Universidade de Aveiro Departamento de Ambiente e Ordenamento
Universidade do Porto Instituto de Ciências Biomédicas Abel Salazar
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Sergio Prats Alegre

Mitigação da erosão do solo após os incêndios florestais

Soil erosion mitigation following forest wildfires

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Dissertação Apresentada para cumprimento dos requisitosnecessários à obtenção do grau de Doutoramento Europeu em Ciências do Mar e do Ambiente Especialidade em Planeamento e Gestão Ambiental; Programa Doutoral PROMAR da Universidade do Porto (Instituto de Ciências Biomédicas de Abel Salazar e Faculdade de Ciências) e da Universidade de Aveiro, realizada sob orientação do Doutor Jan Jacob Keizer, Investigador Auxiliar do Departamento de Ambiente e Ordenamento da Universidade de Aveiro e do Doutor António José Dinis Ferreira, Professor adjunto do Departamento de Ambiente da Escola Superior Agrária de Coimbra do Instituto Politécnico de Coimbra.

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palavras-chave

incêndios florestais, erosão, escorrência, repelência, mulch, efetividade, turvação.

resumo

O presente trabalho centra-se na avaliação da efetividade de quatro técnicas de controlo da escorrência e da erosão após incêndios florestais, adaptadas para o caso de povoamentos florestais no centro norte de Portugal. A seleção e desenvolvimento das técnicas foi efetuada após revisão bibliográfica alargada, mas sobre tudo após a comprovação no campo, efetuando simulações de chuva, de quais os fatores determinantes da erosão nos solos típicos do centro norte do País, caracterizados por serem altamente repelentes ainda antes dos incêndios.

O “mulch” com restos de casca de eucalipto triturada foi um tratamento pioneiro nunca antes testado e deu bons resultados no controlo da escorrência e da erosão em eucaliptais ardidos. O “mulch” com restos florestais não triturados (ramos, paus e folhas) aplicados em um pinhal recentemente ardido não pode ser bem testado devido à protecção natural que forneceram as agulhas do pinheiro que caíram das árvores. No entanto, a sua alta taxa de aplicação desaconselham a sua utilização. O “hidromulch”, uma variante do “mulch” composto por água, fibras orgânicas e sementes utilizadona restauração de taludes e pedreiras, também deu resultados altamente efetivos e foi indicado para o tratamento de áreas especialmente sensíveis. Por outro lado, a utilização de poliacrilamidas (PAM), um agente aglutinante com bastante êxito na redução da erosão em terrenos agrícolas e com alto potencial devido ao seu baixo custo, não obteve resultados satisfatórios, uma vez que não alterou o principal fator envolvido na geração da erosão: o coberto do solo.

No decorrer destas experiências, foi ainda desenvolvido um sensor óptico de turvação que permite facilitar a determinação da concentração de sedimentos nas amostras de escorrência das parcelas de erosão. Atualmente, foi realizado o pedido de patente de um novo protótipo de sensor de turvação da água mais desenvolvido.

keywords

wildfires, soil erosion, runoff, soil water repellence, mulch, effectiveness, turbidity.

abstract

This study aims to measure the effectiveness of four post-fire emergency techniques for reducing overland flow and soil erosion on the central-Portugal typical forest. The selection and development of these techniques was based on the review of the scientific background, but specially after checking throughout field rainfall simulation experiments which factors were the key for runoff and soil erosion on the specific case of high repellent soils.

The forest residue mulch, a new treatment never tested before, was highly effective in reducing runoff and soil erosion in recently burnt eucalypt forest. The logging slash mulch had no obvious effect, but it was attributed to the small amounts of runoff and sediments that the untreated plots produced due to the extensive needle cast following a low severity fire. The hydromulch, a mixture of water, organic fibres, seeds, nutrients and a surfactant used in cutted slopes rehabilitation was also highly successful and was specially indicated for especially sensible areas. The utilization of polyacrylamides, a chemical agent with good performance in agricultural erosion, was not successful in post-fire runoff and soil erosion control, once that did not alter the most important key factor for soil erosion: the ground cover.

The development of a new fibre optic turbidity sensor was a successful development on the soil erosion determination methodology, and its patent is being processed in the mean time.

CONTENTS

CHAPTER 1

Introduction

1. Introduction

1.1. Wildfires and their effects on geomorphic and hydrological processes.

Over the last 400 million years, vegetation has evolved to adapt to fire as a natural component of ecosystem dynamics (Pausas, 2004). Fire-adapted vegetation can recover from the damage provoked by wildfire through regeneration mechanisms such as resprouting from resistant plant parts as well as through germination strategies (Whelan, 1995). During the last millennia, however, fire regime has been strongly modified by human societies (James, 1989, Goren-Inbar et al., 2004). Since the Neolithic age, fire has been used as a source of energy and as a tool in defence, for hunting and for managing the landscape, introducing agricultural and pasture lands in previously forested areas (Bird et al., 2008). As a result, forests had lower fuel loads and became intersected by open spaces, whilst fires became more frequent and, at the same time, of lower intensity (Pausas, 2004). In the Mediterranean Basin, human settlements have shaped the surrounding landscapes during the last 10.000 years.

Also in Portugal, the landscape reflects a long history of intense land management, with a mosaic of (semi-)natural and man-made agricultural and afforested lands, unploughed and ploughed hillslopes connected by a dense net of tracks and roads (Shakesby 2011). Since the 1980´s, however, wildfires have increased dramatically in frequency and extent, aided by a general warming and drying trend but driven primarily by socio-economic changes (Ferreira et al., 2009). These changes were first and foremost the large-scale introduction of commercial plantations of fire-prone tree species such as eucalypt and pine (Figure 1), and the decline in traditional practices like grazing and coppicing, increasing the accumulation of flammable materials (Pereira et al., 2006; Radich and Alves, 2000; Shakesby et al., 1996). Between 1980 and 2010, wildfires have, on average, affected 110.000 ha of rural lands per year in Portugal (AFN, 2012).

Figure 1. Two views of the recently burnt areas studied in this thesis, the Pessegueiro area burnt in 2007 (top; Chapter 3) and the Colmeal area burnt in 2008 (bottom; Chapter 5).

The most evident change produced by a wildfire is the total or partial loss of the vegetation and litter cover. The removal of vegetation and litter reduces rainfall interception and, thereby, enhances throughfall (Díaz-Fierros et al., 1987; Soto et al., 1998) and can increase overland flow (Soto et al., 1993) as well as subsurface and groundwater flow (Lavabre et al., 1993). Vegetation loss also decreases the leaf area from which evapotranspiration occurs, and reduces the obstacles to overland flow (Shakesby and Doerr, 2006). Besides vegetation and litter cover, wildfires can have significant impacts on soil chemical and physical properties, depending to a large extent on fire severity (e.g. Inbar et al., 1998; Robichaud, 2000; Shakesby and Doerr, 2006). Fire has been found to cause (partial) combustion of organic matter, deterioration of soil structure and aggregate stability, increase in bulk density and soil water repellency (DeBano, 2000; Doerr et al., 2000; Fernández et al., 2004; Giovannini et al., 1988; Imeson et al., 1992; Llovet et al., 2009; Soler et al., 1994). These changes often make the soil more susceptible to overland flow generation and removal by rain drop and runoff (Shakesby, 2011). Fire-induced changes in soil water repellency deserve special reference in the case of north-central Portugal. On the one hand, the prevalent forest plantations of Maritime Pine and especially eucalypt commonly exhibit pronounced repellency (Doerr et al., 1996; Ferreira et al., 1997; Keizer et al., 2008; Leighton-Boyce et al., 2007; Malvar et al., 2011; Figure 2) and on the other hand, soil water repellency is widely considered as one of the main factors in enhancing runoff generation and the associated soil losses following wildfire (e.g. Leighton-Boyce et al., 2007; Shakesby and Doerr, 2006; Sheridan et al., 2007). The eucalypt plantations are typically planted as monocultures for paper pulp production. The harvesting cycles are about every 7-14 years, after which the stumps are left to regrow up to four times and a new plantation cycle begins (Ferreira et al., 1997; Keizer et al., 2008; Leighton-Boyce et al., 2007; Malvar et al., 2011).

Figure 2. Illustration of extreme soil water repellency following wildfire, showing three drops with an ethanol concentration of 36 % “sitting” on dry soil.

Soil erosion is a two-stage process, involving detachment of soil particles and their subsequent transport by water, wind or gravity (Morgan 2005). In the case of post-fire soil erosion, fire severity was often found to be a key factor (Benavides-Solorio and

MacDonald, 2005; González-Pelayo, 2006; Rubio et al., 1997; Prosser and Williams, 1998; Robichaud, 2000). However, fire severity must be assessed after "the smoke has gone", using indicators that give qualitative rather than quantitative estimates of the heating regime. Ryan and Noste (1985) developed a fire severity index based on visual estimation of soil cover by litter and duff, and of alteration of soil colour (Figure 3). Vega et al. (2008) recommended using canopy cover consumption and ash colour (Figure 4). Other severity indices such as the Twig Method Severity Index (TMSI; Moreno and Oechel, 1989;) and the NIR-based Maximum Temperature Reached (Guerrero et al., 2007; Maia et al., 2012) can be more precise but also more labour-intensive.

Figure 3. Ash colour indicating higher burnt severity on the left than right on the photograph.

Figure 4. In the Pessegueiro wildfire of 2007, the tree canopies were fully consumed at the eucalypt study site (left) but only partially at the pine study site (right), so that the lower fire severity was associated to "natural" mulching by the subsequent cast of the scorched leaves and needles

Extreme sediment losses of 20 to even 170 Mg ha^{-1} $year^{-1}$ have been observed following wildfires in North America (Spigel and Robichaud, 2007; Riechers et al., 2008; Robichaud et al., 2000) as well as in Europe, (Galicia: Díaz-Fierros et al.,1987; Fernández et al., 2011; Catalonia: Úbeda and Sala, 1998; Marquès and Mora, 1992; France: Lavabre and Martin, 1997; Martin et al.,1997; Greece: Dimitrakopoulos and Seilopoulos, 2002; including Portugal Shakesby et al., 1994; as it was recently reviewed by Shakesby, 2011). Since the 1980´s (Swanson, 1981), it is generally accepted that fire-enhance erosion rates decrease with time-since-fire till they return to background levels at the end of the so-called window-of-disturbance (Shakesby and Doerr, 2006). This window-of-disturbance is estimated to last between 3 and 10 years, depending on fire severity and post-fire climate conditions (e.g. Andreu et al., 2001; DeBano et al., 1998; Robichaud, 2009; Sala et al., 1994). The relation between wildfire and soil erosion, however, is far to be straightforward (Figure 5). Various authors have reported negligible soil erosion rates after wildfires (Kutiel & Inbar, 1993; Kutiel et al., 1995; Lane et al., 2004). In general, in Mediterranean regions, post-fire erosion rates tend to be low (Shakesby, 2011). This is true for Portugal as well (e. g. Ferreira et al., 2008; Malvar et al., 2011, 2013, and Shakesby et al., 1996). In addition, the various factors controlling hydrological processes at different scales complicated even further the measurement and comparison of erosion rates (Robichaud, 2009). Perhaps more importantly, however, is the fact that there is much uncertainty in evaluating erosion rates as "tolerable" or not in terms of net soil loss, as the rate of soil formation continues to be poorly known. The existing estimates point to less than 1.5 Mg ha^{-1} year^{-1}, with large variations between regions (Alexander 1988; Wakatsuki and Rasydin, 1992).

Figure 5. Alluvial fan deposited during a rainfall event in 2010, eight years after the high-severity Hayman fire of 2002, Colorado, USA.

1.2. Mitigation of soil erosion following wildfires

The first efforts aiming to reduce soil erosion following wildfires were probably carried out in the USA (southern California), dating as far back as the late 1800s (Wohlgemuth et al., 2009). From then, the association of wildfire with on-site soil erosion and downstream flooding and massive sediment deposition became increasingly recognized and, in the early part of the last century, led to the first systematic soil erosion control treatments following wildfires (Munns, 1919). An event that alarmed the public opinion and highlighted the need for post-fire rehabilitation, took place on New Year´s Day 1934, in La Crescenta, near Los Angeles. A debris flow deposited river tributaries to depths of up to 5 m, transported boulders the size of automobiles over several kilometres, produced massive damage to properties and killed 16 persons (Kraebel, 1934).

During the first half of the 20th century, post-fire rehabilitations efforts by and large consisted of building engineering structures (check dams) in stream channels to trap the sediments and of seeding hillslopes to increase ground cover. However, these pioneering treatments involved several problems. First, it proved to be unrealistic to build check dams in the often short periods between the occurrence of the wildfires and of the erosion-producing rains, so that they had to be constructed in advance of wildfires in streams and downstream of fire-prone areas (Wohlgemuth et al., 2009). Second, as early as the 1920s, seeding with native shrub species was recognized to be ineffective, since the introduced seeds germinated no earlier than the in-situ seed bank. Subsequent seeding trials with faster-growing, non-native herbaceous species such as Mediterranean mustards (*Brassica* ssp.) led to problems in downstream agricultural areas, where the species were considered to be noxious weeds by the farmers (Gleason 1947). By the 1950s, however, seeding with ryegrass (*Lolium multiflorum* Lam.) had become widely regarded as the most cost-effective treatment for augmenting post-fire ground cover, at least in California (Barro and Conard, 1987). It also proved a good technique to transform shrubland into pasture (Schultz et al., 1955).

The effectiveness of seeding to mitigate post-fire soil erosion started to be questioned during the 1960s, by researchers of the San Dimas Experimental Forest in southern California (Rice et al., 1965). Namely, various studies had found that ryegrass seeding did not markedly reduce erosion even when effectively increased ground cover (Gautier, 1983; Taskey et al., 1989). This ultimately led to a strong controversy on the effectiveness of seeding during the 1988 Symposium on Fire and Watershed Management, where various papers were presented that indicated the effectiveness of alternative post-fire treatments, i.e. application of straw mulch (Gross et al., 1989; Miles et al., 1989) and retaining of slash and residues from post-fire logging on the soil surface (Barker 1989; Poff 1989). From the above-cited studies, however, only one (Taskey et al., 1989) involved direct measurement of soil erosion rates on control and treated plots. Therefore, the symposium concluded to the need for standardization of treatment assessment methods, including a clear definition of time scale and of the potential effects being evaluated, to provide land-use managers with better information to sustain their decisions (MacDonald, 1989). This author draw special attention to the fact that treatment effectiveness - i.e. increase in ground cover - is not necessarily the same as achieving the goal of the treatment – i.e. (reduction of post-fire sediment yields).

During the 1990s and the 2000s, research on post-fire erosion mitigation concerned seeding (e.g. Beyers, 2004; Fernández-Abascal et al., 2003; Groen and Woods, 2008; Peppin et al., 2010; Pinaya et al., 2000; Robichaud et al., 2006), log barrier construction (Robichaud et al., 2008; Wagenbrenner et al., 2006), straw mulching (Bautista et al., 1996; Badía and Martí, 2000; Wagenbrenner et al., 2006) and several combination of two or more techniques (seeding + log erosion barriers + straw mulch in Dean, 2001 and seeding + mulch in Badía and Martí, 2000). In a nutshell, these studies found seeding to be effective in some cases but not in others, log barriers construction to be ineffective unless rain events are few and small, and mulching to be highly effective (Figure 6). The effectiveness of mulching is also well-established for agriculture lands (Harris and Yao, 1923; Meyer et al., 1970; Lyles et al., 1974; Jordán et al., 2010), and cut slopes and unpaved roads (Grismer and Hogan, 2005; Jordán et al., 2008).

The effectiveness of straw mulches for mitigating post-fire erosion has been tested more exhaustively in field trials than that of mulches composed of woody plant materials (Fernández et al., 2011; Kim et al., 2008; Riechers et al., 2008). Often-cited advantages of straw mulches are their wide availability, low costs and low specific weights. Nonetheless, the availability of straw mulches may be limited in many parts of the world (Foltz and Wagenbrenner, 2010). Furthermore, the low specific weights can also be a disadvantage, especially in areas prone to strong winds during the period between straw application and the first heavy rainfall events (Robichaud et al., 2000). Negative ecological effects of straw mulches were pointed out by Kruse et al. (2004), such as the reduction of the density of conifer seedlings and the involuntary introduction of non-native seeds. Whilst woody mulches are (potentially) widely available in forest-dominated areas such as north-central Portugal, their effectiveness under field conditions remains unclear. Wood chips were found to have little effect in reducing post-fire soil losses in various studies (Fernández et al., 2011; Kim et al., 2008; Riechers et al., 2008). This could be due to the shape and size of the chips. Laboratory rainfall simulation experiments found that wood shreds and strands (Figure 7) were as effective as straw, whilst wood chips were not (Smets et al., 2008; Foltz and Dooley, 2003; Yanosek et al., 2006). The first field experiments carried out in Portugal with the application of different quantities of logging litter found positive results (Shakesby et al., 1996).

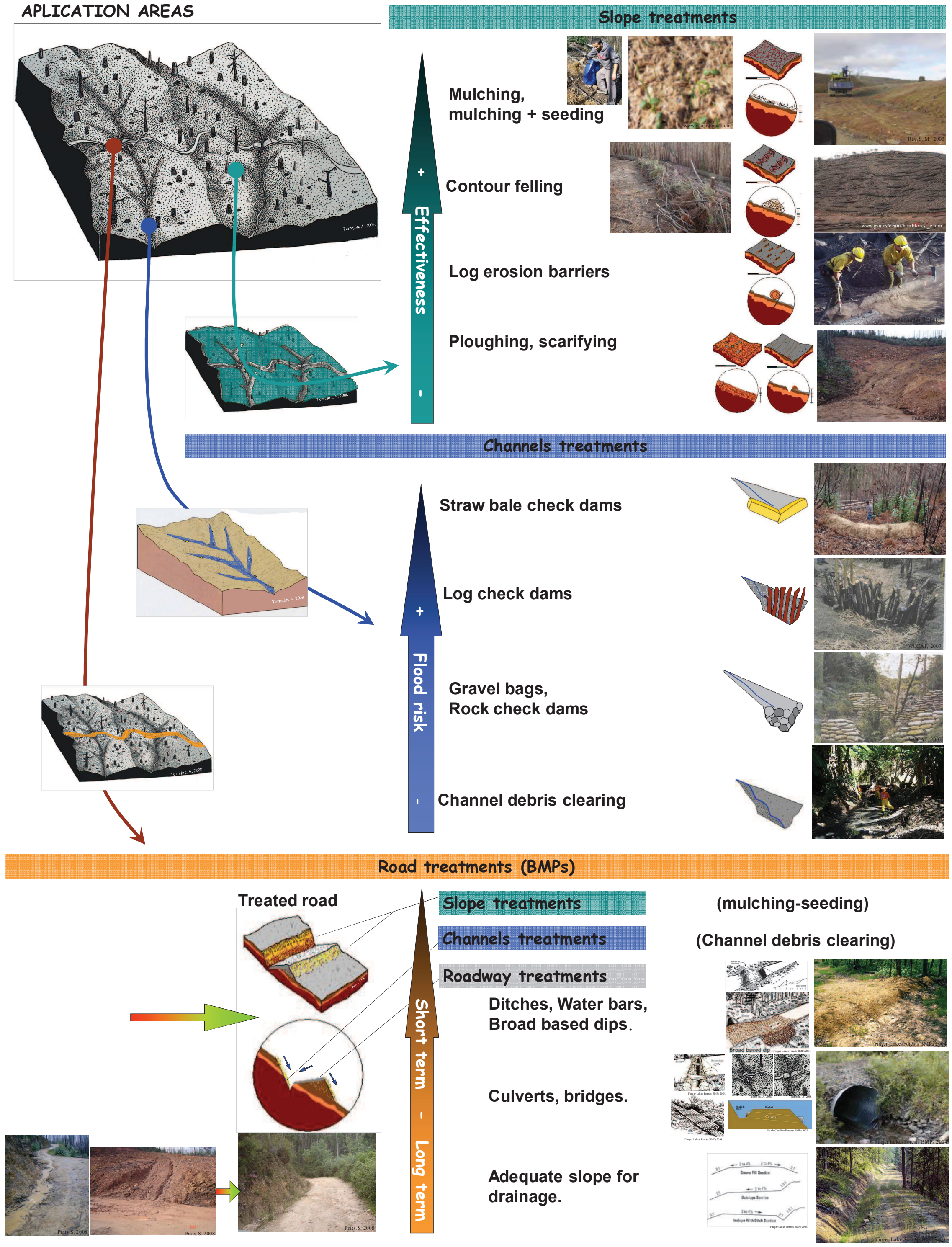

Figure 6. Literature-based overview of treatments for mitigation soil erosion following wildfire at the slope and catchment scale as well as for road networks.

Figure 7. Detail of wood strand mulch used in soil erosion mitigation following the 2007 School wildfire, Idaho, USA.

A recent variant of mulching is that of hydromulching (Figure 8), involving the application of an aqueous mixture of organic fibres with seeds, nutrients, soil binding agents, green colorant and seeds from a jet hose (Naveh, 1975). Hydromulching is a relatively expensive technique, with a cost of \$2300 to \$7400 per ha for ground and aerial spreading, respectively (MacDonald and Larsen, 2009) as compared to \$400-2900 for ground to \$600-2300 for aerial straw mulch spreading (de la Fuente and Blond, 2010; MacDonald and Larsen 2009; Napper, 2006). Hydromulching is typically used in restoration of degraded landscapes such as quarries, road banks, and highway cut slopes (Benik et al., 2003; Emanual, 1976; Robichaud et al., 2010), but has not been extensively used or tested in recently burnt areas (Hubbert et al., 2011; Wohlgemuth et al., 2011). Hubbert et al. (2011) found hydromulching to be effective in reducing post-fire erosion but, at the same time, noted a quick breakdown. By contrast, Wohlgemuth et al. (2011) found it to be ineffective for high-intensity rainfall storms.

Figure 8. Details of the hydromulch applied by the authors in the Colmeal field trial (left) and of the forest residue mulch applied in the Ermida experiment (right).

Also polyacrylamides (PAM) - a family of flocculant agents developed by the agro-chemical sector - were recently introduced for post-fire erosion mitigation (Rough 2007, Wohlgemuth and Robichaud, 2007). Especially in the form of dry granulate (Figure 8), it can be applied easily. During the last two decades, PAM has become widely accepted as for soil erosion control in intensive agriculture with furrow irrigation as well as on steep road embankments (Agassi and Ben-Hur, 1992; Ben-Hur, 2001, 2006; Ben-Hur and Keren, 1997; Ben-Hur and Letey, 1989; Lentz et al., 2002; Levy et al., 1991; Sojka et al., 2007). Since PAM is a generic term for a broad class of hundreds of polymers with differing functional groups and chain lengths, different formulations have been tested to achieve optimal binding of PAM with the soils' specific clay particles, through direct ionic attractions or cation bridges (Theng 1982, Vacher et al., 2003). So far, few field trials have assessed the effectiveness of PAM in recently burnt areas, and these studies have reported opposing results. Whilst Davidson et al. (2009), Riechers et al. (2008) and Inbar (2011) found PAM to be effective in reducing post-fire erosion, Rough (2007), Wohlgemuth and Robichaud (2007) and Macdonald and Larsen (2009) did not.

In Portugal, post-fire emergency treatments have rarely been employed so far, although this may now be changing with the emergency stabilization measures funded by PRODER (under sub-Action 2.3.2.1) for selected, 2010-burnt areas. Likewise, at the start of the studies presented here, field trials on the effectiveness of post-fire soil conservation measures had received surprisingly little research attention, being limited to a single case (Shakesby et al., 1996).

Figure 9. Illustration of the application of the treatments assessed in this study: eucalypt chopped bark mulch (Pessegueiro do Vouga and Ermida, Chapters 3 and 4; top left), eucalypt logging slash mulch (Pessegueiro do Vouga, Chapter 3; top right), dry polyacrylamide (Ermida, Chapter 4; bottom left) and jet-hose hydromulching (Colmeal, Chapter 5; bottom right).

The authors assessed the effectiveness of logging residues for a eucalypt as well as a Maritime Pine plantation. For various reasons, however, the study was somewhat unconventional in that: (i) the treatments were applied two years after the wildfire; (ii) involved different application rates rather than replicate plots; (iii) were evaluated by comparing the erosion rates before and after treatment rather than concurrent erosion rates at treated and untreated plots. According to MacDonald (1994), monitoring of the effectiveness of post-fire rehabilitation measures should ideally involve: (i) application of the treatment as soon as possible after the wildfire, to avoid missing especially the large, initial rainfall events; (ii) a long-term measurement period to assess any decrease in effectiveness during the window-of-disturbance; (iii) inclusion of replicated plots for the control situation (burnt but untreated) as well as for each one of the treatments.

1.3 Tools for measuring soil erosion

A treatment effectiveness monitoring program as suggested by MacDonald (1994; see 1.2) implies major efforts. This is especially true when using runoff plots, due to the large number of runoff samples that need to be analyzed in terms of sediment load. Standard laboratory methods to determine sediment concentration are all rather time- and energy-consuming (APHA, 1998). For example, a site with 10 runoff plots will produce in a typical read-out around 10 runoff samples of 1.5l. Evaporation (hours-days) or filtration (minutes-hours) are time-consuming methods that must invariably be followed by: a weighting, a drying period of 24 h at 105 ºC, a cooling and finally a weighting of the dry sediments (APHA, 1998). In total, 10 samples will demand between 2 to 3 days of laboratory processing besides the storage space. Thus, turbidity sensors have much potential to speed up the process and lower its costs. Turbidity sensors such as the "OBS-3+ Suspended Solids and Turbidity Monitor" (Downing, 2006; Campbell et al., 2005; Figure 10) are available for over two decades now but are poorly suitable for laboratory measurements, especially also due to the large sample volume that is required. They are, however, now widely used for automatic monitoring of stream flow sediment concentrations. Even so, their use is frequently not without problems, since the sensors require complex installations and have a life span that is limited by their electronic parts. Furthermore, the current turbidity sensors (as well as the associated data retrieval and storage systems) have elevated costs, so that hydrometric stations typically involve a single turbidity sensor and, thus, are susceptible to data loss in case of sensor malfunctioning and to bias due to the sensor's position in the water column.

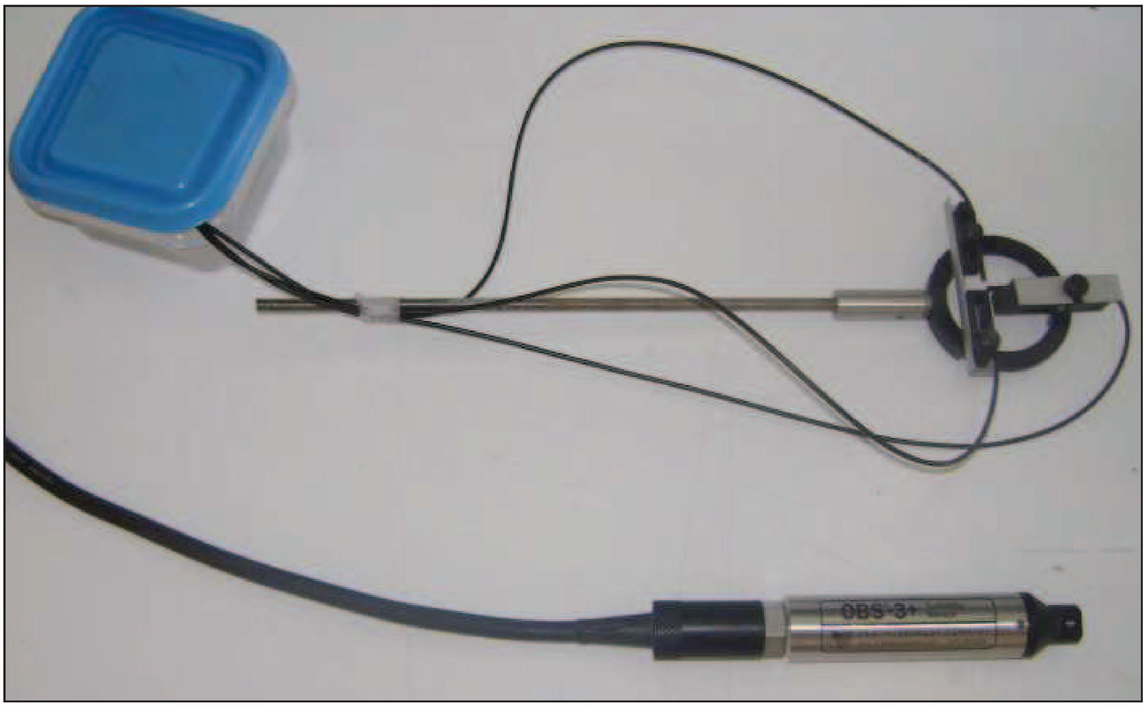

Figure 10. Two optic-based turbidity sensors for sediment concentration determination: the commercial OBS-3+ (below) and the new proto-type (above) co-developed by the author.

Fibre-optics-based sensors and, in particular those using plastic optical fibres (POFs) are now widely recognized to offer various important advantages over traditional methods of sensing (Zienmann, 2008). Besides being inexpensive and easy to handle, they are flexible, small and immune to electromagnetic interferences, and, in general, allow measuring any phenomenon without physically interacting with it. Further advantages that are of special interest for field monitoring of surface water quantity and quality, are the suitability of POFs to operate in a multisensory scheme, to permit monitoring in-situ or remotely, and to be employed under harsh conditions without significant sensor performance deterioration (Yeo, 2008). Fibre-optics-based sensors have, in fact, already been employed in earlier studies to measure turbidity of surface water. Ruhl et al. (2001) focused on low sediment concentrations, whilst Campbell et al. (2005) did not follow-up on his encouraging initial results. More recently, Postolache et al. (2007) also obtained very promising results but the data processing of their multi-beam optical system seems too complex for field monitoring applications. In their recent review on fibre-optical turbidity sensors, Omar and MatJafri (2009) concluded there was a need for more extensive testing, especially with respect to the dependence on particle diameter.

1.4. Aim and objectives of the thesis

The main aim of this thesis was to contribute to a better knowledge of the effectiveness of selected soil conservation methods to reduce runoff and soil erosion following wildfires, with a special emphasis of the principal forest types in north-central Portugal. The specific objectives were the following:

1) to quantify overland flow generation and the associated losses of sediments for recently burnt eucalypt and Maritime Pine plantations, with a high temporal resolution (weekly to monthly) to account for the typically rapid changes in fire-affected ecosystems and at spatial scales from micro-plots of less than 1 m^2 to “regular” plots of 10 m^2 and more;
2) to assess how post-fire runoff and soil erosion at eucalypt and Maritime Pine plantations are modified by the application of: (i) woody mulch composed of chopped eucalypt bark and eucalypt logging slash; (ii) dry polyacrylamide (PAM) and (iii) hydromulching;
3) to identify the key factors that can explain post-fire runoff and erosion in eucalypt and Maritime Pine plantations with and without the four soil conservation measures mentioned under point 1);
4) to determine the potential and limitations of a plastic-fibre-optics-based turbidity sensor for estimating soil erosion rates at runoff plots in recently burnt areas.

1.5. Thesis structure

This thesis can be divided into three parts, based on the specific objectives being addressed.

Part 1. Runoff generation and soil erosion in recently burnt Portuguese forest plantations

The data collected in this study helped to evaluate the factors that triggered runoff and soil erosion in six different burnt slopes in north-central Portugal through different approaches. In Chapter 2 the use of rainfall simulation experiments (rse´s) in eucalypt stands burned in the 2005 Açores wildfire allowed us to determine the key factors for runoff and soil erosion under fixed rainfall. A total of 46 rse´s were carried out during two years following the fire using two rainfall intensities (high- 45 mm h^{-1} and extreme 85 mm h^{-1}) for two eucalypt stands that were unploughed and ploughed before the fire. The independence of the high variability of the natural rainfall led us to depend purely on soil-dependent factors. Non parametric tests were run to infer the significance of key factors for runoff and soil erosion. In the case of Chapter 3, 4 and 5 the datasets were created at high temporal and spatial resolution (weekly intervals in 38 plots with sizes between 0.25 until 16 m^2 in four different burnt slopes), but also for the more important soil properties: soil cover, soil water repellence, soil moisture and soil resistance. This represented a high value to compare the effects of wildfires over the hydrologic processes in central Portugal, and allowed comparison with other regions of the world. Additionally, this body of data served to determine the specific weight of each key factor on the hydrologic, erosive and organic matter losses responses in each wildfire under both control and treatment conditions. This was done with the SAS statistical program (Littell et al., 1996), through the construction of multiple regression models after transformation of the variables for achieving normality of the residuals.

Additionally, the differences between these six burnt slopes had been evaluated. While in Chapter 2 the set up took into account two study areas in order to allow for inferences about land use, such as pre-fire ploughing activities, in Chapter 3 the two study areas provided more information on the effect of fire severity. In Chapters 4 and 5 the set up involved a unique hillslope, and factors such as hillslope position and plot size were also tested.

Part 2. Evaluation of selected post-fire soil erosion mitigation techniques in Portuguese forest plantations.

In Chapters 3, 4 and 5 four new post-fire soil erosion control treatments were assessed. After a thorough review of soil erosion treatments worldwide, mulching appeared as the most effective technique. However, the well documented straw mulch was found not to be highly available in Portugal, and this was the main reason in favour of trying out other materials, i.e., the forest residue mulches in Chapter 3 and 5 provided kindly by SOCASCA, S.A. The more recent use of PAM and hydromulch were possible throughout the support of Quimitecnica S.A. and Serraic Lda., respectively, who kindly provided the treatments for the experiments of Chapter 4 and 5. To the best of our knowledge, these treatments have not been tested before for post-fire soil erosion reduction (with exception of the USA), and the results will provide useful data for cost-benefit analyses for post-fire emergency treatments in Mediterranean regions.

All the experiments were monitored at weekly intervals during, at least, the first year after the fire. The forest residue and PAM experiences (Chapters 3 and 4) were carried out over 1.5 years after the fire. In contrast, the hydromulch experiment (Chapter 5) accounted for the second, the third and the beginning of the fourth post-fire year, despite it was not initiated immediately after the wildfire. These differences were due to the difficulties in applying the experimental set-up at the same time that logging activities were being carried out after the 2008 Colmeal fire.

The analysis of variance of repeated measures was used to assess statistically the differences between each treatment and their controls for runoff, soil erosion, organic matter content and also soil cover. Additionally, in the case of Chapters 3 and 5, the effect of the treatment was also tested over some soil properties such as soil moisture, soil water repellence and soil shear strength.

Part 3. Soil erosion measurement tools.

Although direct measurement of hillslope runoff and erosion is expensive, complex, and labour-intensive, it is the only way to guide adequate future responses to post-fire stabilization and rehabilitation and also to develop and refine predictive models (Robichaud, 2009). In Chapter 6, some of the runoff samples of the 2008 Colmeal wildfire (Chapter 5) were used to check the utility of a new optic-based turbidity sensor as a new and economic measurement tool. This proto-type was developed after the "*Laboratorio I*" and "*Laboratorio II*", both disciplines of the PROMAR PhD-program. The logistic support was provided by direct collaboration between the *Instituto de Telecomunicações* (IT), the *Instituto de Nanotecnologia* (I3N) and the *Departamento de Ambiente e Ordenamento* (DAO) of the University of Aveiro. The proto-type sensor was tested in the laboratory against a commercial sensor provided by the EROSFIRE-II project. The potential of this new tool consists of the substitution of the evaporation-based, time- and energy-consuming classic method. It can also be used for manual measurement of the runoff from field erosion plots as well as for continuous readings of stream flow in a hydraulic channel.

CHAPTER 2

Post-fire overland flow generation and inter-rill erosion under simulated rainfall in two eucalypt stands in north-central Portugal

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Post-fire overland flow generation and inter-rill erosion under simulated rainfall in two eucalypt stands in north-central Portugal ☆

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ABSTRACT

The aim of this study was to improve the existing knowledge of the runoff and inter-rill erosion response of forest stands following wildfire, focusing on commercial eucalypt plantations and employing field rainfall simulation experiments (RSE's). Repeated RSE's were carried out in two adjacent but contrasting eucalypt stands on steep hill slopes in north-central Portugal that suffered a moderate severity fire in July 2005. This was done at six occasions ranging from 3 to 24 months after the fire and using a paired-plot experimental design that comprised two pairs of RSE's at each site and occasion. Of the 46 RSE's: (i) 24 and 22 RSE's involved application rates of 45–50 and 80–85 mm h^{-1}, respectively; (ii) 22 took place in a stand that had been ploughed in down slope direction several years before the wildfire and 24 in an unploughed stand.

The results showed a clear tendency for extreme-intensity RSE's to produce higher runoff amounts and greater soil and organic matter losses than the simultaneous high-intensity RSE's on the neighbouring plots. However, there existed marked exceptions, both in space (for one of the plot pairs) and time (under intermediate soil water repellency conditions). Also, overland flow generation and erosion varied significantly between the various field campaigns. This temporal pattern markedly differed from a straightforward decline with time-after-fire and rather suggested a seasonal component, reflecting broad variations in topsoil water repellency. The ploughed site produced less runoff and erosion than the unploughed site, contrary to what would be expected if the down slope ploughing had occurred after the wildfire instead of several years before it. Finally, sediment losses at both study sites were noticeably lower than those reported by other studies involving repeat RSE's, i.e. in Australia and western Spain. This possibly reflected a history of intensive land use in the study region, including in more recent times after the widespread introduction of eucalypt plantations.

1. Introduction

As thoroughly discussed by Shakesby and Doerr (2006), through their effects on soil properties as well as on vegetation and litter cover, wildfires can lead to considerable changes in geomorphologic and hydrological processes. Previous studies in various parts of the world, including Portugal (e.g. Shakesby et al., 1993, 1996; Walsh et al., 1992, 1995; Ferreira et al., 2005b, 2008), have revealed strong and sometimes extreme responses in runoff generation and associated soil losses following wildfire, especially during the earlier stages of the so-called "window-of-disturbance". Besides wildfire itself, post-fire forestry practices can strongly influence overland flow and erosion in recently burnt areas (e.g. Shakesby et al., 1994; Walsh et al., 1995; Fernández et al., 2007). For example, rip-ploughing during the window-of-disturbance was far more damaging in terms of soil loss than fire (Shakesby et al., 1994).

The need for a model-based tool for assessing erosion risk following wildfire and, ultimately, for guiding post-fire land management, like the Erosion Risk Management Tool (ERMiT) for the Western USA (Robichaud et al., 2007), is overtly evident in the case of Portugal. Over the past decades, wildfires in Portugal have devastated on average around 100.000 ha each year, with dramatically higher figures for dry years like 2003 and 2005 (Pereira et al., 2005). Furthermore, the frequency of wildfires in Portugal is expected to remain the same or to increase in the future (Pereira et al., 2006). In relation to fire occurrence, the widespread introduction of commercial eucalypt plantations (principally of *Eucalyptus globulus* Ait.) in central Portugal (including in the study area) in combination with their proneness to fire deserves special reference. Furthermore, post-fire erosion risk is expectedly higher in eucalypt stands than, for example,

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Maritime Pine forest, another common and fire-prone forest type in central Portugal, namely, eucalypt stands are typically associated with pronounced soil water repellency (Doerr et al., 1996, 1998; Keizer et al., 2005b, 2008; Leighton-Boyce et al., 2005), on the one hand, and on the other, water repellency is widely considered one of the main factors in enhancing runoff generation and the associated soil losses following wildfire (e.g. Shakesby and Doerr, 2006; Leighton-Boyce et al., 2007; Sheridan et al., 2007).

Following the dramatic wildfire season of summer 2003, the EROSFIRE project (Keizer et al., 2006, 2007) set out to develop such an erosion prediction tool tailored to the specificities of post-fire conditions in Portugal's forests. Field rainfall simulation experiments (RSE's) were selected as principal method for gathering the data required for initial calibration of the process-based model MEFIDIS (Nunes et al., 2005) for post-fire conditions, much along the lines of the approach applied in Nunes et al. (2009a, 2009b). In spite of the well-know limitations of RSE's in terms of reproducing natural rainfall events and emulating runoff/erosion processes beyond small spatial scales (e.g. Rickson, 2001), they have been widely used for studying hydrological and erosion processes in recently burnt woodland areas, especially at spatial scales of 1 m^2 and less (e.g. Imeson et al., 1992; Kutiel et al., 1995; Sevink et al., 1989; Benavides-Solorio and MacDonald, 2001; Johansen et al., 2001; Cerdà and Doerr, 2005; Coelho et al., 2005; Ferreira et al., 2005a; Rulli et al., 2006; Leighton-Boyce et al., 2007; Sheridan et al., 2007). However, the bulk of these studies concerned singular moments in time-after-fire, not addressing for example the seasonal component in post-fire runoff and erosion that is often observed in longer-term plot monitoring studies under natural rainfall conditions (e.g. Shakesby et al., 1993, 1994). Also, the individual studies generally involved a single rainfall intensity. As far Portugal is concerned, surprisingly few field RSE studies have been carried out in recently wildfire-burnt stands of eucalypt (Leighton-Boyce et al., 2007) or, for that matter, in other prevailing forest types (Walsh et al., 1998; Coelho et al., 2004; Ferreira et al., 2005a).

The main aim of the present work was to explore repeated field campaigns of RSE's for a better knowledge and understanding of overland flow generation and associated sediment losses in recently burnt commercial eucalypt plantations. To this end, RSE's were carried out in two eucalypt stands on four occasions during the first year following wildfire and on two additional occasions during the second year. Two adjacent sites were selected for expectedly representing contrasting risks of post-fire erosion, with the site that had been rip-ploughed presenting a greater risk than the neighbouring unploughed site.

The specific objectives were to determine how overland flow generation and sediment losses varied at the micro-plot scale with (i) high *vs.* extreme simulated rainfall intensity (45–50 and 80–85 mm h^{-1}); (ii) time since fire and associated changes in initial conditions, in particular soil water repellency; (iii) within- and between-site characteristics at a ploughed *vs.* unploughed slope.

2. Materials and methods

2.1. Study area and sites

The present study was carried out in two adjacent commercial eucalypt (*Eucalyptus globulus* Ait.) plantations in the Açores locality of the Albergaria-a-Velha municipality of north-central Portugal (Fig. 1). The two study sites were located at approximately 40°42′N, 8°29′W and 60–70 m elevation, and comprised steep but short slopes bounded by paths (Table 1).

The study sites burned during early July 2005 in a wildfire that affected a total area of about 16 km^2, which was largely covered by eucalypt plantations. The complete consumption of the litter and herb cover, together with the partial consumption of the shrub layer and tree crowns, suggested that fire severity at both sites had been moderate (Shakesby and Doerr, 2006; Table 1). Judging by remaining tree stumps, the two sites had undergone at least two eucalypt (re)growth cycles prior to the fire. The two sites were selected for their contrasting land management practices and, as mentioned above, expectedly distinct risks of post-fire soil erosion. At the unploughed Açores1 site, trees had been planted without apparent evidence of mechanical ground operations, resulting in an undisturbed soil profile. At the ploughed Açores2 site, a clear pattern of shallow ridges and furrows (up to 20 cm high) running down the slope was present. Rip-ploughing (i.e. mechanical ploughing using a ripper with one to three teeth that rupture the upper soil horizons in a vertical plane without altering their disposition) in preparation for planting is a common practice in this region and, judging by the stand age, would have taken place around 5 years prior to the fire.

The study area is situated at the transition of the region's two major physiographic units, the Littoral Platform dominated by Ceno-Mesozoic deposits and the Hesperic Massif dominated by pre-Ordovician schists and greywackes and Hercynian granites (Ferreira, 1978; Pereira and FitzPatrick, 1995). The soils are mapped – at a scale of 1:1,000,000 – as a complex of Humic Cambisols and, to a lesser extent, Dystric Litosols (Cardoso et al., 1971, 1973). At both the study sites, two soil profiles were excavated in the middle and at the bottom of the study slopes. The soils corresponded to Umbric or Dystric Leptosols (FAO, 1988), depending on the depth of their A horizons. They were shallow (5–40 cm depth) soils developed over schists and had sandy loam textures and high organic matter contents (8.8–10.4%). These soil characteristics differed little between the two sites, which also agreed with the fact that rip-ploughing supposedly does not alter the disposition of the soil layers. Even so, the observed soil differences were duly considered in the discussion of the RSE results.

The climate of the study area can be characterised as humid meso-thermal, with a prolonged dry and warm summer (Köppen Csb) DRA-Centro (1998). Fig. 1 shows the locations of the study sites as well as of the nearest climate station (Estarreja: 40°47′N, 8°35′W, 26 m; 17.5 km distance) and the nearest rainfall station (Albergaria-a-Velha: 40°42′N, 8°29′W, 131 m; 4 km distance). The long-term mean annual temperature at the Estarreja station is 13.9 °C and the mean monthly temperatures range from 8.8 °C in December to 19.1 °C in July (DRA-Centro, 1998). The annual rainfall at the Albergaria-a-Velha station is, on average, 1229 mm and varies between 750 and 2022 mm (DRA-Centro, 1998). Fig. 1 also depicts the stations' seasonal variations in average monthly temperature and rainfall, and the monthly rainfall amounts at the study sites during the first year following wildfire. These latter data were obtained with a tipping-bucket rainfall gauge (Pronamic Professional Rain Gauge) linked to a Hobo Event Logger of Onset Computer Corporation, and were verified using the data from two totaliser rainfall gauges. All three gauges were installed at the foot of the study sites on September 24, 2005. These data were used in this paper to calculate the antecedent daily rainfall for the different field sampling days.

2.2. Rainfall simulation experiments

Between September 2005 and July 2007, a total of 46 rainfall simulation experiments (RSE's) were carried out in the field using two portable simulators as originally designed by Calvo et al. (1988) and later improved by Cerdà et al. (1997). One simulator was equipped with the original nozzle and was calibrated in the laboratory to produce artificial rain with an intensity of approximately 45 mm h^{-1}. The second simulator was equipped with a modified nozzle, using cone nozzle HARDI-1553-14 instead of HARDI-1553-10, to produce intensities of around 80 mm h^{-1}. The former intensity is comparable to the maximum hourly rainfall for a 100-year return period of the Aveiro rainfall station (Brandão et al., 2001). The latter is similar to the maximum hourly rainfall ever recorded in Portugal (Brandão et al., 2001) but a prior RSE study in Portuguese eucalypt forests like Leighton-Boyce et al. (2007) applied still higher intensities (100 mm h^{-1}) and found infiltration capacities exceeding this value. Hereafter, the two intensities will be referred to as "high" and "extreme", respectively. Other modifications to the original simulator design involved the use of a battery-driven pump system with pressure vat and of an approximately square plot (consisting of a square area of 0.50 m × 0.50 m and an outlet area of 0.03 m^2), both of which were introduced by De Alba (1997).

The 46 RSE's were carried out during four separate field campaigns in the first year after the wildfire, and two more campaigns in the second year (Table 2). Before every campaign (with the exception of the second) the two standard and two spare nozzles were (re-)calibrated in the laboratory. Each campaign involved four RSE's on both the ploughed and unploughed site, except in the case of the October 2006 campaign when only the high-intensity RSE's were carried out at the unploughed site due to failure of the extreme-intensity pump system. The four RSE's at a particular site were in general performed on the same day and within less than a week of those carried out at the other study site. Exceptions were the first campaign on the ploughed site and the October 2006 campaign, which took place on September 20 and 22, 2005, and October 12 and 31, 2006, respectively.

The four RSE's at each site and date were carried out using a pair-wise sampling design. High- and extreme-intensity RSE's were run almost

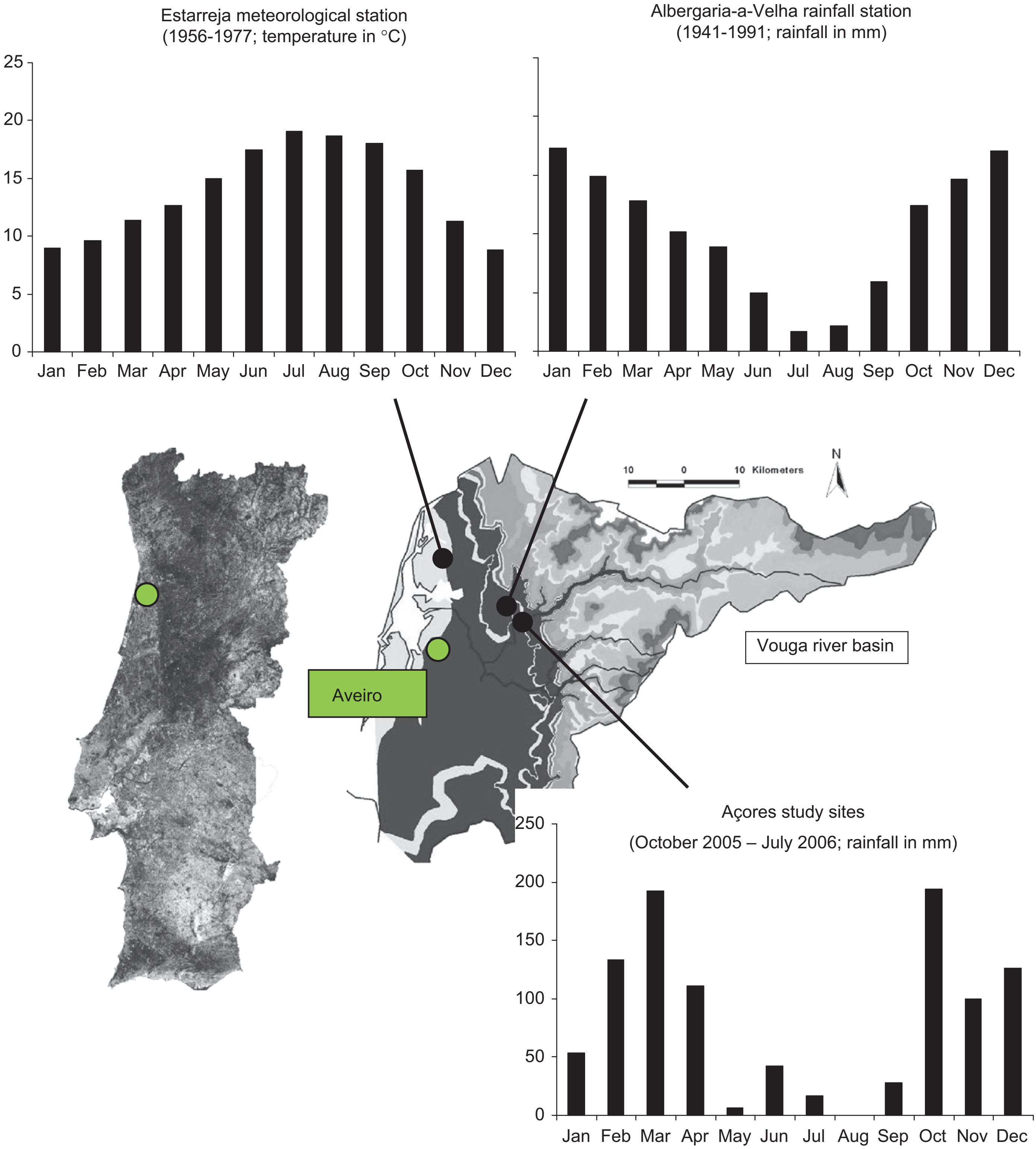

Fig. 1. Location of the study sites and the nearest weather stations, and their respective (average) monthly values.

simultaneously on two neighbouring plots located at about the same elevation on the slope but separated across the slope by 3–5 m. The two pairs of neighbouring plots on each site were placed randomly by installing them halfway the slope's upper and lower half. This was done in a horizontal section of the slope that was specifically reserved for the RSE's. The slope was further divided in a section that was equipped with erosion plots and a section that was used to describe topsoil characteristics at regular intervals (see Keizer et al., 2008). According to their spatial lay-out, the RSE-plots at each site were designated as follows: 1 and 2 were located on upper slope sections, 3 and 4 on the lower slope sections; 1 and 3 concerned high-intensity RSE's, 2 and 4 extreme-intensity RSE's. The prefixes "U" and "P" were used to indicate the plots on the unploughed and ploughed site, respectively.

The RSE's of the first campaign were immediately followed by destructive sampling of the plots as soon as the runoff had stopped. The RSE's of the second and subsequent campaigns, however, were carried out on permanent plots, with the repeat experiments on each plot involving the same intensity as established randomly in October 2006. Each pair of RSE's involved a third "control" plot for destructive measurements and sampling of the initial conditions, in particular

Table 1
General terrain characteristics and fire severity indicators at an unploughed and a ploughed eucalypt site.

Variable	Unploughed	Ploughed
Physiognomy		
Slope section length (m)	20–25	30–40
Slope angle (deg.)	20	15
Aspect	SE	NE
Fire severity indicators		
Eucalypt crown damage	Partial	Partial
Height of eucalypt stem scorching (m)	≤ 9	≤ 12
Combustion of litter/herbs layer	Total	Total
Combustion shrub layer	Partial	Partial
Ash colour	Black	Black

Table 2
Overview of high- and extreme-intensity RSE's ($n \times$ intensity in mm h^{-1}) and their average runoff and erosion results at an unploughed and a ploughed eucalypt stand during the first 2 years following a wildfire.

Campaign	Period	Unploughed		Ploughed	
		High	Extreme	High	Extreme
1	20–22–27/09/2005	2×46	2×84	2×46	2×84
2	10–15/11/2005	2×46	2×84	2×46	2×84
3	30/03–04/04/2006	2×48	2×80	2×48	2×80
4	20–25/07/2006	2×46	2×80	2×46	2×80
5	12–31/10/2006	2×47	2×81	2×47	–
6	03–09/07/2007	2×44	2×76	2×44	2×76
Variables					
Slope angle permanent plots (deg.)		23	23	17	17
Total simulated rainfall (mm)		277	485	277	404
Total runoff (mm)		150	265	94	154
Overall runoff coefficient (%)		54	55	34	38
Total soil loss (g m^{-2})		26	93	14	21
Total organic matter loss (g m^{-2})		18	57	10	14
Specific soil loss (g m^{-2} mm^{-1} runoff)		0.17	0.35	0.15	0.15
Specific o.m. loss (g m^{-2} mm^{-1} runoff)		0.12	0.21	0.11	0.09

regarding soil water repellency and moisture content at various depths. Non-destructive characterisation of the RSE-plots was done prior to all experiments and involved a standard procedure of quantifying the frequency of various cover classes by recording their presence/absence in the 5 cm $\times$ 5 cm cells of a 50 cm $\times$ 60 cm grid laid out over the plots. Photographs were taken and used to check the frequency estimates and convert them into decimal cover classes from 0 to 10.

The destructive sampling of the initial RSE-plots and control plots concerned first and foremost the moisture content and water repellency of the topsoil at 2–3 and 7–8 cm depth. This involved the same methods, equipment and water repellency severity ratings as described in Keizer et al. (2008). In a nutshell, soil moisture content was measured using an ML2 ThetaProbe™ connected to a HH2 ThetaMeter (Delta T-Devices Ltd.) or, in case of probe failure, gravimetrically and then converted based on Saxton et al. (1986) and Costa (1999). Water repellency severity was measured using the 'Molarity of an Ethanol Droplet' (MED) test (e.g. King, 1981; Doerr, 1998), by applying three droplets of increasing ethanol concentration and employing their median ethanol concentration (%vol) as test result. The following nine ethanol classes and corresponding ethanol concentrations were used: 0–0%; 1–1%; 2–3%; 3–5%; 4–8.5%; 5–13%; 6–18%; 7–24%; 8–$\geq$36%. Random roughness was determined using a pin profile metre and the PMPPROJ software (developed by J. Kilpelainen, Agricultural Research Centre, Jokioinen, Finland) for processing the photographs.

All RSE's were carried out using a pre-established protocol and standard field forms that were derived, with some modifications, from those employed in the MEDAFOR project (Shakesby et al., 2002). The protocol's principal elements were the application of artificial rain from a height of 2 m during 1 h, runoff measurements at 1-min intervals and the collection of up to five runoff samples (i.e. one from the start of the runoff till its approximate stabilisation; one from the end of the rainfall till the end of the runoff; three at the start, middle and end of the remaining period). The collected runoff samples were later analysed in the laboratory for their sediment and organic matter loads using the classical evaporation protocol (APHA, 1998) and loss-on-ignition at 550 °C. Soil texture classes were determined by the Soil Laboratory of the Coimbra Higher School of Agriculture, using a combination of mechanical sieving and the pipette method.

2.3. Data analysis

Data analysis was carried out using STATISTICA for Windows Version 9.0, by StatSoft Inc. Rank-based descriptive statistics and non-parametric statistical tests were preferred, in particular because of the limited number of samples and the resulting difficulties in verifying key assumptions underlying the parametric equivalents. The Mann-Whitney *U*-test and the Kruskal–Wallis test were employed to test overall differences, whereas the Wilcoxon's signed-ranks test and the Friedman test were used to assess differences in paired observations, either neighbouring plots or repeat-RSE's on the permanent plots. Besides differences between individual RSE's, also differences in average values of concurrent high-/extreme-intensity RSE's were included in the analyses for being less susceptible to possible noise due to spatial variability. In the case of the temporal patterns, only differences between consecutive campaigns were tested. This was done to restrict the number of multiple, unplanned comparisons to a minimum. Also the significance of the between-site differences of the individual RSE's was assessed using the standard type I error $\alpha = 0.05$ and not using the comparison-wise type I error $\alpha' = 0.025$ following the Dunn–Šidák method for 2 unplanned comparisons (Sokal and Rohlf, 1981).

3. Results and discussion

3.1. Overall runoff and erosion rates

Table 2 summarises the overall runoff and erosion figures obtained over the six field campaigns between September 2005 and July 2007. Direct comparison of the presented values is hampered by the lack of extreme-intensity data at the unploughed site for the October 2006 campaign. Nonetheless, the main differences observable in Table 2 are similar to those for the five common campaigns (explained below).

The two simulated rainfall intensities had a negligible effect on the relative amounts of overland flow generation at the two study sites; runoff coefficients were rather determined by site-specific differences. For the five "common" periods, the overall runoff coefficients amounted to 57–58% and 38–39% for the unploughed and ploughed site, respectively. In terms of absolute runoff amounts, the extreme-intensity RSE's at each site therefore produced, on average, about 70% more overland flow than the high-intensity RSE's on the same site. The extreme-intensity values for the five "common" campaigns were 231 and 154 mm for the unploughed and ploughed site, respectively; the corresponding high-intensity values were 133 and 91 mm.

Total losses of soil and organic matter were determined by a combined effect of site-specific factors and rainfall intensity. The losses at the unploughed site exceeded those at the ploughed site. For the five "common" campaigns, the total soil losses were 25 and 89 g m^{-2} *vs.* 14 and 21 g m^{-2}, respectively, and the corresponding total organic matter losses were 18 and 56 g m^{-2} *vs.* 10 and 14 g m^{-2}. The between-site differences for the separate rainfall intensities were more pronounced. The losses at the unploughed site were almost twice as high in the case of the high-intensity RSE's and more than four times as high in the case of the extreme-intensity RSE's. The intensity-related differences in total losses were bigger at the unploughed than ploughed site. The extreme-intensity RSE's produced, on average, roughly three times more soil and organic matter loss than the high-intensity RSE's at the unploughed site but only 40–50% higher losses at the ploughed site.

The intensity-related differences in total soil and organic matter losses can in the case of the unploughed site be partly attributed to higher specific losses. The specific losses were about twice as high for the extreme- than high-intensity RSE's. By contrast, at the ploughed site the specific losses were basically the same for the two intensities. The contribution of the specific losses to the between-site differences was also not consistent. They were of minor influence in the case of the high-intensity

RSE's, but contributed with roughly a factor two in the case of the extreme-intensity RSE's.

The present results were perhaps most surprising in that the ploughed site produced, on average, less runoff and lower total sediment losses than the unploughed site. In a nearby area, down slope rip-ploughing was found to substantially enhance overland flow responses and sediment loss rates during the first three years after ploughing (Shakesby et al., 1994; Walsh et al., 1995). These results are not directly comparable to those presented here, namely, they concerned much bigger plots (16 m^2) and lower, natural rainfall intensities. Even so, the overall sediment loss rates of the high-intensity RSE's of this study (0.09–0.16 g m^{-2} mm^{-1} rainfall) were much more similar to those reported by Shakesby et al. (1994) for "natural recovery" post-burn sites (0.05–0.10 g m^{-2} mm^{-1} rainfall) than for a recently rip-ploughed site do (3.27 g m^{-2} mm^{-1} rainfall; see also Terry (1996)).

The lower-than-expected sediment losses at the ploughed site could be related to the fact that ploughing took place several years before the wildfire. Shakesby et al. (1994) estimated that sediment losses decline rapidly following rip-ploughing. They attributed this to the formation of a protective stone lag, particularly in the early stages, and to the subsequent development of vegetation and litter cover. There was, however, no evidence that surface stone cover in the RSE-plots was noticeably higher at the ploughed than unploughed site. Walsh et al. (1995) further suggested that rip-ploughing ultimately decreased soil erodibility through selective removal of the fine soil fraction by initial erosion events. This fits in with the lower specific soil losses at the ploughed than unploughed site, especially in the case of the extreme-intensity RSE's. The topsoil (0–5 cm) at the ploughed site has, in fact, somewhat smaller clay and loam fractions than that at the unploughed site (median values of 3 samples: 7 and 20 *vs.* 13% and 24%, respectively). Nonetheless, the lower specific soil losses at the ploughed site could also be due to its smaller runoff amounts as well as to the expectedly lower flow velocities due to its less steep slope angle. Between-site differences are further analysed below.

The overall runoff and erosion values are not easily compared with those from literature, namely, the bulk of the field RSE studies following recent forest wildfires concerned singular moments in time (e.g. Sevink et al., 1989; Kutiel et al., 1995; Benavides-Solorio and MacDonald, 2001; Johansen et al., 2001; Rulli et al., 2006). Focusing on Portugal, only Leighton-Boyce et al. (2007) seem to have carried out RSE's in a recently burnt eucalypt plantation as well. In terms of rainfall intensity (100 mm h^{-1}) and pre-fire ploughing, their RSE's compare best with the extreme-intensity RSE's at the ploughed site. Compared with these RSE's, both the mean runoff coefficient and mean specific sediment loss of Leighton-Boyce et al. (2007) were roughly twice as high (70% and 0.90 g m^{-2} mm^{-1} runoff).

RSE data are also scarce for recently burnt stands of another common and fire-prone forest type in Portugal, that of Maritime Pine. Using basically the same experimental set-up as here, Coelho et al. (2004) and Ferreira et al. (2005a) found runoff coefficients of 55–65%, which is comparable to the overall figure for the high-intensity RSE's at the unploughed site. The specific sediment losses in Coelho et al. (2004), however, were 3–4 times higher (0.90–1.20 g m^{-2} mm^{-1} runoff) than the corresponding values of the present study. Walsh et al. (1998) reported lower runoff coefficients (19–25%) but this was 2 years after a wildfire and involved lower application rates (33–35 mm h^{-1}) as well as larger plots (1 m^2).

Outside Portugal, wildfire-affected eucalypt stands were studied in a particularly exhaustive manner in Australia by Sheridan et al. (2007). This included eight subsequent campaigns of field RSE's but, unlike in this study, using different plots during each campaign. With rainfall intensities of 100 mm h^{-1} applied (during 30 min) on unploughed soils, these RSE's are best compared with the extreme-intensity RSE's at the unploughed site. Over the first 2 years after fire, Sheridan et al. (2007) found a somewhat lower runoff coefficient (41%) than reported here but an almost six times higher specific sediment loss (3.26 g m^{-2} mm^{-1} runoff), possibly reflecting their larger plot size (3 m^2).

Organic matter constituted an important fraction of the sediment losses observed in the present study. It amounted, on average, to some 40% and varied little between the two sites and the two intensities (38–42%). This is held to reflect the export of litter and especially ash particles, since the organic matter content of the 0–5 cm topsoil at both sites was only some 10% (unploughed site = 10.3%; ploughed site = 9.0%; median value of three samples). Unfortunately, comparison with the other studies cited in this section is not possible, since they do not present separate data on organic matter losses.

3.2. Variation with rainfall intensity

Overall differences between the high- and extreme-intensity RSE's tended not to be statistically significant (Table 3). The absolute runoff amounts constituted an exception, with significant differences in three of the four tests. The extreme-intensity RSE's only did not produce significantly more runoff than the high-intensity RSE's in the case of the ploughed site. This can be attributed to a greater spatio-temporal variability in the site's hydrological response, namely, the difference in median runoff amounts is of the same of order of magnitude for the ploughed site as for the unploughed site (approximately 20 vs. 25 mm).

The effect of rainfall intensity was more apparent from the Wilcoxon's Test results, i.e. when eliminating the variability due to differences between the campaigns as well as between the plot pairs. Besides runoff amounts, total soil and organic matter losses revealed statistically significant differences. The significance of these differences and their sign can be perceived from Fig. 2. Thus, extreme-intensity RSE's tended to produce significantly stronger

Table 3
Statistical comparison of runoff and erosion by high- *vs.* extreme-intensity RSE's at an unploughed and a ploughed eucalypt stand. The comparison concerns the two study sites together ("U&P") as well as separately, and the site-wise average values ("mean") as well as the values of the individual RSE-pairs ("Indiv."). The statically significantly outcomes (α = 0.05) of the MW *U*-test and Wilcoxon *S–R* test are indicated with "M" and "W".

Tests and variables	U&P Mean	U&P Indiv.	Unploughed Indiv.	Ploughed Indiv.
Total runoff (mm)	M/W	M/W	M/W	–
Overall runoff coefficient (%)	–	–	–	–
Total soil loss (g m^{-2})	W	M/W	W	–
Total organic matter loss (g m^{-2})	W	W	W	–
Specific soil loss (g m^{-2} mm^{-1} runoff)	W	W	–	–
Specific o.m. loss (g m^{-2} mm^{-1} runoff)	–	–	–	–

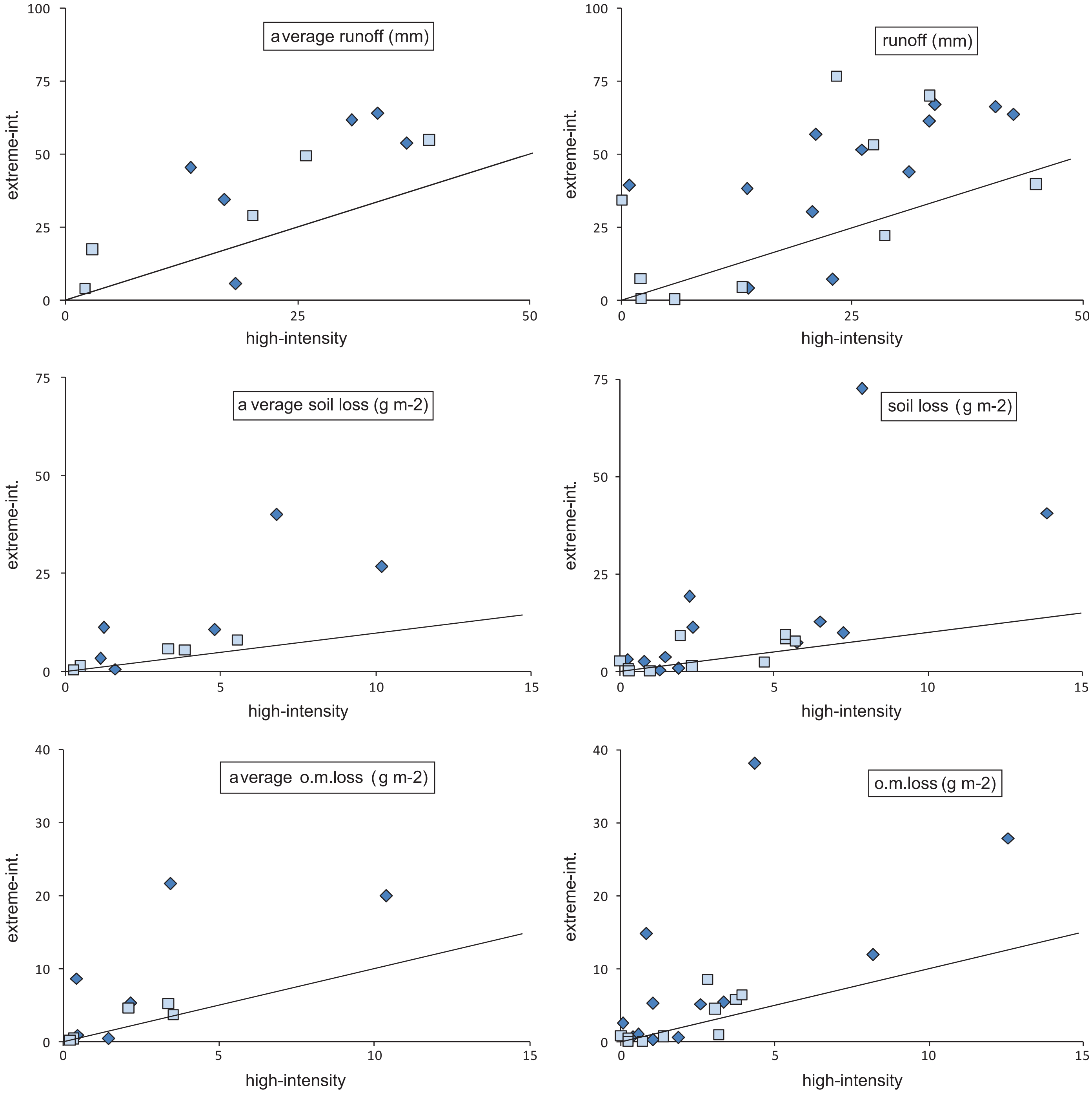

Fig. 2. Runoff and soil and organic matter losses of neighbouring pairs of high- and extreme-intensity RSE's at an unploughed (diamonds) and a ploughed (squares) eucalypt site.

runoff and erosion responses than the simultaneous high-intensity RSE's on the neighbouring plots. This especially applied to (i) absolute as opposed to relative measures; (ii) the two sites together and the unploughed site alone as opposed to the ploughed site alone. The runoff values at the ploughed site revealed a suspicious pattern in Fig. 2, with an equal number of points situated above and below the 1:1 line. Therefore, the Wilcoxon's tests were also applied to the site's separate plot pairs, even though the numbers of paired observations are small ($n=5$). For both plot pairs total runoff was significantly different. However, whilst total runoff was significantly higher for the extreme- than high-intensity RSE's in one case (plots P1 and P2 on the upper section of the slope), it was significantly lower in the other case (plots P3 and P4). In the case of the former plot pair, the extreme-intensity RSE's also produced significantly higher total soil and organic matter losses.

The deviant behaviour of especially one of the plot pairs on the ploughed site could be related to the pre-fire ploughing, leading to more heterogeneous micro-topographic and topsoil conditions in comparison to the undisturbed soil profiles of the neighbouring site. This could involve a combination of factors rather than a single factor *per se*. For example, the slope angle of the high-intensity plot on the lower slope section (P3: 20°) was slightly steeper than that of the adjacent extreme-intensity plot (P4: 18°)

and, at the same time, its random roughness was somewhat smaller (1.1 *vs.* 1.7). Also, spatial variability in topsoil water repellency (0–5 cm) during the first year following the wildfire tended to be more pronounced in the case of the ploughed than unploughed site (Keizer et al., 2008). Litter cover could play a role as well, since the P4 plot had a much higher litter cover than the P3 plot from the second campaign onwards (Fig. 6). This was due to the fall of leaves from scorched eucalypt crowns, which then slowly decomposed *in situ*. Shakesby et al. (1994) also mentioned this phenomenon in burnt eucalypt stands but expected its role in limiting erosion to be short lived. The role of such a litter cover could be direct – through interception storage and protection against rain drop impact – or indirect – by increasing the resistance to overland flow and/or by changing soil moisture as well as water repellency (e.g. Imeson et al., 1992; Lavee et al., 1995; Walsh et al., 1998; Doerr et al., 2000; Pannkuk and Roubichaud, 2003; Leighton-Boyce et al., 2007).

3.3. Temporal patterns

The timing of the RSE's had a significant influence on runoff response in general (Table 4). Overland flow generation varied significantly between the five and six campaigns: (i) in absolute as well as relative amounts; (ii) equally so for the average and individual values of the two sites together as for the values of the ploughed and unploughed site separately; (iii) in terms of both plot-specific and overall differences. The same applied to the total soil and organic matter losses. In the case of the specific losses, however, only the individual values of the two sites together and of the ploughed site separately varied significantly with time-since-fire.

Comparison of the consecutive campaigns revealed that significant changes in hydrological and erosion processes principally occurred between campaigns 2 (November 2005) and 3 (March/April 2006) as well as between campaigns 5 (October 2006) and 6 (July 2007) (Table 4). These last two campaigns differed significantly for all the variables studied here. Total runoff stood out amongst the various variables in that the differences between campaigns 2 and 3 as well as between campaigns 5 and 6 were only statistically significant on a plot-wise basis and not also in general. This probably reflected the significant differences in runoff amounts between the extreme- and high-intensity RSE's (see Table 3), adding to campaign-wise variability.

In close agreement with the above-mentioned statistical results, the temporal variation in runoff and erosion revealed two distinctive patterns (Fig. 3). First, the median values were clearly higher for the first two and the last campaigns than for the third to fifth campaigns. Second, median values of the first five campaigns were noticeably lower than that of the last campaign. The first pattern applied to the absolute and relative runoff amounts as well as to the absolute sediment losses, whilst the second pattern concerned the relative sediment losses. A consistent element in the first pattern was further that the median value of the fourth campaign (July 2006) was lower than those of the third and fifth campaigns).

The temporal patterns in soil water repellency and other potential explanatory variables are shown in Fig. 4. The significant decrease in overland flow and total sediment losses between 4 and 9 months after the wildfire agreed well with a pronounced drop-off at both sites in topsoil water repellency from extremely hydrophobic to hydrophilic. The significant increase in runoff and erosion between 16 and 24 months after the wildfire, however, was less consistent with differences in repellency. Whilst at the ploughed site median ethanol classes were higher in July 2007 than October 2006, at the unploughed site they were basically the same. The limited hydrological impact of the very strong repellency of the unploughed soil in October 2006 could be due to the antecedent rainfall (10 mm in the two preceding days), enhancing the spatial variability in repellency and, thereby, opportunities for re-infiltration of overland flow (e.g. Shakesby et al., 2000; Keizer et al., 2005a).

The results of the summer 2006 campaign also casted doubt on the role of soil water repellency, with the ploughed site presenting the most puzzling case. All four RSE's at this site then produced the least runoff, even though repellency was equally strong as during the first two campaigns and also rather homogeneous (range of ethanol classes: 6–8; $n=10$). In the case of the unploughed site, the reduced runoff production in July 2007 could be due to the moderate median repellency level as opposed to the very strong/extreme level during fall 2005 and summer 2007. The discrepancy in water repellency between the unploughed and ploughed site during the July 2006 campaign can be explained by the 15 mm of rainfall that fell on July 19, 2006,

Table 4
Statistical comparison of runoff and erosion for various RSE campaigns together as well as for consecutive RSE campaigns. The overall comparison concerned the two sites together ("U&P") as well as separately, and the site-wise average values ("mean") as well as the values of the individual RSE-pairs ("Indiv."). The statically significantly outcomes ($\alpha=0.05$) of the Kruskal–Wallis test, Friedman test, MW U-test and Wilcoxon S–R test are indicated with "K", "F", "M" and "W".

Data sets and variables *Campaigns together*	**U&P**		**Unploughed**	**Ploughed**
	Mean	**Indiv.**	**Indiv.**	**Indiv.**
Total runoff (mm)	K/F	K/F	K/F	K/F
Overall runoff coefficient (%)	K/F	K/F	K/F	K/F
Total soil loss (g m^{-2})	K/F	K/F	K/F	K/F
Total organic matter loss (g m^{-2})	K/F	K/F	K/F	K/F
Specific soil loss (g m^{-2} mm^{-1} runoff)		K/F	K/F	
Specific o.m. loss (g m^{-2} mm^{-1} runoff)		K/F	K/F	

Consecutive campaigns	**Campaign *i*/*i*+1**				
	1/2	**2/3**	**3/4**	**4/5**	**5/6**
Total runoff (mm)		W			
Overall runoff coefficient (%)		M/W			M/W
Total soil loss (g m^{-2})		M/W			M/W
Total organic matter loss (g m^{-2})		M/W			M/W
Specific soil loss (g m^{-2} mm^{-1} runoff)					M/W
Specific o.m. loss (g m^{-2} mm^{-1} runoff)				M/W	M/W

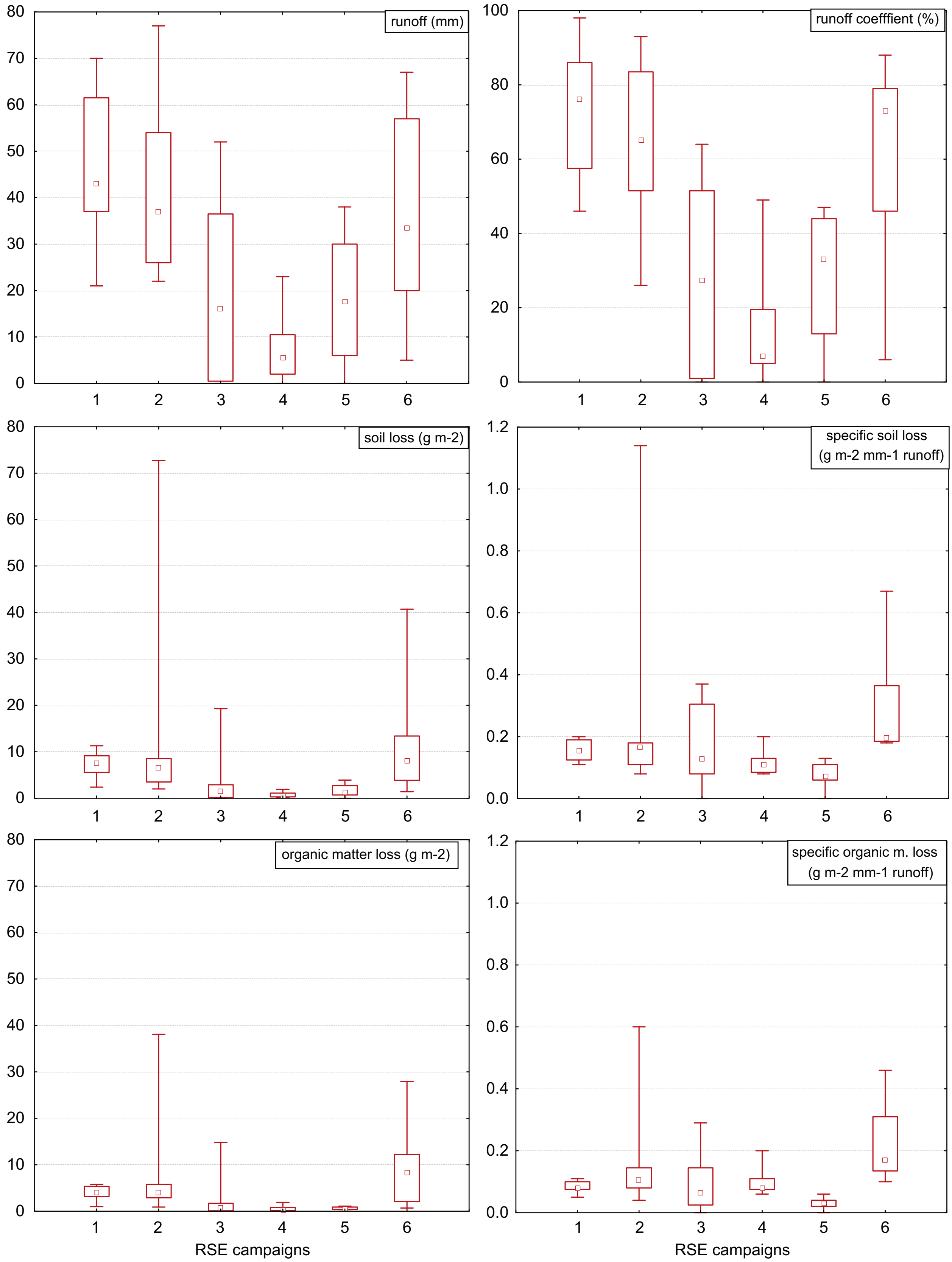

Fig. 3. Box-plots of runoff and soil and organic matter losses by individual RSE's at an unploughed and a ploughed eucalypt site for six field campaigns.

i.e. one *vs.* six days before the RSE's at the unploughed and ploughed site, respectively. On July 10 and July 24, 2006, repellency was very strong at both sites (Keizer et al., 2008).

The overall importance of vegetation recovery in limiting erosion during the study period was minor, as can be inferred from the comparatively high soil and organic matter losses of the

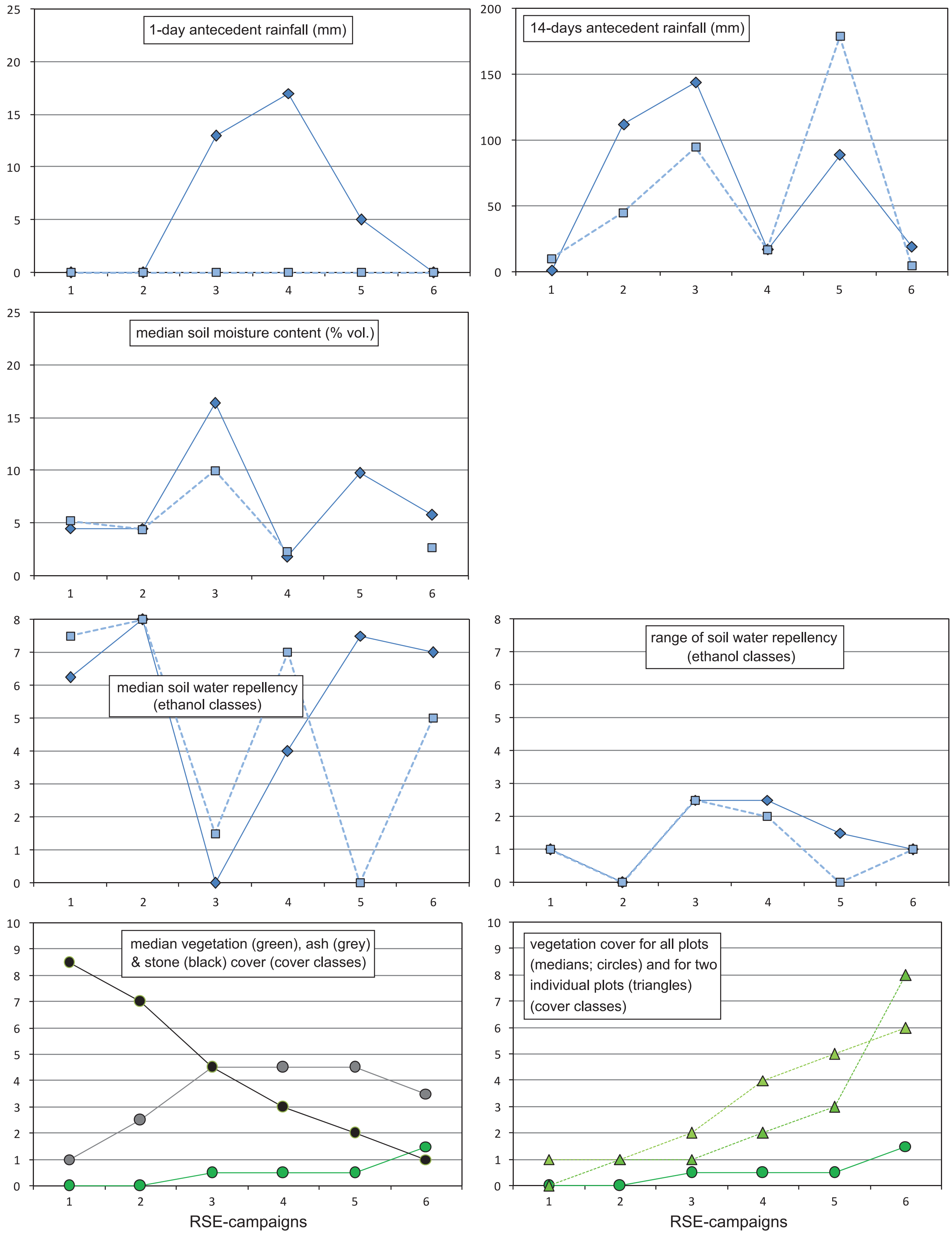

Fig. 4. Antecedent rainfall, initial soil moisture content and water repellency, and vegetation, ash and stone cover at an unploughed (diamonds) and a ploughed (squares) eucalypt site for six RSE campaigns.

last RSE campaign. Recovery of the ground vegetation was in fact limited during these 2 years following the wildfire (Fig. 4). By July 2006, vegetation cover was less than 20% in seven out of eight plots; by July 2007, it was less than 40% in all except the two plots depicted in Fig. 4. Shakesby et al. (1994) also indicated that ground vegetation in eucalypt stands recovered too slowly after fire to be effective within the first 2 years. The possible role of vegetation cover at the scale of individual plots is addressed next.

The RSE's by Sheridan et al. (2007) revealed a better overall agreement between the temporal patterns in runoff coefficient and soil water repellency than found here. Their highest runoff values were not restricted to the first few months after the wildfire but also occurred some 3 years later. Even so, differences in runoff could not be entirely attributed to water repellency, in particular the almost twice as high runoff coefficient 3 years compared to 1 month after fire under equally strongly repellent conditions. Clearly distinct from the current results was Sheridan's et al. (2007) finding of markedly higher sediment concentrations during the first year after fire. Such a decrease suggests a transition from transport- to sediment-limited conditions, as is also commonly observed in post-fire erosion plot studies under natural rainfall (see Shakesby and Doerr, 2006). Their specific sediment losses during the first post-fire year (2.26–7.19 g m^{-2} mm^{-1} runoff) clearly exceeded the present values. Their values for the subsequent 2 years (0.13–1.59 g m^{-2} mm^{-1} runoff), however, were comparable.

The only other study involving a time series of RSE's in wildfire-affected forests is that of Cerdà and Doerr (2005) in Aleppo Pine stands in eastern Spain. They employed the same simulator as in this study (application rate of 55 mm h^{-1}) and also permanent plots. During the first 3 years following fire, their RSE's produced higher runoff coefficients under wet than under dry conditions. This contrasting hydrological response could be explained by the low water repellency levels during this initial post-fire period, likely as a direct effect of fire. The erosion results of Cerdà and Doerr (2005) were also distinct from the present ones. The specific sediment losses dropped sharply from the first to the second year after fire and then more gradually afterwards. Only from the sixth year onwards the specific losses in Cerdà and Doerr (2005) fell below 0.40 g m^{-2} mm^{-1} runoff, thus becoming comparable to the bulk of the high-intensity values presented here. Their values for the first post-fire year (2.50–5.25 g m^{-2} mm^{-1} runoff) were not widely different from Sheridan's et al. (2007) above-mentioned figures for the first post-fire year, even though application rate and plot size were much smaller (55 vs. 100 mm h^{-1}; 0.25 vs. 3 m^2).

3.4. Spatial variability

Within-site differences. Overall differences between the same-intensity plots were not significant for any of the sites or variables (Table 5). This can be attributed to the above-mentioned, significant temporal variability between the various RSE campaigns. Campaign-specific differences, on the other hand, were significant in various instances, all of which involving extreme-intensity plots. The latter suggested that extreme events enhanced the inherent spatial variability in plot characteristics and, consequently, erosion processes. These significant differences, however, had different origins at the two sites. In the case of the ploughed site, the runoff response of the two extreme-intensity plots differed widely (Fig. 5). In turn, this discrepancy in runoff caused significant different total soil and organic matter losses, since the specific losses differed in the opposite sense. In the case of the unploughed site, by contrast, significant differences in specific soil losses contributed markedly to the significant differences in total soil losses.

As discussed before, the significantly lower amount of overland flow generated at one of the extreme-intensity plots at the ploughed site (plot P4) could be explained by its high litter cover (Fig. 6), possibly in combination with other factors. The significantly higher specific soil losses at plot U4 at the unploughed site were more difficult to explain, also because plot-specific data related to soil erodibility were not available. Post-fire vegetation recovery could play a role, since it was basically lacking at plot U4 but pronounced at plot U2 (Fig. 6). It would especially help explain why the plots' specific losses differed considerably less in July 2007 than in November 2005 (with a factor 3 and 6, respectively).

In particular during the campaigns of November 2005 and July 2007, the specific sediment losses recorded at plot U4 stood out amongst the present values. Compared to other studies, however, these values (1.7 and 1.1 g m^{-2} mm^{-1} runoff) were hardly suspicious. Leighton-Boyce et al. (2007) reported a mean value of 2.3 g m^{-2} mm^{-1} runoff for an unburnt eucalypt site where the litter was removed prior to the RSE's. Specific sediment losses in

Table 5
Statistical comparison of within-site and between-site variation in runoff and erosion by high- and extreme-intensity RSE's. The within-site comparison concerned the same-intensity plots at each study site; the between-site comparison concerned the same-intensity plots at different sites, either the site-wise average values ("mean") or the values of the individual RSE's. The statically significantly outcomes ($\alpha = 0.05$) of the MW *U*-test and Wilcoxon *S–R* test are indicated with "M" and "W" or, in the case of the between-site comparison of individual plots, with the codes of the unploughed plots (U1–U4) that are significantly different from the ploughed plots (P1–P4).

Variability and variables *With-in site*	**Unploughed**		**Ploughed**	
	High	**Extreme**	**High**	**Extreme**
Total runoff (mm)				W
Overall runoff coefficient (%)				W
Total soil loss (g m^{-2})		W		W
Total organic matter loss (g m^{-2})				W
Specific soil loss (g m^{-2} mm^{-1} runoff)		W		
Specific o.m. loss (g m^{-2} mm^{-1} runoff)				

Between-site	**Means**		**Individual plots**			
	High	**Extr.**	**High**		**Extreme**	
	Ploughed		**P1**	**P3**	**P2**	**P4**
Total runoff (mm)	W	W	U1		U4	U4
Overall runoff coefficient (%)	W	W	U1		U4	U4
Total soil loss (g m^{-2})		W	U1		U4	U4
Total organic matter loss (g m^{-2})		W			U4	U4
Specific soil loss (g m^{-2} mm^{-1} runoff)						U2
Specific o.m. loss (g m^{-2} mm^{-1} runoff)						U2

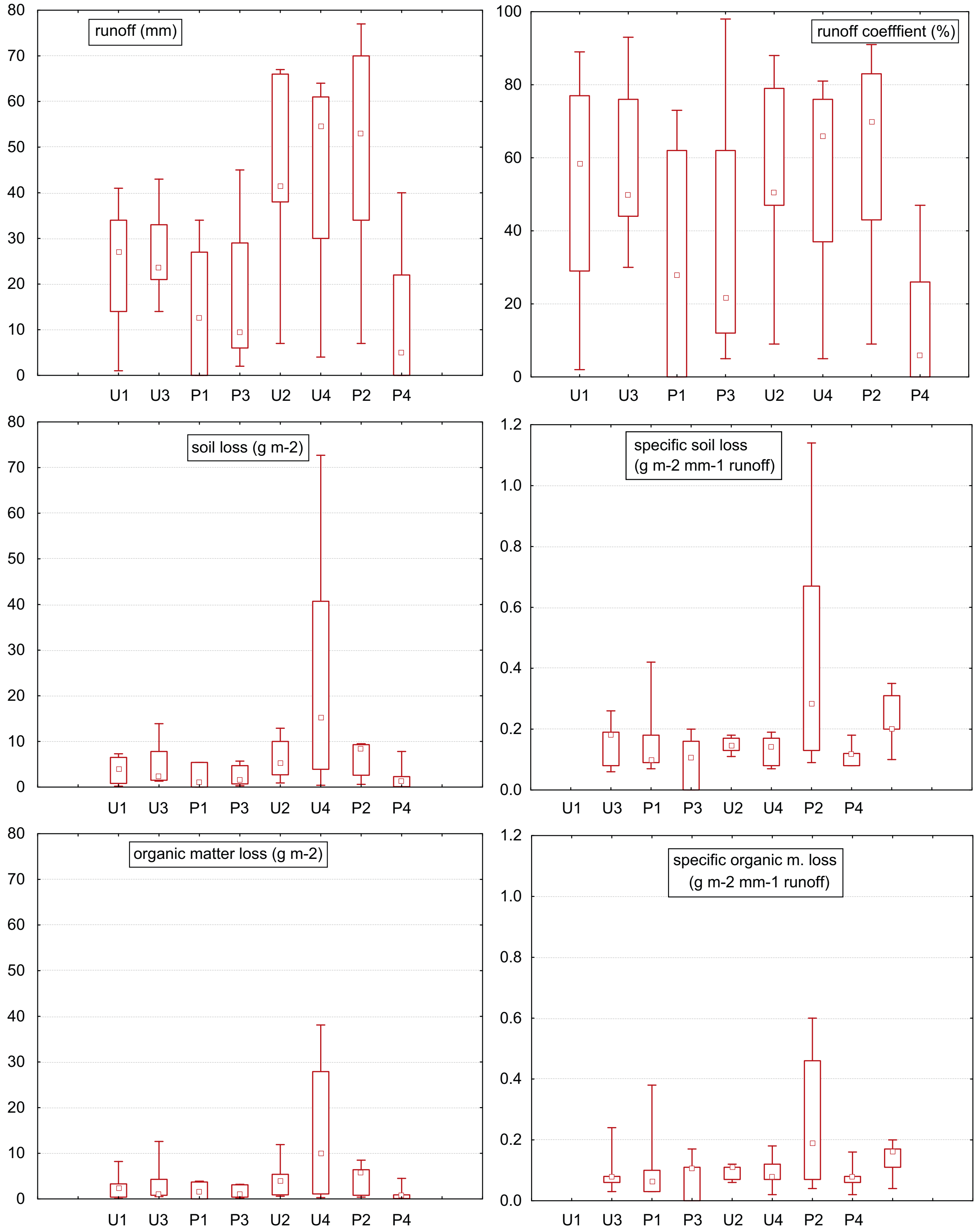

Fig. 5. Box-plots of runoff and soil and organic matter losses over various RSE campaigns for four high- ("1" and "3") and four extreme-intensity ("2" and "4") plots at an unploughed ("U") and a ploughed ("P") eucalypt site.

Cerdà and Doerr (2005) and Sheridan et al. (2007) equally exceeded 2 g m^{-2} mm^{-1} runoff. Also the present spatial variability in specific sediment losses in concurrent RSE's was not extraordinary in comparison to these latter two studies (Cerdà and Doerr, 2005: 2.50–4.46 g m^{-2} mm^{-1} runoff; Sheridan et al., 2007: 7.2–24.3 g m^{-2} mm^{-1} runoff).

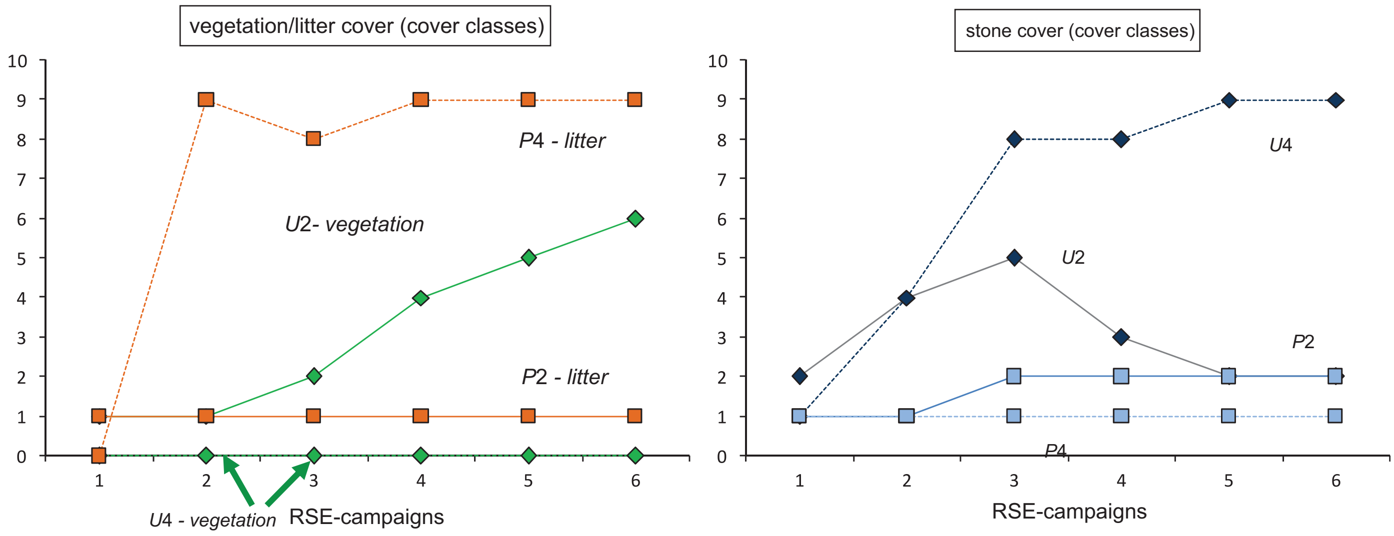

Fig. 6. Vegetation/litter, and stone cover of four extreme-intensity ("2" and "4") plots at an unploughed ("U") and a ploughed ("P") eucalypt site for six RSE campaigns.

The credibility of the relatively high losses at plot U4 was further corroborated by the strong increase in the plot's stone cover (Fig. 6). It remained unclear, however, if this stone lag already existed before the wildfire, becoming increasingly exposed by the subsequent removal of the ash layer and the lack of vegetation recovery, or whether it developed during the study period. The former explanation is perhaps most likely, namely, Shakesby et al. (1993) and Terry (1996) reported a much higher specific sediment loss (11.9 g m^{-2} mm^{-1} runoff) for the initial phase of stone lag formation in an eucalypt stand.

Between-site differences. The unploughed site revealed a significantly stronger average runoff response than the ploughed site (Table 5). This was true for both the absolute and relative runoff amounts and for both the high- and extreme-intensity RSE's, as is also easily perceived from Fig. 7. By contrast, significant differences in average sediment losses were restricted to the total soil and organic matter losses of the extreme-intensity RSE's, again with the values at the unploughed site being highest. Nonetheless, also the high-intensity RSE's revealed some tendency towards higher average soil losses at the unploughed site, namely, the values at the unploughed site were highest in five out of the six RSE campaigns.

Although the Wilcoxon's test results for the individual plots were indicative only, they allowed further insight in the average between-site differences (Table 5). This especially applied to the extreme-intensity RSE's, namely, the significant difference in the average extreme-intensity values was due to pronounced spatial variation at the ploughed site and, more specifically, the deviant behaviour of plot P4, as also readily appreciated in Fig. 5. Plot P4 not only produced, as mentioned above, consistently less sediment and/or runoff than the other extreme-intensity plot at the ploughed site and even the neighbouring high-intensity plot but also then the two extreme-intensity plots at the unploughed site. Thus, the extreme-intensity results of this study were strongly influenced by a highly localised and rather accidental factor like litter fall from scorched crowns.

4. Conclusions

The main conclusions from this study include the following:

- Extreme-intensity RSE's (80–85 mm h^{-1}) tended to generate larger amounts of runoff and, thereby, higher losses of soil and organic matter than high-intensity RSE's (45–50 mm h^{-1}); however, this tendency was not invariable either in space or, at a certain location, through time.
- Within-site variability in runoff and erosion response was more pronounced in the case of the extreme- than high-intensity RSE's, so that their modelling will require greater efforts in terms of model calibration and/or obtaining plot-specific information.
- Runoff and associated sediment losses varied significantly with time-after-fire; however, this temporal pattern did not correspond to a simple decrease with time but had a marked seasonal component, which broadly agreed with the role of topsoil water repellency in enhancing overland flow generation under dry antecedent weather conditions.
- The risk of enhanced runoff generation and erosion in recently burnt eucalypt stands does not necessarily disappear with the first significant rains after the wildfire but can persist through most of the first autumn and also re-appear after subsequent dry spells as long as 2 years later; this could be attributed to the typically pronounced water repellency of eucalypt forest soils combined with an often slow post-fire vegetation recovery.
- Contrary to expected, the unploughed site produced more runoff and erosion than the adjacent unploughed site. Besides soil properties altered by ploughing, the difference in slope angle between the two sites could play a role. These possible explanations will be further explored using the MEFIDIS erosion model as research tool.
- The sediment losses at the two study sites were low compared to those obtained with similar methodologies (i.e. field rainfall simulation experiments on small plots) following wildfires in other parts of the world. Nonetheless, they need to be evaluated against the typically shallow soil depths on the steep hill slopes in the study area, with the elevated organic matter fractions in the observed sediment losses requiring special attention.

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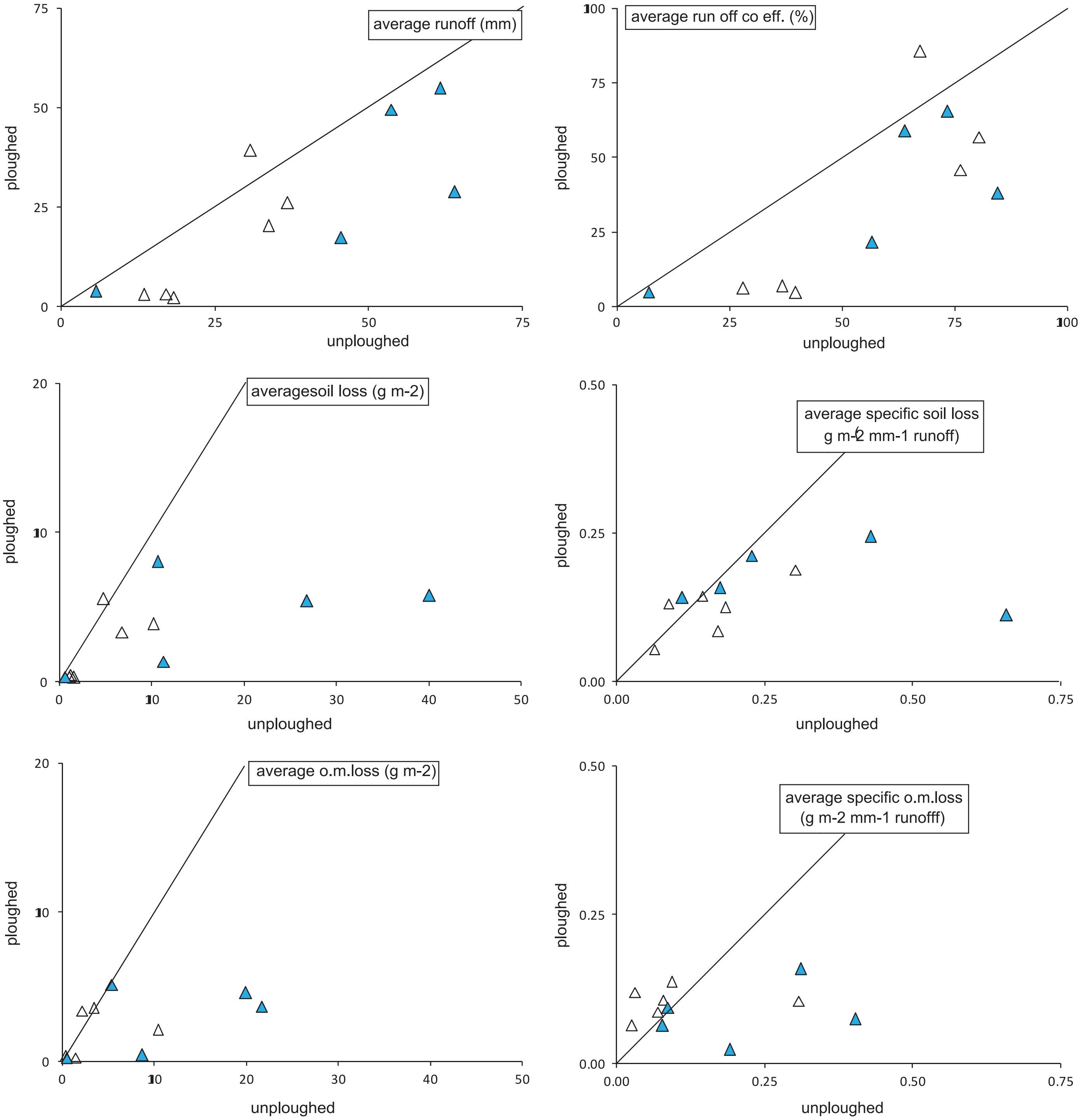

Fig. 7. Average absolute and relative amounts of runoff and soil and organic matter losses by high- (open triangles) and extreme-intensity (closed triangles) RSE's at an unploughed *vs.* a ploughed eucalypt site.

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CHAPTER 3

Effectiveness of forest residue mulching in reducing post-fire runoff and erosion in a pine and a eucalypt plantation in north-central Portugal

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Effectiveness of forest residue mulching in reducing post-fire runoff and erosion in a pine and a eucalypt plantation in north-central Portugal

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ABSTRACT

Fire-enhanced runoff generation and erosion are an important concern in recently burnt areas worldwide but their mitigation has received little public and scientific attention in Portugal. The present study addressed this knowledge gap for the two principal fire-prone forest types in Portugal, testing the effectiveness of a type of mulch that is widely available in the study region but has been little utilized and poorly studied so far. For logistic reasons, two somewhat different forest residue mulches were tested in a eucalypt plantation (eucalypt chopped bark) and a nearby Maritime Pine stand (eucalypt logging slash). Arguably, however, more important differences between the two study sites were those in fire severity, resulting in an elevated litter cover prior to mulching at the pine site but not at the eucalypt site, and in experimental design, with eight bounded erosion plots of 16 m^2 installed at the eucalypt site as opposed to only four at the pine site (due to its limited size). Mulching was applied four months after the wildfire and two months after installation of the plots. Rainfall, runoff and sediment and organic matter losses were measured on a 1- to 2-weekly basis. Mulching proved highly effective at the eucalypt site, on average reducing the runoff coefficient from 26 to 15% and sediment losses from 5.41 to 0.74 Mg ha^{-1}. This mulching effect was also statistically significant, albeit only for the more important runoff and erosion events, and corresponded to a significant role of litter cover in explaining the variation in runoff and erosion. At the pine site, by contrast, mulching had no obvious effect. In all probability, this was first and foremost due to the comparatively small amounts of runoff and sediments produced by the untreated pine plots (5% and 0.32 Mg ha^{-1}) and, as such, due to the extensive needle cast following a low severity fire.

1. Introduction

Wildfires are a common phenomenon in present-day Portugal, having affected on average 110,000 ha of rural lands per year between 1980 and 2010 (AFN, 2011). This can be attributed, besides climate conditions, to a combination of socio-economic factors, in particular the large-scale replacement of native Portuguese forest by commercial plantations of fire-prone tree species such as pine and eucalypt and the decline in traditional practices like grazing and coppicing that reduced the accumulation of flammable materials (Pereira et al., 2006a; Radich and Alves, 2000; Shakesby et al., 1996). The frequency of forest fires in Portugal is also not expected to diminish substantially in the next decades, in part due to an increase in meteorological conditions propitious to wildfires (Pereira et al., 2006a,b).

Wildfires are well documented to increase runoff generation and soil erosion, as mentioned in various studies in Portugal (e.g. Coelho et al., 2004; Ferreira et al., 2008; Malvar et al., 2011; Shakesby et al., 1996). Apart from heating-induced changes in soil properties such as soil water repellency and aggregate stability (Shakesby and Doerr, 2006; Varela et al., 2010), removal of the protective vegetation and litter cover is a key factor in fire-enhanced runoff and sediment losses (Shakesby, 2011). For this precise reason, a commonly applied emergency treatment for reducing post-fire erosion risk, such as mulching, is based on the principle of applying materials that provide an effective ground cover (Cerdà and Doerr, 2008; Robichaud et al., 2000). In Portugal, however, mulching or other types of emergency treatments have rarely been employed in land management of recently burnt areas, although this is changing due to the implementation of PRODER-funded measures (under sub-Action 2.3.2.1) in selected areas that were affected by wildfires during the summer of 2010.

In Portugal, post-fire emergency treatments have also received little research attention. Prior to the present work, the only field study into the effectiveness of post-fire soil conservation measures was that

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reported by Shakesby et al. (1996) and Walsh et al. (1994). Similar to this study, mulch composed of forest residues from logging was applied in a eucalypt and a pine plantation. However, mulching was done two to three years after the wildfire rather than within the first few months—as was the case in this study—when soil erosion risk is supposedly at its maximum (Shakesby and Doerr, 2006).

Outside Portugal, the effectiveness specially of straw mulches has been exhaustively studied under field conditions (e.g. Badía and Martí, 2000; Bautista et al., 1996; Fernández et al., 2011; Groen and Woods, 2008; Riechers et al., 2008; Rough, 2007; Wagenbrenner et al., 2006). This is particularly true in comparison with mulches from woody plant material and, more specifically, wood chips (Fernández et al., 2011; Kim et al., 2008; Riechers et al., 2008). Often-cited advantages of straw, besides its elevated effectiveness in reducing soil erosion, are its wide availability, low costs and low specific weight. Whilst the availability of straw may be limited in many parts of the world (Foltz and Wagenbrenner, 2010), including Portugal, the low specific weight can become a disadvantage in areas with strong winds, especially during the period between straw application and the first rainfall events (Robichaud et al., 2000). In recent years, forest residues have become increasingly harvested in Portugal for use in biomass energy plants and, as such, can be a viable alternative to straw, in spite of the logistic implications of its higher specific weights. Nonetheless, the effectiveness of woody mulches under field conditions remains unclear. Namely, Shakesby et al. (1996) found their forest residue mulch to be ineffective at one of the two study sites, whilst Fernández et al. (2011) and Riechers et al. (2008) reported wood chip mulch to be less effective than straw mulches. Evidence from rainfall simulation experiments suggested that the shape of the woody materials could be of critical importance, with wood shreds and strands rather than wood chips being as effective as straw (Foltz and Dooley, 2003; Yanosek et al., 2006).

The present study had as it main aim to contribute to a better knowledge and understanding of hydrological and erosion processes following wildfire and, in particular, how they are influenced by mulching. More specifically, the following research gaps were addressed: (i) short- to medium-term post-fire conditions, i.e. the first 1.5 years of the fire-induced window-of-disturbance; (ii) high-resolution temporal patterns (approximately weekly) in post-fire runoff and erosion as well as in key explanatory variables, including soil water repellency (for its supposed role in eucalypt stands, especially after Leighton-Boyce et al., 2007; Malvar et al., 2011; Sheridan et al., 2007); (iii) effectiveness of forest residue mulching in the two principal fire-prone forest types in Portugal, i.e. eucalypt and maritime pine plantations.

2. Materials and methods

2.1. Study area and sites

The study area was located in north-central Portugal, in the locality of Pessegueiro do Vouga, municipality of Sever do Vouga (40° 43′ 05″N; 8° 21′15″W; 100 m.a.s.l. of elevation). On 10 August 2007, a wildfire destroyed a relatively small area (approximately 10 ha). The burnt area was predominantly covered by plantations of eucalypt (*Eucalyptus globulus* Labill.) but included a few, comparatively small stands of maritime pine (*Pinus pinaster* Ait.). Although this situation allowed the study of the two predominant fire-prone forest types in Portugal, the limited number and size of the available pine stands implied compromises in terms of site selection as well as experimental design (see Section 2.2). The eucalypt study site was selected for its steep slope and comparatively higher fire severity, as indicated by the total consumption of the canopies. The pine site was chosen for its closeness, comparative slope and exposition to the eucalypt site, although it presented a markedly lower fire severity, with the canopies only partially consumed by the fire (Table 1).

Table 1
General description of the two study sites and of the experimental design. Values followed by different letters are statistically different ($p<0.05$, pair-wise t-test).

Site	Eucalypt	Pine
General characteristics		
Tree	*Eucalyptus globulus* Labill.	*Pinus pinaster* Aiton.
Age and plantation cycle	15; 3rd re-growth	30
Slope angle (°)—average ± sd	25° ± 3.6	24° ± 3.6
Fire severity indicators (Aug. 2007)	Moderate	Low
Ash color	Black, grey	Black
Tree canopy consumption	Total	Partial
Tree scorch height (m)	9	7
Mean litter cover (%)	<10	60
Soil characteristics (0–15 cm)	n = 9	n = 9
Stoniness (%)	54.4 ± 9.3^a	64.7 ± 4.4^b
Sand fraction (%)	39.8 ± 8.3^a	31.6 ± 3.7^b
Silt and clay fraction (%)	5.9 ± 1.3^a	3.6 ± 1.6^b
Soil organic matter (%)	12.2 ± 2.9^a	9.9 ± 2.7^b
Experimental design		
Number of control/treated plots	4/4	2/2
Projected plot surface (m^2)—average ± sd	15.8 ± 1.0	14.3 ± 0.7
Mulching type	Eucalypt chopped bark	Eucalypt logging slash
Application rate (kg m^{-2})	0.87	1.75
Increase in ground cover by mulch (%)	67	76

The climate can be classified as humid meso-thermal with a moderate but extended dry summer (Köppen: Csb; DRA-Centro, 1998). Mean annual temperature at the nearest climate station (Castelo-Burgães: 40° 51′10″N, 8° 22′44″W, 306 m.a.s.l., 1977–2009) is 14.8 °C, while mean monthly temperatures range from 8.9 °C in January to 21 °C in July (SNIRH, 2011). Annual rainfall at the nearest rainfall station (Bouça-Pessegueiro do Vouga: 40° 41′36″N , 8° 22′24″W; 152 m.a.s.l., 1977–2005) is 1546 mm on average but varies strongly from 843 mm in dry years to 2151 mm in wet years (DRA-Centro, 1998).

The soils at both study sites were shallow, 25–30 cm deep Umbric Leptosols (FAO, 1988) developed over Pre-Cambrian schist from the Hespheric Massif (Pereira and FitzPatrick, 1995), as verified by digging out two soil profiles at each site. From the upper 15 cm of these profiles, a total of nine samples were collected in February 2008 and later analysed, using standard laboratory methods (mechanical sieving and loss-on-ignition), to determine the fractions of stones, sand, silt and clay, and organic matter. The topsoil at both study sites was very coarse, with a stone content of over 50% and a sandy texture (Table 1). The between-site differences in the soil fractions were minor but nonetheless statistically significant ($p<0.05$, pair-wise *t*-test).

2.2. Experimental design, field and laboratory measurements

Because of the small size of the maritime pine stands in the burnt area, it was impossible to implement exactly the same experimental design at both study sites. While the eucalypt site was instrumented with eight erosion plots of 2 m wide by 8 m long, only four could be installed at the pine site. The installation of all the plots was completed by 02 October 2007 but the treatment with mulch was not carried out until 10 December 2007. Mulch was applied manually to half of the plots at each site, which were selected randomly. For logistic reasons, somewhat different forest logging residues were used at the two sites (Table 1). For the treatment of the eucalypt plots, chopped bark mulch was obtained at a depot 20 km from the study area, where eucalypt logs are debarked before their transport to a paper pulp factory and the bark is chopped into 10–15 cm wide 2–5 cm long fibers before their transport to a biomass energy plant. On the pine site, in line with Shakesby et al. (1996), the mulch consisted of logging slash residues collected from the soil after clearcutting of an adjacent unburned eucalypt stand (300 m distance from the pine site). Due to the differences in material, a higher application rate of the mulch was needed at the pine treated plots to achieve a ground cover comparable to the eucalypt treated plots (i.e. 70–80%, Table 1).

The erosion plots were delimited using metal sheets of 60 cm long by 15 cm high that were inserted into the soil to a depth of 5 to 10 cm. All the plots had a rhomboid shape, with a trench dug at the upper limit to avoid run-on into the plots. Following the design of Shakesby et al. (1991), a modified gerlach trap (Gerlach, 1967) was installed at the base of each plot to intercept the runoff and retain the coarser material using a net with a mesh width of 0.5 mm. The runoff was routed to a tipping-bucket device using a garden hose, and then to a set of three interconnected 70-liter tanks. The main purpose of the tipping-bucket devices was to verify and correct the runoff measurements. From October 2007 onwards, on a weekly basis, runoff was measured and 1500 ml samples were gathered from all individual tanks. Also the sediments accumulated in the gerlach traps were collected. The runoff and sediment samples were subsequently analyzed using standard laboratory procedures (APHA, 1998) to determine sediment and organic matter loads.

During each field trip, rainfall at the two study sites was measured using two automatic rainfall gauges (sensitivity 0.1 and 0.2 mm) in combination with seven totalizer rain gauges for validation purposes.

The moisture content of the topsoil was monitored in two distinct manners. Within the plots, soil moisture was measured with a non-destructive method, using pultrusion tubes inserted into the soil in which a TDR-type Delta-T® PR2-probe is lowered to carry out readings at different depths (including at 0–10 cm, analyzed in this study). Between 22 October and 20 November 2007, one pultrusion tube was installed in each of the twelve plots, and readings were carried out during 37 fieldtrips. Unfortunately, the two tubes of the pine control plots, one in a eucalypt control and one in a eucalypt treated plot malfunctioned most of the time, so the data were not included here. Destructive measurements of soil moisture content were taken outside the plots, in a slope section that was specifically reserved for that purpose and considered representative of the control conditions. In these slope sections, a 20-m long transect comprising three equidistant points was laid out at shifting positions on a total of 31 sampling occasions between October 2007 and December 2008. At each transect point, soil moisture was then measured three times at two depths (0–5 and 5–10 cm), using a Delta-T® ML2-sensor. For technical but especially logistic reasons, destructive moisture readings were not possible on 7 dates in the case of the pine site.

Besides soil moisture, soil water repellency was measured along the above-mentioned transects on each possible sampling occasion following the 'Molarity Ethanol Drop test' (Doerr, 1998). In each transect point, three replicate measurements at four different sampling layers were carried out (soil surface and 0–5, 5–10 and 10–15 cm soil depth). Each measurement involved applying three droplets of increasing ethanol concentration to fresh parts of the soil until infiltration of at least two of three droplets of the same concentration within 5 s. Like in Keizer et al. (2005, 2008), the following nine volumetric ethanol percentage concentrations and, in between brackets, corresponding ethanol classes were used: 0 (0), 1 (1), 3 (2), 5 (3), 8.5 (4), 13 (5), 18 (6), 24 (7), 36 (8). In this study, the overall frequency of the two highest ethanol classes measured in all the depths was analyzed as a combined indicator of repellency severity and homogeneity.

The ground cover within the 12 erosion plots was measured eight times at regular intervals between 31 October 2007 and 2 June 2008, and again at the end of the study period. A grid of 1×1 m divided in rows and columns of 10 cm wide was placed at three fixed positions in the lower, middle and upper parts of each plot. At the 100 intersection points between rows and columns, the ground cover was recorded in the field according to the following four categories: "stones" (rock outcrop and stones bigger than 2 mm); "bare soil" (which included ashes and charcoal); "litter" (including the applied mulch) and vegetation.

2.3. Data analysis

The effect of mulching in overland flow and sediment losses was tested by means of a two-way repeated measures analysis of variance (Ott and Longnecker, 2001). The number of days since wildfire and the read-out dates were used as periods of repeated measurements for runoff amount, runoff coefficient, sediment losses and organic matter losses. The underlying assumptions of normality and homoscedacity were verified, and both the runoff and erosion values had to be transformed, by taking the square and fourth roots, respectively, for the Kolmorogonov–Smirnov test not to reject normality at $\alpha=0.05$. In addition, the three smallest rainfall events (<3.6 mm) had to be excluded for the transformed data to meet the normality assumption.

Multiple regression models were constructed to determine how well the observed runoff and erosion could be explained by selected independent variables. This was done using a stepwise forward selection procedure, i.e. the REG procedure (Littell et al., 1996), in which the independent variables were selected in order of their significant contribution ($p<0.05$) to the explained variance. As in the repeated measures ANOVA, the square and fourth roots, for runoff amount and sediment losses were used, since model residuals met the normality assumption without exception. Due to missing data, various data sets comprising different combinations of read-outs and sets of independent variables were analysed. The complete data set involved 32 read-outs and six independent variables, i.e. rainfall amount and intensity, and the above-mentioned four ground cover classes. The "limited" data set involved 5 fewer read-outs but one more independent variable (i.e. soil moisture as measured with the PR-probe); whilst the "partial" data set involved 12 fewer read-outs but two more independent variables (i.e. soil moisture as measured with the ML2-sensor and frequency of extreme repellency).

3. Results

3.1. Overall rainfall, runoff and erosion values

In terms of total rainfall, the treatment period agreed well with average climate conditions. Between 10 December 2007 and 23 December 2008 1546 mm of rainfall were registered in the study area, exactly the same as the above-mentioned, long-term mean annual rainfall at the nearby Bouça station. Rainfall was much less during the pre-treatment period (138 mm) and even insignificant between the occurrence of the wildfire on 10 August 2007 and the completion of the plot installation on 02 October 2007 (approximately 10 mm, Fig. 1).

Prior to mulching, the control and the to-be-treated plots at the eucalypt site produced, on average, basically the same runoff amounts as well as sediment and organic matter losses (Table 2). At the pine site, by contrast, the control plots generated, on average, 40% less runoff and 55–60% less sediment and organic matter than the to-be-treated plots. As for between-site comparability, the to-be-treated pine plots differed little from the eucalypt plots in average runoff amounts (5–10% less) but noticeably more in average sediment losses (34–40% less).

Following mulching, the control and treated plots at the eucalypt site revealed marked differences in runoff and especially erosion, with the treated plots producing 43% and even 86% lower amounts of overland flow and sediment losses, respectively (Table 2). In the case of the pine site, on the other hand, the average differences between the control and treated plots were almost inexistent, but still

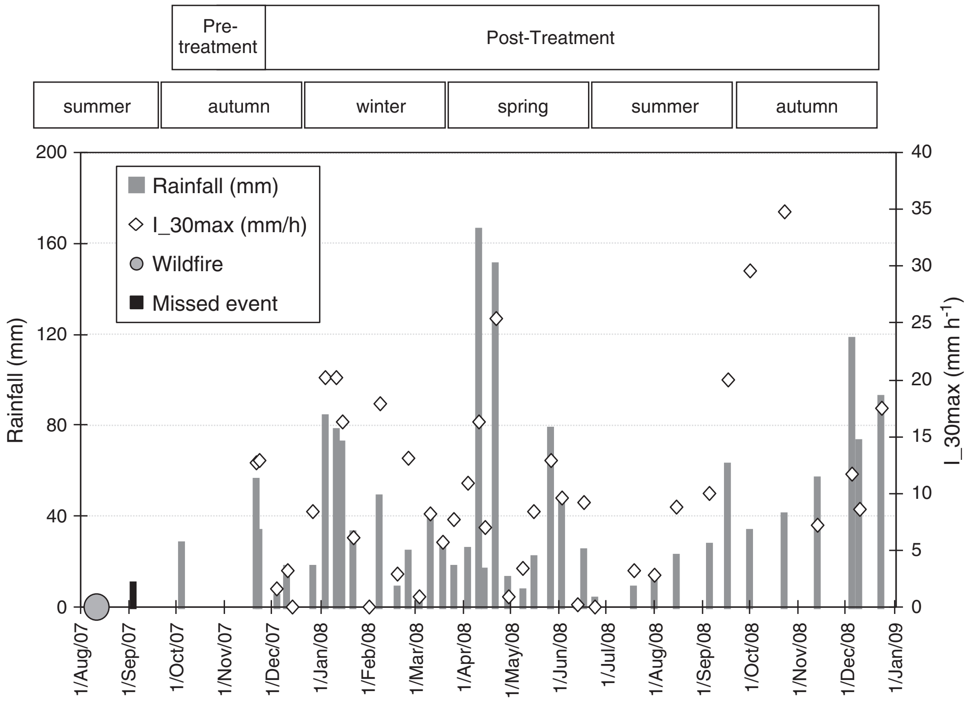

Fig. 1. Rainfall total and intensity of the individual measurements periods during the first 16 months after the wildfire in August 2007.

the control plots produced less runoff and sediments (10% and 16%, respectively) than the treated ones. Even so, the runoff ratios of post- to pre-treatment phases suggested that mulching was effective at the pine site as well. Namely, because those ratios were noticeably higher in the case of the control plots than of the treated plots, for runoff (4.1 vs. 2.6) and also for sediment losses (5.8 vs. 2.8). Similarly, the differences that existed between the control plots at the two study sites during the pre-treatment period were markedly amplified during the subsequent study period. Compared to the concurrent values at the eucalypt site, overland flow generation at the pine control plots dropped, on average, from 55% to 20%, and sediment and organic matter losses from roughly 30% to just over 5%.

3.2. Temporal patterns in rainfall, runoff and erosion

Despite the fact that the control and treated plots at the eucalypt site differed strongly in their overall runoff and erosion figures, the repeated measures ANOVAs did not reveal an unequivocal role of the mulching with chopped eucalypt bark. This was due to the presence of significant interactions between treatment and time-since-fire effects ($p<0.05$). The runoff coefficient constituted an exception ($p=0.3$), varying significantly with both factors individually ($p<0.05$). In the case of the absolute runoff response, the significant interaction could be attributed to the smaller rainfall events. Removal of the 11 read-outs with less than 17.5 mm rainfall from the original data set of 32 read-outs turned the interaction effect insignificant (albeit only just: $p=0.05$), such that both the treatment and the time-since-fire came to have a separate significant effect on runoff amounts. In the case of the sediment and organic matter losses, on the other hand, the bulk of the read-outs (27) needed to be removed from the data set to eliminate the significant interaction effect, reflecting the fact that sizeable sediment losses occurred much less frequently than substantial runoff amounts (Fig. 2).

In the case of the pine site, the repeated measures ANOVAs did not even hint at significant interaction effects, either for runoff amounts and coefficients or for sediment and organic matter losses ($p=0.8$). From the individual factors, mulching with eucalypt logging slash did not play a significant role in the case of any of these four parameters ($p=0.3$) but time-since-fire did in all four instances ($p<0.05$).

The pronounced temporal variation in rainfall, runoff and sediment losses was summarized by season (Table 2), and so were the corresponding temporal patterns in treatment effectiveness (Fig. 3). The overland flow generated by the untreated eucalypt plots exhibited a strong seasonal variation. It varied with roughly a factor 3 from around 50 mm during the driest seasons (autumn 2007 and summer 2008) to about 150 mm during winter 2007/08 and autumn 2008, whilst the rainiest season (spring 2008) assumed an intermediate position with 110 mm. Spring 2008 also stood out for producing comparatively little runoff in the case of the untreated pine plots, its mean runoff coefficient being at least twice as low as that of the other four seasons. Amongst these other seasons, autumn 2007 or,

Table 2
Pre- and post-treatment and season-wise runoff and erosion for the control plots (EC and PC) and treated plots (ET and PT) at the Eucalypt and Pine study sites.

	No. of read-outs	Rainfall amount	Total runoff (mm)				Mean runoff coefficient (%)				Total sediment losses (Mg ha^{-1})				Total organic matter losses (Mg ha^{-1})			
		(mm)	EC	ET	PC	PT	EC	ET	PC	PT	EC	ET	PC	PT	EC	ET	PC	PT
Pre-treatment (autumn 2007)	5	138	41	43	23	39	30	31	16	28	0.21	0.22	0.06	0.13	0.11	0.11	0.03	0.08
Post-treatment (winter 2007–autumn 2008)	35	1546	466	267	93	102	30	17	6	7	5.41	0.74	0.32	0.37	2.47	0.32	0.17	0.13
Winter 2007–2008	11	434	158	98	42	57	36	23	10	13	1.40	0.22	0.09	0.15	0.65	0.10	0.05	0.05
Spring 2008	12	565	110	42	12	12	19	7	2	2	2.07	0.12	0.03	0.03	0.95	0.06	0.02	0.01
Summer 2008	6	135	48	36	11	10	36	26	8	8	0.26	0.10	0.08	0.09	0.14	0.04	0.04	0.03
Autumn 2008	6	412	150	92	28	24	36	22	7	6	1.69	0.30	0.12	0.10	0.72	0.12	0.06	0.03

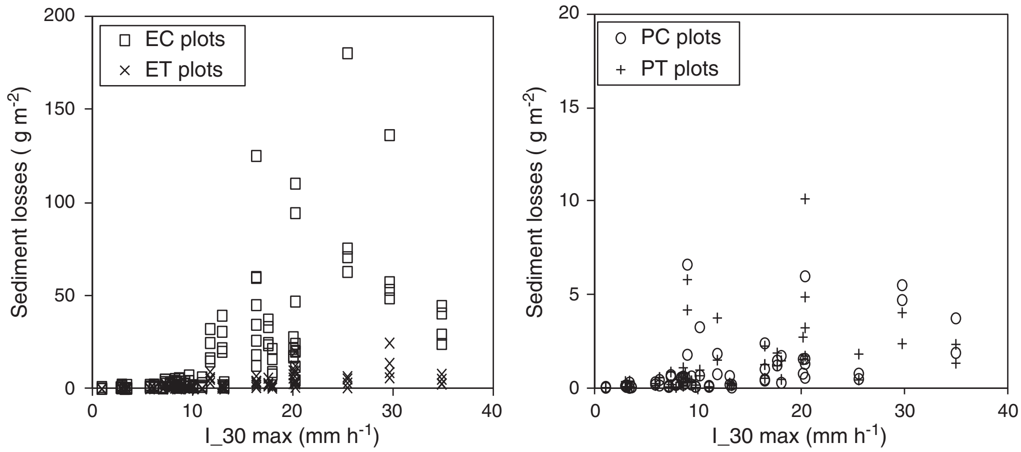

Fig. 2. Relationship of sediment losses with rainfall intensity for the treated (ET and PT) and control (EC and PC) plots at the eucalypt site (left) and pine site (right).

in other words, the first season following the wildfire was exceptional for precisely the opposite reason but only in the case of the untreated plots at the pine site.

Unlike runoff, sediment losses at the untreated eucalypt plots closely followed the seasonal pattern in rainfall. They were, on average, a factor 10 higher during spring 2008 than during autumn 2007 (2.07 vs. 0.21 Mg ha^{-1}). The discrepancy between the hydrological and erosion response of the untreated eucalypt plots was to a large extent due to two extreme events of roughly 150 mm that occurred during April 2008 and produced 80% of the season's sediment losses as opposed to 50% of the season's runoff. In the case of the untreated pine plots, the seasonal variation in sediment losses neither agreed well with that in runoff nor with that of rainfall. Instead, specific sediment losses were clearly lower during the first three seasons (averaged to 0.22 g m^{-2} mm^{-1} runoff) than during autumn and especially summer 2008 (0.41 and 0.76 g m^{-2} mm^{-1} runoff, respectively).

The effectiveness of mulching with eucalypt chopped bark varied between the four seasons in much the same manner as rainfall did (Fig. 3). The relative reductions in both runoff and sediment losses at the eucalypt site were at their minimum during the driest season (summer 2008) and at their maximum during the rainiest season (spring 2008). Throughout the treatment period, mulching was consistently more effective in reducing erosion than overland flow at the eucalypt site, and markedly so. Mulching effectiveness revealed completely distinct patterns at the pine site compared to the eucalypt site. First, effectiveness contrasted sharply between the first post-treatment season and the three subsequent seasons (i.e. between markedly negative and roughly zero to marginally positive) and second, effectiveness differed little between runoff and erosion.

3.3. Temporal patterns in ground cover, soil moisture and water repellency

In November 2007, the plots at the two study sites differed markedly in their ground cover (Fig. 4). Whilst the mean litter cover of the control and to-be-treated eucalypt plots was around 10%, that of the pine plots was roughly 50%, mainly due to needle cast from the scorched pine canopies (which continued until late January 2008). At the same time, a major discrepancy also existed in the total cover of bare soil and ashes, amounting to 70% at the eucalypt plots vs. 40% at the pine plots. These site differences were by and large eliminated by mulching, resulting in a mean litter cover of about 70% for the treated plots at the pine as well as eucalypt site. One year later, however, the mean litter cover was basically the same at the treated pine plots but had decreased noticeably at the treated eucalypt plots (20%). Also in the case of the untreated plots, the pine site revealed less pronounced cover changes than the eucalypt site, where an increase in average stone cover of roughly 20% occurred at the expense of a decrease in particular in ash cover. Worth special

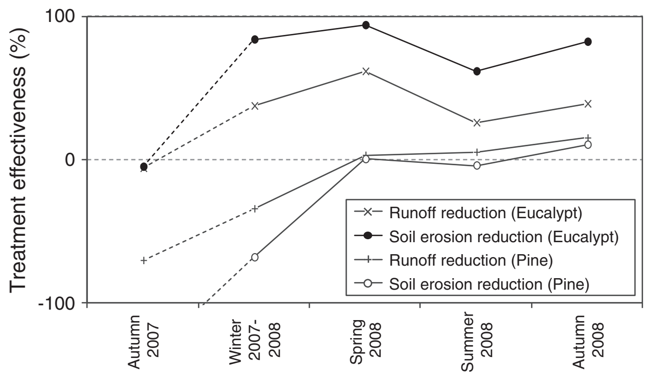

Fig. 3. Seasonal patterns in the average effectiveness of mulching with eucalypt chopped bark at the eucalypt site and with eucalypt logging slash at the pine site in reducing runoff and sediment losses.

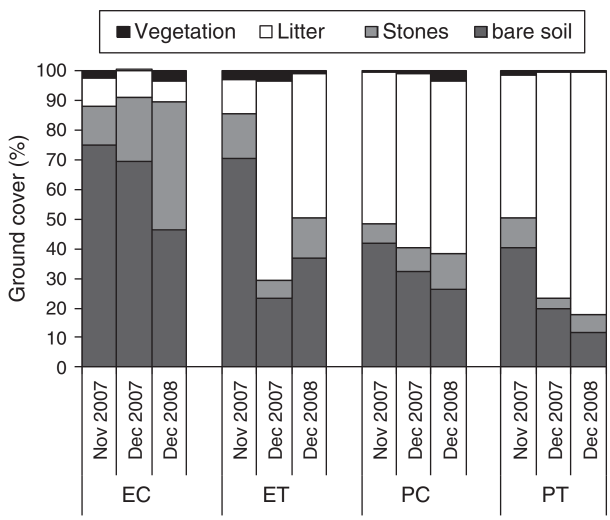

Fig. 4. Mean cover of the four cover categories at the control and treated plots at the eucalypt and pine study sites (EC, ET, PC and PT, respectively) immediately before and after the mulching (November and December 2007) and at the end of this study (December 2008).

mention is perhaps the very limited recovery of the vegetation, even by December 2008.

Soil moisture content varied markedly in the course of this study but revealed straightforward temporal patterns. As shown for the PR2-probe values (Fig. 5), the mean moisture content tended to: (i) increase—more or less gradually—from minimum values during autumn 2007 to maximum values during spring 2008; (ii) decrease again during the summer of 2008, albeit to higher values than at the start of the measurement period; (iii) attain higher values towards the end of the study than one year earlier. The PR-probe data also suggested that mulching significantly increased soil moisture content at the eucalypt site (repeated measures ANOVA: $n=32$, $p<0.05$), with the overall mean value being 15 and 20% vol. for the control and treated plots, respectively. In spite of the above-mentioned technical problems with the PR-tubes in the pine control plots, such a treatment effect could also be inferred for the pine site. Namely, the mean PR-probe values did not differ significantly between the treated pine and the treated eucalypt plots (repeated measures ANOVA: $n=32$, $p=0.3$), on the one hand, and on the other, the mean ML2-sensor values—measured in untreated slope parts—did not differ significantly between the pine and eucalypt site (pair-wise Student *t*-test: $n=29$, $p=0.06$). Although the two sensors gave distinct results in terms of absolute values, they did produce broadly similar temporal patterns, as evidenced by the strong relationship between PR-probe and ML-sensor on the untreated conditions at the eucalypt site (Pearson correlation coefficient: 0.70, $n=29$, $p<0.05$).

The frequency of extreme repellency (%FR) revealed more irregular temporal patterns than soil moisture (Fig. 5). Even so, both study sites revealed a broad tendency at their untreated slope parts for %FR to: (i) increase from October 2007 to maximum values in December 2007; (ii) decrease subsequently to minimum values during spring 2008; (iii) again increase towards the summer of 2008, most notably so at the pine site. The actual %FR values, however, tended to be noticeably lower at the pine compared to the eucalypt site. Extreme repellency was also found to be significantly less frequent at the pine site than at the eucalypt site during the winter of 2007/2008 (pair-wise Student *t*-test: 42 vs. 81%, $n=10$, $p<0.05$) as well as during the spring of 2008 (pair-wise Student *t*-test: 18 vs. 49%, $n=7$, $p<0.05$).

3.4. Key factors explaining runoff and erosion

The hydrological and erosion response of the 12 plots throughout the treatment period could be clearly explained the two rainfall and the four ground cover variables included in the forward selection procedure (Table 3). Almost 60% of the variation in (square-root transformed) runoff could be accounted for by four of the variables, whilst three variables sufficed to explain 70% of the variation in (fourth-root transformed) sediment losses. In both instances, the principal covariate concerned rainfall; however, it corresponded to rainfall intensity in the case of sediment losses as opposed to rainfall total in the case of runoff. Amongst the cover-related variables, litter cover was the most important factor, explaining roughly a third less variance than rainfall total and rainfall intensity, respectively for runoff and sediment losses models.

The regression results for the eucalypt site alone were similar to those for both sites together. This equally applied for the treatment

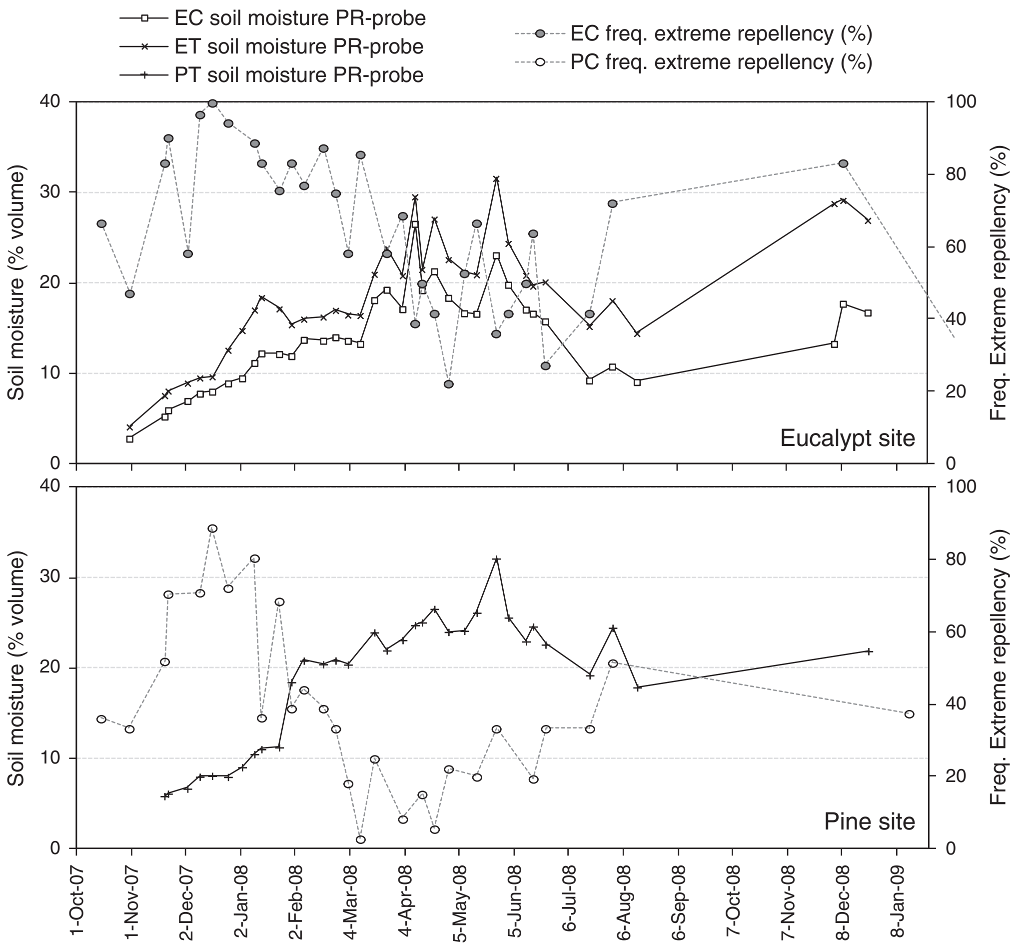

Fig. 5. Mean values of soil moisture content (0–10 cm depth; PR2-probe) and frequency of extreme soil water repellency (0–15 cm depth) for the individual measurement periods at the two study sites. Moisture data were available for the control and treated plots at the eucalypt site (EC and ET) and for control plots at the pine site (PT), whereas the repellency data were limited to the untreated conditions at both sites (EC and PC).

Table 3
Multiple regression models of runoff and sediment losses for various combinations of erosion plots (treated and untreated plots at the two study sites together and separately, and untreated plots at both study sites), measurement periods (32, 27 and 20 read-outs) and sets of independent variables (6, 7 and 8 covariates).

Selected variable	Runoff (mm)			Sediment losses (g m^{-2})		
	Parameter estimate	Variable name	Partial r^2	Parameter estimate	Variable name	Partial r^2
Global model: all 4 pine and all 8 eucalypt plots						
Complete dataset: 32 read-outs						
Covariates: 6—rainfall total (P_tot) and intensity (I_30), covers of bare soil, stones, litter and vegetation						
Intercept	3.39			1.04		
1st var.	0.02	P_tot	0.33	0.04	I_30	0.37
2nd var.	−0.04	Litter	0.20	−0.01	Litter	0.28
3rd var.	0.05	I_30	0.05	0.00	P_tot	0.05
4st var.	−0.03	Stones	0.01			
Cum. r^2			0.59			0.70
Eucalypt model: all 8 eucalypt plots						
Complete dataset: 32 read-outs						
Covariates: 6—rainfall total (P_tot) and intensity (I_30), covers of bare soil, stones, litter and vegetation						
Intercept	3.02			0.92		
1st var.	0.03	P_tot	0.47	0.04	I_30	0.47
2nd var.	−0.04	Litter	0.09	−0.01	Litter	0.21
3rd var.	0.05	I_30	0.04	0.01	P_tot	0.08
4th var.	−0.03	Stones	0.01			
Cum. r^2			0.61			0.76
Pine model: all 4 pine plots						
Complete dataset: 32 read-outs						
Covariates: 6—rainfall total (P_tot) and intensity (I_30), covers of bare soil, stones, litter and vegetation						
Intercept	0.39			0.50		
1st var.	0.01	I_30	0.36	0.03	I_30	0.42
2nd var.	0.06	P_tot	0.04			
Cum. r^2			0.41			0.42
Eucalypt model: all 8 eucalypt plots						
Limited dataset: 27 read-outs						
Covariates: 7—rainfall total (P_tot) and intensity (I_30), covers of bare soil, stones, litter and vegetation, and soil moisture (PR2-probe).						
Intercept	2.52			1.05		
1st var.	0.03	P_tot	0.51	0.04	I_30	0.48
2nd var.	−0.07	Moisture	0.10	−0.01	Litter	0.24
3rd var.	−0.02	Litter	0.04	0.01	P_tot	0.04
4th var.	0.07	I_30	0.02	−0.01	Moisture	0.01
5th var.	0.03	Veget.	0.02			
Cum. r^2			0.68			0.77
Untreated plots model: all 6 untreated plots						
Partial dataset: 20 read-outs						
Covariates: 8—rainfall total (P_tot) and intensity (I_30), covers of bare soil, stones, litter and vegetation, soil moisture (ML2-sensor) and frequency of extreme repellency (						
Intercept	0.09			0.93		
1st var.	0.03	P_tot	0.44	0.01	P_tot	0.55
2nd var.	0.03	Repellency	0.24	−0.01	Litter	0.21
3rd var.	−0.02	Litter	0.03	0.03	I_30	0.02
Cum. r^2			0.73			0.79

period as a whole as well as for the "limited" data set. Even so, the removal of the four pine plots increased somewhat the importance of the principal, rainfall-related covariate, in absolute terms but especially compared to the subsequent covariates (explaining at least twice as much variance). The relationship of sediment losses to the principal covariate—rainfall intensity—was shown in Fig. 2. Furthermore, litter cover was substituted by soil moisture as the second most important factor explaining runoff at the eucalypt site. This was not the case, however, for the site's sediment losses.

The regression results for the pine site alone differed in two important aspects from those for the eucalypt site (Table 3). First, rainfall intensity was the principal factor explaining not just sediment losses (Fig. 2) but also runoff. Second, litter cover did not explain a significant fraction of the variance in either runoff or sediment losses (and neither did any of the other cover categories). This was in line with the above-mentioned finding that there was no significant treatment effect at the pine site.

Soil water repellency and, in particular, the frequency of extreme repellency was found to have a significant effect on runoff at the two study sites but not on sediment losses (Table 3). Nonetheless, the role of extreme repellency of enhancing overland flow generation was clearly secondary compared to that of rainfall total, explaining roughly 50% less variance. Sediment losses varied significantly not just with rainfall total but also with litter cover, notwithstanding the fact that only the untreated plots were included in the analysis.

4. Discussion

The present findings coincided in many aspects (including in terms of plot design) with the results of Shakesby et al. (1996). This was especially true for the post-treatment periods of both studies, differing much less in rainfall amounts than the pre-treatment periods (1546 vs. 1470 mm as opposed to 138 vs. 645 mm). As far as the control plots were concerned, key points of agreement were: (i) the overall runoff coefficients of the eucalypt plots (30 vs. 20%); (ii) the overall sediment losses of the eucalypt plots (5.4 vs. 4.9 Mg ha^{-1}); (iii) the specific sediment loss rates of the eucalypt plots (0.35 vs. 0.33 g m^{-2} mm^{-1} rainfall); (iv) the markedly lower sediment losses of the pine compared to the eucalypt plots (amounting to only 6 vs. 16% of the eucalypt plots). Shakesby et al. (1996) suggested various factors that could contribute to the contrast in sediment losses between their pine and eucalypt plots, of which especially the presence of a pine needle "carpet" would seem relevant in the present context. Pannkuk and Robichaud (2003) equally found needle cast to be effective in reducing post-fire erosion rates. The regression results of the present study also supported that, even in the case of the control plots, litter cover played a significant role in reducing sediment losses. Worth nothing in this respect was that the mean litter cover at the untreated pine plots was approximately 60%, i.e. a commonly accepted threshold for mulch cover to be effective (Robichaud et al., 2000).

As far as the effectiveness of mulching with forest residues was concerned, the results of this study and those of Shakesby et al. (1996) coincided in two aspects: (i) a major decrease in overall sediment losses at the eucalypt site (with 86 vs. 91%); (ii) the lack of such an obvious reduction at the pine site, with sediment losses actually being higher at the control than at the treated plots (16 vs. 50%). Shakesby et al. (1996), however, did not assess treatment effectiveness in the same way as was done here (or in the other post-fire treatment studies listed in Table 4), comparing pre- to post-treatment values instead of the values of treated and untreated plots. Furthermore, Shakesby et al. (1996) opted for testing various mulch application rates at single plots rather than for testing one single rate at various replicate plots. Also in the case of the present study, the limited number of replicate plots implied special caution in interpreting the effectiveness figures for the pine site in particular. The suggestion of an erosion-enhancing effect of mulching at the pine site might well be due to the comparatively low runoff and sediment losses of the untreated pine plots, on the one hand, and, on the other, a marked variability amongst the plots in their hydrological and erosion response, implying a need for more replicate plots.

Even the untreated eucalypt plots did not produce excessive sediment losses during the first 1.5 year after wildfire when compared to the figures reported by some of the other studies on post-fire erosion treatment listed in Table 4. This fitted in well with the well-established tendency for erosion rates to be low in Mediterranean regions, in particular in cases—like the present one and those of Badía and Martí (2000) and Bautista et al. (1996)—where shallow soils and elevated surface stone cover bear witness to a long history of land use (Shakesby, 2011). Nonetheless, the large fraction of organic

Table 4
Compilation of field studies into the effectiveness of mulching-based treatments in reducing post-fire runoff and erosion. The meaning of the abbreviations are as follows: C, control; Effect., effectiveness; Euc., Eucalypt; GT, gerlach trap; LEB, log erosion barrier; mod., moderate; –, not reported; p., pine; PAM, polyacrilamide; plant., plantation; RS, rainfall simulation; sev., severity; SF, silt fence; PW, paired watershed; T, treated.

Treatment type (Mg/ha^{-1}, % cover)	Location, forest type	Fire sev.	Slope (%)	Method/ plot size (m^2)	n of plots		Study period	Annual rainfall	Total ground cover (%)		Runoff/rainfall (%)			Soil erosion (Mg ha^{-1})			Reference
					C	T	year, month	mm yr^{-1}	C	T	C	T	Effect. (%)	C	T	Effect. (%)	
Forest residue mulch																	
Chopped bark (8.7; 67)	C Portugal, Euc. plant.	Mod.	56	GT/16	4	4	yr0	1546	31	77	30	17	41	5.4	0.7	86	This study
Logging slash (17.5; 76)	C Portugal, P. plant.	Low	53	GT/16	2	2	yr0	1546	68	80	6	7	−10	0.3	0.4	−16	
Euc. Logging (46; 89)	C Portugal, Euc. plant.	Mod.	44	GT/16	2	1	yr2	1471	48	95	20	19	3	4.9	0.4	91	Shakesby et al. (1996)
Pine logging (18; 8)	C Portugal, P. plant.	Low	44	GT/16	2	2	yr3	2027	76	78	22	16	28	0.8	1.2	−50	
Wood chip mulch																	
Wood chip (4; 45)	NW Spain, shrub	High	40	SF/500	4	4	yr0	1520	19	56	–	–	–	35.0	33.0	6	Fernandez et al. (2011)
Wood chip (17; 70)	W Korea, Japanese p.	Mod.	51	GT/30	3	3	yr3	1115	43	80	19	11	42	7.6	3.8	51	Kim et al. (2008)
Wood chip (–; 70)	AR USA, Ponderosa p.	High	27	SF/4100	1	1	mth3	487	42	86	–	–	–	65.6	15.9	76	Riechers et al. (2008)
Straw mulch																	
Straw (2.5; 80)	NW Spain, shrub	High	40	SF/500	4	4	yr0	1520	19	84	–	–	–	35.0	12.0	66	Fernandez et al. (2011)
Straw + seeds (1; 53)	NE Spain, semi-arid shrub	Mod.	45	GT/8	4	4	yr1	268	38	99	–	–	–	2.6	0.4	83	Badía and Martí (2000)
		Mod.	45	GT/8	4	4	yr2	268	47	69	–	–	–	3.5	1.4	59	
Straw + seeds (1; 27)	NE Spain, semi-arid shrub	Mod.	45	GT/8	4	4	yr1	268	70	100	–	–	–	1.0	0.4	59	
		Mod.	45	GT/8	4	4	yr2	268	73	85	–	–	–	2.0	0.7	64	
Straw (2; 42)	E Spain, semi-arid p.	Mod.	42	GT/16	3	3	yr1	293	67	89	5	0	91	1.1	0.1	89	Bautista et al. (1996)
Straw (2.2; 78)	CO USA, Ponderosa p.	High	29	SF/16,000	8	3	yr0	198	33	74	–	–	–	6.2	8.8	−42	Wagenbrenner et al. (2006)
		High	29	SF/16,000	12	4	yr1	198	50	75	–	–	–	9.5	0.5	95	
		High	29	SF/16,000	12	4	yr2	198	68	89	–	–	–	1.2	0.0	98	
		High	29	SF/16,000	12	4	yr3	198	88	89	–	–	–	0.7	0.0	100	
Straw (2.2; 100)	MO USA, spruce-fir p.	High	15	RS/0.5	10	10	yr1	480	1	100	47	36	23	7.2	1.0	86	Groen and Woods (2008)
		High	15	RS/0.5	4	3	yr2	480	38	34	27	27	0	4.2	2.2	48	
Straw + seeds (2.2; 94)	CO USA, Ponderosa p.	High	22	SF/2830	4	4	yr1	402	32	55	–	–	–	13.2	0.7	95	Roughs, (2007) (unp.)
		High	22	SF/2830	4	4	yr2	402	58	72	–	–	–	11.0	2.5	77	
Straw rice (4.5; –)	AR USA, Ponderosa p.	High	27	SF/4100	1	1	mth3	487	42	88	–	–	–	48.4	9.1	81	Riechers et al. (2008)
Straw + seeds	NM USA	–	24	SF/25	6	6	yr0	52	–	–	–	–	–	8.3	2.5	70	Dean 2001 (unp.)
		–	24	SF/25	6	6	yr1	156	–	–	–	–	–	12.6	0.7	95	
Hydromulch											–	–	–				
Aerial (2.4; 94)	CO USA, Ponderosa p.	High	22	SF/2830	4	4	yr1	402	32	56	–	–	–	7.2	0.4	94	Roughs,(2007) (unp.)
		High	22	SF/2830	4	4	yr2	402	58	57	–	–	–	4.5	2.3	49	
Hand (2.4; 88)	CO USA, Ponderosa p.	High	22	SF/2830	4	4	yr1	402	32	54	–	–	–	10.2	8.5	17	
		High	22	SF/2830	4	4	yr2	402	58	53	–	–	–	8.5	6.9	19	
Aerial (−; 50)	CA USA, Chaparral	High	23	PW/55,000	1	1	yr0	415	–	50	–	–	–	15.0	21.0	−40	Wohlgemut et al. (2006)
Aerial (−; 100)		High	23	PW/55,000	1	1	yr0	415	–	100	–	–	–	15.0	7.0	53	
PAM Pellets	AR USA, Ponderosa p.	High	27	SF/4100	1	1	mth3	487	42	71	–	–	–	59.2	30.4	49	Riechers et al. (2008)
Barriers																	
Shrub barriers (10 m)	NW Spain, shrub	High	40	SF/500	4	4	yr0	1520	19	24	–	–	–	35.0	30.0	14	Fernandez et al. (2011)
LEB (2–2 m)	W Korea, Japanese p.	Mod.	51	GT/30	3	3	yr3	1115	43	–	19	18	7	7.6	7.5	2	Kim et al. (2008)
LEB + straw + seeds	NM USA	–	24	SF/25	6	6	yr0	52	–	–	–	–	–	8.3	1.9	77	Dean (2001) (unp.)
		–	24	SF/25	6	6	yr1	156	–	–	–	–	–	12.6	0.5	96	

matter observed in the sediment losses should be noted, not only for the implications for medium- to long-term land-use sustainability (e.g. Ferreira et al., 2008; Malvar et al., 2011; Thomas et al., 1999) but also for off-site pollution with pyrolitic toxic organic compounds (Vila-Escalé et al., 2007).

In comparison with other field studies that tested the effectiveness of wood chips mulches (Table 4), the eucalypt chopped bark was highly effective. Riechers et al. (2008) and Kim et al. (2008) reported substantial reductions of 76 and 51% respectively, while Fernández et al. (2011) found that wood chips decreased erosion by a mere 6%. These discrepancies can be due not only to the differences in the application rates, but also, as noted by the last two studies, to the fact that the 5–2 cm long chips pieces floated and were removed along with the sediments. This did not occur with the 10–15 cm long fibres of the chopped bark mulch. In fact, the mulching effectiveness at the eucalypt site can be compared more favourably with the range of values compiled for straw mulch in Table 4 (48–100%).

Arguably, the present results justified the decision to measure runoff and erosion with a high temporal resolution to compensate a possible lack of replicate plots. The repeated measures experimental design allowed valuable statistical inferences on treatment effectiveness as well as on the role therein of selected explanatory variables. An important insight was that even in the case of the eucalypt site the effectiveness of mulching was not time-invariant, being statistically significant only for the larger runoff and erosion events. This reflected the presence of thresholds, below which runoff amounts and sediment losses were too low for the mulching effect to prevail over the inherent variability in the plots' runoff generation and sediment transport processes. Litter cover played a more important role in sediment losses than runoff amounts. This coincided with the effects of mulching described by Smets et al. (2008), decreasing runoff generation by increasing surface storage as well as soil moisture content, on the one hand, and, on the other, decreasing sediment transport by decreasing splash erosion (sediment availability) as well as by increasing resistance to flow (transport capacity). Visual inspection of the treated and untreated eucalypt plots indeed suggested that mulching not only decreased splash erosion (pedestal formation) but also enhanced deposition of ashes and fines. From the few prior studies that assessed mulching effects in terms of both overland flow and erosion, Groen and Woods (2008) and Shakesby et al. (1996: eucalypt site) found a clearly greater impact on sediment losses than runoff. Bautista et al. (1996) and Kim et al. (2008), on the other hand, reported comparable reductions in runoff and erosion, notwithstanding the fact that the effectiveness varied greatly between these studies (42 to 91%).

The role of litter cover, whilst significant, was secondary compared to that of rainfall. With one exception, both rainfall total and rainfall intensity explained significant fractions of the variations in runoff and sediment losses, as was also observed by Bautista et al. (1996). The relative importance of the two rainfall variables, however, tended to differ for runoff and erosion, with rainfall total explaining better runoff amounts and rainfall intensity explaining better sediment losses. The former agreed with the findings of Kim et al. (2008), whereas the latter was in accordance with Wagenbrenner et al. (2006) but not with Fernández et al. (2011). This discrepancy could be due to differences in rainfall regime. The rainfall intensities in the present study were in fact more similar to those in Wagenbrenner et al. (2006) than to those in Fernández et al. (2011), notwithstanding the fact that the former study was carried out in the Colorado Front Range, USA, and the latter in Galicia, north-east Spain.

Following rainfall, soil water repellency was the most important variable explaining overland flow generation but this was only assessed for the untreated conditions. The role of water repellency in enhancing overland flow has often been inferred for burnt as well as unburnt eucalypt stands in particular (e.g. Coelho et al., 2005; Ferreira et al., 2005a; Malvar et al., 2011; Sheridan et al., 2007). However, it has rarely been established in an unequivocal manner, especially due to the destructive nature of repellency measurements and the relationship of repellency with other potential explanatory variables (Shakesby and Doerr, 2006), except perhaps by Leighton-Boyce et al. (2007) using surfactants in rainfall simulation experiments. Even so, water repellency could have been of minor importance at the mulched plots, since mulching was found to increase the soil moisture content at the eucalypt plots.

5. Conclusions

The principal conclusions of this study into the short- to medium-term effects of mulching with forest residues on runoff generation and sediment losses in a recently burnt eucalypt as well as maritime pine plantation in north-central Portugal were:

- whilst sediment losses at the untreated eucalypt plots were not excessively high for post-fire conditions worldwide, those at the untreated pine plots were low even by Mediterranean standards;
- mulching with eucalypt chopped bark was, on average, highly effective at the eucalypt site, with an increase in litter cover from 10 to 70% resulting in a decrease in 45% of runoff amount and in 85% of sediment losses;
- the effect of mulching at the eucalypt site was statistically significant, albeit for noticeably more runoff than erosion events due to the latter's highly irregular nature, and coincided with the significant role that litter cover played in explaining runoff and especially sediment losses;
- mulching at the pine site did not result in less runoff and erosion at the treated plots compared to the untreated plots, probably due to the already elevated effectiveness of the "natural" mulching by needle cast from the scorched pine canopies in combination with a marked variability in hydrological and erosion response amongst the plots;
- rainfall total and intensity explained runoff and sediment losses markedly better than any of the other six variables included in this study, but, besides litter cover, also soil moisture and soil water repellency could explain a significant fraction of the variation in overland flow generation.

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CHAPTER 4

Polyacrylamide application versus forest residue mulching for reducing post-fire runoff and soil erosion

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Polyacrylamide application versus forest residue mulching for reducing post-fire runoff and soil erosion

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HIGHLIGHTS

- The effectiveness of two soil erosion control treatments was contrasted after a wildfire.
- Chopped bark mulch reduced runoff and soil erosion, whereas dry polyacrylamide did not.
- Rainfall amount and soil cover were key factors respectively for runoff and soil erosion.
- Fire intensity across the burnt slope also affected soil erosion and organic matter content on the eroded sediments.

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ABSTRACT

For several years now, forest fires have been known to increase overland flow and soil erosion. However, mitigation of these effects has been little studied, especially outside the USA. This study aimed to quantify the effectiveness of two so-called emergency treatments to reduce post-fire runoff and soil losses at the microplot scale in a eucalyptus plantation in north-central Portugal. The treatments involved the application of chopped eucalyptus bark mulch at a rate of 10–12 Mg ha^{-1}, and surface application of a dry, granular, anionic polyacrylamide (PAM) at a rate of 50 kg ha^{-1}. During the first year after a wildfire in 2010, 1419 mm of rainfall produced, on average, 785 mm of overland flow in the untreated plots and 8.4 Mg ha^{-1} of soil losses. Mulching reduced these two figures significantly, by an average 52 and 93%, respectively. In contrast, the PAM-treated plots did not differ from the control plots, despite slightly lower runoff but higher soil erosion figures. When compared to the control plots, mean key factors for runoff and soil erosion were different in the case of the mulched but not the PAM plots. Notably, the plots on the lower half of the slope registered bigger runoff and erosion figures than those on the upper half of the slope. This could be explained by differences in fire intensity and, ultimately, in pre-fire standing biomass.

1. Introduction

In the last few decades, wildfires have become a common and widespread phenomenon in Portugal (Pereira et al., 2005; Shakesby, 2011). One of the principal effects of wildfires is widely held to be a partial or total loss of vegetation and litter cover (e.g. Soto and Diaz-Fierros, 1997; Shakesby, 2011). The resulting reduction in both rainfall interception and plant transpiration enhances runoff generation as well as soil exposure to the direct impact of raindrops (Soto et al., 1998; Wagenbrenner et al., 2006; Ben-Hur et al., 2011; Fernández et al., 2011). Direct effects of wildfires due to soil heating, such as breakdown of aggregates and increased soil water repellency, are generally considered to be key factors in the strong and sometimes extreme hydrological and erosion responses of recently burnt areas (e.g. Coelho et al., 2004; Doerr et al., 2006; Ferreira et al., 2008; Keizer et al., 2008; Varela et al., 2010; Malvar et al., 2011). Fire-enhanced generation of runoff and the associated export of sediments, organic matter, nutrients and pollutants not only have negative consequences for on-site land-use sustainability, but also can endanger downstream aquatic and flood-zone habitats and associated human infrastructures (Shakesby and Doerr, 2006; Ferreira et al., 2008; Robichaud, 2009).

It is generally accepted that fire-enhanced erosion rates are maximal immediately after the wildfire (e.g. 35 Mg ha^{-1} during the first post-fire year in Fernández et al., 2011) and decrease with time to background levels at the end of the so-called window of disturbance (up to 10 years after the wildfire as reported in Swanson, 1981 and in Shakesby and Doerr, 2006). However, the intensity and extent of this period, which depends on fire severity and post-fire climate conditions,

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are still highly uncertain and difficult to quantify (Neary et al., 1999; Cerdà and Doerr, 2005; Cerdà and Lasanta, 2005; Robichaud, 2009).

A variety of measures have been identified that can effectively reduce post-fire soil erosion (e.g. Miles et al., 1989; MacDonald and Larsen, 2009; Robichaud et al., 2013). Arguably, the most widely accepted measure is mulching, i.e., the application of a cover of organic compounds on the soil surface to modify energy and water fluxes and to protect the soil from direct raindrop impact (Bautista et al., 2009). Mulching has been found to successfully control post-fire runoff and soil erosion in many field trials (e.g. Miles et al., 1989; Bautista et al., 1996; Wagenbrenner et al., 2006; Fernández et al., 2011; Prats et al., 2012). A mulch cover of 60% is widely considered the minimum threshold for a significant reduction in soil loss (Pannkuk and Robichaud, 2003; Cerdà and Doerr, 2008; Robichaud et al., 2010). In the case of straw mulch, this threshold cover is typically achieved by applying 2 Mg of straw per ha (Miles et al., 1989; Bautista et al., 1996; Badía and Martí, 2000; Wagenbrenner et al., 2006; Groen and Woods, 2008; Fernández et al., 2011), with costs ranging from 600 to 1200 USD ha^{-1} for aerial and manual application, respectively (Napper, 2006).

Although burnt areas are commonly mulched with straw, this has various disadvantages: high cost, potential introduction of non-native plants, and susceptibility to wind-scattering (Bautista et al., 2009). In recent years, there has been increasing interest in alternative mulch types derived from forest residues, using fibers of different shapes and sizes (Yanosek et al., 2006; Smets et al., 2008). In laboratory experiments, 6-cm long wood strands applied at rates of 4 to 8 Mg ha^{-1} were found to be highly effective, reducing erosion rates by 80% (Foltz and Copeland, 2009; Foltz and Dooley, 2003; Foltz and Wagenbrenner, 2010). In field trials, mulching with 10- to 15-cm long chopped eucalyptus bark fibers markedly reduced post-fire erosion during the first year after the fire (Prats et al., 2012), while mulching with wood chips did not (Fernández et al., 2011). The mulch employed by Prats et al. (2012) had the further advantages of being readily available in the study region (due to the widespread occurrence of eucalyptus plantations in north-central Portugal), not being susceptible to removal by wind, decaying more slowly than straw, and not introducing invasive weeds. The cost of applying the chopped bark mulch, however, differed little from that of applying straw, as the lower costs per Mg were offset by the higher application rates needed to achieve the 60% cover threshold.

A more recent measure to control post-fire erosion is the application of polyacrylamides (PAMs; Rough, 2007; Robichaud et al., 2010). PAMs refer to a family of flocculant agents, comprising a broad class of chemical compounds with different chain lengths, charge types and charge densities. Different PAM formulations have been developed to ensure effective binding with clay particles through direct ionic attractions or cation bridges (Theng, 1982; Vacher et al., 2003). The application of PAMs constitutes a remarkable soil- and water-management technique, due to their extremely low cost (~3 USD per kg), their safety, and their capacity to influence physicochemical processes (Sojka et al., 2007). During the last two decades, the use of PAMs has proven effective for erosion control in furrow irrigation in intensive agriculture (Ben-Hur, 2006; Sojka et al., 2007). Application rates as low as 1 to 50 kg ha^{-1} have been found to noticeably reduce soil losses from agricultural fields as well as from steep road embankments (Agassi and Ben-Hur, 1992; Ben-Hur, 2001; Ben-Hur and Keren, 1997; Ben-Hur and Letey, 1989; Lentz et al., 2002; Levy et al., 1991). The effectiveness of PAMs in reducing post-fire erosion, however, is poorly established. The few studies which have been carried out have produced inconsistent results. Davidson et al. (2009), Riechers et al. (2008) and Inbar (2011) found PAM to be effective, whereas Rough (2007) and Wohlgemuth and Robichaud (2007) did not.

The main objective of the present study was to evaluate the effectiveness of two erosion-mitigation techniques – mulching with forest residues (chopped bark) and surface application of a dry granular anionic PAM – during the first year after a wildfire in a eucalyptus plantation in north-central Portugal. The specific objectives were to: (i) assess the performance of both techniques at a high temporal resolution (monitoring every 1 or 2 weeks); (ii) determine the spatial variation in overland-flow generation and soil losses from the base to the top of a 40-m long slope; and (iii) determine the key factors explaining overland flow and soil losses for the treatments, together and separately.

2. Material and methods

2.1. Study area

The study area was located near the Ermida hamlet in the Sever do Vouga municipality of north-central Portugal. The area was affected by a wildfire that consumed 295 ha between 26 and 28 July 2010 (AFN, Autoridade Florestal Nacional, 2012). The burnt area not only consisted mainly of eucalyptus (*Eucalyptus globulus* Labill.) plantations, but also included some maritime pine (*Pinus pinaster* Ait.) plantations and a stand of cork oak (*Quercus suber* L.). The eucalyptus trees in the region are typically planted as monocultures for paper pulp production, and harvested every 7–14 years. After logging, the eucalyptus trees are left to regrow from the stumps two or three times, after which a new plantation cycle is begun (Ferreira et al., 1997; Leighton-Boyce et al., 2005; Prats et al., 2012).

The climate of the study area can be classified as humid mesothermal (Csb in the Köppen classification), with moderately dry but extended summers (DRA-Centro, Direcção Regional do Ambiente do Centro, 1998) when the bulk of the wildfires occurs. The mean annual temperature at the nearest weather station of "Castelo Burgães" (40°51′16″N, 8°22′55″W, 306 m a.s.l.; 1990–2010; SNIRH, Serviço Nacional de Informação dos Recursos Hídricos, 2011) was 14.9 °C, while mean monthly temperatures ranged from 9.0 °C in January to 21.1 °C in July. Annual rainfall at the nearest rainfall station of "Ribeiradio" (40°44′39″N, 8°18′05″W; 228 m a.s.l.; 1990–2010; SNIRH, Serviço Nacional de Informação dos Recursos Hídricos, 2011) varied between 960 and 2530 mm, with an average of 1609 mm.

The study area is situated in one of the region's major physiographic units, the Hespheric Massif. The area consists mainly of pre-Ordovician schists and graywackes, but includes Hercynian granites at several locations (Ferreira de Brum, 1978). Within the study area, a steep (25°) but short (40 m) slope with southwest aspect was selected for this study (40°44′05″N, 8°21′18″W, 200 m a.s.l.; Fig. 1). The eucalyptus trees in the study site had been cut just before the fire, as evidenced by the tree logs that were piled up at the base of the slope and were partially charred by the wildfire. Judging from the remaining tree stumps (with diameters of roughly 1 m), the stand had undergone three prior harvestings, and had originally been planted some three decades before the 2010 wildfire. The overall severity of the 2010 wildfire was estimated to be moderate, as inferred from the complete consumption of the logging slash residues, the understory vegetation and the litter layer, as well as from the prevalence of a 1- to 4-cm thick layer of black ash (Table 1). At the base of the slope, however, the presence of gray and white ashes suggested moderate to high severity.

2.2. Experimental setup

At the end of August 2010, before any significant rainfall events (Fig. 2), the study site was instrumented with two rainfall gauges (one tipping-bucket gauge with a resolution of 0.2 mm and one storage gauge for validation purposes), and 12 square erosion plots of approximately 0.28 m^2 were established (Fig. 1). The 12 plots were organized into four sets (blocks) that were located at about equal distances from the base to the top of the slope (Table 1), while the three plots of each block were placed at 1- to 3-m distance from each other. The plot outlets were connected to tanks with a storage capacity of 30 l for overland-flow collection. The spatial variation in soil properties across the study slope was examined in February 2011 by excavating a soil profile in each block, measuring soil depth, and collecting two samples from

Fig. 1. Overview of the hillslope during the installation of the microplots.

each of two soil depths (0–5 and 5–10 cm). These 16 samples were analyzed in the laboratory for bulk density (Porta et al., 2003), granulometric composition (Guitian and Carballas, 1976) and organic matter content (Botelho da Costa, 2004) (Table 1). Whereas soil depth tended to decrease in the upslope direction, the other soil parameters revealed less straightforward spatial patterns. The upper 10 cm of the soils overlying pre-Ordovician schists of the Hespheric Massif (Pereira and FitzPatrick, 1995) had a sandy loam texture and high contents of stones (50–55%) and organic matter (7.9–11.6%).

Table 1
General description of the study site and details of the studied treatments. The ground cover corresponds to the average values of the three plots at each slope position, whereas the values of bulk density, stoniness, texture fraction and organic matter content correspond to the average values of the indicated samples collected at 0–5 cm and 5–10 cm depth.

	Block number			
	I	II	III	IV
General characteristics				
Position (m from base of slope)	11	18	27	36
Slope angle (degrees)	26	25	24	27
Projected plot area (m^2)	0.22	0.23	0.23	0.22
Ground cover immediately after wildfire (01 September 2010)				
Black ashes (%)	82	88	91	92
Gray and white ashes (%)	8	6	2	0
Stones (%)	5	5	4	2
Litter (%)	5	2	3	5
Soil characteristics				
Soil depth (cm)	74	43	35	35
Bulk density (g cm^{-3})	1.1	1.1	1.2	1.0
Stoniness (>2 mm, %)	53.1	55.3	54.8	50.9
Sand fraction (%)	62.7	66.9	69.6	58.8
Silt fraction (%)	20.5	18.2	16.7	22.7
Clay fraction (%)	16.7	14.8	13.6	18.5
Organic matter content (%)	10.9	11.6	7.9	11.1
Treatments				
PAM application rate (Mg ha^{-1})	0.05	0.05	0.05	0.05
Mulch application rate (Mg ha^{-1})	11.2	11.6	10.4	10.1
Mulch cover (%; on 04 October 2010)	86	89	80	78

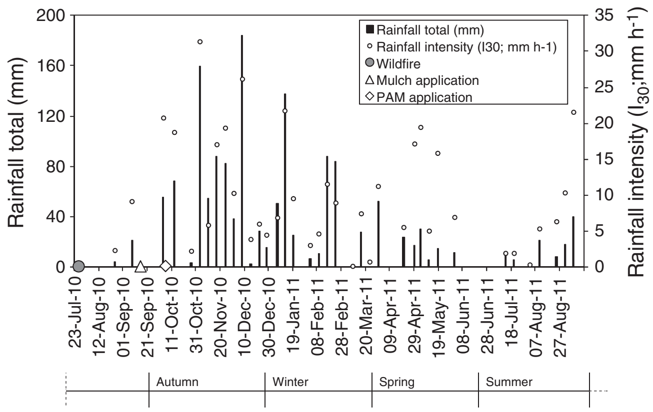

Fig. 2. Rainfall total and maximum intensity during 30 min (i30) of the individual, 1- to 2-weekly readouts during the first year after wildfire. Besides the occurrence of the wildfire, also the application dates of the two treatments (forest residue mulch and PAM) are indicated.

A randomized block design was employed to assess the effectiveness of the two erosion-mitigation techniques. The two treatments were randomly allocated to two of the three plots in each block, leaving the last plot untreated (control). The forest residue mulch consisted of chopped eucalyptus bark and was purchased from the Socasca S.A., at the standard market price of 30 € per Mg. The mulch was applied manually on 15 Sep 2010 at a rate of 10–12 Mg ha^{-1}, which provided 80–90% ground cover (Table 1). A dry granular anionic PAM with high molecular weight (Superfloc 110-c Series N/A-100) was chosen for this study, because of its effectiveness in prior studies (Chaudhari and Flanagan, 1998; Flanagan et al., 2002; Yu et al., 2003; Ajwa and Trout, 2006), including in a recently burnt area (Inbar, 2011). It was spread out manually over the soil surface on 4 October 2010 at a rate of 50 kg ha^{-1}. The delay in the PAM application relative to the mulching was due to difficulties in obtaining the Superfloc 110-c polymer. As a consequence of this delay, rainfall prior to the PAM application was considerably higher than that prior to the mulching (81 mm vs. 25 mm, respectively). Therefore, the present study does not include the initial post-fire period up until 4 October 2010.

2.3. Field data collection and laboratory analyses

From 1 September 2010 to 7 September 2011, the rainfall accumulated in the storage gauge and the overland flow collected in the tanks were measured at 1- to 2-week intervals, depending on the occurrence of rainfall. Whenever there was more than 250 ml of runoff in a tank, a sample was collected (in a 1.5-l bottle) and transported to the laboratory for analysis. In total, some 400 runoff samples were collected during 34 readouts. The sediment concentration of these samples was determined in the laboratory by filtration, using a paper filter with a pore diameter of 12 μm, followed by drying at 105 °C for 24 h. Subsequently, the organic matter content of the filtered and dried sediments was measured by loss-on-ignition method (550 °C for 4 h).

The ground cover of the 12 erosion plots was determined on six occasions during the study period, i.e., immediately before and after applying the treatments (on 1 Sep and 3 Nov 2010), and then at 2- to 4-month intervals until November 2011. The following five cover categories were recognized: bare soil, stones (including rock outcrop), litter (including the applied mulch), ash (including charred plant material), and vegetation. Ground cover was quantified by laying a square grid of 0.5 m × 0.5 m at a fixed position over the plots, and recording the cover category at the 100 points of intersection between the grid's 10 equidistant rows and 10 equidistant columns.

2.4. Data analysis

The SAS system (Littell et al., 1996, 2006) was used to carry out the following statistical analyses: (i) one-way ANOVA, to assess whether the three treatments (control, PAM, and mulching) resulted in significant differences among their overall values of runoff (specific), soil losses and organic matter content of the eroded sediments over the entire study period (4 October 2010–7 September 2011) as well as in the cover of the five cover categories immediately after the wildfire (September 2010) and 1 year later; (ii) two-way ANOVA, to determine the (combined) effects of the three treatments and of the plots' four positions across the slope on the overall values of runoff (specific), soil losses and organic matter content of the eroded sediments; (iii) two-way repeated-measures ANOVA, to assess the (combined) effects of the three treatments and the time-since-treatment on the 1- to 2-weekly values of runoff (specific), soil losses and organic matter content of the eroded sediments; (iv) post-hoc tests of least squares differences (LSDs) adjusted by the Tukey–Kramer method (Tukey, 1953; Kramer, 1956), to assess whether the plots treated with mulch and PAM produced significantly different overall or 1-/2-weekly values of runoff, (specific) soil losses and organic matter content compared to the untreated plots; and (v) multiple linear regression, using the REG stepwise forward selection procedure in combination with the collinearity test to select, among a set of 10 independent variables, those that explained a significant ($p \leq 0.05$) fraction of the variation in the 1- to 2-weekly values of runoff, (specific) soil losses and organic matter content and, at the same time, had a condition index below 30 (Belsley et al., 1980; Littell et al., 1996). The 10 independent variables included in the REG procedure consisted of two rainfall-related variables ("rain" — rainfall amount; "i30" — maximum rainfall intensity in 30 min), the five cover categories and three time-invariant variables ("depth" — soil depth, "position" — position of the plots across the slope, "angle" — slope angle of the individual plots).

In the case of the two-way repeated-measures ANOVAs, the assumption of normality of the residuals was rejected for the original values of runoff (mm), soil loss (g m^{-2}) and specific soil loss (g m^{-2} mm^{-1} runoff) (Kolmogorov–Smirnov test: $p < 0.05$). To remediate this, the runoff and (specific) soil loss data were log10 fourth root transformed, respectively, and the six readouts with the least rainfall (<6 mm) were eliminated from the data set. The resulting data sets were also used in the multiple linear regression analyses. The variance–covariance structure of the repeated-measures ANOVAs was modeled with the heterogeneous auto-regressive variance, because it gave the smallest values for the Akaike Information Criterion (AIC; Akaike, 1987) and the −2 restricted log likelihood (Littell et al., 2006).

3. Results

3.1. Overall rainfall, runoff and erosion values

Total rainfall during the entire study period from 1 September 2010 until 7 September 2011 amounted to 1500 mm, closely approximating the long-term mean annual rainfall at the nearest Ribeiradio station (1609 mm). From the 1481 mm of rain that fell during the post-treatment period (i.e. after 4 October 2010), more than half (55%) was, on average, converted to overland flow over the untreated plots (control treatment) and produced 848 g m^{-2} of soil loss (Table 2). This soil loss was accompanied by an even greater loss of organic matter, as the sediments eroded from the control plots had an average organic matter content of 61%. Mulching had a significant and prominent impact on runoff generation, but in particular on soil loss (one-way ANOVA: $p < 0.05$ and $p < 0.01$, respectively). The runoff in the mulched plots was, on average, 52% lower than in the control plots, whereas the associated soil losses were 93% lower. The effect of PAM, on the other hand, was less marked and not significant (one-way ANOVA: $p = 0.3$) and, at the same time, opposite for runoff and erosion, reducing the average runoff by 16% while increasing the average soil losses by 23%. Thus, the overland flow generated by the PAM plots transported, on average, 50% more soil per unit of runoff than the overland flow produced by the control plots (1.58 vs. 1.05 g m^{-2} mm^{-1} runoff), and this difference was statistically significant (one-way ANOVA: $p < 0.05$). The same was not applied to the organic matter losses, as they made up equivalent fractions of the sediments eroded from the PAM, mulched and control plots (51 vs. 61%).

Table 2
Average values of total runoff volumes, total and specific soil losses, and organic matter contents in the eroded materials for control (untreated), polyacrylamide (PAM) and mulched plots over the entire post-treatment period (4 October, 2010–7 September, 2011). PAM and mulch effectiveness exhibits positive and negative signs in order to highlight the enhancing or reducing effect of the treatment. Significant differences between the untreated and treated plots, according to one-way ANOVA, are in bold ($p < 0.05$) or underlined and bold ($p < 0.01$).

	Runoff	Soil losses		Organic matter content
	Volume (mm)	Total (g m^{-2})	Specific (g m^{-2} mm^{-1})	(% w/w)
Control	785	848	1.05	61
PAM	657	1047	1.58	51
Mulch	378	63	0.17	63
PAM effectiveness (%)	−16	+23	**+50**	−16
Mulch effectiveness (%)	**−52**	<u>**−93**</u>	<u>**−84**</u>	+3

Overall (specific) soil losses over the entire post-treatment period differed significantly among the three treatments as well as among the four slope positions (Table 3). In contrast, overall runoff volumes did not differ significantly among treatments or among slope positions. Overall organic matter contents in the eroded sediments also did not differ significantly among the treatments but they did among the slope positions. The specific contrasts of the treated (mulching/PAM) vs. control plots were in line with the above-reported one-way ANOVA results. Mulching resulted in reductions in overall (specific) soil losses and runoff that were highly ($p < 0.001$) and marginally ($p = 0.05$) significant, respectively. Applying PAM, on the other hand, only produced a significant change in specific soil losses ($p < 0.01$) and this corresponded to an increase rather than a reduction.

The significant role of slope position was more obvious for the overall soil losses compared to the specific soil losses, especially for the control and PAM plots compared to the mulched plots (Fig. 3). From the base to the top of the slope, overall soil losses of the control and PAM plots decreased from 1800 to 1300 g m^{-2}, respectively, to roughly 400 g m^{-2}. Albeit not significant, a similar trend of decreasing values in the upslope direction was also observed for the runoff volumes of the control and PAM plots in particular. In contrast, the organic matter contents in the eroded sediments revealed a clear tendency toward an increase in the upslope direction.

3.2. Temporal patterns in rainfall, runoff and erosion

During the study period from 1 September 2010 to 7 September 2011, rainfall was measured on a total of 34 occasions (Fig. 2). In three instances, rainfall exceeded 100 mm, twice during the autumn of 2010 (159 and 184 mm) and once during the winter of 2010/11 (138 mm). These highest rainfall totals coincided with the highest maximum rainfall intensities, with i30 values amounting to 31, 26, and 22 mm h^{-1}, respectively. The most extreme rainfall events in autumn 2010 produced the two principal peaks in runoff and soil losses in the control and PAM plots, but only in runoff in the mulched plots (Fig. 4).

The two-way repeated-measures ANOVAs of the 1- to 2-weekly runoff volumes and (specific) soil losses revealed significant effects for both factors – treatments and time-since-treatment – but also for their

Table 3
Two-way ANOVA of the effects of control (untreated), polyacrylamide (PAM), and mulch treatments and slope position on total runoff volumes, total and specific soil losses, and organic matter contents in eroded materials over the entire post-treatment period (4 October, 2010–7 September, 2011). Significant F-values and t-values – in the case of the specific contrasts between treated and untreated plots – are in bold ($p < 0.05$) or underlined and bold ($p < 0.01$). Abbreviation "DF num, den" are degrees of freedom for numerator and denominator.

Source of variation		DF num, den	Runoff	Soil losses		Organic matter content (% w/w)
			Volume (mm)	Total ($g\ m^{-2}$)	Specific ($g\ m^{-2}\ mm^{-1}$)	
Between effects	Treatment	2.6	2.72	**46.33**	**186.33**	1.96
	Slope position	1.6	2.70	**8.89**	**5.53**	**8.17**
	Treatment × slope	2.6	0.19	1.73	2.58	1.12
Specific contrasts	Control vs. PAM	6	0.67	−0.80	**−4.01**	1.54
	Control vs. mulch	6	2.27	**7.91**	**14.35**	−0.31

interaction (Table 4). Thus, the role of the treatments in overland flow generation and soil erosion was not unequivocal during the entire post-treatment period. Nonetheless, the specific contrasts of the mulched vs. control plots revealed significant differences in runoff as well as (specific) soil losses ($p < 0.01$). Furthermore, the interaction terms could be rendered insignificant by removing the readouts with the smallest rainfall amounts from the data set, while the individual factors continued to be significant. In the case of runoff, this could be achieved by eliminating the 11 readouts with less than 22 mm rainfall; in the case of (specific) soil losses, however, it required excluding all but 4 of the 28 readouts. The two-way repeated-measures ANOVA of the organic matter contents revealed a significant role of time-since-treatment but not of the treatments themselves. For all plots together, there was an overall decrease of 4% in the organic matter contents of the sediments eroded during the autumn of 2010 and those eroded during the summer of 2011. This decrease was most pronounced for the PAM plots (5.2%) and least pronounced for the control plots (2.6%).

For the individual readouts, LSDs between control and mulched plots were usually statistically significant, both in terms of runoff (23 readouts) and soil loss (27 readouts) (Fig. 4). In contrast, LSDs between control and PAM plots were only significant on one occasion for runoff and soil losses. Fig. 4 illustrates the importance of the interaction term between treatments and time-since-treatment for the soil losses. In the first two readouts after the wildfire, the PAM plots produced noticeably more erosion than the control plots, whereas the opposite was true in early spring 2011 (Fig. 4).

The reduction in average runoff and soil losses in the treated (PAM and mulching) vs. untreated plots was plotted (as percentage of the untreated plot values) against the weekly maximum rainfall intensities (i30; Fig. 5). The reduction in runoff decreased in a clear and similar manner with increasing i30 for both treatments, although mulching was consistently more effective than PAM at reducing runoff. The effectiveness of PAM in reducing soil loss also appeared to diminish with increasing i30, although variability between readouts was more pronounced than for runoff. In contrast, the effectiveness of mulching in decreasing soil losses was basically unaffected by i30.

3.3. Statistical modeling of the temporal runoff and erosion patterns

The hydrological and erosion response of all 12 treated and untreated plots together could be explained by the 10 independent variables included in the forward selection procedure (Table 5: 70–80% of the total variance). This was clearly less valid for the organic matter contents in the eroded sediments (40% of the total variance being explained). In the case of the (log-transformed) runoff volumes, 66% of the variation could be explained by a single variable—rainfall amount. In the case of the (fourth-root-transformed) soil losses, on the other hand, 61% of the variation was explained by two factors of similar importance—maximum rainfall intensity (i30) and litter cover. Runoff, soil losses and organic matter contents were plotted against the principal explanatory variables (Fig. 6).

The multiple regression models explaining runoff were basically the same for each of the three treatments separately as well as for the 12 plots together, showing a consistent prevalence of the role of rainfall amount (Table 5). The treatment-specific models explaining soil losses were also similar for the three treatments. However, they differed markedly from the model for all 12 plots together, as litter cover was no longer a key explanatory variable. In a similar fashion, bare soil cover was no longer an important factor in explaining the organic matter contents in the individual treatments. The separate models explaining organic matter contents lacked a clear consistency, including the range of the explained variation from 27% in the case of the mulched plots to roughly twice as much (56%) in the case of the PAM plots.

The important role of litter cover in the erosion model for all 12 plots together reflected a conspicuous difference in the mulched vs. PAM and control plots. Even at the end of this study, in September 2011, this difference was, on average, about 65% (Fig. 7). Aside from litter cover, the concurrent stone, ash and bare soil covers differed significantly among the treatments (one-way ANOVA: $p < 0.01$), being, for obvious reasons, lower in the mulched vs. PAM and control plots.

4. Discussion

4.1. Post-fire erosion risk in recently burnt eucalyptus plantations

The soil losses in the control plots plainly justified the application of emergency measures immediately after the wildfire. The roughly 8 $Mg\ ha^{-1}\ yr^{-1}$ clearly exceeded the range of values compiled by Shakesby (2011) for recently burnt Mediterranean ecosystems (0.3–3 $Mg\ ha^{-1}\ yr^{-1}$), as well as the threshold of 1 $Mg\ ha^{-1}\ yr^{-1}$ for tolerable soil loss proposed by Verheijen et al. (2009). The present figures were also somewhat higher than those reported by Shakesby et al. (1996): 4.9 $Mg\ ha^{-1}\ yr^{-1}$, and Prats et al. (2012): 5.4 $Mg\ ha^{-1}\ yr^{-1}$, for recently burnt eucalyptus stands in north-central Portugal. An explanation for these latter differences could be a scaling effect (e.g., Boix-Fayos et al., 2007; Ferreira et al., 2008), since Shakesby et al. (1996) and Prats et al. (2012) employed much larger plots than those in the present study (16 vs. 0.25 m^2). However, recent studies (Cerdà et al., 2013; Garcia-Estringana et al., 2013) showed that the scaling effect would influence first and foremost the generation of overland flow, but not so clearly the soil erosion. Furthermore, the specific soil losses in the control plots of the present study (1.05 $g\ m^{-2}\ mm^{-1}$ runoff) were lower than those in Prats et al. (2012) and especially Shakesby et al. (1996) (1.15 and 1.68 $g\ m^{-2}\ mm^{-1}$ runoff, respectively). It is worth stressing that, aside from mineral soil, organic matter was also eroded in large quantities from the control plots, on average some 5 $Mg\ ha^{-1}\ yr^{-1}$. The implications of these organic matter losses are not restricted to on-site soil fertility (e.g., Malvar et al., 2011; Shakesby, 2011), but extend to off-site impacts of ash-loaded runoff, which has been recently shown to induce eco-toxicological effects (Campos et al., 2012).

4.2. Effectiveness of mulching

The present results on mulching's overall effectiveness agreed well with those of the two previous studies that tested the effectiveness of

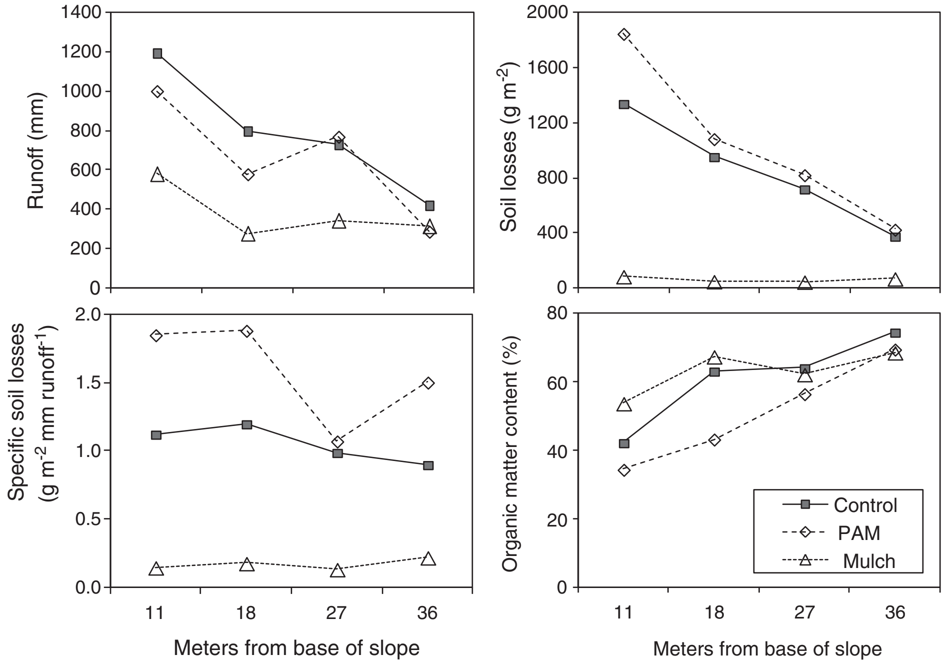

Fig. 3. Overall values of runoff, (specific) soil losses and organic matter content of the eroded sediments for the individual microplots over the entire post-treatment period (4 October 2010–7 September 2011).

forest residue mulching in recently burnt eucalyptus stands (Shakesby et al., 1996; Prats et al., 2012). All three studies found an overall reduction in soil losses on the order of 90% (Fig. 8: studies 1, 2 and 3). Moreover, the overall reduction in runoff was similar in this study (52%) and in Prats et al. (2012; 41%), whereas it was markedly lower in Shakesby et al. (1996; 3%). The slightly greater reduction in runoff found here compared to Prats et al. (2012) could be due to the slight difference in

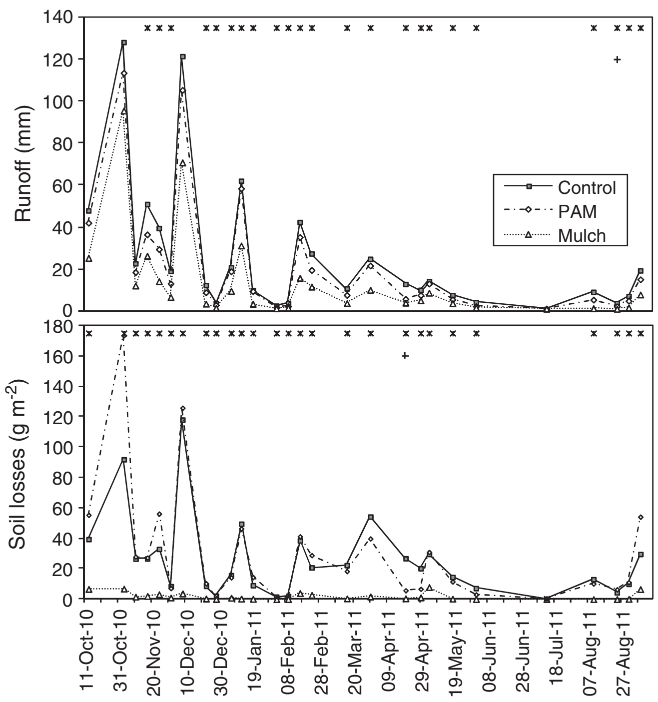

Fig. 4. Temporal patterns in average runoff and soil loss values for the three treatments (control, PAM and mulch) during the post-treatment period (4 October 2010–7 September 2011). Significant LSD's ($p < 0.05$) between the control plots and the mulched and PAM plots for the individual readouts are marked with an asterisk (*) and a cross (+), respectively.

mulch-application rates (10–12 vs. 9 Mg ha^{-1}), possibly combined with the aforementioned scaling effect. The major difference in runoff reduction compared to Shakesby et al. (1996) is more difficult to explain, but could involve methodological aspects. The mulch in Shakesby et al. (1996) was applied at a much higher rate (46 Mg ha^{-1}) but was composed of eucalyptus residues that came directly from logging, i.e. they were not chopped like the residues applied in this study and by Prats et al. (2012). As a result, the mulch in Shakesby et al. (1996) might have acted principally as a low-vegetation cover rather than as a litter layer, intercepting rainfall but not slowing down overland flow or enhancing its (re-)infiltration.

Mulching with forest residue, as described in this study, seems to constitute a more effective post-fire treatment than mulching with wood chips (e.g., Kim et al., 2008; Riechers et al., 2008; Fernández et al., 2011; Fig. 8: studies 4, 5 and 6). A key factor was probably the greater size of the fibers, promoting adherence to the soil surface. Riechers et al. (2008) found that an initial 80% cover of wood chips is drastically reduced as the chips float off under sufficient overland flow.

The temporal patterns of mulching effectiveness throughout this study also fit well with other studies with comparable data sets (Bautista et al., 1996; Badía and Martí, 2000; Prats et al., 2012). The differences in mulch type (straw or forest residue) and experimental design (especially monitoring intervals) notwithstanding, these three prior studies and the present one agreed in that: (i) mulch effectiveness was not unequivocal due to a significant interaction between treatment and time-since-treatment; (ii) mulch effectiveness was more often significant for large and intense compared to small and weak rainfall events; (iii) mulch effectiveness was greater in terms of reducing soil erosion compared to overland flow; (iv) soil erosion produced by mulched plots was less easily explained than soil erosion produced by untreated, control plots. Furthermore, in the case of the present study, the short monitoring intervals (1 to 2 weeks) highlighted the fact that runoff reduction by mulching is dependent on rainfall characteristics (intensity and amount), whereas soil erosion reduction was basically constant throughout the post-treatment period. The two last readouts with elevated maximum rainfall intensities (~20 mm h^{-1}) suggested a decrease in the mulch's effectiveness in reducing soil losses, which could be due to decomposition of the chopped bark mulch. Even so,

Table 4
Two-way repeated-measures ANOVA of the effects of treatment and time-since-treatment on the 1- to 2-weekly values of runoff, (specific) soil losses and organic matter contents during the post-treatment period (4 October 2010–7 September 2011: 28 readouts). Significant F-values and t-values – in the case of the specific contrasts between treated and untreated plots – are in bold ($p < 0.05$) or underlined and bold ($p < 0.01$). Abbreviation “DF num, den” are degrees of freedom for numerator and denominator.

Source of variation		DF num, den	Runoff (mm)	Soil losses		Organic matter content (% w/w)
				(g m^{-2})	(g m^{-2} mm^{-1})	
Within effects	Treatment	2.9	**15.45**	**73.40**	**68.52**	3.96
	Time	27,243	**209.52**	**38.07**	**6.57**	**2.82**
	Treatment × time	54,243	**2.98**	**2.79**	**2.37**	1.27
Specific contrasts	Control vs. PAM	9	1.77	−0.9	−2.03	2.2
	Control vs. mulch	9	**5.45**	**10.45**	**8.97**	−0.41

the mulch cover was found to decrease in a roughly linear fashion, by some 2% per month. These results are in close agreement with the value reported by Prats et al. (2012) for eucalyptus residue mulch, but markedly lower than the 4–5% found by Badía and Martí (2000) and Fernández et al. (2011) for straw mulch. This indicates a clear advantage of applying forest residue vs. straw mulch, especially when the window of disturbance is prolonged due to slow recovery of the spontaneous vegetation.

4.3. Effectiveness of PAM

As mentioned above, only a few field trials have assessed the effectiveness of PAM in reducing post-fire erosion, giving contradictory results. Comparisons are difficult, mainly due to the differences in PAM type and experimental design in each study (Fig. 8: studies 7 to 12). The greatest reduction was reported by Rough (2007; 80%), but this involved applying PAM mixed into an amended slurry. Riechers et al. (2008) found a 50% reduction in post-fire erosion, but they only measured the first few rainstorms after the fire. The authors applied PAM attached to dry pellets of compressed straw, so that the effect of PAM could not be separated from the effect of the 80–90% ground cover provided by the pellets. Similarly, Davidson et al. (2009) reported a 40% reduction in post-fire erosion by applying PAM attached to compressed paper pellets for a ground cover of 50%. Of the prior studies that also applied PAM in dry granular format, Inbar (2011) found

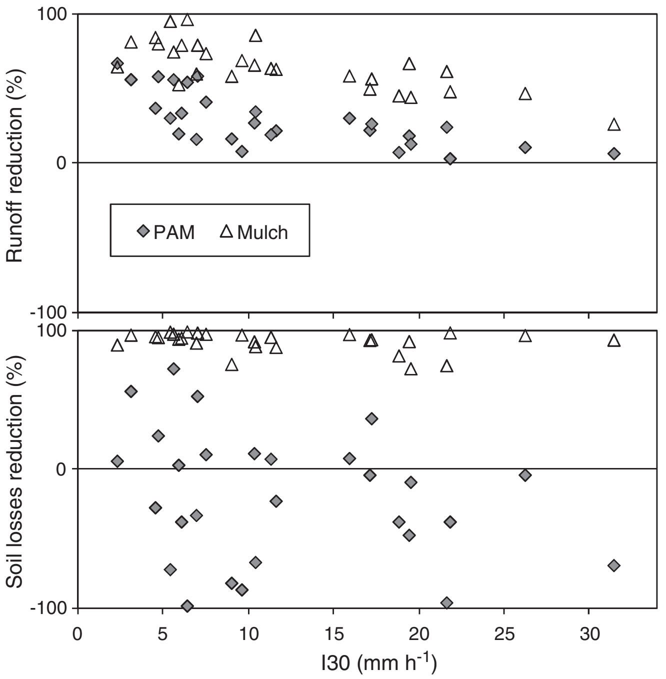

Fig. 5. Average reduction in runoff and soil losses at the mulched and PAM plots in relation to maximum rainfall intensity in 30 min (i30; mm h^{-1}) for the 28 individual readouts with >6 mm of rainfall during the first year after wildfire.

23% and 50% reductions in post-fire erosion at application rates of 25 and 55 kg ha^{-1}, respectively. However, whereas Inbar (2011) used the exact same type of PAM as we did, they removed the ashes before applying it, differing from all of the other studies referred to here. Rough (2007) and Wohlgemuth and Robichaud (2007) reported that applying 5.6 kg ha^{-1} of dry granular PAM does not reduce post-fire soil erosion.

The above-mentioned divergent findings on the effectiveness of PAM in reducing post-fire runoff and erosion could be the result of a number of factors, such as not only type of PAM, its application rate and method, but also soil type and texture. PAM is widely held to be most suitable for soils with high clay contents, high cation exchange capacities and divalent, exchangeable cations (Ben-Hur, 2001, 2006; Sojka et al., 2007). Nevertheless, selection of the most suitable PAM formulation for a specific soil is rather complex, since the many PAM formulations have distinct properties due to differences in molecular weight, charge type and charge density. Moreover, the selection of optimal application rate and method is not straightforward either, as clearly demonstrated by Theng (1982), McLaughlin and Brown (2007) and Inbar (2011). At present, the best option for applying PAM in recently burnt areas would appear to be in combination with paper/straw pellets; nevertheless, the added value of adding PAM to the pellets remains questionable, including in economic terms.

The mechanisms by which PAMs reduce post-fire soil erosion are not completely understood, but some aspects have become clear. The present results suggest that poor effectiveness of PAM in recently burnt areas could involve a combined effect of ashes and soil water repellency. PAM might preferentially bind the ashes instead of the soil (Rough, 2007), and both materials might then be removed after the first rainfall events by the repellency-enhanced overland flow (Wallace and Wallace, 1986). The study site exhibited strong to extreme soil water repellency during the initial post-fire period, as is common in recently burnt eucalyptus stands in north-central Portugal (Keizer et al., 2008; Malvar et al., 2011; Prats et al., 2012). A substantial reduction in the ash cover was also observed during the three first rainfall events after the PAM application.

4.4. Key factors in post-fire erosion with and without emergency treatments

In this study, litter cover – mainly composed of mulch – was slightly more important than rainfall total or intensity in explaining the differences in soil loss among the 12 plots. A crucial role for protective soil cover in post-fire erosion was also found by Pietraszek (2006), analyzing the evolution of the spontaneous ground cover in a large data set comprising 10 different wildfires of varying ages (0–10 post-fire years). In Pietraszek's (2006) case, bare soil cover explained more than 50% of the variation in erosion rates. As in this study, other multiple linear regression models have been carried out in the north-central Iberian Peninsula, with post-fire mulched and control (Prats et al., 2012), prescribed burnt and unburnt (Vega et al., 2005), and agriculture plowed and vegetated field (Nunes et al., 2011) plots. As in the present study, rainfall intensity was identified as the key factor for soil erosion. This was especially true for the “bare” plots in their experimental designs. For the “cover-protected” plot data sets (mulched, unburnt or

Table 5
Stepwise multiple linear regression models of the 1-to 2-weekly values of runoff, soil losses and organic matter contents of eroded sediments during the post-treatment period (4 October 2010–7 September 2011), for the three treatments together as well as separately. The full names of the variables are given in Section 2.4.

	All plots (n = 12)			Control plots (n = 4)			PAM plots (n = 4)			Mulched plots (n = 4)		
	Param. estimate	Variable name	Partial r^2	Param. estimate	Variable name	Partial r^2	Param. estimate	Variable name	Partial r^2	Param. estimate	Variable name	Partial r^2
Runoff (mm)												
Intercept	−0.01			0.93			0.53			0.03		
1st var	0.01	Rain	0.66	0.01	Rain	0.68	0.01	Rain	0.70	0.01	Rain	0.76
2nd var	0.01	Stones	0.07	−0.01	Position	0.07	0.02	Litter	0.05	0.02	i30	0.04
3rd var	0.02	i30	0.02	0.02	i30	0.02	0.02	i30	0.02	−0.02	Veg	0.03
4th var	0.00	Depth	0.01				−0.01	Position	0.02	0.06	Stones	0.02
Total r^2			0.77			0.77			0.79			0.85
Soil losses (g m^{-2})												
Intercept	1.75			1.76			1.93			0.47		
1st var	−0.01	Litter	0.32	0.04	i30	0.45	0.05	i30	0.46	0.04	i30	0.44
2nd var	0.04	i30	0.29	−0.02	Position	0.08	−0.03	Position	0.14	−0.03	Veg	0.04
3rd var	−0.03	Position	0.04	0.01	Rain	0.06	0.01	Rain	0.06			
4th var	0.00	Rain	0.03									
5th var	−0.03	Veg	0.01									
Total r^2			0.70			0.59			0.66			0.48
Organic matter content (% of sediments)												
Intercept	47.8			64.3			18.3			82.9		
1st var	−0.54	Bare	0.30	−1.55	Depth	0.40	1.40	Position	0.56	−1.76	Stones	0.18
2nd var	0.67	Position	0.11							−0.28	Depth	0.04
3rd var										−3.53	Bare	0.04
Total r^2			0.41			0.40			0.56			0.27

vegetated), the soil erosion models tended to be weaker, with restricted dependency on rainfall intensity and a smaller number of contributing variables. Those findings can be attributed to the buffer effect exerted by an organic cover relative to bare soil. Aside from the provision of higher rainfall interception, Smets et al. (2008) reported that mulching reduces the amount of runoff due to higher storage capacity and soil moisture content, and reduces soil erosion due to both decreased splash erosion and an increased resistance to flow.

Rainfall did not have a significant effect on the organic matter contents of the eroded materials, either in the entire data set or for any of the three treatments alone. In contrast, protective soil cover (or, rather, the lack of it) was a key explanatory variable but only when analyzing all plots together. At the same time, however, time-since-treatment had a significant effect on organic matter contents, whereas treatment did not. These rather complex results probably reflect an increase in bare soil cover in the control and PAM plots combined with an overall

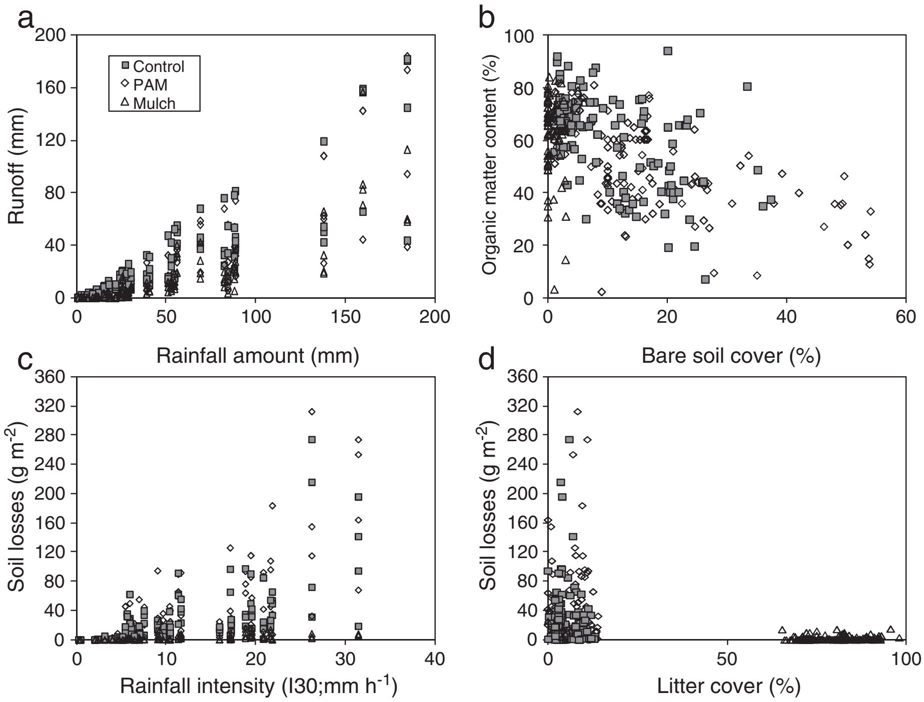

Fig. 6. Relationships of runoff and organic matter content of the eroded sediments (top figures) and soil losses (bottom figures) with the principal explanatory variables of the global multiple regression models (see Table 5) for each one of the plots during the first year after wildfire.

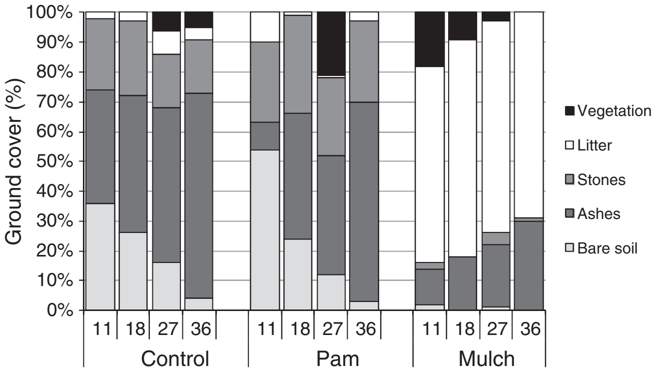

Fig. 7. Ground cover percent of the five cover categories at the individual microplots organized by treatment and slope position (in m from the base) in September 2011, one year after wildfire.

minor decrease in organic matter content from 56 to 53%. A similar decrease in organic matter content was observed by Thomas et al. (1999), although this was during the second year after a wildfire. Overall, post-fire organic matter losses have been poorly studied but the few existing data clearly point to their importance and consequently, the urgent need for further studies into the transport of ashes as the principal source of such high organic matter contents, well above that in the topsoil.

The observed spatial pattern of decreasing runoff and erosion in the upslope direction was unexpected, especially since soil depth did tend to decrease in this direction as well. Moreover, the other soil properties measured in this study offered no plausible explanations, as they revealed no obvious spatial patterns. An exception was the cover of gray-white ash, even though it differed only little across the slope (from 0 to 8%). The role of gray-white ash was probably indirect, reflecting differences in soil burn severity and the associated changes in soil properties (e.g. Shakesby and Doerr, 2006; Varela et al., 2010). The hydrological response at the base of the slope – even seen in the mulched plots – was due to higher fire severity, as suggested by the presence of white ash. Bodí et al. (2011a) also found that soils covered with white ash produce a stronger hydrological and erosive response than those covered with black ash. Another possible explanation for the role of gray-white ash is related to its apparently greater susceptibility to being blown away by the wind, giving rise to bare spots. Various studies, such as Leighton-Boyce et al. (2007), Woods and Balfour (2010) and Bodí et al. (2011b) have shown that the presence of ash can decrease the generation of overland flow.

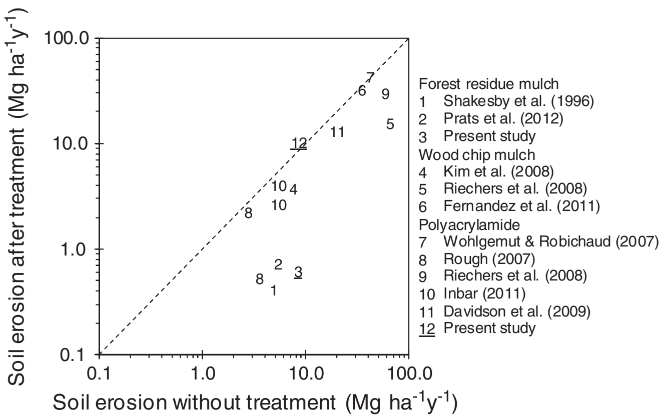

Fig. 8. Annual post-fire soil erosion rates on this and other studies assessing the effectiveness of emergency treatments such as forest residue mulch (1 to 3), wood chip mulch (4 to 6) and PAM (7 to 12).

5. Conclusions

The main conclusions of the present study on the short-term effectiveness of chopped bark mulch and dry anionic PAM during the first year after a wildfire in a eucalyptus plantation in north-central Portugal were the following:

- a litter cover of 80% provided by the chopped eucalyptus bark was highly effective in reducing runoff and especially soil losses throughout the first post-fire year. These results warrant follow-up studies with longer temporal and spatial scales, as well as with different application rates;
- PAM application did not result in a significant reduction of either runoff or soil losses, except for a very few isolated rainfall events. However, its potential advantages do warrant further research, especially in combination with mulching;
- soil losses from the untreated plots during the first year after the wildfire were comparatively high, both for the study region and for the Mediterranean Basin;
- post-fire runoff and soil losses could be well explained by rainfall- and cover-related variables, opening perspectives for the prediction of treatment effectiveness with a temporal resolution compatible with weather predictions;
- post-fire overland flow generation on a microplot scale depended first and foremost on rainfall amount, whereas the associated interrill soil losses were best related to maximum rainfall intensity.

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CHAPTER 5

Effectiveness of hydromulching to reduce runoff and erosion in a recently burnt pine plantation in central Portugal.

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EFFECTIVENESS OF HYDROMULCHING TO REDUCE RUNOFF AND EROSION IN A RECENTLY BURNT PINE PLANTATION IN CENTRAL PORTUGAL

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ABSTRACT

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.

KEYWORDS: wildfire; post-fire erosion; overland flow; soil water repellency; ash

INTRODUCTION

Soil erosion is a key process in the functioning of Mediterranean ecosystems (Cantón *et al*., 2001; Ceballos *et al*., 2003; Cerdà *et al*., 2010), and wildfires represent one of a number of disturbances in forests and shrublands that can greatly increase soil and fertility losses (Cerdà, 1998a, 1998b; Shakesby & Doerr, 2006; Shakesby, 2011). The consumption of the vegetation and litter layer by fire increases both overland flow—because of the reduction of rainfall interception and resistance to flow—and sediment losses by increasing the splash erosion by raindrops (Soto & Diaz-Fierros, 1997; Llorens & Domingo, 2006). Additionally, the fire-induced heating of the soil can reduce aggregate stability, decrease porosity, and increase soil water repellency (SWR), and these changes can decrease infiltration and increase soil erodibility (DeBano, 2000; Ferreira *et al*., 2008; Keizer *et al*., 2008; Malvar *et al*., 2011; Prats *et al*., 2012).

The association of wildfire with on-site soil erosion and downstream flooding and massive sediment deposition has become increasingly recognized (Kraebel, 1934) and, in the early part of the last century, led to the first systematic soil erosion control treatments following wildfires (Munns, 1919). The first post-fire rehabilitation efforts consisted of building engineering structures (check dams) in stream channels to trap the sediments and of seeding hillslopes to increase ground cover (Wohlgemuth *et al*., 2009). However, it was proved to be unrealistic to build check dams in the short periods between the occurrence of the wildfires and the occurrence of the erosion-producing rains; also, various studies started to question the effectiveness of seeding to reduce soil erosion during the 1980s (Gautier, 1983; Taskey *et al*., 1989).

During the 1990s and the 2000s, research on post-fire erosion mitigation concerned seeding (e.g., Pinaya *et al*., 2000; Fernández-Abascal *et al*., 2003; Beyers, 2004; Robichaud *et al*., 2006; Groen & Woods, 2008; Peppin *et al*., 2010), construction of erosion barriers by using logs (Wagenbrenner *et al*., 2006; Robichaud *et al*., 2008), and straw mulching (Bautista *et al*., 1996; Badía & Martí, 2000; Wagenbrenner *et al*., 2006). In a nutshell, these studies found seeding to be effective in some cases but not in others, log erosion barriers to be ineffective unless rain events are few and small, and mulching to be highly effective. The effectiveness of mulching was also well-established for agriculture lands (Harris & Yao, 1923; Meyer *et al*., 1970; Lyles *et al*., 1974; Meyer *et al*., 1999; Wilson *et al*., 2004; García-Orenes *et al*., 2009, 2010; Giménez-Morera *et al*., 2010; Jordán *et al*., 2010), cut slopes, and unpaved roads (Grismer & Hogan, 2005; Jordán & Zavala, 2008).

Post-fire straw mulching at rates of c.a. 2 Mg ha^{-1} has been proved to reduce sediment yields by more than 80%

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(Bautista *et al*., 1996; Badía & Martí, 2000; Wagenbrenner *et al*., 2006; Groen & Woods, 2008; Fernández *et al*., 2011; Robichaud *et al*., 2013b). However, straw may be available in only limited quantities in certain regions, including Portugal (Prats *et al*., 2012), and may be redistributed by strong winds as a result of its low weight (Robichaud *et al*., 2000). Straw application can also introduce invasive weeds and inhibit native species recovery (Kruse *et al*., 2004). Despite the increased application costs, other mulches of higher specific weight have also been tested. Forest residues, at application rates of 8 Mg ha^{-1} in Prats *et al*. (2012) and 46 Mg ha^{-1} in Shakesby *et al*. (1996), or wood strands mulch, at rates of 4–12 Mg ha $^{-1}$ in Robichaud *et al*. (2013a), were found to be as effective as straw mulch, whereas wood chips mulch was found to be much less effective (Kim *et al*., 2008; Fernández *et al*., 2011).

Mulching is effective against erosion because it reduces runoff and erosion rates by two mechanisms. First, it increases interception storage capacity, which reduces the amount of rain available for producing runoff, it reduces runoff velocity, and it increases soil moisture (Bautista *et al*., 2009). Second, mulch protects the soil surface against the kinetic energy of rainfall drops and decreases the hydrodynamic power of flowing water (Smets *et al*., 2008).

A recent variant of mulching is that of hydromulching, which refers to the application of a water-based mixture of organic fibers, seeds and a green colorant. It is easily applied because it can be sprayed onto slopes by a jet hose (Naveh, 1975). It also tends to bind strongly to the soil surface by the action of the soil-binding agent, so it is particularly useful on steep slopes and strongly modified areas such as quarries, construction sites, and cut and fill slopes along roads (Emanual, 1976; Benik *et al*., 2003; Robichaud *et al*., 2010). Runoff and soil erosion will be reduced because the hydromulch increases interception storage and protects the soil surface. Additionally, the introduced seeds are intended to increase the vegetative cover, especially when the mulch starts decompose. In burnt areas, seeding requires careful selection of species that are adapted to the target environment, both to guarantee that the seeding produces an adequate cover and to avoid that the introduced species come to behave as invasive weed (Kruse *et al*., 2004). An important disadvantage of hydromulching is its elevated costs, which can range from $3,700.00 to $10,300.00 per ha for aerial application (Hubbert *et al*., 2012). By contrast, the costs for straw mulching are on the order of $600.00 and $1,200.00 per ha for application by helicopter and by hand-spreading, respectively (Napper, 2006). Despite this greater expense, hydromulching has been used especially in the USA after some fires when access was difficult, the slopes were too steep or subject to wind to use straw mulch and when there were particularly important 'values at risk', such as water reservoirs, cultural or natural heritage sites, or industrial plants.

The effectiveness of hydromulching in reducing post-fire runoff and erosion has not yet been fully established. Although Robichaud *et al*. (2013b) found no marked decrease in post-fire runoff, Hubbert *et al*. (2012), Rough (2007), and Robichaud *et al*. (2010, 2013a) did report substantial reductions in erosion rates (with 65–95%). However, these reductions were restricted to the first year after hydromulching, which the authors attributed to the rapid breakdown of the mulch layer. Wohlgemuth *et al*. (2011) also found hydromulching to markedly reduce overall erosion rates (by 60–80%) but not the sediment losses produced by high-intensity storms. Robichaud *et al*. (2010) suggested that hydromulching would be most effective on short slopes (10–20 m), where interrill erosion is the dominant process and the hydromulch mat is less likely to be detached by rill incision. However, Rough (2007) found aerial hydromulching to be highly effective on long hillslopes with elevated rill densities (0·1 rill m^{-2}).

Given the elevated potential of hydromulching for post-fire rehabilitation, there is a clear need to test its effectiveness in geographical regions outside the USA. Although hydromulch can include surfactants, the effectiveness of hydromulching has been poorly assessed for vegetation types associated with strong or extreme SWR, such as the eucalypt and pine plantations that dominate in north-central Portugal (Ferreira *et al*., 2008, Keizer *et al*., 2008; Prats *et al*., 2012). Also, the effectiveness of hydromulching after post-fire salvage logging is poorly known in spite of being perhaps the most common practice following wildfires in north-central Portugal. Salvage logging was typically being used to recover timber values and reduce the risk of insect infestation (McIver & Starr, 2000), but it can trigger runoff and soil erosion through soil alteration and forest floor disturbances (Rab, 1994; Castillo *et al*., 1997; Edeso *et al*., 1999; Fernández *et al*., 2004, 2007).

The overall aim of the present research was to study the effectiveness of hydromulching to reduce runoff and erosion over a three-year period in a recently burnt and logged pine plantation in north-central Portugal. The specific objectives were to (i) assess the effectiveness of hydromulching in reducing runoff volumes and sediment yields at the plot scale; (ii) analyze the changes in runoff and soil erosion over time and across plot size (0·25, 0·5, and 10 m^2 plots); and (iii) determine the effect of hydromulching on key soil properties, surface cover, and vegetative recovery, and the extent to which these mulching-induced changes can explain the observed differences in runoff and erosion between the hydromulched and untreated plots.

MATERIAL AND METHODS

Study Area and Site

This study was conducted near the village of Colmeal in the Góis municipality of north-central Portugal (N 40º08′42″, W 7º59′16″; 490 m asl). On 27 August 2008, a wildfire burnt 68 ha of forest lands. A west-facing 25 degree steep hillslope was selected to study post-fire vegetation recovery (Maia *et al*., 2012a, 2012b), and, at a later stage, also for this study. The hillslope had been planted with maritime pine (*Pinus pinaster* Ait.) some 25 years before the wildfire, at a density of 2,600 saplings per ha. The undergrowth was composed of a mixture of Mediterranean and Atlantic shrubs and was dominated by *Calluna vulgaris* I. and *Arbutus unedo* L. (Maia *et al*., 2012b). The study area has a Mediterranean climate with a mean annual temperature of 10–12·5°C (according to Köppen; APA, 2011). The annual precipitation as recorded by the nearest weather station (Cadafaz, N 40º08′02″, W 8º32′40″; 12 km W^{-1} from the study area; 25 years of data) was, on average, 1,130 mm but varied from 717 mm to 1,872 mm (SNIRH, 2012). The soils were shallow, 30- to 35-cm deep Humic Cambisols (WRB, 2007), overlying schist, as was observed from four soil pits dug during November 2008 (Table I). A soil sample was collected at 0–5 cm depth in each pit, and later analyzed, using standard laboratory methods, for bulk density (Porta *et al*., 2003), porosity, and grain-size distribution (Guitian & Carballas, 1976). Percent organic carbon was determined by a carbon analyzer (Flash EA 1112 series by Thermo Finnigan, USA) and multiplied by the van Bemmelen factor (1·724) in order to obtain the organic matter content on the soil (Jackson, 1958).

Experimental Design, Field Data Collection, and Laboratory Analyses

At the location selected for this experiment, the 2008 wildfire had completely consumed the pine crowns, so there was basically no needle cast after the fire (Table I). On 11 December 2008, 106 days after the fire, more than half of the soil surface corresponded to black ashes, a third to stones, and less than 10% to bare soil. The fire severity was classified as moderate according to various severity indices described in Maia *et al*. (2012b) at locations some 5–10 m distance from the present experiment. For example, the maximum temperature reached (Guerrero *et al*., 2007) by the soil at 0–3 cm depth, estimated with near-infrared spectroscopy, was, on average, 78°C; the twig diameter index (Maia *et al*., 2012a), which ranged between 0 (unburnt) and 1 (very intense wildfire) was, on average, 0·4 (Table I).

Table I. Indicators of fire severity, ground cover, and mean soil properties from 0- to 5-cm depth ($n = 4$)

Site characteristics	Average	±	SD
Overall fire severity		Moderate	
Tree canopy consumption		Total	
TDI	0·4	±	0·1
MTR (°C)	78	±	30
Ground cover in December 2008 (%)			
Litter	2	±	1·3
Black ashes	56·6	±	9·7
Bare soil	7·2	±	3·7
Stones (>2 mm)	34·2	±	8·3
Soil properties			
Soil depth (cm)	35·3	±	4·3
Slope (º)	24·5	±	3·4
Bulk density (g cm^{-3})	0·8	±	0·1
Porosity ($cm^3 cm^{-3}$)	0·5	±	0·1
Organic matter (%)	16·4	±	1·6
Soil texture			
Clay (%)	8·4	±	1·9
Silt (%)	35·8	±	9·0
Sand (%)	55·8	±	12·8
Stoniness (>2 mm) (%)	36	±	15·0
USDA soil texture class		Sandy loam	

TDI, twig diameter index; MTR, maximum temperature reached, following Maia *et al*. (2012a, 2012b); SD, standard deviation; USDA, United States Department of Agriculture.

Because the National Forestry Authority had decided to log the stand as soon as possible because of the risk of nematode infestation, the experimental set up of this study involved four phases. The first phase comprised the installation of a tipping-bucket rain gage (Pronamic professional rain gauge with an event logger) in combination with a storage gage for validation purposes. This was carried out on 15 September 2008, prior to any rainfall following the wildfire. After that, the rainfall was measured weekly from the storage gage, and the maximum weekly or monthly 30-min rainfall intensity ('I30', in mm h^{-1}) was calculated for each period from the tipping-bucket rain gage data series.

On 5 November 2008, the pretreatment period started with the installation of four plots bounded with metal sheets. Two were micro-plots of approximately 0·5 × 0·5 m, whereas the other two were small plots of approximately 0·5 m wide and 1·0 m long. The outlets of each plot were connected, using garden hose, to 30 L tanks, where the runoff was collected. The runoff volume in each tank was measured at 1- to 2-weekly intervals, depending on rainfall, from 5 November 2008 to 12 October 2010, except during March 2008 when the runoff measurements had to be interrupted because of the logging activities. This 23-month period was divided in a pretreatment and posttreatment period, as further specified in Table II. Whenever runoff exceeded 250 ml, a sample was collected for determination of sediment and organic matter contents by using standard laboratory methods (filtration at 14 μm, drying for 24 h at 105°C and loss-on-ignition for 4 h at 550°C; APHA, 1998).

The third phase began on 30 March 2009, after the logging had been completed, when two more micro-plots and two more small plots were installed at close distances from the previous micro-plots (<5 m) along with six sediment fences (Robichaud & Brown, 2002) that had been set up at some 10–20 m distance in the upslope direction. Following the design by Fernández *et al*. (2011), these sediment fence plots ('SF plots') of roughly 2-m wide and 5-m long were bounded by means of a geotextile fabric and delimited by metal sheets to avoid run-on into the plots. The geotextile fabric filtered the runoff, and only the sediments accumulated at the bottom of the SF plots were collected at monthly intervals from 31 March 2009 to 12 October 2010. Afterwards, the SF plots were emptied

Table II. Overall figures of rainfall, overland flow, soil losses, and effectiveness of hydromulching during the first 3 years after a wildfire in a maritime pine plantation

		Year 1		Year 2	Year 3
Period		Pre	Post	Post	Post
Start date		5 November 2008	31 March 2009	21 September 2009	12 October 2010
End date		11 February 2009	21 September 2009	12 October 2010	28 November 2011
Rainfall (mm)		609	282	1464	1527
Overland flow					
Number of plots (C/Hm)		*4/0*	*4/4*	*4/4*	—
Runoff (mm)	C	363	140	691	—
	Hm	—	61	152	—
Runoff coefficient (%)	C	60	50	47	—
	Hm	—	22	10	—
Erosion					
Number of plots (C/Hm)		*4/0*	*7/7*	*7/7*	*3/3*
Soil loss ($g\,m^{-2}$)	C	86	217	361	247
	Hm	—	36	63	109
Specific soil loss	C	0·14	0·77	0·25	0·16
($g\,m^{-2}\,mm\,rain^{-1}$)	Hm	—	0·13	0·04	0·07
Organic matter content (%)	C	48	50	52	—
	Hm	—	57	57	—
Effectiveness of	Runoff	—	−56	−78	—
hydromulching (% change)	Soil losses	—	−83	−83	−56
	OM %	—	15	10	—

C, control; Hm, hydromulching; OM, organic matter.

on a single occasion, on 28 November 2011, comprising the fourth phase of this study. The collected sediments were later analyzed for their moisture and organic matter contents by using standard laboratory methods (drying for 24 h at 105°C and loss-on-ignition for 4 h at 550°C; APHA, 1998).

On 31 March 2009, the hydromulch was applied to two of the four micro-plots, two of the four small plots, and three of the six SF plots, all of which were selected randomly. In addition, it was applied to one of two adjacent soil strips of 5-m wide and 10-m long, which had been delineated for monitoring of selected soil properties by using destructive techniques. The hydromulch was provided and applied by Serraic, Lda. by using a jet hose operated by a person on foot. It consisted of an aqueous mixture of wood fibers, seeds, a surfactant, nutrients, a natural bio-stimulant and a green colorant applied at a nominal ratio of 3.5 Mg ha^{-1}. The formulation is confidential, but the company guaranteed that the components are nontoxic for humans or the environment. The seed composition was also confidential, but detailed descriptions of the floristic composition in the SF plots suggested that it included grass (e.g., *Lolium perenne* L.) as well as shrub species [*Cytisus striatus* (Hill), *Ulex minor* Roth.].

Ground cover was measured at seven occasions between 31 March 2009 and 12 October 2010 and finally on 11 November 2011. The ground cover was recorded at each intersection point of a 5 × 5-cm grid in the case of the micro-plots and small plots, and of a 10 × 10-cm grid in the case of the SF plots, that is, at 100, 200, and 400 points, respectively. Each recording involved classifying the ground cover according to seven categories: stones bigger than 2 mm ('Stone'), bare soil ('Bare'), ashes ('Ash'), litter ('Litter'), hydromulch ('Hm'), native vegetation ('Natveg'), and vegetation introduced by hydromulch ('Introveg'). The data also were grouped into two lumped categories: total vegetation ('Tveg') and total protective ground cover ('Hlv'), with the latter being the sum of hydromulch, litter, and vegetation.

The soil strips were sampled at monthly intervals from 22 April 2009 to 11 August 2010 for a total of 17 occasions. Sampling involved destructive measurements of soil shear strength, using a torvane (vane tester, Eijkelkamp), and of SWR, using the molarity ethanol drop (Doerr, 1998). At the bottom of each 50 m^2-strip, 15 equally spaced measurements were made along a horizontal transect, and this transect was then shifted approximately 0·5 m upslope for the next sampling occasion. Before measuring shear strength or repellency, any hydromulch, stones, litter, or ashes were removed. The molarity ethanol drop test was slightly modified in accordance with our prior studies (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 s, three drops with successively higher ethanol concentrations were applied until two of the three drops infiltrated within 5 s. The nine ethanol concentrations used were 0, 1, 3, 5, 8·5, 13, 18, 24, and 36%. In data analysis, the overall median of the relative frequency of any ethanol concentrations higher than 0%, calculated over the total measurements in each strip, was called SWR frequency.

Volumetric soil moisture content was monitored at a depth of 0–5 cm at eight locations: four within the untreated SF plots and four within the hydromulched SF plots. This was carried out using eight EC-5 sensors linked to two Em5b data loggers (Decagon Devices, Inc.) and recording data at 10 min intervals. For each read-out period, initial soil moisture content ('Sm') was calculated as the soil moisture at the start of the largest rainfall event during that 1- to 2-weekly period by using the data of the automatic rainfall gage to identify this event.

Data Analysis

For the statistical analyses described in the succeeding text, runoff volumes and (specific) soil losses were fourth-root transformed so that the residuals did not fail the assumption of normality according to the Kolmogorov–Smirnov test at $\alpha \le 0{\cdot}05$, whereas runoff coefficients were square-root transformed for the same reason. Furthermore, 16 read-outs with low rainfall amounts (less than 6 mm) had to be removed from the data set to prevent non-normality of the residuals.

The effects of hydromulching, plot size, and time-since-hydromulching on the dependent variables (runoff volume, runoff coefficient, soil losses, specific soil losses, and organic matter content of the eroded sediments) were assessed by means of a three-way repeated measures analysis of variance (ANOVA) (Ott & Longnecker, 2001). The variance–covariance structure of each dependent variable was selected according to the lowest values of the Akaike information criterion and the restricted maximum likelihood (REML) fit (Littell *et al.*, 2006). The heterogeneous first-order auto-regressive variance–covariance structure was selected for all dependent variables except runoff coefficient, for which a spatial power structure was selected. In addition, specific contrasts between the treated and control plots, for each individual read-out as well as between the three plot sizes, were tested by means of the least squares means and adjusted by the Tukey–Kramer method (Kramer, 1956). Repeated measures ANOVA was also used to test the treatment and time effects on the seven ground cover categories and the initial soil moisture content. In the case of soil resistance and SWR frequency, however, the treatment effect could only be tested using a nonparametric test, that is, the Mann–Whitney *U*-test ($\alpha \le 0{\cdot}05$).

Stepwise multiple linear regressions using the REG procedure in SAS (Littell *et al.*, 1996) were used to determine how well the weekly runoff volumes ($n = 35$) and the monthly soil losses ($n = 17$) could be explained by a set of independent variables. These variables were selected sequentially in a forward selection procedure, in order of decreasing significance by using a minimum p value of 0·05. The 16 independent variables were plot size ('Plotsz'), rainfall amount ('Rain'), 30-min maximum rainfall intensity ('I30'), days since the last rainy day ('Drain'), the seven individual ('Stone', 'Bare', 'Ash', 'Litter', 'Hm', 'Natveg', and 'Introveg'), the two lumped categories ('Tveg' and 'Hlv'), soil shear strength ('Storv'), SWR frequency, and initial soil moisture content ('Sm'). Especially because the various cover categories can be expected to reveal strong correlations, collinearly tests were included in the stepwise procedure, removing independent variables with a condition index higher than 30 (Belsley *et al.* 1980) from the regression models.

RESULTS

Rainfall Amount and Intensity

Rainfall was considerably lower during the first year after the wildfire (1,014 mm) than during the two subsequent years (1,464 and 1,527 mm, respectively; Table II). Even though this study did not commence until 8 December 2008 and had to be interrupted, because of the salvage logging, during March 2009, the present analysis covered almost 90% of the rainfall during the first post-fire year (891 mm; Figure 1). From these 891 mm, 609 mm fell before the logging and the hydromulch application (designated here as 'pretreatment period'), and 282 were measured until the end of post-fire year 1. The highest rainfall amounts were measured during winter, in January 2009 and 2010 with 244 and 262 mm, respectively. The highest rainfall intensities, however, occurred during different times of the first

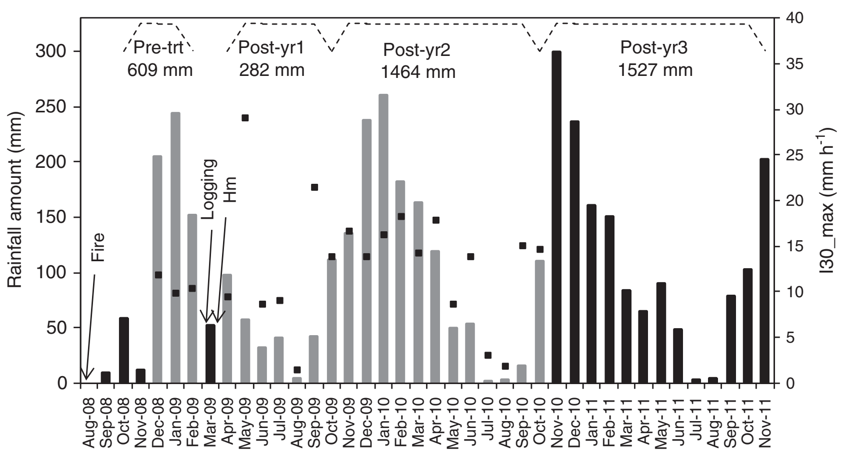

Figure 1. Monthly rainfall (mm) and maximum monthly 30-min rainfall intensity over the study period. Black columns represent total rainfall where no rainfall intensity data were collected. Arrows indicate the date of the fire, logging, and the hydromulch application (Hm), respectively.

post-fire year, during May 2009 and September 2009 with maximum I30 of 29 mm h^{-1} and 21 mm h^{-1}. During the second post-fire year, I30s of 15 mm h^{-1} occurred at least once a month from October 2009 to April 2010.

Ground Cover

At the start of this study, in December 2008, half of the soil surface was covered by ashes, and less than 10% was bare (Figure 2; Table I). By 26 March 2009, after the logging had been completed, ash cover had decreased to 28%, the bare soil cover had increased to 17%, and the stones had become the predominant cover category with, on average, 42%. The recovery of the vegetation was very slow on the control plots, as vegetative cover continued to be near zero 1 year after the fire (August 2009), but reached 30% after the second year (October 2010) and a mere 36% at the beginning of the fourth post-fire year (November 2011). Immediately after its application, on 31 March 2009, the hydromulch provided a cover of 80% on average, but this cover was significantly higher at the two micro-plots and two small plots (90% ± 4%) than at the three SF plots (64% ± 2) (ANOVA, $p < 0{\cdot}05$). This difference was no longer significant after five months (August 2009), even though the hydromulch cover continued higher at the four runoff plots (64% ± 12) than at the three SF plots (47% ± 7; ANOVA, $p = 0{\cdot}06$). There was a marked decrease (5·3% per month) in the average of the hydromulch cover during the first 5 months after its application. After 1 year from the application (1 April 2010), the hydromulch cover decreased to 27% on average (an annual decay rate of 4·6% per month). This decrease in hydromulch cover was, by and large, compensated by an increase in protective soil cover due to the native and introduced vegetation (including the litter it produced). The cover of the introduced vegetation was at its maximum (22%) in June 2010 and became practically zero by November 2011. The native vegetation recovered slowly on the hydromulched plots as well but by November 2011 did attain a clearly higher cover than at the control plots (52% vs. 36%). The total protective ground cover (lumped into the 'hlv' category) was around 75% through all the post-treatment period. When the stone cover is included, a protective layer consistently covered 90% of the surface.

Soil Properties

The monthly values of soil shear strength, frequency of SWR, as well as the soil moisture content over the post-treatment period are depicted in Figure 3. The three variables oscillated across the monitoring period according to the rainfall amounts. Soil shear strength and soil moisture varied in the wake of the rainfall variations. By contrast, SWR showed the lowest values during the rainiest months.

Overall, soil resistance to detachment was lower at the untreated than treated strip ($2{\cdot}4 \pm 0{\cdot}7$ kg cm^{-2} vs. $2{\cdot}8 \pm 0{\cdot}5$ kg cm^{-2}; *U*-test: $Z = -5{\cdot}04$; $p < 0{\cdot}01$). Shear strength was clearly lowest at the control strip during 12 out of 17 months as opposed to 2 months at the hydromulched strip, when shear strength was also greater than during the remaining months.

The hydromulched strip, overall, was less repellent than the control (15% vs. 35% SWR frequency; *U*-test: $Z = -6{\cdot}07$; $p < 0{\cdot}01$) and consequently had higher soil moisture (18·1% volume ± 9·7 vs. 14·3% ± 6·7; ANOVA: $F = 7$; $p < 0{\cdot}05$). In certain periods, however, the opposite was true, as is well-illustrated by Figure 3. In the case of soil moisture content, these periods were confined to the dry season of summer 2009; in the case of SWR, it also happened during summer 2010.

Overall Runoff and Soil Losses

Roughly half of the rainfall was converted into runoff on the control plots (Table II). This corresponded to 360 mm of runoff [runoff coefficient (rc) = 60%] during the pre-treatment period, 140 mm during the post-treatment

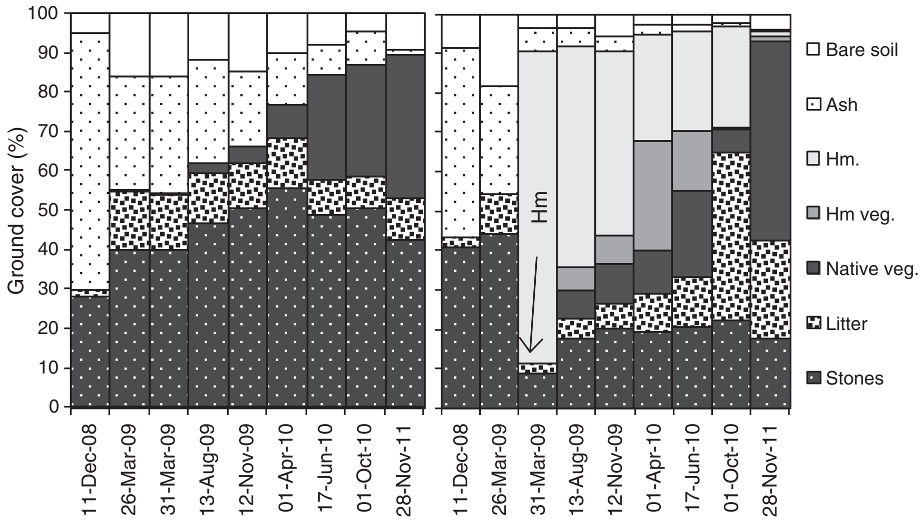

Figure 2. Mean ground cover (%) of the seven categories analyzed in the seven control plots (left) and seven hydromulched plots (right). The arrow indicates the date of the hydromulch application (Hm).

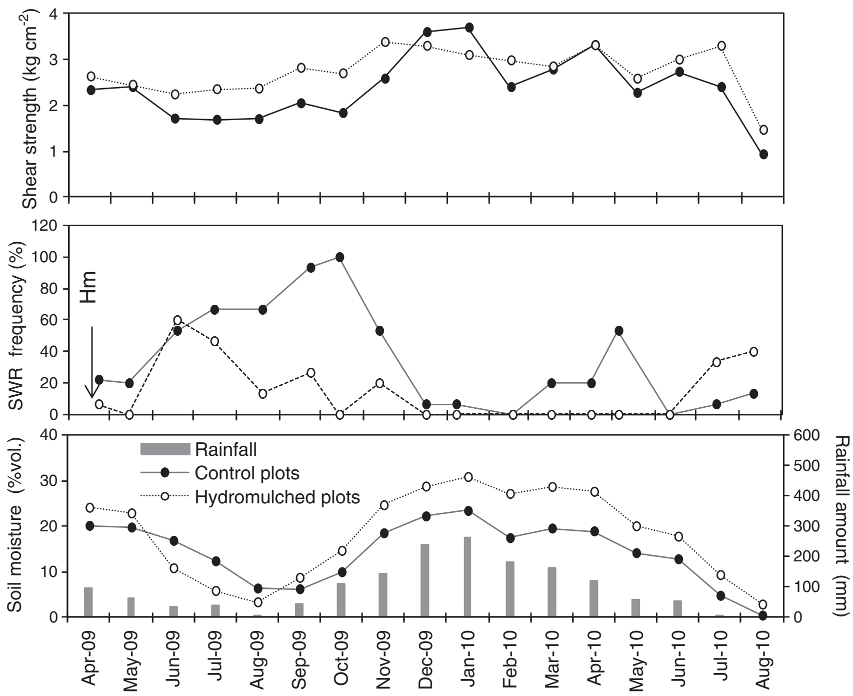

Figure 3. Monthly average values of soil shear strength (top), frequency of soil water repellency (middle) and initial soil moisture content (i.e., prior to rainfall events) and rainfall (bottom) for the control and hydromulched strips.

period of the first post-fire year (rc = 50%), and 691 mm during the second post-fire year (rc = 47%). These differences coincided with the variations in rainfall amount. However, the same was not true in the case of soil losses. The control plots produced, on average, 86 g m^{-2} during the pre-treatment period, 217 g m^{-2} during the post-treatment period of the first post-fire year, and 361 g m^{-2} during the second post-fire year. There was a fivefold increase in the specific soil losses between the pre-treatment and post-treatment periods (from 0·14 to 0·77 g m^{-2} mm rain^{-1}), and after that, the specific soil losses decreased progressively until reaching values similar to those prior to the logging during the third year (0·16 g m^{-2} mm rain^{-1}; Table II).

Hydromulching was highly effective in reducing overland flow, with, on average, 56% during the first post-fire year and even 78% during the subsequent year (Table II). Hydromulching effectiveness in decreasing soil losses exceeded the effectiveness at reducing overland flow to a marked extent, amounting to 83% during both years. During the third post-fire year, however, the effectiveness in mitigating erosion reduced to 56%. Hydromulching did, however, increase somewhat the relative amounts of organic matter in the eroded sediments to 57% as opposed to 50% and 52%.

The ANOVA analysis of Table III showed that the treatment effect strongly influenced all the variables, especially

Table III. Summary of the three-way repeated measures analysis of variance of the 1- to 2-weekly runoff amounts (fourth-root transformed), runoff coefficients (square-root transformed), as well as of the monthly soil losses, specific soil losses (fourth-root transformed) and organic matter contents of the eroded sediments during the posttreatment period (31 March 2009–12 October 2010)

Variable Unit *n*	Df num, den	Runoff amount mm 35	Runoff coeffient % 35	Df num, den	Soil losses g m^{-2} 17	Specific soil losses g m^{-2} mm^{-1} rain 17	Organic matter content % 17
Treatment	1,4	**80·2**	**176·3**	1,8	**71·7**	**63·7**	**9·3**
Size	1,4	1·0	0·0	2,8	3·3	2·6	2·7
Size*treatment	1,4	3·2	3·9	2,8	1·7	1·4	0·3
Time	34,136	**116·6**	**17·3**	16,124	**27·8**	**21·2**	**3·0**
Treatment*time	34,136	**8·4**	**3·2**	16,124	**5·0**	**4·5**	**1·9**
Size*time	34,136	**2·1**	0·7	30,124	**3·8**	**3·6**	**1·7**
Size*treatment*time	34,136	**2·1**	1·1	30,124	**3·1**	**3·0**	1·5

Df, degrees of freedom; num, numerator; den, denominator.
The *F* values in bold, or both in bold, and underlined were statistically significant at $\alpha = 0{\cdot}05$ and 0·01, respectively.

in the case of runoff coefficient (F value of 176) and less important in the case of the organic matter content ($F = 9$). The strong treatment effect, especially in the case of runoff coefficient as highlighted by the big F value (176), contrasted with the lack of effect of the plot size.

In Figure 4 it can be observed that the differences in runoff between plot sizes were very low (in the order of 12–20%, for micro-plots and small plots, respectively). The runoff on the control plots decreased with increasing plot size mainly because of the low runoff amount of one of the small plots (684 mm), whereas the same was true but in the opposite sense in the case of one small hydromulched plot (309 mm). These opposite tendencies resulted in a higher hydrological effectiveness of hydromulching for the micro-plots compared with the small plots (on average, 80% vs. 68%). Plot size also did not play a clear-cut role in soil losses, but the variance increased, especially in the case of the control SF plots (up to 70%). Consequently, the overall reduction in soil losses on the micro-plots and small plots was somewhat higher compared with the SF plots (90%, 89%, and 76%, respectively).

Temporal Patterns in Overland Flow and Soil Losses

The average monthly runoff amounts produced by the untreated plots revealed a marked seasonal pattern in which peak runoff values appeared to antecede the maximum monthly rainfall values during the winter season (Figure 5a). As a result, runoff coefficients were highest during the autumn months, varying between about 80% to 90% in December 2008, November 2009, and October 2010. High runoff coefficients were also observed during late spring and early summer, when rainfall amounts were comparatively small (<53 mm), attaining 62% in July 2009 and 81% in June 2010. The average monthly soil losses at the untreated plots revealed a less obvious temporal pattern (Figure 5b). The four peak losses of 50 $g\,m^{-2}$ $month^{-1}$ or more occurred during autumn (December 2008, September and November 2009) and spring (May 2009). Apparently, the latter peak was associated with the elevated maximum rainfall intensity ($I30 = 29\,mm\,h^{-1}$), whereas the December 2008 and November 2009 ones were rather related to runoff peaks. The average specific soil losses suggested a contrast between the two months with the highest maximum rainfall intensities—that is, May and September 2009—and the remaining months. The specific losses during these two months amounted to 0·8 and 1·2 $g\,m^{-2}\,mm\,rain^{-1}$, respectively, as opposed to the baseline monthly average of 0·25 $g\,m^{-2}\,mm\,rain^{-1}$ for the rest of the study period.

The hydromulched plots produced, on average, consistently lower amounts of monthly runoff as well as monthly soil losses than the untreated plots (Figure 5a and 5b). In the case of runoff, these monthly differences were statistically significant from July 2009 onwards, with the exception of the summer 2009 and 2010 months with little to no rainfall. In the case of soil losses, however, the monthly differences were also statistically significant for the first 2 months following hydromulching and, thus, for basically all of the 19 months with noticeable rainfall. Even so, the three-way ANOVA results indicated that hydromulching did not have an unequivocal statistically significant effect on monthly soil losses, as the triple interaction term of treatment x time-since-mulching x plot size was statistically significant (Table III). The same applied to the corresponding specific soil losses as well as to the 1- to 2-weekly runoff volumes and *mutatis mutandis* (i.e., because of a significant treatment x time-since-mulching interaction) to the runoff coefficients and the organic matter content of the eroded sediments.

Hydromulching failed to produce significant reductions in overland flow generation (average 1- to 2-weekly values) across the whole range of maximum rainfall intensities (Figure 6). There was, however, a tendency for the hydrological effectiveness of hydromulching to decrease with maximum rainfall intensity, reflecting first and foremost the comparatively low effectiveness (<50%) for the two more intense measurement periods that happened in May and September 2009. Also, the effectiveness of hydromulching to reduce average monthly soil losses was comparatively low for these two highest maximum rainfall intensities, albeit it still amounted to some 80% and corresponded to a statistically significant difference between the hydromulched and untreated plots. In overall terms, however, the reduction in soil losses lacked an obvious relationship with rainfall intensity.

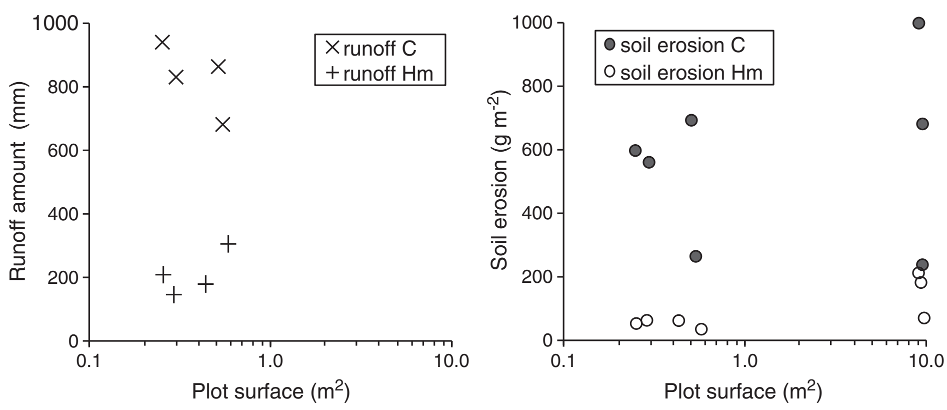

Figure 4. Total overland flow (mm) and total soil losses ($g\,m^{-2}$) of the individual untreated and hydromulched plots over the first and second year of the posttreatment period (31 March 2009 to 12 October 2010).

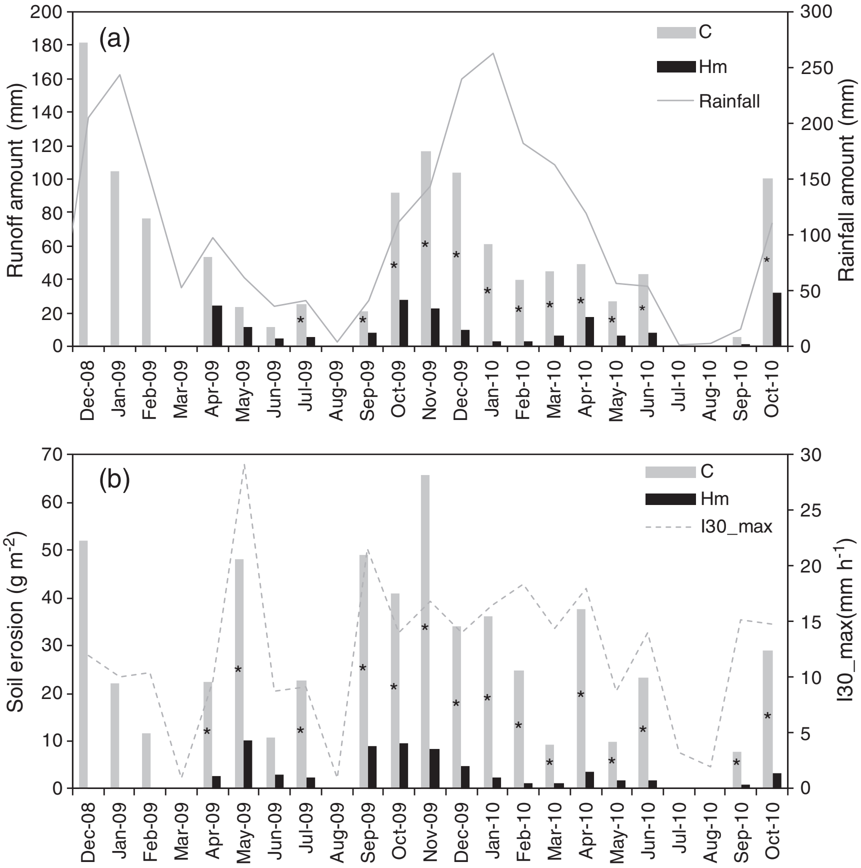

Figure 5. Average monthly values of rainfall (mm) and overland flow (mm) (5a) and of 30-min maximum rainfall intensity (I30; mm h^{-1}) and soil losses (g m^{-2}) (5b) for the untreated and hydromulched plots from the fourth through the twenty-sixth month after the wildfire. Asterisks denote significant least squares mean differences between hydromulched and control plots ($p < 0{\cdot}05$).

Key Factors Explaining Runoff and Soil Losses

Stepwise multiple linear regression with all eight hydromulched and untreated runoff plots together ('global model') revealed that the total protective ground cover ('hlv') stood out as the principal factor in overland flow generation, explaining more than twice as much of the variation in fourth-root transformed runoff amount than the second factor, I30 (31% vs. 13%; Table IV). The hydrological response of the untreated plots alone, however, could clearly be explained best by rainfall amount (41% of variance), whereas that of the hydromulched plots alone was mainly controlled by maximum rainfall intensity, albeit to a lesser degree (19% of variance). Initial soil moisture content was the second most important (and significant) explanatory variable of the runoff produced by the untreated but not the hydromulched plots. The negative sign of its coefficient suggested that the role of initial soil moisture was indirect, with SWR increasingly enhancing overland flow generation as soils dry out. Figure 7 illustrated well that the hydrological response of the untreated plots was stronger under drier than wetter soil conditions. A similar tendency was suggested for the hydromulched plots but just for rainfall amounts below 60 mm, as higher rainfall amounts were associated with wetter soils at the hydromulched than untreated strips.

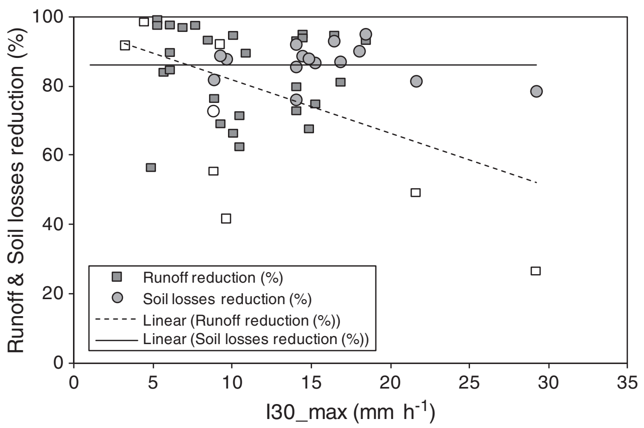

Figure 6. Weekly runoff (squares) and monthly soil losses (circles) reductions at the hydromulched plots compared with untreated plots in relation to 30-min maximum rainfall intensity for the posttreatment period (31 March 2009 to 12 October 2010). Gray-filled/white-filled symbols correspond to significant/not significant least squares mean differences between control and hydromulched plots (at $\alpha = 0{\cdot}05$). Dotted and continuous lines correspond to linear regression equations fitted to runoff and soil loss reductions, respectively.

The predominant role of total protective ground cover ('hlv') was even more pronounced in the case of the global model for soil losses than that for runoff volumes, explaining over half of the variation (55%; Table IV). The most conspicuous contrast between the erosion and runoff

Table IV. Multiple regression models for 1- to 2-weekly runoff amounts ($n = 35$) and monthly soil losses ($n = 17$) for all plots together ('Global') and for the untreated ('Control') and hydromulched plots separately

Selected variable		Global models: Parameter estimate	Global models: Variable name	Global models: Partial r^2	Control models: Parameter estimate	Control models: Variable name	Control models: Partial r^2	Hydromulching models: Parameter estimate	Hydromulching models: Variable name	Hydromulching models: Partial r^2
Runoff amount (mm; 4th root transformed)	Intercept	1·86			1·97			0·40		
	1st variable	−0·01	Hlv	0·31	0·01	Rain	0·41	0·05	I30	0·19
	2nd variable	0·02	I30	0·13	−0·03	Sm	0·11	0·01	Hm	0·05
	3rd variable	−0·02	Sm	0·06	−0·01	Tveg	0·03			
	4th variable	0·01	Rain	0·03						
	Cumulative r^2			0·53			0·54			0·24
Soil losses ($g\,m^{-2}$; 4th root transformed)	Intercept	1·65			1·58			0·76		
	1st variable	−0·01	Hlv	0·55	0·03	Bare	0·26	0·08	Bare	0·35
	2nd variable	0·03	Bare	0·07	0·03	I30	0·11	0·02	I30	0·08
	3rd variable	0·03	I30	0·05	−0·01	Hlv	0·06			
	Cumulative r^2			0·68			0·43			0·43

The independent variables selected (statistically significant at $\alpha = 0{\cdot}05$) were: Rain, rainfall amount; I30, 30-min rainfall intensity, total protective ground cover; Hlv, the sum of hydromulch, litter, and vegetation cover; Hm, hydromulch cover; Tveg, total vegetation cover; Bare, bare soil cover; Sm, initial soil moisture.

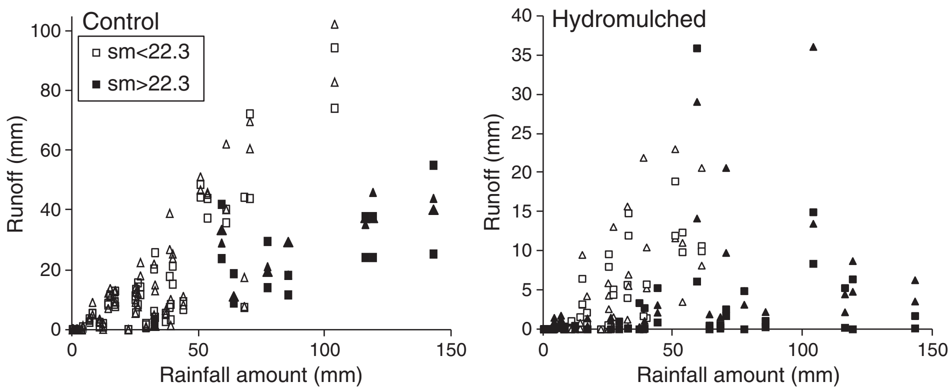

Figure 7. Runoff versus rainfall amounts for the untreated (left) and hydromulched (right) micro-plots (triangles) and small plots (squares) under contrasting initial soil moisture conditions, of less versus greater than 22·3% (open and filled symbols, respectively). Note the different scales of the two Y axes.

results, however, was evidenced by the treatment-specific models. Bare soil cover clearly outranked rainfall amount/intensity as the prime factor explaining soil losses, not only at the untreated plots (26% vs. 11% of variance) but also at the hydromulched plots (35% vs. 8% of variance).

DISCUSSION

Post-Fire Hydrological and Erosion Response in Pine Sites of Central Portugal

Post-fire runoff coefficients as high as observed here were also reported by previous studies in north-central Portugal, such as Ferreira *et al*. (2008) and Malvar *et al*. (2011) by using rainfall simulation experiments. Both prior studies related their strong hydrological response to extreme SWR. In the present study, however, the role of SWR would be limited to the first year after the wildfire, when repellency was moderate, and mostly hydrophilic after November 2009. This reduced importance of SWR was also suggested by the multivariate linear regression model that was fitted to the runoff data from the control plots. The global regression model attested that it was rather ground cover that played a key role in overland flow generation. Pierson *et al*. (2009) likewise argued that ground cover exerted a greater influence on post-fire hydrological response than SWR. Various studies in Portugal (Shakesby *et al*., 1996; Ferreira *et al*., 2008; Prats *et al*., 2012) have furthermore attributed low post-fire runoff coefficients in pine stands to needle cast from scorched tree crowns (Shakesby *et al*., 1996; Cerdà & Doerr, 2008; Ferreira *et al*., 2008; Prats *et al*., 2012).

The soil losses from the control plots during the first post-fire year (302 g m^{-2}) were higher than the range of 80–220 g m^{-2} year^{-1} reported by other studies in burnt pine plantations (Shakesby *et al*., 1996; Fernández *et al*., 2007; Ferreira *et al*., 2008; Prats *et al*., 2012). This could be due to the salvage logging activities that took place during late winter/early spring 2009, as was also suggested by the markedly higher specific soil losses immediately after logging than during the pretreatment period. Logging-enhanced erosion rates were also reported by Inbar *et al*. (1997) and suggested by Malvar *et al*. (2013) but not by Fernández *et al*. (2007). The latter authors attributed their findings to the low severity of the fire, the low rainfall erosivity, and the reduced perturbations of the soil by the machinery employed. To minimize the erosion effects of post-fire logging, it is widely recommended to delay the logging activities until litter fall from scorched tree canopies has provided a 'natural' mulching (Rab, 1994; Castillo *et al*., 1997; Edeso *et al*., 1999; Fernández *et al*., 2004, 2007; Cerdà & Doerr, 2008).

The soil losses during the first post-fire year fitted in well with the low values that were reported by Shakesby (2011) for moderate severity on field plots in the Mediterranean region (321 g m^{-2} year^{-1}), which was attributed to an intensive land-use history. By contrast, in regions of lower forest interventions such as North America, post-fire erosion rates can be one order of magnitude higher, amounting to 2,500 g m^{-2} year^{-1} (Spigel & Robichaud, 2007). The discrepancy between these two geographical regions seems to be much smaller for organic matter losses, with values of 200 and 150 g m^{-2} year^{-1}. High losses of organic matter are of particular relevance as they can easily compromise soil fertility and, thus, on-site land-use sustainability and downstream surface water quality through pollution with toxic pyrogenic organic compounds (Vila-Escalé *et al*., 2007; Campos *et al*., 2012).

A protective ground cover was also the most important factor explaining the monthly soil losses observed in this study and the differences therein between the treated and untreated plots. This agreed well with the bulk of post-fire soil erosion studies (e.g., Benavides-Solorio & MacDonald, 2001; Pannkuk & Roubichaud, 2003; Benavides-Solorio & MacDonald, 2005; Fernández *et al*., 2008; Larsen *et al*., 2009). At the same time, bare soil cover played a key role in the differences in soil losses among the hydromulched plots, as well as among the control plots. Pietraszek (2006) equally attested to the relevance of bare soil cover for soil losses from untreated areas. It could explain 50% of the variability in soil erosion produced by ten sites that had burnt from less than one up to 10 years earlier.

Effectiveness of Hydromulching in Reducing Runoff and Soil Losses

The hydromulch was a complex mixture which contained water, wood fibers, seeds, surfactants, seed-growing biostimulants, nutrients and a green colorant. It is intended that each component affected some of the pieces of the post-fire runoff erosion process.

Runoff was highly reduced at the treated plots, between 56% and 73%, which is higher than in other post-fire mulching experiments, both with straw (Bautista *et al*., 1996; Groen & Woods, 2008) and forest residues (Shakesby *et al*., 1996; Prats *et al*., 2012). Probably, this high effectiveness could be related to the effect of the wood fibers, because it increases the surface water storage capacity, but also due to the effect of the surfactants, a wetting agent that reduces SWR and increases soil infiltration (Leighton-Boyce *et al*., 2007; Madsen *et al*., 2012).

Soil losses were highly reduced in the hydromulch plots during the 3 years after the wildfire. Ground cover was pointed out as the main factor controlling soil losses, but the hydromulch mat showed a rapid decay during the first year after the application. This was identified as one of the disadvantages of hydromulchings (MacDonald & Robichaud, 2007). In the present study, the decayment rates of the hydromulch ranged between 4% and 6% per month, very similar to other research with hydromulch (Hubbert *et al*., 2012; Robichaud *et al*., 2013a). In contrast to those sites, our hydromulch was highly conducive to germination and growth of plants from seeds. Thus, the introduced seeds compensated for the loss of hydromulch with progressively more plant and litter cover, which resulted in more than 70% protective ground cover since the hydromulch application until the third post-fire year (Figure 2).

Besides the composition, the application technique can influence the hydromulch effectiveness. In this study, the area was already logged and the plots were small, which a priori will facilitate the spread of the hydromulch from a jet hose operated on foot. However, the hydromulch cover was significantly lower on the SF plots despite being sufficient to reduce soil erosion. Rough (2007) and Robichaud *et al*. (2010) reported that the hydromulch sprayed from vehicles was intercepted by the standing trees, and they recommended special caution when applying the mixture in areas with a high density of dead trees and from long distances. Aerial hydromulch can be a better and less expensive option, but Hubbert *et al*. (2012) checked that the intended application rates of 50% and 100% hydromulch cover resulted in only 20–26% and 56%.

Unsuccessful hydromulch experiences were first attributed to extreme rainfall events (Wohlgemuth *et al*., 2011) or to the long length of the plots (Napper, 2006). Robichaud *et al*. (2010) pointed out that hydromulch effectiveness depended on slope length, only being effective at slopes shorter than 10–20 m, when interrill erosion was the dominant process instead of rill erosion. The former authors hypothesized that in their long slope sections, the smooth and dense hydromulch mat had little resistance against the sheer force of concentrated flow. But on the other hand, the research of Rough (2007) showed that aerial hydromulching was highly effective and was carried out at the hillslope scale (2,500 m^{-2}, on average), where rills were frequent (0·1 rills m^{-2}) and after extreme rainfall events (I30 = 40 mm h^{-1}). Many other hydromulch formulations are available and are being evaluated for their capacity to reduce soil losses. As concluded by Robichaud *et al*. (2013a), the differences in hydromulch components, application techniques, and application rates can greatly impact hydromulch effectiveness. However, Napper (2006) referred that one of the major problems is the difficulty in knowing the specific chemical composition that was applied in a given situation because most of the hydromulch formulations are kept confidential.

Hydromulching Effects in Soil Properties

Soil properties in agriculture had been typically improved by mulching (Smets *et al*., 2008) by materials such as manure, stones, straw, forest residue, and wood shreds (Harris & Yao, 1923; Mulumba & Lal, 2008; Foltz & Copeland, 2009). Regarding post-fire soil shear strength, the results are not conclusive. Bautista *et al*. (1996) and Fernández *et al*. (2011) found no differences between control and straw mulch plots. Fernández *et al*. (2007) found lower figures in logged compared to unlogged plots. They related these lower values to the absence of roots, once that the logged plots showed a much lower vegetation cover. Agreeing with them, the statistically higher soil shear strength measured on the hydromulch strip could be related to a higher vegetation cover compared to the control strip. Regarding soil water properties, our results are consistent with other mulch experiments (Smets *et al*., 2008; Bautista *et al*., 2009; Prats *et al*., 2012) in which higher soil moistures were found on the mulched areas. The hydromulching layer acted as a water adsorbent dense mat, which effectively increased the soil water retention capacity. It prevented sunlight from reaching the soil surface and thereby decreased soil temperatures. Still, the surfactants included on the hydromulch could have a role in increasing soil infiltration and improve the seed germination (Madsen *et al*., 2012). Besides the positive impacts over plant recovery and soil microbial activity (Bautista *et al*., 2009), a major insight suggested by Prats *et al*. (2012) supported the fact that mulching affected the SWR regime of the burnt forest, promoting the hydrophilic soil conditions. However, this was not true during the dry seasons. Probably, the higher plant cover of the hydromulch (13% vs. 3% during the first post-fire summer) could increase the transpiration and thus lowering soil moisture and increasing SWR. Brainard *et al*. (2012) reported a higher water demand of plants during water stress periods in agriculture, and Soto & Diaz-Fierros (1997) found lower soil moisture on the vegetated areas as compared with bare and burnt plots during the first post-fire summer.

CONCLUSIONS

The main conclusions of this study in the effectiveness of hydromulching to reduce runoff and erosion in a recently burnt and logged pine plantation were as follows: (i) hydromulching, providing coverage of 80%, produced marked changes in SWR and soil moisture, especially in the soil cover. Despite a decrease of up to 30% after 1 year from the application, the treatment induced a highly protective ground cover because of an increase of both vegetative and litter cover; (ii) hydromulching was highly effective during the first 19 months after its application, reducing total runoff volumes by 70% and total soil losses by 83%, and continued effectively during the third year following the wildfire, reducing erosion by 56%; (iii) hydromulching was less effective in reducing runoff (around 30%) but not in reducing soil losses (80%) for the more intense storms (I30 higher to $20\,\mathrm{mm\,h^{-1}}$); (iv) the protective soil cover provided by hydromulch, in combination with litter and vegetation, explained runoff and soil losses better than any other variable, however, rainfall intensity and soil moisture explained a considerable portion of the variation in runoff generation; (v) the application of hydromulch was lower than expected on the larger plots (only a 64% hydromulch cover as compared with 90% in the smaller plots), despite both applications having significantly reduced soil losses. Further research will be needed to determine the effective ground cover in order to match hydromulch decayment rate and vegetative cover increase over time, especially to minimize application costs; and (vi) soil losses were similar across the range of plot sizes studied here (0·25–10 m ^{2}). This, plus the small size of the plots, indicates that interrill erosion was the dominant erosion process. Further research is needed to determine how the effectiveness of hydromulching may vary with increasing slope length when rill erosion is more likely to occur.

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CHAPTER 6

Design and performance assessment of a plastic optical fibre-based sensor for measuring water turbidity

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TECHNICAL DESIGN NOTE

Design and performance assessment of a plastic optical fibre-based sensor for measuring water turbidity

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Abstract
A turbidity sensor based on a plastic optical fibre is presented. The sensor is based on transmission and 90° scattering variations with the total suspended particles in a solution. Transmitted and scattered output signals were characterized and evaluated for different configurations for a large range of clay concentrations. The developed system, in comparison with the OBS-3+ standard system, is more robust, of low cost and has a user-friendly design. A good correlation between the systems was accomplished.

Keywords: turbidity sensor, plastic optical fibres, transmittance, nephelometry, management of sediments, risk assessment

(Some figures in this article are in colour only in the electronic version)

1. Introduction

Turbidity sensors are becoming increasingly used in soil erosion studies and operational water quality monitoring programs for continuous measurement of suspended sediment concentrations. However, the costs of commercially available sensor systems for continuous monitoring of soil losses constitute an important constraining factor. Automatic samplers, for measurements of sediment fluxes at the slope and especially at the catchment's scale, have existed for a couple of decades but can only gather limited numbers of samples, whereas the more recent turbidity sensors continue to be rather costly and, therefore, are generally employed to produce single readings at a fixed height of the water column [1]. However, a low-cost turbidity meter system would allow the employment of multiple sensors across the channel section and with the depth of the water column. Furthermore, as has been proposed by EPA (Environmental Protection Agency) guidance manual, turbidity measurement systems require complex installation and extensive calibration, and present some durability problems because of the electronic parts involved [2].

Fibre-optic-based sensors are suitable to be used in an environment of a potentially hazardous nature without significant sensor performance deterioration and also in situations where multi-sensor operation and *in situ* and remote monitoring are required and offer a new approach to the measurement problems of conventional sensors [3]. In spite of important advances in the last couple of years, the deployment of fibre-optic-based sensors in field research or operational environmental monitoring programs is a largely unexplored area of research. In the literature, some studies can be found which report turbidity sensors, namely for underwater applications [4] and wine industrial processes [5] but mainly for low concentration of suspended sediments, typically 2–4 g l^{-1}. Campbell *et al* presented a fibre optic in-stream transmissometer for high-concentration measurements; however, the authors did not address the scattering dependence

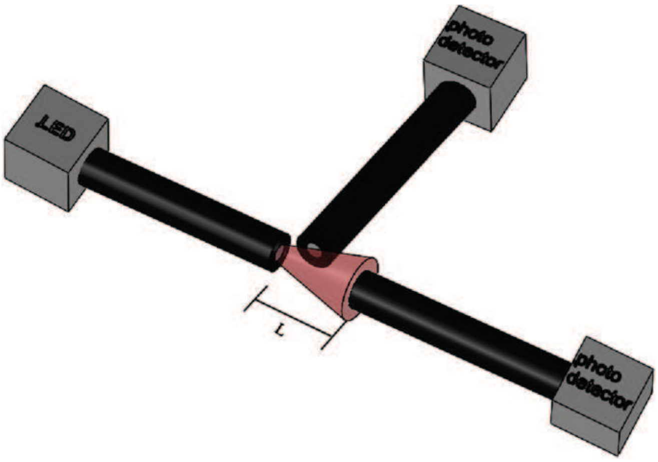

Figure 1. Schematic design of the sensor.

on the concentration of the suspended particles [6]. More recently, Postolache *et al* obtained very promising results, but the data processing of their multi-beam optical system seems too complex for field monitoring applications and the performance was studied for only four turbidity calibration solutions [7].

The turbidity of a medium is directly dependent on its transparency. Suspended matter in a liquid results in the scattering and absorption of light rays. The attenuation and scattering of a light beam passing through a suspension depends on several parameters, namely particle concentration, particle sizes, size distribution and refractive indices of the particle and medium. Here we report on the first design and performance assessment of a plastic optical fibre (POF) turbidity sensor for different clay particle concentrations and thus make a proof of concept.

2. Description

As can be seen in figure 1, the intensity-based POF system design presented here is used to quantify both the amount of light transmitted through a liquid and the amount scattered at an angle of 90° from the incident beam (nephelometry). The system is based on a LED (IF-E96), with a centre wavelength of 660 nm, connected to the emitter optical fibre (HFBR-RUS100), and on two receiver fibres placed at 90° (scattered light) and 180° (transmitted light), each connected to a photodetector (IF-D91). Both output signals were acquired using a NI DAQ board (USB 6008) with a 2 Hz frequency. Experimental results were obtained through a time average procedure of a 3 min acquisition and error bars refer to their SD. A simple application in LabView^{TM} was developed as a user interface, allowing (i) the control of USB 6008, (ii) visualization of the collected data and (iii) data storage.

The system performance was evaluated to empirically determine the best configuration with respect to longitudinal separation of two fibres, L, using several single clay suspensions with a large range of concentrations, up to 10 g l^{-1}, with the particle size distribution between 0.001 and 0.002 mm. Three distances were tested: 2, 5 and 10 mm. Validation of the method was accomplished through the comparison of the selected configuration with a standard commercial system (Campbell OBS-3+) using samples of overland and stream flow collected from the burned study area of Colmeal (Central Portugal). The homogeneity of all suspensions was accomplished by means of a magnetic agitator.

3. Results

For the three established distances between emitter and receiver fibres, the transmitted output signal decreases with increasing concentration of suspended clay particles (figure 2(*a*)). Moreover, in accordance with the Beer–Lambert law, exponential models provided an excellent fit to the measurement results for all three materials (all correlation coefficients were 0.999). Comparing the different configurations, it can be seen that a distance of 2 mm provides higher resolution and range of operation when compared with the distances 5 mm and 10 mm. This is due to the dependence of the light coupling on the axial distance of the fibres. The noise level (4.22 mV) is achieved at 5 g l^{-1} and 9 g l^{-1} for 10 mm and 5 mm, respectively, and extrapolating data are expected to be attained 40 g l^{-1} for 2 mm. However, the 5 mm spacing was preferred for being less susceptible to clogging up under field conditions, especially by the coarser ash and plant particles that are commonly eroded from hill slopes during the initial phases after wildfire.

The scattered output signal (figure 2(*b*)) revealed similar behaviour for all configurations because the scattering receiver was always kept at the same position: as close as possible to the emitting fibre but avoiding direct light. It can be seen that the scattered light only starts to be detected at 1 g l^{-1}. After this threshold, a strong linear correlation ($R^2 = 0.995$) with clay concentration is accomplished, at least up to 10 g l^{-1} (figure 2(*b*)). Trials to place the scattering receiver at greater distances resulted invariably in the total loss of the scattered signal.

Due to the dependence of both the output signals on other variables than the particle concentration, results shown in figure 2 cannot be understood as global calibration curves of each design, being valid only for the specific conditions of this test: clay particles with a size range of 0.001–0.002 mm suspended in water (RI $\sim$ 1.33). However, results suggest that the transmitted and scattered output signals can be used for low and high clay particle concentrations, respectively.

Figure 3 shows the results obtained for 29 runoff samples collected in a Colmeal fire, which were analysed with a commercial backscatter sensor, OBS-3+, and the new developed plastic optical sensor. Scattering results were not used because the concentration of suspended particles in runoff samples was within the threshold. The POF-sensor values (figure 3(*a*)) agree well with those obtained in the initial test with similar concentrations of clay for 5 mm configuration. However, the runoff sample with the highest

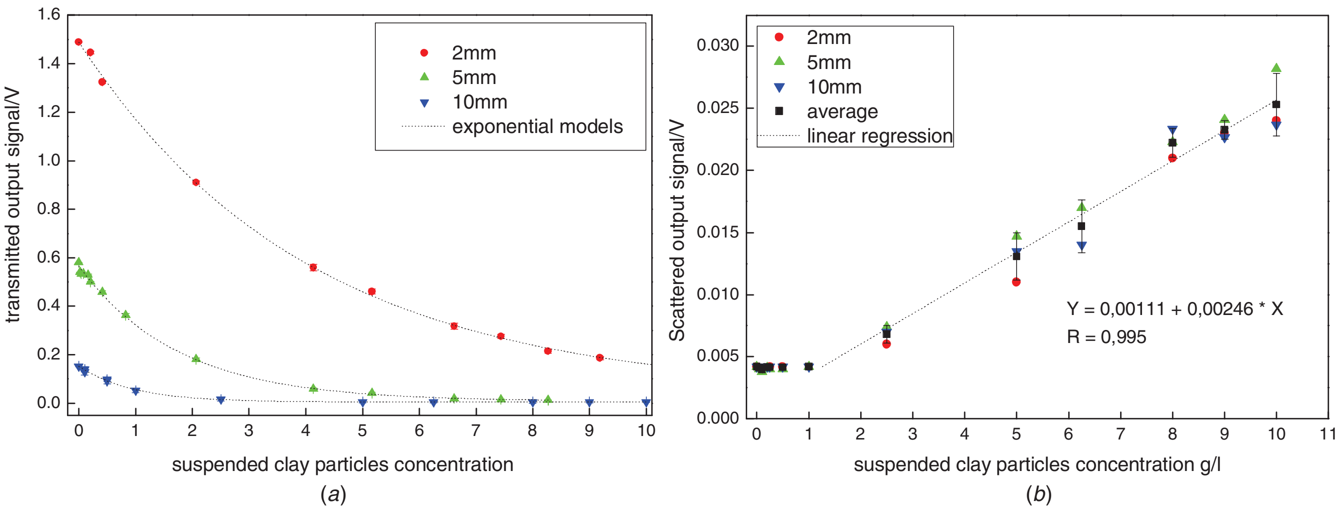

Figure 2. Transmitted (*a*) and scattered (*b*) output signal with varying concentrations of clay for three configurations.

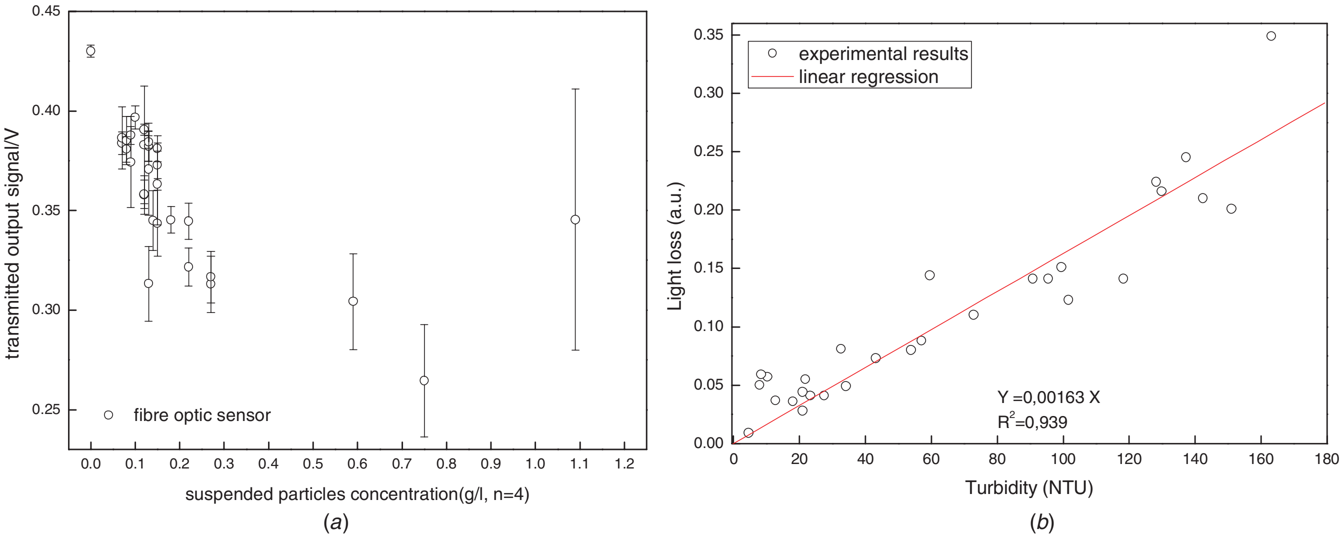

Figure 3. (*a*) Transmitted output signal ± SD of the 29 runoff samples; (*b*) correlation between sensors: POF-sensor measurements displayed in light losses (obtained from $1 - V_{\mathrm{trans}}/V_{0\ \mathrm{g\ l^{-1}}}$) and OBS-3+ measurements in NTU.

sediment concentration (1.09 g l^{-1}) did not follow the same tendency. The higher-than-expected transmitted output signal of this sample was confirmed by OBS-3+ with a comparatively low turbidity. A possible explanation is suggested by the fact that the sample's POF-sensor values are more variable than the values of the other runoff samples. This greater variability could be due to the larger suspended particles, which were detected by visual inspection, suggesting sensitivity of the sensor to the particle size. In fact, particles of larger dimensions can more easily interrupt the coupling of light between the emitter and receiver fibres when passing between them, increasing signal variability. This study will be addressed in further work. With respect to the validation of the proposed method, the POF-sensor values for the runoff samples are closely related to the corresponding OBS-3+ values (figure 3(*b*)). This relationship can be fitted very well by a linear regression equation.

4. Discussion

A new low-cost and robust POF-based system for turbidity evaluation of suspended particle solutions was presented and showed viability on the determination of sediment concentration. From the three configurations tested, a distance of 5 mm between the emitter and transmitted light receiver was selected because it presented the best balance between the sensitivity of the sensor and its capacity to operate with suspended particles of large dimensions. The proof of concept of our system is accomplished but, for the accurate estimation of particle concentration with the proposed sensor, other variables have to be considered and studied, namely particle size. As indicated by this study, preliminary results on this matter suggest that, not only the average transmitted output signal is dependent on the particle size class, but also the output signals variability.

By comparing OBS-3+ and POF-based system performance, a good correlation was obtained. Nonetheless the operation mode is easier with the newly developed system since the homogenization of samples is more difficult with OBS-3+ because measurements have to be performed in 3L tanks (sensor output depends on the tank used). The output of the POF does not depend on the tank, support or specific position and it is cost-effective. The small-sized optical systems make it highly mobile for field measurements. It must be emphasized that the developed system is cost-effective, opening new opportunities for soil erosion and operational water quality monitoring studies.

Further investigation will also be focused in the study of the effect on the system performance of several sediment properties, such as reflectivity, sediment colour and optical properties of the medium, in different field conditions.

Acknowledgments

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CHAPTER 7

Final discussion, conclusions and recommendations

7. Final discussion, conclusions and recommendations

7.1. Final discussion

7.1.1. Post-fire overland flow and soil erosion rates in central Portugal

One of the major advantages of this research was to intensively monitor the overland flow and soil erosion on six different slopes during the first post-fire year, with different plot sizes under artificial and natural rainfall. However, the comparison of post-fire overland flow data with other studies was difficult due to the absence of similar datasets. The few studies that monitored the runoff during the first post-fire year in central Portugal used 16 m^{-2} plots (Ferreira et al., 2008; Shakesby et al., 1996), and exhibited runoff coefficients in the same range as our Pessegueiro plots (12 to 20 % versus 6 to 30 %). However, these studies differed also in time-since fire or surface cover. Other researchers in Eastern Iberian Peninsula have also shown low runoff rates (Bautista et al., 1996; Cerdà et al., 1995; Cerdà, 1998a; Cerdà, 1998b; Cerdà & Lasanta, 2005; Cerdà & Doerr, 2007) but conditions such as the bigger plot sizes, calcareous parent material and the more arid rainfall regime make the comparison of results difficult.

In the case of soil erosion, this research can be compared with both the Portuguese post-fire scenario and other studies around the world (Figure 1). The soil erosion on the untreated plots was low, especially when compared with other studies in North America and NW Iberian peninsula, but more comparable to the Mediterranean figures of Portugal and East Spain, except in the case of Ermida, which reported soil erosion rates as high as 10 Mg ha^{-1}. The low rates were attributed first and foremost to a long history of human landscape impact up to the present days (Shakesby, 2011). This was especially true in the case of the ploughed site of the Açores wildfire (Figure 2; cross “+” symbols). The lower-than expected erosion rates could be related to the fact that ploughing took place several years before the wildfire. The soil erosion could firstly be enhanced immediately after ploughing (as referred to by Ferreira et al., 1997) and decreased several years later, once the soils became depleted and exhausted. On the other hand, the low erosion rates in the Pessegueiro pine plots (Figure 2; triangles) were attributed to lower fire intensity, especially when compared to the Ermida site. In terms of runoff, the effect of pre-fire ploughing seemed not to be as important as fire intensity. In Figure 2 it can be confirmed

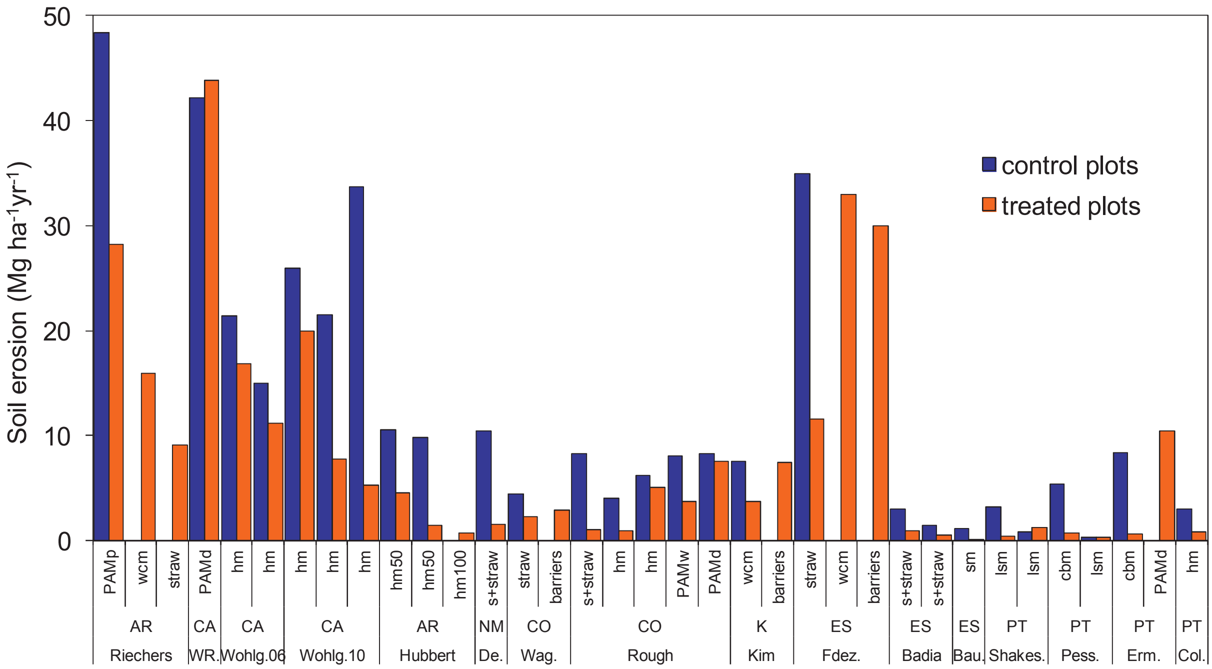

Figure 1. Post-fire soil erosion rates measured on control and mulched treated areas around the world (AR, Arizona; CA, California; NM, New Mexico; CO, Colorado; K, Korea; ES, Spain; PT, Portugal). Treatment abbreviations are: cbm,chopped bark mulch; hm,hydromulch; hm50 and hm100, hydromulch at 50 and 100 % ground cover application; barriers, log or shrub erosion barriers; lsm, logging slash mulch; PAMd, PAMp and PAMw , polyacrylamide dry, pellets and wet formulations; straw, straw mulch; s+straw, seeding and straw mulch; wcm, wood chip mulch. Author abbreviations are: Riechers, Riechers et al. (2008); WR., Wohlgemtu and Robichaud (2007); Wohlg., Wohlgemut et al. (2006, 2010); Hubbert, Hubbert et al. (2011); De., Dean (2001); Wag, Wagenbrenner et al. (2006); Rough, Rough (2007); Kim, Kim et al. (2008); Fdez., Fernández et al. (2011); Badia, Badía and Martí (2000); Bau., Bautista et al. (1996); Shakes., Shakesby et al. (1996); Pess, Prats et al. (2012); Erm, Prats et al. (2013b); Col, Prats et al. (2013a).

that the runoff coefficient remained very low at the Pessegueiro pine plots, while the ploughed Açores plots did not show differences within the mean tendencies. Plot size seemed to affect the overland flow in the range of plots tested. Runoff coefficient tended to decrease with increasing plot size, however that effect was not visible on soil erosion. Given that rill erosion was not observed, interrill erosion was assumed to be the main process across plot sizes.

Some considerations can be derived about soil fertility when analyzing the organic matter percentage in the eroded sediments. This fraction -composed mostly of particulate pieces of charcoal and black ashes- was almost invariable around 50 % and contains a substantial part of the nutrient stocks of the forest (Ferreira et al., 2008; Soto, 1993). Compared with other studies in Galicia (Soto and Diaz-Fierros, 1998) and North America (Spigel and Robichaud, 2007), the annual organic matter losses were in the same range (1 to 5 Mg ha^{-1}), despite the fact that they reported soil erosion rates to be an order of magnitude higher (13 to 20 Mg ha^{-1} at slope scale plots up to 100 m^2) compared to this research. Soto (1993) found that the main factor influencing soil nutrient losses after wildfires was the soil erosion rates. From this point of view, mulching was able to fix carbon and nutrient on post-fire forest ecosystem and also to prevent off-site pollution with pyroxitic toxic organic compounds (Vila-Escalé et al., 2007; Campos et al., 2012). In the medium- and long-term land-use sustainability context, mulching can be of major concern, since the velocities of soil formation are known to be extremely low (Alexander 1985; Alexander 1988). Shakesby (2011) summarized that “the post-fire nutrient losses in mono-specific plantations of pine and, particularly eucalypt on already degraded soils seem to be most at risk because post-fire nutrient depletion exacerbates an already considerable loss of nutrients caused by clear-felling and timber removal alone”.

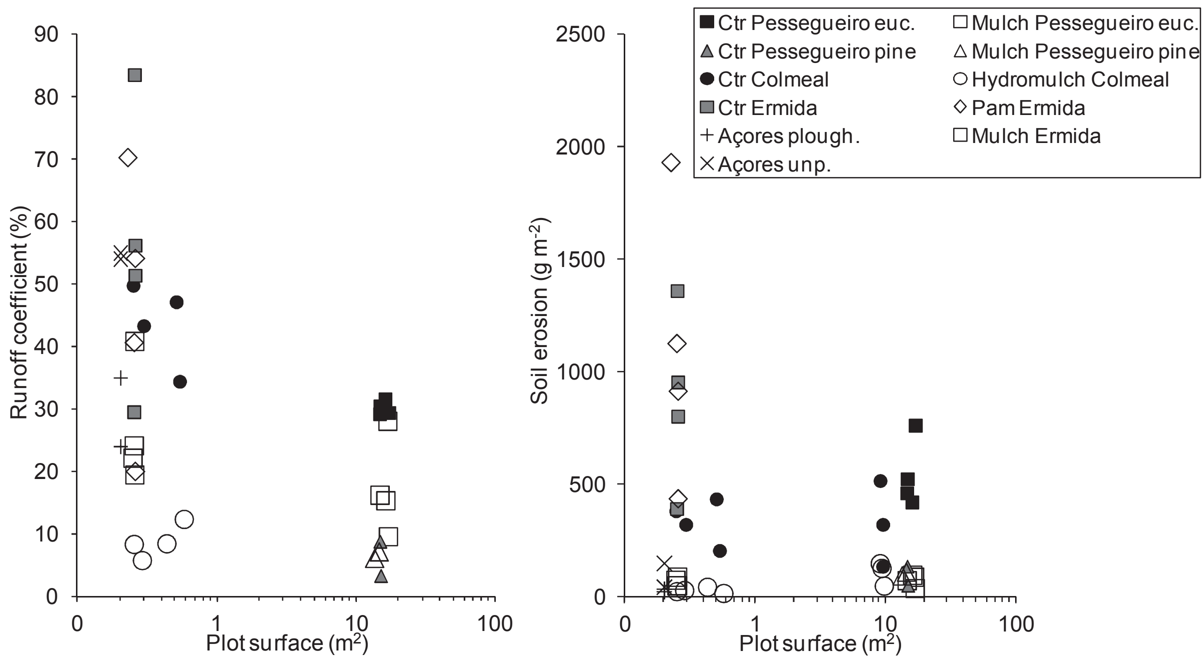

Figure 2. Runoff coefficient and soil erosion versus plot size measured during the first year after the treatment for the entire plot studied in this thesis.

The quantification of the probability of damaging runoff and erosion events after a wildfire cannot be determined as easily as the mitigation treatments or the values at risk. One of the main goals of the soil erosion models was to overcome these uncertainties, but if there are no real data to contrast the model results, the hydrologic and erosive predictions can be highly unrealistic. In the case of the USA, the first steps in post-fire risk assessment -carried out during the 1970s- had to rely on the collective experience and the perceptions of interdisciplinary teams of specialists (Robichaud, 2009). Forty years of research resulted in the creation of a large body of post-fire soil erosion data as well as of post-fire scientific assessment on treatment effectiveness (such as Wagenbrenner et al., 2006). This large body of data permitted the development of specific assessment tools (such as soil burnt severity assessing with hyperspectral satellite imaging, in Robichaud et al., 2007a) and post-disturbance erosion risk prediction tools (such as the probabilistic ErMiT model, in Robichaud et al., 2007b) in order to be less subjective when determining the likelihood of post-fire runoff and soil erosion. In the Portuguese context, post-fire soil erosion studies for periods longer than a year is spectacularly scarce. Consequently, the models predictions can be highly uncertain, once these models were not calibrated or developed for scenarios different to the Portuguese burnt forest. Some FCT-founded research projects tried to solve this lack of information (for example, the IBERLIM, EROSFIRE, EROSFIRE-II, FIRECNUTS and RECOVER project, mainly at the University of Aveiro) and are still gathering or processing soil erosion data. The EROSFIRE project (POCI/AGR/60354/2004), aimed to compare the INAG soil erosion predictions for Portugal, based on the USLE model, against measured values. The preliminary results of the EROSFIRE decision-support tool for post-fire land management revealed that the measured soil erosion can be much lower compared to the predicted figures (Keizer et al. 2012). Recently, the ICNF soil erosion predictions for the Tavira wildfire, which destroyed a total of 24000 ha in the Algarve region (Catraia Technical Report MAMAOT and ICNF, 2012), ranged between less than 5 to more than 200 Mg $ha^{-1}year^{-1}$, which is much higher than the bulk of figures reported on this thesis and also on the extensive review of Shakesby (2011). Far to be optimistic, these results lead to think that soil erosion already happened, and that the Portuguese soils started to get exhausted. It is necessary to consider that the funding for post-fire rehabilitation can be as high as several dozens of millions of euros (€4 millions in the Catraia 2012 wildfire) or dollars ($72 millions in the Cerro Grande 2000 wildfire; Wagenbrenner et al., 2006). It is compulsory to be effective and reach the goals of the post-fire rehabilitation policies. If the runoff and the soil erosion remains unknown, thus the efforts to restore will be inadequate or inappropriate. The “no intervention strategy" could be a realistic technique, but undoubtedly, it is compulsory measuring and checking the extent of the post-fire hydrologic and erosive response. In order to overcome these uncertainties, and until the Portuguese soil erosion dataset can be large enough to feed the models in an adequate manner, a close collaboration between researchers and forest managers must be enhanced, especially when designing post-fire managements strategies or applying technical measures.

7.1.2. The effectiveness of selected post-fire erosion mitigation treatments.

The effectiveness of mulching for reducing post-fire soil erosion has been studied most exhaustively with respect to straw (Badía et al., 2000; Bautista et al., 1996; Dean, 2001; Fernández et al., 2011; Groen and Woods, 2006; MacDonald and Larsen, 2009; Riechers et al., 2008; Robichaud et al. 2010; Rough, 2007; Wagenbrenner et al., 2006), to a lesser extent with respect to wood chips (Fernández et al., 2011; Kim et al., 2008; Riechers et al., 2008) as well as hydromulch (Wohlgemuth et al., 2006; 2011; Hubbert et al., 2012; Prats et al., 2013a) and only rarely with regard to forest residues (Shakesby et al., 1996; this thesis, Chapters 3 and 4; Figure 1). Chopped eucalypt bark mulch providing an initial ground cover of 70-80 % was found here to reduce post-fire runoff and erosion to a similar extent as straw was reported to do by the bulk of the existing literature. At the same time, the chopped eucalypt bark mulch had the important advantages of being readily available, of not being susceptible to removal by wind, and of decaying at slow rates. Reductions in both overland flow and soil losses were somewhat lower at the eucalypt plantation in the Pessegueiro than Ermida study area (40 % *vs.* 50 % and 85 % vs. 90%, respectively), possibly due to the higher application rate in the latter case (8.7 *vs.* 10.8 Mg ha^{-1}).

The logging slash mulch applied at the Pessegueiro pine site, however, appeared to resort little effect on post-fire runoff and erosion. The same applied to the nearby pine site that Shakesby et al. (1996) treated with logging slash mulch (Figure 1). This reduced effectiveness was probably due to the "natural" mulching of the untreated plots by leaf and needle cast from the scorched tree canopies, resulting in low runoff and erosion figures under control conditions. A marked reduction in erosion by post-fire needle cast was also reported by Cerdà and Doerr (2008). Arguably, however, the main disadvantage of mulching with logging slash is the elevated application rate of 18 to 47 Mg ha^{-1}) that is needed to achieve the widely-recommended ground cover of 70 %.

The hydromulching applied at the Maritime Pine plantation in Colmeal was highly effective, reducing runoff in 70 % and soil losses with 80 %. These figures were somewhat better than those reported by the three prior studies evaluating hydromulching in recently burnt areas (Rough, 2007; Hubbert et al., 2012; Wohlgemuth et al., 2011); possibly due to the lower-than-hoped application rates in these three studies as a result of the interception of the spraying jet by the burnt but still upright trees. The observed decrease in hydromulch cover was pronounced (4 - 5 % per month) but comparable to the three above-mentioned field trials. In this study, this decay was by and large compensated by the introduced seeds which did not happened on the previous USA studies. Beyers (2004) pinpointed to the risks of introducing invasive weeds but the plants introduced at the Colmeal plots with the hydromulch had almost disappeared two years later. Despite the elevated costs of hydromulching, its use could be justified where “values at risk” are high, whether in economic, cultural or safety terms, or where recuperation of the spontaneous vegetation is strongly compromised.

The dry granular polyacrylamide applied here was ineffective to decrease post-fire runoff or the associated soil losses. This was possibly due to the preferential binding of the PAM to the ashes in combination with the subsequent removal of the bulk of these ashes by the runoff, similar to what was reported by Rough (2007) and Wallace and Wallace (1986a). Future testing of PAM in recently burnt areas should perhaps focus on find if it could be advantageous when combined with mulching (Davidson et al, 2009; Riechers et al., 2008). On the other hand, the possibility exists that other PAM formulations could be more appropriate for the soils studied here (with their relatively low clay contents) (Sojka et al., 2007) and/or in the presence of a noticeable ash layer. Whilst PAM is a very promising product, including in terms of costs, its successful application in recently burnt areas is thus far from straightforward, as also found by Rough, (2007) and Wohlgemuth and Robichaud (2007).

Several considerations must be taken into account when selecting a treatment to reduce the risk of post-fire soil erosion. First and foremost, the selected treatment must be effective in reducing runoff and erosion. Far from being obvious, innumerous examples exist in which treatment effectiveness was confounded with treatment goal. For example, citing Wagenbrenner et al. (2006): “Studies on the effectiveness of seeding have tended to measure changes in cover rather than erosion rates”. Second, the potential treatment should be cost-effective compared to alternative treatments. Table 1 shows a cost-benefit analysis for the most commonly applied post-fire soil erosion control treatments not only in

Portugal but also in the USA (following Napper, 2006 and Wagenbrenner et al., 2006). The difference in the application rates between straw and forest residues (2.2 vs. 8 Mg ha^{-1}) will largely compensate their differences in price per unit of weight (roughly 150 vs. 30 € per Mg). Mulching with straw and chopped bark were the most cost-effective, especially when compared to the hydromulching, which despite to be effective was very expensive. The choice between these two alternatives will easily come to depend on the availability of the straw and chopped bark in sufficient quantities. In north-central Portugal, this will most likely be chopped eucalypt bark. However, in the case of recently burnt pine, oak or shrublands the application of chopped eucalypt bark seems less recommendable than chopped bark from native tree or shrub species.

Table 1. Estimated costs of the forest residue mulches, PAM and hydromulch used in this study. The values between brackets were averaged from Napper et al. (2006). In order to allow comparisons, the costs of straw mulch, seeding and barriers were also calculated for Portugal and compared to Wagenbrenner et al. (2006) between brackets. The costs of materials, manpower and transportation are approximate and can vary largely depending on material availability, wildfire accessibility, and country regions.

	Effectiveness	Material			Manpower		Transportation		Other	Final cost per hectare	
Treatment	% reduction in soil erosion	Cost	Application rate		persons	€ ha^{-1} day^{-1}	Vehicle type	€ ha^{-1} day^{-1}	€	€ ha^{-1}	\$ ha^{-1}
		€ Mg^{-1}	Mg ha^{-1}	€ ha^{-1}	ha day^{-1}						
Chopped bark mulch	86	30	8	240	3	150	truck	40	50	480	(na)
Logging slash mulch	-16	0	17-47	0	4	200	jeep	30	50	280	(1500)
PAM	-23	20000	0.05	1000	1	50	jeep	30	50	1130	(na)
Hydromulch	83	na	na	2500	2	100	truck	100	50	2750	(6200)
Straw mulch	(95)	150	2	300	2	100	truck	40	50	490	(1000)
Seeding	(-26)	20	0.05	1	1	50	jeep	30	50	131	(220)
Barriers (shrub, LEB)	(40)	0	0	0	6	300	jeep	30	50	380	(1000)

The log and shrub erosion barriers have been used extensively, mostly because the materials (logs, stems, shrubs) are already on the field. However, their efficacy has proved to be much lower, dependent on log storage capacity and the occurrence of small rainfall events, whilst the costs are similar to the mulching (Wagenbrenner et al., 2006). In this sense, the study of the Lourizán Forestry Research Center (Fernández et al., 2011) illustrates very well the low effectiveness of shrub erosion barriers when compared to straw (see Figure 1). In the words of Susana Bautista: “It is very difficult to fail in applying mulch, but it is very easy to fail in installing log erosion barriers”. In fact, Robichaud et al. (2008) verified that 32 % of their log erosion barriers did not have good contact with the ground surface, and 38 % were moved off contour. More precise measurements showed that less than half of the total length of the contour-felled logs effectively stored the runoff and the sediments.

Some labor intensive treatments as contour trenches across the slope with a bulldozer, channel stabilization structures, side slope stabilizations (Rice et al., 1965) and scarification of the soil surface (MacDonald and Larsen, 2009) have been shown to be ineffective for reducing soil erosion. Some of these ground disturbing measures altered the sediment fluxes across the slope and continue to persist long after the emergency is over (Wohlgemuth, 2003). In Portugal, post-fire ground interventions such as ploughing and rip-ploughing increased dramatically the runoff and the soil erosion (Ferreira et al., 1997; Shakesby et al., 2002).

In regions such as the USA, with a large experience in post-fire soil erosion control, the use of mulched based treatments has increased as seeding and erosion barriers have decreased (Robichaud, 2009). This shift in the selection of the treatments was supported by the big bulk of post-fire research compiled by Robichaud (2010). Innovative mulches such as chopped bark, wood strands and the in-situ tree chopping mulching as well as wood and strand mulching were successfully applied during the 2000s (Napper, 2006; Riechers et al. 2008). The aerial application methods were found to be very useful for reaching inaccessible areas by roads, and in these situations they can be more economic compared to hand or ground applications (Napper, 2006). In the Iberian Peninsula context, the first straw helimulch had been carried out by the Lourizán Forestry Research Center (www.vtelevision.es) and applied after the 2010 wildfire seasons in Galicia (NW Spain). However, land managers are still unaware about the advantages of the mulch. Frequently, other treatments different than mulch, are being still applied for post-fire soil erosion control all over Europe, and, worst of all, most of the times its effectiveness in soil erosion reduction is not being assessed.

7.1.3. Key factors for post-fire runoff and soil erosion

The rainfall simulations experiments came to highlight the predominant role of rainfall intensity (measured at 45 and 80 mm h^{-1}) followed by a combination of site-specific factors – such as vegetation and litter, stone cover and soil water repellence - as significant factors for the untreated runoff and soil erosion. It was found that soil water repellence affected the hydrologic response of all our burnt areas, despite the difficulties in comparing it between studies. For example, the soil water repellence measured at the burnt and untreated Colmeal site ("0" median MED class; i.e., hydrophilic during post-fire year 2) was lower than other burnt pine sites in Portugal ("6" annual median in the Pessegueiro pine site and "6" to "8" median MED class in Coelho et al., 2004; Ferreira et al., 2005a and Ferreira et al., 2008, despite to be measured punctually after the wildfire), but still much lower than other burnt eucalypt sites ("8" annual median MED class in both Açores and Pessegueiro wildfires; in Keizer et al., 2008a and Prats et al., 2012 respectively). However, the runoff coefficients in the Colmeal control plots were as high as in the other studies. The highest seasonal runoff coefficient (90 % in autumn of post-fire year 2) coincided with the highest soil water repellence measured in the area (seasonal median value of 3 MED class). Summer and specially the early autumn accounted for the highest repellency levels and the highest runoff coefficients, whereas during winter, accordingly with the lower soil water repellence, runoff coefficient reached minimum values. The broad seasonal variations of soil water repellence and runoff still coincided with other research in Australia (Sheridan et al., 2007) and also in unburnt eucalypt forest (Leighton-Boyce et al., 2007) as well in lignite mines in Germany (Lemnitz et al. 2008).

However, the tests on the relative contribution of each individual factor on the Pessegueiro and Colmeal wildfires datasets coincided also with Larsen et al. (2009) and confirmed that soil water repellence was secondary for runoff amount and that soil cover was the mean factor for soil erosion. Furthermore, our findings revealed valuable data about the proportions at which each factor explained the variation in runoff and soil erosion. With slight differences between sites, half of the variation on runoff was explained by rainfall amount, and between 20 to 5 percent by litter cover. No more than 10 % was explained inversely by soil moisture, which confirmed the secondary role of soil water repellence. In the case of soil erosion, rainfall intensity explained a third of the variability, while the presence of an organic cover explained between 30 to 50 % of the variation, independently of the cover consisting of litter, treatment or vegetation. Other site-specific factors accounted for lower amounts (<5 %) in the models. Regarding the organic matter content in the sediments, it was proven that plot-specific factors such as fire intensity, through the changes in some soil cover variables, were more important than rainfall characteristics. The organic matter content tended to decrease with increasing bare soil and stone cover, despite the resulting models performing worse compared to the runoff and soil erosion models.

The effect of mulch, in a pool including all the control and treated plots, consisted of a shifting to, or strength of rainfall intensity (as also confirmed by Nunes et al., 2010 on their unburnt bare versus vegetated and abandoned areas) on both runoff and soil erosion. The importance of litter increased, explaining from a third to half of soil erosion. The physical factors (rainfall characteristics, soil water repellence, soil moisture and time-invariant soil properties) may not be as important for the treatments that provide immediate ground cover, (chopped bark mulch, slash logging mulch and hydromulching) due to the protection that the organic cover provided over soil detachment and increased soil water storage, and thereby immediately reduce overland flow and soil erosion. However, this was not true for the PAM because it did not affect the ground cover. The beneficial effect of an organic layer has long been studied by many researchers (Harris and Yao, 1923; Morgan, 2005) and is still being developed (Smets et al., 2008; Foltz and Dooley, 2003; Jordan et al., 2010, Wagenbrenner et al., 2006). The strongest position of litter as a key factor for soil erosion was achieved by the most effective treatments (i.e., 93 % reduction for the chopped bark mulch in Ermida, during 12 months) but also by the longest monitoring period (i.e., 80 % reduction on the hydromulch during two years). Similarly to Pietraszek (2006), these findings revealed the importance of having long term series of data and enough replications for a clear picture of the key explanatory factors. The major advantage of this research was to assess with enough replications the spatial and temporal variability of runoff, soil erosion and also organic matter within the same slope, under different fire severities in six different slopes, monitored at short time intervals, which made this dataset especially valuable.

7.1.4. Development of a turbidity sensor.

The knowledge about soil losses other than on small spatial and short temporal scales continues to be rather poor. In part, this can be attributed to the difficulties and expense in measuring sediment fluxes at the slope and especially the catchment's scale (Shakesby and Doerr, 2006). The development of a new POF-based turbidity sensor for the determination of sediment concentration took place under the Lab I and Lab II disciplines of the PhD PROMAR program (Prats et al. 2009a). The good correlation obtained between the commercial OBS-3+ and the POF-based system for a first set of runoff samples opened new opportunities for soil erosion and operational water quality monitoring studies. The new sensor can allow the monitoring of more experimental units and effectively assess the high spatial variability in soil erosion not only on the catchment scale but also on the slope and plot scale. The main constraints in soil erosion monitoring projects pointed out by MacDonald (1994), – to have enough replicates and long time series- can be overcome with the use of this new tool.

The proof of concept of our system is accomplished but, for accurate estimation of sediment concentration, other variables have to be considered and studied, namely particle size and colour. The same is true in the case the optic OBS sensor (Optical Backscatter Sensor; Downing, 2006) where particle size, aggregation and fouling were found to be sources of inaccurate OBS data (Downing, 2006). In burned areas, the presence of ashes deserves special mention. Prats et al. (2010a) realized that the ashes will behave as perfect black bodies absorbing all the light and resulting in anomalous turbidity series. The new prototype has the advantage of measure turbidity in two ways: the 90º scattering output (in which light losses are calculated from the light reflected from the particles in an angle of 90º between emitter and the receiver fibres) and also a direct output (in which the light losses are produced by the particles inside the space between the emitter and the confronting-180ºreceiver fibres). While the scattering output presented the same difficulties in calibration as the OBS-3+, with the A,B, and C regions of measurement (Downing, 2006), the direct light was less dependent on particles reflectivity and could be calibrated through linear or polynomial correlations. As a result of this, turbidity measurements in recently burned areas based on direct light loss will be more robust, given that the scatter series will need complex calibrations, for both the OBS-3+ (Prats et al. 2009b) and the new POF-based turbidity sensor (Bilro et al. 2011).

7.2. Final conclusions

The main conclusions of this thesis on the effectiveness of four selected post-fire mulching techniques in reducing runoff and soil erosion in central Portugal are as follows:

1. The chopped bark mulching was the most cost-effective treatment, reducing runoff by 40 % and soil erosion by 85 % through an increase in ground cover of 70%;
2. The slash logging mulch at the pine site was not effective, probably due to the elevated effectiveness of the needle cast on the untreated plots from the scorched pine canopies. However, it is hardly recommendable as post-fire emergency treatment especially because of the heavy loads (17 Mg ha^{-1}) needed to achieve the recommended ground cover of over 70 %;
3. The polyacrylamides that were applied in dry form did not markedly reduce soil erosion and, thus, cannot be recommended for mitigating post-fire soil erosion. Because of PAM's elevated potential and low costs, however, further work seems justified, especially to diminish the possible preferential biding to ash rather than soil particles;
4. The hydromulch was basically as effective in reducing post-fire runoff and erosion as the chopped bark mulch, but its much higher costs would seem to limit its applications to situations in which the "values at risk" are high and especially vulnerable;
5. the observed soil losses suggested that in north-central Portugal erosion rates during the first year after wildfire tend to be relatively low when compared with other parts of the world, including Mediterranean Europe; nonetheless, the Ermida study site illustrated well that this tendency is not without exceptions;
6. the losses of organic matter observed during the first year after wildfire in particular were comparable to the highest figures reported across the globe, suggesting that wildfire effects on soil fertility losses may be more important for future land-use sustainability than soil losses *per se*;
7. after rainfall total or intensity, ground cover was found to be the main factor explaining runoff and soil erosion differences between mulched and untreated plots.
8. a newly developed fibre-optics-based turbidity sensor produced encouraging results for estimating sediment concentrations of runoff samples, including from recently burnt areas.

7.3. Final considerations

From a land management point of view, this research can conclude that certain field indicators can trigger non tolerable post-fire soil erosion rates in north-central Portugal. Wildfires on slopes up to 20 º, with less than 10 % of litter and leaves covering the soil and complete canopy combustion will result in soil erosion rates up to 5 Mg ha^{-1}. Furthermore, if the ashes colour is white, grey or red in more than 10 % of the soil surface, the risk of soil erosion can rise as far as 10 Mg ha^{-1}. These findings can be used as indicators to correctly implement post-fire emergency soil erosion control treatments but only if the wildfire can trigger on-site and off-site effects that can compromise important values at risk. In situations where the previous conditions were not meet (such as the Pessegueiro pine site or the Açores sites) the "no intervention" option would be preferred (Robichaud, 2009; Bautista et al., 2009). In the case of an urgent intervention, mulching will become the more advantageous technique. Stakeholders, land managers, governments and forestry institutions must be aware of these improvements. More divulgation in post-fire soil erosion control (www.phoenixefi.org/uploads/tecnicas_rel.pdf) is needed in order to save time, efforts and money. In the case of the USA, the most recent development on research is on-line each year through the General Technical Reports (GRT´s, United States Department of Agriculture) and land managers can access it. But the same is not true in the Iberian Peninsula. However, there are several points that require further development.

The next steps with chopped bark mulching could be addressed in order to find lighter materials. Forest residue mulch was highly effective, but lighter materials will allow faster and cheaper implementations. It could be possible to find an optimum of soil erosion control by selecting the longer fibres and removing the shorter ones, or by restricting the application to the spots that presented the field indicators described above. In light of the good results with post-fire mulching all over the world, we strongly recommend to direct the future research towards experimenting with different mulches (needles, agricultural residues, long wood chips) and with spread methods (hand, blowing, aerial, chopping in situ).

The next steps with chopped bark mulching could be addressed in order to find lighter materials. New formulations with hydromulch can be especially valuable in the case of highly sensible ecosystems. The slurry can be modified to include some native seeds that are of interest to the managers. In this way, the hydromulch has a strong potential for restoration of highly degraded areas, as is the case of recurrent burnt slopes, arid and extreme conditions, exotic species, etc. Moreover, the most important constraint would be the expense of the technique. The major advantage of the lightness on the application of PAM will empower future research with this technique. However, it would be necessary to develop new formulations capable of supporting ash-covered sites. The success of soil erosion control with chemical polymers depends on a large number of factors (PAM characteristics, soil texture, soil clay, bedrock, presence of ashes, etc.) and a dichotomic manual or a treatment protocol will be needed in order to identify the specific PAM for the specific burnt site.

Long term series were pointed out by MacDonald (1994) as one of the requisites for monitoring post-fire soil erosion, and research efforts must be directed to evaluate the extent of the “window-of-disturbance”. The first line of a burnt areas emergency strategy must be directed to control soil erosion, but once the immediate risk is controlled, other issues such as forest ecosystem functions, will need further assessment for at least 5 years.

There is an important gap that still needs to be addressed: the upscaling of runoff and soil erosion at the catchment scale. With the range of plots tested, it was possible to test on-site changes in the hydrological and erosive processes, but not the off-site effects. Future studies must check the effect of wildfires and emergency treatments on runoff peak flows, soil and nutrient transportation and deposits in streams and reservoirs. In other words, to develop further researches with the main aim of determine the extent at which wildfires can increase the hydrologic and erosive connectivity of the watersheds, and thus, affect the downstream values at risk. Off-site effect such as water quality degradation, nutrient exportations and loss of volume storage in dams and reservoirs are still unknown. For example, in the “Rias Baixas” region (Galicia, Spain), the ash and sediment deposits ruined the mollusc collection economic activities after the 2006 wildfire season. In a similar manner, roads and structures such as fluvial beaches can be completely destroyed (Lourenço, 2010).

The management of post-fire areas constitutes a delicate compromise between the costs of the treatment and the benefit that the technique will provide. The large body of data of this thesis can serve for develop, calibrate and validate a robust and useful management tool for Portugal. The first steps are currently being developed, and the MMF model has being applied to the Pessegueiro wildfire. Runoff and soil erosion were successfully predicted, differentiating accurately not only the low and high soil erosion rates related to the different burnt severity at the pine and eucalypt sites respectively, but also the reduction in soil erosion between control and forest residue mulch treatments. In that sense, to have a robust model that had been calibrated from real data has undoubtedly a strong potential to save economic and human resources in future similar scenarios.

Further investigation with the new turbidity sensor will be focused on the study in continuum of several sediment properties, such as concentration, reflectivity or sediment colour, particle size, and nutrients, in different field conditions. Currently, the ongoing QREN-founded TRANSFIBRA project has as its main aim the development and further calibration of a low-cost and robust prototype. The next steps will be directed to linking of the sensor to a logger for automatic measuring and storage of turbidity data. The ease of replication can be a major advantage, not only for measuring turbidity at different heights in the water column in a stream, but also in a group of plots. A wireless prototype has been tested in the laboratory and can be of interest to complement the monitoring of the plots. The turbidity sensor has also demonstrated its potential in the case of waste water monitoring and other applications.

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Outputs and Publications

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- Patents

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-Supervising experience

(Co-)Tutorship of Master Thesis of university Students:

1) Eng. Martinho Martins

01/09/2010 - 01/12/2011

Avaliação dos efeitos de “mulching” na resposta hidrologica e de erosão do solo após fogo

(Co-)Tutorship of traineeships of university graduates:

1) Lic. Luis P. Gomes Amaral (RECOVER fellowship)

01/06/2007 - 31/03/2008

Soil erosion conservation techniques and erosion assessment after wildfires.

2) Lic. Alessandra Queirós Pinheiro

01/09/2007-31/03/2008

hidrologic data collection

(Co-)Tutorship of traineeships of university students:

1) Federico Barragan Pimentel (Geology finalist student)

15/09/2007 - 30/06/2008

Soil properties assessment and monitoring in burned and unburned areas.