The role of soil water repellency in overland flow generation in pine and eucalypt forest stands in coastal Portugal. Aust. J. Soil Res. 43 (3), 337–350.

Keizer, J.J., Doerr, S.H., Malvar, M.C., Prats, S.A., Ferreira, R.S.V., Oñate, M.G., Coelho, C.O.A., Ferreira, A.J.D., 2008a. Temporal variation in topsoil water repellency in two recently burnt eucalypt stands in north-central Portugal. Catena 74, 192– 204.

Keizer, J.J., Nunes, J.P., Palacios, E., Beekman, W., Malvar, M.C., 2008b. Soil erosion modeling for two recently burnt eucalypt slopes: comparing MEFIDIS, MMF and USLE. ISCO2008. In: 15th Conference of the International Soil Conservation Organization. May 18–23, Budapest, Hungary.

Kirkby, M.J., 1976. Hydrological slope models: the influence of climate. In: Derbyshire, E. (Ed.), Geomorphology and Climate. Wiley, London, pp. 247–267.

Kirkby, M.J., Irvine, B.J., Jones, R.J.A., Govers, G., Boer, M., Cerdan, O., Daroussin, J., Gobin, A., Grimm, M., Le Bissonnais, Y., Kosmas, C., Mantel, S., Puigdefabregas, J., Van Lynden, G., 2008. The PESERA coarse scale erosion model for Europe. I. — Model rationale and implementation. Eur. J. Soil Sci. 59, 1293–1306.

Larsen, I.J., MacDonald, L.H., 2007. Predicting post-fire sediment yields at the hillslope scale: testing RUSLE and Disturbed WEPP. Water Resour. Res. 43, W11412.

Leighton-Boyce, G., 2002. Spatio-temporal Dynamics and Hydrogeomorphic Implications of Soil Water Repellency within Eucalyptus Forests in NorthCentral Portugal. Unpublished PhD thesis, University of Wales Swansea.

Malvar, M.C., Prats, S.A., Nunes, J.P., Keizer, J.J., 2011. Post-fire overland flow generationandinter-rillerosionundersimulatedrainfallintwoeucalyptstands in north-central Portugal. Environ. Res. 111, 222–236.

Malvar, M.C., Martins, M.A.S., Nunes, J.P., Robichaud, P.R., Keizer, J.J., 2013. Assessing the role of pre-fire ground preparation operations and soil water repellency in post-fire runoff and inter-rill erosion by repeated rainfall simulation experiments in Portuguese eucalypt plantations. Catena 108, 69–83.

Mataix-Solera, J., Cerdà, A., Arcenegui, V., Jordán, A., Zavala, L.M., 2011. Fire effects on soil aggregation: a review. Earth Sci. Rev. 109 (1–2), 44–60.

Martins, M.A.S., Machado, A.I., Serpa, D., Prats, S.A., Faria, S.R., Varela, M.E.T., Gonzalez-Pelayo, O., Keizer, J.J., 2013. Runoff and inter-rill erosion ina Maritime Pine and a eucalypt plantation following wildfire and terracing in north-central Portugal. J. Hydrol. Hydromechanics 61 (4), 261–269.

Martínez-Zavala, L., Jordán-López, A., 2009. Influence of different plant species on water repellency in Mediterranean heathland soils. Catena 76, 215–223.

Meyer, L.D., Wischmeier, W.H., 1969. Mathematical simulation of the process of soil erosion by water. Trans. Am. Soc. Agric. Eng. 12, 754–758.

Moody, J.A., Martin, D.A., Cannon, S.H., 2008. Post-wildfire erosion response in two geologic terrains in the western USA. Geomorphology 95, 103–118.

Moreira, F., Vaz, P.J.G., Catry, F.X., Silva, J.S., 2009. Regional variations in wildfire susceptibility of land cover types in Portugal: implications for landscape management to minimize fire hazard. Int. J. Wildland Fire 18, 563–574.

Morgan, R.P.C., 2001. A simple approach to soil loss prediction: a revised Morgan–Morgan–Finney model. Catena 44, 305–322.

Morgan, R.P.C., 2005. Soil Erosion and Conservation. Blackwell, Oxford.

Morgan, R.P.C., Morgan, D.D.V., Finney, H.J., 1984. A predictive model for the assessment of erosion risk. J. Agric. Eng. Res. 30, 245–253.

Moriasi, D.N., Arnold, J.G., Van Liew, M.W., Bingner, R.L., Harmel, R.D., Veith, T.L., 2007. Model evaluation guidelines for systematic quantification of accuracy in watershed simulations. Trans. ASABE 50, 885–900.

Naveh, Z., 1990. Fire in the Mediterranean – a landscape ecological perspective. In: Goldammer, J.G., Jenkins, M.J. (Eds.), Fire in ecosystem dynamics. Pp. 1–20. SPB Academic Publishing, The Hague, Netherlands.

Nash, J.E., Sutcliffe, J.V., 1970. River flow forecasting through conceptual models. Part 1: A discussion of principles. J. Hydrol. 10, 282–290.

Nearing, M.A., 1998. Why soil erosion models over-predict small soil losses and under-predict large soil losses. Catena 32, 15–22.

Nearing, M.A., Foster, G.R., Lane, L.J., Finkner, S.C.,1989. A process-based soil erosion model for USDE-water erosion prediction project technology. Am. Soc. Agr. Eng. 32, 1587–1593.

Nearing, M.A., Govers, G., Norton, L.D., 1999. Variability in soil erosion data from replicated plots. Soil Sci. Soc. Am. J. 63, 1829–1835.

Pereira, J., Carreira, J., Silva, J., Vasconcelos, M., 2006a. Alguns conceitos básicos sobre os fogos rurais em Portugal. In: Pereira, J.S., Pereira, J.M.C., et al. (Eds.), Incêndios florestais em Portugal – caracterização, impactes e prevenção, pg 133–16. ISAPress, Lisbon.

Pereira, J., Correia, A.V., Correia, A.C., Ferreira, M., Onofre, N., Freitas, H., Godinho, F., 2006b. Florestas e biodiversidade. In: Santos, F., Miranda, P. (Eds.), Alterações climáticas em Portugal—cenários, impactos e medidas de adaptação (Projecto SIAM II). Gradiva, Lisbon, pp. 301–343.

Pietraszek, J.H., 2006. Controls on post-fire erosion at the hillslope scale. Colorado Front Range, M.S. thesis. p. 124. Plus appendices, Colorado State Univ., Fort Collins.

Prats, S.A., MacDonald, L.H., Monteiro, M., Ferreira, A.J.D., Coelho, C.O.A., Keizer, J.J., 2012. Effectiveness of forest residue mulching in reducing post-fire runoff and erosion in a pine and a eucalypt plantation in north-central Portugal. Geoderma 191, 115–124.

Prats, S.A., Malvar, M.C., Vieira, D.C.S., MacDonald, L., Keizer, J.J., 2013. Effectiveness of Hydromulching to reduce runoff and erosion in a recently burnt Pine plantation in Central Portugal. Land Degrad. Dev.. http://dx.doi.org/10.1002/ldr.2236.

Prosser, I.P., 1990. Fire, humans and denudation at Wangrah Creek, Southern Tablelands, N.S.W. Aust. Geogr. Stud. 28, 77–95.

Quinton, J.N., 1997. Reducing predictive uncertainty in model simulations: a comparison of two methods using the European Soil Erosion Model (EUROSEM). Catena 30, 101–117.

Renard, K.G., Foster, G.R., Weesies, G.A., Mc Cool, D.K., Yoder, D.C. (cords.), 1997. Predicting soil erosion by water: a guide to conservation planning with the revised universal soil loss equation (RUSLE). Agriculture Handbook nº 703. Washington DC. USDA, Natural Resources Conservation Service.

Robichaud, P.R., Elliot, W.J., Pierson, F.B., Hall, D.E., Moffet, C.A., 2007. Predicting postfire erosion and mitigation effectiveness with a web-based probabilistic erosion model. Catena 71, 229–241.

San-Miguel, J., Camia, A. 2009. Forest fires at a glance: facts, figures and trends in the EU. In: Birot, Y. (Ed.), Living with wildfires: what can science tell us, Chp. 1, pp 11–19. European Forest Institute, Discussion Paper 15.

Santos, J.M., Verheijen, F.G.A., Wahren, F.T., Wahren, A., Gosch, L., Bernard-Jannin, L., Rial-Rivas, M.E., Keizer, J.J., Nunes, J.P., 2013. Soil water repellency dynamics in Pine and Eucalypt plantations in Portugal – a high resolution time series. Land Degrad. Dev. http://dx.doi.org/10.1002/ldr.2251.

Scott, D.F., Versfeld, D.B., Lesch, W., 1998. Erosion and sediment yield in relation to afforestation and fire in the mountains of the Western Cape Province, South Africa. S. Afr. Geogr. J. 80, 52–59.

Shakesby, R.A., 2011. Post-wildfire soil erosion in the Mediterranean: review and future research directions. Earth Sci. Rev. 105, 71–100.

Shakesby, R.A., Doerr, S.H., 2006. Wildfire as a hydrological and geomorphological agent. Earth Sci. Rev. 74, 269–307.

Shakesby, R.A., Walsh, R.P.D., Coelho, C.O.A., 1991. New developments in techniques for measuring soil erosion in burned and unburned forested catchments, Portugal. Zeitschrift für Geomorphologie, Supplementband 83, 161–174.

Shakesby, R.A., Coelho, C.O.A., Ferreira, A.D., Terry, J.P., Walsh, R.P.D., 1993. Wildfire impacts on soil erosion and hydrology in wet Mediterranean forest, Portugal. Int. J. Wildland Fire 3, 95–110.

Shakesby, R.A., Boakes, D.J., Coelho, C.O.A., Gonçalves, A.J.B., Walsh, R.P.D., 1996. Limiting the soil degradational impacts of wildfire in pine and eucalypt forests in Portugal. Appl. Geogr. 16, 337–356.

Shakesby, R.A., Doerr, S.H., Walsh, R.P.D., 2000. The erosional impact of soil hydrophobicity: current problems and future research directions. J. Hydrol. 231–232, 178–191.

SNIRH, 2012. Sistema Nacional de Informação de Recursos Hídricos. http://snirh.pt/. Accessed 1 March 2012.

Soto, B., Díaz-Fierros, F., 1998. Runoff and soil erosion from areas of burnt scrub: comparison of experimental results with those predicted by the WEPP model. Catena 31, 257–270.

Thornthwaite, C.W., Mather, J.R., 1957. Instructions and tables for computing potential evapotranspiration and the water balance: Centerton, N.J., Laboratory of Climatology, Publications in Climatology. vol. 10, No. 3: 185–311.

Urbanek, E., Shakesby, R.A., 2009. The impact of stone content on water flow in water-repellent sand. Eur. J. Soil Sci. 60, 412–419.

Vega, J.A., Díaz-Fierros, F., 1987. Wildfire effects on soil erosion. Ecologia Mediterranea 13 (4), 119–125.

Veléz, R., 2009. The causing factors: a focus on economic and social driving forces. In: Birot, Y. (Ed.), Living with wildfires: what can science tell us? p. 21–26. European Forest Institute, Discussion Paper 15.

Varela, M.E., Benito, E., Keizer, J.J., 2010. Effects of wildfire and laboratory heating on soil aggregate stability of pine forest in Galicia: the role of lithology, soil organic matter content and water repellency. Catena 83, 127–134.

Vieira, D.C.S., Malvar, M.C., Nunes, J.P., Keizer,J.J., 2010. Modelação da erosão do solo e sua verificação numa área florestal recentemente ardida em Açores, centronorte de Portugal. Territorium 17, 127–134.

Walsh, R.P.D., Coelho, C.O.A., Shakesby, R.A., Terry, J.P., 1992. Effects of land use management practices and fire on soil erosion and water quality in the Águeda River Basin, Portugal. GeoÖkoPlus 3, 15–36.

Walsh, R.P.D., Boakes, D.J., Coelho, C.O.A., Gonçalves, A.J.B., Shakesby, R.A., Thomas, A.D., 1994. Impact of fire-induced water repellency and post-fire forest litter on overland flow in northern and central Portugal. In: Proceedings of the

Second International Conference on Forest Fire Research, November 1994. Coimbra, Portugal, Volume II, pp. 1149–1159.

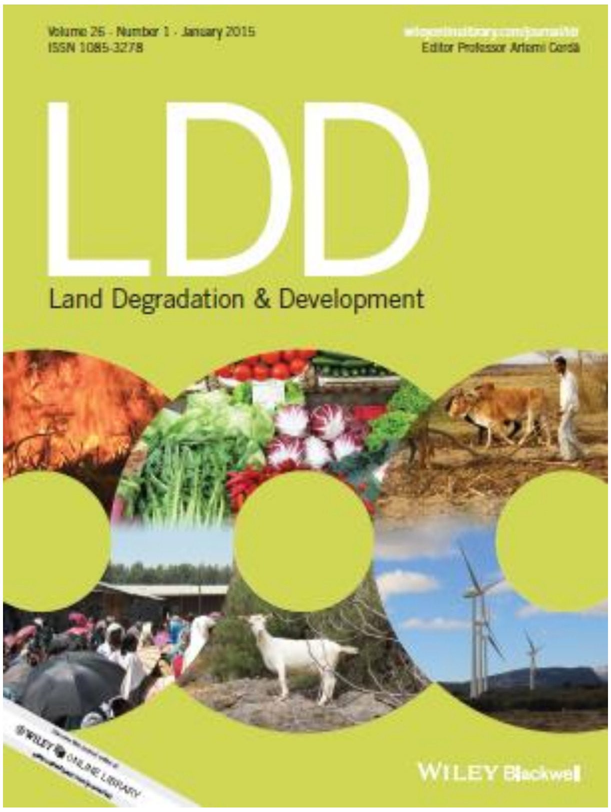
Walsh, R.P.D., Boakes, D.J., Coelho, C.O.A., Ferreira, A.D., Shakesby, R.A., Thomas, A.D., 1995. Post-fire land use management and runoff responses to rainstorms in Portugal. In: McGregor, D.F.M., Thompson, D. (Eds.), Geomorphology and Land Management in a Changing Environment. Wiley, Chichester, pp. 283–308.

Wells, W.G., 1981. Some effects of brushfires on erosion processes in coastal southern California. Erosion and Sediment Transport in Pacific Rim Steeplands, International Association of Hydrological Sciences Publication No. 132. IAHS Press, Institute of Hydrology, Wallingford, UK, pp. 305–342.

Wischmeier, W.H., Smith, D.D., 1978. Predicting rainfall-erosion losses – a guide to conservation planning. Agriculture Handbook. No. 537. USDA. Washington, DC, p. 58.

IV. Other contributions

IV.I Post-fire rehabilitation treatments

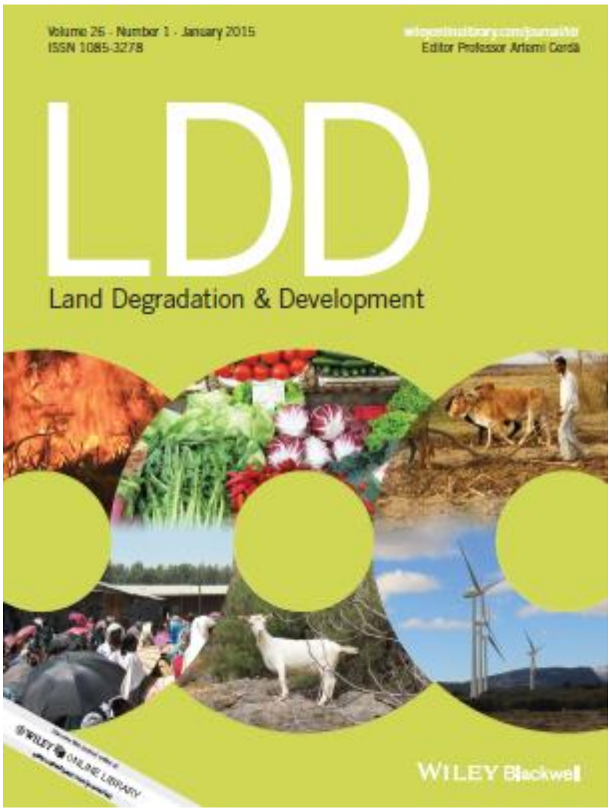


Fernández C. Vega J.A., Jiménez, E., Vieira, D.C.S., Merino, A., Ferreiro, A., Fontúrbel, T. (2012).

Seeding and mulching+seeding effects on post-fire runoff, soil erosion and species diversity in Galicia (NW Spain)

Land Degradation and Development 23, 150-156.

The effects of two different soil rehabilitation treatments on runoff, infiltration, erosion and species diversity were evaluated in a shrubland area in Galicia (NW Spain) after an experimental fire by means of rainfall simulations. The treatments compared were: seeding, seeding + mulching and control (untreated). Rainfall simulations were conducted 9 months after fire and the application of soil rehabilitation treatments. A rainfall rate of 67 mm h−1 was applied for 30 min to each runoff plot. Seeding significantly increased plant species richness in the treated plots relative to the control plots, although it had no effect on diversity or evenness. Rehabilitation treatments did not significantly increase soil cover or affect runoff and infiltration. Soil losses were low in all cases, varying from 75·6 kg ha−1 in the seeded + mulched plots to 212·1 kg ha^{-1} in the untreated plots. However, there were no significant differences in sediment yields between treatments. The percentage of bare soil appeared to be a critical variable in controlling runoff and erosion.



Prats, S.A., Malvar, M.C., Vieira, D.C.S., MacDonald, L., Keizer, J. J.

Effectiveness of Hydromulching to reduce runoff and erosion in a recently burnt Pine plantation in Central Portugal

Land Degradation and Development (2013)

Forest fires can greatly increase runoff and surface erosion rates. Post-fire soil erosion control measures are intended to minimize this response and facilitate ecosystem recovery. In a few recent cases, hydromulch has been applied, and this consists of a mixture of organic fibers, water and seeds. The objectives of this research were to (i) analyze the effectiveness of hydromulch in reducing post-fire runoff and sediment production and (ii) determine the underlying processes and mechanisms that control post-fire runoff and erosion. After a wildfire occurred in August 2008, 14 plots ranging in size from 0·25 to 10 m^2 were installed on a 25 degree slope in a burnt pine plantation that had also been subjected to salvage logging. Half of the plots were randomly selected and treated with hydromulch. One of two slope strips adjacent to the plots was also hydromulched and used for monitoring some soil properties. Measurements made in each of the first 3 years following the wildfire included (i) the plot-scale runoff volumes and sediment yields; (ii) soil shear strength, soil moisture, and soil water repellency; and (iii) surface cover. The hydromulch reduced overland flow volume by 70% and soil erosion by 83%. The decrease in runoff was attributed to the increase in soil water retention capacity and the decrease in soil water repellency, whereas the reduction in soil erosion was initially attributed to the protective cover provided by the hydromulch and lately to an enhanced vegetative regrowth in the third year after burning.

V. General discussion

V.I From pre-fire to post fire

One of the most noticeable changes brought by wildfires is the dramatic reduction of vegetation. This absence of protective cover after fire has been used as one of the main justifications for enhanced post-fire hydrologic and erosive behaviour (Neary et al., 1999; Shakesby and Doerr, 2006; Pausas et al., 2008; Shakesby, 2011). The degree of change from pre to post fire conditions is frequently related with burn severity, and its metric is usually based on the loss or decomposition of organic matter, both aboveground and belowground (Keeley, 2009).

Burn severity classification methodologies worldwide show some inconsistencies between authors. These burn severities classifications were associated with a significant increase in erosion rates from low to moderate and to high burn severity (Figure 31, Vieira et al, 2015). These results are even more surprising, considering the additional variability found within reviewed studies regarding types of fires, environments or rainfall simulations methodological procedures. Thus, burn severity can be considered as one of the most important factors influencing post-fire erosive response, and could also be the most reliable and robust predictor of erosion risk.

This review study (Vieira et al., 2015) further indicated that this fire effect is more clearly pronounced for erosion response than for hydrological response. It also showed a tendency towards a smaller increase in runoff generation following burning at high severity than at low or moderate severity. This tendency could be explained by the role of soil water repellency in post-fire overland flow generation (e.g. Crockford et al., 1991; Burch et al., 1989; Shakesby et al., 1993; Doerr et al., 1998; Scott, 2000), in combination with the non-linear relationship of soil water repellency formation with soil heating (e.g. DeBano, 2000; Doerr and Moody, 2004; Varela et al., 2005). However, this possible role of soil water repellency could not be analysed within this review database, since most of the studies did not present data regarding SWR or just initial soil moisture content. Nevertheless, this highlighted a possible research gap, i.e. the need to clarify the link between SWR, burn severity and overland flow generation needs to be clarified. Furthermore, Chapter II.I (Vieira et al., 2015) highlighted the absence of several auxiliary variables of potential interest, among the reviewed studies regarding post-fire rainfall simulation experiments (Table 3). The absence of information regarding burn severity and plots/experiments, restricted the statistical meta-analysis greatly, and should be a concern for the future post-fire research studies.

The impacts of fires in the studied ecosystems are known to last several years, depending on various factors, such as vegetation recovery, post-fire climate conditions, sediment availability, and basin morphology (Rowe et al., 1954; Cerdà, 1998; Moody and

Martin, 2001; Gartner et al., 2004; Shakesby et al., 2007; Sheridan et al., 2007; Cannon et al., 2010, Moody et al., 2013). In this sense, some authors recently referred that the window of disturbance model (Swanson, 1981; Prosser and Williams, 1998, Shakesby and Doerr, 2006), might vary from the original form. That is the case, of MacDonald and Larsen (2009), mentioning that sediment yield peak and recovery period varies according to different burn severity conditions, and Wittenberg and Inbar (2009), that projected an enlargement of the 'classical' window of disturbance with fire recurrence (Figure 31).

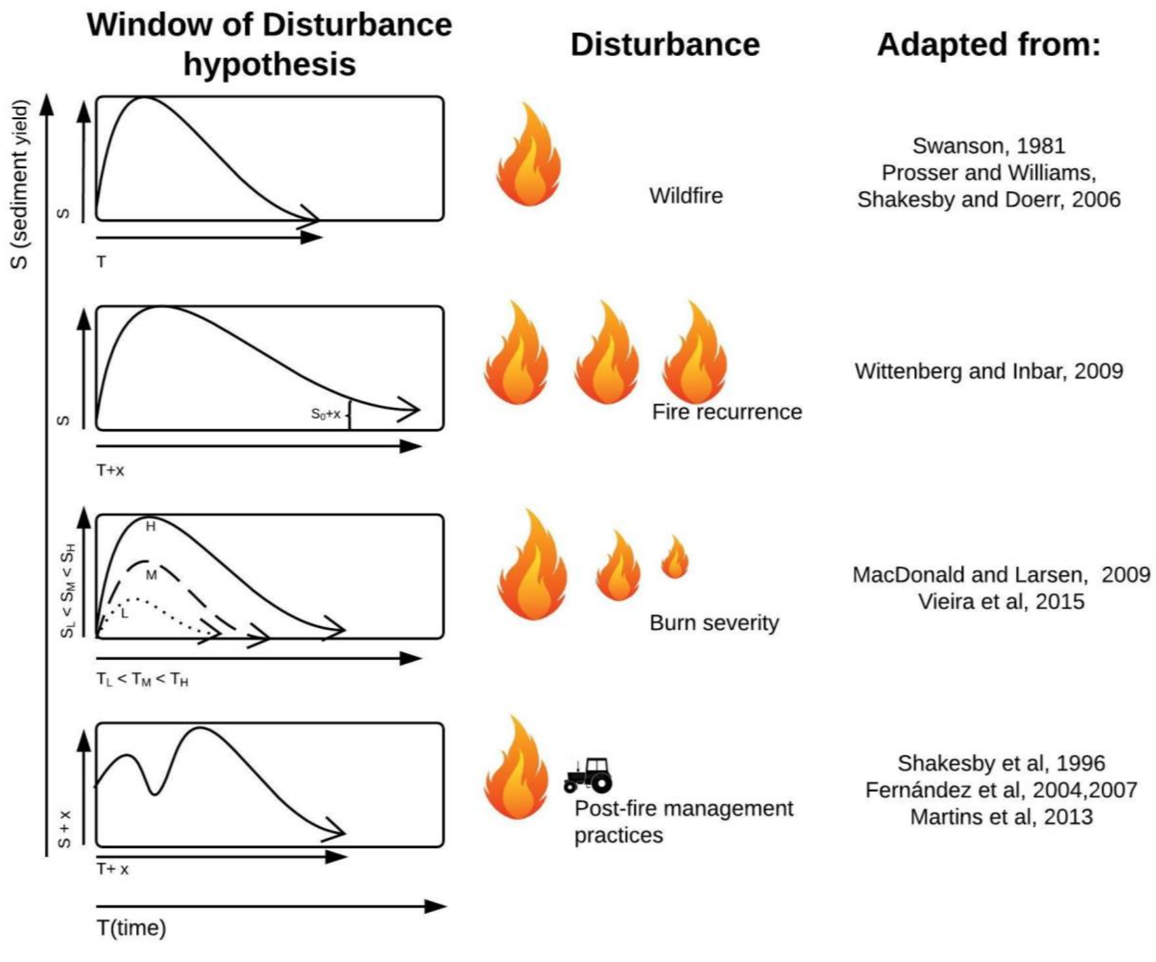


Figure 31 – Post-wildfire soil erosion patterns associated to (a) single wildfire event, (b) after multiple fire events, leading to an increase of the window of disturbance and return to higher background levels, (c) after different burn severities, with the increase of burn severity higher erosion and higher recovery periods are achieved, and (d) wildfire followed by management practices. Note: x: undefined addition of time or sediment yields from the disturbance in comparison to a single wildfire; L: low burn severity, M: moderate burn severity and H: high burn severity.

Due to the reduced number of observations, the compiled RSE's database (Vieira et al, 2015), didn't allow a separated meta-analysis with time-since-fire factor for each individual severity class. In the future, this analysis is expected to be done, determining this way how much the window of disturbance change with burn severity. That could only be possible if the number of the database observations increase, with likely data inclusion from future post-fire RSE's studies. In the case of the Wittenberg and Inbar (2009) model, more studies are required to validate such enlargement of recovery period. Nevertheless unveils a research gap regarding the role of pre-fire disturbances as a contribution for post-fire response.

Chapter II.II, emphasized the possibility of pre-fire disturbances, such as the ones involving a deep soil mobilization together with a previous wildfire, being a major influence in post-fire hydrological and erosive behaviour. Although the burn severity in the studied locations was very similar, the type of management and the time since they were implemented might help to understand their differences. This influence could be seen by the higher hydrological and erosive response at the most recently plowed site (eucalypt contour-plowed), in comparison to another deeply plowed 18 year before (eucalypt downslope plowed), and even higher when compared to the unplowed one (eucalypt unplowed). However, this doesn't explain why annual post-fire hydrological and erosive response during the 4 monitored years, didn't show any clear decline with time since fire. Instead, the observed pattern generally followed the rainfall pattern. Moreover, in the case of runoff coefficients an increase from the first to the 4th year of the study was visible, accompanied by a general decrease of sediment concentration in runoff. This unexpected response was accompanied with a low, and in some sites almost inexistent, vegetation recovery. After 4 years no more that 40% of ground level vegetation grew, whereas the shallow soils were still protected by a very representative stone and litter cover.

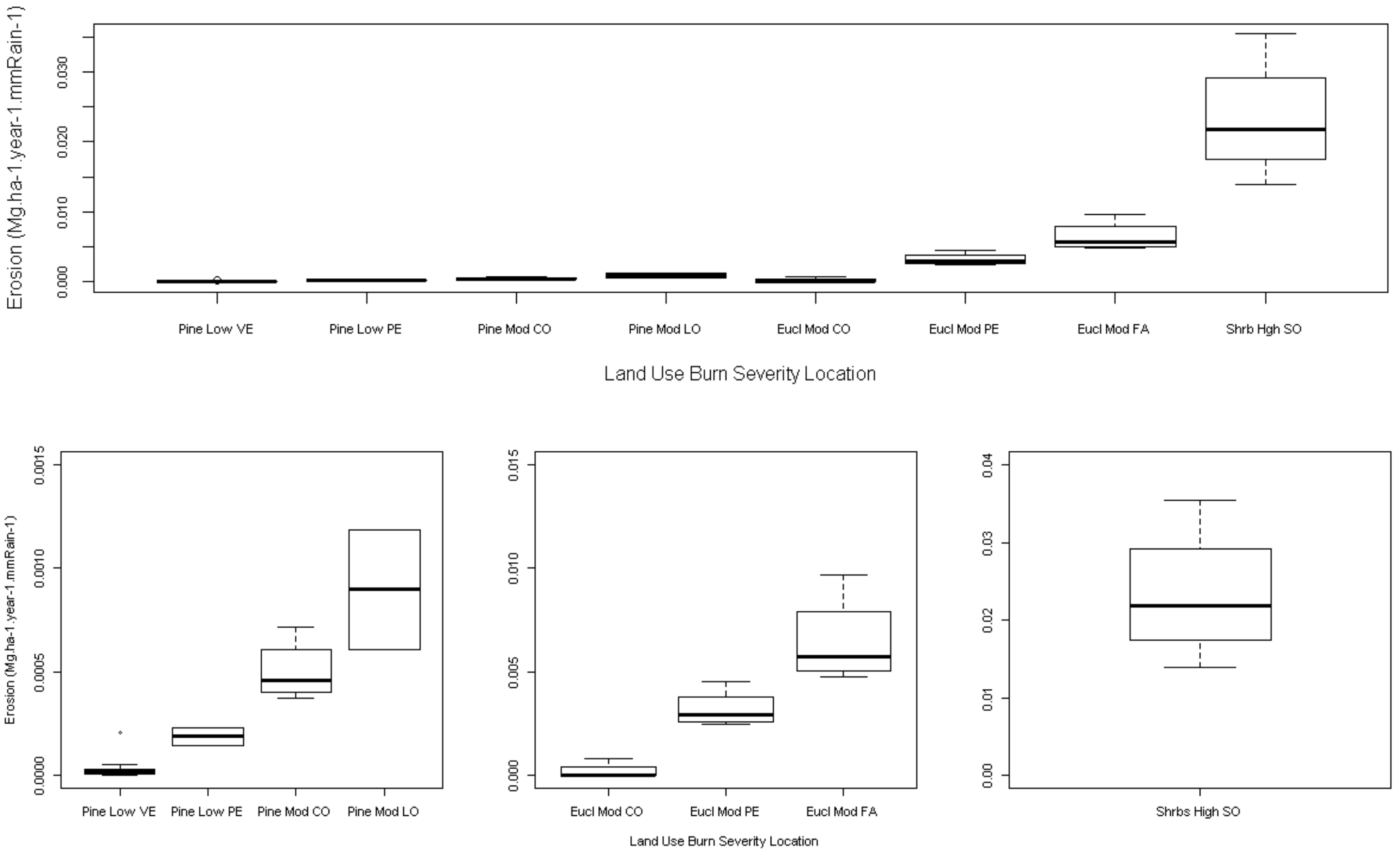
Chapter II.II also highlighted how past and recent forest management practices led to an escalated degradational effect over forest soils in this location. Very often, post-fire related studies all over the Mediterranean basin are also associated to impacting forestry practices (Shakesby et al., 1996, Martins et al., 2013; Malvar et al., 2015), land abandonment (Llovet, 2005; Pausas et al., 2008) and fire recurrence (Wittenberg and Inbar, 2009). The long human exploitation of the Mediterranean soils for cultivation has been the distinctive justification of many authors of post-fire studies when characterizing these soils as highly degraded and with low erosion rates, in comparison to other post-fire erosion rates elsewhere (Shakesby and Doerr, 2006, Shakesby, 2011) or other forms of disturbances (García-Ruiz et al., 2015). These soils have been also described as stony and shallow (also called as skeletal soils), and although sediment losses are low, they are still very important due to the removal thin and vulnerable near surface organic matter and nutrients. Moreover, the soil small depth leads to reduced water retention, promoting runoff and erosion of any sediment and organic matter, thus tending to maintain the degraded state of the soil (Wainwright, 2009).

Looking into the studied sites in chapter II.II, only contour-plowed eucalypt (ECP) presented erosion rates above the tolerable soil losses limits (Verheijen et al., 2009), during the second and the third year after wildfire. This site was also the location that was most recently subjected to soil mobilization (6 -12 years ago) for cultivation purposes, but at the same time is also the only location where the applied management practice is considered less damaging for soil losses when compared to downslope plowing (Morgan, 2005). Nevertheless, during the monitoring period it also showed several signs of elevated degradation, such as an increase of stone cover until 80% and no ground vegetation recovery whatsoever. The other sites, however, revealed lesser erosion rates, but still with evidences of elevated degradation, such as reduced soil depth (7 cm) in the unplowed pine site (PU), absence of significant vegetation recovery after 4 years of study (8%) in the unplowed eucalypt site (EU) and the elevated presence of erosion features among the downslope ridges and furrows as a consequence of a deep plowing from 18 years before in the eucalypt downslope plowing (EDP) site.

Thus, the knowledge of past disturbances seems to be important to understand the future response of that same location, even after such an impacting event as a wildfire. These factors have been poorly explored among post-fire studies, whereas the wildfire usually takes credit for all the enhanced hydrologic and erosive response. Alternatively, as mentioned before, the inclusion of control plots in the experimental design would allow to identify the effect of the wildfire alone.

Analysing the overall erosive response between all the datasets concerning natural rainfall used in this research study, it is possible to verify similar increase in erosion rates with burn severity (Figure 32) as the one from Vieira et al. (2015). It was also verified that moderate burn severity at pine sites, showed a generally lower erosive response in comparison to the eucalypt sites at the same severity. This can be justified with the appearance of the post-fire pine needle cast in all pine sites, without exception (Shakesby et al., 1996; Fernández et al., 2007, Fernández et al., 2011; Prats et al., 2012, chapter II.II), providing efficient mulch over the recently burned area.

Between various unplowed eucalypt sites, subjected to the same burn severity, is possible to observe some variability regarding the erosion response. In this case, past disturbances such as plowing operations, or fire recurrence, could clarify this variability.



Note:
i) land uses: pine, eucalypt, shrubs;
ii) burn severity:low, moderate, high;
iii) locations: Verín (VE), Pessegueiro do Vouga(PE), Colmeal (CO), Lourizela (LO), Falgueirinho (FA), Soutelo (SO).

Figure 32 – Overall erosion rates comparison among used datasets, (a) at several land uses, burn severities and locations; (b) zoomed and separated by land use; during the first year after the wildfire.

Comparison between different land uses subjected to low, moderate and high burn severity, under natural rainfall could highlight how burn severity methodologies require adaptations to different land use or land cover. A quantitative methodology for burn severity impact determination, and their relationship with erosion rates is required within post-fire research. This could work integrated within an existing model application, or just built as an individual burn severity model.

V.II From field measurements to model improvements

One of the main difficulty regarding model improvements for post-fire conditions is to translate fire induced changes into different model inputs. Usually model calibration with a burned area dataset might overcome the limitations associated to site variability or even with burn severity characteristics. However, few post-fire models have been developed exclusively for this purpose, and the others (like the ones used in this study) are mostly hydrological models initially projected for agricultural areas adapted to forest and burned situations. Therefore they require an investment in model adaptation by the alteration or addition of new parameters that were not contemplated before, besides the usual model calibration.

The first attempt performed by these authors to estimate post fire erosion rates within two distinct burn severities, was made with two distinct empirical models RUSLE and revised MMF (Fernández et al., 2010). Burn severity was not included in each model directly as an input, instead, during model evaluation was verified if they were able to capture such difference by themselves. Burn severity differences were captured indirectly from variances in important factors, as ground cover (GC, %), canopy cover (CC) and hydrological soil depth (EHD, m) (Fernández et al., 2010).

Soil alterations due to fire could be easily contemplated in the RUSLE model, but not in the case of the revised MMF. Many authors proposed that these changes should be considered in the K factor, where the soil erodibility would increase with burn severity (Larsen and MacDonald, 2007). However, many soil characteristics cannot be considered for post-fire conditions in the revised MMF, or every time the ground cover is extremely reduced. As a consequence, the revised MMF model considers that sediment losses are transport limited, since the equations for sediment detachment by rainfall would easily overestimate sediment losses in such bare ground situation (Vieira et al., 2014). Nevertheless, the revised MMF was considered to be the most flexible model for post-fire response. Firstly due to its capacity of estimating runoff and erosion rates even not considering post-fire soil changes, while RUSLE is only focused over sediment losses. Secondly, the revised MMF revealed a higher efficiency in comparison to the RUSLE model.

In this first approach an attempt to model post-fire rehabilitation treatments was also performed, but both models revealed a week efficiency results, where RUSLE was worse than revised MMF. The causes for that low efficiency were related to the nature of those rehabilitation treatments. Two treatments (wheat straw and wood chips) concerned two types of mulching, thus could provide an increase of ground cover, and that could be effectively included in both models. While the third treatment (barriers), provided lower cover and higher resistance for overland flow generation. That became very difficult to model, since runoff couldn't be estimated with RUSLE model and the dataset didn't contemplate runoff data for calibration.

The follow-up study of that first attempt concerned only one model and was focused over land use differences, and less over burn severity differences. The chosen model was the revised MMF, due to the high efficiency previously obtained (Fernández et al., 2010), and also because it was possible to calibrate the model for runoff and erosion response. In Vieira et al (2014) several conceptual model configurations were explored, whereas changes in the model such as time step changes and inclusion of soil water repellency into the model were implemented. The separation of the original annual modelling into seasonal modelling was performed in order to accommodate seasonal patterns in runoff and erosion as had been measured in the field. The incorporation of SWR in overland flow generation hasn't been (as far as we know) explicitly incorporated in any modelling of post-fire runoff and erosion.

In the end, the revised MMF model improved from the original annual basis (FP) to the seasonal approach (SM), and from that also improved to the seasonal approach with SWR (SM-SWR). The obtained model efficiencies were consistent with those previously observed in burned areas in Portugal (Vieira et al., 2010), NW of Spain (Fernández et al., 2010) and with the results by Larsen and MacDonald (2007) for periods of 1–10 years after wildfire. There was a significant improvement from the previous version of this model from how was implemented by Fernández et al. (2010) to the version of Vieira et al. (2014). In the last version, it was also effectively possible to model post-fire rehabilitation treatments.

The next step of these model improvements would be, to test and calibrate barriers treatments with runoff data, and also to include pre-fire disturbances or a control sediment yield response, so that the background levels before the wildfire could be represented. Other possible improvement could be related to the response of different land use to burn severity, since pine site with low or moderate burn severity, is followed by a natural and significant needle cast cover formation. And this info might be crucial for a modelling tool with the aim of helping forest management and decision making regarding post-fire rehabilitation treatments.

VI. General conclusions and future perspectives

VI.I Conclusions

This study allowed identifying several factors affecting post-fire runoff and erosion. Regarding the burn severity factor it is possible to conclude that fire occurrence had a significant effect on the hydrological response, increasing the runoff coefficient in comparison to unburnt conditions for each of the burn severity classes. However, runoff effect size did not vary significantly between the three soil burn severities. Furthermore, post-fire erosion response ratio significantly increased with increasing burn severity.

Land use is a factor that might be responsible for runoff generation variability, whereas pine site constantly produced higher runoff amounts and coefficients than the eucalypt one. Erosion rates were also higher in the pine site, however not significantly.

Different plowing techniques and the since they have been implemented might be determinant for both runoff and erosive processes. Plowed sites seem to present a higher erosion risk than the unplowed ones. However, this comparison between plowed and unplowed site, might be influenced by the highly degraded soil conditions the unplowed soils already have been subjected to, the time since plowing and the type of technique used.

Fire recurrence, together with several forest interventions might compromise the sustainability of this site. Several indicators of a continuous degradation have been observed in this area. And due to that, an increased importance is given to the low erosion rates verified in such poor and damaged soils.

Sediment concentration results highlighted a possible decrease of the erosion risk in the area. However, annual runoff and erosion amounts do not seem to be attenuated with time since fire, most likely due to the reduced ground vegetation recovery. Which, together with the constant elevated stone cover during the monitoring period, might indicate: either the period of monitoring didn’t allow observing a similar behaviour to the classic window of disturbance model; or this window of disturbance was possibly enlarged due to past disturbances in this specific place.

Modelling studies allowed to compare post-fire soil losses predicted by the RUSLE and Morgan–Finney model in two burned areas with different levels of fire severity in NW Spain. An acceptable efficiency index was only obtained with the MMF model although it slightly underestimates post-fire soil losses.

Despite their limitations, both models were able to clearly distinguish situations of high and low post-fire erosion risk. This shows the applicability of both models to be used as operational tools in terms of prioritizing management areas. However revised MMF shown a greater potential for post-fire runoff and erosion predictions.

Revised MMF model was able to predict plot-scale runoff and erosion rates of the year following the wildfire with acceptable accuracy in two forest land uses. And these estimations were improved by addressing seasonal changes and by incorporating soil water repellency into the runoff predictions.

It has been shown that the revised MMF model can easily provide a set of simple criteria for management decisions for runoff and erosion in burned areas. The successful predictions of runoff and erosion at the validation site attest to its applicability to other eucalypt and pine sites in Portugal, and suggest that it may well have wider applicability to post-fire conditions for other vegetation types elsewhere in the Mediterranean.

No accurate prediction of soil erosion after soil rehabilitation was achieved with the tested models at first attempt. However, after model modifications to the revised MMF in the second study, its capacity to determine the efficiency of ‘mulch type’ soil rehabilitation treatments in preventing post-wildfire runoff and erosion was improved.

VI.II Future studies and perspectives

The elaboration of this research work highlighted several research gaps that deserve further investigation. That is the case of the burn severity, and its relationship with overland flow generation and SWR occurrence. Additionally, it would be important to know how much impact burn severity has in the ‘window of disturbance’ model regarding sediment yield variation and required post-fire recovery period. It would be also interesting, to test the possibility to determine burn severity quantitatively either by developing a model for that or just by the integrating it in a post-wildfire erosion model.

Other research subjects should be investigated, such as site history or pre-fire disturbances and how can they affect post-fire studies. Moreover, how do these common plowing techniques and management practices affect the Mediterranean basin in the long term?

In the post-fire modelling framework, the improvement of a tool for forest managers regarding erosion risk and to help in the decision making of post-fire rehabilitation measures, applicable for Portugal or elsewhere in the Mediterranean basin it’s still necessary.

The potential of the data collected within this study, can still reach additional research gaps, especially regarding modelling post-fire runoff and erosive response. The integrated fieldwork installation in the Colmeal study area, can reach other scales besides the presented one (micro plot) and be upscalled until the catchment. This would allow understanding and integrating in models, processes transitions and contributions through the scale increase.

Another fact is that this small-catchment field installation that lasted for 4 years could provide calibration inputs for further event modelling within bigger basins just by being an intermediate size between slope and catchment scale (Figure 33). Furthermore, this is one of the few post-fire Mediterranean catchments being monitored for so long.

The inclusion of burn severity in post-fire erosion models is already halfway done, by the fact that models already have the potential to distinguish erosion rates for areas subjected to different burn severities (e.g. Fernández et al., 2010). The quantification of such qualitative metrics would allow distinguishing two areas with the same burn severity that presents distinct responses. That would be the case for different land uses as already seen before, whereas pine sites seem to produce less erosion than eucalypt sites from low to moderate severity, due to the formation of a needle cast. This would also allow to determine a clearer boundary between burn severity classifications, whereas moderate burn severity seemed to be very difficult to classify, since it’s possible to find indicators for high, moderate and low burn severity in the same area.

Regarding the modelling, the determination of the impact of stone cover or stone content in the infiltration process could be important for modelling development in the Mediterranean Basin. Its presence is very representative in these highly degraded soils, and might have a role attenuating the effect of SRW. Model improvement in the rehabilitation treatment with barriers, would be an interesting contribution, in the sense that it’s commonly applied after the wildfires, using the vegetation leftovers or from the surrounding areas. The output of the model would help the managers deciding if those treatments were sufficient to reduce the impact or extra mitigation measures were still required.

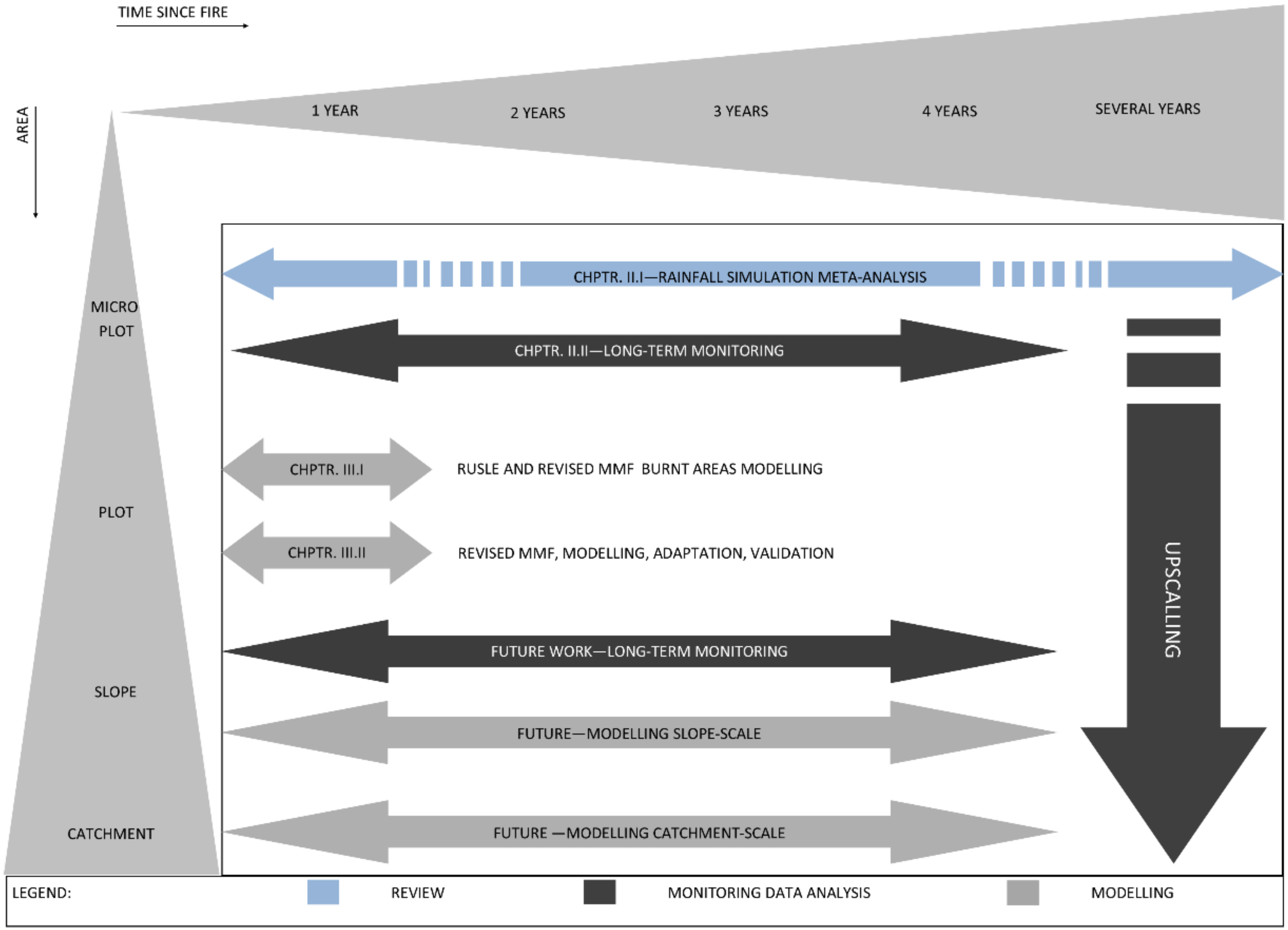


Figure 33 - Temporal and spatial scale for each thesis publication/chapter together with potential future publications.

References

Anderson, H.W., Hoover, M.D., Reinhart, K.G., 1976. Forests and Water: Effects of Forest Management on Floods, Sedimentation, and Water Supply. General Technical Repost PSW-18. United states Department of Agriculture, Forest Service, pacific Southwest Forest and Range Experimental Station, Berkeley CA.

Arcenegui, V., Guerrero, C., Mataix-Solera, J., Mataix-Beneyto, J., Zornoza, R., Morales, J., Mayoral, A.M., 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.

Bachelet, D., Lenihan, J.M., Nielson, R.P., 2007. Wildfires and Global Climate Change. The Importance of Climate Change for Future Wildfire Scenarios in the Western United States, Excerpted From Ebi, Kristie, et al., Regional Impacts of Climate Change, Four Case Studies in the United States, Pew Center on Global Climate Change (20 pp. Available at http://www.pewclimate.org/docUploads/Regional-Impacts-West.pdf, last accessed April 2015).

Barrett, S.W., DeMeo, T., Jones, J.L., Zeiler, J.D., Hutter, L.C., 2006. Assessing ecological departure from reference conditions with the Fire Regime Condition Class (FRCC) mapping tool. In ‘Fuels Management – How to Measure Success’. (Eds PL Andrews, BW Butler) USDA Forest Service, Rocky Mountain Research Station, Proceedings RMRS-P-41 pp. 575–585. (Fort Collins, CO).

Belillas, C.M., Feller, M.C., 1998. Relationships between fire severity and atmospheric and leaching nutrient losses in British Columbia’s coastal Western Hemlock zone forests. International Journal of Wildland Fire 8, 87–101.

Benavides-Solorio, J., MacDonald, L.H., 2001. Post–fire runoff and erosion from simulated rainfall on small plot, Colorado Front Range. Hydrological Processes. 15, 2931–2952.

Benavides-Solorio, J.D.D., MacDonald, L.H., 2005. Measurement and prediction of post-fire erosion at the hillslope. International Journal of Wildland Fire 14, 457–474.

Beschta, R.L., Rhodes, J.J., Kauffman, J.B., Gresswell, R.E., Minshall, G.W., Karr, J.R., Perry, D.A., Hauer, F.R., Frissell, C., 2004. Postfire management on forested public lands of the western United States. Conservation Biology 18, 957–967.

Beukema, S.J., Kurz, W.A., 1998. ‘Vegetation dynamics development tool – Users Guide Version 3.0.’ (ESSA Technologies: Vancouver, BC).

Bisson, M., Fornaciai, A., Coli, A., Mazzarini, F., Pareschi, M.T., 2008. The vegetation resilience after fire (VRAF) index: development, implementation and an illustration from central Italy. International Journal of Applied Earth Observation and Geoinformation 10, 312–329.

Blake, W.H., Doerr, S.H., Shakesby, R.A., Wallbrink, P.J., Humphreys, G.S., Chafer, C.J., 2006. Tracing eroded soil in a burnt water supply catchment, Sydney, Australia: linking magnetic enhancement to soil water repellency. In: Owens, P.N. Collins, A.J. (Eds.), Soil erosion and sediment redistribution in river catchment. CAB International, pp. 62–69.

Burch, G.J., Moore, I.D., Burns, J., 1989. Soil hydrophobic effects on infiltration and catchment runoff. Hydrological Processes. 3, 211–222.

Cannon, S.H., Gartner, J.E., Rupert, M.G., Michael, J.A., Rea, A.H., Parrett, C., 2010. Predicting the probability and volume of post-wildfire debris flows in the intermountain west, USA. Geological Society of America Bulletin. 122, 127–144.

Carey, A., Evans, M., Hann, P., Lintermans, M., MacDonald, T., Ormay, P., Sharp, S., Shorthouse, D., Webb, N., 2003. Wildfires in the ACT 2003: Report on initial impacts on natural ecosystems. Australian Capital Territory, Technical Report 17, Environment ACT, Canberra, Australia. 123 pp.

Carvalho A, Flannigan M, Logan K, Miranda A, Borrego C. 2008. Fire activity in Portugal and its relationship to weather and the Canadian Fire Weather Index System. International Journal of Wildland Fire 17: 328–338.

Carvalho A, Flannigan M, Logan K, Gowman L, Miranda AI, Borrego C, 2010. The impact of spatial resolution on area burned and fire occurrence projections in Portugal under climate change. Climatic Change 98: 177–197

Cerdà, A., 1998. Changes in overland flow and infiltration after a rangeland fire in a Mediterranean scrubland. Hydrological Processes. 12, 1031–1042.

Certini, G., 2005. Effects of fire on properties of forest soils: a review. Oecologia 143, 1-10.

Chafer, C.J., 2008. A comparison of fire severity measures: An Australian example and implications for predicting major areas of soil erosion. Catena. 74, 235–245.

Clark, J., McKinley, R., 2011. Remote sensing and geospatial support to Burned Area Emergency Response Teams. Fire Management Today 71, 15–18.

Connaughton, C.A., 1935. Forest fire and accelerated erosion. Journal of Forestry 33, 751-752.

Crockford, S., Topadalis, S., Richardson, D.P., 1991. Water repellency in a dry sclerophyll forest—measurements and processes. Hydrological Processes. 5, 405–420.

DeBano, L.F., 2000. The role of fire and soil heating on water repellency in wildland environments: A review. Journal of Hydrology. 231–232, 195–206.

DeBano, L.F., Neary, D.G., Ffolliott, P.F., 1998. Fire's Effects on Ecosystems. John Wiley & Sons, New York. 159–196.

Díaz-Fierros, F., Benito, E., Pérez, R., 1987. Evaluation of the U.S.L.E. for prediction of erosion in burnt forest areas in Galicia (NW Spain). Catena 14: 189-199.

Doerr, S.H., Moody, J., 2004. Hydrological impacts of soil water repellency: on spatial and temporal uncertainties. Hydrological Processes 18: 829– 832.

Doerr, S.H., Shakesby, R.A., Walsh, R.P.D., 1998. Spatial variability of soil hydrophobicity in fire-prone eucalyptus and pine forests, Portugal. Soil Science. 163, 313–324.

Doerr, S.H., Shakesby, R.A., Blake, W.H., Chafer, C.J., Humphreys, G.S., Wallbrink, P.J., 2006. Effects of differing wildfire severities on soil wettability and implications for hydrological responses. Journal of Hydrology. 319, 295–311.

Esteves, T.C.J., Kirkby, M.J., Shakesby, R.A., Ferreira, A.J.D., Soares, J.A.A., Irvine, B.J., Ferreira, C.S.S., Coelho, C.O.A., Bento, C.P.M., Carreiras, M.A. 2012. Mitigating land degradation caused by wildfire: Application of the PESERA model to fire-affected sites in central Portugal. Geoderma, doi:10.1016/j.geoderma.2012.01.001

European Commission, 2011. Forest fires in Europe, Middle East and North Africa 2011. JRC Scientific and Technical Report, EUR 25483 EN. European Commission. 108 pp.

Fernández, C., Vega, J.A., Gras, J.M., Fontúrbel, T., Cuiñas, P., Dambrine, E., Alonso, M., 2004. Soil erosion after Eucalyptus globulus clearcutting: differences between logging slash disposal treatments. Forest Ecology and Management 195: 85–95. doi:10.1016/J.FORECO.2004.02.052

Fernández, C., Vega, J.A., Fontúrbel, T., Pérez-Gorostiaga, P., Jiménez, E., Madrigal, J., 2007. Effects of wildfire, salvage logging and slash treatments on soil degradation. Land Degradation and Development 18, 591–607.

Fernández, C., Vega, J.A., Vieira, D.C.S., 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, 774-87.

Fernández, C., Vega, J.A., Jiménez, E., Fontúrbel, T., 2011. Effectiveness of three post-fire treatments at reducing soil erosion in Galicia (NW Spain). *International Journal of Wildland Fire* 20, 104–114. http://dx.doi.org/10.1071/WF09010

Ferreira, A.J.D., Coelho, C.O.A., Ritsema, C.J., Boulet, A.K., Keizer, J.J., 2008. Soil and water degradation processes in burned areas: lessons learned from a nested approach. Catena 74: 273–285.

Flanagan, D.C., Livingston, S.J., eds. 1995. WEPP user summary. NSERL Report No. 11. West Lafayette, IN: U.S. Department of Agriculture, Agricultural Research Station, National Soil Erosion Research Laboratory. 131 p.

Flannigan, M.D., Stocks, B.J., Wotton, B.M., 2000. Climate change and forest fires. Science of the Total Environment 262, 221–229.

Fox, D.M., Maselli, F., Carrega, P., 2008. Using SPOT images and field sampling to map burn severity and vegetation factors affecting post forest fire erosion risk. Catena 75, 326–335. doi:10.1016/J.CATENA.2008.08.001

García-Ruiz, J.M., Beguería, S., Nadal-Romero, E., González-Hidalgo, J.C., Lana-Renault, N., Sanjuán, Y., 2015. A Meta-Analysis of soil erosion rates across the world, Geomorphology, Available online 17 March 2015, ISSN 0169-555X, http://dx.doi.org/10.1016/j.geomorph.2015.03.008.

Gartner, J.E., Bigio, E.R., Cannon, S.H., 2004. Compilation of post wildfire runoff-event data from the western United States. (online USGS Open-File Report 2004–1085, http://pubs.usgs.gov/of/2004/1085 [accessed: 17 July 2013]).

Gimeno-Garcia, E., Andrea, V., Rubio, J.L., 2007. Influence of vegetation recovery on water erosion at short and medium-term after experimental fires in a Mediterranean shrubland. Catena. 69, 150–160.

Good, P., Moriondo, M., Giannakopoulos, C., Bindi, M., 2008. The meteorological conditions associated with extreme fire risk in Italy and Greece: Relevance to climate model studies. International Journal of Wildland Fire 17: 155–165.

Goode, J.R., Luce, C.H., Buffington, J.M., 2012. Enhanced sediment delivery in a changing climate in semi-arid mountain basins: implications for water resource management and aquatic habitat in the northern Rocky Mountains. Geomorphology 139–140, 1–15.

Guay, T., 2011. Rapid assessment of vegetation condition after wildfire. Fire Management Today 71, 5–8.

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 (3), 1029–1037.

Hammill, K., Bradstock, R.A., 2006. Remote sensing of fire severity in the Blue Mountains: influence of vegetation type and inferring fire intensity. International Journal of Wildland Fire 15, 213 - 226.

Harding, A., Palutikof, J., Holt, T., 2009. The climate system. In: Woodward, J. (Ed.), The Physical Geography of the Mediterranean. Oxford University Press, Oxford, pp. 69–88.

Hendricks, B.A., Johnson, J.M., 1944. Effects of fire on steep mountain slopes in central Arizona. Journal of Forestry 42, 568-571.

ICNF, 2013. 6º Inventário Florestal Nacional. Áreas dos usos do solo e das espécies florestais de Portugal continental. Resultados preliminares. Fevereiro 2013. Instituto da Conservação da Natureza e Florestas. (Available at http://www.icnf.pt/portal/florestas/ifn/resource/ficheiros/ifn/ifn6-res-prelimv1-1 , last accessed April 2015)

Jain, T.B.; Pilliod, D.S.; Graham, R.T.; Lentile, L.B.; Sandquist, J.E. 2012. Index for characterizing post-fire soil environments in temperate coniferous forests. Forests. 3: 445-466.

Keane, R.E., Holsinger, L., Pratt, S., 2006. Simulating historical landscape dynamics using the landscape fire succession model LANDSUM version 4.0. USDA Forest Service, Rocky Mountain Research Station, RMRS-GTR-171CD. (Fort Collins, CO).

Keeley, J.E., 2009. Fire intensity, fire severity and burn severity: a brief review and suggested usage. International Journal of Wildland Fire. 18, 116–126.

Key, C.H., Benson, N.C., 2006. Landscape Assessment: Ground measure of severity, the Composite Burn Index; and Remote sensing of severity, the Normalized Burn Ratio. In D.C. Lutes; R.E. Keane; J.F. Caratti; C.H. Key; N.C. Benson; S. Sutherland; and L.J. Gangi. 2006. FIREMON: Fire Effects Monitoring and Inventory System. USDA Forest Service, Rocky Mountain Research Station, Ogden, UT. Gen. Tech. Rep. RMRS-GTR-164-CD: LA 1-51.

Kirkby, M.J., Irvine, B.J., Jones, R.J.A., Govers, G., Boer, M., Cerdan, O., Daroussin, J., Gobin, A., Grimm, M., Le Bissonnais, Y., Kosmas, C., Mantel, S., Puigdefabregas, J, van Lynden, G., 2008. The PESERA coarse scale erosion model for Europe. I. — model rationale and implementation. European Journal of Soil Science 59: 1293–1306.

Kotliar, N.B., Reynolds, E.W., Deutschman, D.H., 2008. American Three-toed Woodpecker response to burn severity and prey. Fire Ecology Special Issue 4, 26–45.

Kutiel, P., Naveh, Z., 1987. The effect of fire on nutrients in a pine forest soil. Plant and Soil 104, 269-274.

Larsen, I.J., MacDonald, L.H., 2007. Predicting post-fire sediment yields at the hillslope scale: Testing RUSLE and Disturbed WEPP. Water Resources Research 43: W11412.

Lentile, L.B., Holden, Z.A., Smith, A.M.S., Falkowski, M.J., Hudak, A.T., Morgan, P., Lewis, S.A., Gessler, P.E., Benson, N.C., 2006. Remote sensing techniques to assess active fire characteristics and post-fire effects. International Journal of Wildland Fire 15, 319–345.

Lewis, S.A., Wu, J.Q., Robichaud, P.R., 2006. Assessing burn severity and comparing soil water repellency, Hayman fire, Colorado. Hydrological Processes 20, 1–16.

Littell, J.S., McKenzie, D., Peterson, D.L., Westerling, A.L., 2009. Climate and wildfire area burned in western U.S. ecoprovinces, 1916–2003. Ecological Applications 19, 1003–1021.

Lloret, F., Piñol, J., Castellnou, M., 2009. Wildfires. In: Woodward, J. (Ed.), The physical Geography of the Mediterranean. Oxford university Press, Oxford, pp.541-560.

Llovet, J., 2005. Degradación del suelo posterior al fuego en condiciones mediterráneas. Identificación de factores de riesgo. PhD thesis, University of Alacant

Lotan, J.E., Alexander, M.E., Arno, S.F., [and others]., 1981. Effects of fire on flora: A state-of-knowledge review: Proceedings of the national fire effects workshop; 1978 April 10-14; Denver, CO. Gen. Tech. Rep. WO-16. Washington, DC: U.S. Department of Agriculture, Forest Service. 71 p.

Lyon, L.J.; Crawford, H.S.; Czuhai, E.; Fredriksen, R.L.; Harlow, R.F.; Metz, L.J.; Pearson, H.A. 1978. Effects of fire on fauna. Gen. Tech. Rep. WO-6. Washington, DC: U.S. Department of Agriculture, Forest Service. 22 p.

MacDonald, L.H., Larsen, I.J., 2009. Effects of forest fires and post-fire rehabilitation: a Colorado Case study. In: Fire Effects on Soils and Restoration Strategies, edited by A. Cerda and P.R. Robichaud. Science Publishers, Enfield, NH, pp. 423-452.

Maia, P., Pausas, J.G., Arcenegui, V., Guerrero, C., Pérez-Bejarano, A., Mataix-Solera, J., Varela, M.E.T., Fernandes, I., Pedrosa, E.T., Keizer, J.J., 2012. Wildfire effects on the soil seed bank of a maritime pine stand — The importance of fire severity. Geoderma. 191 (2012) 80–88.

Malvar, M.C., Prats, S.A., Keizer, J.J., 2015. Runoff and inter-rill erosion affected by Wildfire and pre-fire ploughing in eucalypt plantations of north-Central Portugal. Land Degradation and Development.

Martin, R.E., Anderson, H.E., Boyer, W.D., [and others]., 1979. Effects of fire on fuels: A state-of-knowledge review: Proceedings of the national fire effects workshop; 1978 April 10-14; Denver, CO. Gen. Tech. Rep. WO-13. Washington, DC:U.S. Department of Agriculture, Forest Service. 64 p.

Martins, M.A.S., Machado, A. I., Serpa D., Prats, S.A., Faria, S.R., Varela, M.E.T., Gonzalez-Pelayo, O., Keizer, J.J., 2013. Runoff and inter-rill erosion in a Maritime

Pine and a eucalypt plantation following wildfire and terracing in north-central Portugal. Journal of Hydrology and Hydromechanics 61, 4, 261-269

Mayor, A.G., Bautista, S., Llovet, J., Bellot, J., 2007. Post-fire hydrological and erosional responses of a Mediterranean landscape: seven years of catchmentscale dynamics. Catena. 71, 68–75.

Meyer, G.A., Pierce, J.L., Wood, S.H., Jull, A.J.T., 2001. Fire, storms, and erosional events in the Idaho Batholith. Hydrological Processes 15: 3025– 3038.

Miller, J.D., Nyhan, J.W., Yool, S.R., 2003. Modeling potential erosion due to the Cerro Grande Fire with a GIS-biased implementation of the Revised Universal Soil Loss Equation. International Journal of Wildland Fire 12, 85–100.

MMA, 2010. Los incendios forestales en España. Ministerio de Medio Ambiente, Madrid.

Moody, J.A., Martin, D.A., 2001. Initial hydrologic and geomorphic response following a wildfire in the Colorado Front Range. Earth Surface Processes and Landforms 26, 1049–1070.

Moody, J.A., Martin, D.A., 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, J.A., Martin, D.A., Cannon, S.H., 2008. Post-wildfire erosion response in two geologic terrains in the western USA. Geomorphology 95: 103-118.

Moody, J.A., Shakesby, R.A., Robichaud, P.R., Cannon, S.H., Martin, D.A., 2013. Current research issues related to post-wildfire runoff and erosion processes. Earth-Science Reviews. 122, 10–37.

Moreira, F., Vaz, P.J.G., Catry, F.X., Silva, J.S., 2009. Regional variations in wildfire susceptibility of land cover types in Portugal: implications for landscape management to minimize fire hazard. Int. J. Wildland Fire 18, 563–574.

Moreno, J.M., 2005. Impacto sobre los riesgos naturales de origen climático. Riesgo de incendios forestales. In Moreno JM (coord.), Evaluación preliminar de los impactos en España por efecto del cambio climático. Ministerio de Medio Ambiente (Madrid); 1–64.

Moreno, J.M., 2009. Impacts on potential wildfire risk due to changes in climate. In Birot Y (ed.). Living with Wildfires: What Science Can Tell Us. A Contribution to the Science-Policy Dialogue, EFI Discussion Paper 15; European Forest Institute. Joensuu.

Moreno, J.M., Vázquez, A., Vélez, R., 1998. Recent history of forest fires in Spain. In: Moreno, J.M. (Ed.), Large Forest Fires. Backhuys Publishers, Leiden, pp. 159-185.

Morgan, R.P.C., 2001. A simple approach to soil loss prediction: A revised Morgan–Morgan–Finney model. Catena 44: 305–322.

Morgan, R.P.C., 2005. Soil Erosion and Conservation. Blackwell, Oxford.

Morgan, P., Hardy, C.C., Swetnam, T.W., Rollins, M.G., Long, D.G., 2001. Mapping fire regimes across time and space: understanding coarse and fine-scale fire patterns. International Journal of Wildland Fire 10, 329–342.

Morgan, P., Keane, R.E., Dillon, G.K., Jain, T.B., Hudak, A.T., Karau, E.C., Sikkink, P.G., Holden, Z.A., Strand, E.K., 2014. Challenges of assessing fire and burn severity using field measures, remote sensing and modelling. International Journal of Wildland Fire 23, 1045–1060.Moritz et al., 2010

Naveh, Z., 1990. Fire in the Mediterranean – a landscape ecological perspective. In: Goldammer, J.G. and Jenkins, M.J. (eds.), Fire in ecosystem dynamics. Pp. 1-20. SPB Academic Publishing, The Hague, Netherlands

Nearing, M.A., Foster, G.R., Lane, L.J., Finkner, S.C., 1989. A process-based soil erosion model for USDA-water erosion prediction project technology. American Society of Agricultural Engineers 32: 1587–1593.

Neary, D.G., Gottfried, G.J., 2002. Fires and floods: post-fire watershed responses. In: Viegas, V.R. (Ed.), Forest Fire Research &Wildland Fire Safety. Millpress, Rotterdam, pp. 1–9.

Neary, D.G., Klopatek, C.C., DeBano, L.F., Ffolliott, P.F., 1999. Fire effects on belowground sustainability: a review and synthesis. Forest Ecology and Management. 122, 51–71.

Neary, D.G., Ryan, K.C., DeBano, L.F., 2005. Wildland fire in ecosystems. Effects of fire on soil and water. USDA Forest Service, Rocky Mountain Research Station. General Technical Report RMRS-GTR-42-vol 4, Ogden, UT.

NWCG, 2014. National Wildfire Coordinating Group (2014) Glossary of Wildland Fire Terminology. (Available at http://www.nwcg.gov.sci-hub.org/pms/pubs/glossary/ last accessed April 2015)

Nicks, A.D., Lane, L.J., Gander, G.A., 1995. Weather generator. In: Flanagan, D.C., Nearing, M.A. (Eds.), U.S. Department of Agriculture—Water Erosion Prediction Project Hillslope Profile and Watershed Model Documentation. NSERL Report No. 10. U.S. Department of Agriculture, Agricultural Research Service, National Soil Erosion Research Laboratory, West Lafayette, IN, pp. 1.1–2.22.

Pardini, G., Gispert, M., Dunjó, G., 2003. Erosion and nutrient depletions in five Mediterranean soils of NE Spain under different land use. The Science of the Total Environment 309, 213-224.

Parsons 2003

Pausas, J.G., 2004. Changes in fire and climate in the eastern Iberian Peninsula (Mediterranean Basin). Climate Change 63, 337-350.

Pausas, J.G., Ouadah, N., Ferran, A., Gimeno, T., Vallejo, R., 2003. Fire severity and seedling establishment in Pinus halepensis woodlands, eastern Iberian Peninsula. Plant Ecology 169, 205–213.

Pausas, J.G., Llovet, J., Rodrigo, A., Vallejo, R., 2008. Are wildfires a disaster in the Mediterranean basin?-A review. International Journal of Wildland Fire. 17:713-723.

Pereira, J., Carreira, J., Silva, J., Vasconcelos, M., 2006a. Alguns conceitos basicos sobre os fogos rurais em Portugal. In: Pereira J.S., Pereira J.M.C. et al. (eds.), Incêndios florestais em Portugal – caracterização, impactes e prevenção, pg 133-16, ISAPress, Lisbon.

Pereira, J., Correia, A.V., Correia, A.C., Ferreira, M., Onofre, N., Freitas, H., Godinho, F., 2006b. Florestas e biodiversidade. In: Santos, F., Miranda, P. (Eds.), Alterações climáticas em Portugal—cenários, impactos e medidas de adaptação (Projecto SIAM II). Gradiva, Lisbon, pp. 301–343.

Pierce, J., Meyer, G., 2008. Long-term fire history for alluvial fan sediments: the role of drought and climate variability, and implications for management of Rocky Mountain forests. International Journal of Wildland Fire 17, 84–95.

Pierson, F.B., Robichaud, P.R., Spaeth, K.E., 2001. Spatial and temporal effects of wildfire on the hydrology of a steep rangeland watershed. Hydrological Processes. 15, 2905–2916.

Prats, S.A., MacDonald, L.H., Monteiro, M., Ferreira, A.J.D., Coelho, C.O.A., Keizer, J.J., 2012. Effectiveness of forest residue mulching in reducing post-fire runoff and erosion in a pine and a eucalypt plantation in north-central Portugal, Geoderma 191: 115-124.

Prosser, I.P., Williams, L., 1998. The effect of wildfire on runoff and erosion in native Eucalyptus forest. Hydrological Processes. 12, 251–265.

Renard, K.G., Foster, G.R., Weesies, G.A., Mc Cool, D.K., Yoder, D.C., (co-ordinators). 1997. Predicting soil erosion by water: A guide to conservation planning with the revised universal soil loss equation (RUSLE). Agriculture Handbook n- 703. USDA, Natural Resources Conservation Service: Washington, DC.

Robichaud, P.R., Brown, R.E., 1999. What happened after the smoke cleared: onsite erosion rates after a wildfire in eastern Oregon. in: Olsen, D.S., Potyondy, J.P. (Eds.), Proceedings Wildland Hydrology Conference, June 1999, Bozeman, Mt Hernon, Virginia. American Water Resource Association, pp. 419– 426.

Robichaud, P.R., 2000. Fire effects on infiltration rates after prescribed fire in northern Rocky Mountain forests, USA. Journal of Hydrology 231–232, 220–229.

Robichaud, P.R.; Beyers, J.L.; Neary, D.G., 2000. Evaluating the effectiveness of postfire rehabilitation treatments. Gen. Tech. Rep. RMRS-GTR-63. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 85 p.

Robichaud, P.R., MacDonald, L., Freeouf, J., Neary, D., Martin, D., Ashmun, L., 2003. Postfire rehabilitation of the Hayman Fire. In ‘Hayman Fire Case Study’. (Ed. RT Graham) pp. 293–313. USDA Forest Service, Rocky Mountain Research Station RMRS-GTR-114. (Ogden, UT).

Robichaud, P.R., Elliot, W.J., Pierson, F.B., Hall, D.E. Moffet, C.A., Ashmun, L.E., 2007. Predicting postfire erosion and mitigation effectiveness with a web-based probabilistic erosion model. Catena. 71, 229–241.

Rowe, P.B., Countryman, C.M., Storey, H.C., 1954. Hydrologic Analysis Used to Determine Effects of Fire on Peak Discharge and Erosion Rates in Southern California Watersheds. USDA Forest Service, California Forest and Range Experimental Station (49 pp.).

Ryan, K.C., Noste, N.V., 1985. Evaluating prescribed fires. In 'Wilderness Fire Symposium Missoula, MT', 22–26 September 2983, Missoula, MT. (Eds JE Lotan, BM Kilgore, WC Fischer, RW Mutch) pp. 230–237. USDA Forest Service, Intermountain Research Station, INT-182. (Ogden, UT).

Soto, B., Díaz-Fierros, F., 1998. Runoff and soil erosion from areas of burnt scrub: comparison of experimental results with those predicted by the WEPP model. Catena 31: 257-270.

Sandberg, D.V., Pierovich, J.M., Fox, D.G., Ross, E.W., 1979. Effects of fire on air: a State-of-knowledge review. Gen. Tech. Rep. WO- 9. Washington, D.C.: U.S. Department of Agriculture, Forest Service. 40 p.

San-Miguel, J., Camia, A. 2009. Forest fires at a glance: facts, figures and trends in the EU. In: Birot, Y. (Ed.), Living with wildfires: what can science tell us, Chp. 1, pp 11–19. European Forest Institute, Discussion Paper 15.

Schmidt, M.W., Noack, A.G., 2000. Black carbon in soils and sediments: analysis, distribution, implications and current challenges. Global Biochemical Cycles. 14, 777–793.

Scott, D.F., 2000. Soil wettability in forested catchments in South Africa: as measured in different methods and as affected by vegetation cover and soil characteristics. Journal of Hydrology 231-232, 87-104.

Scott and Van Wyk, 1990

Scott et al., 2004

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

Shakesby, R.A., Coelho, C.O.A., Ferreira, A.D., Terry, J.P., Walsh, R.P.D., 1993. Wildfire impacts on soil erosion and hydrology in wet Mediterranean forest, Portugal. International Journal of Wildland Fire. 3, 95–110.

Shakesby, R.A., Coelho, C.O.A., Ferreira, A.D., Terry, J.P., Walsh, R.P.D., 1994. Fire, post-burn land management practice and soil erosion response curves in eucalyptus and pine forests, north-central Portugal. In Sala, Rubio, J.L. (Eds.), soil Erosion as a Consequence of Forest Fires. Geoforma, logroño, Spain, pp. 111-132.

Shakesby, R.A., Boakes, D.J., Coelho, C.O.A., Gonçalves, A.J.B., Walsh, R.P.D., 1996. Limiting the soil degradational impacts of wildfire in pine and eucalypt forests in Portugal. Applied Geography 16: 337–356.

Shakesby, R.A., Doerr, S.H., 2006. Wildfire as a hydrological and geomorphological agent. Earth Science Reviews. 74, 269–307.

Shakesby, R.A., Wallbrink, P.J., Doerr, S.H., English, P.M., Chafer, C.J., Humphreys, G.S., Blake, W.H., Tomkins, K.M., 2007. Distinctiveness of wildfire effects on soil erosion in south-east Australian eucalypt forests assessed in a global context. Forest Ecology and Management. 238, 347–364.

Sheridan, G.J., Lane, P.N.J., Noske, P.J., 2007. Quantification of hillslope runoff and erosion processes before and after wildfire in a wet Eucalyptus forest. Journal of Hydrology. 343, 12–28.

Swanson, F.J., 1981. Fire and geomorphic processes. In: Mooney, H.A. Bonnicksen, T.M. Christiansen, N.L. Lotan, J.E. Reiners, W.A. (Eds.), United States Department of Agriculture, Forest Service, General Technical Report WO-26. United States Government Planning Office, Washington DC, pp. 401–421.

Tiedemann, A.R., Conrad, C.E., Dieterich, J.H. Hornbeck, J.W., Megahan, W.F., Viereck, L.A., Wade, D.D., 1979. Effects of Fire on Water. A State-of-art Review. United States Department of Agriculture, Forest Service, General Technical Report WO-10. 28 pp.

Úbeda, X., Outeiro, L., 2009. Physical and chemical effects of fire on soil. In: A. Cerdà, P.R. Robichaud (Eds.), Fire Effects on Soils and Restoration Strategies. Science Publishers, Enfield, NH, USA, pp. 105-132.

Varela, M.E., Benito, E., de Blas, E., 2005. Impact of wildfires on surface water repellency in soils of northwest Spain. Hydrological Processes. 19, 3649-3657.

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

Vega, J.A., Fernández, C., Jiménez, E., Ruiz, A.D., 2009. Evidencias de cambio climático en Galicia a través de la tendencia de los índices de peligro de incendios forestales. In: Evidencias e impactos del cambio climático en Galicia. Xunta de Galicia, Santiago de Compostela, pp 173-194.

Vega, J.A., Fontúrbel, T., Merino, A., Fernández, C., Ferreiro, A., Jiménez, E., 2013. 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.

Verheijen, F.G.A., Jones, R.J.A., Rickson, R.J., Smith, C.J., 2009. Tolerable versus actual soil erosion rates in Europe. Earth-Science Reviews 94: 23–38.

Vieira, D.C.S., Malvar, M.C., Nunes, J.P., Keizer, J.J. 2010. Modelação da erosão do solo e sua verificação numa área florestal recentemente ardida em Açores, centro-norte de Portugal. Territorium 17: 127-134.

Vieira, D.C.S., Prats, S.A., Nunes, J.P., Shakesby, R.A., Coelho, C.O.A., Keizer, J.J. 2014. Modelling runoff and erosion, and their mitigation, in burned Portuguese forest using the revised Morgan-Morgan-Finney model. Forest Ecology and Management 314: 150-165 doi: 10.1016/j.foreco.2013.12.006

Vieira, D.C.S., Fernández, C., Vega, J.A., Keizer, J.J., 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, Volume 523, April 2015, Pages 452-464, ISSN 0022-1694, http://dx.doi.org/10.1016/j.jhydrol.2015.01.071

Wainwright, J., 2009. Weathering, soils, and slope processes. In: Woodward, J. (Ed.), The Physical Geography of the Mediterranean. Oxford University Press, Oxford, pp. 169-202.

Wells, C.G., Campbell, R.E., DeBano, L.F., Lewis, C.E., Fredriksen, R.L., Franklin, E.C., Froelich, R.C., Dunn, P.H., 1979. Effects of fire on soil. A state-of-knowledge review. Gen. Tech. Rep. WO-7. Forest Serv., US Department of Agriculture: Washington, DC; 34.

Westerling, A.L., Gerchunov, A., Brown, T.J., Cayan, D.R., Dettinger, M.D., 2003. Climate and wildfire in the western United States. Bulletin of the American Meteorological Society 84 (5), 595–604.

Westerling, A.L., Turner, M.G., Smithwick, E.A.H., Romme, W.H., Ryan, M.G., 2011. Continued warming could transform Greater Yellowstone fire regimes by mid-21st century. Proceedings of the National Academy of Sciences 108 (32), 13165–13170.

Wischmeier, W.H., Smith, D.D., 1978. Predicting rainfall-erosion losses- a guide to conservation planning. Agriculture Handbook. No. 537. USDA. Washington, D.C; 58.

Wittenberg, L., Inbar, M., 2009. The role of fire disturbance on runoff and erosion processes — a long-term approach, Mt. Carmel case study, Israel. Geographical Research 47, 46–56.

Zarriello, T.J., Knick, S.T., Rotenberry, J.T., 1995. Producing a burn/disturbance map for the Snake River Birds of Prey National Conservation Area. In ‘Snake River Birds of Prey National Conservation Area Research and Monitoring Annual Report’. (Boise, ID)