UNIVERSITÀ DEGLI STUDI DI SASSARI

CORSO DI DOTTORATO DI RICERCA

Scienze Agrarie

Curriculum Agrometeorologia ed Ecofisiologia dei Sistemi Agrari e Forestali

Ciclo XXXI

Wildfire spread simulation modeling for risk assessment and management in Mediterranean areas

Liliana Del Giudice

Coordinatore del Corso	Prof. Ignazio Floris
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Abstract

Wildfires are a key problem in many terrestrial ecosystems, particularly in the Mediterranean Basin, and climate change will likely cause their increase in future years. Wildfire behavior simulator models are very useful to characterize wildfire risk, identify the valued resources more exposed to wildfires and to plan the best strategies to mitigate risk. In this work, we first carried out a review of wildfire spread and behavior modelling, and then focusing on FLAMMAP model. Then, we evaluated the effects of diverse strategies of fuel treatments on wildfire risk in an agro-pastoral area of the North-central Sardinia (Italy) that has been affected by the largest Sardinian wildfire of recent years (Bonorva wildfire, about 10,500 ha burned, on July 2009). Finally we analyzed the combined effects of fuel treatments and post-fire treatments with the aim to mitigate wildfire and erosion risk, linking the minimum travel time algorithm with the Ermit modeling approach in a study area located in Northern Sardinia (Italy), mostly classified as European Site of Community Importance. Overall, the results obtained showed that wildfire behavior simulator models can support forest fire management and planning and can provide key spatial information and data that can be helpful to policy makers and land managers.

General introduction

Wildfire is a key problem in many terrestrial ecosystem (Pyne et al. 1996; Pausas et al. 2009), particularly in Mediterranean ecosystems. In recent decades, Mediterranean areas suffered an increase in the occurrence of large fires (Moreno et al. 1998; Mouillot e Field 2005; Trigo et al. 2006; Viegas et al. 2006; Riaño et al. 2007; Costa Alcubierre et al., 2011; Alcasena et al., 2015). This intensification is due to various factors, firstly the increased frequency of extreme weather conditions, with hot temperatures, strong winds, low relative humidity and prolonged drought, that can even cause the lengthening of fire seasons (Trigo et al., 2006; Viegas et al., 2009; Koutsias et al., 2012; Pausas and Fernandez-Munoz, 2012; Cardil et al., 2013, 2014; Salis et al., 2014, 2016). Therefore, climate change can influence wildfire occurrence, particularly in conjunction with changes in land uses and consequent land cover variations. Indeed, abandonment of rural areas has promoted a considerable decline in the extent and management of agricultural areas, which have been progressively covered by natural vegetation (e.g. Mediterranean maquis), often highly flammable and capable of causing high intensity fires (Velez 2000; Badia *et al.* 2002; Pausas 2004; Bonet and Pausas, 2007; Castellnou e Miralles 2009; Ruiz-Mirazo et al., 2012); the increased pressure in coastal and urban areas caused an increase in the risk and in the number of fires in these zones (Pellizzaro et al., 2012; Alcasena et al., 2015; Salis et al., 2016). In addition, a considerable increase in fire suppression costs in the last period caused decreased investments in fuel management and fire prevention activities (Calkin et al., 2005; Stephens and Ruth, 2005; Prestemon et al., 2008; Hand et al., 2014).

An extensive and applied research on wildfire risk management is needed to mitigate the growing incidence of large fires impacting forests, urban interfaces and touristic areas. Simulation models and tools can help researchers to develop risk assessment and mitigation strategies: in particular, wildfire simulation models of the latest generation as FlamMap or Randig allow to work at fine resolutions. Different works based on the Minimum Travel Time (MTT) allowed to simulate thousands of fires (Ager et al. 2007, 2010) employing relatively reduced calculation times. The MTT calculates the fastest fire travel times along straight lines connecting cells in a grid. The MTT can be helpful to study the potential effects of climate change, land use change, vegetation succession and fuel management programs on wildfire behaviour and risk. Furthermore, we can assess other factors as carbon cycling (Ager et al., 2010b) or soil erosion (Robichaud et al., 2009) using a probabilistic framework able to quantify expected losses. For example, soil erosion models can be coupled with MTT methodologies to evaluate post-fire erosion, which affected several areas of Sardinia (Vacca et al. 2000; Canu et al. 2009).

Some studies have calibrated and validated wildfire simulator models in weather conditions and vegetation that characterize the Mediterranean areas (Arca et al. 2007; Salis et al. 2013, 2016). With the term risk we define the probability that something negative will happen. Wildfire risk can be obtained combining the likelihood that a wildfire occurs at a given intensity and the response in terms of losses which can be caused by that fire intensity. We can mitigate wildfire risk changing the excepted output, by reducing wildfire probability, wildfire intensity or the landscape response or susceptibility (Finney 2005; Ager, 2013; Finney, 2013; Scott et al. 2013). Many studies have examined the potential effect of fuel treatments on fire behaviour and risk by evaluating the potential variation in burn probability or fire intensity. Relevant studies have been implemented successfully in Canada and USA, with a limited application of this methodology in Mediterranean areas (e.g.: Finney, 2001, 2006; Finney et al., 2007; Ager et al., 2007, 2010, 2013; Miller et al., 2008; Moghaddas et al., 2010; Liu et al., 2013; Scott et al., 2013; Salis et al., 2016). The effects of fuel treatments on wildfire growth and behaviour depend both on the characteristics of fuel treatments as for instance: patterns (thickness, spacing and overlap), treatment intensity and size, wind speed and direction, and ignition patterns. Even after fuel treatments, it is basically impossible to eliminate wildfire risk because there are numerous unobstructed straight line wildfire paths, influenced by wind direction and ignition point locations (Finney, .2013? 2007?; Tehrune, 2013), and because fuel treatments typically cover small percentages of a study area.

The aims of the following three chapters are:

1) To review state of the art of wildfire spread and behaviour modelling, focusing on FLAMMAP, describing its principal applications.
2) To present a methodology that can be helpful to design and optimize fuel treatments in order to mitigate wildfire risk in an agro-pastoral area of the North-central Sardinia, Italy.
3) To link fire simulation modeling approach based on the application of the MTT algorithm (Finney 2002) with the Ermit modeling approach (Robichaud 2007) to

characterize post-fire erosion in Northern Sardinia, Italy, and to investigate the combined effects of fuel treatments with aim to reduce the wildfire probability and intensity, and post-fire treatments aimed to mitigate the erosion.

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Chapter 1: Introduction to wildfire behavior modeling and potential of the Minimum Travel Time (MTT) fire spread algorithm

1.1 Introduction

In the last years several works attempted to model the behavior of wildland fires and to simulate their spread across the landscape (Sullivan, 2009a; 2009b; 2009c). Wildfire behavior simulator models are very useful to help manager in environmental policy, to identify the risk factors in the landscape and to identify the valued resources and assets more exposed to fire. Furthermore, they are helpful to plan fuel treatments to mitigate fire risk (Scott et al., 2013; Calkin et al., 2011) and to support forest fire fighting and fire management (Guariso and Baracani, 2002)

There are different typologies of fire models, and several authors provided different classifications of them: for instance Sullivan (2009) suggested a classification based on the heat transfer and distinguished four main types of fire behavior models: physical, quasi-physical, quasi-empirical and empirical. Physical models are based on physical and chemical low (Albini, 1986; Balbi et al., 2007, 2010), empirical models don't consider physical and chemical theories but are based on observed data or experiments (McArthur, 1966), quasi-empirical models are based on physical and empirical models, and quasi-physical models only consider physical laws.

Overall, the fire behavior simulator models are characterized by a fire simulation technique, that describes the spreading of fire through the landscape. The difference between each fire simulation technique is the way in which the landscape and the spreading process are represented (Albright and Meisner, 1999). Among these techniques, we can mention the cellular technique, the elliptical wave propagation, the minimum travel time (MTT) (Finney, 2002), and the level set method (Rehm and McDermott, 2009).

A group of wildfire behavior simulator models, as for example FlamMap or Randig, can simulate thousands of fires in relatively limited calculation times, using the minimum travel time approach (Ager et al. 2007, 2010). The MTT calculates the fastest fire travel times along straight lines connecting cells in a landscape grid that represent the area under study. The MTT is able to predict fire behavior and can take into account different weather conditions and wind directions for the set of simulations defined in a given study area.

The aim of this work is to provide a brief review of fire behavior simulator models and to describe the principal applications of FlamMap MTT.

1.2 Fire behavior simulator models: classifications

1.2.1 Classification based on the heat transfer modeling

Wildfires are the result of combination of heat transfer across fuel due to combustion process and the related chemical reactions (Sullivan, 2009a). In the last years, several authors developed models with the aim to simulate wildfire behavior and spread.

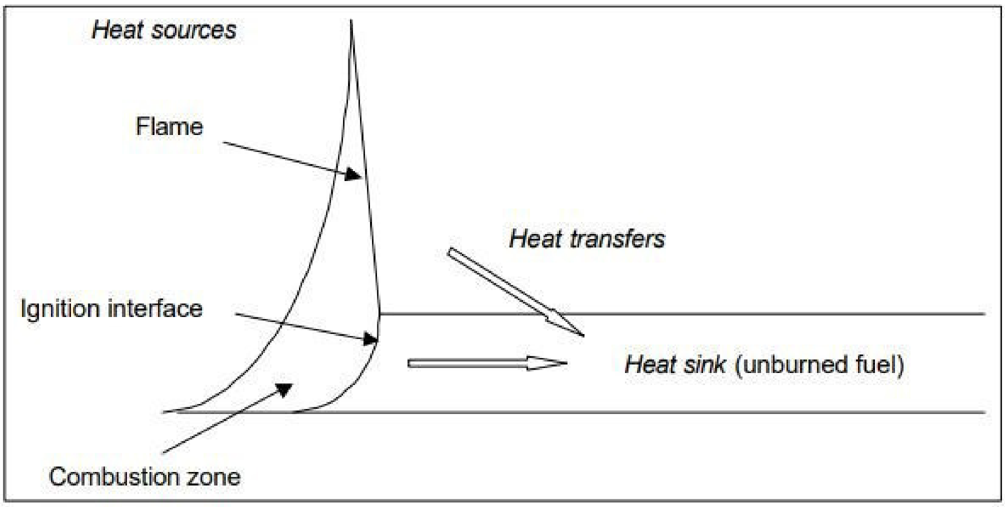

Fig. 1. Classical description on flame spread mechanism. Picture from: Dupuy et al., 1999.

These models consider the space in which there is the fire into heat source, that includes combustion zone, flame and ignition interface, and heat transfer, that comprises heat sink (unburned fuel) (Fig. 1) (Dupuy et al., 1999). The models can be classified into typologies that are different in complexity because there are physical models, that use physical and chemical theories involved in fire combustion to simulate fire behavior (Albini, 1986; Balbi et al., 2007, 2010), empirical models that don't consider physic and chemical theoretical laws and are based on observed data or experiments (McArthur, 1966), quasi-empirical models that are based on physical and empirical models, and finally quasi-physical models that only use physical laws. Several authors proposed diverse classifications of the fire behavior simulator models, among which Perry (1998), Pastor et al. (2003) and Sullivan (2009a, 2009b, 2009c), are the most complete.

1.2.1.1 Physical models

Physical models replicate a complex mix of chemical and physical laws generated by wildfire, in particular combustion process and transfer of energy to generate new ignition and heat transfer, without modeling of wildfire-atmosphere interaction (Arca et al., 2015; Salis, 2008; Mazzoleni and Giannino, 2014; Perry, 1998; Albright and

Meisner 1999; Pastor et al., 2003; Johnston et al., 2005) and it is fundamental to have details of heat sources in addition to spread factors (Dupuy et al., 1999). These models are based on known physical and chemical laws, so comparing different cases is very simple (Chandler et al., 1983; Salis, 2008) and this property facilitates their scaling (Arca et al., 2015).

The development of a physical model to model wildfire fire behaviour can be split into five steps, according to Grishin (1997): (1) To understand the physical phenomenon of wildland fire spread, particularly the transfer of energy from burned to unburned area. (2) Determination of coefficients and structural parameters, and identification of the most appropriate equations. (3) Choice of numerical solutions of the problem. (4) Checking of the model (numerical solution and system of equations). (5) Testing of model with the reality (Sullivan, 2009a).

These models do not consider the interactions between wildfire and atmosphere and use packing ratio, moisture content and the surface area to volume ratio as fuel characteristics. This can be interesting to study wildfire guided by heat transfer because heat flux and flame properties are considered fix (Weber, 1991).

Other physical models consider non-steady propagation (Weber, 1989), convection heat term (Albini, 1986) and temperature gradient inside the particles (Thomas, 1967).

In recent works, physical models include the degradation of the vegetation or the turbulent/reactive flow resulting from the mixing between the ambient gas and the pyrolizate using the computational fluid methods (Morvan and Dupuy, 2001, 2004).

1.2.1.2 Quasi-physical models

Quasi-physical models are based on physical laws such conservation of energy (heat), like physical models, but in this case the heat transfer is not quantified by chemistry, such as physical models, but often it is determinate using an empirical approach. This model needs to data of the flame geometry to close the system of equations. Another characteristic is that it may not be internally self-consistent (Sullivan, 2009a).

1.2.1.3 Quasi-empirical models

These models combine physical and empirical models (Burrows, 1999a, 1999b; Catchpole and de Mestre, 1986; Marsden-Smedley and Catchpole 1995a, 1995b; Simeoni et al., 2001, 2003). In order to model fire behavior, physical theories about combustion and heat transfer are combined to models with statistical correlations obtained in laboratory from fire experiments (Albright and Meisner, 1999; Pastor et al., 2003; Salis 2008). Validation of these models is simpler than the physical ones and can be applied in situations different from laboratory (Pastor et al., 2003; Johnston et al., 2005).

The most used quasi-empirical model in Mediterranean area is the one proposed by Rothermel in 1972 (Pastor et al., 2003; Sullivan, 2009b) and was the basis of the National Fire Danger Rating System (Deeming et al. 1977; Burgan 1988) and the fire behaviour BEHAVE (Andrews 1986). This model is based on the physical model of Frandsen (1971) and on experimental data obtained by Rothermel and Anderson in 1966 using wind tunnel experiments in artificial fuel beds of varying characteristics and by McArthur in 1966 in Australia, with input variables and a range of wind speed conditions. After a preliminary calibration, this model is applicable to various contexts.

In Canada the most used quasi-empirical model is the Fire Behavior Prediction (FBP) System (Forestry Canada Fire Danger Group 1992), which is included in the Canadian Forest Fire Danger Rating System (CFFDRS) (VanWagner 1998; Taylor and Alexander 2006). The model was the result of a long research and many experiments and it is applicable in several countries such as Asia, Mexico and New Zealand. Almost 500 fires were used in the construction of the FBP system; 400 were field experiments, the remainder well-documented observations of prescribed and wildfires. The CFFDRS has been introduced and implemented in several countries, including New Zealand, Mexico and several countries of south-east Asia.

1.2.1.4 Empirical models

Statistical correlations can be extracted from experimental fires reproduced in laboratory or in the field, with the aim of determinate the characteristics of fire behavior (rate of spread, head fire, perimeter, etc.) and the characteristics of flames (Noble et al., 1980; Cheney and Gould, 1995, 1997; Cheney et al., 1998, Mc Arthur, 1966; Sullivan, 2009b). These characteristics can be obtained also observing fire with aim of hazard reduction or prescribed fires (Sullivan, 2009b). Empirical models describe wildfire features without considering the physical mechanisms which drive the fire process (Perry, 1998). These models are linked to experimental conditions, so it is difficult to apply them to other situations (Marsden-Smedley, 1993).

The development of a empirical model to model wildfire fire behaviour can distinguish into four steps (Sullivan, 2009b): (1) determination of physical and quantitative characteristics of the fuel and description of terrain data (slope, aspect, etc); (2) characterization of the atmospheric environment (wind, temperature, moisture, etc); (3) observation of fire and measurement of its variables (spread, flame, smoke, etc.); (4) finally statistical correlation between the variables.

The first empirical model was very simple and had the aim of evaluate wildfire spread linked to wind and slope, and so aid to plan suppression actions (Chandler et al. 1983). Given their simplicity the empirical models were one-dimensional and were accepted by wildland fire authorities because of own immediate use (Sullivan, 2009b).

In Australia the most used empirical model to predict fire spread were the McArthur Grassland (McArthur1965, 1966) and Forest (McArthur 1967) Fire Danger Rating Systems (FDRS), and the Forest Fire Behaviour Tables for Western Australia (Red Book) (Sneeuwjagt and Peet 1985. In recent years McArthur Grassland FDRS was replaced by CSIRO Grassland Fire Spread Meter (GSFM) (Cheney and Sullivan1997), which is based on empirical model of Cheney (1998) (Sullivan, 2009b) and was the first model which used experimental plots greater than or equal to 1 ha.

1.2.2 Other classification based on the modelled physical system

There are other classifications of fire models, based on the physical system studied in the simulations (Albright and Meisner, 1999; Pastor et al., 2003; Johnston et al., 2005). We can classify them into surface fire models, crown fire models, spotting models, and ground fire models.

1.2.2.1 Surface fire models

To simulate fire spreading through fuels contiguous to the ground, we can use these models. Surface fuels refer to fuels lower than 2 m in height, i.e. grass, shrubs, small trees. The final aim is to determine surface fire rate of spread, fire line intensity, flame height and fire length-to-width ratio, that are the most important fire characteristics. The Rothermel's model is the most used to evaluate surface fire behavior (Rothermel, 1972). Rothermel's model is based on law of the conservation of energy linked to radiative heat transfer and simulate rate of spread using environmental data of fuel, weather and topography.

1.2.2.2 Crown fire models

These models simulate fires that burn canopies. We can divide the crown fire predictive models into two groups: crown fire initiation models and crown fire spread models (Pastor et al., 2003). The first provides an analysis of surface to crown fire transition, the second characterizes crown fire behavior.

There are several studies that predict fire line and rate of spread conditions for passive, active and independent crown fire transition, for instance Van Wagner (1993), Rothermel (1991) and Dickinson et al. (2007). These models are largely used in spite of their empirical nature.

1.2.2.3 Spotting models

Spot fires originated from fire front can cause an independent new fire that needs a different modelling. These models are largely integrated into the Forest Service calculation system (Rothermel, 1983; Alexander et al., 2004). Overall, spot fires are modelled by few authors, including Albini (1979, 1981, 1983).

1.2.2.4 Ground fire models

Ground fires can cause many human and assets losses, they burn without flames and at very slow spread rates, but damage organic layer of the soil and heat the inorganic layer (Pastor et al., 2003). Probability of ground fire ignition was studied by Frandsen (1987) and Hartford (1989) with a series of experiments. Schneller and Frandsen (1998) and Frandsen (1998) carried out a study on heat flux in a ground fire.

1.3 Fire simulation techniques

Fire simulation techniques characterize each fire model simulator and describe the spreading of fire through the landscape. Each fire simulation technique is characterized by the way in which represent the landscape and the spreading process (Albright and Meisner, 1999). Four principal fire simulation techniques to simulate the spread of fire through the landscape can be distinguished: cellular technique, elliptical wave propagation technique, minimum travel time and level set method.

1.3.1 Cellular technique: cellular automata models

Cellular automata is a discrete model studied in various sectors, like mathematics, physics and biology and it is very used in the complex systems science (Karafyllidis and Thanailakis, 1997; Ohgai et al., 2007; Gao et al., 2008). The cellular automata model is a regular grid of cells, each with a set of possible values, such as fuel type, elevation, slope, etc., and an initial state before ignition (Albright and Meisner, 1999). This technique is very used because of its inherent parallelism, regularity and modularity (Cohen et al., 2002; Zeigler, 1987). How does the cellular automata models about fire work? There is a spread mechanism from a cell to his neighbors that is guided by group of cells with similar state, either active or inactive. This technique allows to perform a huge number of simulations, but we need to use techniques for high performance computing (Innocenti et al., 2009).

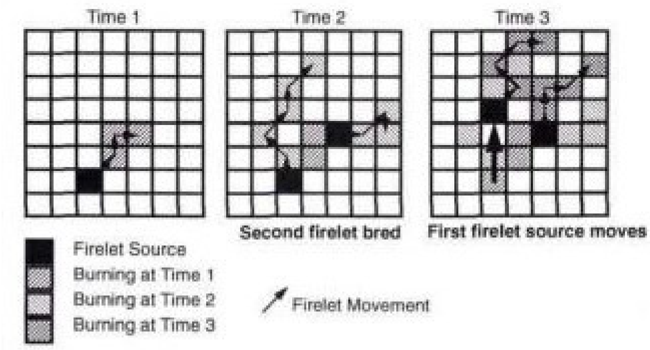

Fig. 2. Example of fire behavior simulation using cellular automata. Picture from: Clarke et al., 1994.

1.3.2 Elliptical wave propagation technique

The elliptical wave propagation technique is based on the Huygens' wavelet principle, that was originally proposed for the propagation of light waves and that can be applied to fire spread simulation (Sullivan, 2009c). The Huygens' wavelet principle states that each point on a fire perimeter is a theoretical source of a new fire. This new fire is characterized by the given fire spread model and the conditions of the location of the origin of the new fire. It is assumed that the new fires around the perimeter ignite simultaneously, and each new fire attains a certain size and shape. The outer surfaces of all the individual fires become the new fire perimeter for that time (Anderson et al., 1982; Sullivan 2009c; Arca et al., 2015). With this technique, we can simulate correctly wildfire with not heterogeneous environmental conditions (French, 1992). The model fire shape under uniform condition is the simple ellipse (Van Wagner, 1969) and other alternative and more complex ellipse shapes have been proposed by Dorrer (1993) and Wallace (1993). Several studies demonstrated the applications of the use of Huygen‘s principle to modelling surface fire growth, for instance Coleman and Sullivan (1996), and Richard and Brice (1995).

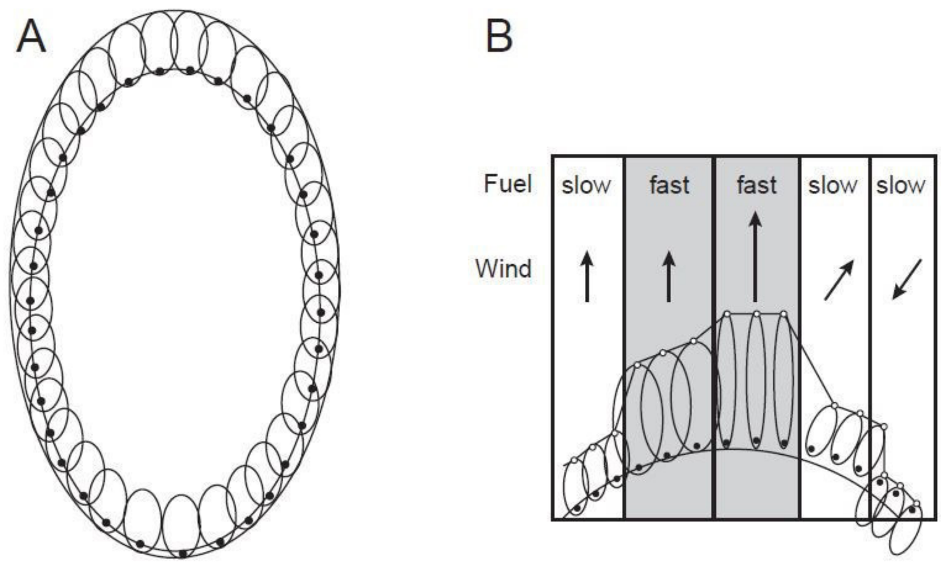

Fig. 3. Illustration of Huygens' principle using elliptical wavelets. (A) Uniform conditions use wavelets of constant shape and size to maintain the elliptical fire shape over time. (B) Nonuniform conditions showing the dependency of wavelet size on the local. Picture from: Finney, 1998

1.3.3 Minimum travel time

The minimum travel time algorithm was developed by Finney (2002). This technique calculates the fastest fire travel time along straight-line transects connecting nodes (cell corners) of the grid (Finney 2002, 2006). Fire-behavior values in the underlying cells of the grid are used to evaluate fire behavior and travel times along the line segments.

Fire behavior characteristics depend on different spatial data themes as major fire-spread direction, maximum spread rates and intensities, and the elliptical fire shape dimensions. The fire shape defines the Cartesian expansion rates of an elliptical fire along the ground surface (Anderson et al. 1982; Finney, 2002). The MTT considers all environmental conditions constant in time to calculate fire growth (Finney, 2006).

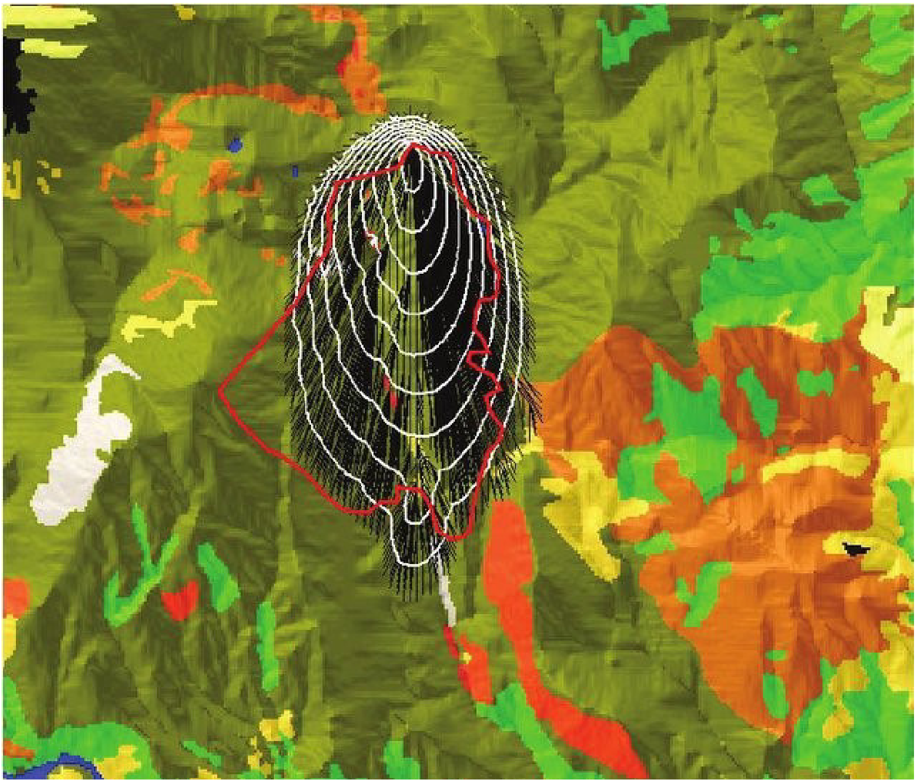

Fig. 4. Example of wildfire simulation using the MTT. Picture from: Opperman et al., 2006

1.3.4 Level set method

The level-set method is an efficient and versatile method used in recent years to describe wildfire propagation. The method describes the fire front as a discretized set of cells that expand at a given rate of spread and it does not require information on the shape of the fire front. This method calculates the fire spread using information of the fuel properties and environmental condition typical of landscape. This information is attributed to each cell of grid and determines the state of the cell (Chen et al., 2018).

The level-set method can calculate the normal vector to the fire front, models wind-aided fire spreads, and can merge separate fire fronts automatically and the ignition points naturally evolved into an elliptical form, according to the test conducted by Rehm and McDermott (Rehm and McDermott, 2009). The level-set method is very useful to be implemented and coupled with physical based models, considering that the same grid can be used by both models (Chen et al., 2018). The level-set method has been incorporated by many fire models such SWWS (Ghisu et al., 2014).

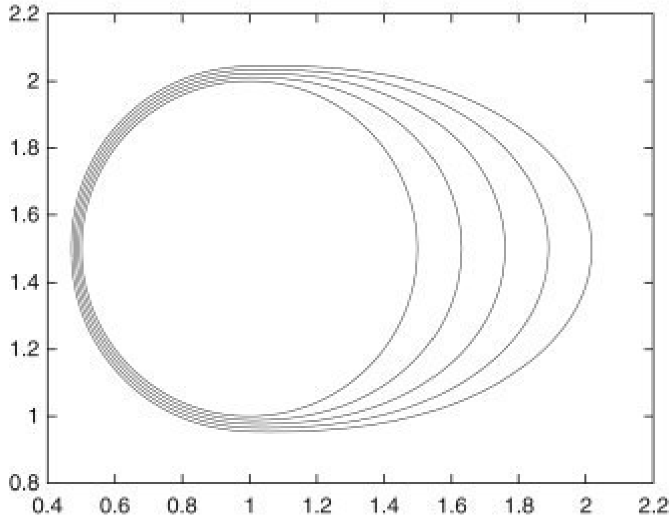

Fig. 5. Basic simulation of wildfire using Level set method. Picture from: Mallet et al., 2009

1.4 Main wildfire simulators

Wildfire simulators are built to be practical, easy to implement and to provide timely information on the progress of fire spread for wildland fire authorities (Sullivan, 2009c). There are several wildfire simulators: in this work we will only describe some of them.

In 1990, Green et al. developed IGNITE, a raster-based fire spread model that simulates fires in landscape with heterogeneous fuels. It utilizes fire spread models of McArthur (1966, 1967) as primary spread model and the propagation model of Green et al. (1983). The model is provided with a geographic information system that permits of importing and editing maps (Green at al., 1990).

Another program is FIREMAP, developed by Vasconcelos and Guertin in 1992 that permits to estimate wildfire behavior in discrete time steps and in spatially non-uniform environments. This simulation system uses Rothermel's behavior prediction model (Rothermel, 1972) as primary spread model and permits to show the outputs as digital maps (Vasconcelos and Guertin, 1992). It can simulate the fire suppression actions (Papadopoulos and Pavlidou, 2011)

FARSITE was developed by Finney in 90's (Finney, 1994) and is a two-dimensional fire spread simulation model. The base of FARSITE is the semi-empirical model of Rothermel (1972), and the spread of fire through the landscape technique is based on the Huygens' principle (Richards, 1990; Finney, 1998). The model requires 5 mandatory input layers such elevation, slope, aspect, fuel models, canopy cover, in the form of ASCII files. 3 other layers (crown base height, stand height, and crown bulk density) can be also included in the so-called Landscape file. Furthermore, the model needs the fuel bed characteristics and the fuel moisture. Fuel bed characteristics can be summarized using fuel models; the fuel models can be standardized (Anderson, 1982; Cruz, 2005; Scott and Burgan, 2005) or customized for some distinguishing vegetation type. FARSITE also requires weather and wind data.

In 1994 Coleman and Sullivan proposed a model called SiroFire. This program is a DOS protected mode application and could be applied to the grass and forest in Australia. It uses GIS-derived geographic maps and digital elevation models to simulate the fire behavior and provides graphical outputs. This model uses fire spread model of McArthur (1966, 1967) and of Cheney et al. (1998) as primary spread model and as propagation method Knight and Coleman (1993) (Coleman and Sullivan, 1996).

Karafyllidis and Thanailakis in 1997 model presented the Thrace model, that can simulate fire in homogeneous and inhomogeneous forests and incorporates weather conditions and land topography. The simulator was based on the Rothermel (1972) spread model (Karafyllidis and Thanailakis, 1997).

Pyrocart is a model implemented by Perry et al. in 1999 that simulates fire behavior. The Rothermel spread model was integrated with geographic information system (GIS) by PYROCART. It uses the propagation method of Green at al. (1983). This model was validated in a work of Perry at al. (1999) and the predictive accuracy of the model was estimated to be 80%. The model was dominated by slope and fuel (Perry et al., 1999).

FlamMap is a fire behavior model implemented by Finney in 2002. FlamMap uses the minimum travel time to simulate potential fire behavior characteristics, fire perimeters, and burn probabilities. We will examine in depth FlamMap in the 6th chapter.

Guariso e Baracani in 2002 presented a software system to simulate fires in Mediterranean area at small-scale called PdM. This model uses Valette's classes of flammability. The fuel is composed of two-layer cellular automata, one for the crowns of the trees and one for the surface coverage. In the model they introduce slope, air temperature and moisture data as coefficients following Rothermel's model and consider wind as Alexander's ellipse theory. The software can be a support for fire-operators with real-time data. Indeed, the simulator automatically retrieves GIS coverage and digital terrain model data, shows on the screen all the active operators by GPS positioning, saves and re-runs any step of the simulation to test different fire management actions (Guariso and Baracani, 2002).

Lopes et al. (2002) presented the software FireStation. This is a semi-empirical model that simulates fire in landscape with complex topography and provides as outputs fire rate of spread. The model requires topographic inputs, fuel characteristics and wind data. The primary spread model is Rothermel (1972). Fire shape is developed following an ellipse-type model. Wind field was simulated by two models that are based on local observations. The model has a graphical interface that help the user to use it (Lopes et al., 2002).

Table 1. Summary of simulation models published in the literature 1990–2007- Modified by Sullivan, 2009c

Model	Reference	Origin	Simulation type	Propagation type	Propagation method	Primary spread model
IGNITE	Green et al., 1990	Australia	Raster	Epidemic or template	Green et al., 1983	McArthur, 1967
FIREMAP	Vasconcelos and Guertin, 1992	Portugal	Raster	Minimum travel time	–	Rothermel, 1972
FARSITE	Finney, 1994	USA	Vector	Huygens	Richards, 1995	Rothermel, 1972
SiroFire	Coleman and Sullivan, 1996	Australia	Vector	Huygens	Knight and Coleman, 1993	McArthur, 1967 ; Cheney et al., 1998
Thrace	Karafyllidis and Thanailakis, 1997	Greece	Raster	Minimum travel time	–	Rothermel, 1972
PYROCART	Perry et al. 1999	New Zealand	Raster	Epidemic or template	Green et al., 1983	Rothermel, 1972
FlamMap	Finney, 2002	USA	Raster	Minimum travel time	–	Rothermel, 1972
SWWS	Ghisu et al., 2014	Italy		Level-set method	Richards, 1995	Rothermel, 1972
PdM	Guariso and Baracani, 2002	Italy	Raster	Minimum travel time	–	Rothermel, 1972

Table 1. Cont.

Model	Reference	Origin	Simulation type	Propagation type	Propagation method	Primary spread model
FireStation	Lopes et al., 2002	Portugal	Raster	Epidemic or template	–	Rothermel, 1972
Prometheus	CWFGM Steering Committee, 2004	Canada	Vector	Huygens	Richards, 1995	FCFDG, 1992
BehavePlus	Andrews, 2007	USA				Rothermel, 1972
ForeFire	Filippi et al., 2010	France		Discrete Event Simulation	–	Balbi et al., 2007
FSim	Finney et al., 2011	USA		Minimum travel time		Rothermel, 1972
Wildfire Analyst	Ramirez et al., 2011	Spain		Minimum travel time	–	Rothermel, 1972
SWWS	Ghisu et al., 2014	Italy		Level-set method	Richards, 1995	Rothermel, 1972

The Canadian Wildland Fire Growth Model Project Team (2004) developed Prometheus (Forestry Canada Fire Danger Group 1992), that is a deterministic wildland fire growth simulation model based on the Fire Weather Index (FWI) and Fire Behavior Prediction (FBP) sub-systems of the Canadian Forest Fire Danger Rating System (CFFDRS). Prometheus uses primary spread model of FCFDG and Richards (1995) as propagation method. The model simulates fire behavior and spread considering heterogeneous fuel, topography and weather conditions. It produces GIS compatible outputs.

The first version of BehavePlus was released in 2002 (Andrews, 2007). This program can be used for any fire management application and to calculate fire behavior. Its primary spread model is Rothermel (1972). It produces outputs such surface and crown fire rate of fire spread and intensity, probability of ignition, fire size, spotting distance, and tree mortality. The program needs type and moisture fuel data to simulate wildfire (Andrews, 2007).

ForeFire was developed by Filippi et al. (2010) to estimate fire spread and was based on the physical model of Balbi et al. (2007). A set of custom fuel models was also developed to calibrate the model. ForeFire is based on the Discrete Event Simulation method (DEVS) (Ziegler, 1987).

FSim is a fire simulation model that focuses on the simulation of large-fire, and was developed by Finney et al. in 2011. The model can simulate spatial and temporal variation in weather and fuel moisture with run of thousands years in order to capture rare fire events. It is composed of different modules, each one devoted to a component of wildfires: weather data, fire ignition, fire suppression and fire growth. The weather generation module simulates wind speed, wind direction, and fuel moistures by percentage of dry weight for six fuel categories, and generates a fire danger rating index called Energy Release Component (ERC). Large fire ignitions are evaluated by analyzing the relationship of historic large fire ignitions with ERC and are calculated by logistic regression. Fire suppression is estimated using statistical model of containment based on large fire records. Fire growth and behavior are calculated using Rothermel (1972) to evaluate the spread and intensity of surface fire and Rothermel (1991) and Van Wagner (1977) to calculate crown fire. The propagation is based on the MTT (Finney, 2002). FSim includes spotting modeling from torching trees (Albini 1979).

Wildfire Analyst (WFA) is a desktop software application developed by Ramirez and Monedero (2011). This model can provide real time analysis of wildfire progression, fire behavior, suppression capabilities and impact analysis during a wildfire. The primary spread model in the WFA tool is the Rothermel (1972) model and the modifications proposed by Albini (1976). The propagation technique is the MTT (Finney, 2002). WFA can simulate, among other things, the evacuation time mode, that is the minimum time required for a fire to reach the defined evacuation points (Ramirez and Monedero, 2011).

The Sardinian web-based wildfire simulator (SWWS) is a software application developed by National Research Council of Italy, Institute of Biometeorology. This model uses the level-set method to simulate the wildfire propagation. The fire-spread model is based on Rothermel's surface fire spread model (Rothermel, 1972; Ghisu et al., 2014; Arca et al., 2018).

1.5 Main Assumptions and Limitations of Fire Spread and behavior Models

"All models are wrong, but some are useful" (George Box, 1979). This sentence summarizes the uncertainty and the nature of models. Models are simplification of reality, that reduce complex phenomena, as wildfires, in mathematical equations. Considering that there are several factors involved in fire spread and behavior, as topography, vegetation and meteorology, and that their interrelations are very complex, it is impossible to perform a simulation without uncertainty (Viegas, 2002).

Riley and Thompson (2017) analyzed the uncertainty of wildfire modeling and highlight three important dimension of uncertainty: nature, locations and levels. Nature can be linked to knowledge or to variability; the first case is reducible because the limitation is referred to understanding, the second is irreducible because is pertained to natural and anthropic systems. Uncertainty can have different locations, that are context (basic assumptions of model), inputs of model, model structure, model technics and parameters. Finally, levels of uncertainty pass from total determinism to total ignorance and between these endpoints there are 1) statistical level, that can be mitigate probabilistically or quantitatively, 2) scenarios, for which we know results but not their likelihood, and 3) recognized ignorance, in which we know the source of uncertainty but not the different possibilities or their likelihoods.

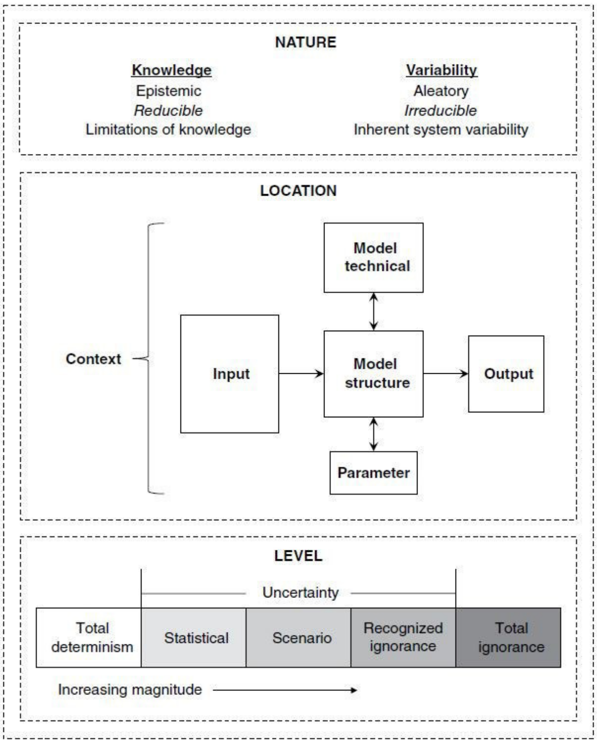

Fig. 6. Representation of the three dimensions of uncertainty (nature, location, level). Picture from: Riley and Thompson, 2017.

One of the principal simplifications used by several fire model is to consider continuous and homogeneous fuels. This problem is very important because real vegetation is different: this influences the goodness of prediction, especially for the works at high resolution (Parsons et al. 2011). This simplification interests both moistures as well the physical characteristics of fuels (Cheney 1981). Several authors tried to solve this question, for example Rothermel in 1983, which introduced two-fuel model concepts and the use of GIS based fire growth models such as FARSITE (Finney, 1994). However, future research still should focus on this topic (Parson et al., 2011).

Several models consider fuel bed like a single layer and continuous to the ground. In the last years, some authors improved the studies on this topic using both physical models, for example Linn and Cunningham. (2005), as well empirical models, as Van Wagner (1977), Cruz and Alexander (2013), and Cheney et al. (2012) (Alexander and Cruz, 2013).

Wildfire is influenced by wind, slope and fuels. Some models consider these factors separately to ease simulations, but this is a simplification (Viegas, 2004). Several authors tried to combine the effects of diverse factors: for instance, Rothermel in 1972 introduced factors of wind and slope with the same directions and gradients in his fire model, and in 1983 he modified the interactions of wind and slope considering a not parallel effect. This model is the more used by most fire behavior prediction systems (Viegas, 2006).

As already mentioned, wildfires spread from burned to unburned fuels by heat transfer (Drysdal, 1998). However, various works do not reach the general conclusion on the ignition and spread of wildfires (Finney et al., 2013). This fact causes an uncertainty in the simulations, because the limited knowledge in fundamental principles of fire propagation (Finney et al. 2015). For instance, many models are based on the theory that radiation produced by the combustion process is the main heat source responsible for fuel particle ignitions and first phases of spread of wildland fire, but recent works revealed that only radiation is insufficient alone to support fire spread (Finney et al., 2015). Finney et al. (2015) also state that beyond radiation also convection has a key role in fire spread and ignitions, and this topic needs to be further investigated.

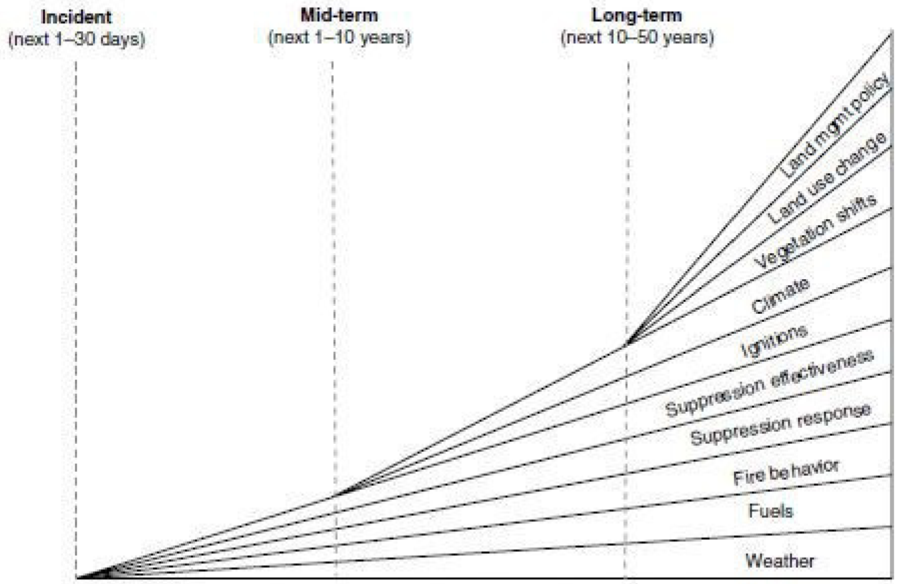

Fig. 7. Compounding uncertainty across planning levels. As modeling frameworks move from shorter to longer-term planning contexts, additional sources of uncertainty come into play, and existing sources of uncertainty grow in magnitude. Picture from: Riley et al., 2017.

The time frame used in wildfire simulations is very important, because the planning horizon is directly proportional to uncertainty. If we consider a time frame of 10 years we do not have certainty of ignitions, weather, landscape, and management data for all years. It is better to consider a brief time range to have better and more certain simulations (Fig. 6) (Riley and Thompson, 2017).

Finally, we have to know which variability is involved and the context to mitigate the uncertainty.

1.6 FlamMap: potential and main applications

FlamMap is a software, free to download, that analyzes and maps fire behavior, calculating potential fire behavior characteristics like spread rate, flame length and fireline intensity using the Minimum Travel Time (MTT) algorithm (Finney 2002). FlamMap considers constant weather and fuel moisture conditions in the single temporal propagation unit and does not simulate their temporal variations in fire behavior (Papadopoulos, 2011).

Improved versions of FlamMap is the so-called MTT version, as well as Randig, a command line version of MTT. These models can simulate thousands of fire events using multiple weather scenarios selecting a random sequence among the scenarios defined in a specific input file according to their relative probability (Kalabokidis et al., 2014).

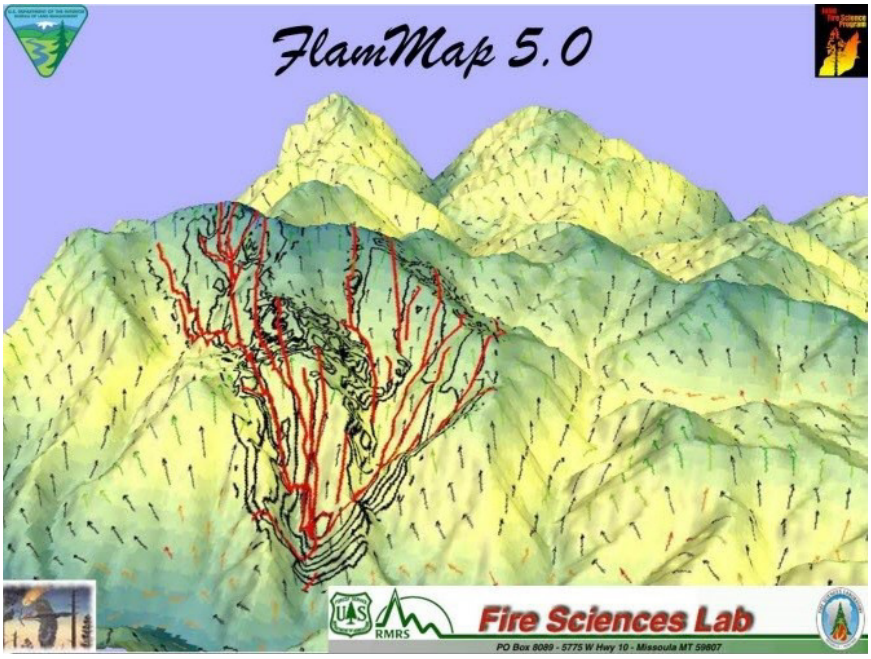

Fig. 8. Main menu of FlamMap

Using the MTT it is possible measure the areas where there are overlapping fires, and so it is possible crate burn probability maps. The maps of outputs produced by FlamMap are helpful to identify particular combinations of fuels, topography and wheatear that can cause a greater fire risk, so we can focus on these areas the prevention and suppression actions (Stratton, 2004)

There are various applications of FlamMap in several papers. We will analyze some of these, in particular the evaluation of wildfire exposure, the effects of fuel treatments, the effects on soil losses and the risk transmission. Often these applications can be combined in several works. In the Mediterranean area there are lots of works on wildfire exposure that used FlamMap MTT or Randig.

1.6.1 Evaluation of wildfire exposure

Arca et al. analyzed differences in term of burn probability and fire severity variations among different weather scenarios in Sardinia (24,000 km2). These scenarios derived from the Regional Climate Model (RCM) EBU-POM, implemented by the Belgrade University and the Euro-Mediterranean Center for Climate Change (CMCC). In this study Arca et al. considered as baseline climate scenario the current Mediterranean climate (1961-1990) and the future period 2071-2100. They used Randig (Finney, 2006) and simulated 100,000 random fires using ignition points obtained by historical ignition points (Arca et al., 2012).

Ager et al. (2012) evaluated wildfire exposure in the Deschutes National Forest near Bend, Oregon (6530 km^2). They used Randig (Finney et al., 2006) to simulate wildfire and generated maps of burn probability, conditional flame length and fire size. They also evaluated wildfire risk transmission and calculated the source-sink ratio (SSR) of wildfire calculated as the ratio of fire size generated by an ignition to burn probability. 50,000 wildfires were simulated at a 90 m spatial resolution (Ager et al., 2012).

Salis et al. (2013) applied a burn probability modelling approach in Sardinia (24,000 km2), to evaluate wildfire exposure from large fire events of social, economic and ecological resources. They simulated 100,000 wildfires at 250 m resolution. Similar approaches, working at resolution of 200 m and 150 m, were used to evaluate spatiotemporal variations in wildfire intensity and size, and burn probability in Sardinia (Salis et al. 2014, 2015).

Kalabokidis et al. (2014) used Randig to evaluate fire exposure of two areas in Greece, *i.e.* Lesvos Island (in North Aegean Sea, 1,650 km^2) and Messenia (in Southwestern Peloponnesus, 3,000 km^2). They calculated burn probabilities, fire size and flame length probability and furthermore they evaluated fire hazard and vulnerability taking into consideration values-at-risk. They simulated 100,000 fire events for the two study areas (Kalabokidis et al. 2014)

In 2015 in another work Kalabokidis et al. carried out a work on fire regime in Messinia, (in Southwestern Peloponnesus, 3,000 km^2), for the present (1961–1990) and the future (2071–2100) climate projections. These projections derived from simulations of the KNMI regional climate model RACMO2, under the SRES A1B emission scenario. To achieve this goal, they divided the whole study area in three landscapes and simulated using Randig 100,000 fires for 300 min of wildfire duration for each landscape at 60 m resolution. They obtained burn probability, potential fire spread and intensity values (Kalabokidis et al. 2015).

Mitsopolous et al. (2015) evaluated the effects of three different burning condition scenarios on the wildland urban interface in Greece, Mt. Penteli (160 km^2) and calculated wildfire risk components as burn probability, conditional flame length, fire size, and source–sink ratio using FlamMap. They created custom fuel models specific of the study area using the field fuel parameters and they localized the different fuel types and residential structures in the study area using photointerpretation procedures of large scale natural color orthophotographs. They simulated 10,000 fires at 30 m resolution (Mitsopoulos et al. 2015).

In a work of 2016, Mitsopoulos et al. used FlamMap to evaluate fire behavior in Greece (Mt. Penteli, 160 km^2). They considered four summer periods, one referring to present, 2000, and three to future, 2050, 2070 and 2100, under the A1B emissions scenario. They used as ignition point for all simulations the starting spot of a large fire in 2009 that burnt 14,000 ha in the study area. The duration of fire was 480 min at 30 m resolution. They obtained outputs useful for fire management planning across the landscape and being related to climate change they are valuable components of long terms fire prevention and management (Mitsopoulos et al. 2016).

Alcasena et al. (2015) used FlamMap 5 (Finney, 2006) to simulate wildfires and evaluate the burn probability and intensities at landscape scale. They analyzed and mapped wildfire exposure for the different Highly Valued Resources (HVR) structures in province of Nuoro, Sardinia (680 km^2). They simulated 90,000 fires at 40 m resolution considering extreme moisture conditions and wind by historical data and historical fire ignitions (Alcasena et al., 2015).

Fréjaville et al. (2015) carried out a study in two areas of Provence (south-eastern France) affected by wildfires. These areas were similar in terms of climate, but had different fuel types and landscape characteristics. They simulated with FlamMap 125,000 fires per weather scenario, obtaining the burn probability and fire behavior metrics. They combined these indicators and calculated the fire severity index (FSI) that linked the probability of burning of an area with the intensity and residence time of occurring fires (Fréjaville et al., 2015).

In another work of 2016, Alcasena quantified wildfire exposure of Highly Valued Resources and Assets (HVRAs) in an area of 280 km^2 in Central Navarra (Spain) using MTT algorithm in FlamMap 5 (Finney, 2006). They used Lidar data to characterize canopy fuel, and using recent fire weather and moisture conditions data and historical ignition patterns. They simulated 30,000 fires at 20 m resolution and they evaluated burn probability, conditional flame length, fire size, and source–sink ratio (Alcasena et al., 2016).

Fréjaville et al. (2016) used FlamMap to simulate wildfire in an area of Western-Alps (31,710 km^2) with the aim of evaluating fire spread and intensity and the effects of climate, vegetation composition and fuel moisture on fire behavior. They performed various simulations for different scenarios of wind speed and fuel moisture, with fixed wind direction. In this study they linked wildfire simulations with multivariate analysis (Fréjaville et al., 2016).

Mallinis et al. (2016) carried out a work with the objective to analyze the spatial variation of wildfire regime in Holy Mount Athos in Greece (330 km^2) that includes 20 monasteries and other structures frequently interested by wildfires. They used FlamMap to simulate 30,000 fire for 480 min of duration at high resolution and they calculated evaluate burn probability (BP), conditional flame length (CFL), fire size (FS), and source-sink ratio (SSR). They created custom fuel models determined by fuel sampling and high-resolution images (Mallinis et al., 2016).

Thompson et al. (2016) proposed a new risk assessment approach that integrates complementary pixel-based outputs of fire behavior and polygon based outputs to simulate final fire perimeters in an area of 74,000 km^2 in Colorado. This approach improved the evaluations of potential wildfire impacts to highly valued resources and assets (HVRAs). They used FlamMap 5 for the deterministic fire behavior modelling, but they used also FSim (Thompson et al., 2016).

The first application of a high resolution methodology landscape wildfire modelling to evaluate impacts of climate changes on wildfire exposure at national scale in Europe is the work of Lozano et al. (2017). They used simulation modelling to assess potential climate change impacts on wildfire exposure in Italy and Corsica (France) (310,000 km^2) and simulated 620,000 fires for three climatic periods (1981–2010, 2011–2040, and 2041–2070), at 250 m resolution using Randig (Lozano et al., 2017).

Mitsopoulos et al. (2017) studied fire suppression difficulty in three different ecosystems in Eastern Europe, Mt. Menoikio (Greece, 22 km^2), The Bayam Forest District (Turkey, 160 km^2), and the Yalta Mountain-Forest Natural Reserve (Ukraine, 25 km^2). The used very high resolution satellite imagery and landscape fire behavior modelling using FlamMap. The results are a fire suppression difficulty map that could foster cooperation between national authorities and would also maximize the efficiency of firefighting procedures (Mitsopoulos et al., 2017).

Fréjaville et al. (2018) evaluated the effect of warm and dry climate on variations of Potential Fire Intensity (PFI) and Crown Fire Likelihood (CFL) in species ranges in the western Alps, in an area of 31 710 km^2. They used FlamMap (Finney, 2006) to simulate PFI and CFL, considering fine fuels for live biomass and fine- to medium-sized fuels for dead biomass. They considered a different scenario of dead moisture fuels (range 5%-14%) and repeated simulations for this scenario (Fréjaville et al., 2018).

1.6.2 Evaluation of fuel treatments effects on fire exposure

In 2008, Stratton presented in a paper a methodology to evaluate the effects of landscape fuel treatments on wildfire behavior in an area of 8 km^2 in Utah (USA). Fuel Treatments are localized using the Bureau of Land Management (BLM) that is based on the threat of fire to communities and the need for range and wildlife improvement. Stratton (2008) calculated a fire density grid, using the BLM's fire start layer that identifies the historical high ignition areas, and used Fire Family Plus and FLAMMAP. The first simulate the weather scenario starting from historical data. These data were used in FARSITE and FlamMap to model pre- and post-treatment effects on fire growth, spotting, fire line intensity, surface flame length, and the occurrence of crown fire. This method can help managers to plan fuel treatments and the forest fire policy management (Stratton, 2008).

Moghaddas et al. (2010) evaluated effects of fuel treatments on fire regime in an area of 186 km^2 within the Meadow Valley in the northern Sierra Nevada. They used FlamMap and FARSITE and created landscape files using high-resolution aerial (IKONOS) imagery, ground-based plot data, ArcFuels and the Forest Vegetation Simulator. They evaluated crown fire potential, flame length, and conditional burn probabilities on 11 land allocation types in which was divided the study and that was associated with predefined management direction, standards, and guidelines. They simulated with FlamMap 1,000 fires for the pre- and post-treatment landscapes with maximum simulation time for each ignition of 900 min at 30 m resolution. They used FARSITE to simulate a single ‘‘problem fire’’ (Bahro et al. 2007), that are ‘‘hypothetical wildfire that could be expected to burn in an area that would have severe or uncharacteristic effects or result in unacceptable consequences’’ (Bahro et al. 2007), in the study area in pre and post-treatment conditions (Moghaddas et al., 2010).

Ager et al. (2010) examined the effect of fuel treatments on burn probability and intensity within treated stands and the carbon impacts in an area of 163 km^2 in Oregon. They hypothesized two spatial priorities and six treatment percentage of a whole landscape. The first spatial priority is the protection of residential area, while the second is the forest protection. The change in wildfire behavior was translated into expected carbon flux and compared to the treated and untreated landscapes. They used Randig and simulated 120,000 fires at 30 m resolution (Ager et al., 2010).

Chung et al. (2013) optimized a model useful to long-term planning at a landscape scale in term of fuel management. The model has been incorporated into a OptFuels, that is a spatial decision-support system and comprise three simulation and optimization components: Fire and Fuels Extension to the Forest Vegetation (FVS-FFE), FlamMap and a heuristic solver to schedule fuel treatments to minimize the total expected loss over the planning horizon (Jones and Chung 2011). The model evaluated optimal locations and timing of fuel treatments considering changes in forest dynamics over time, fire behavior and spread, values at risk, and operational feasibility in an area of 140 km^2 located on the west side of the Bitterroot Valley in Montana (USA) . The model used the Minimum Travel Time algorithm in FlamMap and the Fire and Fuels Extension to the Forest Vegetation Simulator to evaluate effects in pre and post fuel treatments condition. The final aim was to minimize losses due to wildfires. (Chung et al. 2013).

In 2013 Collins et al carried out a work that had the aim to study the treatment effects on fire regime in an area of 192 km^2 in Sierra Nevada. The work was divided into three sections and objectives, first to evaluate fire regime with and without the treatment network, then to project hazardous fire potential over several decades to assess fuel treatment network longevity and finally to assess fuel treatment effectiveness and longevity over a range of two critical fire modelling inputs: surface fuel models and canopy base height. They used Randig and simulated 10,000 fires for each landscape at 60 m resolution, and each fire has a burning period of 240 min. They selected five time steps to project the simulations: 2010, 2020, 2030, 2040, 2050 and have in total 30 different scenarios (Collins et al. 2013).

In a work of 2014 Ager et al. studied the effect of fuel treatments on wildfire transmission in the Deschutes National Forest in central Oregon (7566 km^2). They took into consideration two areas with the same shape and size and same index of wildfire exposure, and they observed that these areas burn alike, but if one characteristic was different the wildfire changed risk transmission among the parcels, for example the vegetation. It is very important to know what this factor is to hypothesize fuel treatments. They simulated 200,000 fires using Randig (Ager et al., 2014)

Salis et al. (2016) evaluated the effects of fuel treatments on an area in North-East Sardinia (680 km^2). They hypothesized various level of treatments, 3-9-15 per cent of the whole landscape, and located the treatments according to different priorities to protect values such as roads (ROAD) and wildland urban interface (WUI), or randomly located (RAND). Randig was used to simulate 25,000 fires for each treatment alternative at 25 m resolution (Salis et al., 2016).

In 2016 Stevens et al. evaluated the effectiveness of fuel treatment for the protection of different objectives in California, in an area of 78 km^2. These treatments were the main of defender WUI zone (WUI), to reduce potential fire severity across the entire landscape (FHR) and restore the variable forest structure associated with frequent fire and to increase the diversity of wildlife habitat within the treatment (RES). Furthermore, they evaluated if FHR strategies had negative effects on wildlife diversity, if WUI strategies exposed other portions of the landscape to wildfire and if fuel treatment had any effect on emissions. They used Randig and simulated 10,000 random ignitions with 12 h of fire during at 30 m resolution (Stevens et al., 2016).

Oliveira et al. (2016) evaluated effect of fuel break network on an area affected by wildfire in Portugal, in an area of 7,878 km^2. They treated 3% of whole study area and hypothesized two scenarios to be compared with a no treatment scenario. The fuel break network was built on buffer of 120m, in an area that was interested by historical wildfires, near roads, rivers, irrigated valleys or mountain ridges. The first scenario was the treatment of whole buffer, while the second was the treatment of 120 m wide strip and not removal canopy cover at 22% and an understory discontinuous litter, litter and short grass. They used Randig and simulated 150,000 fires per scenario at 90 m resolution (Oliveira et al., 2016).

Martin et al. (2016) carried out a study on forest management planning in eucalypt plantations (14 km^2) focused on fuel treatments and based on economical, ecological and fire prevention criteria. The fuel treatments were created with the aim of minimizing losses from wildfire and to meet budget constraints and demands for wood supply for the pulp industry and conserve carbon. They used FlamMap 5 (Finney, 2006; Martin et al., 2016).

Chiono et al. (2017) evaluated effects of fuel treatments on wildfire in Sierra Nevada, California and the consequences of treatments on carbon stock and quantified the biomass harvested in an area of 554 km^2. They simulated fuels reduction treatments and wildfire and evaluated the carbon balance of the treatment scenarios. Therefore, they first quantified the carbon contained in the forest biomass harvested in each treatment scenario, then quantified the carbon emitted during prescribed fire and wildfires, and finally quantified the carbon remaining within onsite pools. They simulated 80,000 fires at 90 m resolution using Randig (Chiono et al., 2017).

Alcasena et al. (2018) carried out a study still on fuel treatments in an area of 1300 km^2 in Catalonia. They had three principal aims: first to increase the resiliency of sub-Mediterranean forest ecosystems disrupting major fire movements, then to facilitate fire suppression and to reduce ember emissions, and finally to protect wildland urban interface rural communities from catastrophic events reducing the likelihood of large fires burning into residential communities. In this work they used LTD (Vogler et al. 2015) to create fuel treatments and FlamMap to simulate wildfire at 40 m resolution, and using extreme weather condition, so condition of 97th percentile (Alcasena et al. 2018).

Fitch et al. (2018) analyzed treatment effectiveness in reducing wildfire suppression costs. They evaluated the effects of fuel treatments on fire behavior for Four Forest Restoration Initiative and they selected three treatment alternatives from the Draft Environmental Impact Statement: no treatments, the medium treatment option and the preferred treatment option, most aggressive in term of treatment thinning intensity.

In the most aggressive option they hypothesized to treat 175,640 ha using mechanically treated across the entire 4FRI treatment area, then, 240,072 ha using prescribed fires. In the medium treatment option, they proposed to treat 157,221 ha using mechanically treating, and 72,356 ha using prescribed fire. They used to simulate fire behavior FlamMap 5 (Finney, 2006) and used the fuel moisture characteristics from the Schultz

Fire that burned 6070 ha in the landscape and the wind and weather condition were obtained from the local weather station. Finally, they obtained the suppression cost estimation by a regression analysis with the outputs of the fire simulations (Fitch et al., 2018).

1.6.3 Other applications of FlamMap

Fuel treatments can have effects on soil erosion, for instance Sidman et al. (2016) evaluated the effects of fuel treatments on fire reduction and so in post-fire hydrological responses in south-western Utah. They used three model, FuelCalc and FlamMap within the Wildland Fire Assessment Tool (WFAT) to create fuel treatments, the First Order Fire Effects Model (FOFEM) within WFAT to model wildfire and evaluate the effectiveness of the fuel treatments and finally. KINEROS2 within the Automated Geospatial Watershed Assessment tool (AGWA) to model post wildfire hydrological response. They hypothesized a planned prescribed fire at Zion National Park and a planned mechanical thinning at Bryce Canyon National Park and they performed simulations for treated and untreated landscapes (Sidman et al., 2016).

In another work written by Elliot et al. (2016), the authors described the effects of fuel treatments on fire behavior and erosion. They used FlamMap to simulate fire behavior before and after fuel treatments, FSim system (Finney et al., 2011) to model 40,000 potential fire seasons and predict the perimeters of fires in these fire seasons and the WEPP model (Laflen et al. 1997) predicted hill slope erosion using FlamMap outputs. FSim run at 90 m resolution and FlamMap and WEPP at 30 m resolution. They modelled four conditions: current vegetation fuel conditions in the absence of fire, after a fire without fuel treatments, after fuel treatments and after a fire considering fuel treatments (Elliot et al., 2016).

Analysis on wildfire risk transmission is another application of FlamMap, for instance Haas et al. (2015) studied wildfire risk transmission and located the areas of highest exposure of human populations to wildland fires under severe weather events in Colorado (USA). They identified different levels of exposure considering how much population was potentially interested by the risk of wildfire. They used Randig (Finney et al., 2006) to simulate wildfire and to investigate where fire ignitions were most likely to cause the highest impact on human communities, and the human causes that influenced the transmission of risk. They simulated 50,000 fires at 90 m resolution (Haas et al., 2015).

Alcasena et al. (2017) evaluated both the wildfire transmission and potential economic losses to residential houses in a rural area of central Navarra (Spain). They quantified expected losses considering individual structure level in 14 rural communities using FlamMap to evaluate burn probability and fire intensity and using a response function (RF). Fire exposure was estimated by simulating 50,000 fire events that considering extreme historical fire weather conditions (97th percentile) at 20 m resolution (Alcasena et al., 2017).

1.7 Conclusions

In this chapter, a short introduction on wildfire simulation modeling was presented. First of all, different fire prediction models and fire prediction techniques were described, then several fire spread simulators were analyzed.

Among the fire spread models, one of the most used in the Mediterranean area is the MTT as developed by Finney (2002). There are many applications of this model: an exhaustive state of the art on the use of the MTT approach to analyze a number of wildfire issues was presented.

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Chapter 2: Modeling the effects of different fuel treatment mosaics on wildfire spread and behavior in a Mediterranean agro-pastoral area

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2.1 Introduction

The occurrence of large wildfire events is mainly associated with extreme weather conditions and the presence of highly flammable, unmanaged and continuous forest fuels (Cardil et al., 2017; Dimitrakopoulos et al., 2011; Fernandes et al., 2012; Keeley et al., 2012; Pausas and Vallejo, 1999; San-Miguel-Ayanz et al., 2013; Xanthopoulos et al., 2009). Nonetheless, in the Mediterranean Basin, wildfires often affect areas largely characterized by the presence of herbaceous flashy fuels, such as open woodlands (e.g.: dehesas and montados), meadows and pastures, or dryland crops (e.g.: wheat, barley, and oat) (Bajocco et al., 2017; Levin et al., 2016; Naveh, 1973; Salis et al., 2014). In these areas, the presence of cured herbaceous fuels througout the fire season favors the ignition and propagation of large wildfires, even with moderate weather and low fuel loads, as well as the ignition of short-distance spot fires in advance of the main fire front, which further enlarges fire perimeters and complicates fire suppression efforts (Colin et al., 2002; Costa-Alcubierre et al., 2011; Nudda et al., 2015, 2016, 2017; Salis et al., 2016a). Furthermore, in several Mediterranean areas herbaceous fuels are preferential sites of fire ignition, and thus can be a source of large events that can later spread into forests or anthropic values (Alcasena et al., 2017; Gonzalez-Olabarria et al., 2015; Ricotta and Di Vito, 2014). For instance, the largest event that affected the island of Sardinia (Italy) in the last 20 years (Bonorva wildfire, July 2009, 10,600 ha burned), one of the largest wildfire events ever occurred in Italy, mainly affected herbaceous-type land tenures (Salis et al., 2012; Schmuck et al., 2010). This wildfire spread for two days under extreme weather conditions, presented maximum spread rates close to 4 kmh^{-1}, and caused substantial losses to agro-pastoral farms and inland rural communities (Fois, 2015); moreover, even aerial resources had limited success in containing the wildfire spread.

Wildfire management within the Mediterranean Basin continues to increase in complexity, due to a number of converging drivers that amplify potential threats to ecological, social and economic values (Bovio et al., 2017; Corona et al., 2015; Curt and Frejaville, 2017; Moreira et al., 2011; Salis et al., 2016b). Major drivers include the increasing presence of anthropic values and activities into fire-prone areas, budget constraints in promoting wildfire prevention and mitigation policies, the progressive ageing of the population and associated land abandonment in forest and rural areas, the lack of adequate fuel management, and climate change (Bedia et al., 2014; Bonet and Pausas, 2007; Brotons et al., 2013; Chergui et al., 2017; Curt et al., 2013; EEA, 2017; Fernandes, 2013; Karali et al., 2013; Lozano et al., 2017; Madrigal et al., 2017; Oliveira et al., 2017; Pausas and Fernandez-Munoz, 2012; Pellizzaro et al., 2012; Ruiz-Mirazo et al., 2012; Salis et al., 2014; Turco et al., 2015; Velez, 2002; Xanthopoulos et al., 2006). Consequently, there is a growing interest in wildfire risk assessment tools that can support land managers and policy makers in mapping wildfire exposure, prioritizing fuel treatment efforts, developing comprehensive strategies for risk mitigation and climate change adaptation, and optimizing strategies and investments with finite budgets while accounting for diverse operational constraints (Ager et al., 2011, 2017; EEA, 2017; Piqué-Nicolau et al., 2014; Thompson et al., 2012, 2013). To induce relevant changes in fire spread and behavior, it is widely accepted that the most efficient approach is the alteration of fuel conditions (e.g.: load and continuity) at the landscape scale (Agee and Skinner, 2005; Reinhardt et al., 2008). Fuel management is primarily intended to modify wildfire behavior and growth through strategic placement and arrangement of treatment units at strategic locations (Ager et al., 2010, 2013; Cochrane et al., 2012; Finney, 2001; Graham et al., 2004; Liu et al., 2013; Oliveira et al., 2016; Parisien et al., 2007; Salis et al., 2016b; Schmidt et al., 2008). Moreover, treating fuels can help fire crews suppress wildfires by enlarging safety areas or escape routes, and hence can enhance their capacity to contain an event (Agee et al., 2000; Calkin et al., 2014; Montiel and Kraus, 2010; Weatherspoon and Skinner, 1996).

The integration of fuel management strategies into wildfire management poses a number of tradeoffs for land managers tasked with identifying the best spatial arrangements and treatment solutions while taking into account management goals, and financial, social, legal and physical constraints (Agee and Skinner, 2005; Ager et al., 2010, 2013, 2017; Argañaraz et al., 2017; Collins et al., 2010; Corona et al., 2015; Finney et al., 2007; Hand et al., 2014; Hudak et al., 2011; O'Connor et al., 2016; Parsons et al., 2017; Reinhardt et al., 2008; Schmidt et al., 2008; Scott et al., 2016; Thompson et al., 2012, 2017; Thompson and Calkin, 2011; Vogler et al., 2015). Overall, fuel treatments will not stop or eliminate fires (Calkin et al., 2014; Finney and Cohen, 2003; Price and Bradstock, 2010); in fact, scattered widespread fuel treatments can be bypassed or eluded by large events (Finney, 2004, 2007; Reinhardt et al., 2008).

Yet, fuel treatments and land management strategies are supported by relatively little research, particularly in the Mediterranean Basin context, on how treatment strategies and the spatial arrangement of treated units affect wildfire transmission and behavior, and on the effectiveness of fuel treatments to limit wildfire growth and exposure at landscape scales (Alcasena et al., 2017; Duguy et al., 2007; Fernandes et al., 2004; Gonzalez- Olabarria and Pukkala, 2011; Mitsopoulos et al., 2016; Oliveira et al., 2016; Salis et al., 2016b). Preliminary work has shown that the maximum efficiency in fuel treatment effectiveness while minimizing area treated could be obtained by the creation of patterns of rectangular treatment units, and regular mosaic patterns were proved to be more efficient than random arrangements, particularly when small areas are treated (Bevers et al., 2004; Finney, 2001, 2004; Loehle, 2004; Schmidt et al., 2008). Promising results have been obtained with the development of fuel treatment optimization models, which can mitigate fire risk while taking into account fuel management multi-objective perspectives or specific needs (Ager et al., 2013; Alcasena et al., 2018; Arca et al., 2015; Chung et al., 2013; Finney, 2007; Kennedy et al., 2008; Rytwinski and Crowe, 2010; Vogler et al., 2015). The final evaluation of the effectiveness of fuel treatments typically requires the estimation of altered wildfire spread and behavior before and after the implementation of fuel treatment strategies (Ager et al., 2010, 2014; Finney et al., 2007; Kim et al., 2009; Schmidt et al., 2008; Stratton, 2004). In recent years, spatial fire growth simulators and burn probability modeling approaches based on the MTT algorithm (Finney, 2002) have emerged as useful tools for analyzing the influence of fuel treatments on wildfire growth and behavior, and for performing risk-based simulation of fuel treatment efficiency (Finney, 2005, 2007; Miller et al., 2008; Riley and Thompson, 2017; Thompson et al., 2012).

The goals of this study were to: (1) analyze the effects of different fuel treatment arrangements, unit sizes, and percentages of treated area on simulated wildfire exposure metrics at the landscape scale, and (2) determine to what extent treatment effectiveness is conditioned by diverse wind speed conditions. With this purpose, we simulated fire spread and behavior considering the driest fuel moisture conditions in a study area of about 625 km2, mainly covered by herbaceous surface fuels, and located in Northern Sardinia, Italy. Fuel treatments were constrained to specific herbaceous land use classes and changed the treated fuels to unburnable state. The methodology and findings presented in this study can support the design and optimization of fuel management programs and wildfire risk mitigation policies in agropastoral areas of the Mediterranean Basin.

2.2 Material and methods

2.2.1 Study area

The study area is located in Northern Sardinia, Italy, and has nearly 62,500 ha of land (Fig.1). Overall, the area is characterized by the presence of large flat zones, with the highest peaks (Goceano Mountains) located in the southeastern portion of the territory. The elevation ranges from about 180 m a.s.l. to 970 m a.s.l., with an average elevation of about 400 m a.s.l. (Fig. 1). The climate is Mediterranean, with relevant variations in temperature and precipitation between the hot and dry period and the cold and wet winter. The average annual precipitation is about 650 mm; peaks of more than 750mm are common at the highest elevations (Chessa and Delitala, 1997). The highest precipitation is observed in November and December, while July is the hottest and driest month of the year. The average annual temperature is about 13 °C; maximum temperatures are often above 30 °C in the summer season.

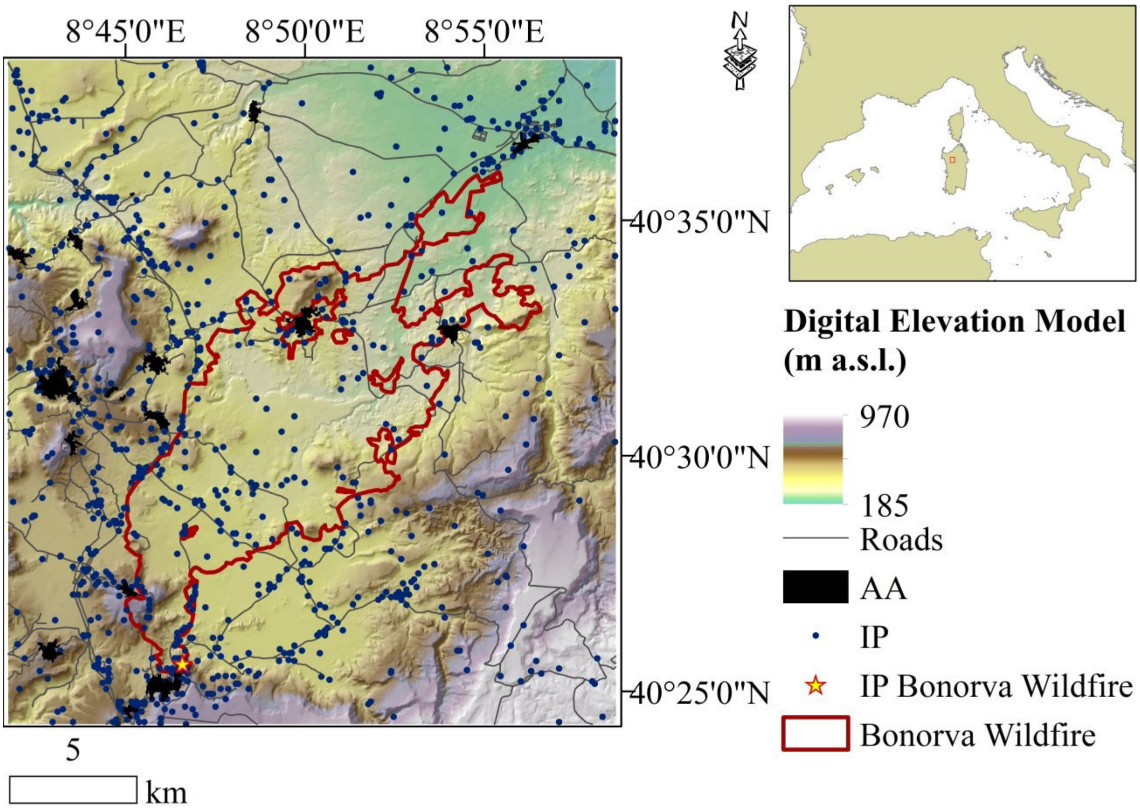

Fig. 1. Digital elevation model (DEM) of the study area (North Sardinia, Italy) along with roads and urban and anthropic areas (AA). The study area was affected by a very large wildfire (Bonorva, 23 July 2009, about 10,600 ha of size (red polygon)), which was one of the largest events ever observed in Sardinia since the 1990s. The fire ignition points (IP) of the study period 1998e2015 are showed in blue.

The study area is one of the most important agro-pastoral areas of Sardinia. In fact, sheep (about 800 farms and 300,000 head) and cattle farms (about 450 farms and 15,000 head) are key components of the productive sector of the area. Moreover, about 1,700 farms (with at least 1 ha of land) are involved in agricultural production (ISTAT, 2010). The area consists of a number of small municipalities, with about 25,000 residents (ISTAT, 2011); urban and anthropic areas cover approximately 1,400 ha of land (Figs.1 and 2). The vegetation is largely characterized (about 65%) by the presence of herbaceous fuels, most of which is classified as grasslands and pastures (Fig. 2).

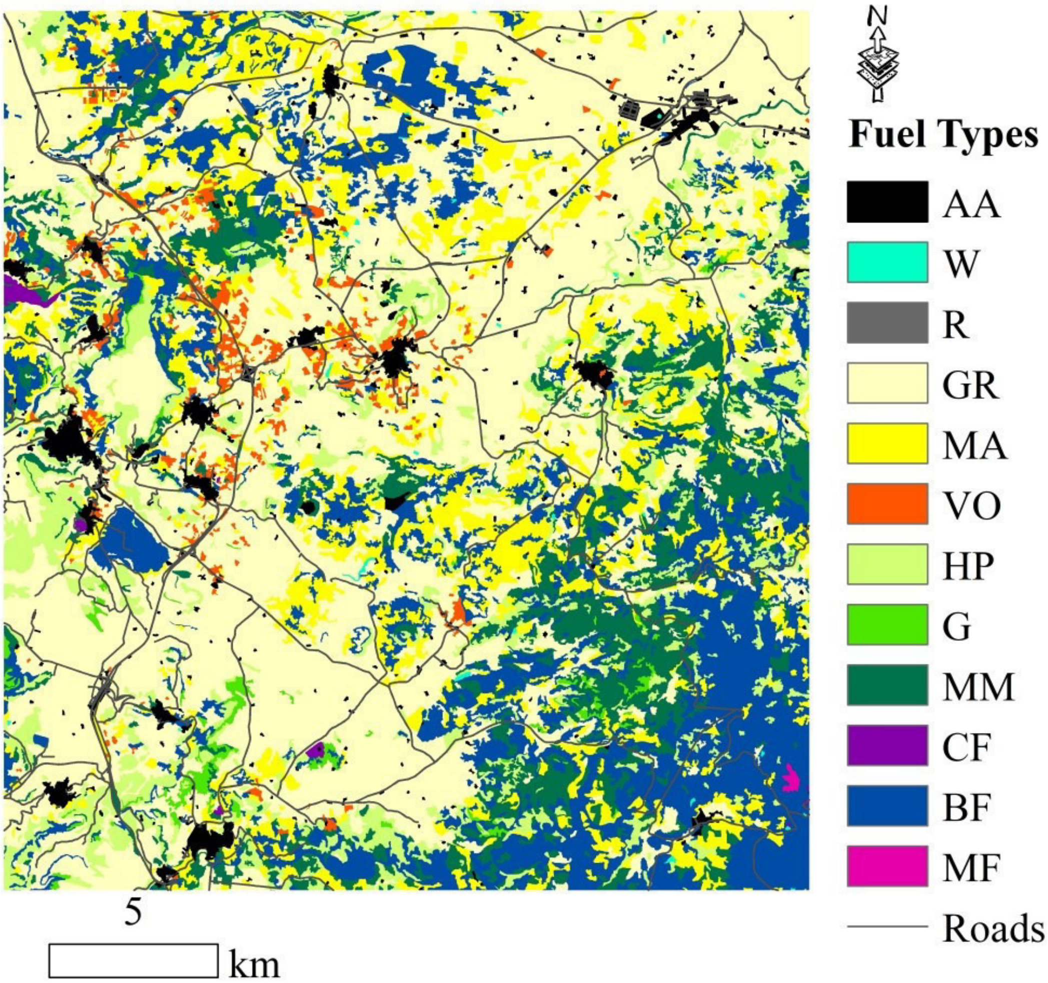

Fig. 2. Main fuel types of the study area. AA=urban and anthropic areas; W=waterMain AA=urban and anthropic areas; W=water bodies; R=rocks; GR=grasslands; MA=mixed agricultural areas; VO=vineyards and orchards; HP= herbaceous pastures; G=garrigue; MM=Mediterranean maquis; CF=conifer forests; BF=broadleaf forests; MF=mixed forests.

Herbaceous and open wooded pastures, as well as marginal shrublands and woodlands, play a key role in the economy and needs of the local livestock farms. Grasslands are mainly devoted to herbaceous autumn-winter crop productions. Shrubland formations (8%) are relatively tall and complex in the most of the study area, and comprise *Olea europaea* L. var. *oleaster* Hoffgg. Et Link, *Phyllirea* spp., *Pistacia lentiscus* L., low-height *Quercus* spp.; low brushes (e.g.: *Cistus* spp., *Pyrus* spp.; *Prunus* spp.) are present in the most degraded and grazed lands. Broadleaf forests (17%) are mainly confined to hills and mountain areas (Fig. 2), and are principally constituted by *Q. pubescens* Willd., *Q. suber* L., and *Q. ilex* L.. Fruit-bearing areas are represented by sparse and family-farm vineyards, olive groves and cherry-trees, and cover about 2% of the area, mainly concentrated in the western plains (Fig. 2).

2.2.2 Wildfire data

We used the 1998-2015 fire database provided by the Sardinia Forest Service. This database contains information on ignition point coordinates, municipality, ignition date, and fire size. In the above period, the study area experienced about 950 fire ignitions (Fig. 1), and the total area burned was close to 19,500 ha, that is on average about 55 wildfires and 1,080 ha of land burned per year. Overall, wildfires above 100 ha accounted for 82% of the total area burned, and only for 1.5% of the total fire number in the study area. These large events were concentrated during late June to late August. Fires were frequently ignited near roads, villages and small family-run farms (Fig. 1). The main wind direction (SW) associated with large wildfires (>100 ha) in the study area was derived from wildfire reports, weather data, and personal communication with the Sardinia Forest Service. SW winds contributed to about 79% of the total area burned by wildfires >100 ha in the period 1998-2015. The main weather pattern associated with these large events is related to the movement of hot and dry air masses from North-Africa (which in northwestern Sardinia often flow from the SW due to orographic effects), originated by a low-pressure cell moving eastward across the western Mediterranean Sea. The majority of the total area burned was related to Bonorva fire (Fig. 1), which ignited on July 23, 2009, spread for more than 20 km and burned approximately 10,600 ha in about 36 h. The largest fire growth was observed during 11 a.m.e7.00 p.m.. The day of the fire was characterized by extreme weather conditions across the entire island in terms of temperature, relative humidity and high-intensity wind (ARPAS, 2009).

2.2.3 *Input data for wildfire modeling*

All input data were assembled to generate a 25 m resolution gridded landscape file as required by FlamMap (Finney, 2006). The terrain characteristics (elevation, slope and aspect) were derived from 10-m digital elevation data of the island (Sardinia Region geoportal, 2017). Surface fuels were interpreted from the 2008 Sardinian Land Use Map (Sardinia Region geo-portal, 2017). We assigned to each land use class either a standard or custom surface fuel model (Anderson, 1982; Arca et al., 2009; Salis et al., 2013; Scott and Burgan, 2005) (Fig. 2 and Suppl. Table 1). Canopy metrics (canopy cover, canopy bulk density, canopy base height and canopy height) for forest areas were estimated using as reference Q. suber L. and Q. pubescens L. stands, using data from the National Inventory of Forests and Forest Carbon Sinks (INFC, 2005) (Suppl. Table 1). Fuel moisture content (FMC) for 1-h and 10-h time lag dead fuel was determined by the data and methods of Pellizzaro et al. (2005, 2007) and Salis et al. (2015), and focusing above the 97th percentile values. Considering that most of the study area is flat and that preliminary tests with WindNinja (Forthofer, 2007) showed limited variation between constant and simulated wind data in the fuel treatment areas, fire simulations were performed using constant wind attributes. Specifically, wind direction was held constant (225°), while three different wind speed conditions (16, 24 and 32 km h^{-1}) were set as reference. Finally, we selected all fire ignition locations for the period 1998-2015 in the study area and derived a smoothed historical fire ignition density map. The fire ignition density map was held constant for all fire simulations.

2.2.4 *Fuel treatment alternatives*

Overall, we generated 13 fuel treatment alternatives, which consisted of the untreated condition (NO-TREAT, that is the control) and 12 treatment scenarios obtained by the combination of three landscape treatment percentages with four different spatial treatment unit selection strategies (Fig. 3). Each fuel treatment alternative was represented by a specific 25 m x 25 m surface fuel raster map for wildfire simulations (Fig. 3). We imposed specific criteria for the spatial selection of the single land use units to be treated (Table 1).

Fuel treatments were simulated on single land use units classified with the codes 241, 211, and 212 by the 2008 Sardinian Land Use Map (Table 1). Only single land units between 0.5 and 50 ha were identified as possible targets for fuel treatment. To avoid potential soil erosion issues in case of heavy rain events after the treatments, we limited the possibility of performing the treatments to areas with terrain slope <10°. As indicated in Table 1, fuel treatments converted the treated units into unburnable areas sensu NB models of Scott and Burgan (2005). Fuel treatments were applied to 2% (≈1200 ha), 5% (≈3000 ha), and 8% (≈4800 ha) of the landscape area (Fig. 3). We identified specific priority areas to locate the fuel treatment units for all the strategies tested; these priority areas were held constant for all the strategies taken into account (Table 1). Three fuel treatment strategies focused on the design of disconnected single treatment units characterized by different extents: low size (LOW strategy, 0.5-10 ha), medium size (MED strategy, 10-25 ha), or large size (LAR strategy, 25-50 ha) land use units. In addition, we included a fourth fuel treatment alternative which selected treatment units in a 100-m buffer surrounding the road network (ROAD strategy).

Table 1. Criteria used to select fuel treatment unit polygons. Overall, we tested 13 fuel treatment conditions (NO-TREAT + 4 fuel treatment alternatives x 3 percentages of area treated).

CRITERIA	**DESCTIPTION**
Treatment constraints	We allowed treatments in areas covered by specific herbaceous fuels (annual crops with permanent crops (241), non-irrigated arable land (211), and permanently irrigated land (212) (as derived from Sardinia Land Use Map 2008)). Treatments were constrained to areas with low terrain slope (<10°).
Percentage of landscape treated	We treated 2% (≈1,200 ha), 5% (≈3,000 ha) and 8% (≈4,800 ha) of the study area
Single treatment units	We performed treatments in single units with given size classes [low size (LOW, 0.5-10 ha), medium size (MED, 10-25 ha), and large size (LAR, 25-50 ha)] + in a 100 m buffer around the road network (ROAD).
Spatial strategy	For all treatment alternatives, the single units were located near random priority areas, which determined the reference center of each treatment block. We first randomly attributed a treatment prioritization order to each LAR polygon. We then selected a set of LAR fuel treatment units so that a landscape area treated of 2%, 5% and 8% was obtained. The fuel treatment units of the MED and LOW strategies were selected in the closest neighboring of the selected LAR polygons, and was constrained to a 25 x 25 m Fishnet cells included in the LAR polygons or located within a distance of 100m to the LAR polygons, only in treatable areas (slope < 10°, Corine classes 211, 212, 241). We also imposed that the distance among single units was greater than or equal to 100 m, so that the treated patches were close but not jammed together. The selection of the ROAD polygons was constrained by the intersection of the road network buffer and the 25 x 25 m Fishnet cells included in the LAR polygons or located within a distance of 2500 m to the selected LAR polygons, only in treatable areas (slope < 10°, Corine classes 211, 212, 241)
Fuel treatment type	We converted treated units to non-burnable areas (by superficial tillage & prescribed burning (241 & 211) or summer irrigation (212)).

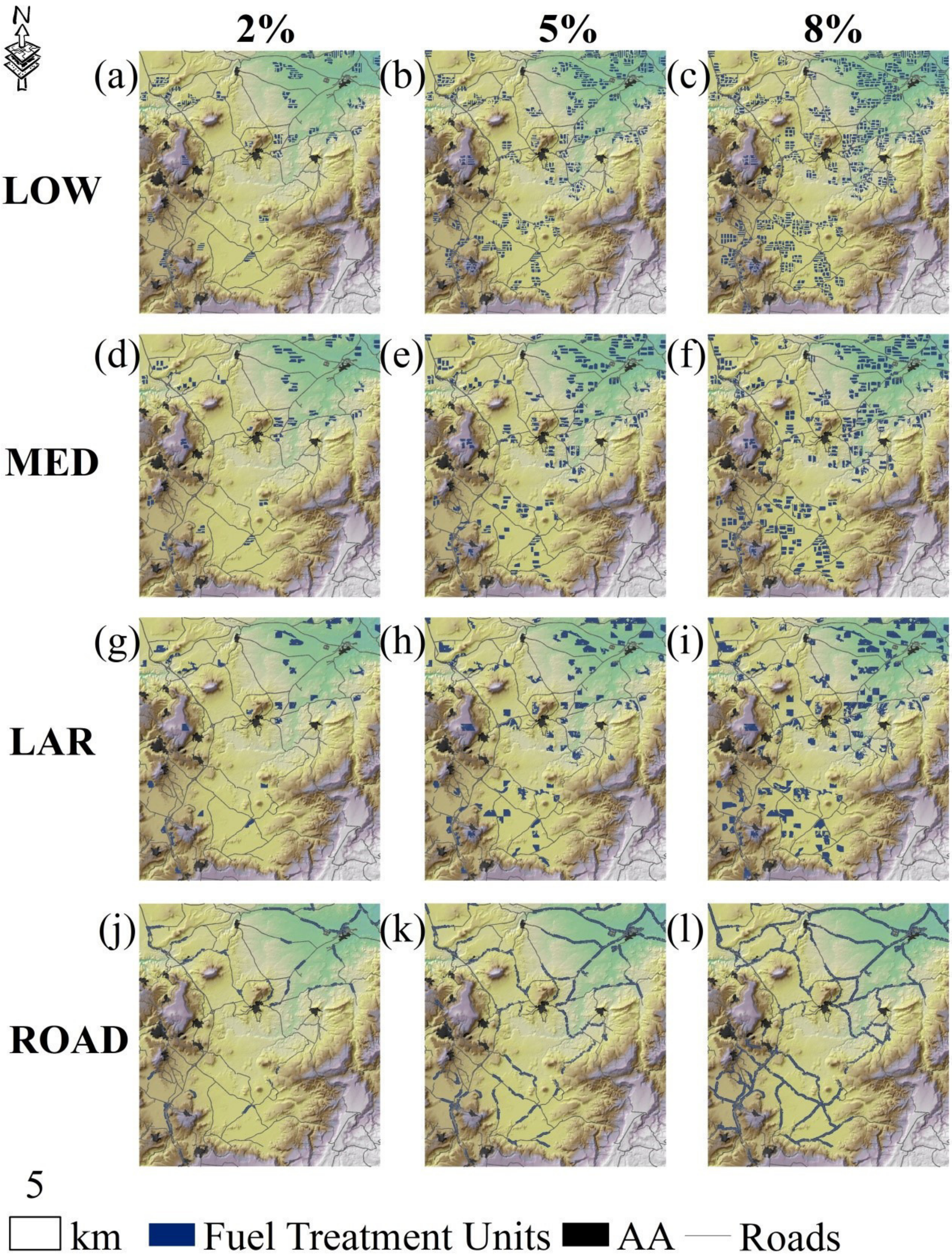

Fig. 3. Maps of the fuel treatment alternatives tested: low size treatment units (LOW, a, b, c), medium size treatment units (MED, d, e, f), large size treatment units (LAR, g, h, i), treatment units nearby roads (ROAD, j, k, l), considering 2%, 5% and 8% of the landscape area treated.

2.2.5 Spatial data on selected anthropic values

We obtained spatial data on selected anthropic values in the study area from the Sardinia Land Use Map (2008). The selected features consisted of continuous urban fabric (CUF, ≈ 445 ha), discontinuous urban fabric (DUF, ≈490 ha), industrial and commercial units (ICU, ≈ 280 ha), and sport and green urban areas (SGU, ≈ 111 ha), and covered about 1,325 ha of the study area. In order to measure simulated wildfire metrics around the above values, we considered a reference zone that consisted of a 150-m buffer surrounding the individual polygons. This distance was adequate to capture general fire behavior in the vicinity of the value, and to focus on the most important human features of the community. Overall, the buffer area used to investigate wildfire behavior around the selected anthropic values totalled close to 7,900 ha, comprised mostly by area surrounding DUF values (5,300 ha). Simulated burn probability and flame length values were used as key wildfire exposure metrics to characterize the probability that a wildfire could affect a given anthropic value, and the potential average intensity at which a wildfire would burn each buffer pixel, respectively.

2.2.6 Wildfire simulation modeling

The wildfire simulations were performed using the minimum travel time (MTT) spread algorithm of Finney (2002), as implemented in Randig. The MTT algorithm has been widely used and is routinely applied to fire management problems, at a broad range of scales and with multiple purposes (Miller and Ager, 2013; Salis et al., 2013). The MTT algorithm models two-dimensional fire growth under constant weather following the Huygens' principle, where fire edge growth and behavior are modeled as a vector or wave front (Finney, 2002; Knight and Coleman, 1993; Richards, 1990). Randig calculates surface fire spread according to Rothermel's equation (1972); crown fire initiation and spread are calculated according to Van Wagner (1977) and Rothermel (1991), respectively. We simulated 5,000 wildfires for each fuel treatment alternative. The ignitions points were located within the burnable fuels of the study area, according to the ignition probability grid originated from the historical fire database. Simulations were performed at 25 m resolution, consistent with the input data, with constant fuel moisture and wind direction (225°), and a burning period of 8 h, which reflected the major fire growth duration of the Bonorva wildfire. Three different wind intensities (16, 24 and 32 km h^{-1}) were set as reference and were used as input for the wildfire spread modeling. Regarding spot fires, in preliminary work we found that spotting probabilities in the range of 1-2% were the best compromise to accurately model large fire events in Sardinia in conditions of intense winds (Alcasena et al., 2015, Salis et al., 2013, 2016a, b). In this study, we used a spot probability of 1% as reference for each fire simulation due to the fact that the study area is largely covered by herbaceous fuels, which typically have lower potential to originate embers than forests or shrublands. Suppression activities were not taken into account by the simulation exercise. The wildfire simulations generated a conditional burn probability (BP) as well as a frequency distribution of flame lengths (FL) in 0.5 m classes for each pixel in the study area. The conditional burn probability is the chance that a pixel will burn at a specific flame length interval, given an ignition in the study area. From the frequency distribution of FL values at each pixel we derived the weighted flame length, which is the conditional flame length (CFL). We then calculated the potential fire size (FS) grid, which was obtained by smoothing the fire size output using inverse distance weighting (search distance 1,000 m) in ArcMap. Burn probability, flame length and fire size were used as indicators to analyze the wildfire response to variations in percentage of landscape treated, wind speed and spatial arrangement of fuel treatments. We considered 2.5 m as a flame length threshold to identify the areas where fire intensity can potentially overwhelm ground crew fire suppression capabilities (Andrews et al., 2011). Statistical differences between fuel treatments and the NO-TREAT control were carried out by the Wilcoxon signed rank test using an alpha value of 0.05.

2.3 Results

2.3.1 Wildfire exposure at the landscape scale

2.3.1.1 Burn probability

On a pixel basis, landscape burn probability (BP) ranged from a low of 0 to a maximum of 0.1606 for the NO-TREAT condition and the highest wind speed value (Table 2 and Fig. 4). Burn probability in all fuel treatment alternatives, including the NO-TREAT condition, was strongly influenced by wind speed. In fact, increments in wind speed promoted growth in average BP values, which for the NO-TREAT condition increased from 0.0136 (16 km h^{-1}) to 0.0284 (24 kmh^{-1}) up to 0.0442 (32 km h^{-1}) (Tables 2 and 3). Regardless of spatial arrangement, wind speed and percentage of area treated, the statistical tests revealed significance differences in BP due to treatment strategy. The Wilcoxon test identified significant differences between the control and all treatment strategies, in particular when 5% and 8% of the landscape was treated, regardless of treatment type or wind speed scenario. Regarding the road strategy, significant differences were also obtained when 2% of the landscape was treated, regardless of the wind scenario. Average BP decreased following a non-linear trend with increasing percentage of landscape treated (Table 2). For instance, at the highest values of wind speed and for the ROAD strategy, average BP dropped from 0.0407 (2% of landscape treated) to 0.0221 (8% of landscape treated). We observed a clear effect of the treatment alternatives on BP: ROAD was unequivocally the most efficient strategy, while for the other three strategies average BP increased moving from large to low size treatment units (Tables 2 and 3). For instance, we found that treating 5% of the landscape using the ROAD strategy was more efficient than treating 8% of the study area with the LOW strategy, even at the lowest wind speed conditions. Furthermore, at 32 km h^{-1} wind speed conditions, treating 8% of the landscape using the ROAD strategy can halve BP with respect to the NO-TREAT conditions (Table 2 and Fig. 5). BP maps showed a marked spatial variability, depending on landscape characteristics, the effects of the spatial arrangement of the treatment alternatives, the percentage of landscape treated and the wind speed conditions (Figs. 4 and 5). The areas with the highest values of BP were associated with: 1) the major wildfire flow paths obtained from the Randig simulations, and 2) historical fire ignition density. Overall, the differences in average

BP containment among alternatives were emphasized by increasing wind speed conditions and treated areas (Fig. 5).

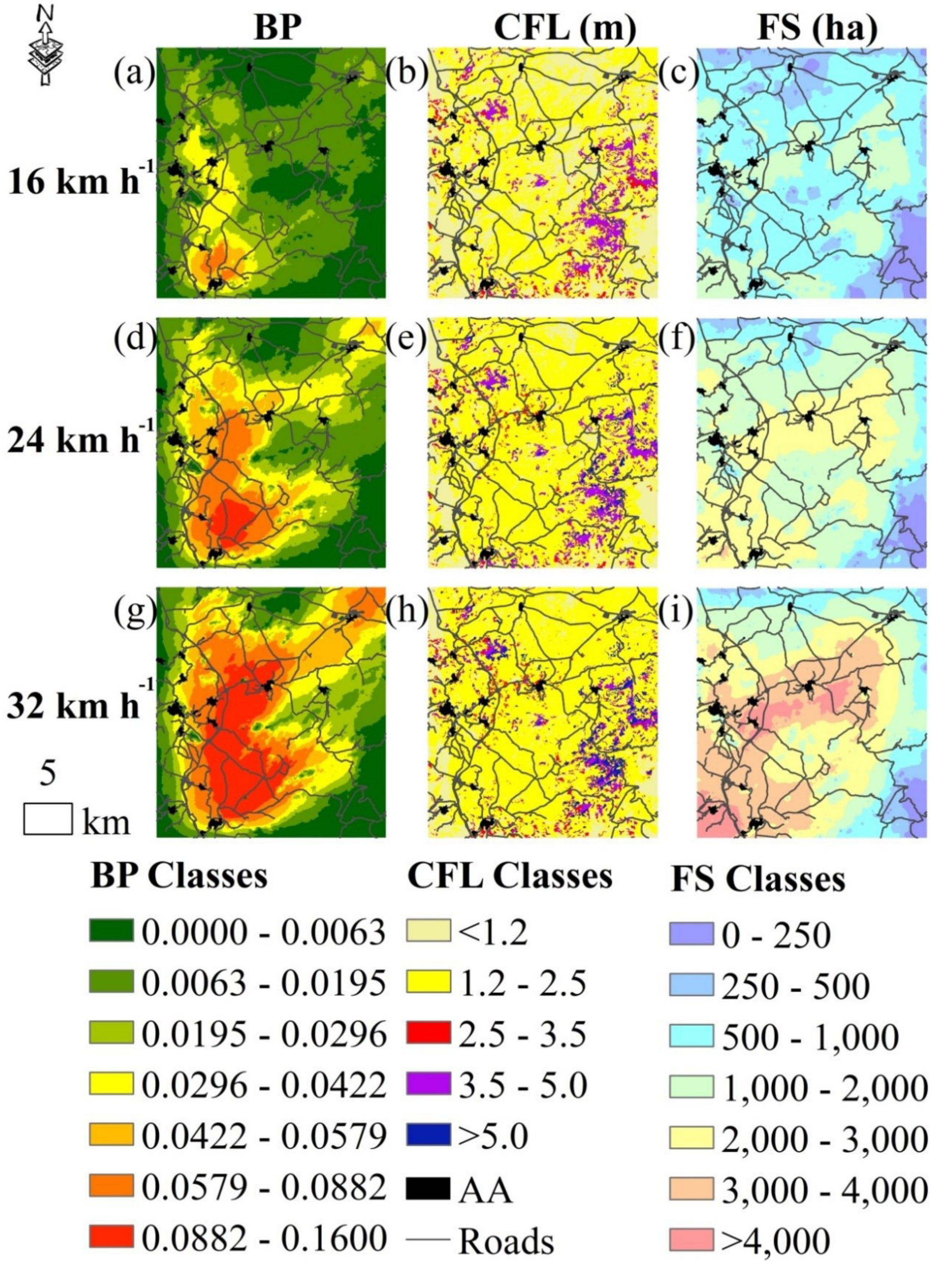

Fig. 4. Maps of burn probability (BP (a, d, g)), conditional flame length (CFL (b, e, h)) and fire size (FS (c, f, i)) for the NO-TREAT condition, considering different wind speed conditions (16, 24 and 32 km h^{-1}).

Table 2. Average and standard deviation values of burn probability (BP) at the landscape scale for each fuel treatment alternative, percentage of landscape treated and wind speed condition.

Wind speed (km h^{-1})	**Landscape treated (%)**	**Fuel treatment alternative**				
		NO-TREAT	LOW	MED	LAR	ROAD
16		1.36E-02 ±1.22E-02				
	2%		1.31E-02 ± 1.15E-02	1.31E-02 ± 1.18E-02	1.31E-02 ± 1.21E-02	1.28E-02 ± 1.17E-02
	5%		1.24E-02 ± 1.13E-02	1.24E-02 ± 1.17E-02	1.19E-02 ± 1.13E-02	1.14E-02 ± 1.12E-02
	8%		1.17E-02 ± 1.08E-02	1.11E-02 ± 1.05E-02	1.09E-02 ± 1.04E-02	9.33E-03 ± 1.03E-02
24		2.84E-02 ±2.33E-02				
	2%		2.77E-02± 2.38E-02	2.74E-02 ± 2.27E-02	2.71E-02 ± 2.24E-02	2.65E-02 ± 2.30E-02
	5%		2.54E-02 ± 2.15E-02	2.56E-02 ± 2.29E-02	2.47E-02 ± 2.18E-02	2.31E-02 ± 2.19E-02
	8%		2.41E-02 ± 2.12E-02	2.31E-02 ± 2.04E-02	2.20E-02 ± 2.08E-02	1.66E-02 ± 1.78E-02
32		4.42E-02 ± 3.61E-02				
	2%		4.21E-02 ± 3.51E-02	4.19E-02 ± 3.50E-02	4.20E-02 ± 3.60E-02	4.07E-02 ± 3.47E-02
	5%		3.89E-02 ± 3.39E-02	3.87E-02 ± 3.45E-02	3.74E-02 ± 3.33E-02	3.38E-02 ± 3.22E-02
	8%		3.62E-02 ± 3.15E-02	3.42E-02 ± 3.00E-02	3.27E-02 ± 3.06E-02	2.22E-02 ± 2.45E-02

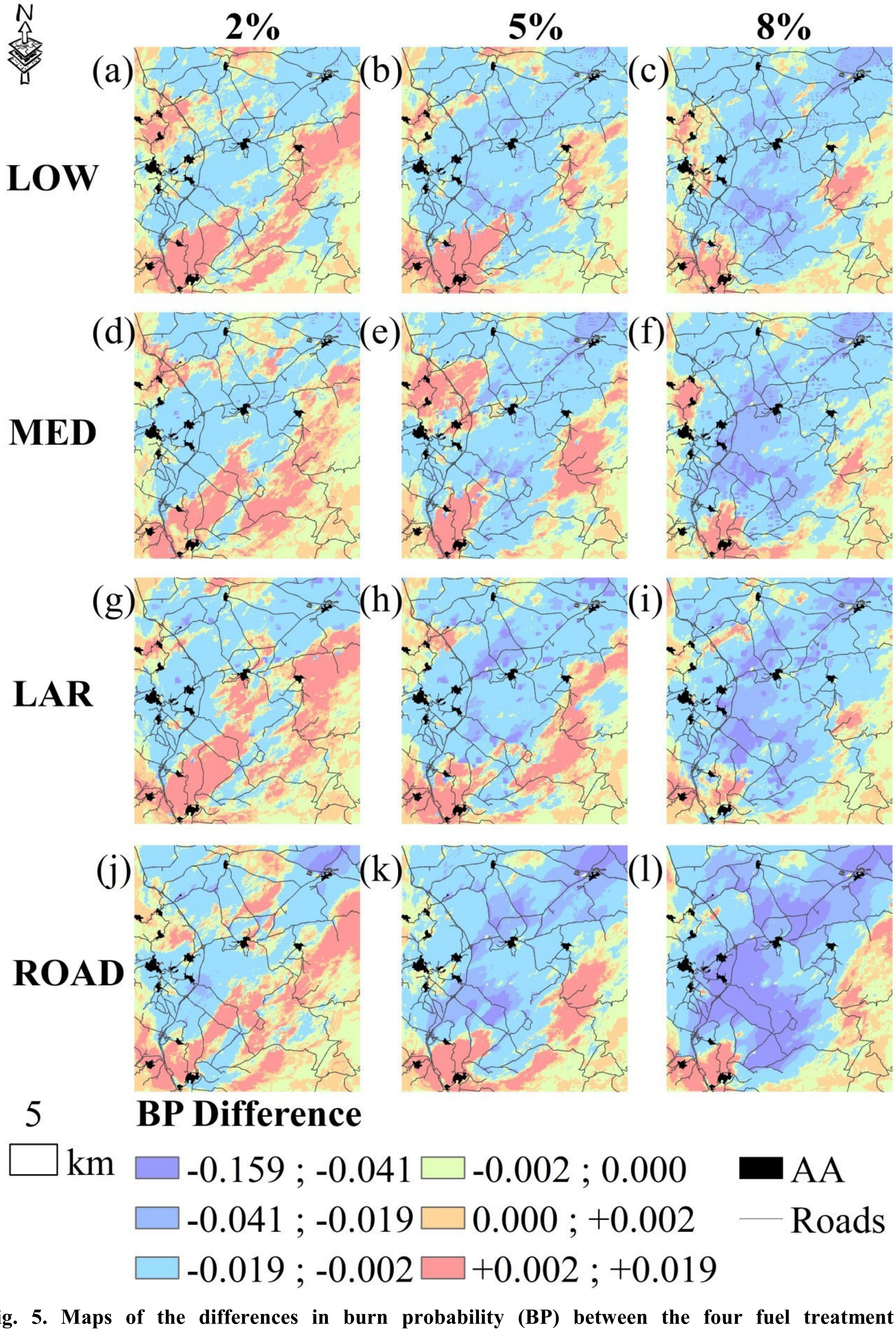

Fig. 5. Maps of the differences in burn probability (BP) between the four fuel treatment alternatives (LOW, MED, LAR, ROAD) and the NO-TREAT condition, considering the three percentages of landscape treated (2%, 5%, and 8%), and a wind speed of 32 km h^{-1}.

2.3.1.2 Fire size

The highest FS value was about 5200 ha and was observed for the NO-TREAT condition and the highest wind speed value (Table 4 and Fig. 4). As observed for BP outputs, FS was also strongly influenced by the percentage of the landscape treated, treatment strategy, and wind speed (Tables 3 and 4). The Wilcoxon test showed that all the differences between treatments and the control were significant with the exception of a few pairwise comparisons for the 2% of treated area scenario (Table 3). Under the NO-TREAT condition wind speed increased the average FS values at the landscape scale from 769 ha (16 km h^{-1}) to 1,555 ha (24 km h^{-1}) to 2,326 ha (32 km h^{-1}) (Table 4). The treatment strategies tested decreased average FS even at the lowest percentages of area treated. Again, average FS decreased with increasing percentage of the landscape treated (Table 4), with the ROAD strategy the most efficient in limiting fire growth. In fact, among the fuel treatment alternatives tested, the spatial arrangement associated with the ROAD strategy was able to promote the most relevant reductions in average FS for all wind speed and treatment intensities. For instance, at the highest values of wind speed and 8% of the landscape treated, average FS dropped from 1,879 ha with the LOW strategy to 1,193 ha with the ROAD strategy. In addition, at the highest wind speed value, the ROAD strategy guaranteed a reduction in average FS values compared to the NO-TREAT condition close to 10%, 25% and even 50% for treatment intensities of 2%, 5% and 8%, respectively (Table 4). At the lowest wind speed, treating 8% of the area with the ROAD strategy decreased the number of very large fires (>1,000 ha) by about 60% with respect to NO-TREAT. As observed for BP, we also found that for all scenarios tested average FS values decreased moving from low to large size treatment unit alternatives (Table 4). The maps of the differences in FS between the whole set of fuel treatment alternatives and NO-TREAT conditions for the study area are presented in Fig. 6.

2.3.1.3 Conditional flame length

As far as CFL is concerned, the effects of fuel treatment alternatives in reducing flame length at the landscape scale compared to the control condition were much more limited than those observed for BP and FS (Tables 3 and 5). Overall, treating 2% of the landscape did not produce significant differences between NO-TREAT and the diverse strategies, while 5% and 8% of area treated always produced significant differences with respect to NO-TREAT. The highest average CFL values were in general observed

Table 3. P-values for the pairwise comparisons between NO-TREAT and the fuel treatment alternatives for BP, FS and CFL values and the wind speed conditions tested in this study. Significant differences between NO-TREAT and treatment alternatives were indicated by an asterisk (P-value <0.05). Pairwise comparisons were carried out by the nonparametric Wilcoxon signed rank test.

TREATMENT PAIRWISE COMPARISON	BP			FS			CFL		
	Wind Speed (km h^{-1})								
	16	24	32	16	24	32	16	24	32
NO-TREAT vs LOW-2%	0.051	0.08	0.000*	0.000*	0.000*	0.000*	0.16	0.11	0.030*
NO-TREAT vs MED-2%	0.07	0.008*	0.000*	0.000*	0.000*	0.000*	0.32	0.051	0.021*
NO-TREAT vs LAR-2%	0.030*	0.000*	0.000*	0.000*	0.000*	0.000*	0.06	0.053	0.004*
NO-TREAT vs ROAD-2%	0.000*	0.000*	0.000*	0.000*	0.000*	0.000*	0.07	0.09	0.010*
NO-TREAT vs LOW-5%	0.000*	0.000*	0.000*	0.000*	0.000*	0.000*	0.000*	0.000*	0.000*
NO-TREAT vs MED-5%	0.000*	0.000*	0.000*	0.000*	0.000*	0.000*	0.000*	0.000*	0.000*
NO-TREAT vs LAR-5%	0.000*	0.000*	0.000*	0.000*	0.000*	0.000*	0.000*	0.000*	0.000*
NO-TREAT vs ROAD-5%	0.000*	0.000*	0.000*	0.000*	0.000*	0.000*	0.000*	0.000*	0.000*
NO-TREAT vs LOW-8%	0.000*	0.000*	0.000*	0.000*	0.000*	0.000*	0.000*	0.000*	0.000*
NO-TREAT vs MED-8%	0.000*	0.000*	0.000*	0.000*	0.000*	0.000*	0.000*	0.000*	0.000*
NO-TREAT vs LAR-8%	0.000*	0.000*	0.000*	0.000*	0.000*	0.000*	0.000*	0.000*	0.000*
NO-TREAT vs ROAD-8%	0.000*	0.000*	0.000*	0.000*	0.000*	0.000*	0.000*	0.000*	0.000*

In the NO-TREAT condition, whose average values ranged from 1.28m (16 km h^{-1}) to 1.58m (32 km h^{-1}) (Table 5). Moreover, for the NO-TREAT condition, the area with CFL above 2.5m increased of about 12% and 20% moving from 16 to 24 and to 32 km h^{-1} wind speed, respectively. All the treatment alternatives decreased average CFL slightly with respect to NO-TREAT. Average CFL decreased with increasing percentage of the landscape treated and moving from the highest to the lowest wind intensities, as expected (Table 5). The ROAD strategy was the most efficient spatial arrangement of fuel treatment units in reducing fire intensity, even if the differences at the landscape scale with the other treatments were quite small. For instance, at the highest values of wind speed and 8% of the landscape treated, average CFL moved from 1.49m (LOW strategy) to 1.42m (ROAD strategy). The highest CFL values were observed in the south-western zone of the study area, corresponding to forests and shrublands and complex topography (Fig. 6). The maps of the differences between fuel treatment alternatives and NO-TREAT conditions are presented in Fig. 7.

2.3.2 Wildfire exposure to anthropic values

Scatterplots of average BP vs. CFL, for FS levels, for the buffer areas of the selected anthropic features showed considerable variation in exposure factors among and within features in terms of magnitude and spatial pattern depending on fuel treatment alternative, area treated, and wind speed (Fig. 8). Overall, the fuel treatment strategies that focused on treating nearby roads (ROAD) were highly efficient in protecting the vicinity of the selected anthropic values, while the LOW strategy was often the least efficient one (Fig. 8). In some cases, the ROAD strategy applied over 5% of the study area was even more effective in reducing BP and FS than the other strategies applied to 8% of the landscape, and this effect was particularly strong at the highest wind speed. On the whole, as observed at landscape scale, only ROAD treatments when applied to 8% of the study area clearly maximized the reduction in exposure factors in the proximity of all the selected values. Also at the anthropic values scale, the increase in the area treated resulted in significant benefits by dropping the average BP and FS. In addition, as expected, the shift from 16 to 32 km h^{-1} wind speed caused positive variation in the fire exposure factors. In fact, for all fuel treatment alternatives, burn probability, flame length and fire size showed the highest values under the most intense winds. Only in a few cases, and only at the lowest wind speed conditions and percentages of area treated, protection of areas near anthropic values was not enhanced

Table 4. Average and standard deviation values of fire size(FS) at the landscape scale for each fuel treatment alternative, percentage of landscape treated and wind speed condition.

Wind speed (km h^{-1})	Landscape treated (%)	Fuel treatment alternative				
		NO-TREAT	LOW	MED	LAR	ROAD
16		769± 302				
	2%		743± 300	740± 301	732± 304	723± 303
	5%		704± 300	696± 300	677± 297	641± 301
	8%		662± 287	640± 295	628± 292	543± 303
24		1555± 681				
	2%		1503± 664	1494± 662	1482± 673	1440± 671
	5%		1396± 626	1385± 631	1348± 635	1234± 628
	8%		1317± 596	1281± 592	1213± 579	939± 571
32		2326± 1178				
	2%		2211± 1142	2193± 1144	2186± 1155	2105± 1133
	5%		2040± 1066	2011± 1063	1950± 1059	1719± 984
	8%		1879± 962	1820± 940	1727± 915	1193± 788

by the fuel treatment alternatives in terms of BP and FS compared to NO-TREAT (Fig. 8). Focusing on the selected anthropic values, we found that continuous urban fabrics (CUF) were the most exposed category in terms of average CFL and FS for all the scenarios analyzed, as well as for most of the simulations when considering average BP. On the contrary, industrial and commercial units (ICU) and discontinuous urban fabrics (DUF) experienced the lowest values of CFL and BP, respectively, for almost all scenarios tested. Due to the high presence of herbaceous fuels in the study area and the type of treatments performed, the effects of wind speed, area treated and spatial arrangement of fuel treatments on BP and FS were more evident than those on CFL. For instance, considering the NO-TREAT scenario, average BP for the selected anthropic values ranged from a low of 0.0147 for DUF with 16 km h^{-1} wind speed to a maximum of 0.0544 for SGU with 32 km h^{-1} wind speed conditions. As far as CFL is concerned, focusing on the NO-TREAT scenario, the values ranged from 1.04m for ICU with 16 km h^{-1} wind speed to 1.49m for CUF with 32 km h^{-1} wind speed.

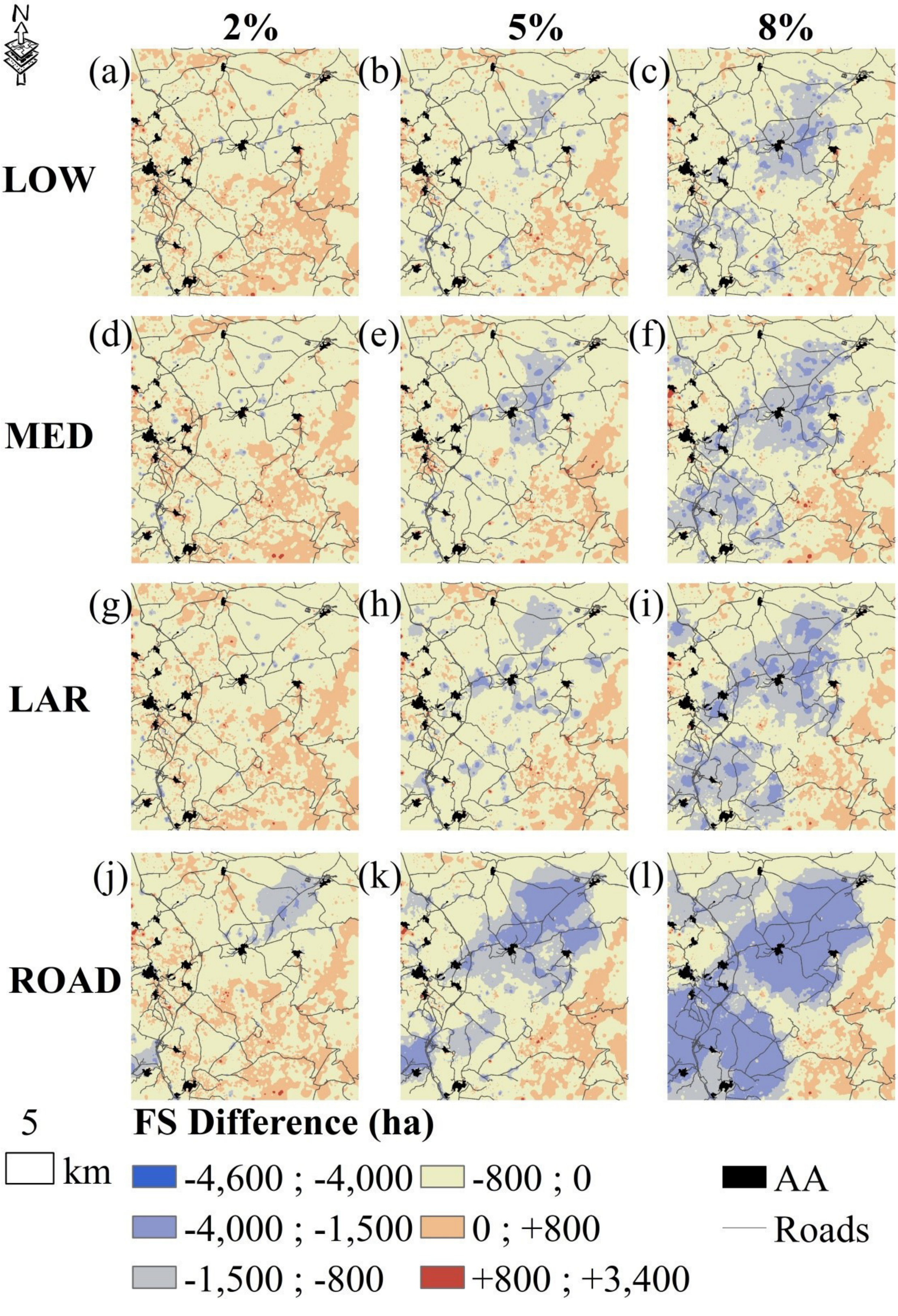

Fig. 6. Maps of the differences in fire size (FS) between the four fuel treatment alternatives (LOW, MED, LAR, ROAD) and the NO-TREAT condition, considering the three percentages of landscape treated (2%, 5%, and 8%), and a wind speed of 32 km h^{-1}.

2.4 Discussion

In this study, we performed fire spread simulations based on the MTT algorithm to test the response of wildfire exposure variables (namely burn probability, flame length and fire size) to variations in percentage of area treated and spatial arrangement of fuel treatments in a fire-prone Mediterranean area. The study area has large portions of land covered by herbaceous surface fuels, mainly related to agricultural (autumn-winter crop productions) and pastoral uses for animal feed (herbaceous and open wooded pastures, as well as degraded shrublands), and for these reasons represents a relevant example of dry Mediterranean agro-pastoral landscapes.

We found that strategic fuel treatments designed near roads were the most effective in limiting fire growth for all wind speed conditions and percentages of area treated. Similar findings were obtained in previous work conducted in a Mediterranean landscape (Northern Sardinia, Italy) mainly covered by oak forests and shrublands (Salis et al., 2016b). However, the use of a low spotting probability (1%) in our study could have increased the effectiveness of continuous fuel treatments nearby roads vs. other patchy arrangements. Linear fuel break networks have also been suggested to be more efficient and cost-effective than dispersed fuel treatments by Fernandes et al. (2012) and Oliveira et al. (2016). On the whole, this opens many options for roads being used as preferential fire control lines when the road network sufficiently covers a given landscape (Eastaugh and Molina, 2012; Gill, 2008; Price and Bradstock, 2010), even considering that road networks can limit fire spread both through creation of fuel breaks and by favoring placement of fire management resources (Narayanaraj and Wimberly, 2011). The fact that the ROAD treatment strategy was the most effective solution to mitigate fire size and propagation could strengthen regional fire regulation and planning guidelines (Sardinia Regional Government, 2017), which impose fuel management in the vicinity of the road network as a general wildfire prevention activity. On the other hand, to achieve significant results, it would be more appropriate to expand road treatment buffers (e.g.: 100m buffers): this would be crucial especially in strategic locations or hot-spot areas (Ager et al., 2013; Eastaugh and Molina, 2012; O'Connor et al., 2017; Oliveira et al., 2016). Plus, managing fuels around roads enhances the prevention of arson and accidental fire ignitions (e.g.: cigarettes), largely increases the potential of roads to act as barriers even in case of spotting, can make firefighting operations more effective, and increases the size of safe zones or escape routes

Table 5. Average and standard deviation values of conditional flame length (CFL) at the landscape scale for each fuel treatment alternative, percentage of landscape treated and wind speed condition.

Wind speed ($km\ h^{-1}$)	**Landscape treated (%)**	**Fuel treatment alternative**				
		NO-TREAT	LOW	MED	LAR	ROAD
16		1.28±0.88				
	2%		1.26± 0.90	1.26± 0.89	1.26± 0.89	1.26± 0.90
	5%		1.23± 0.91	1.24± 0.91	1.23± 0.92	1.22± 0.92
	8%		1.21± 0.92	1.21± 0.93	1.20± 0.94	1.19± 0.94
24		1.46± 1.04				
	2%		1.45± 1.06	1.45± 1.05	1.44± 1.06	1.45± 1.05
	5%		1.42± 1.07	1.41± 1.07	1.41± 1.08	1.40± 1.08
	8%		1.39± 1.08	1.39± 1.09	1.38± 1.09	1.35± 1.09
32		1.58± 1.10				
	2%		1.57± 1.11	1.57± 1.12	1.56± 1.11	1.55± 1.11
	5%		1.52± 1.14	1.52± 1.14	1.52± 1.13	1.50± 1.14
	8%		1.49± 1.15	1.48± 1.15	1.47± 1.15	1.42± 1.14

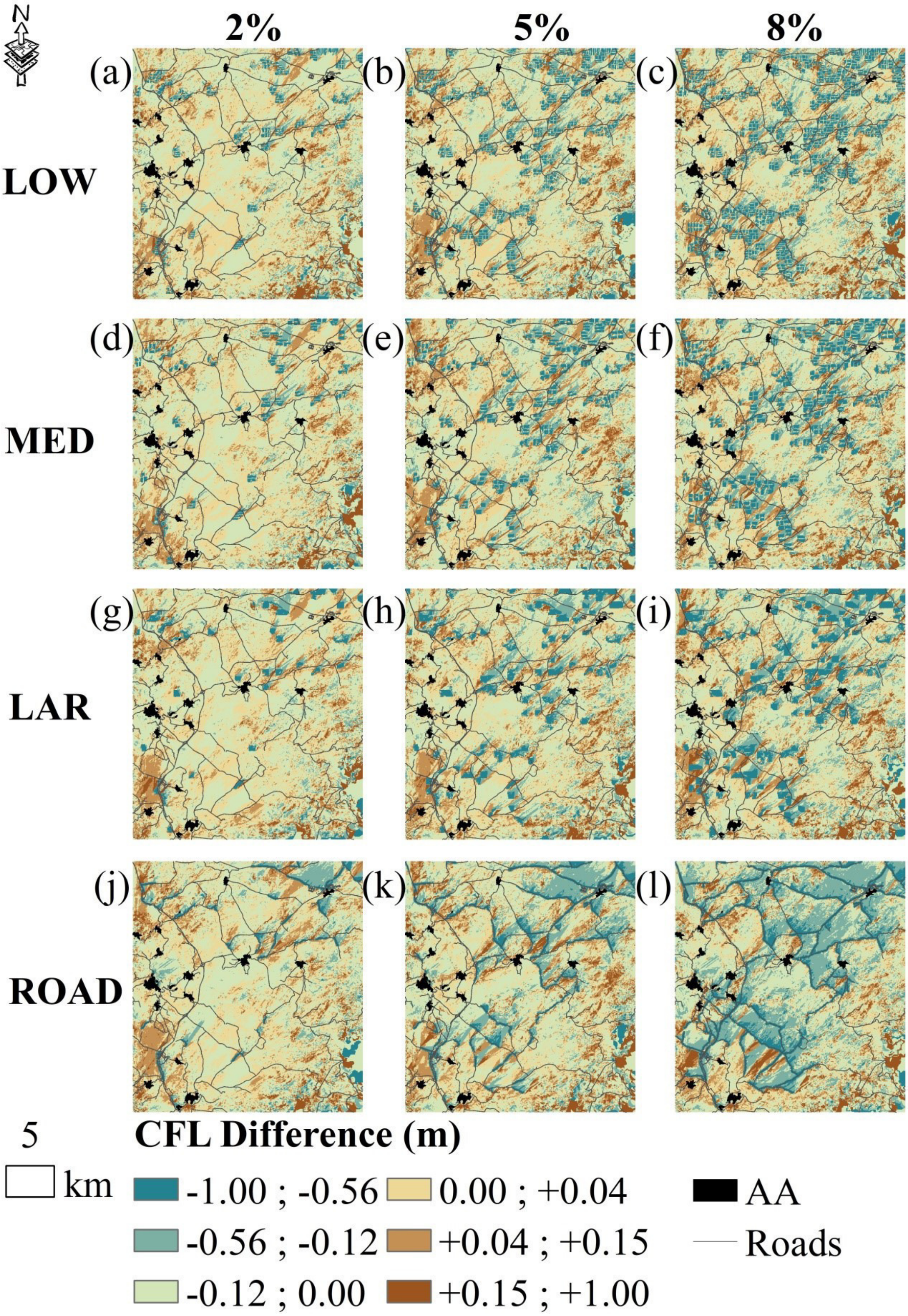

Fig. 7. Maps of the differences in conditional flame length (CFL) between the four fuel treatment alternatives (LOW, MED, LAR, ROAD) and the NO-TREAT condition, considering the three percentages of landscape treated (2%, 5%, and 8%), and a wind speed of 32 km h^{-1}.

(Weatherspoon and Skinner, 1996; Catry et al., 2009; Ganteaume et al., 2013; O'Connor et al., 2017; Xanthopoulos et al., 2006). Regarding this last point, it is dramatically remarkable that the large majority of the victims of the Portugal wildfire events of June 2017 lost their lives nearby roads, and that 30 people lost their lives in a single road section of about a 400 m-length (Viegas et al., 2017). Even if it is likely that slower fire growth rates and the increased presence of unburnable areas after fuel treatment would have improved fire suppression capacity and safety, we did not take into consideration fire suppression in the fire modeling exercise. This was due to the fact that: 1) current fire suppression operations in Mediterranean areas mainly focus on civil protection issues and disregard fire perimeter control (Beighley and Quesinberry, 2004; Oliveira et al., 2016), and 2) coordinating suppression activities based on fuel management infrastructures during large events is challenging (Finney and Cohen, 2003; Keeley, 2002; Oliveira et al., 2016; Rigolot and Viegas, 2002). Yet, as indicated by Oliveira et al. (2016), the high costs of fuel management strategies require that fire suppression operations take advantage of the presence of treated areas to reduce area burned beyond a passive effect.

We observed a general pattern in terms of treatment effectiveness related to single land use size (LAR, MED and LOW strategies): overall, the smallest treatment units (LOW strategy) were less effective than the largest (LAR strategy) in their effect on fire spread. This points out that, in agro-pastoral areas and for treatments that convert treated fuels to non-burnable state, the creation of large and extended fuel treatment units (unit size 25-50 ha) ensures a greater efficiency in reducing fire exposure with respect to small treatment units (0.5-10 ha). Moreover, from an operational point of view, the LAR strategy is more cost-effective, less time-consuming and easier to implement, as it concentrates fuel management operations in well-defined large areas. The fact that the LAR strategy was superior to the other two could be related to the reduction in fuelbrand overflight possibilities and the associated ignition of spot fires, as well as to an enhanced potential to block heading fire spread and to enable mostly flanking propagation (Finney, 2007).

As expected, we found that, apart from the fuel treatment strategy, the increase in the percentage of landscape treated (from 2% to 8%) resulted in a reduction of fire exposure indicators. Our results highlighted that in several cases treating 5% of the landscape using the ROAD strategy was more efficient than treating 8% of the study area with other strategies, even at the lowest wind speed conditions. Although we were aware that the increase in the treated areas would have positively influenced the potential to limit fire propagation, we chose to treat relatively small areas (2, 5, and 8% of the landscape), considering that, as indicated by previous work and according to local land managers' indications, performing fuel treatments for vast portions of land (e.g.: >10% of a study area) is very challenging or even practically impossible (Calkin et al., 2014; Finney, 2007; Moghaddas et al., 2010). As supported by other studies (Ager et al., 2007; Bradstock et al., 2012; Price, 2012; Salis et al., 2016b; Syphard et al., 2011), treating a small proportion of the landscape (2%) resulted in minimal reduction in wildfire exposure profiles and potential area burned. Yet, preliminary simulations (treated landscape=0.5% and 1%) showed very limited or null differences among treatment strategies and NO-TREAT conditions in terms of BP, CFL and FS. Despite this, our work showed that even treating low percentages of the landscape (e.g.: 5% of the study area) can provide excellent results in limiting fire growth when combined with an efficient localization of fuel treatments (e.g.: ROAD strategy).

The results revealed significant variation in the fire exposure profiles in relation to wind intensity, with an apparent increase in the average values of BP, CFL and FS at both the landscape and selected anthropic values scales as wind speed increased. Simulating fire growth and behavior under severe weather conditions such as intense winds can help identify wildfire preferential pathways and hot-spot areas, or estimate potential losses from fires. This is relevant in the light of climate change and the increased frequency of extreme weather (EEA, 2017). Furthermore, testing different wind intensity conditions highlighted how diverse fuel treatment strategies and treated area percentages would be able to lower fire growth and behavior. As a general rule, fires burning under mild wind speed conditions and low percentages of area treated are less affected by the spatial pattern of fuel treatments because fire growth is smaller and the relative spread rates in the treatment scenarios are not dissimilar to those in the untreated condition (Ager et al., 2010; Finney, 2001). From this point of view, our findings confirmed that the differences in the effectiveness of the fuel treatment scenarios were accentuated by stronger wind conditions (32 km h^{-1}), that is by those conditions associated with the major extreme-behavior fires that could overcome the suppression capabilities of firefighters.

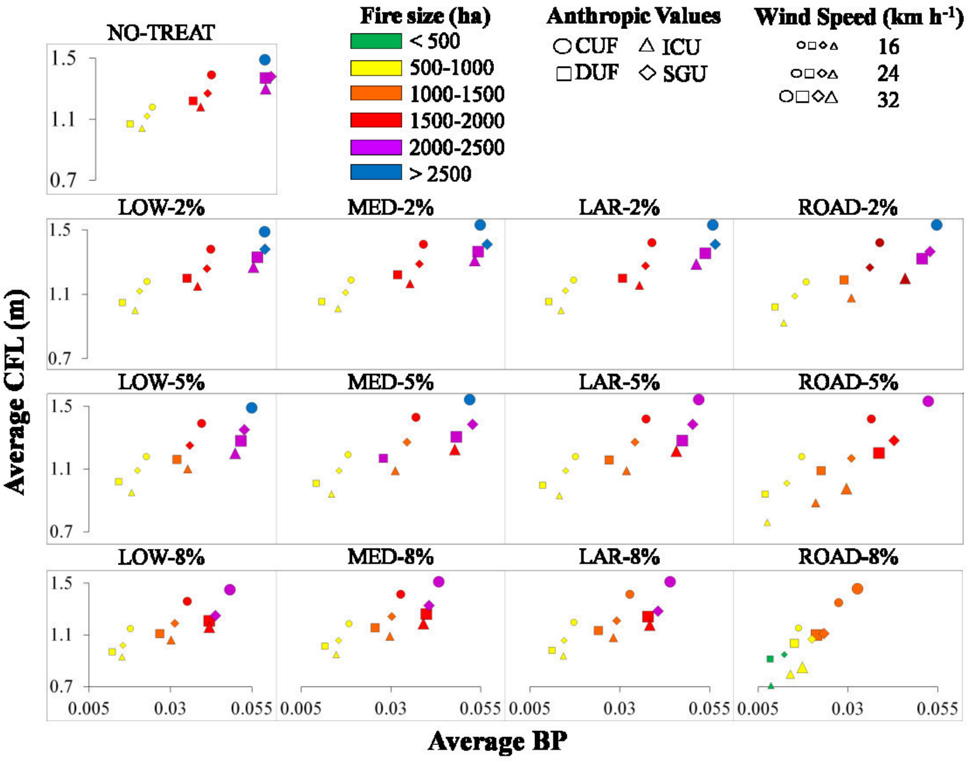

Fig. 8. Scatterplots of average burn probability (BP) vs average conditional flame length (CFL) in the vicinity (150-m buffer) of selected anthropic values (continuous urban fabric (CUF), discontinuous urban fabric (DUF), industrial and commercial units (ICU), and sport and green urban areas (SGU)). We show the results obtained for the whole set of fuel treatment alternatives and wind speed conditions analyzed in this study. Each symbol is colored and symbolized according to the average simulated fire size and wind speed scenario, respectively.

In this work, we simulated treatments on herbaceous land use classes to change the treated units to unburnable areas. The treatments simulated (prescribed burning, superficial tillage, and summer irrigation, depending on the land use type) are overall low-cost treatments and could be financially supported by specific EU rural policies and programs (European Commission, 2017) with the aim of preserving and enhancing ecosystems related to agriculture and forestry. In fact, we show that efficient fuel treatment mosaics can limit wildfire growth and behavior, and can therefore reduce both wildfire losses and suppression costs (e.g.: less aerial interventions). If financially supported, the above treatments can also produce positive economic, social and ecological effects on fire-prone Mediterranean areas by linking preventive actions to EU payments to local farmers. Furthermore, fuel management approaches can reduce the relevant gap between fire prevention and suppression in terms of organizational hierarchy and budget (Bovio et al., 2017; Gebert et al., 2007; Gonzalez-Olabarria and Pukkala, 2011; Thompson et al., 2013). The effects of fuel treatments on fire spread and exposure that we tested in this study are only temporary. For instance, the possibility of vegetation resprouting or germination of annual herbs after tilling and/or prescribed burning performed in mid-late June in Sardinia, as well as in other dry climate Mediterranean Basin areas, is typically very low, particularly in terms of the potential to create a continuous surface fuel bed able to support surface fires. This is mostly due to the fact that rain events, from June until September (which is the typical fire season period), are quite rare and limited in terms of total amount, and the maximum temperatures are often above 30° during summer, which limits soil water content and plant resprouting or growth in that period of the year, after the treatments. The limited longevity of individual treatments would therefore impose a scheduled program of summer irrigation or late-spring prescribed burning. Regarding the latter point, land managers could also promote the selection of land use units according to a 2-3 year spatial rotation criteria, and dynamic single treatments units could be added to priority fuel management target areas.

The use of grazing animals as a cost-effective, non-toxic, and non-polluting solution for reducing 1-hr and 10-hr fuel loads and continuity and limiting fire behavior was proposed in previous work for different ecosystems (Diamond et al., 2009; Franca et al., 2012; Green and Newell, 1982; Hart, 2001; Lovreglio et al., 2014; Ruiz-Mirazo and Robles, 2012). However, several Sardinian wildfires were found to spread fast in grazed areas, and in recent years the largest wildfire events on the island were not blocked but only slowed down when they encountered grazed landscapes (Nudda et al., 2015, 2016, 2017; Salis et al., 2012). In addition, in Mediterranean areas, common concerns with herbivores are mostly related to overgrazing, soil erosion and even degradation of shrublands and forests, particularly for goats (Caballero et al., 2009; Kairis et al., 2015; Vacca et al., 2003). For the above reasons, we did not use grazing as a preferential fuel treatment option.

The application of fire spread models, previously calibrated and validated for Mediterranean fire-prone ecosystems and landscapes, may help in designing optimized fuel management strategies and spatial arrangements, as well as prioritizing the most exposed areas. The methodology proposed in this paper can be replicated in other Mediterranean areas and elsewhere and simulates diverse fuel management scenarios while analyzing their performance and effectiveness by objective measures like burn probability, fire intensity and fire size. The proposed approach could have a large application in Sardinia, as the most recent regional programs for rural and inner areas development, as well as forest and fire management directives and planning, highlight the relevance of fire prevention and land management to reduce wildfire risk, preserve valued landscapes and ecosystems, promote the multifunctional use of agricultural areas, and protect anthropic values under current conditions and those expected in the future under climate change (Sardinia Regional Government, 2014, 2016, 2017). Likewise, ongoing regional fuel treatment programs aimed at reducing fire risk are mainly based on expert-based evaluations and decisions and are limited by a number of constraints, and could benefit from a large-scale, comprehensive and optimized spatial design of fuel treatments according to preliminary quantitative assessments of fuel treatment effects on wildfire spread and behavior. Yet, assessing quantitatively wildfire exposure over large landscapes remains challenging, since several factors that affect fire ignition, spread and suppression potential are difficult to assess (Ager et al., 2014; Calkin et al., 2015; Fernandes, 2013). In addition, even if the MTT fire models family (FSim, FSPro, Flammap, Randig) was proved to have potential in quantitatively replicating large wildfires, in terms of predicting potential area burned, size and shape of perimeters, or potential burn probability and fire intensity (e.g.: Ager et al., 2014; Alcasena et al., 2016; Finney et al., 2011; Salis et al., 2013), these models have a number of limitations. For instance, in this work: (i) fire-atmosphere interactions are not modeled, so that crown fire activity, spotting phenomena and spread rates may have been underestimated with respect to actual events (Cruz and Alexander, 2010); (ii) the spatial input data used for surface and crown fuels were assigned according to Corine land-cover classes and forest inventory data, which can add additional uncertainty; (iii) the 25-m spatial resolution may not fully capture fine scale fuel bed characteristics and conditions of both treated and untreated areas; and (iv) a 1% constant spot probability for the three wind speed scenarios might represent a simplified condition.

2.5 Conclusions

This work presents a wildfire exposure assessment framework, based on the MTT fire spread algorithm, that characterizes the performance of diverse fuel treatment mosaics related to diverse spatial arrangement strategies for limiting wildfire spread in an agro-pastoral Mediterranean area. The proposed approach highlights the variation in wildfire exposure profiles due to different treatment scenarios and differentiates the strategies according to their effectiveness using an objective quantitative assessment approach. We demonstrate that fuel treatment buffers surrounding the road network represent the most efficient spatial strategy for herbaceous fuel type dominated landscapes. The methodology and the findings of this work can provide guidelines and suggestions for land managers and policy makers in the study area and neighboring Mediterranean areas, particularly for rangelands and wooded pastures (e.g., dehesas or montados). A number of considerations, preferences and constraints used in this study for the spatial localization, priorities and objectives of fuel treatments has the potential to be finely tuned for strategic planning of landscape scale fuel treatments and fire management programs. This work increases knowledge and awareness of spatial arrangements of fuel treatments in herbaceous areas with limited portions of land to be treated, and may support the identification and planning of the most effective strategies and spatial locations of fuel treatments.

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Supplementary data

Table 1. Fuel model and canopy data used for the wildfire simulations. The data refer to the 625 km^2 study area. FT = fuel type; FL = fuel load; FD = fuel depth; CH = average canopy height; CBD = average canopy bulk density; CBH = average canopy base height; CC = canopy cover. Treated units (TU) were considered as non-burnable fuels.

FM CODE	CORINE CODES	FT	AREA (%)	DEAD FL ($t\ ha^{-1}$)	LIVE FL ($t\ ha^{-1}$)	FD (cm)	CH (m)	CBD ($100^*\ kg\ m^{-3}$)	CBH (m)	CC (code)
FM21	1	AA	2.33	-	-	-	-	-	-	-
FM22	4; 5	W	0.10	-	-	-	-	-	-	-
FM23	332	R	0.01	-	-	-	-	-	-	-
FM25	211; 212; 213; 231	GR	46.06 [a]	1.2	0.0	20	0	0	0	0
FM26	241; 242; 243; 244	MA	13.26 [a]	1.2	0.0	30	0	0	0	0
FM27	221; 222; 223	VO	1.85	1.0	2.0	80	10	11	1	1
FM28	321	HP	8.68	2.5	0.0	35	0	0	0	0
FM29	333; 334	G	0.81	5.3	4.1	45	0	0	0	0
FM30	322; 323; 324	MM	9.48	15.0	12.5	135	12	14	1	1
FM31	312	CF	0.18	10.0	1.0	25	14	11	2	4
FM32	311	BF	17.19	12.0	2.0	70	12	14	2	3
FM33	313	MF	0.04	12.0	2.0	70	14	13	2	3
FM41	241, 211, 212	TU	- [a]	-	-	-	-	-	-	-

[a] These values refer to the untreated condition (NO-TREAT)

Table 2. Total area treated by Corine classes for each fuel treatment strategy and percentage of area treated at landscape scale. The Corine codes refer to annual crops with permanent crops (241), non-irrigated arable land (211), and permanently irrigated land (212). According to the 2008 Sardinia Land Use Map and focusing on areas with terrain slope below 10°, the treatable areas of the above Corine classes cover respectively 6.70%, 30.55%, and 10.15% of the whole study area

Fuel Treatment Strategy	Corine Code	Area Treated			
		NO-TREAT	2%	5%	8%
LOW					
	211	0.00%	1.05%	2.86%	4.78%
	212	0.00%	0.75%	1.57%	2.30%
	241	0.00%	0.20%	0.56%	0.91%
MED					
	211	0.00%	0.99%	2.66%	4.74%
	212	0.00%	0.83%	1.75%	2.35%
	241	0.00%	0.18%	0.59%	0.91%
LAR					
	211	0.00%	1.04%	2.74%	4.70%
	212	0.00%	0.82%	1.74%	2.41%
	241	0.00%	0.14%	0.53%	0.89%
ROAD					
	211	0.00%	1.02%	2.85%	5.27%
	212	0.00%	0.88%	1.76%	2.02%
	241	0.00%	0.10%	0.39%	0.71%

Chapter 3: Linking burn probability and erosion models to quantify post-fire erosion risk: a case study from Northern Sardinia, Italy

3.1 Introduction

In Sardinia, Italy, roughly 3,000 wildfires occur every year which burn about 15,000 ha (Nudda *et al.* 2016). Although there has been a reduction in fire numbers and area burned during recent years in comparison to 1970-1990 wildfire seasons, the occurrence of extreme weather events, concomitant high wildfire ignitions and increased fuel loads due to land abandonment has resulted in high severity events and mega-fires, such as those happened in 2009 in Sardinia (Salis *et al.* 2014, 2018). Large fire events are also occurring for neighboring areas (Viegas *et al.* 2017; Ruffault *et al.* 2018; San-Miguel-Ayanz *et al.* 2018). Future climate and socio-economic changes are expected to further influence the risks posed by large, severe wildfires in Mediterranean forests and shrublands (Brotons *et al.* 2013; Chergui *et al.* 2017; Lozano *et al.* 2017; Turco *et al.* 2018). These high-severity wildfires can be responsible for several negative impacts on ecosystems (DeBano *et al.* 1998; Certini 2005). Among these impacts, several researchers emphasized the effects on soils, which are affected by the fire removal of the vegetative cover, the creation or enhancement of water repellent soil layers resulting in increases in surface runoff and erosion potential (Cerdá and Doerr 2007; Larsen *et al.* 2009; Shakesby 2011; Robichaud *et al.* 2013; Fonseca *et al.* 2017). Large and severe wildfires are a major threat to watershed health, because they can alter hydrologic and geomorphic processes, and can lead to changes 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 recognized that the impacts of wildfire on hydrology and geomorphology depends 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). For instance, intense rainstorms following wildfires can promote the risk of extensive flooding and high sediment delivery (Onodera and Van Stan 2011; Sankey *et al.* 2017; Srivastava *et al.* 2018). For the above reasons, there is need to evaluate post-wildfire erosion risks across landscapes and to design mitigation strategies accordingly (Robichaud and Ashmun 2013; Thompson *et al.* 2013). In the post-fire context, the priority is watershed stabilization and rehabilitation efforts within the area burned; whereas in the pre-fire context, the exact timing and location of the wildfires are uncertain and require subjective or stochastic approaches (Scott *et al.* 2012; Hyde *et al.* 2017). A common pre-fire risk mitigation approach which has been adopted worldwide is hazardous fuels reduction treatments, which can be designed to reduce fire intensity and severity within treated areas, as well as to lessen the likelihood of high burn probability and fire intensity outside of treated areas (Ager *et al.* 2014; Buckley *et al.* 2014; Sidman *et al.* 2015; Elliot *et al.* 2016; Vaillant and Reinhardt 2017). Overall fuel reduction treatments have been shown to be effective in modifying fire behavior and burn probability in Mediterranean areas and elsewhere (Reinhardt *et al.* 2008; Ager *et al.* 2010; Cochrane *et al.* 2012; Chung *et al.* 2015; 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; Xanthopoulos *et al.* 2006; Molina *et al.* 2011; Bovio and Ascoli 2013; Corona *et al.* 2015; Salis *et al.* 2016). Risk mitigation is strongly linked to landscape fuel management and may involve a range of primary objectives, strategies and spatial patterns depending on fire management and protection objectives, land use laws, social and physical constraints, and budget (Reinhardt *et al.* 2008; Ager *et al.* 2013; Hand *et al.* 2014; Valor *et al.* 2015; Parisien *et al.* 2018; Alcasena *et al.* 2018; Salis *et al.* 2018). Thus, land and forest managers need to systematically prioritize the more important areas for treatments, while taking into consideration a number of constraints in budgets, time, and laws. From this point of view, geospatial risk-based analytical tools provide a systematic mechanism that can guide assessment and prioritization tasks at landscape and regional scale (Ager *et al.* 2016; Alcasena *et al.* 2017). Recently, spatial wildfire risk assessment was based on burn probability modeling which aims in detecting spatial variability in potential wildfire likelihood, size and behavior stemming from historically-derived variations in ignition points, wind and weather patterns, and fuel types and conditions (Finney 2002; Ager *et al*. 2007, 2010; Salis *et al.* 2013). A number of scientific works performed in the Mediterranean area used the minimum travel time (MTT) fire spread algorithm, which can be parallelized and allow to run thousands wildfire simulations in relatively short time and with good results (Salis *et al.* 2013; Alcasena *et al.* 2016). The coupling of MTT fire spread algorithm and erosion models can assist in targeting fuel management practices, particularly in landscapes characterized by spatial heterogeneity in climate, topography, fuels and soil characteristics (Moody *et al.* 2013; Elliot *et al.* 2016; Srivastava *et al.* 2018). Previous works used both wildfire behavior and erosion modeling to quantify post-fire sediment delivery. Miller *et al.* (2011) estimated burn severity and post-fire ground cover with the First Order Fire Effects Model (FOFEM), and then applied the GeoWEPP model for predicting post-fire erosion. Scott *et al.* (2012) combined geospatial analysis, large-fire simulations with Fire SIMulation system (FSim), and burn probability modeling to examine pixel-based measures of wildfire hazard and watershed exposure with the aim of identifying watersheds that are likely to burn at high intensity which can be used to inform mitigation and prioritization efforts in the Beaverhead-Deerlodge National Forest in Montana (USA). Thompson *et al.* (2013) generated spatially resolved estimates of wildfire likelihood and intensity by 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 (USA). They reported substantial variation in the exposure of and likely effects to highly valued watersheds throughout the study area, and suggested that a large amount of risk could be mitigated via hazardous fuel reduction treatments. Sidman *et al.* (2015) modeled 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 modeling erosion from both wildfire and fuel management on treatment areas. Srivastava *et al.* (2018) combined FlamMap and WEPP to identify high-risk erosion hillslopes following wildfire and to evaluate the effects of fuel treatments on hydrological response of a watershed located in Washington (USA).

The aim of this work is to analyze at the landscape scale the combined effects of (i) fuel treatments aimed to reduce the wildfire probability and intensity, and (ii) post-fire treatments aimed to mitigate the erosion, by estimating pre-fire and post-fire erosion risk. We used the MTT fire spread algorithm as implemented in Randig (Ager *et al.* 2010; Salis *et al.* 2013) to simulate hundred thousand wildfire simulations and capture the spatial variability in wildfire behavior and intensity, and we used the MTT outputs to feed the Erosion Risk Management Tool (ERMIT, Robichaud 2007*a*, 2007*b*) to simulate pre- and post-fire sediment delivery in a 68,000 ha Mediterranean fire-prone area located in Northern Sardinia, Italy. The effects of soil burn severity, time since fire, vegetation type and recovery, the sediment delivery exceedance probability and sediment yields were investigated. We then examined the potential of four different fuel management strategies (a control condition plus three diverse fuel treatment strategies located nearby urban areas (WUI), located nearby roads (ROAD), or randomly located (RAND)), which used different spatial approaches to reduce burn probability and fire severity, and thus in turn modified post-fire sediment delivery inside and nearby the treated areas.

3.2 Material and methods

3.2.1 Study area

The study area covers about 68,000 ha of land and is located in Northeastern Sardinia, Italy (Fig. 1).

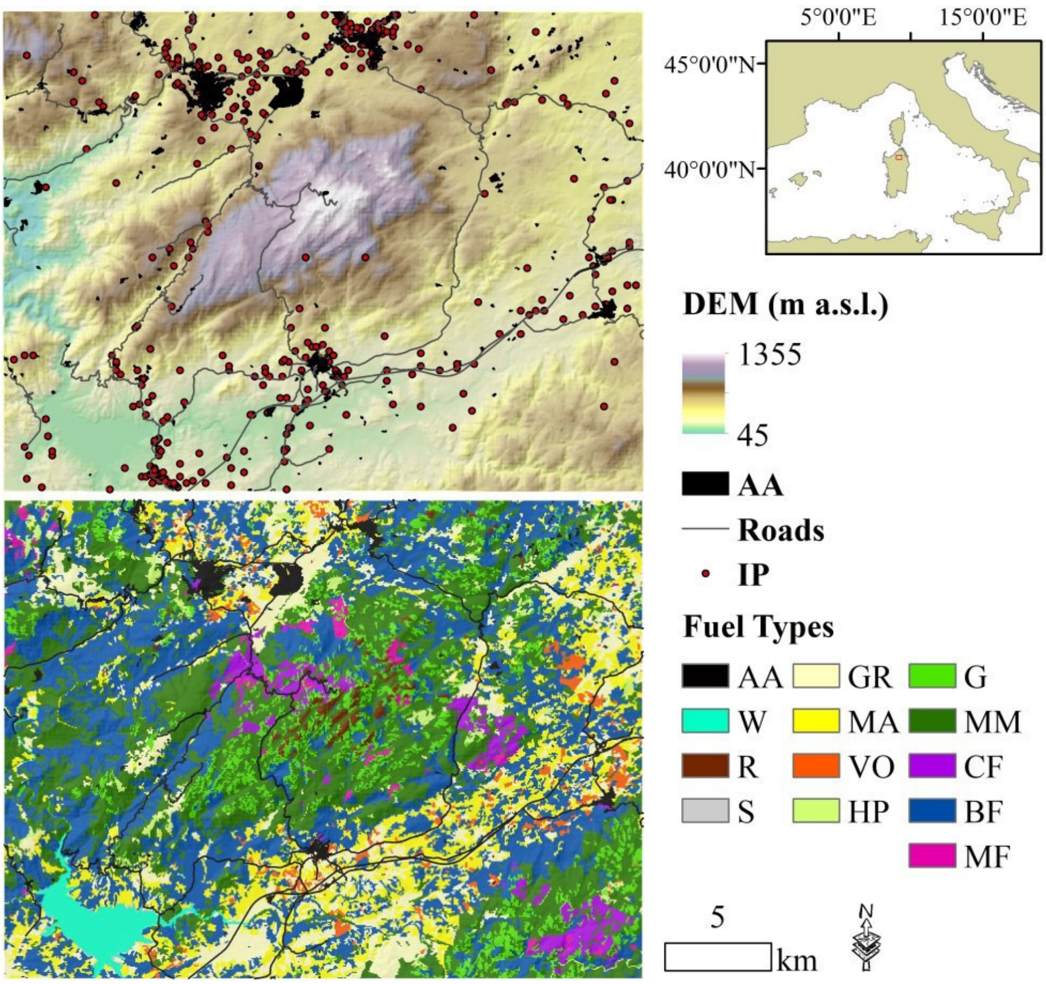

Fig. 1. Map of the study area, located in Northern Sardinia, Italy. The topmost map shows the terrain elevation of the study area, together with roads, anthropic areas (AA), and wildfire ignition points of the period 1980-2010. The bottom map presents the main fuel types of the study area, as derived from the Sardinia Land Use Map (2008). AA = anthropic areas; W = water bodies; R = rocks; 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

The topography of the area is complex: the terrain elevation ranges from about 45 m a.s.l. to about 1,350 m a.s.l., with several hills and low mountains (Fig. 1). The climate is Mediterranean, and is overall characterized by drought conditions from late May until September. The average annual precipitation is greater than 1,000 mm at the highest elevations where summer storms are frequent, and about 650 mm in lower elevation areas. The rainiest months are typically November and December. The mean annual temperature of the study area is about 13°C, with relevant variations between mountain peaks and lowest areas (Chessa and Delitala 1997). The vegetation is largely characterized by the presence of shrublands and forests, which occupy about 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* ssp. laricio Poir, even though their presence is limited. High and dense Mediterranean maquis cover large amount 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 characterized by a higher presence of *Cistus monspeliensis* L., *Pistacia lentiscus* L. and low shrubs (Fig. 1). Anthropic areas cover approximately 1% of the study area and include the town of Tempio Pausania. Fruit-bearing areas, mostly sparse vineyards and olive groves, cover about 2300 ha located in flat areas and nearby urban areas. Grasslands and agricultural areas are mainly herbaceous and horticultural productions and characterize about 20% of the study area, particularly in the plains (Fig. 1).

Recent wildfire history during 1980-2010 indicates the study area experienced about 800 ignitions; wildfires smaller than 10 ha were the 95% of these ignitions, while the remaining wildfires were responsible for 90% of the total area burned. The largest wildfire was in 1983, burned 18,000 ha near the town of Tempio Pausania and caused 9 fatalities in the northern part of the study area. The majority of the ignitions was concentrated in the hottest months of the year (June to September); about 60% of the ignitions happened from mid-July to late August. The most common areas of ignitions are roads and surroundings of anthropic zones (Fig. 1).

3.2.2 Input data for wildfire modeling

To generate the gridded landscape file for Flammap (Finney 2006) we assembled all input data at 25 m resolution. The topographic inputs (elevation, slope and aspect) data were derived from 10-m digital elevation data of the island (www.sardegnageoportale.it/). 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 threshold. Q. suber L. and Q. 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 3 different fuel treatment scenarios carried out nearby Wildland Urban Interface (WUI) or roads (ROAD), or randomly located (RAND) (Fig. 2). WUI and ROAD scenarios were obtained by the application of a spatial treatment optimization software (LTD, landscape treatment designer (Ager *et al.* 2013; Vogler *et al.* 2015)).

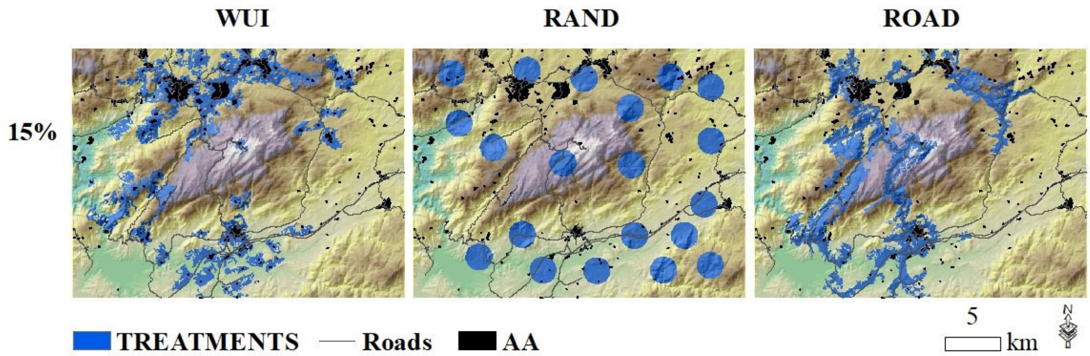

Fig. 2. Spatial location of the fuel treatments tested in this work. WUI = wildland-urban interface protection; RAND = random location; ROAD = road protection. The area treated for each of the above three strategies was 15% of the study area.

Each fuel treatment scenario was performed for a total area of 10,000 ha (15% of the study area) (Fig. 2). The treatments hypothesized modeled common fuel management operations such as pruning of the lowest branches, removal of dead fuels and part of the understory for shrublands, forest understory, and herbaceous pastures (Sardinia Forest Agency, personal communication 2014). Fuel moisture content (FMC) for the 1-h and 10-h time lag dead fuels was estimated using historic moisture data above the 97th percentiles, according to sampling campaigns carried out 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 directions which characterized about 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 correspond to 97th percentile of historic conditions. Finally, we developed 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.

3.2.3 Wildfire simulation modeling

To simulate wildfire spread and behavior in the study area, we used the minimum travel time (MTT) spread algorithm of Finney (2002) as implemented in Randig. The MTT uses the Huygens' principle to simulate fire growth (Richards 1990; Finney 2002) considering both behavior and growth modelled by vector or wave front (Finney 2002; Ager *et al.* 2010) and surface fire spread is predicted by the Rothermel's equation (1972). Crown fire initiation and spread are modeled 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 target fuel treatments and evaluate wildfire exposure and risk (Salis *et al.* 2013, 2016, 2018; Mitsopoulos *et al.* 2015; Alcasena *et al.* 2016, 2017, 2018; Kalabokidis *et al.* 2016; Oliveira *et al.* 2018; 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 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, which are associated with the largest wildfires on the island. 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 given a fire occurs (Scott 2006).

3.2.4 *Input data for erosion modeling*

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 by 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). This tool allows the user to create 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 stochastic climate of the study area for 50 years, which are used to account for temporal 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. For these parameters, we used the climate data of the Tempio Pausania weather station, as reported in Arrigoni (1968). The stochastic weather data generate by ERMiT Rock:Clime are summarized in Table 1

Table 1. Climate data of the weather station of Tempio Pausania, as reported in Arrigoni (1968). Tmax = average maximum temperature; Tmin = average minimum temperature; PP = cumulated precipitation. The average values of the stochastic climate variables provided by ERMiT Rock:Clime tool are reported under parenthesis for each month and climate variable

Month	Tmax - °C	Tmin - °C	PP - mm	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)**

The rock content percentage and texture soil layers for the study area were derived from Carta dei suoli della Sardegna (Aru *et al.* 1990) and used to build the soil input files for WEPP.

To delineate watersheds and create the polygon terrain slope length, steepness and width files 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).

The hillslope horizontal length is composed of the three slope sections (top, middle, and toe) and represents the length of the hillslope being modeled. 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 lower 10%.

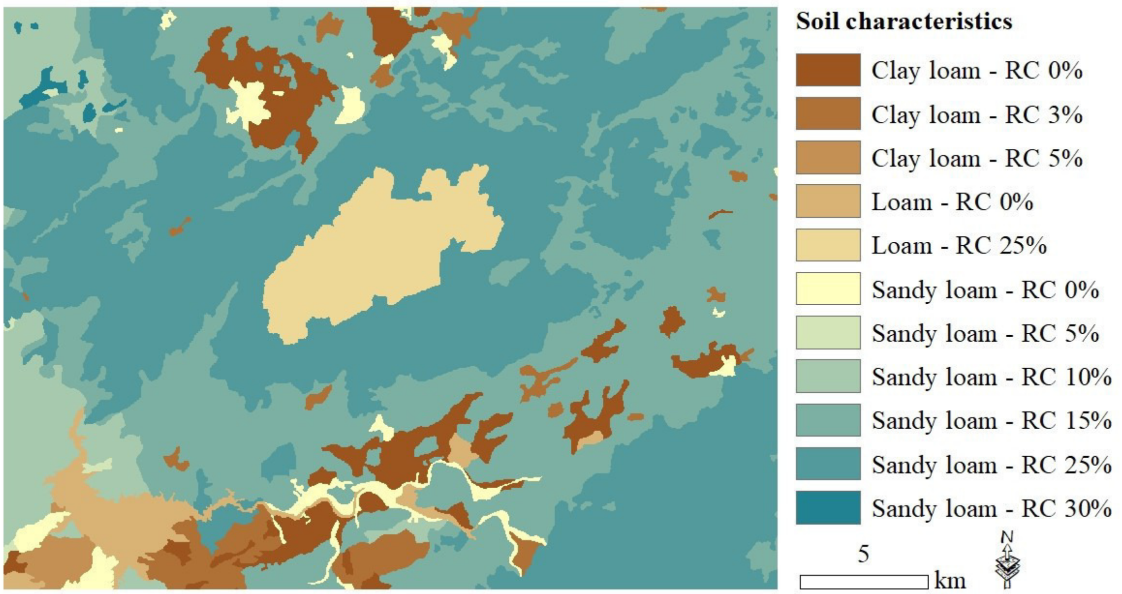

Fig. 3. Map of soil texture and rock content characteristics for the study area, as derived from Carta dei suoli della Sardegna (Aru et al. 1990).

Land-cover data were obtained from the 2008 Sardinia land use map (http://www.sardegnageoportale.it/navigatori/sardegnamappe/), and we then reclassified the land-cover data layer into ERMIT cover types (e.g. forest, chaparral, range).

For modelling post-wildfire conditions, the conditional flame length (CFL) outputs of Randig before and after fuel treatment strategies were used to associate to each pixel a value of potential soil burn severity, which is a description of the impact of a fire on the soil and litter (Robichaud *et al.* 2007*a*). CFL data allowed to discriminate areas characterized by different levels of potential soil burn severity, should a wildfire occur. For this purpose, as proposed by Andrews *et al.* (1982), we identified 4 classes of fire intensity, from unburned to high, which were used as reference for discriminating 4 soil burn severity classes (Fig. 3): CFL = 0 – 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) (Table 2).

Table 2. Flame length values and corresponding soil burn severity classes used in the study

Fire intensity classes	Soil burn severity classes
0<FL<0.1	Unburned
0.1<FL<1.2 m	Low
1.21<FL<2.4 m	Moderate
FL>2.41 m	High

We integrated flame length pixel values from Randig for each hillslope with the severity class breaks previously defined. The areal distribution of unburned, low, moderate, and high potential soil severity considering the actual vegetation in the study area was 1.7%, 64.2%, 24.6% and 9.5%, respectively.

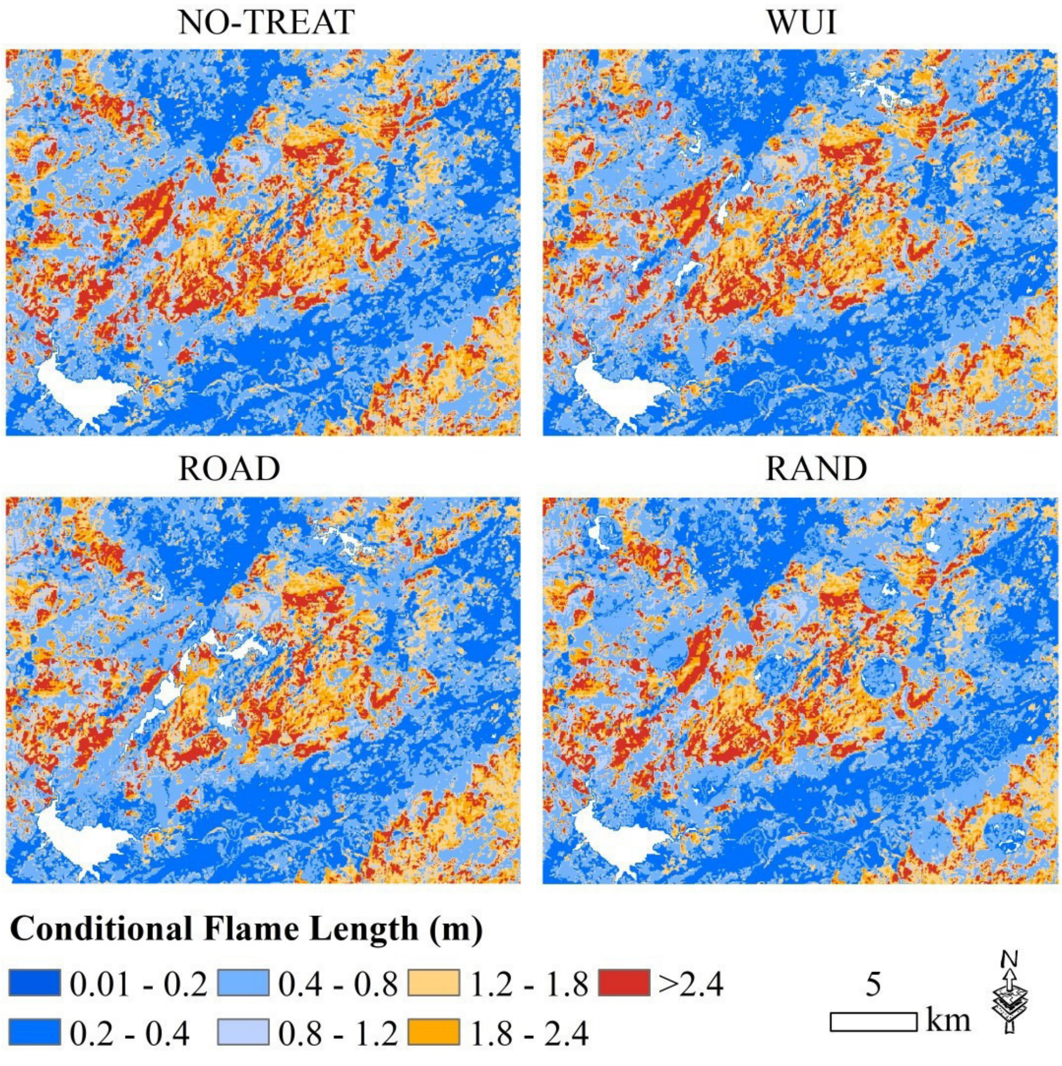

Fig. 4. Map of the conditional flame length, as obtained from fire spread simulations, for the study area considering actual fuel conditions, and WUI, ROAD and RAND fuel treatment strategies applied for 15% of the study area. These maps were used to derive soil burn severity classes.

Even though Keeley (2009) suggests that severity often plays a more limited role on post-wildfire erosion than topography or climate, the use of flame length as indicator of burn severity was previously done by other works (Elliott 2013; Elliott *et al.* 2016; Srivastava *et al.* 2018).

3.2.5 Post-fire erosion modeling

Post fire erosion was simulated by using the Erosion Risk Management Tool (ERMiT) (Robichaud *et al.* 2007*a*), which is a probability-based risk assessment tool for quantifying post-fire disturbance erosion modeling and evaluating rehabilitation treatment effectiveness. ERMiT provides probabilistic estimates of single-storm postfire 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 (Water Erosion Prediction Project) 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, infiltration, runoff, soil detachment, sediment transport, and sediment deposition to predict runoff and erosion at the hillslope scale (Flanagan and Livingston 1995; Elliott *et al.* 2016). As previously reported, ERMiT needs five main input data: a) climate parameters, which are created through Rock:Clime (Elliot 1999, 2004); b) vegetation types (forest, range, chaparral); c) soil types and rock content; d) topography (slope length and gradient); 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 to 100 percent (Robichaud *et al.* 2007*a*, 2007*b*).

In this work, considering the extension of the study area, all simulations were performed using the Batch ERMiT interface spreadsheet (https:\\forest.moscowfsl.wsu.edu\FSWEPP), and the input data b-e were produced in a GIS environment. The ERMiT sediment delivery predictions have an associated probability of occurrence, which is calculated as the product of the occurrence probabilities due to each source of variation (Robichaud *et al*. 2007*a*). Sediment delivery predictions are paired with their respective combined occurrence probability, and sorted in descending order of sediment delivery amounts. The "exceedance probability" for each sediment delivery prediction is computed as the sum of the occurrence probabilities for all greater sediment yield predictions (Robichaud *et al.* 2007*b*).

ERMiT batch produced sediment erosion prediction files for each hillslope of the study area, which were linked to spatial maps in order to produce erosion maps for each of the conditions analyzed.

3.2.6 *Modeling fuel reduction effects on post-fire sediment delivery*

To analyze the benefits of fuel reduction treatments on the study area, sediment erosion was modeled considering the following conditions: 1) actual fuel conditions, in the absence of wildfire disturbances; 2) actual fuel conditions, in the presence of wildfire disturbances; 3) wildfire disturbances after different spatial strategies of fuel management and post-fire erosion reduction treatments.

We then tested the effects of different factors in the post-fire sediment erosion rates for the whole study area. These factors included: a) sediment delivery exceedance probabilities (from 1 to 95); b) two different post-fire treatment strategies to reduce erosion (untreated and seeding); 3) the years (from 1 to 5) after the wildfire events; 4) the three land cover types (range, chaparral and forest); 5) three slope classes (below 10°, from 10 to 20°, above 20°); 6) the four soil burn severity categories (unburned, low, moderate, high).

3.3 Results and Discussion

3.3.1 Post fire erosion for actual vegetation conditions

Considering actual fuel conditions in the absence of wildfire disturbances and 50% exceedance probability, the simulated average sediment delivery in the study area was about 0.01 t ha^{-1}, and varied from 0 to a maximum of 0.17 t ha^{-1} across all hillslopes (Table 3). In these conditions, the total area which showed erosion was about 11,900 ha over 68,000 ha. Previous studies on soil erosion carried out in Sardinia confirmed the low values of mean sediment delivery in the absence of wildfire disturbances: Acutis *et al.* (1996) measured mean erosion rates close to 0.02 t ha^{-1} yr^{-1} in north-western Sardinia, while Rivoira *et al.* (1989) reported mean soil losses of about 0.03 t ha^{-1} yr^{-1} in northern Sardinia. An overview of post-wildfire soil erosion characteristics under natural or simulated rainfall conditions in the Mediterranean basin, as reported by previous studies is provided in the supplementary data.

The areas that presented the highest erosion rates in the absence of wildfire disturbances were those characterized by the steepest and longest terrain slopes (Fig. 1). 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): they evaluated soil loss data for diverse crops growing in hilly areas, and during two experimental campaigns observed mean soil losses of 2.55 and 0.86 t ha^{-1} yr^{-1}. Canu *et al.* (2015) measured post-fire sediment delivery in cork oak areas of NW Sardinia in the range 0.05-0.86 t ha^{-1}, with average values below 0.1 t ha^{-1}: the measurements were carried out at the third year after the fire. Overall, the values obtained in our study for the unburned conditions are also not dissimilar to those reported by Cerdan *et al.* (2010): they reported that mean soil erosion in Mediterranean Europe amounted to about 1.2 t ha^{-1} yr^{-1} for the whole CORINE area.

The average sediment delivery, both in terms of average and maximum sediment delivery, and the total area with potential erosion issues were largely dependent from the exceedance probability (Table 3). For instance, considering actual fuel conditions and absence of wildfire disturbances in the study area, the variation from 50% to 20% exceedance probability resulted in an increase of about 500% of the average sediment yields, and of about 900% of the maximum sediment delivery.

Table 3. Effects of the sediment delivery exceedance probability on average and maximum sediment delivery, and on the total area with sediment delivery, considering actual fuel conditions, in the absence of wildfire disturbances.

Exceedance probability	Average sediment delivery ($t\ ha^{-1}$)	Max sediment delivery ($t\ ha^{-1}$)	Total area with sediment delivery (ha)
1%	2.789	51.07	41,327
3%	1.593	34.33	37,980
5%	0.787	23.92	37,595
10%	0.155	9.73	34,624
20%	0.024	1.50	27,842
30%	0.013	0.91	22,726
40%	0.006	0.67	13,864
50%	0.005	0.17	11,904
60%	0.004	0.06	10,589
70%	0.002	0.04	9,522
80%	0.002	0.04	9,511
90%	0.001	0.03	4,765
95%	0.000	0.02	1,286

The occurrence of fire has significant effects on the increase of the sediment delivery coefficient compared to unburnt conditions, even after moderate fires (Gimeno-García *et al.* 2000; Keeley 2009; Stoof *et al.* 2015; Vieira *et al.* 2015). In our work, we found that, in the post-wildfire simulations with actual fuel conditions, 80% of sediment delivery was generated by only 17.6% of the hillslopes of the study area, when considering an exceedance probability of 80% (Fig. 5). The reduction in the exceedance probability promoted the increase of the hillslope areas that contributed to about 80% of the sediment yields (22.5% and 24.8% for an exceedance probability of 50% and 20%, respectively).

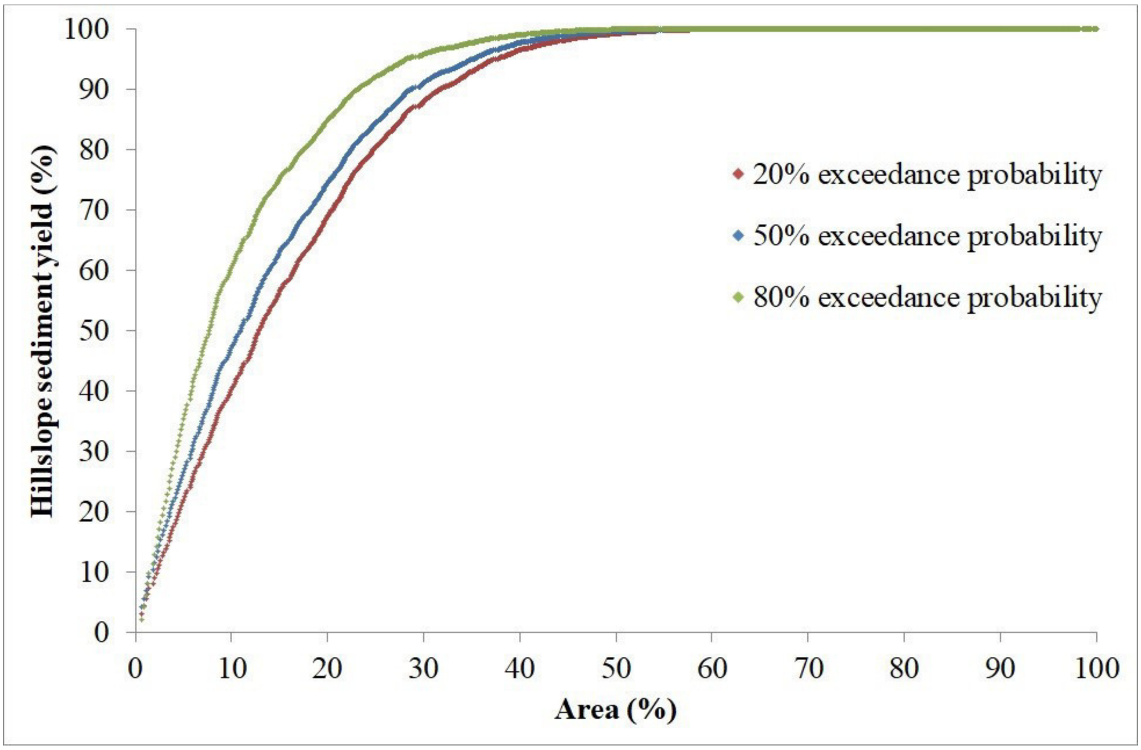

Fig. 5. Effects of sediment delivery exceedance probability (20%, 50%, and 80%) on cumulative hillslope area and simulated cumulated average sediment yield with actual vegetation and the first year after wildfires.

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; Badia and Martì 2008). In fact, infrequent but intense rainstorms can cause high runoffs and soil losses within short periods, as proved by several studies (Moody and Martin 2001; Cannon *et al.* 2011; Hosseini *et al.* 2016).

The application of ERMiT allowed to highlight how the distribution of runoff event rates can affect the sediment yields exceedance probabilities in the post-fire conditions. In our study area, the average sediment delivery was strongly influenced by the exceedance probability in terms of both spatial variation and absolute sediment delivery, as showed in Figure 6. The highest values of sediment delivery were observed in the steepest areas with the lowest exceedance probabilities: for instance, at 20% exceedance probability, only about 10% of the landscape exhibited sediment yields greater than 24 t ha^{-1} in the first year after the wildfires (Figure 6). The increase in the exceedance probability promoted the reduction of the extent of the areas characterized by sediment yields. The fact that low rain intensity rates after the fires can pose limited problems of soil erosion was confirmed by previous studies (Moody *et al.* 2013; Haas *et al.* 2017).

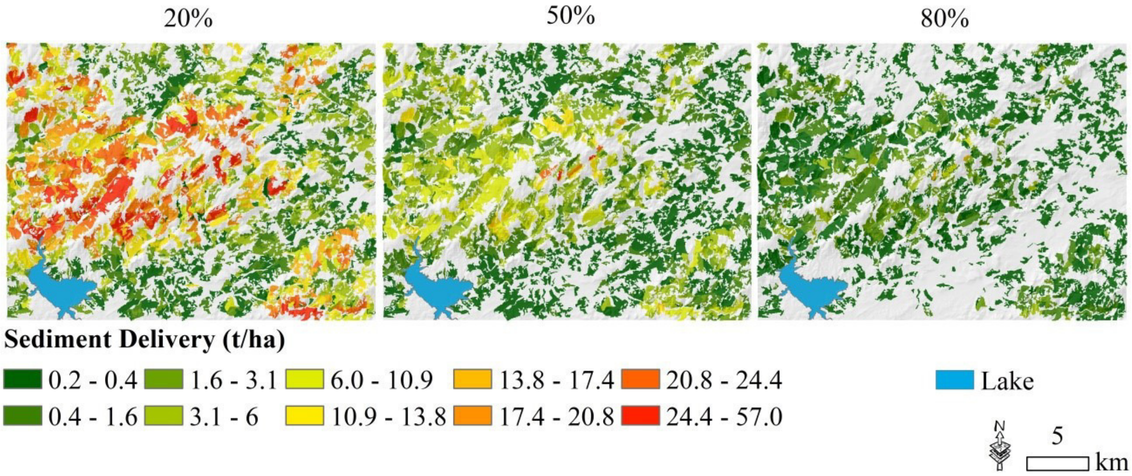

Fig. 6. Maps of the sediment delivery at the landscape scale, considering the first year after the fires with actual fuel conditions, and an exceedance probability of 20% (left), 50% (middle), and 80% (right).

The fire effects were particularly relevant in the first years after the fires (Fig. 7). The fact that the highest impacts in terms of post-fire erosion are generally observed the first year after the fires was confirmed by a number of works (Shakesby *et al.* 2011; Hosseini *et al.* 2016). In our study, for instance, focusing at the 50% exceedance probability, the average sediment delivery in the study area ranged from 1.7 t ha^{-1} of the first year after the fires to 0.04 t ha^{-1} at the fifth year after the events. Limited spots located in the steepest slopes showed peaks above 25 t ha^{-1}, with 50% exceedance probability. When taking into consideration 20% and 80% exceedance probability, the average sediment delivery was respectively 6.4 t ha^{-1} and 0.3 t ha^{-1} on the first year after the fires, and 0.6 t ha^{-1} and about 0 t ha^{-1} on the fifth year after the fires. These values are in line with those reported in previous works that focused on the Mediterranean basin. For instance, Shakesby *et al.* (2011) reported mean post-wildfire erosion rates (measured on field plots) one year after the fire equal to 0.39 t ha^{-1} for low, 3.28 t ha^{-1} for moderate, and 10.80 t ha^{-1} for high severity fires. Pausas *et al.* (2008) indicated that post-fire erosion rates measured in the Mediterranean Basin are rarely higher than 10 t ha^{-1} and are often lower than 1 t ha^{-1} on the first year after the fire. Other studies evidenced relatively low erosion rates in Mediterranean 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 was confirmed by Cerdan *et al.* (2010), which compared erosion plot data in Europe and evidenced the more limited rates observed in the Mediterranean compared with other European areas: they attributed this difference to the stoniness and the thinness of the Mediterranean soils (Shakesby *et al.* 2011). On the other hand, sediment delivery rates above 10 t ha^{-1} the first year after the fire were showed by Soto and Diaz-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 areas, particularly in the U.S (Robichaud *et al.* 2013; Elliott *et al.* 2016). For instance, post-fire erosion rates from the Cannon Fire, in California (USA), ranged from 2.5-15 t ha^{-1} (Robichaud *et al.* 2008), while erosion rates measured following wildfires in the Sierra Nevada Mountains were 46 t ha^{-1} in the Cedar Fire (Robichaud *et al.* 2013).

We obtained that the effects of the fires on erosion processes tended to becoming insignificant 4-5 years after the fires (Fig. 7): at that time, the differences between burned and unburned landscapes was basically irrelevant. Indeed, soil erosion yields decline through time mostly due to the regeneration of the burned vegetation, which progressively tends to return to values typical of pre-burning conditions, typically within 5 years after the fire disturbances (Fox *et al.* 2006; Robichaud *et al.* 2007*a*; Shakesby *et al.* 2011). Nonetheless, a number of works highlighted that the responses of the areas burned last less than 7 years, and depend not only on vegetation recovery, but also on post-fire weather, sediment availability, morphology, and fire 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).

Fig. 7. Exceedance probability vs. average sediment delivery at the landscape scale, for five years after the wildfires, considering the actual fuel conditions

The role played by soil burn severity (SBS) classes in determining post-fire erosion at the landscape scale in the first two years after the wildfires is characterized by highest sediment yields corresponding to high soil burn severity (Figure 8*a*). Sediment yields were on average 3.1 t ha^{-1} in the first year after the fires, and 1.3 t ha^{-1} in the second year after the fires using the 50% sediment delivery exceeding probability. In the hillslopes with lower SBS values, the above values were on average 1.5 and 0.4 t ha^{-1} for moderate SBS, and 0.08 and 0.04 t ha^{-1} for low SBS, respectively. Therefore, the magnitude of sediment delivery from high severity burn hillslopes was about 2 times greater than from moderate and 38 times greater than from low severity burn hillslopes. Previous works confirmed our results: in Galicia, Spain, Soto and Diaz-Ferros (1998) reported sediment delivery rates one year after the fire of 12.4 and 4.9 t ha^{-1} on high-severity and low-severity plots respectively, whereas the erosion measured in the control plot was around 2.0 t ha^{-1}. Gimeno-García *et al.* (2000), using experimental fires in Mediterranean shrublands, observed that 1-year erosion rates are low (< 0.1 t ha^{-1} yr^{-1}) in unburned conditions, while soil losses become significant after a fire, and increased with fire severity (2.3 and 2.9 t ha^{-1} yr^{-1} in moderate- and high-severity fires). Vega *et al.* (2005) analyzed the first-year effect of two different prescribed burning treatments in shrublands of Galicia, Spain: the most intense burning caused significantly greater soil erosion (5-8 times) compared with the unburned areas.

As expected, 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 the fires and exceedance probabilities (Figure 8*b*). Focusing on 50% exceedance probability, the average sediment delivery rate for the first year after the fires at the landscape scale decreased from 4.0 t ha^{-1} for steep slopes to 2.2 t ha^{-1} for moderate slopes to 0.6 t ha^{-1} for low slopes. The relevant role played by steepness in sediment delivery rates was also highlighted by previous works (Pelletier and Orem 2014; DeLong *et al.* 2018). Plus, 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, due to the quicker vegetation recovery and the higher presence of organic matter in north-facing slopes than in southern ones.

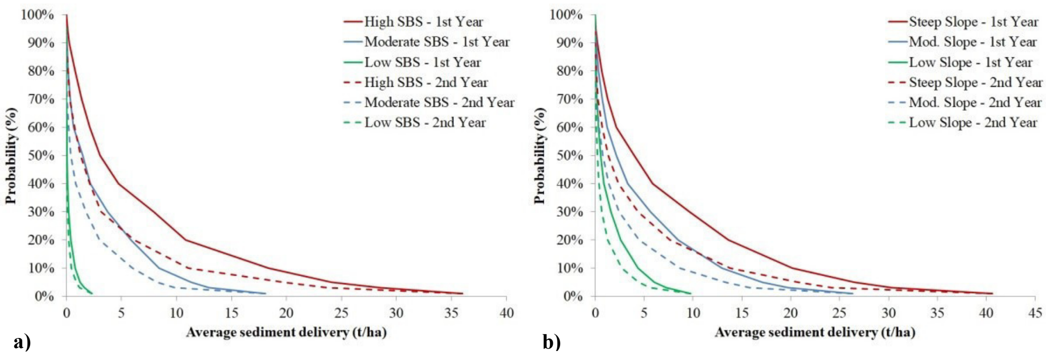

Fig. 8. a) Exceedance probability vs. average sediment delivery at the landscape scale, for the first two years after the fires and the three soil burn severity (SBS) classes, considering the actual fuel conditions; b) Exceedance probability vs. average sediment delivery at the landscape scale, for the first two years after the fires and three terrain slope classes (low, < 10°; moderate, 10-20°, steep, >20°), considering the actual fuel conditions

Regarding the effects of vegetation types, we observed that the highest sediment delivery was always located in areas covered by chaparral, which showed an average sediment delivery of 2.5 t ha^{-1} in the first year after the fires, and of 0.05 t ha^{-1} in the fifth year after the events, for an exceedance probability of 50% (Fig. 9). This can be partially explained by the fact that chaparral mostly covers areas characterized by steep terrain, and is very limited in flat areas and plains. On the contrary, herbaceous fuels presented the lowest average sediment delivery rates at the landscape scale: post fire erosion ranged from a maximum of 0.5 t ha^{-1} immediately after the fires to a minimum of 0.04 t ha^{-1} (Figure 9). Forest vegetation types showed sediment delivery values not far from chaparral in the first year after the fires, then post-fire erosion was more limited, particularly after the third year after the fires: in fact, at that time-step the average sediment delivery for forests was 0.04 t ha^{-1}, the lowest among fuel types, reaching 0.02 t ha^{-1} in the fifth year after the fire events. On the whole, our results are in line with the data obtained by Vacca *et al.* (2000) in some burned sites located in southern Sardinia: the mean yearly soil loss on burned herbaceous pastures was 0.06 t ha^{-1}, while soil losses on slopes covered with shrubs and Eucalyptus spp. were higher and corresponded to 0.11 t ha^{-1} and 0.23 t ha^{-1}. However, the high post-fire sediment delivery rates of shrublands and forests, particularly in mountains and hilly areas, are 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).

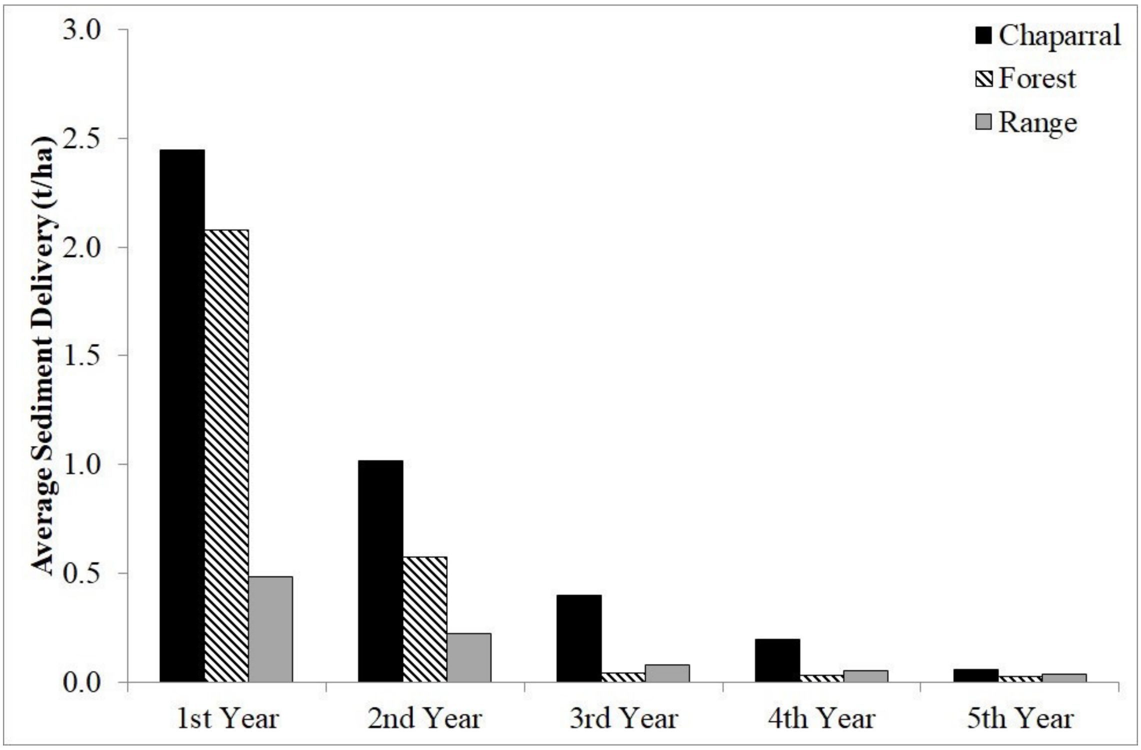

Fig. 9. Histograms of the average sediment delivery for chaparral, forest and range at the landscape scale, considering the first five years after the fires with actual fuel conditions, and a 50% exceedance probability.

Finally, we tested the effects of seeding post-fire treatments on sediment yields for the study area. As expected, we found that this post-fire treatment was able to reduce erosion, particularly in the second year after the wildfires, where we observed a maximum difference between seeding and untreated scenarios close to 22 t ha-1, with an exceedance probability of 20%. After the third year post-fire, the differences between seeding and no treatments were progressively less relevant. As expected, the variation in sediment delivery induced by post-fire treatments was higher when the probability exceedance was lower (Fig. 10). Previous papers agreed that any notable relationship between establishment of vegetative cover and reduction of erosion within the first year after fire can be found (Robichaud et al. 2000; Beschta et al. 2004; Beyers 2004; Peppin et al. 2010; Rulli et al. 2012): in fact, the most relevant sediment movements frequently occurs before plant cover is established (Robichaud et al. 2000). The sediment yield reduction was overall confirmed to disappear by the third and subsequent years after fire (Peppin et al. 2010) However, seeding was proved to be very effective in some cases and areas, but not in others (Prats et al. 2014)

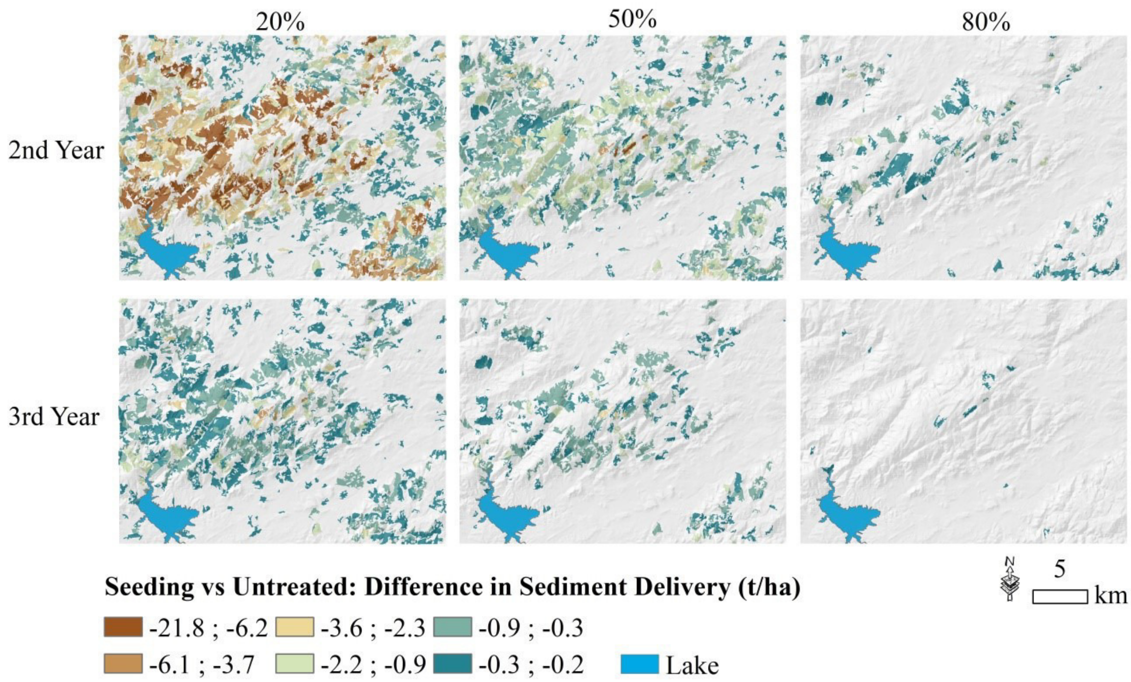

Fig. 10. Maps of the difference in sediment yields for the study area considering post-fire seeding treatments vs. no treatments, focusing on the second and third years after the wildfire events, and exceedance probabilities of 20%, 50% and 80%.

3.3.2 Post fire erosion for the different fuel management scenarios and erosion treatments

Even if the three strategies of fuel treatments tested in the study areas had the main goal of reducing burn probability, and were applied to a limited portion (15%) of the study area, we also observed positive effects on post-fire sediment delivery at both landscape scale and inside the treated areas with respect to actual fuel conditions. Looking at a 50% exceedance probability and the first year after wildfires, the average sediment delivery at the landscape scale dropped from 1.7 t ha^{-1} of the actual fuel conditions to 1.5 t ha^{-1} of the WUI strategy and 1.5 t ha^{-1} of the ROAD strategy, which was the most effective in reducing post-fire sediment yields (Figure 11). Conversely, the RAND strategy was less effective than the previous two in reducing average sediment delivery at the landscape scale the first year after the wildfires (-0.7 t ha^{-1} with respect to actual fuel conditions). The second year after the wildfires, the average sediment delivery at the landscape scale was much lower than the previous year, and the differences between fuel treatment strategies and actual vegetation were less; the best performance against post-fire erosion was obtained by the ROAD strategy (0.5 t ha^{-1} vs. 0.6 t ha^{-1} in the actual fuel conditions, 50% exceedance probability). At the third year after the wildfires, and in the following years, the differences among fuel treatment strategies and actual fuel conditions were small.

The fact that the differences in post-fire soil erosion induced by fuel treatments could be relatively small was also reported by previous studies, the most of which carried out in the U.S. (Robichaud *et al.* 2010). Moreover, fuel treatments efforts to minimize wildfire severity can oftentimes conflict with those meant to reduce the potential for erosion (Shakesby *et al.* 1993; Harrison *et al.* 2016). Indeed, the presence of woody fuels, litter, or a continuous cover of surface fuels limit erosion by protecting the soil, reducing sediment yields, increasing infiltration rates (Robichaud 2000). However, continuous and dense surface fuels also increase the potential wildfire spread and intensity, when wildfire ultimately occurs (Silins *et al.* 2009; Harrison *et al.* 2016).

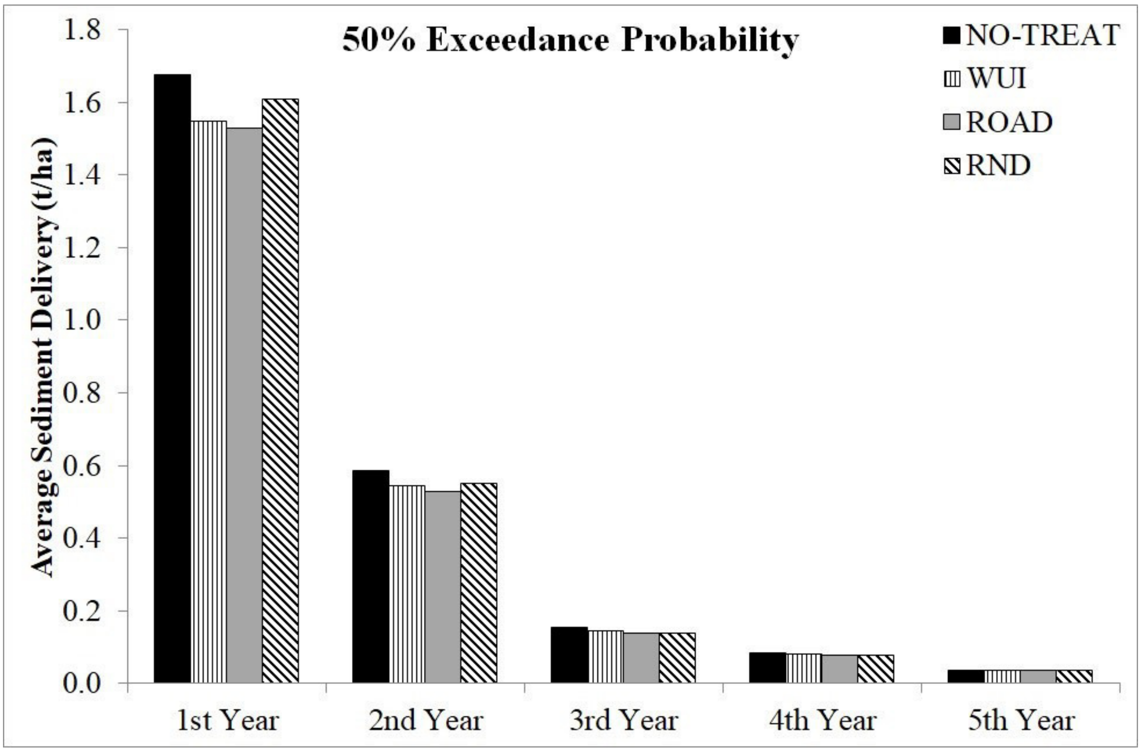

Fig. 11. Average sediment delivery at the landscape scale after the wildfires, from the first to the fifth year after the events, considering three different spatial fuel treatment strategies (WUI, RAND, and ROAD) applied for 15% of the study area, and actual fuel conditions (NO-TREAT). The results refer to 50% sediment delivery exceedance probability

The spatial impact of fuel treatment strategies in reducing post-fire sediment yields with respect to NO-TREAT conditions (considering 50% exceedance probability) shows that the location of fuel treatments was able to lower wildfire intensity for the study area for all years decreasing since the year of the fire (Figure 12). The role played by the location of fuel treatments on post-fire erosion by reducing fire severity was also highlighted by Elliot *et al.* (2016) and Srivastava *et al.* (2018) using fire spread modeling approach.

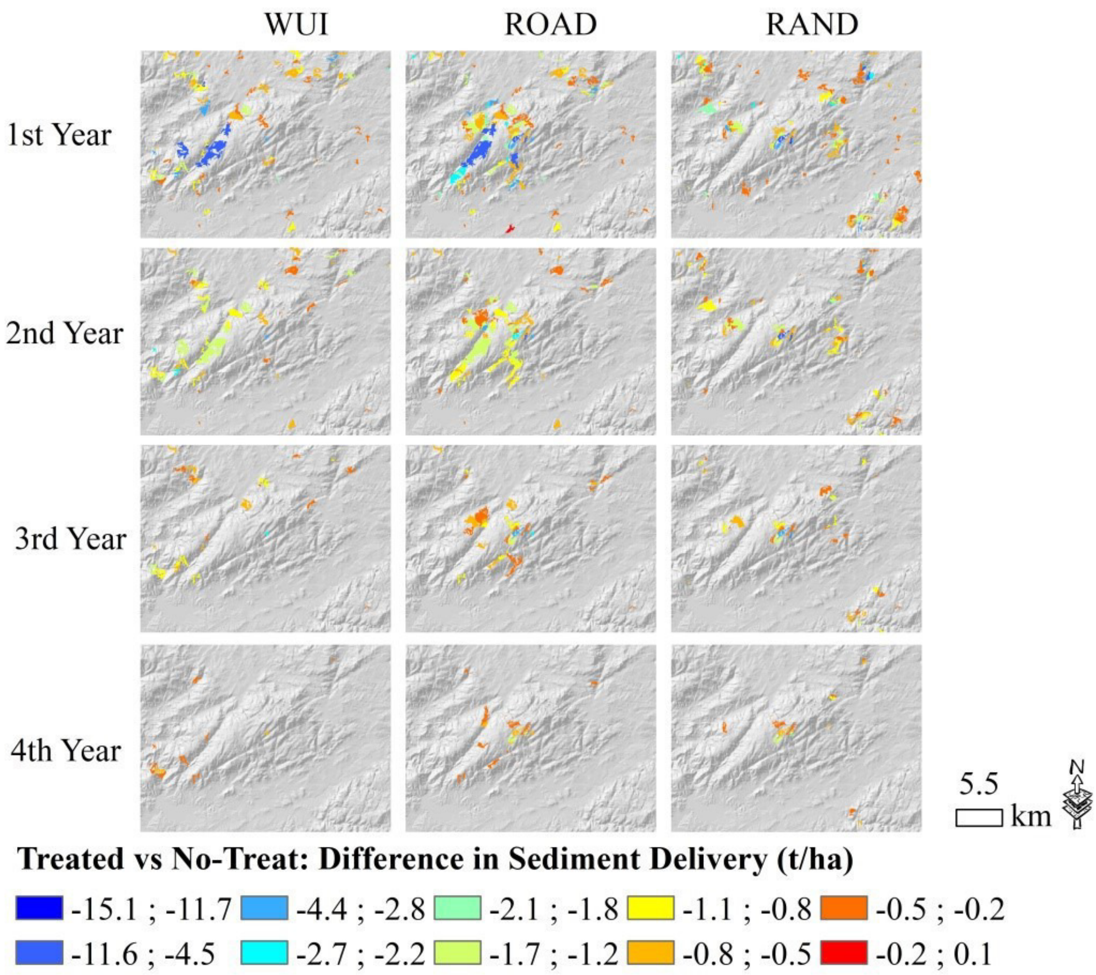

Fig. 12. Differences in sediment delivery between actual fuel conditions (NO-TREAT) and WUI (left), ROAD (middle) and RAND (right) strategies, considering the first four years after the wildfires, and a reference exceedance probability of 50%.

Focusing at the first year after the wildfires, the simulations confirmed the relevant 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 diverse exceedance probabilities (Figure 13). For instance, moving from 20% to 80% exceedance probability resulted in a decrease of the average sediment yield from 6.0 t ha^{-1} to 0.3 t ha^{-1} for ROAD fuel treatment strategy. The increase of the exceedance probability emphasized the differences among fuel treatment strategies and actual 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 t ha^{-1} (6.0 vs. 6.4 t ha^{-1}) when considering 20%

exceedance probability, while it decreased to 0.03 t ha^{-1} (0.25 vs. 0.28 t ha^{-1}) with 80% exceedance probability (Figure 13).

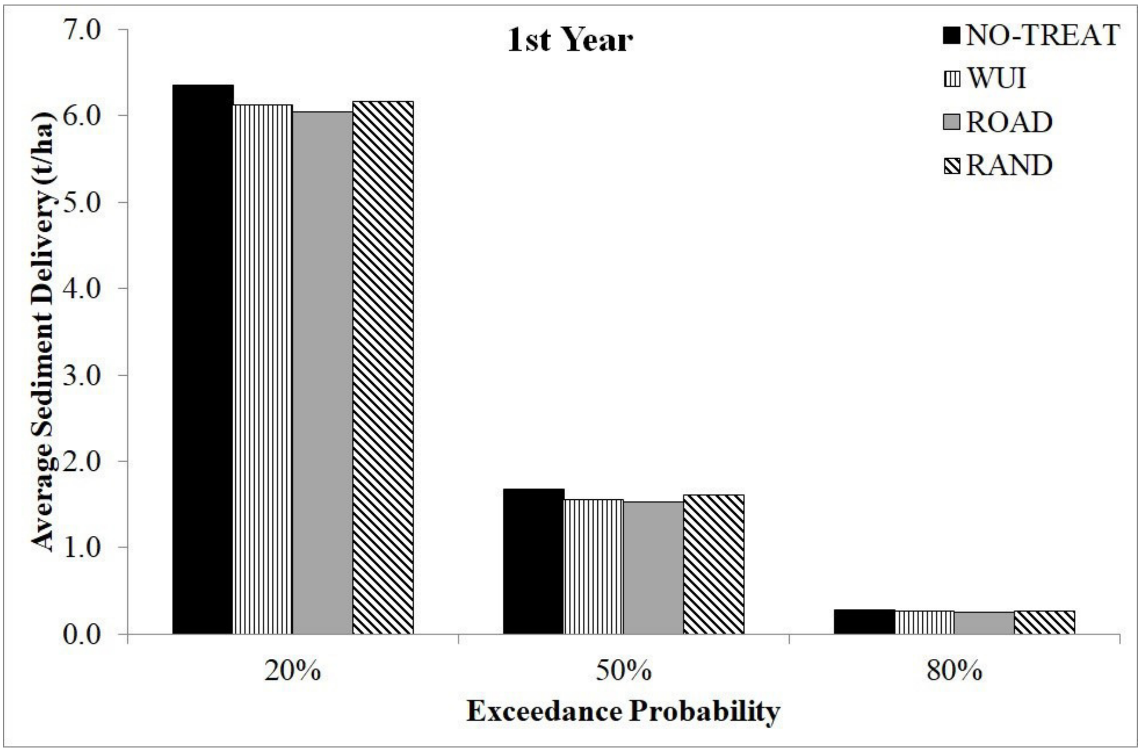

Fig. 13. Average sediment delivery at the landscape scale focusing on the first year after the wildfires with three sediment delivery exceedance probabilities (20%, 50%, and 80%), and three different spatial fuel treatment strategies (WUI, RAND, and ROAD), applied for 15% of the study area, plus actual fuel conditions (NO-TREAT).

Overall and particularly in the first years after wildfires, the reduction in sediment yields promoted by fuel treatments is quite important, mainly when taking into consideration WUI and ROAD treated areas (Table 4). In fact, the first year after the wildfire events, the average sediment delivery in WUI areas dropped from 1.6 to 1.1 t ha^{-1}, and even in ROAD areas from 2.1 to 1.3 t ha^{-1}. Only five years after the wildfires, the differences in sediment delivery between fuel treatment strategies and no-treatment condition is irrelevant. The variation in sediment yields as affected by post-fire seeding was evident for the second, third and fourth year after the wildfires (Table 4). Looking at the second year post-fire, the sediment delivery was more than halved by the application of seeding with respect to the untreated conditions, and this was observed for all fuel management strategies.

Table 4. Average sediment yields inside the areas treated by the three fuel management strategies (WUI, ROAD, and RAND) and when considering actual fuel conditions (NT) in the respective treated areas. A 50% sediment delivery exceedance probability was set. Sediment yields vary depending on the year after wildfire events and on the post-fire erosion strategy (Untreated vs. Seeding).

Post-fire Timeframe	**Fuel Treatment Strategy**	**Sed. Yields (t ha^{-1})**	**Sed. Yields (NT) (t ha^{-1})**	**Sed. Yields (SEED) (t ha^{-1})**	**Sed. Yields (NT-SEED) (t ha^{-1})**
		Post-fire erosion strategy: Untreated		*Post-fire erosion strategy: Seeding*	
1ST YEAR					
	WUI	1.108	1.570	1.108	1.570
	ROAD	1.296	2.075	1.296	2.075
	RAND	1.002	1.261	1.002	1.261
2ND YEAR					
	WUI	0.367	0.544	0.159	0.260
	ROAD	0.448	0.750	0.191	0.360
	RAND	0.352	0.506	0.161	0.244
3RD YEAR					
	WUI	0.080	0.133	0.058	0.085
	ROAD	0.114	0.202	0.076	0.123
	RAND	0.094	0.164	0.061	0.010
4TH YEAR					
	WUI	0.057	0.075	0.032	0.057
	ROAD	0.075	0.110	0.044	0.079
	RAND	0.061	0.087	0.038	0.064
5TH YEAR					
	WUI	0.030	0.030	0.030	0.030
	ROAD	0.044	0.044	0.044	0.044
	RAND	0.036	0.036	0.034	0.034

3.4 Conclusions

Wildfires cause an increase of soil erosion because modify chemical and physical soil characteristics, reduce vegetation cover, and promote soil water repellency (DeBano *et al.* 1998, Badia and Martì 2000; Malkinson *et al.* 2011; Hosseini *et al.* 2016). The variability in post-fire sediment delivery rates and the uncertainties when predicting future wildfire effects or fuel and environmental scenarios pose relevant challenges in post-fire erosion modeling (Scott *et al.* 2012; Elliot *et al.* 2016). In this work, we showed how fire spread and behavior models can support the identification of areas with diverse 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. The post fire erosion analysis was based on stochastic simulations and allowed to proactively estimate and map a range of possible pre- and post-fire soil sediment delivery events. Given the strong variability in fire location, size and intensity and the complex interactions between landscape and wildfires, the proposed approach pERMiTs to obtain spatial information on the areas characterized by high severity and burn probability, which can suffer the most relevant impacts in terms of soil erosion after a fire event. Furthermore, the stochastic approach proposed offers a range of fire and soil erosion hazard metrics which are intuitive and easy-to-use and allows to compare multiple wildfire and sediment delivery scenarios across large study areas, and under variable rainfall intensity rates (Haas *et al.* 2017). Findings from this study have significant implications for risk-based strategic management of fuels and land in Mediterranean climate areas, and can help targeting more efficient fuel reduction treatments in the watershed more exposed to severe wildfire events and to erosion processes. Moreover, considering the limitations in budgets, time and specialized teams, the identification of the watersheds that have the highest combined hazard can guide the identification of priority areas where mitigation efforts can produce the most effective and convenient effects to lessen post-fire debris flows. By limiting the potential negative effects of post-fire debris flows before a fire happens, policy makers, forest managers and local communities can more efficiently face the threats posed by fires and subsequent post-fire sediment delivery yields, and thus mitigate the risks related to these hazards, particularly in the light of future climate changes and the predicted increase in the occurrence of extreme weather events (Thompson *et al.* 2013). Future work will focus on the evaluation of the tradeoffs between fire severity reduction and erosion control, that is how much does the fuel treatments cost in terms of erosion relative to the benefits in terms of avoided future erosion.

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Supplementary Data

Table 1. Post-wildfire soil erosion characteristics under natural or simulated rainfall conditions in the Mediterranean basin, as reported by previous studies. Modified by Shakesby 2011.

Country	Location	Vegetation	Rainfall (mm)	Slope	Soil texture/soil type	Fire severity	Post-fire measurement period	Erosion (t ha−1)	Source
Spain	Sa Murtera, Pariatge County, Andratx, Balearic Islands	forest and scrubland	531.7	6		Moderate and high	Yr 1	0.02	García-Comendador et al. (2017)
Spain	Sa Font de la Vila, Pariatge County, Andratx, Balearic Islands	forest and scrubland	531.7	6		Moderate and high	Yr 1	0.06	
NW Spain	Nr. Santiago de Compostela, Galicia	Various (pine, eucalyptus, heather and grasses)	1400	3–29	Loam/sandy loam	N.A.	Yr 1	56 (average) 15–170 range	Díaz-Fierros et al. (1982, 1987); Benito, E. et al. (1991)
NW Spain	Monte Pedroso, Santiago de Compostela, Galicia	Ulex europaeus Ulex europaeus	1474	17	Sandy loam	High and low	Yr 1	12.4 4.9	Soto and Díaz-Fierros (1998)
NW Spain	Verín, SE Orense, Galicia	Pinus pinaster	680 (800)	15–22	Loamy sand	Low	Yr 1	0.03	Fernández et al. (2007)
NW Spain	Ponte-Caldelas, Galicia	Shrubland	1600/3000	37 (27–43)	Granite/sandy-loam	High	Yr 1	42.9	Fernandez et al. (2016)
							Yr 2	11	
NW Spain	Oia-O Rosal, Galicia	Pine stand/shrubland	1572/3000	48 (42–53)	Granite/sandy-loam	High	Yr 1	31.6	
							Yr 2	5.9	
NW Spain	Carnota, Galicia	Pine stand	946/2000	50 (48–52)	Granite/sandy-loam	High	Yr 1	7.4	
							Yr 2	-	
NW Spain	Negreira, Galicia	Shrubland	1950/2500	46 (41–49)	Granite/loamy-sand	High	Yr 1	40.9	
							Yr 2	9.2	
NW Spain	Ribasieira, Galicia	Shrubland	1443/2500	45 (43–48)	Granite/sandy-loam	High	Yr 1	11.3	
							Yr 2	-	
NW Spain	Monte Coirego, Cotobade Mountains, Pontevedra, Galicia	Ulex europaeus	1800	17	Sandy loam	High low	Yr 1	0.60 H, 0.25 L	Vega et al. (2005)
NW Spain	A Estrada, Pontevedra, Galicia	shrubland	1810	25	Sandy loam	Moderate and high	Yr 1	3.6	Vega et al. (2014)

Table 1. Cont.

E Spain	Benidorm, Valencia, Valencian Community	P. halepensis and shrubs	405 (293)	14–20	Xeric Torriorthent	High	18 months	0.2–2.9	Bautista et al. (1996)
E Spain	Nr Guadalest Reservoir, Alicante, Valencian Community	P. halepensis and shrubs	475		Sandy loam	N.A.	Yr 1	0.17	Llovet et al. (2009)
E Spain	Xortà Mt range, Alicante, Valencian Community	P. halepensis and shrubs	658		Silty clay loam	High	Yr 1	0.1	Mayor et al. (2007)
E Spain	Sierra Calderona, Valencia, Valencian Community	P. halepensis, Quercus suber and shrubs	498 (S)	19	Loam	Low	8 months	0.07 (S)	Rubio et al. (1997)
E Spain	Sierra Calderona, Valencia, Valencian Community	P. halepensis, Quercus suber and shrubs	422 (NE)	19	Loam	High	9 months	4.34 (NE)	
E Spain	Sierra Calderona, Valencia, Valencian Community	P. halepensis and shrubs		17–19	Sandy loam	High	Yrs 1–3	0.24 (NE) 0.48 (S)	Andreu et al. (2001)
E Spain	Central Ebro valley, Cantabria	P. halepensis and shrubs	446	18-22	Gypsiferous/ calcareous soils	Moderate	20 months	2.6–3.5 (gypsiferous)/ 1.0–2.0 (calcareous)	Badía and Martí (2000)
E Spain	Montserrat area, Catalonia	P. halepensis and shrubs	675	18-30		High	18 months	3.5 (N)- 21.8 (S)	Marquès and Mora (1992)
E Spain	Cadiretes Mts, Girona, Catalonia	P. pinaster and Q. suber	675	5-6	Sandy loam	Low (M), Moderate (M), High (H)	Yr 1	0.2 (L), 8.5 (M), 40.7 (H)	Úbeda and Sala (1998)
E Spain	Cap de Creus Peninsula, Girona, Catalonia	Cistus monspeliensis and Erica arborea	450	16-17	Sandy loam	Moderate	6 months	0.016–0.021	Pardini et al. (2004)
E Spain	Onil, Valencia, Valencian Community	Mediterranean gorse shrubland Ulex parviflorus Cistus albidus and Rosmarinus officinalis	273	26	Sand/clay/silt	High	2 months	0.3-8.42	De Luis et al. (2003)
E Spain	Lliria, Valencia, Valencian Community	sclerophyllous shrub	400		Sandy loam	Moderate and high	Yr 1	2.9 H 2.3 M	Gimeno-García *et al.* 2000

Table 1. Cont.

Portugal	Águeda Basin, North Portugal	Eucalyptus globulus and P. pinaster plantations	1300–1900	17–19	Sandy loam	Moderate	Yr 1	05-2.2	Shakesby et al. (1996)
							Yr 2	3.2-6.6	
Portugal	Lourizela, Águeda Basin, North Portugal	P. pinaster plantation	1300–1900	-	Sandy loam	Moderate	Yr 1	2	Ferreira et al. (1997)
Portugal	Raivo, Águeda Basin, North Portugal	E globulus plantations	1155 (1300–1900)	Various	Sandy loam	Moderate	9 months	45	Shakesby et al. (1994, 2002)
Portugal	Montesinho Natural Park, Braganza,Trás-os-Montes, North Portugal	Erica australis, Chaemespartium tridentatum, Cystus ladanifer	850	N.A.	Clay loam	Low	Yr 1	0.96-2.77	Fonseca et al. (2017)
Portugal	Vouga River , North Portugal	Pinus Pinaster Pterospartum tridentatum	1200/2000	N.A.	Loam/sandy loam,	High	Yr 1	Degraded=2.57 Semidegraded=0.31 Control=0.04	Hosseini et al. (2016)
							Yr 2	Degraded=3.79 Semidegraded=0.84 Control=0.00	
Portugal	Agueda, North Portugal	Eucalypt	1470	N.A.		Moderate	Yr 1	4.9	Prats et al. (2014)
Portugal	Agueda, North Portugal	Pine	2000	N.A.		Low	Yr 1	0.8	
Portugal	Pessegueiro, South Portugal	Eucalypt	1540	N.A.		Moderate	Yr 1	5.4	
Portugal	Pessegueiro, South Portugal	Pine	1540	N.A.		Low	Yr 1	0.3	
Portugal	Ermida, North Portugal	Eucalypt	1600	N.A.		Moderate and high	Yr 1	8.5	
Portugal	Ermida, North Portugal	Eucalypt	1600	N.A.		Moderate and high	Yr 1	8.5	
Portugal	Colmeal village, Góis, North Portugal	Pine	1100	N.A.		Moderate	Yr 1	2.2	

Table 1. Cont.

Portugal	Colmeal village, Góis, North Portugal	Eucalypt	1133	N.A.	Sandy loam	Moderate	Yr 1	0.25 unplowed site, 0.45 downslope plowed site, 0.55 contour plowed site	Vieira et al. (2016)
							Yr 2	0.33 unplowed site, 0.44 downslope plowed site, 1.37 contour plowed site	
							Yr 3	0.54 unplowed site, 0.82 downslope plowed site,1.32 contour plowed site	
							Yr 4	0.14 unplowed site, 0.38 downslope plowed site, 0.71 contour plowed site	
							Overall period	1.26 unplowed site, 2.09 downslope plowed site, 3.94 contour plowed site	
France	Rimbaud catchment, Massif des Maures, Var	P. pinaster and shrubs	824	11	Sandy loam Rankers soil on gneiss	High	Yr 1	8.8	Lavabre and Martin (1997)
			766				Yr 2	16.3	
			921				Yr 3	8.3	
			1011				Yr 4	0.3	
France	Rimbaud catchment, Massif des Maures, Var	P. pinaster and shrubs	< 1100	9 (Ave)	Sandy loam Rankers soil on gneiss	High	Yr 1	5.7	
							Yr 2	0.7	
							Yr 3	0.8	
France	Massif des Maures, Var	P. pinaster and shrubs	< 1100	< 20	Sandy loam Rankers soil on phyllites	High	Yr 1	12.0 (Gageai 1 catch.)	Martin et al. (1997)
								19.7 (Gageai 2 catch.) >10.6 (Saute catch.)	
Italy	Ittiri, Sardinia	Q. suber	730	11-17	Sandy clay	Low	Yr 3	0.84-0.05 (range)	Canu et al. (2015)

Table 1. Cont.

Italy	SW Sardinia	Macchia scrub	540	20	Sandy loam		Overall period	0.06 (herb past) 0.11 (shrubs) 0.23 (eucalyptus)	Vacca et al. (2000)
Italy	Tuscany	Matorral shrubs		15	N.A.	Low (L) and high (H)	Yr 1	0.04 0.1(L), 0.7 (H)	Giovannini (1997)
Italy	Sant'Angelo creek, Sarno, Campania	oak trees typical Mediterranean scrub	1000-1500	35	Gravelly muddy sand	Moderate and high	1 month	19.8-33.1	Esposito et al. (2017)
Italy	Rio Mannu, North Sardinia	Macchia scrub	500-700	8.5 (10-63)	Granite/sandy-loam	High	Yr 1	7.18-45	Rulli et al. (2012)
Italy	Ottava, Sassari, Sardinia	Grassland, herbaceous pastures	542	17	Sandy loam	N.A.	N.A.	2.55-0.86	Porqueddu et al. (2001)
Italy	Bonassai, Sardinia	Grassland, herbaceous pastures	500	0	Clay loam	N.A.	N.A.	0.02	Acutis et al. (1996)
Italy	Pattada, Sardinia	Quercus suber L., Cistus monspeliensis	650	14	Sandy loam	N.A.	N.A.	0.03	Rivoira et al. (1989)
Croatia	Zrnovnica river basin, nr Split	P. halepensis	826	30	Skeletal colluvial soil	High	Yr 1	0.1	Butorac et al. (2009)
							Yr 2	0.0002	
							Yr 3	0.0012	
							Yr 4	0.0025	
Greece	SE of Thessaloniki	P. halepensis and shrubs	420	N.A.	N.A.	High	9–26 months	0.8	Spanos et al. (2005)
Greece	Lesvos	P. brutia	614 (658)	19	Chromic luvisol	Moderate and high	6 months	1.25 (M), 12–22 (H)	Dimitrakopoulos and Seilopoulos (2002)
Israel	Bet Oren, Mt Carmel	P. halepensis and Quercus calliprinos and shrubs	713 (Yr 1)	11-17	Sandy loam	High	Yr 1	0.9 (N)	Inbar et al. (1997); Wittenberg and Inbar (2009)
			501 (Yr 2)				Yr 2	3.7 (S)	
			1207 (Yr 3)				Yr 3	0.10 (N), 0.30 (S)	
			717 (Yr 4)				Yr 4	0.0014 (N), 0.0009 (S)	

Table 1. Cont.

Israel	Galim, Mt Carmel	P. halepensis and Quercus calliprinos and shrubs	713 (730)	N.A.	N.A.	High	Yr 1	0.036	Inbar et al. (1998)
Israel	Yoqneam forest, Mt Carmel	P. halepensis and P. brutia plantation	550	24	Sandy loam	Low and moderate	5 months	0.0005	Kutiel and Inbar (1993)
Israel	Bet Oren, Mt Carmel	P. halepensis and Quercus calliprinos and shrubs	690	11-17	Sandy loam	Low and moderate	Yr 1	0.12-0.6	Lavee *et al.* (1995)

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Final conclusions

I showed the effects of different treatment scenarios on wildfire exposure in an area dominated by herbaceous fuel type and demonstrated that the treatments realized near to road are the most effective in this case study. This work can be useful to investigate the use of fuel treatments and their spatial arrangements in predominantly herbaceous landscape with space limits designated for treatments.

Moreover I highlighted the use of fire spread and behavior models linked to ERMiT modeling approach to identify the areas affected by major fire and erosion risk. I evaluated the effect of fuel treatments on post-fire sediment yields and compared multiple wildfire and sediment delivery scenarios across large study areas. The results showed that this study can be helpful to plan fuel treatments in areas exposed to severe wildfire events and to consequent erosion processes.

The methodologies proposed in this thesis can be helpful for land managers and policy makers to plan the best strategies to mitigate risk related to wildfires, which pose serious threats in the Mediterranean Basin, particularly in the areas historically prone to large wildfires.

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