Mitigating land degradation caused by wildfire: Application of the PESERA model to fire-affected sites in central Portugal

T.C.J. Esteves a,⁎, M.J. Kirkby c, R.A. Shakesby d, A.J.D. Ferreira a, J.A.A. Soares b, B.J. Irvine c, C.S.S. Ferreira a, C.O.A. Coelho b, C.P.M. Bento a, M.A. Carreiras a

a Department of Environment, Escola Superior Agrária de Coimbra, Coimbra, 3040-316, Portugal
b Department of Environment and Planning, Universidade de Aveiro, Aveiro, 3810-193, Portugal
c School of Geography, University of Leeds, Leeds, LS2 9JT, United Kingdom
d Department of Geography, Swansea University, Swansea, SA2 8PP, United Kingdom

⁎ Corresponding author.
E-mail address: tanya@esac.pt (T.C.J. Esteves).

Abstract

Wildfires represent an important agent of land degradation in temperate sub-humid ecosystems, including southern European Mediterranean countries. Identification of integrated conservation approaches that can reduce or prevent degradational impacts is the aim of the EU-funded DESIRE research program, part of which is concerned with quantifying the likely benefit of acceptable alternative conservation strategies to wildfire. The overall aim of this paper is to apply a modification of the Pan-European Soil Erosion Risk Assessment (PESERA) model in order to compare predicted soil erosion rates of one possible conservation strategy, the regular application of prescribed fire, with that of wildfire. The model is applied to two fire-prone study areas in central Portugal (Góis and Mação) and predicts runoff and erosion at much larger spatial (regional) and temporal (decadal) scales than is usually possible with field monitoring. Simulation using the model was carried out for 50 years based on a historical climate time-series. Even assuming very frequent management burns (every 2 years) and infrequent wildfires (100 years), the model suggests that this conservation measure can generally reduce soil erosion relative to infrequent wildfires, although the predicted soil losses for both types of fire are large compared even with those obtained from small-scale field monitoring. The benefits, limitations, scope for improvement and application to future climatic scenarios of the model in a fire context are discussed.

© 2012 Elsevier B.V. All rights reserved.

1. Introduction

Annually, around 50000 wildfires affect on average 600000 ha of mainly forests and shrubland in southern European Mediterranean countries (Rulli et al., 2006). The sub-humid climatic characteristics of these countries make them particularly susceptible to wildfire occurrence, due to strong vegetative growth during the wet season, producing a high fuel load, and pronounced hot and dry summer periods. Together these provide conditions conducive to fire propagation (Lloret et al., 2009). Although fire is a natural phenomenon in the Mediterranean region (Mataix-Solera et al., 2011), with vegetation adapted to it, and fire is important for the management of the dynamics of the vegetation and their fire-prone ecosystems, wildfires are nevertheless considered by many to be one of the main present day soil erosion and degradation agents in the region (Ferreira et al., 2008; Lloret et al., 2009; Mataix-Solera et al., 2011; Pausas et al., 2008; Trabaud, 2002).

The substantial increase in wildfire activity in recent decades in the Mediterranean (Ferreira et al., 2009; Llovet et al., 2009; Pausas et al., 2008; Shakesby, 2011) has turned this natural phenomenon into a serious problem. This increase can be attributed to a range of mainly human-related factors, chief among which are rural outmigration and abandonment of land and associated practices which, in large degree, reflect socio-economic and demographic changes and resulting land use changes (e.g. Moreira and Russo, 2007). These changes have resulted in weaker control and greater accumulation of biomass, allowing changes in the ecological natural processes associated with a substantial increase in wildfire risk (Moreno et al., 1998). Furthermore, this human impact on wildfire occurrence has been compounded by increased temperatures and decreased total rainfall though still with high-intensity rainstorms over the same period (e.g. Badía and Martí, 2008; Pausas, 2004).

Wildfires have presented a major problem for the Portuguese authorities particularly since the mid-1970s. In one year alone (2003), approximately 4.6% of the country was affected by wildfire (Direcção de Unidade de Defesa da Floresta, 2011). The difficulty of dealing with such events lies not only in their destruction of the vegetation, but also in their impacts on the soil. Wildfire is a
phenomenon with one of the highest environmental impacts in the Iberian Peninsula, and wildfires are thought by many researchers to contribute significantly to land degradation and desertification over large areas (e.g. Moreno et al., 1998). For parts of fire-prone western USA, DeBano et al. (2005) actually suggested that wildfires are probably the most important single agent of geomorphological change, although this significance has been questioned as regards south-east Australia (Tomkins et al., 2007) and the Mediterranean (Shakesby, 2011). The effects of wildfires are not limited to the destruction of vegetation. They can also alter the hydrological response of soil, reducing its resistance to erosion and increasing overland flow production, during post-fire rainstorms (Mataix-Solera et al., 2011; Rulli et al., 2006). As part of a major EU-funded five-year project (DESIRE) concerned with the identification of suitable land degradation and desertification mitigation strategies, this paper aims to investigate the value of applying the Pan-European Soil Erosion Risk Assessment (PESERA) soil erosion model to judge the effectiveness of regular, frequent prescribed fire relative to infrequent wildfire in limiting post-fire land degradation in two fire-prone study sites in Portugal (Mação and Góis). A background to soil erosion and the relevant specific characteristics of post-fire erosion are discussed before considering briefly the characteristics of the study areas, describing the PESERA model and adaptations necessary to apply it to fire occurrence and post-fire soil erosion, assessing the value of the model in a fire context and considering the scope for improvement and wider application.

2. Background

2.1. Soil erosion in the post-fire context

Soil erosion has long been identified as an important global concern, with implications for the maintenance of fertile soil and crop yields (Kirkby et al., 2008). By removing the most fertile topsoil with its organic matter and nutrients, erosion reduces soil productivity, leading, where soils are shallow, to a progressive and ultimately irreversible soil productivity loss, and in vulnerable areas this is one major cause of desertification (Kirkby et al., 2008). Wildfire effects on soil erosion can be diverse and complex, depending on many factors including fire behavior, fire severity (based on the degree of biomass destruction), local meteorological conditions, season and frequency in which fires occur, fuel characteristics, nature of the terrain, such as the slope and topography and soil characteristics (e.g. texture, moisture content and organic content) (Ferreira et al., 2008; Neary et al., 1999). These and other factors and their interrelations make the kind of generalizations necessary for modelling of wildfire effects very difficult.

By destroying much of the standing vegetation and the litter, wildfires typically reduce the rates of rainfall interception and storage together with reducing transpiration and infiltration capacity, which all tend to enhance overland flow (e.g. Cerda and Doerr, 2005; Certini, 2005; González-Pérez et al., 2004; Swanson, 1981). Thus, soil properties that contribute to the successful functioning of the hydrological system and constitute key factors in ecosystem sustainability, namely infiltration capacity, porosity, hydraulic conductivity and soil water storage capacity, may be affected negatively by the fire (Neary et al., 1999; Powers et al., 1990). The complex role of the layer of burnt vegetation and litter that forms ash and charred matter in influencing overland flow and erosion is being increasingly recognized (e.g. Woods and Balfour, 2008). In soils affected by fire, overland flow generation may also be enhanced by the development and/or reinforcement of a water repellent layer (e.g. DeBano, 1971, 2000; Doerr et al., 1996; Imeson et al., 1992). If present, such a layer may prevent water from wetting the aggregates except under prolonged wet conditions and infiltration capacity can be considerably reduced (e.g. Doerr and Shakesby, 2009; Llovet et al., 2009; Neary et al., 1999; Shakesby et al., 2000).

Post-fire intense rainfall events typically increase soil erosion to many times that experienced during pre-burn conditions or that experienced in similar unburned sites (Ferreira et al., 2008; Hyde et al., 2007; Shakesby and Doerr, 2006). With large rainfall events that exceed the storage capacity of an ash layer if present, erosion rates usually increase markedly due to the destruction of the vegetation layer, the availability of highly erodible material and any changes in soil physical and hydrological properties, leading to an increase in runoff and a decrease in the strength of the soil surface that increases the detachability and transportation of sediment (Llovet et al., 2009). It has been shown that fire can reduce the structural stability of a soil and its aggregates, which makes it more easily eroded (e.g. Certini, 2005; Giovannini, 1994; Mataix-Solera et al., 2011; Scott et al., 1998). In addition, fire may also consume part of the root systems, further contributing to the loss of soil cohesion (Hyde et al., 2007; cf. Shakesby et al., 2007). Post-fire erosion rates decline over a period varying from months to years, which is commonly referred to as the “window of disturbance” (Prosser and Williams, 1998). The decline occurs as a result of a range of factors, including exhaustion of easily removed material, build-up (or re-exposure) of a surface stone “lag” and vegetation recovery (e.g. Shakesby and Doerr, 2006; Shakesby et al., 1994; Swanson, 1981) (Fig. 1). In addition to loss of mineral-organic soil material, there can be significant losses of organic matter and available nutrients (e.g. Gimeno-García et al., 2000; Thomas et al., 1999). There is a risk that climate change will cause changes in fire activity and increased soil losses in the Mediterranean. Generally, most authorities agree that temperatures will rise, with most warming occurring in summer, together with an increase in the length of summer dry periods (Bento-Gonçalves et al., 2011; Hoinka et al., 2007; Santos et al., 2002). Although overall, rainfall totals are expected to decrease, torrential rainfall will be more frequent (Badía and Martí, 2008; Bento-Gonçalves et al., 2011; Santoro et al., 2002). In addition to on-site soil and nutrient losses, there may also be off-site impacts including sediment accumulations in channels and on roads and around buildings (Kirkby et al., 2008).

The significance of post-fire erosion as regards the medium-term sustainability of soil in the Mediterranean remains uncertain (Pausas et al., 2008; Shakesby, 2011). Although apparently alarming when compared with unburnt terrain, measured post-fire erosion rates are actually usually relatively modest when seen in the light of other disturbance agents (e.g. agricultural practices). Data concerning measured rates of soil loss usually relate to small spatial and short temporal scales. What is generally lacking in assessments of wildfire impacts is a larger-scale, medium- to long-term perspective. One
way of achieving this is via modelling, which allows the effects of different scenarios (specifically, wildfires and prescribed fire) to be assessed. This process can help to identify locations most at risk as well as being able to judge the relative effectiveness of different erosion-limiting practices over a number of fire cycles under both present and possible future climate scenarios.

2.2. Study areas: Mação and Gós, Portugal

The PESERA model inputs have been based on knowledge of relief, soil and vegetation characteristics, and climate data from two study areas in Portugal, which are the fire-prone municipalities of Mação and Gós. The Mação study area, located in the lower Tejo river basin (Fig. 2), has an “Atlantic–Mediterranean” climate. The area comprises mainly Pinus pinaster and Eucalyptus globulus plantations. Soils are typically very shallow and stony Humic Cambisols on steep slopes (>20°), overlying metamorphic schist and greywacke bedrock. The mean annual rainfall varies from 1000 mm in the north to less than 700 mm per year in the south, with wet winters and dry summers. Fire occurred in 1998 after which pine regeneration and shrub encroachment occurred. Much of the municipality burned again in the catastrophic 2003 fires, and part of what had remained unburned was affected by fire in 2005. In these two recent fires, more than 70% of the Mação municipality was affected. Natural soil degradation and tree regeneration without intervention, together with the effects of mitigation techniques (mainly clearance of fire breaks to manage the fuel build-up) are being assessed (Fig. 3A and B).

Research in Gós municipality is focused primarily on the small 9.7-ha Vale Torto catchment (Fig. 2). It is located on the flank of a quartzite ridge crest, which forms the north-eastern part of Lousã Mountain (“Penedos de Gós”). Gós has a mean annual rainfall of about 1200 mm, concentrated mainly during the winter season. It was affected by several wildfires during the 1970s and early 1980s and, using prescribed fire in the early 1990s, it was burned to provide grazing areas for the surrounding villages. The entire shrub-vegetated catchment was burned in an experimental fire (broadly equivalent to a prescribed fire near the catchment boundaries but hotter toward the main drainage line) in February 2009 (Fig. 4A and B). The soils are very stony and shallow Lithosols, and the catchment is characterized by very steep slopes (>20°). Fieldwork has focused on monitoring soil loss and has been carried out both before and after the fire at small-plot to hillslope scales to assess the efficacy of this tool in reducing soil degradation.

For the purposes of this paper, the focus is on the impact of fire (both prescribed fire and wildfire) on soil erosion. More than two years of post-fire monitoring from the Vale Torto site and from a wildfire-affected site with similar characteristics nearby provide useful ‘real-world’ data with which to compare with erosion values predicted by the PESERA model.

3. The PESERA model

The Pan European Erosion Risk Assessment (PESERA) has been used to provide an objective, physically-based and spatially explicit methodology that has been applied across Europe at 1 km resolution (Kirkby et al., 2008), allowing detailed local applications to be placed in a broader spatial context. In this paper the model is applied at representative points, in order to assess the likely temporal changes resulting from alternative fire management scenarios.
The model is constructed around a central water balance, which separates available precipitation into its possible pathways (Fig. 5). Allowance is therefore made for interception losses, evapotranspiration from both the vegetation canopy and patches of bare ground, overland flow, runoff and infiltration. Infiltration behavior controls overland flow generation by either infiltration-excess or saturation-excess mechanisms (Beven and Kirkby, 1979; Cerdà, 1998), with the former dominating in both disturbed Portuguese sites. Water entering the soil is then further partitioned between subsurface drainage (assumed to be minimal in this environment) and water supply for plant growth via transpiration, which is therefore controlled by potential evapotranspiration and by the availability of soil water within the rooting zone. The model also takes account of leaf-fall and organic matter decomposition, to estimate the biomass of soil organic matter, and this contributes to the runoff threshold and infiltration capacity of the soil.

In the previously-developed version, infiltration capacity is dynamically modified by vegetation cover and cultivation practices. The model assumes that bare areas quickly become crusted during rains, and crusts decay under vegetation cover, or are destroyed by tillage (as can occur in replanting Eucalyptus stands) following wildfire, whereas, under the vegetation, soil retains its fundamental

![Fig. 3. A: Land degradation after fires in the Caratão catchment study area. The area is steep (>20°), with stony cambisols over metamorphic rocks. The former forests of Pinus pinaster and Eucalyptus globulus were 70% burned 1998–2005. General view of catchment. B: Caratão study area. Detail of vegetation cover.](image)

Please cite this article as: Esteves, T.C.J., et al., Mitigating land degradation caused by wildfire: Application of the PESERA model to fire-affected sites in central Portugal, Geoderma (2012), doi:10.1016/j.geoderma.2012.01.001
texture-related infiltration status. The extent and state of the modelled crust are re-assessed monthly, dynamically responding to seasonally changing vegetation cover and accumulating rainfall. Since most overland flow and erosion occur within storms, total rainfall is assessed making use of the frequency distribution of daily rainfalls. Shorter-period rainfall distributions would, of course, be preferable, but they are not available sufficiently widely to be incorporated in the model. For each month, the distribution of daily rainfalls is fitted to a gamma distribution which can be used to assess the longer-term average and frequency distribution of damaging erosion events. In this application, this distribution has been used to generate a series of realizations of climate over a 50-year period. These realizations either use the frequency distributions based on historic records or on a perturbed distribution, so that scenarios of future climate change can be incorporated.

The hydrological component of the model involves dynamic interactions with vegetation, either natural or constrained by land use choices. It is then combined with soil factors that determine erodibility and with topography, based on local relief determined from the best available DEM (90 m SRTM or better), in a physically significant way, to estimate either the long-term average erosion rates (Kirkby et al., 2008) by summing over the frequency distribution of storms in every month, or the erosion forecast for a specific climatic time series, which can either be directly based on historic records or drawn from

Fig. 4. A: Experimental fire in Vale Torto catchment on February 2009. B: View of the Vale Torto catchment, 4 months after the prescribed fire.
the frequency distribution as one possible realization of future conditions. Generally speaking therefore, PESERA combines the effects of topography, climate, soil and land-use into a single integrated forecast of runoff and soil erosion (Fig. 5). Although there are inherent difficulties in formally validating a model of this kind, there has been some evaluation at a number of sites (e.g. Licciardello et al., 2009).

To initiate a model run for PESERA, current rainfall and soil conditions are repeated in an annual cycle, using suitable weightings for the distribution of rainfall events for each month until the annual cycle for hydrology, vegetation and soil organic matter is exactly repeated in an annual cycle, using suitable weightings for the frequency distribution as one possible realization of future conditions. To run the model in time-series mode, for an explicit realization sequence of long-term average rates of runoff, erosion and so on. To run the model in time-series mode, for an explicit realization sequence of daily rainfalls, the time series is started after completion of this spin-up initiation phase.

Any process model, and particularly a coarse-scale one like PESERA, has a number of inherent disadvantages compared with simpler models, including that by Kirkby et al. (2008). These disadvantages include:

1. The need for input data that may not be freely or readily available.
2. The need to rely on spatial soil data collected nationally, using criteria that differ from country to country, combined into soil types that are not completely uniform, and only partially harmonized.
3. Concentration on the relevant, most widespread dominant processes, in this case Hortonian overland flow, so that erosion by saturation overland flow, for example, is poorly estimated at a low value.

Nevertheless, there are advantages in using these types of models:

1. They apply the same objective criteria to all areas, and can be applied throughout a region, provided suitable generic data are available.
2. They provide a quantitative estimate of the erosion rate, which can be compared with long-term average values.
3. The methodology can be repeatedly applied with equal consistency as improved data sources become available to past and present scenarios of changed climate and land use.

Overall, the PESERA model has a secure theoretical base, although forecasting accuracy is limited by the restriction (based on data quality and availability) to daily rainfall data, and to a greatly simplified representation of topography. Within these constraints, the model responds both rationally and in accordance with established principles to variations in climate, land use and topography (Kirkby et al., 2008).

4. Application of PESERA to modelling fire impacts

The PESERA model has been modified to represent both fire likelihood and at least some aspects of the post-fire response, although because of the complexities concerning the impact of some factors (e.g. stone lag development), they have not been incorporated, in the absence of a clear consensus as to their magnitude of their input. A fire ignition model has been implemented within PESERA, using simplified versions of algorithms developed and tested independently (Venevsky et al., 2002) for the Iberian Peninsula. A fire danger index (FDI) is calculated as:

$$ FDI = 1 - \frac{1}{\alpha N} \left[ 1 - \exp \left( -\alpha N \right) \right] $$

where $$\alpha = 0.00037$$.

and $$N = T \left( \frac{T_b}{2} + 4 \right) D_{>-3}$$

where $$T$$, $$T_b$$ are respectively the mean monthly temperature and temperature range, and $$D_{>-3}$$ is the number of days in the month with more than 3 mm of rain.

The generated number of wildfire ignitions depends on two factors; the number of lightning strikes (0.1 to 10 per km² per year) and the number of visitors. The former is the dominant factor in, for example, the Sahel, Africa, whereas in southern Europe the vast majority of fires are attributable to human actions. Fire probability is then calculated as the number of ignitions × the Fire Danger Index. Once started, wildfire extent is calculated from the rate of spread, which decreases with the potential fuel load (dry vegetation biomass) and increases with the wind speed. Within the PESERA model, the fire area cannot exceed one complete grid cell (conceptually taken as 1 km²), which is adequate for many fires, and is commonly represented by fire ignitions in many adjacent cells. In establishing the equilibrium state, fire is ignored. However, for a time series, there are options to include random fires (drawn at random with the calculated fire probability) and prescribed fires (regularly applied in a selected month of the year). These fires are assumed to destroy a variable fraction of the vegetation biomass and soil organic matter within the fire area, depending on fire severity, reducing the biomass in the grid cell, with knock-on effects for runoff and erosion in subsequent years. An additional factor has been added to the original model, to allow for burn severity, and the subsequent destruction of living biomass and soil organic matter during the fire. The severity is to some extent inverse to the rate of spread, with highest severity assumed to occur where the dry fuel load is greatest. The proportion of standing biomass burned is estimated as saturating to 100% as fire severity increases:

$$ p(\text{Veg burned}) = 1 - \exp \left( -\frac{S}{S_0} \right) $$

and the proportion of soil organic matter also responds, but less immediately, as:

$$ p(\text{SOM burned}) = 1 - \left( 1 + \frac{S}{S_0} \right) \cdot \exp \left( -\frac{S}{S_0} \right) $$
where $S$ is the fire severity and $S_n$ acts as a scaling threshold in each case.

Immediately post-fire, the principal modelled effect is the loss of vegetation and litter cover. In addition, depending on burn severity, soil erodibility is assumed to increase considerably as a result of soil surface disturbance. This raised erodibility is made to decrease exponentially with increasing cumulative rainfall, following the results of Ferreira et al. (2005) and Coelho et al. (2004). These components, of burn severity and its impact of soil disturbance, have been included on an experimental basis, and require additional calibration before being ready to be applied more widely. Other factors known to influence post-fire response to date have not been incorporated into the model. In particular, it is hoped to include some of the effects of ash production and water repellency in a later version.

To run the model with these adaptations, the spin-up was completed as before, without any burn effect. Within the time series, there is the possibility of including either prescribed burns or wildfires. The former are modelled as regular occurrences always in the same month. Wildfires use the probability of model-generated ignition to produce random events throughout a year, according to conditions. In each case, the burn severity and area affected are estimated by the model using the algorithm described above. The main impact of each fire is therefore modelled through its effects on biomass and soil organic matter. For a fire of high severity and covering a large area, the soil is very strongly exposed to erosion until the vegetation begins to regenerate. Characteristics of the first rainy season are therefore critical for soil recovery. With no major storm, rapid recovery of the vegetation is likely, and the fire may have little long-term impact on erosion rates; but if there is a major storm, severe erosion may occur, in the worst cases contributing to what could potentially be irreversible degradation. Overall, the normal pattern of disturbance and recovery is visualized as in Fig. 6, with the proviso that the level of fire-induced additional sediment yield depends both on fire severity and the magnitude of the largest storm event within the “window of disturbance”.

Following spin-up, the model was run for 50 years, using a particular realization of the climate time-series drawn from the historical frequency distribution. Fig. 7 shows the mainly seasonal fluctuations in simulated fire probability. Diamonds represent the fires generated in this run, only three of which affected more than 10% of the area. Fig. 8 shows that a large fire followed by a moderate storm (25 mm) produces much more erosion than the largest event (49 mm) when there is no fire. Using the PESERA algorithms unchanged from the basic version to grow the post-fire semi-natural vegetation, full vegetation recovery is estimated to take as much as five years, even with little erosion.

Four model realizations are given in Fig. 9, all using the same climate time series as in Fig. 8, and using the same algorithm and parameters for estimating fire probabilities. Erosion is almost doubled in these four runs, the largest cumulative erosion occurring when fire immediately precedes the 49 mm event in 2012, which increased erosion from this event alone by more than five times. Comparison of these runs emphasizes the highly variable nature of future outcomes, with largely independent stochasticity in the climate and the fire record. This level of uncertainty can also be seen from Fig. 10, where fifteen runs with fire, and five without are summarized as regards cumulative erosion. Both the climate time series and fire incidence have been allowed to vary. For each series (with and without fires), cumulative erosion and the upper and lower envelope of all runs are shown. Although the fires increased mean erosion by about 15%, there is a substantial overlap between the two series, indicating difficulty with erosion prediction.

In Fig. 11, the system response to regular prescribed burns is simulated. As prescribed burn intervals change, wildfire ignition frequency varies little, but the resulting fires are significantly smaller in both area and severity. Managed prescribed fires with a very high frequency (every 2 years) reduce the average overall biomass three- to fourfold compared with infrequent (every 100 years) unmanaged burns. The model also indicates that frequent fires generally reduce erosion relative to the infrequent ones. Thus it suggests some soil conservation benefit from using prescribed fires.

5. Assessment of the application

This tentative application of modelling to the dynamics of forest fires and post-fire recovery is still at an exploratory stage. A number of factors need further consideration, including the influence of ash (e.g. Cerdà and Doerr, 2008; Woods and Balfour, 2008), and the roles of water repellency (e.g. Doerr et al., 2009; Fox et al., 2007) and high stone content (Cerdan et al., 2010; Urbanek and Shakesby, 2009) in modifying the hydrological and soil erosion response of the soil, particularly after fire. Other factors need improved parameterization. For example, there is a need to consider in more depth the relationship between burn severity and the extent of destruction of both above-ground biomass and soil organic matter, which are known to have an effect on post-fire soil erosion (e.g. DeBano et al., 2005), but have not been included at this stage. In its present form, the model treats the whole area as a uniform cell, so that there is also scope for examining the spatial heterogeneity of a fire and its effects on soil erosion. Further analysis of field data for the study sites will hopefully begin to address some of these issues. The methodology developed here explicitly makes use of climatic variables derived from recent weather records. There is scope for applying the PESERA model, therefore, to exploring the response of Mediterranean fire-prone terrain to alternative future climate scenarios.

In its present form, compared with field data, the model appears to over-estimate post-fire soil erosion in the context of the thin stony soils characteristic of much fire-prone terrain in central Portugal and elsewhere in the Mediterranean, which reflect a long history of human intervention. Much higher post-fire erosion is predicted in the model for the 1 km² grid cells than is commonly measured at hillslope- and even plot-scales in the Mediterranean, and Portugal in particular. Post-fire erosion rates measured at plot scales in the Mediterranean relatively rarely exceed 10 t/ha in the first year after fire and many studies have found rates of <1 t/ha (Pausas et al., 2008; Shakesby, 2011). For 16 m² plots in Pinus and Eucalyptus plantations in similar fire-affected terrain in north-central Portugal, erosion rates for an entire fire cycle were estimated at a comparatively modest 2–29 t/ha (Shakesby et al., 1996, 2002). Even with a short fire interval of just 10 years over the modelled period of 1999–2049, the total erosion from successive fire cycles would therefore amount to 8–116 t/ha, which at its greatest is still under half the PESERA estimate for the same period (Fig. 9). Furthermore, these field data were derived from small plots, and erosion would be expected to be progressively lower for increasing measurement scales up to the regional...
scale considered in PESERA. On the other hand, the modelled erosion rates are actually lower than those indicated by $^{137}$Cs inventories at the hillslope scale in burnt shrub-covered terrain in Galicia, northwest Spain (Menéndez-Duarte et al., 2009). These inventories show annual rates of 6.6–6.7 t/ha since the 1960s, which amount to a total of 335 t/ha if extrapolated for a 50-year period. This area of Spain has high rainfall, and rock type and soils differ from the two Portuguese sites considered here, so that different erosion rates might be anticipated. However, erosion rates estimated from cosmogenic nuclide inventories are also much higher than those derived from field monitoring for other Mediterranean land covers and land uses. For example, Schoorl et al. (2004) used $^{137}$Cs inventories to determine erosion rates of up to as much as 69.1 t/ha/yr on ploughed agricultural land near Malaga, Spain, and even up to 8.9 t/ha/yr for unburned pine and shrubland. Establishing whether actual erosion rates lie close to these broader-scale estimates or to small-scale field measurements is important for achieving the goal of Sustainable Land Management and should therefore be an important aim in wildfire impact studies in the Mediterranean (Shakesby, 2011).

The applications of models such as PESERA and of cosmogenic nuclide inventories to wildfire impact on soil erosion both provide promising alternative routes toward assessing medium- to long-term impacts of this landscape-disturbing agent. Development of both approaches should be important foci in wildfire erosion studies as they provide potentially invaluable broad-scale perspectives to complement typically small-scale and short-term field monitoring. Important advantages of modelling using PESERA are: (1) its flexibility; (2) its applicability to future as well as past climate and vegetation scenarios; and (3) its ability to isolate fire-induced from 'background' erosion. Modification of the model parameters to take...
into account some of the factors not so far included should help in improving the assessment of the significance of post-fire erosion.

6. Conclusions

Wildfires are often said to pose a major threat to the sustainability particularly of forest ecosystems in the Mediterranean, not only because they can curtail forest production at least in the short- to medium-term, but also because they generally promote accelerated erosion of forest soils that far exceeds that measured in unburnt terrain. Loss of soil mineral particles may be important if it exceeds the rate of replacement through weathering, but probably of more interest in the short- to medium-term is the accompanying loss of organic matter and nutrients, which could be significant over repeated fire cycles. Most studies of post-fire erosion have considered empirical data and have been carried out at relatively small scales and over short periods, but it is extremely important to understand the degradation processes and effects at larger scales, both spatial and temporal. One means of achieving this perspective is to use functional tools such as the PESERA modelling framework, which has the potential to optimize the management of forest areas, providing some guidance on the effectiveness of alternative broad management strategies for soil conservation and for the maintenance of biodiversity, both now...

Fig. 9. Four realizations of 50-yr wildfire regime. The climate time series is the same in each case, and only random occurrence of wildfires changes between the runs. The vertical scale of erosion is approximately the same in each case. Diamonds indicate fires.

Fig. 10. Envelopes of cumulative erosion, with and without fires. The envelope curves incorporate variability both in the climate realization and in the incidence of ignition events. In this environment the curves show an average 25% increase in erosion due to wildfires, but this can be masked by variability in the weather. The contrasts are strongest when comparing paired catchments — with and without fire.

Fig. 11. Changes in average biomass, frequency of wildfires and erosion under regimes of managed burns. The horizontal axis shows the regular interval between managed burns. The vertical scale shows the average number of additional wildfires per year (diamonds), the average above ground biomass in kg.m$^{-2}$ (squares) and the average erosion in T Ha$^{-1}$ y$^{-1}$ (triangles). Lines and curves indicate only general trends.
and under future climate and land use scenarios. This exploratory study has shown that there is considerable potential in modifying the PESERA model to take into account the episodic nature of wildfire and subsequent erosion in contrast to the more regular seasonal disturbance modelled in agricultural situations, for which the model was developed.

Acknowledgments

This paper forms part of the research financed by project DESIRE (037046): “Desertification Mitigation and Remediation of Land – A Global Approach for Local Solutions” conducted in the framework of the EC-DG RTD – 6th Framework Research Programme (sub-priority 1.1.6.3), which aims to identify alternative desertification and land degradation strategies for using and protecting vulnerable areas. An acknowledgment should also be made to the Recover project (PTDC/AGR-AAM/73350/2006). We would also like to thank Catherine Stoof for organizing the experimental fire and for supplying the images of the Vale Torto study area. The opinions expressed in this paper are those of the authors and do not necessarily reflect the views of the European Commission.

References


