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Coupling wildfire spread and erosion models to quantify post-fire erosion before and after fuel treatments

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Abstract. Wildfires are known to change post-fire watershed conditions such that hillslopes can become prone to increased erosion and sediment delivery. In this work, we coupled wildfire spread and erosion prediction modelling to assess the benefits of fuel reduction treatments in preventing soil runoff. The study was conducted in a 68 000-ha forest area located in Sardinia, Italy. We compared no-treatment conditions *v.* alternative strategic fuel treatments performed in 15% of the area. Fire behaviour before and after treatments was estimated by simulating 25 000 wildfires for each condition using the minimum travel time fire-spread algorithm. The fire simulations replicated historic conditions associated with severe wildfires in the study area. Sediment delivery was then estimated using the Erosion Risk Management Tool (ERMiT). Our results showed how post-fire sediment delivery varied among and within fuel treatment scenarios. The most efficient treatment alternative was that implemented near the road network. We also evaluated other factors such as exceedance probability, time since fire, slope, fire severity and vegetation type on post-fire sediment delivery. This work provides a quantitative assessment approach to inform and optimise proactive risk management activities intended to reduce post-fire erosion.

Additional keywords: fire behaviour, fire management, fire prevention, post-fire impacts.

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Introduction

Wildfire regimes play a key role in structuring many communities of the Mediterranean Basin. Climate and socioeconomic changes can alter wildfire regimes in the future, and increase the risks posed by large and severe wildfires in Mediterranean forests and shrublands (Brotons *et al.* 2013; Lozano *et al.* 2017; Viegas *et al.* 2017; Chergui *et al.* 2018; Ruffault *et al.* 2018; San-Miguel-Ayanz *et al.* 2018; Turco *et al.* 2018; Salis *et al.* 2019). High-severity wildfires can be responsible for negative impacts on ecosystems (DeBano *et al.* 1998; Certini 2005). Among these impacts, several researchers have emphasised the negative effects on soils, which are affected by the removal of vegetative cover and the creation or enhancement of water-repellent soil layers, resulting in increasing surface runoff and erosion potential (Cerdà and Doerr 2007; Larsen *et al.* 2009; Shakesby 2011; Robichaud *et al.* 2013; Fonseca *et al.* 2017; Capra *et al.* 2018). Large and severe wildfires are a potential threat to watershed conditions and can have manifold effects on hydrologic processes including change in flow regimes, flood frequency, erosion and debris flows (Shakesby 2011; Thompson *et al.* 2013; Zavala *et al.* 2014). Wildfires can also lead to changes in stream water chemistry, and post-fire sediment-driven transport can increase contaminant loads, with the related significant consequences for human health, safety and aquatic habitats (Stephens *et al.* 2004; Zavala *et al.* 2014; Nunes *et al.* 2018; Rust *et al.* 2018). It is recognised that the impacts of wildfire on hydrology and geomorphology depend on several inter-related factors including burn severity, soil characteristics, terrain configuration, fuel types and post-fire weather conditions (Shakesby and Doerr 2006; Prats *et al.* 2014; Zavala *et al.* 2014; Hyde *et al.* 2017). For instance, intense rainstorms following wildfires can increase the risk of extensive flooding and high sediment delivery (Onodera and Van Stan 2011; Shakesby 2011; Sankey *et al.* 2017).

Predicting post-wildfire erosion risk can be used to inform fuel, forest and other management investments in terms of scope, extent, and location of treatments (Robichaud and Ashmun 2013; Thompson *et al.* 2013). For instance, hazardous fuel reduction treatments designed to reduce fire intensity and severity (Ager *et al.* 2014; Buckley *et al.* 2014; Elliot *et al.* 2016; Sidman *et al.* 2016; Vaillant and Reinhardt 2017) can be located in areas where fires are most likely to impact runoff. A number of simulation studies have concluded that fuel management can be effective in modifying fire behaviour and burn probability in mediterranean-climate areas and elsewhere (Reinhardt *et al.* 2008; Ager *et al.* 2010; Oliveira *et al.* 2016; Salis *et al.* 2016, 2018; Alcasena *et al.* 2018; Palaiologou *et al.* 2018). Fuel management strategies employ a combination of surface fuel loading, depth and continuity reduction treatments, and silvicultural practices to change tree crown structure (e.g. thinning and pruning), as well as the creation of infrastructures and safety areas to facilitate fire suppression activities (e.g. road networks, fire breaks and water sources) (e.g. Fernandes and Botelho 2003; Molina *et al.* 2011; Corona *et al.* 2015; Salis *et al.* 2016).

One approach to predict and prioritise areas for treatment is using a risk-based framework where fire simulation models are coupled with erosion models to predict high-impact areas. Fuel management can then be modelled to determine the net benefit of treatments (Elliot *et al.* 2016; Sidman *et al.* 2016). For instance, Miller *et al.* (2011) estimated burn severity and post-fire ground cover with the First Order Fire Effects Model (FOFEM), and then applied the Geospatial Water Erosion Prediction Project (GeoWEPP) model for predicting post-fire erosion in the western USA. Scott *et al.* (2012) combined geospatial analysis, large-fire simulations with the Fire SIMulation system (FSim) and burn probability modelling to examine pixel-based measures of wildfire hazard and watershed exposure with the aim of identifying watersheds that were likely to burn at high intensity, which could be used to inform mitigation and prioritisation efforts in the Beaverhead–Deerlodge National Forest in Montana (USA). Thompson *et al.* (2013) generated spatially resolved estimates of wildfire likelihood and intensity with FSim, and coupled that information with spatial data on watershed location and erosion potential to quantify watershed exposure and risk on National Forest System lands in the Rocky Mountain region in the USA. Sidman *et al.* (2016) modelled fire severity in the Bryce Canyon National Park in Utah (USA) with FuelCalc, FlamMap and FOFEM, and post-fire hydrology and erosion effects with the KINEROS 2 model. Elliot *et al.* (2016) coupled FlamMap and FSim to predict respectively burn severity and probability in a study area in California (USA), and then performed GeoWEPP simulations to estimate sediment yields for undisturbed, burned and managed hillslopes and to evaluate the costs of fuel treatments to reduce fire severity. Elliot and Miller (2017) used FlamMap in Idaho (USA) for predicting burn severity and GeoWEPP for modelling erosion from both wildfire and fuel management on treatment areas. Srivastava *et al.* (2018) combined FlamMap and Water Erosion Prediction Project (WEPP) to identify high-erosion-risk hillslopes following wildfire and to evaluate the effects of fuel treatments on the hydrological response of a watershed located in Washington (USA).

In the present paper, we build on previous work by examining the effects of a range of multi-objective fuel treatment strategies, as well as of post-fire erosion control treatments, on potential soil erosion risk in the Mediterranean Basin. The study area was a 68 000-ha fire-prone area located in north-eastern Sardinia, Italy. We used fire simulation models to estimate spatial variability in wildfire behaviour and intensity, and used these outputs to feed the Erosion Risk Management Tool (ERMiT, Robichaud 2007*a*, 2007*b*). The effects of soil burn severity, time since fire, vegetation type and recovery, slope steepness, and sediment delivery exceedance probability on sediment yields were investigated. We then examined the potential of three competing fuel treatment strategies that prioritised treatments: near wildland–urban interfaces (WUI), near roads (ROAD) or randomly located (RAND). We discuss the results in terms of efficient strategies for allocating scarce funding for fuel management or maximising benefits to watershed conditions.

Material and methods

Study area

The study area covers ~68 000 ha of land and is located in north-eastern Sardinia, Italy (Fig. 1). The topography of the area is complex: terrain elevation ranges from ~45 to ~1350 m above sea level (a.s.l.), with several hills and low mountains (Fig. 1). The climate is Mediterranean, which is overall characterised by drought conditions from late May until September. The average annual precipitation is greater than 1000 mm at the highest elevations where summer storms are frequent, and ~650 mm in lower-elevation areas. The rainiest months are typically November and December. The mean annual temperature of the study area is ~13°C, with significant variations between mountain peaks and lowest areas (Chessa and Delitala 1997). The vegetation is largely characterised by the presence of shrublands and forests, which occupy ~46 000 ha of the study area (Fig. 1). Oak woodlands (*Quercus ilex* L. and *Quercus suber* L.) are the most important forest type in the study area. Conifer species are mainly represented by artificial plantations of *Pinus pinea* L., *Pinus pinaster* Aiton and *Pinus nigra* ss. *laricio* Poir, even though their presence is limited. High and dense Mediterranean maquis covers large parts of the study area, particularly in the hilly and mountainous areas of Monte Limbara, with *Erica arborea* L. and *Arbutus unedo* L.; grazed and degraded areas are characterised by greater presence of *Cistus monspeliensis* L., *Pistacia lentiscus* L. and low shrubs (Fig. 1). Anthropic areas cover ~1% of the study area and include the town of Tempio Pausania. Fruit-bearing areas, mostly sparse vineyards and olive groves, cover ~2300 ha located on flat terrain and near urban areas. Grasslands and agricultural areas are mainly herbaceous and horticultural production and characterise ~20% of the study area, particularly in the plains (Fig. 1).

Recent wildfire history from 1980 to 2010 indicates the study area experienced ~800 ignitions; wildfires smaller than 10 ha

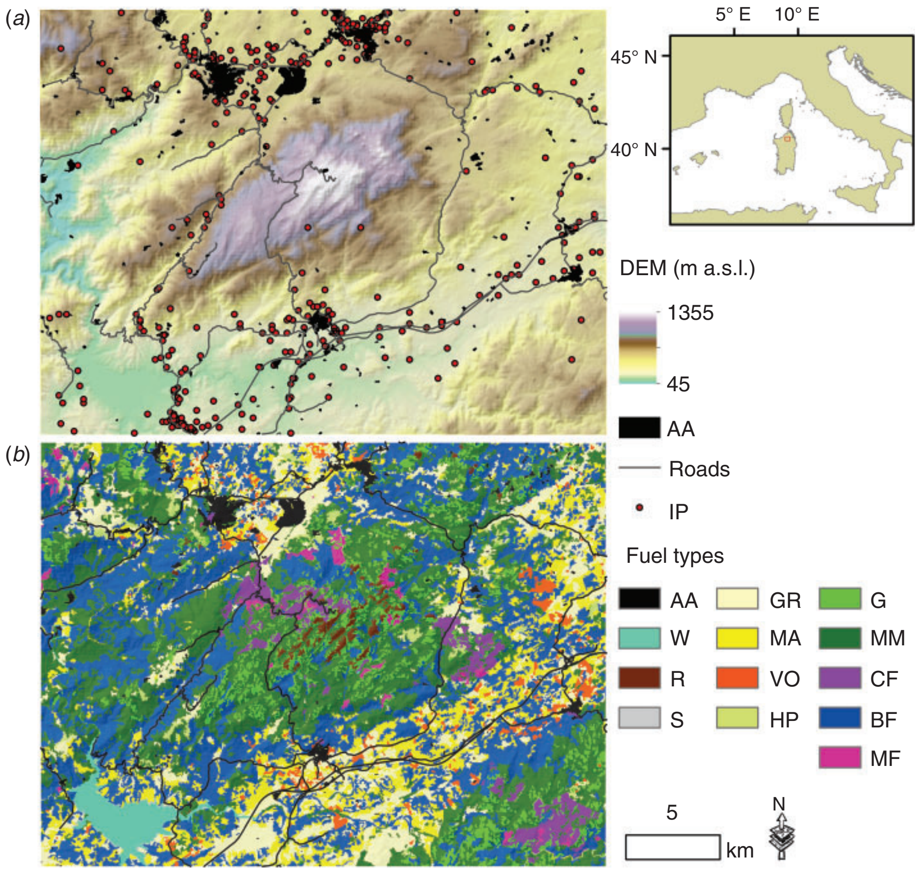

Fig. 1. Map of the study area, located in northern Sardinia, Italy. The topmost map (*a*) shows the terrain elevation (DEM, digital elevation model) of the study area, together with roads, anthropic areas (AA), and wildfire ignition points (IP) of the period 1980–2010. The bottom map (*b*) presents the main fuel types, as derived from the 2008 Sardinia land-use map (http://www.sardegnageoportale.it/). AA, anthropic areas; W, water bodies; R, rocks; S, sands; GR, grasslands; MA, mixed agricultural areas; VO, vineyards and orchards; HP, herbaceous pastures; G, garrigue; MM, Mediterranean maquis; CF, conifer forests; BF, broadleaf forests; MF, mixed forests.

comprised 95% of these ignitions, although the remaining wildfires were responsible for 90% of the total area burned. The largest wildfire was in 1983; it burned 18 000 ha near the town of Tempio Pausania and caused nine fatalities in the northern part of the study area. The majority of the ignitions were concentrated in the hottest months of the year (June to September); ~60% of the ignitions happened from mid-July to late August. The most common areas of ignitions were roads and the areas surrounding anthropic zones (Fig. 1).

Input data for wildfire modelling

To generate the gridded landscape file for FlamMap (Finney 2006), we assembled all input data at a 25-m resolution. The topographic input data (elevation, slope and aspect) were derived from a 10-m digital elevation model of Sardinia (www.sardegnageoportale.it/, accessed 30 July 2019). Surface and canopy fuels were interpreted from the 2008 Sardinian Land Use Map (www.sardegnageoportale.it/): we identified 13 fuel types, for which we associated standard or custom fuel models (Anderson 1982; Scott and Burgan 2005; Arca *et al.* 2009). As described in Salis *et al.* (2016), we used different models for forest fuels depending on elevation, using 600 m as a threshold. *Quercus suber* L. and *Quercus ilex* L. stands were used as reference to estimate canopy bulk density, canopy base height and canopy height (INFC 2005). Regarding fuels, we also generated three different fuel treatment scenarios carried out within the wildland–urban interface (WUI), near roads (ROAD), or randomly located (RAND) (Fig. 2). WUI and ROAD scenarios were obtained by the application of the *LTD* (*Landscape Treatment Designer*) spatial treatment optimisation software (Ager *et al.* 2013; Vogler *et al.* 2015).

Each fuel treatment scenario was performed for a total area of 10 000 ha (15% of the study area) (Fig. 2). The treatments modelled common fuel management operations including pruning of the lowest branches, and removal of dead fuels and part of the understorey for shrublands, forest understorey and herbaceous pastures (Sardinia Forest Agency, pers. comm. 2014). Fuel moisture content (FMC) for the 1- and 10-h-time lag dead fuels was estimated using historic moisture data above the 97th percentile, according to sampling campaigns carried out

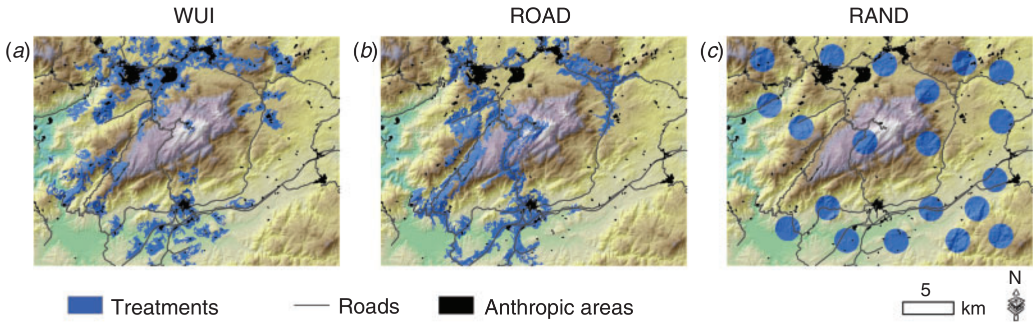

Fig. 2. Spatial location of the fuel treatments tested in this work. WUI, wildland–urban interface protection (*a*); ROAD, road protection (*b*); RAND, random location (*c*). The area treated for each of the above three strategies was 15% of the study area.

in Sardinia in previous years, as described in Pellizzaro *et al.* (2005, 2007) and Salis *et al.* (2015). Wind directions for wildfire simulations were NW and W, which characterised ~65% of days with wildfire occurrence, and S and SW directions, which are associated with the largest wildfires in Sardinia. We also used a fixed value of wind speed, 35 km h^{-1}, which corresponds to the 97^{th} percentile of historic conditions. Finally, we generated a smoothed fire ignition probability grid for the study area, using the 1980–2010 fire occurrence database. The ignition probability grid, which was held constant for all wildfire simulations, was created considering all observed fire ignitions, and using the inverse distance weighting algorithm (*ArcGIS 10.1* software) with a search distance of 1 km.

Wildfire simulation modelling

To simulate wildfire spread and behaviour in the study area, we used the minimum travel time (MTT) spread algorithm of Finney (2002) as implemented in Randig. The MTT uses Huygens' principle to simulate fire growth (Richards 1990; Finney 2002) considering both behaviour and growth modelled by vector or wave front (Finney 2002; Ager *et al.* 2010) and surface fire spread is predicted by the equation of Rothermel (1972). Crown fire initiation and spread are modelled respectively according to Van Wagner (1977), as implemented by Scott and Reinhardt (2001), and Rothermel (1991). The MTT algorithm is widely used in Mediterranean areas to assess wildfire exposure and risk and to target fuel treatments (Salis *et al.* 2013, 2016, 2018; Mitsopoulos *et al.* 2015; Alcasena *et al.* 2017, 2018, 2019; Kalabokidis *et al.* 2016; Oliveira *et al.* 2016; Palaiologou *et al.* 2018; Parisien *et al.* 2018). We simulated 25 000 wildfires for each fuel treatment scenario, including the untreated condition, using a reference resolution of 25 m, consistent with the input data. The ignition points were selected within the ignition probability grid developed from the historical database and burnable fuels of the study area. We considered constant fuel moisture and wind speed and a fixed burning period of 10 h for each wildfire simulated. Wind directions were NW, W, SW and S. The wildfire simulations generated a burn probability (BP) and a frequency distribution of flame lengths (FL) in 0.5-m classes for each pixel. BP measures the likelihood that a pixel will burn given an ignition in the study area. The distribution of FL values for each pixel was used to calculate the conditional flame length (CFL), which defines the probability weighted flame length if a fire occurs (Scott 2006).

Input data for erosion modelling

We obtained data on climate, soil characteristics, topography, land cover and potential soil burn severity in the study area, as needed for ERMiT simulations (Robichaud *et al.* 2007*a*). Climate parameter files for the study area were obtained with the ERMiT Rock:Clime tool (Elliot *et al.* 1999), using the integrated Rock: Clime web interface (https://forest.moscowfsl.wsu.edu/cgi-bin/fswepp/ermit/erm.pl, accessed 30 July 2019). This tool allows creating custom climate parameter files for a given area by providing monthly precipitation amount, monthly maximum and minimum temperatures, and monthly number of wet days in an existing climate parameter file. The tool generates a stochastic climate of the study area for 100 years, which accounts for year-to-year variability of storms and rain event patterns. ERMiT uses these data to generate a WEPP formatted stochastic daily weather data file, which includes: (1) daily precipitation amount, duration, time-to-peak and peak intensity; (2) minimum, maximum and dewpoint temperatures; (3) solar radiation; (4) wind velocity and direction. To estimate these values, we used the climate data of the Tempio Pausania weather station, as reported in Arrigoni (1968). The observed and stochastic weather data generated by ERMiT Rock:Clime are summarised in Table 1.

The rock content percentage and texture soil layers for the study area were derived from the Soil Map of Sardinia (Aru *et al.* 1990) and used to build the soil input files for WEPP.

To delineate watersheds and create the polygon terrain files (slope length, steepness and width) needed to run ERMiT, we clipped the 10-m digital elevation model (DEM) of Sardinia (http://www.sardegnageoportale.it/) to the study area and we then applied the Hillslope Delineation Toolbox (https://forest.moscowfsl.wsu.edu/fswepp/batch/HillslopeDelineationToolbox.html, accessed 30 July 2019). The hillslope horizontal length is composed of the three slope sections (top, middle and toe) and represents the length of the hillslope being modelled. These gradients are different percentages of the hillslope: top is the upper 10% by length, middle – the main portion – 80% by length, and toe is the steepness of the lowest 10%.

Land-cover data were obtained from the 2008 Sardinia land use map (http://www.sardegnageoportale.it/), and we then

Table 1. Climate data from the weather station of Tempio Pausania, as reported in Arrigoni (1968)

T_{max}, average maximum temperature (°C); T_{min}, average minimum temperature (°C); PP, total precipitation (mm). The average values of the stochastic climate variables provided by the ERMiT Rock:Clime tool are reported in parentheses

Month	T_{max}	T_{min}	PP	Rainy days (#)
Jan	8.5 (8.5)	3.6 (2.7)	99.1 (102.7)	9.53 (9.80)
Feb	9.1 (9.1)	3.6 (3.0)	101.1 (110.7)	9.73 (11.13)
Mar	12.2 (12.2)	5.5 (5.2)	86.1 (80.7)	8.30 (8.53)
Apr	15.3 (15.3)	7.6 (7.4)	80.0 (87.1)	7.70 (8.07)
May	19.5 (19.6)	10.8 (10.5)	57.9 (61.2)	5.60 (5.97)
Jun	24.2 (24.2)	14.3 (14.2)	20.1 (16.9)	1.93 (1.47)
Jul	27.6 (27.5)	17.4 (17.4)	7.1 (8.1)	0.67 (0.73)
Aug	27.2 (27.2)	17.9 (17.8)	19.1 (24.3)	1.83 (1.70)
Sep	24.1 (24.1)	15.5 (15.4)	61.0 (61.5)	5.87 (5.93)
Oct	18.4 (18.5)	11.6 (11.1)	98.0 (112.3)	9.44 (10.73)
Nov	13.3 (13.2)	8.0 (6.9)	115.1 (109.0)	11.07 (10.63)
Dec	9.9 (9.9)	5.1 (4.1)	118.1 (117.0)	11.36 (10.93)
	17.4 (17.4)	10.1 (9.6)	862.7 (891.6)	83.03 (85.62)

reclassified the land-cover data layer into ERMiT cover types (e.g. forest, chaparral, range). For this purpose, we created the following classes: (1) forests, in the areas classified as broadleaf, conifer and mixed forests (36% of the study area), which have a high canopy cover; (2) Mediterranean shrublands, in the areas classified as 'dense vegetation formation with the presence of shrub species as well as by small evergreen trees' (33% of the study area), and which have characteristics quite similar to California chaparral (Alexander 1999); (3) grasslands, in the areas covered by herbaceous pastures, *Quercus suber* L. dehesas (canopy cover between 5 and 25%), and other zones with herbaceous vegetation. Grasslands, particularly those located in the most complex and steep areas, are frequently grazed owing to the high presence of sheep and goats.

For modelling post-wildfire conditions, the CFL outputs before and after fuel treatment strategies were used to associate with each pixel a value of potential soil burn severity, which is a description of the impact of a fire on soil and litter (Robichaud *et al.* 2007*a*). Flame length as an indicator of burn severity has previously been used in other work (Elliot 2013; Elliot *et al.* 2016; Srivastava *et al.* 2018). CFL data allowed discrimination between areas characterised by different levels of potential soil burn severity should a wildfire occur. For this purpose, as proposed by Andrews and Rothermel (1982), we identified four classes of fire intensity, from unburned to high, which were used as reference for discriminating four soil burn severity classes (Fig. 3): CFL <0.01 m (unburned); 0.01 ÷ 1.2 m (low burn severity); 1.21 ÷ 2.4 m (moderate burn severity); ≥2.41 m (high burn severity). We integrated CFL pixel values from Randig for each hillslope with the severity class breaks previously defined.

Post-fire erosion modelling

Post-fire erosion was simulated using ERMiT (Robichaud *et al.* 2007*a*), which is a probability-based risk assessment tool for quantifying post-fire disturbance erosion modelling and evaluating rehabilitation treatment effectiveness. ERMiT provides probabilistic estimates of single-storm post-fire hillslope erosion by incorporating variability in rainfall characteristics, topography, land cover, soil burn severity and soil characteristics into each prediction (Robichaud *et al.* 2007*a*). ERMiT uses WEPP technology as the runoff and erosion calculation engine. WEPP is a process-based model that predicts runoff and sediment yields and simulates both inter-rill and rill erosion processes (Flanagan and Nearing 1995). It incorporates the processes of evapotranspiration, water infiltration and runoff, soil detachment, sediment transport and sediment deposition to predict soil runoff and erosion at the hillslope scale (Flanagan and Livingston 1995; Elliot *et al.* 2016). As previously reported, ERMiT needs five main input data types: (a) climate parameters, which are created through Rock:Clime (Elliot *et al.* 1999, 2004); (b) vegetation types (forest, range, chaparral); (c) soil types and rock content; (d) topography (slope length and gradient); and (e) soil burn severity classes (unburned, low, moderate and high). The general process by which ERMiT incorporates parameter variability is to: (1) determine the range of possible parameter values; (2) select representative values from the range; and (3) assign an 'occurrence probability' to each selected value such that the sum of assigned occurrence probabilities adds up to 100% (Robichaud *et al.* 2007*a*, 2007*b*). All simulations were performed using the ERMiT Batch interface spreadsheet (https://forest.moscowfsl.wsu.edu/fswepp/batch/bERMiT.html, accessed 30 July 2019), and the input data for b-e were prepared in a Geographic Information System (GIS) environment. ERMiT sorts the results to determine the exceedance probability of a given sediment delivery, as described in Robichaud *et al.* (2007*a*). The ERMiT Batch produced sediment delivery files that were linked to spatial maps in order to produce sediment delivery maps for each of the fuel treatment strategies analysed.

Modelling fuel reduction effects on post-fire sediment delivery

To analyse the benefits of fuel reduction treatments in the study area, sediment delivery was modelled considering the following conditions: (1) current fuel conditions in the absence of wildfire disturbance; (2) current fuel conditions in the presence of wildfire disturbance; (3) wildfire disturbance after different spatial

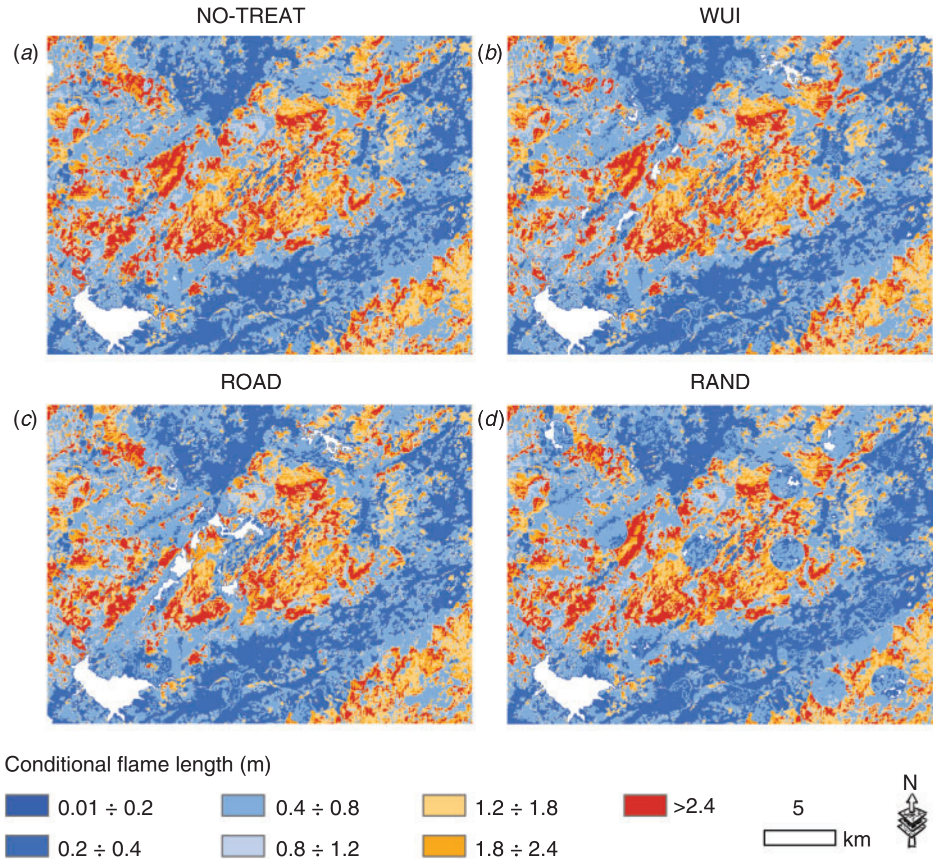

Fig. 3. Conditional flame length, as obtained from fire spread simulations, for the study area considering current fuel conditions (*a*), and WUI (wildland–urban interface) (*b*), ROAD (road protection) (*c*) and RAND (random location) (*d*) fuel treatment strategies applied for 15% of the study area. These maps were used to derive the soil burn severity classes, as described in the methods.

strategies of fuel management and post-fire erosion reduction treatments.

We then tested the effects of different factors in the post-fire sediment delivery for the whole study area. These factors included: (i) sediment delivery exceedance probabilities (from 1 to 95%); (ii) two different post-fire conditions, untreated and a seeding treatment strategy, which is a common post-fire treatment in the Mediterranean Basin; (iii) the number of years (from 1 to 5) after the wildfire events; (iv) the three land cover types (range, chaparral and forest); (v) the three slope steepness classes (below 10°, from 10° to 20°, above 20°); (vi) the four soil burn severity categories (unburned, low, moderate and high).

Results and discussion

Post-fire erosion for current vegetation conditions

The sediment delivery, both in terms of average (weighted by the area of each burned hillslope) and maximum (maximum of all of the burned hillslopes) sediment delivery, and the total area with potential erosion issues in the study areas decreased with increasing exceedance probability (Table 2). For instance, considering current fuel conditions and the absence of wildfire disturbance in the study area, the difference between 50 and 20% exceedance probability resulted in an increase of ~100% in average sediment yields, and of ~750% in maximum sediment delivery.

Considering current fuel conditions in the absence of wildfire disturbance and 50% exceedance probability, the simulated average sediment delivery in the study area was ~0.01 Mg ha^{-1}, and varied from 0 to a maximum of ~0.2 Mg ha^{-1} across all hillslopes (Table 2). In these conditions, the total area contributing to sediment delivery was ~11 900 ha out of 68 000 ha. Previous studies on soil erosion carried out in Sardinia also found relatively low values of sediment delivery in the absence of wildfire disturbances. For example, Acutis *et al.* (1996) measured mean erosion rates close to 0.02 Mg ha^{-1} year^{-1} in north-western Sardinia, whereas Rivoira *et al.* (1989) reported mean soil losses of ~0.03 Mg ha^{-1} year^{-1} in northern Sardinia. The steep and long sloping terrain areas showed the highest erosion rates in the absence of wildfire disturbance. The role played by terrain slope on soil erosion was highlighted in a previous study carried out in north-western Sardinia by Porqueddu *et al.* (2001) where soil loss data for diverse crops growing in hilly areas were evaluated during two experimental periods. They observed mean soil losses of 2.6 and 0.9 Mg ha^{-1} year^{-1}. Canu *et al.* (2015) measured post-fire sediment delivery in cork oak

Table 2. Effects of the sediment delivery exceedance probability on average and maximum sediment delivery, and on the total area with sediment delivery, considering current fuel conditions, in the absence of wildfire (no fire, NF) and 1 year after the wildfire disturbances (post-fire, PF)

Exceedence probability	Average sediment delivery (Mg ha^{-1})		Max sediment delivery (Mg ha^{-1})		Total area with sediment delivery (ha)	
	NF	PF	NF	PF	NF	PF
1%	2.8	20.3	51.1	151.1	41 327	50 301
3%	1.6	15.2	34.3	115.2	37 980	41 194
5%	0.8	13.1	23.9	98.0	37 595	37 843
10%	0.2	9.8	9.7	75.8	34 624	36 188
20%	0.0	6.4	1.5	57.0	27 842	35 633
30%	0.0	4.3	0.9	46.0	22 726	35 275
40%	0.0	2.5	0.7	38.3	13 864	34 508
50%	0.0	1.7	0.2	32.1	11 904	33 463
60%	0.0	0.9	0.1	29.8	10 589	31 930
70%	0.0	0.6	0.0	25.2	9522	29 572
80%	0.0	0.3	0.0	16.7	9511	26 149
90%	0.0	0.1	0.0	9.8	4765	19 102
95%	0.0	0.0	0.0	6.0	1286	4858

areas of NW Sardinia in the range 0.05–0.9 Mg ha^{-1}, with average values below 0.1 Mg ha^{-1} 3 years after a fire. Overall, the values obtained in our study for unburned conditions are similar to those reported by Cerdan *et al.* (2010) who reported mean soil erosion in Mediterranean Europe amounting to ~1.2 Mg ha^{-1} year^{-1} for the whole CORINE land cover area (https://land.copernicus.eu/pan-european/corine-land-cover, accessed 30 July 2019).

Fire occurrence has significant effects on the increase of sediment delivery compared with unburned conditions (Table 2), even after moderate fires (Gimeno-García *et al.* 2000; Keeley 2009; Stoof *et al.* 2015; Vieira *et al.* 2015). We found that in post-wildfire simulations with current fuel conditions, 80% of sediment delivery was generated by only 18% of the hillslope of the study area at an exceedance probability of 80% (Fig. 4). At lower exceedance probabilities (50 and 20%), hillslope area that contributed ~80% of the sediment yields covered 23 and 25% of the area. High post-fire soil erosion rates are frequently related to extreme weather, and particularly to intense rainfall events (De Luis *et al.* 2003; Mayor *et al.* 2007; Badía and Martí 2008). In fact, infrequent but intense rainstorms can cause high runoff and soil losses within short periods, as observed in several studies (Moody and Martin 2001; Cannon *et al.* 2011; Hosseini *et al.* 2016).

The application of ERMiT allowed examining how the distribution of runoff rates affected sediment yield exceedance probabilities in the post-fire conditions. In our study area, the average estimated sediment delivery was strongly influenced by the exceedance probability in terms of both spatial variation and absolute sediment delivery (Fig. 5). The highest values of sediment delivery were observed in the steepest areas with the lowest exceedance probabilities. For instance, at 20% exceedance probability, only ~10% of the landscape exhibited sediment yields greater than 24 Mg ha^{-1} in the first year after wildfire (Fig. 5). The increase in exceedance probability resulted in a reduction of areas that had sediment yields. The fact that low rainfall rates after the fire pose only limited problems of soil erosion was also found in previous studies (Moody *et al.* 2013; Haas *et al.* 2017).

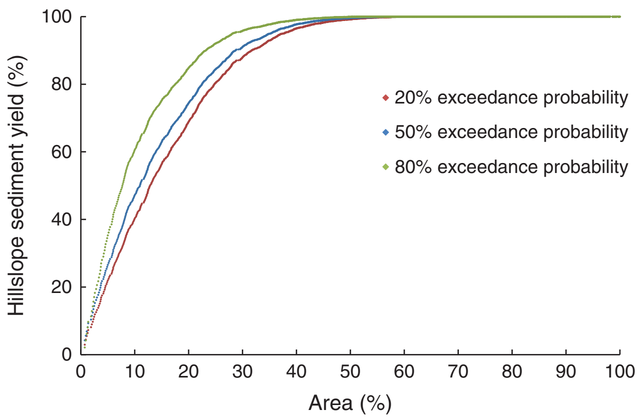

Fig. 4. Cumulative area and simulated average annual sediment yield, considering the first year after the fire and current vegetation and three sediment delivery exceedance probabilities (20, 50 and 80%).

The fire effects were particularly significant in the first years after fire (Fig. 6). The fact that the highest impacts in terms of post-fire erosion are generally observed in the first year after fire has been confirmed by others (Shakesby 2011; Hosseini *et al.* 2016). In our study, for instance, focusing on the 50% exceedance probability, the average sediment delivery in the study area ranged from 1.67 Mg ha^{-1} the first year after fire to 0.04 Mg ha^{-1} the fifth year after the event. Specific areas located on the steepest slopes showed peaks above 25 Mg ha^{-1}, with 50% exceedance probability. When taking into consideration 20 and 80% exceedance probability, the average sediment delivery was 6.4 and 0.3 Mg ha^{-1} respectively the first year after fire, and 0.6 and 0.01 Mg ha^{-1} the fifth year after fire. These values are similar with those reported in previous work that focused on the Mediterranean basin. For instance, Shakesby (2011) reported mean post-wildfire erosion rates (measured on field plots) 1 year after fire equal to 0.4 Mg ha^{-1} for low-severity, 3.3 Mg ha^{-1} for moderate-severity and 10.8 Mg ha^{-1} for high-severity fires.

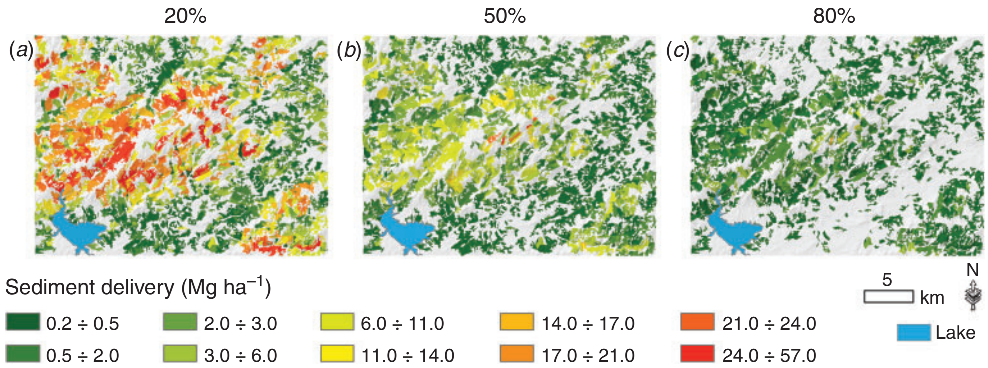

Fig. 5. Sediment delivery at the landscape scale, considering the first year after the fire with current fuel conditions and an exceedance probability of 20% (left), 50% (middle), and 80% (right).

Pausas *et al.* (2008) indicated that post-fire erosion rates measured in the Mediterranean Basin are rarely higher than 10 Mg ha^{-1} and are often lower than 1 Mg ha^{-1} in the first year after fire. Other studies have reported relatively low erosion rates in Mediterranean Basin environments (Imeson *et al.* 1992; Kutiel and Inbar 1993; Lavee *et al.* 1995; Rubio *et al.* 1997). The relatively low post-wildfire erosion rates in Mediterranean areas compared with other European areas was confirmed by Cerdan *et al.* (2010): they attributed this difference to the stoniness and thinness of Mediterranean soils (Shakesby 2011). However, sediment delivery rates above 10 Mg ha^{-1} the first year after the fire were reported by Soto and Díaz-Fierros (1998) in Galicia (Spain), Úbeda and Sala (1996) and Marquès and Mora (1992) in Catalonia (Spain), Lavabre and Martin (1997) in southern France, and Dimitrakopoulos and Seilopoulos (2002) in Greece. Field measurements of annual erosion rates following wildfires in other areas reported higher sediment delivery than in the Mediterranean Basin, particularly in the US (Robichaud *et al.* 2013; Elliot *et al.* 2016). For instance, post-fire erosion rates from the Cannon Fire in California (USA) ranged from 2.5 to 15 Mg ha^{-1} (Robichaud *et al.* 2008), whereas erosion rates measured following wildfires in the Sierra Nevada mountains were 46 Mg ha^{-1} in the Cedar Fire (Robichaud *et al.* 2013).

The effects of fire on estimated sediment delivery became less significant 4 years after fire (Fig. 6); the fifth year after the fire, as programmed in ERMiT, there were no differences between burned and unburned sediment yields. In effect, the regeneration of the burned vegetation, which progressively tends to return to values typical of pre-burning conditions, typically occurs within 5 years after fire disturbances (Fox *et al.* 2006; Robichaud *et al.* 2007*a*; Malkinson *et al.* 2011; Shakesby 2011). Nonetheless, others have noted that the erosion responses of burned areas last for less than 7 years, and depend not only on vegetation recovery, but also on post-fire weather, sediment availability, morphology and burn severity (Moody and Martin 2001; Gartner *et al.* 2004; Shakesby *et al.* 2007; Sheridan *et al.* 2007; MacDonald and Larsen 2009; Cannon *et al.* 2010; Moody *et al.* 2013; Vieira *et al.* 2015).

The role played by soil burn severity (SBS) classes in determining post-fire erosion at the landscape scale in the first 2 years after wildfire was characterised by highest sediment

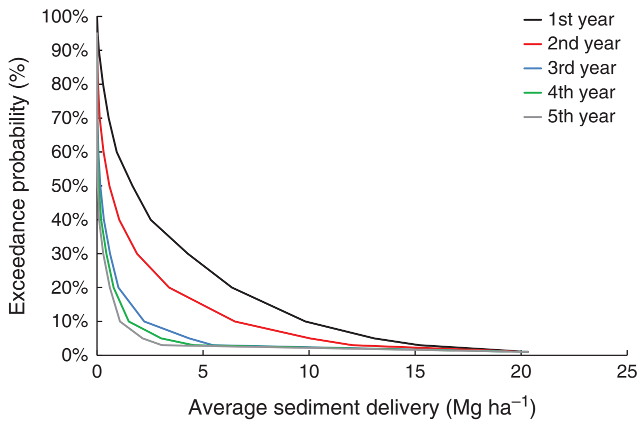

Fig. 6. Exceedance probability *v.* average sediment delivery at the landscape scale, for different years after the fire, considering current fuel conditions.

yields corresponding to high soil burn severity (Fig. 7*a*). Sediment yields were on average 3.1 Mg ha^{-1} in the first year after fire, and 1.3 Mg ha^{-1} in the second year after fire using the 50% sediment delivery exceeding probability. In the hillslopes with lower SBS values, the values were on average 1.5 and 0.4 Mg ha^{-1} for moderate SBS, and 0.08 and 0.04 Mg ha^{-1} for low SBS respectively. Therefore, the magnitude of sediment delivery from high-severity burn hillslopes was ~2 times greater than from moderate-severity-burn and 38 times greater than from low-severity-burn hillslopes. Previous work from Galicia, Spain, reported sediment delivery rates 1 year after the fire of 12.4 and 4.9 Mg ha^{-1} on high-severity and low-severity plots respectively, whereas the erosion measured in the control plot was ~2.0 Mg ha^{-1} (Soto and Díaz-Fierros 1998). Gimeno-García *et al.* (2000), using experimental fires in Mediterranean shrublands, observed that 1-year erosion rates were low (<0.1 Mg ha^{-1} year^{-1}) under unburned conditions, whereas soil losses became significant after a fire, and increased with fire severity (2.3 and 2.9 Mg ha^{-1} year^{-1} in moderate- and high-severity fires). Vega *et al.* (2005) analysed the first-year erosion effect of two different prescribed burning treatments in shrublands of Galicia, Spain; the most intense burning caused greater

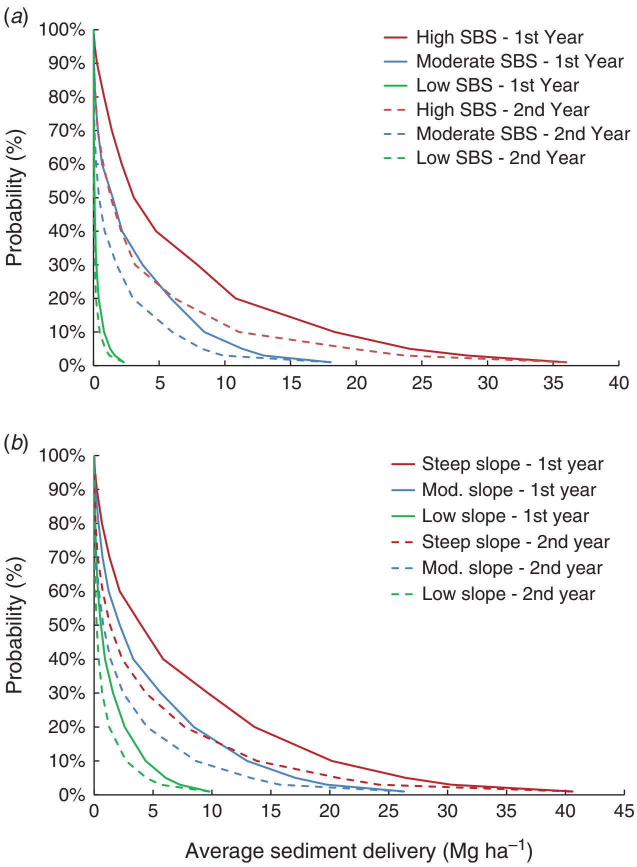

Fig. 7. (*a*) Exceedance probability *v.* average sediment delivery at the landscape scale, for the first 2 years after the fire and the three soil burn severity (SBS) classes considering the current fuel conditions; (*b*) exceedance probability *v.* average sediment delivery at the landscape scale, for the first 2 years after the fire and three terrain slope classes (low, <10°; moderate, 10–20°, steep, >20°) considering the current fuel conditions.

soil erosion compared with unburned areas (0.56 Mg *v.* 0.096 Mg ha^{-1} year^{-1}).

As expected, the post-fire erosion process was also affected by terrain slope; sediment delivery rates increased as the steepness of the terrain increased. This was observed for the different years after fire and exceedance probabilities (Fig. 7*b*). Focusing on 50% exceedance probability, the average sediment delivery rate for the first year after fire decreased from 4.0 Mg ha^{-1} for steep slopes to 2.2 Mg ha^{-1} for moderate slopes to 0.6 Mg ha^{-1} for low slopes. The strong role played by slope steepness in sediment delivery rates has also been highlighted in previous work (Pelletier and Orem 2014; DeLong *et al.* 2018). In addition, Marquès and Mora (1992), Cerdà *et al.* (1995) and Pausas *et al.* (1999) reported that even the terrain aspect can affect post-fire sediment delivery, owing to quicker vegetation recovery and the higher presence of organic matter on north-facing slopes than on south-facing ones.

We observed the highest sediment delivery always occurred in areas covered by Mediterranean shrublands with an average sediment delivery of 2.5 Mg ha^{-1} in the first year, and of 0.05 Mg ha^{-1} in the fifth year after the fire for an exceedance

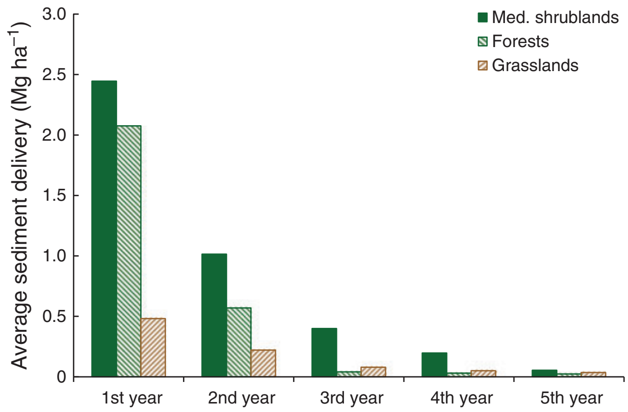

Fig. 8. Average sediment delivery for Mediterranean shrublands, forests and grasslands, at the landscape scale, considering the first 5 years after the fire with current fuel conditions and a 50% exceedance probability.

probability of 50% (Fig. 8). This can be partially explained by the fact that shrubs mostly cover steep terrain areas and are very limited in flat areas and plains. In contrast, grasslands presented the lowest average sediment delivery rates at the landscape scale: post-fire erosion ranged from an average of 0.5 Mg ha^{-1} immediately after fire to 0.04 Mg ha^{-1} during the fifth year after fire with 50% exceedance probability (Fig. 8). Forest vegetation types showed sediment delivery values not far from those of chaparral the first year after fire, then post-fire erosion was more limited, particularly after the third year after fire: in fact, the average sediment delivery for forests was 0.04 Mg ha^{-1}, the lowest among fuel types, reaching 0.02 Mg ha^{-1} the fifth year after fire. Our results are similar to data obtained by Vacca *et al.* (2000) in some burned sites located in southern Sardinia; the mean annual soil loss on burned herbaceous pastures was 0.06 Mg ha^{-1}, whereas soil losses on slopes covered with shrubs and eucalyptus were higher, 0.11 and 0.23 Mg ha^{-1} respectively. However, the high post-fire sediment delivery rates of shrublands and forests, particularly in mountains and hilly areas, were counterbalanced by the reduction in stream flow, soil erosion and transport due to the replacement of historical highly erosive cereal fields with dense shrubs and forests in the absence of fires (Beguería *et al.* 2003, 2006; Symeonakis *et al.* 2007; García-Ruiz 2010).

Finally, we tested the effects of seeding post-fire treatments on sediment yields for the study area. During the first year after the fire, ERMiT is programmed to indicate no benefit from seeding (Robichaud *et al.* 2007*a*). The assumption of no sediment yield reduction in the first year is reasonable for Sardinian conditions as well; the few measured data on burned plots in Sardinia highlighted relatively low differences between burned pastures and seeded areas, and reported the lowest sediment delivery in the burned plots (Rivoira *et al.* 1989; Porqueddu and Roggero 1994; Vacca *et al.* 2000). We found that post-fire seeding reduced erosion, particularly in the second year after fire, where we observed a maximum difference between seeding and untreated scenarios close to 22 Mg ha^{-1} with an exceedance probability of 20%. By the third year post-fire, the differences between seeding and no treatments reduced progressively. Variation in sediment delivery induced by post-fire treatments

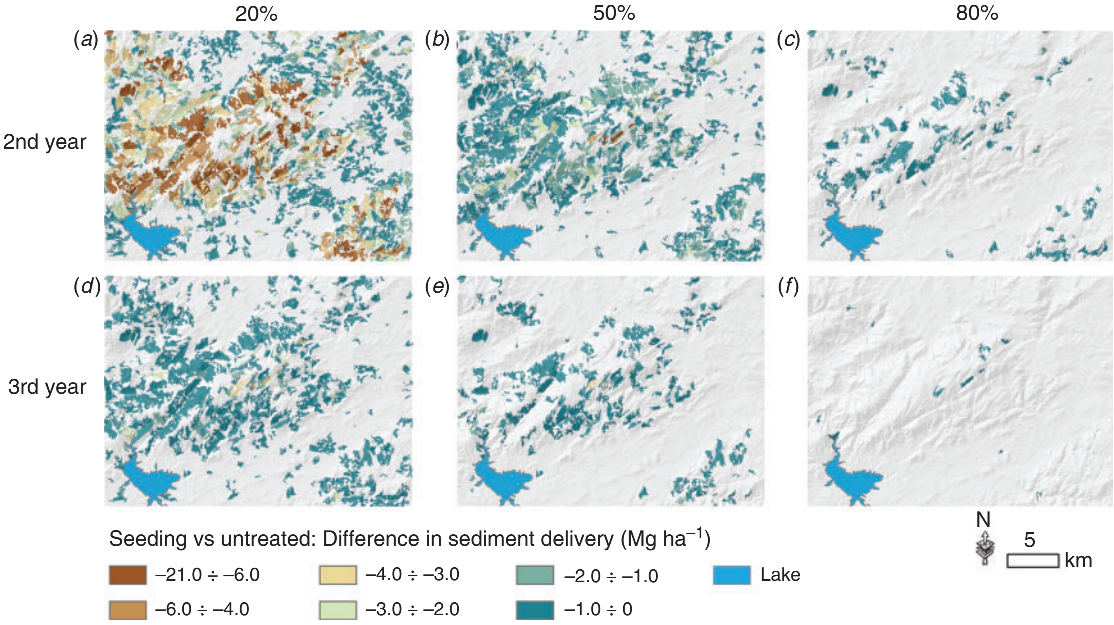

Fig. 9. Maps of the difference in sediment yields for the study area considering post-fire seeding treatments *v.* no treatments, focusing on the second and third years after the fire and exceedance probabilities of 20, 50 and 80%.

was higher when the probability exceedance was lower (Fig. 9). Previous research agrees that establishment of vegetative cover reduces erosion within the first year after fire (Robichaud *et al.* 2000; Beschta *et al.* 2004; Beyers 2004; Peppin *et al.* 2010; Rulli *et al.* 2012). Often the largest sediment delivery occurs before plant cover is established (Robichaud *et al.* 2000). The sediment yield was reduced by the third and subsequent years after fire (Peppin *et al.* 2010). However, seeding proved to be very effective in some cases and locations but not in others, often related to rainfall timing and amounts (Prats *et al.* 2014).

Post-fire erosion for the different fuel management and erosion treatment scenarios

The three simulated fuel treatments strategies, which were applied to a limited portion (15%) of the study area, had the main goal of reducing burn probability and fire severity. We observed positive effects on post-fire sediment delivery at both landscape scale and in the treated areas. At a 50% exceedance probability during the first post-fire year, the average sediment delivery at the landscape scale dropped from 1.7 Mg ha^{-1} of the current fuel conditions to 1.6 Mg ha^{-1} for the WUI strategy and to 1.5 Mg ha^{-1} for the ROAD treatment strategy, which was the most effective in reducing post-fire sediment yields (Fig. 10). Conversely, the RAND strategy was less effective than the other two in reducing average sediment delivery at the landscape scale the first year after fire (−0.07 Mg ha^{-1} with respect to current fuel conditions). The second year after fire, the average sediment delivery at the landscape scale was much lower than the previous year, and the differences between fuel treatment strategies and current vegetation were smaller; the best performance against post-fire erosion was obtained with the ROAD strategy (0.5 *v.* 0.6 Mg ha^{-1} of the current fuel conditions, 50% exceedance probability). The third year after fire, and in the

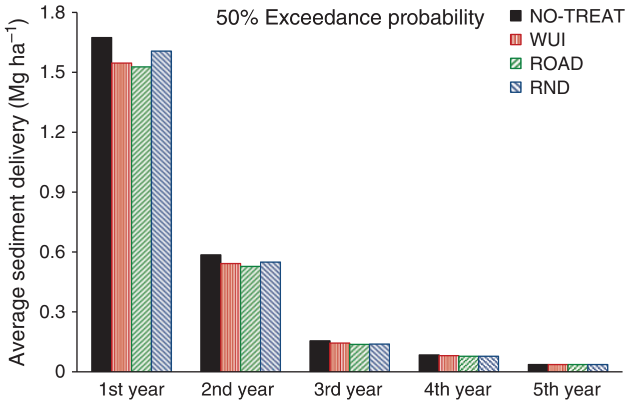

Fig. 10. Average sediment delivery at the landscape scale after the fire, from the first to the fifth year after the events, considering three different spatial fuel treatment strategies (WUI (wildland–urban interface), RAND (random location), and ROAD (road protection)) applied for 15% of the study area and current fuel conditions (NO-TREAT). The results refer to 50% sediment delivery exceedance probability.

following years, the differences among fuel treatment strategies and current fuel conditions were limited.

The limited reduction in estimated erosion from fuel treatments at the landscape scale can be related to the low percentage of area treated (15% of the landscape). However, as presented in Fig. 11, the local effect of fuel treatments on simulated sediment delivery is substantial and supports the need to optimise the locations of fuel treatments to maximise their effect on burn severity mitigation and post-fire soil erosion reduction at the landscape scale. Reducing fire severity with fuel treatments can also lower threats to several ecosystem services including the availability of clean water, recreation opportunities, timber production, and protecting habitats (Ager *et al.* 2007; Elliot *et al.* 2016). Small differences in post-fire soil erosion induced by fuel treatments were also reported by previous studies, most of which were carried out in the USA (Robichaud *et al.* 2010). Moreover, fuel treatments can reduce wildfire severity and resulting sediment after a wildfire, but also increase sediment when the treatment is implemented (Shakesby 1993; Harrison *et al.* 2016). Indeed, the presence of woody fuels, litter, or a continuous cover of surface fuels limits erosion by protecting the soil, reducing sediment yields, and increasing infiltration rates (Robichaud 2000). However, continuous and dense surface fuels also increase potential wildfire spread and intensity, when wildfire ultimately occurs (Silins *et al.* 2009; Harrison *et al.* 2016).

The spatial impact of fuel treatment strategies in reducing post-fire sediment yields with respect to NO-TREAT conditions (considering 50% exceedance probability) showed that the presence of fuel treatments was able to lower wildfire intensity for several years, and that this effect decreased from the first to the fifth year after the fire (Fig. 11). The location of fuel treatments and its effect on post-fire erosion by reducing fire severity were also highlighted by Elliot *et al.* (2016) and Srivastava *et al.* (2018) using a fire spread modelling approach.

After the first year following wildfire, the simulations confirmed the considerable effect of sediment delivery exceedance probability on average post-fire sediment yields at the landscape scale, and that overall, the ROAD fuel treatment strategy was the most effective among those tested for the various exceedance probabilities (Fig. 12). For instance, moving from 20 to 80% exceedance probability resulted in a decrease of the average sediment yield from 6.0 to 0.3 Mg ha^{-1} for the ROAD fuel treatment strategy. The increase in exceedance probability emphasised the differences among fuel treatment strategies and current fuel conditions in terms of post-fire sediment yields. For instance, the difference in average sediment delivery between the ROAD fuel treatment strategy and NO-TREAT was 0.3 Mg ha^{-1} (6.0 *v.* 6.3 Mg ha^{-1}) when considering 20% exceedance probability, whereas it decreased to 0.03 Mg ha^{-1} (0.25 *v.* 0.28 Mg ha^{-1}) with 80% exceedance probability (Fig. 12).

Overall, and mostly in the first years after fire, the reduction in sediment yields resulting from fuel treatments was quite notable, particularly for WUI and ROAD treatment strategies (Table 3). In fact, the first year after the wildfire events, the average sediment delivery in WUI areas dropped from 1.6 to 1.1 Mg ha^{-1}, and even in ROAD areas from 2.1 to 1.3 Mg ha^{-1}. The differences in sediment delivery between fuel treatment strategies and no-treatment condition were no longer significant 5 years after fire. However, the variation in sediment yields as affected by post-fire seeding was evident for the second, third and fourth year after fire (Table 3). Looking at the second year

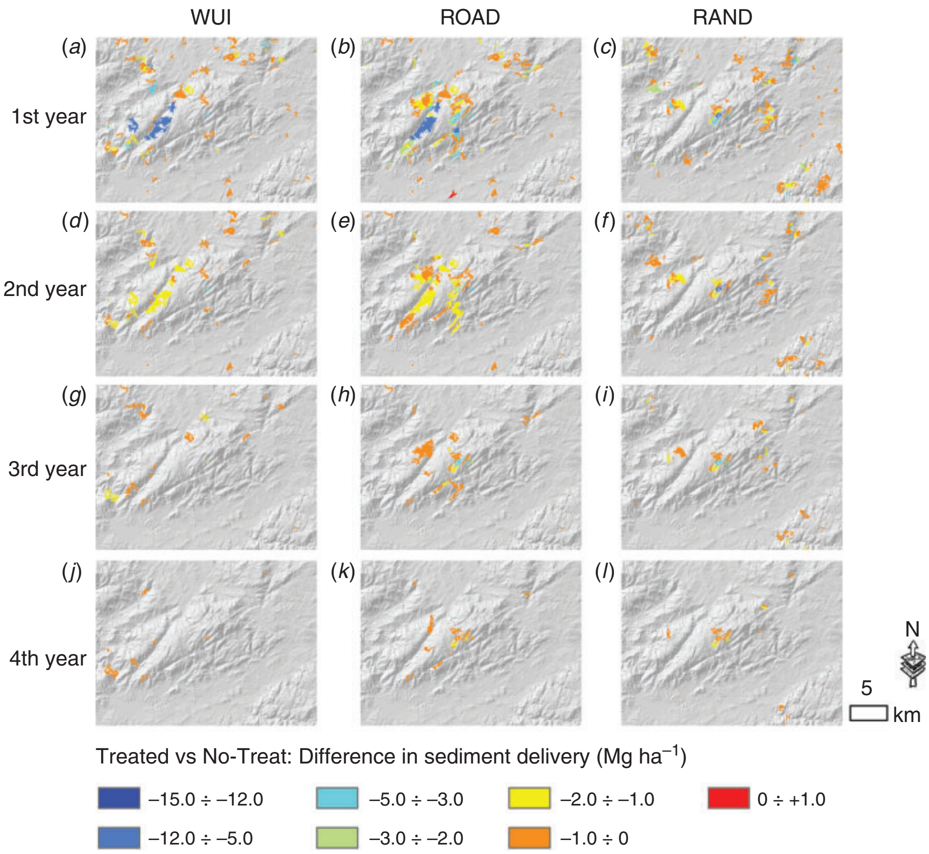

Fig. 11. Differences in sediment delivery between WUI (wildland–urban interface) (left), ROAD (road protection) (middle), and RAND (random location) (right) fuel treatment strategies and current fuel conditions (NO-TREAT) considering the first 4 years after the fire and a reference exceedance probability of 50%.

post fire, sediment delivery was more than halved after the application of seeding with respect to untreated conditions, and this was observed for all fuel management strategies.

Conclusions

The quantification of post-fire sediment delivery rates and the uncertainties when predicting future wildfire effects or fuel treatment scenarios pose serious challenges. In this work, we have demonstrated how fire spread and behaviour models can be used to identify areas with various levels of fire intensity, and therefore with different erosion potential, and can inform the evaluation of the effects of fuel management strategies on post-fire sediment yields and preserving ecosystem services. The post-fire erosion analysis was based on stochastic simulations and allowed proactive estimation and mapping of a range of possible pre- and post-fire soil sediment delivery events. Given the large variability in fire location, size and intensity and the complex interactions between landscape and wildfires, the proposed approach allows for spatial information on those areas characterised by high severity and burn probability that would have the largest impact on soil erosion after a fire event. Furthermore, the stochastic approach proposed offers a range of fire and soil erosion hazard metrics that are intuitive, easy to use and allow users to compare multiple wildfire and sediment delivery scenarios across large study areas. Findings from this study have significant implications for risk-based strategic management of fuels and land in the Mediterranean Basin as well as helping target more efficient fuel reduction treatments in those watersheds more exposed to severe wildfire events and erosion processes. Additionally, considering the limitations in budgets, time and specialised teams, identification of the watersheds that have the highest combined hazard can guide the prioritisation of areas where mitigation efforts can reduce the probability of post-fire erosion and sediment delivery. Using this information, policymakers, forest managers and local communities can more efficiently face the threats posed by fires and subsequent

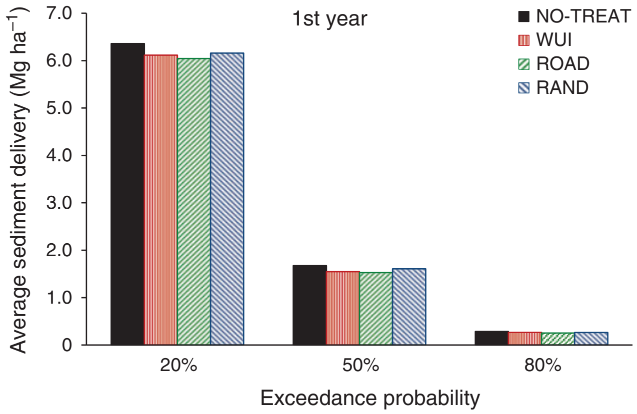

Fig. 12. Average sediment delivery at the landscape scale focusing on the first year after the fires with three sediment delivery exceedance probabilities (20, 50 and 80%) and three different spatial fuel treatment strategies (WUI (wildland–urban interface), RAND (random location), and ROAD (road protection)) applied for 15% of the study area plus current fuel conditions (NO-TREAT).

Table 3. Average sediment yields in the areas treated with the three fuel management strategies (WUI (wildland–urban interface), ROAD ((road protection) and RAND (random location)) and considering current fuel conditions (NT, No Treatments) in the respective treated areas
A 50% sediment delivery exceedance probability was set. Sediment yields vary depending on the year after fire and on the post-fire erosion strategy (untreated *v.* seeding (SEED)).

Post-fire timeframe	Fuel treatment strategy	Sediment yields (Mg ha^{-1})	Sediment yields, NT (Mg ha^{-1})	Sediment yields, SEED (Mg ha^{-1})	Sediment yields, NT-SEED (Mg ha^{-1})
		Post-fire erosion strategy: untreated		Post-fire erosion strategy: seeding	
1st year					
	WUI	1.1	1.6	1.1	1.6
	ROAD	1.3	2.1	1.3	2.1
	RAND	1.0	1.3	1.0	1.3
2nd year					
	WUI	0.4	0.5	0.2	0.3
	ROAD	0.4	0.7	0.2	0.4
	RAND	0.3	0.5	0.2	0.2
3rd year					
	WUI	0.1	0.1	0.1	0.1
	ROAD	0.1	0.2	0.1	0.1
	RAND	0.1	0.2	0.1	0.1
4th year					
	WUI	0.1	0.1	0.03	0.1
	ROAD	0.1	0.1	0.04	0.1
	RAND	0.1	0.1	0.04	0.1
5th year					
	WUI	0.03	0.03	0.03	0.03
	ROAD	0.04	0.04	0.04	0.04
	RAND	0.04	0.04	0.03	0.03

post-fire sediment delivery yields. This is particularly important with future climate change and the predicted increase in the occurrence of extreme weather events. Future work will focus on the evaluation of the economic trade-offs between fire severity reduction, erosion control and how these fuel treatment costs, in terms of erosion reduction, relate to the benefits of reduced future erosion.

Conflicts of interest

The authors declare no conflicts of interest.

Acknowledgements

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