



## Wildfires in Europe: Burned soils require attention

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### ABSTRACT

Annually, millions of hectares of land are affected by wildfires worldwide, disrupting ecosystems functioning by affecting on-site vegetation, soil, and above- and belowground biodiversity, but also triggering erosive off-site impacts such as water-bodies contamination or mudflows. Here, we present a soil erosion assessment following the 2017's wildfires at the European scale, including an analysis of vegetation recovery and soil erosion mitigation potential. Results indicate a sharp increase in soil losses with 19.4 million Mg additional erosion in the first post-fire year when compared to unburned conditions. Over five years, 44 million Mg additional soil losses were estimated, and 46% of the burned area presented no signs of full recovery. Post-fire mitigation could attenuate these impacts by 63–77%, reducing soil erosion to background levels by the 4th post-fire year. Our insights may help identifying target policies to reduce land degradation, as identified in the European Union Soil, Forest, and Biodiversity strategies.

### 1. Introduction

According to the Global Wildfire Information System (Boschetti et al., 2021), on average, 448 million hectares of land burn every year (2012–2021), whereas 1.4 million hectares correspond to the European Union (EU) and the United Kingdom (UK) alone. This global phenomenon is often responsible for substantial environmental, social and economic losses, which together with land abandonment, droughts, absence of appropriate land management and urban development planning, are expected to aggravate land degradation. In addition, wildfires are becoming a persistent threat, since the fire risk is expected to increase in a context of a warmer and drier climate (Bednar-Friedl et al., 2022).

Under post-fire conditions, it is possible to observe a reduction in the infiltration by the soil, a reduction of ground cover protection in the form of litter and vegetation due to its combustion, and the alteration of soil aggregate stability due to the impact of the high temperatures reached during the wildfire event (Santín and Doerr, 2016; Moody et al., 2013; Shakesby and Doerr, 2006). These three factors combined can lead to a significant increase (mean 2.5–15.9-fold) in soil erosion when compared to the background levels (i.e. pre-fire conditions), but with a highly variable outcome between geographical regions and burn severity levels (Vieira et al., 2015). Higher burn severity increases

on-site erosion, but also off-site impacts downstream of the burned area such as destructive floods and debris flows (Moody et al., 2013).

Despite being regarded as an important hydrological and geomorphological agent (Shakesby and Doerr, 2006), post-fire soil erosion has mostly been assessed from plot to catchment scale by means of field measurements and from slope to regional scale using modelling approaches (Lopes et al., 2021). Model-based soil erosion assessments made at larger spatial scales, such as European (Panagos et al., 2015, 2020) and Global scale (Borrelli et al., 2017, 2020), haven't considered the effects of wildfires in the entire soil erosion budget. Notwithstanding, Borrelli et al. (2016) estimated the combined impact of wildfires and forest management in European forests, however, several drivers of post-fire soil erosion such as burn severity, vegetation recovery, and mitigation were not addressed.

What differs a post-fire soil erosion assessment from the latest soil erosion assessments is the level of uncertainty in which wildfires affect soil surface physical, chemical, and biological properties, combined with the degree of the vegetation consumption and their post-fire recovery (Lopes et al., 2021). Post-fire hydrological and erosive response are expected to increase substantially after the wildfire event, however, the degree of this increase highly depends on the soil burn severity, post-fire weather dynamics, pre-fire ecosystem condition, history of antecedent disturbances in the area, and if any post-fire management

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operations are taking place (Basso et al., 2022; Moody et al., 2013; Shakesby, 2011; Shakesby and Doerr, 2006).

The most widely used models to estimate post-fire soil erosion are empirically based, being the Universal Soil Loss Equation (USLE; Wischmeier and Smith, 1978) and its revised form RUSLE (Renard et al., 1997) on the top of the researcher's choice for burned (Lopes et al., 2021) and unburned conditions (Borrelli et al., 2021). Such wide application of RUSLE is related to the fact that it requires less demanding input data when compared to physically-based models (Merritt et al., 2003), benefiting also by large-scale data availability in comparison to other available models (Vieira et al., 2018). However, USLE and RUSLE were developed to estimate soil erosion in agricultural lands, and are not adapted to consider the impacts of wildfires on vegetation (Hosseini et al., 2018; Morrison and Kolden, 2015) and soil properties (Fernández and Vega, 2018; Nunes et al., 2018a; Moody et al., 2013).

The need to adapt models to post-fire conditions and validate their predictions has been stressed in several studies (Parente et al., 2022; Fernández and Vega, 2018; Vieira et al., 2018; Larsen and MacDonald, 2007). Most of these adaptations are typically achieved by introducing an empirical 'fire factor' or by adjusting input parameters known to be impacted by wildfires such as ground cover, surface roughness, or soil hydraulic properties (Lopes et al., 2021; Vieira et al., 2018). However, accurate post-fire predictions of hydrological and geomorphological effects in fire-affected scenarios still present several obstacles (Lopes et al., 2021), not only because of regional differences in climate, soil properties, and wildfire characteristics, but also due to model limitations in representing soil erosion processes (Nearing, 1998). Notwithstanding, there is still a need to quantify the impacts of wildfires at larger scales, evidencing not only the expected increase in soil erosion in these locations, but also their off-site effects such as the destructive transport of debris or the high ash and sediment loads transport onto downstream water bodies and infra-structures (Basso et al., 2020, 2022; Santi and Rengers, 2020). These predictions also urge to include the latest developments on the post-fire mitigation measures (Girona-García et al., 2021) which have shown to effectively reduce the on-and-off site effects.

Therefore, the aim of this study is to upscale the existent methodologies to estimate post-fire soil erosion at the European scale. To accomplish that, we estimated the soil erosion following the 2017's wildfires for a 5-years period, compared it against pre-fire condition, and post-fire mitigation measures application. In this way it was possible to assess not only the impact of the wildfires *per se*, but also to identify areas in which ecosystems failed to recover during such time span and evidence the role of restoration practices halting land degradation.

## 2. Methods

### 2.1. Methodological approach

The widely use of open access data at European and Global scales evidences the importance in the provision of spatial products by the Joint Research Centre and Copernicus for land management purposes. Available large scale data by the European Soil Data Centre (ESDAC, 2021), on soil properties (Orgiazzi et al., 2018), rainfall erosivity (Panagos et al., 2017), or global soil erosion assessment (Borrelli et al., 2017), but also from the European Forest Fire Information System (EFFIS, San-Miguel-Ayanz et al., 2012) on current and historical wildfire data, motivated us to provide a large scale post-fire soil erosion map. Therefore, we opted for a European post-fire soil erosion assessment with RUSLE model (Panagos et al., 2015), for the following reasons:

- **Follow-up EU assessment** – Previous soil erosion assessment at this scale (Borrelli et al., 2016) have not considered distinct wildfires impacts according to its burn severity (Vieira et al., 2015), neither a window of disturbance profile on soil erosion as function of vegetation recovery (MacDonald and Larsen, 2009);

- **Data availability and model representability** - Despite being an empirical model, RUSLE is by far the most used soil erosion model worldwide (Borrelli et al., 2021), and its inputs data are widely available and recognized;
- **Bibliographic support** – There is already a substantial number of scientific studies approaching post-fire soil erosion worldwide (Girona-García et al., 2021; Lopes et al., 2021; Vieira et al., 2015) which can be used to support the development of a post-fire factor, and although such studies are not evenly distributed worldwide their physical principles still apply.

To accomplish an European post-fire soil erosion assessment, this study involved three main sources of data (Fig. 1), a meta-data resulting from bibliographic research on field and modelling post-fire studies, previous soil erosion assessments at European scale for baseline estimations, and spatial data for the determination of wildfires impact on ecosystems and the correspondent vegetation dynamics from pre-fire conditions to post-fire recovery.

The reference Pan-European assessment made for 2016 (Panagos et al., 2020), and the reference dataset for land cover fractions used of 2015 (CGLS-LC100, Buchhorn et al., 2020), led us to choose the 2014–2016 interval for the assessment of pre-fire conditions. Allowing a close temporal comparability between the previous assessment, and also a minimum of three years for calibration without interference from previous wildfires. While the choice of performing a 5-year assessment was motivated by the possibility of including a more complete picture of the post-fire window of disturbance given the fact that most of the post-fire studies only assess soil losses in the first 1–3 post-fire years (Basso et al., 2022; Lopes et al., 2021).

### 2.2. Post-fire conditions

In order to implement the fire factor into the RUSLE baseline predictions, a bibliographic review was performed so that the model adaptations would consider the latest knowledge development under post-fire soil erosion predictions as well as the best post-fire soil erosion measurements available. Our bibliographic search approached post-fire data published in peer-review journals in order to obtain four important indicators for post-fire soil erosion and its mitigation:

- **Post-fire modelling methods and inputs**, from studies performing modelling assessments with RUSLE in order to understand which methodologies were more often applied and accepted by the scientific community, and also to understand which inputs are more frequently used to represent wildfire impacts on soil erosion (Supplementary 3);
- **Post-fire anomalies**, of runoff and erosion under simulated rainfall from pre-to post-fire conditions in order to estimate the alteration of these parameters indirectly according to their burn severity (Supplementary 4, Vieira et al., 2015);
- **Post-fire soil erosion measurements**, of field studies assessing post-fire soil erosion under natural rainfall conditions in order to provide model prediction boundaries and data for model validation (Supplementary 1).
- **Post-fire mitigation measures efficiency**, of post-fire runoff and erosion under natural rainfall following Girona-García et al. (2021) in order to integrate their impact in the soil erosion estimations.

Such reviews were performed in Scopus database under the specifications used for Lopes et al. (2021) and Vieira et al. (2015), involving different combinations of search terms (fire, wildfire, erosion, sediment yields, hydrology). Following the data retrieval, all datasets required some treatment and harmonization. Inputs used were subjected to a quality assessment, by verifying if the resulting post-fire soil erosion predictions were compared and evaluated against field-based erosion rates measurements, and by evaluating author's methodological

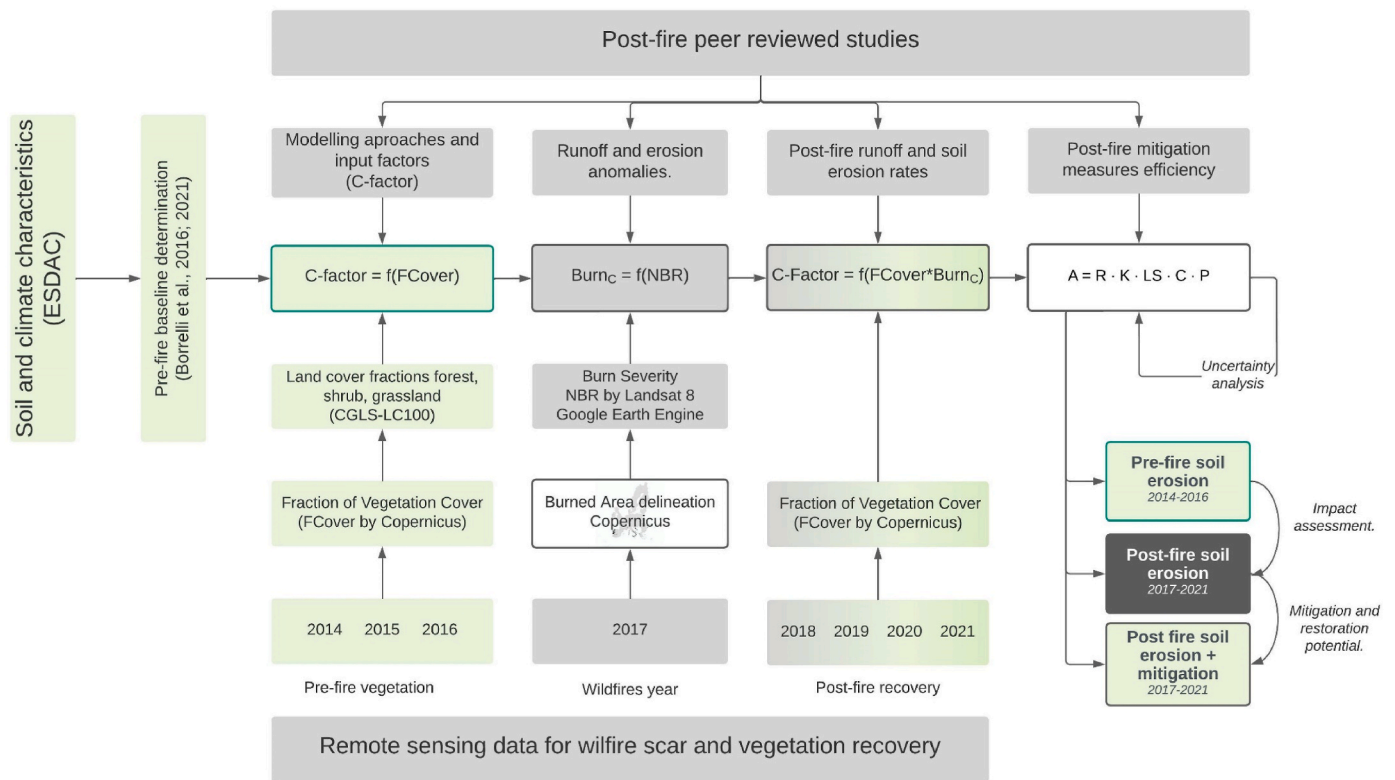


Fig. 1. Overall methodological modelling approach.

approaches for inputs estimation. Then all the data was temporally and geographically located and categorized within co-variables of interest, such as burn severity, time-since fire, pre-fire land use, and land cover attribute.

### 2.3. RUSLE model and pre-fire conditions

This study used the Universal Soil Loss Equation (USLE), as modified by Dismeyer and Foster (1984) for forest, shrubland and grasslands (Buchhorn et al., 2020), in order to model the average soil erosion potential in the absence of wildfire by including ground cover changes described by Hansen et al. (2013) which have been also modelled by Borrelli et al. (2016). This baseline ad model will then be used to determine the additional impact of forest fires at the global scale for the short term (1–5 post-fire years). USLE model uses the following factors to estimate soil loss:

$$A = R \cdot K \cdot LS \cdot C \cdot P \quad (1)$$

where,  $A$  = soil loss ( $\text{Mg ha}^{-1} \text{ yr}^{-1}$ ),  $R$  = rainfall and runoff factor ( $\text{MJ mm h}^{-1} \text{ ha}^{-1} \text{ yr}^{-1}$ ),  $K$  = soil erodibility factor ( $\text{Mg h MJ}^{-1} \text{ mm}^{-1}$ ),  $LS$  = slope length and the slope steepness factor (dimensionless), and  $CP$  = cover management practice factor (dimensionless).

The  $K$  and  $R$  factors were expressed spatially using the latest maps created by the soil research team of the Joint Research Centre of the European Commission (Panagos et al., 2015).

In the original equation, the cover and management factor ( $C$ ) was developed for agricultural croplands. It has to be adjusted to the targeted ground cover characteristics (forest, shrub and grasslands) to be able to spatially differentiate the various cover species, canopy cover densities and ground floor vegetation (Panagos et al., 2015; Dismeyer and Foster, 1984). Considering the size of the study area, the  $C$ -factor used to predict the soil loss potential was estimated as Borrelli et al. (2016), whereas the vegetation density was quantified by manipulating the biophysical parameter Fraction of Vegetation Cover (FCOVER, Fuster et al., 2017).

This provided an estimation of the fraction of the vegetation that is visible vertically, allowing to differentiate a  $C$  factor for undisturbed conditions ( $C_U$ ) between bare or protected soil for forest, shrub, and grasslands (Fig. 2).

### 2.4. Post-fire estimations and uncertainty

With regard to the wildfire disturbance the state-of-the-art identified that post-fire soil erosion rates and vegetation recovery, are closely related to burnt severity and bare soil cover (Girona-García et al., 2021; Vieira et al., 2015). Notwithstanding, as evidenced by Johansen et al. (2001), forests, shrubland, and grasslands are distinctively impacted by wildfires, whereas contrary to the undisturbed condition, burned forests present much higher post-fire soil erosion rates in comparison to shrubland and grasslands due to higher fire residence times and burn severities. Therefore, the  $C$  factor for burned conditions (Fig. 2) was approximated as follows:

$$C_{max_{B_{Forest}}} = C_{max_{U_{Forest}}} \times B_{severity} \times 10 \text{ for } FCOVER = 0.2, \text{ max} = 0.5 \quad (2)$$

$$C_{max_{B_{Shrub}}} = C_{max_{U_{Shrub}}} \times B_{severity} \text{ for } FCOVER = 0.2, \text{ max} = 0.3 \quad (3)$$

$$C_{max_{B_{Grass}}} = C_{max_{U_{Grass}}} \times B_{severity} \text{ for } FCOVER = 0.2, \text{ max} = 0.3 \quad (4)$$

where, for equations (2)–(4), the maximum  $C$  factor for burned conditions ( $C_{max_B}$ ) was estimated by multiplying the maximum  $C$  factor for undisturbed conditions ( $C_{max_U}$ ) and the burn severity anomaly factor ( $B_{severity}$ ) as derived by Vieira et al. (2015). Such maximum value was determined for ground cover (FCOVER) values of 0.2, similarly to the approach taken by Wischmeier and Smith (1978). Regarding the individual cover types, for forest (2) the  $C$  factor was increased one order of magnitude for all the severities and limited to a maximum of 0.5, while shrubland (3) and grasslands (4) were represented by low and moderate severities and limited to a maximum of 0.3. In this way was possible to accommodate the impacts determined by Vieira et al. (2015) for

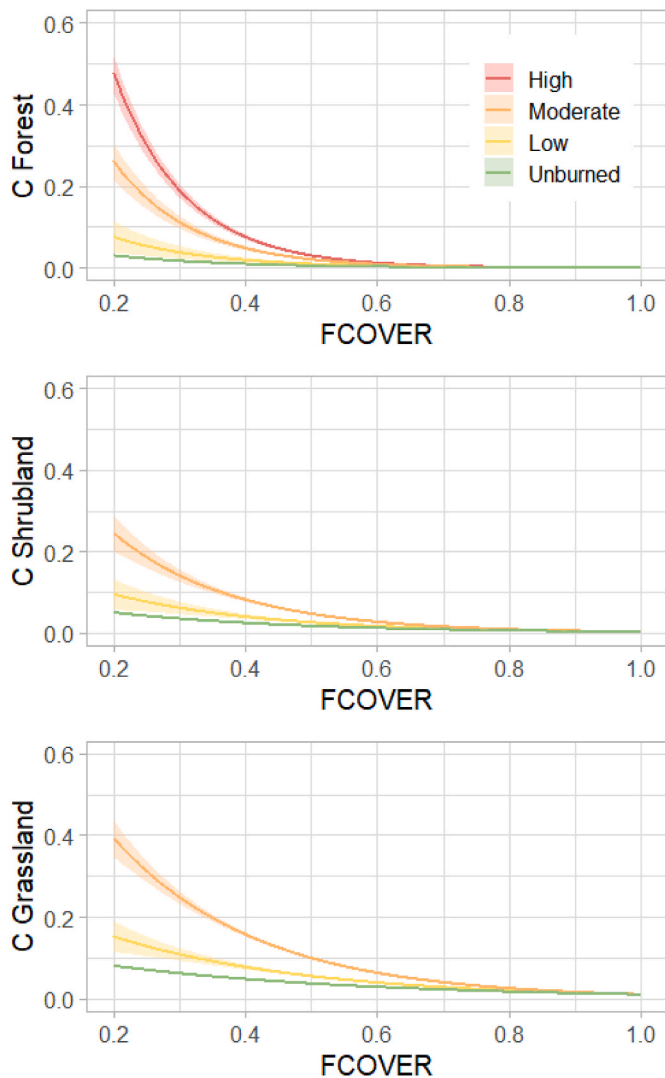


Fig. 2. C factor derived for unburned and burned conditions for forest, shrubland and grassland, and according to low, moderate and high burn severity.

distinctive severities, as also the conceptual model of Johansen et al. (2001) for distinct cover types (Fig. 2).

The recovery of the burned areas was addressed by using vegetation cover indicators as proxies, which consisted in merging two criteria:

- 1) a burned area is considered recovered once the ground cover reaches 100%;
- 2) a burned area is considered recovered once the ground cover reaches the pre-fire conditions.

In this way, it was possible to determine erosion reduction with the increase of the protective cover, but also consider the initial characteristics of the local vegetation before the wildfires.

For the first point, a minimum C factor was estimated:

$$C_{min_B} = C_{min_U} \text{ for } FCOVER = 1.0 \tag{5}$$

Being the minimum C factor for burned conditions ( $C_{min_B}$ ) approximated to the ( $C_{min_U}$ , Panagos et al., 2015) for each land cover (forest, shrub, and grassland) whenever the ground cover (FCOVER) reached its maximum value of 1.

This formulation allowed to include the impact of the burn severity in the soil erosion estimations for when protective cover presented the

minimum value according to the burn severity impact, and approximate the recovery of the burned areas under an exponential curve in function of FCOVER over a 5-years' time period (Fig. 2). The choice of an exponential base for such estimations goes in line with the Johansen et al. (2001) conceptual model after observing a non-linear relationship between sediment yields and bare soil cover, and a sharp increase of the sediment yields on the 60–70% bare soil cover.

The second point of the recovery to the background levels, was motivated by the FCOVER distribution sample for pre-fire conditions. For the 3 years preceding the wildfires, the mean FCOVER value rounded 0.43, thus representing the mixture of Forest, Shrubs and Grasslands within our AOI. This sample presented a normal distribution among the 3 pre-fire years, and the mean FCOVER value obtained for this period was used to assess the rate of recovery.

Therefore a recovery rate (RCOVER) was calculated, and several classes were defined as follows:

$$RCOVER_t = \frac{FCOVER_t}{FCOVER_{pre-fire}} \tag{6}$$

No Recovery :  $0.0 \leq RCOVER < 0.3$

Ongoing Recovery :  $0.3 \leq RCOVER < 0.6$

Near Recovery :  $0.6 \leq RCOVER < 1.0$

Recovered :  $RCOVER \geq 1.0$

The consideration of a RCOVER class allowed us to adopt a succession of C-factor equations according to the status of the vegetation recovery progress. For the first 2 classes 'No Recovery' and 'Ongoing Recovery' the C factor would be calculated as described for equations (2)–(5) according to the initial burn severity classification and land cover attributes. In the case of 'Near Recovery' class, this would led to the adoption of the equation with one burn severity level lower than the original one, likewise the 'Recovered' class, would led to adoption of the equation corresponding to unburned conditions. This formulation allow us to use FCOVER as constant input for C estimations, while reducing the impact of severity proportionally to the pre-fire conditions.

Finally, the impacts of the application of mitigation measures was also simulated for the Area of Interest (AOI), and for the entire study period according to three scenarios:

- I. The entire burned area was treated with mulching immediately after the fire (full mitigation), describing the maximum mitigation potential for such technique.
- II. The treatment with mulching was restricted to areas burned at high and moderate burn severity (burn severity risk based), describing a potential scenario of decision-making without having access to modelling tools.
- III. The treatment with mulching was restricted to areas where the soil erosion risk was unsustainable (soil erosion risk based), whereas soil losses were above the  $2 \text{ Mg ha}^{-1} \text{ y}^{-1}$  threshold similarly to the one considered in Panagos and Katsoyiannis (2019) and Panagos et al. (2020).

For those estimations, the P factor was used according to the effectiveness values obtained by Girona-García et al. (2021), considering effect of mulching treatments without specifying the material (e.g. straw mulching, forest residues) or the technique used (e.g. manual spreading, hellmulching). Should be noted that the potential increase in ground cover originated from those treatments was not simulated, therefore this represents a conservative estimation based only in the mitigation treatment efficiency obtained from field studies.

## 2.5. Error propagation and model validation

Soil erosion predictions uncertainty ranges were determined individually for unburned, post-fire and for both mitigation scenarios tested. Given the fact that RUSLE is a linear model and that a confidence interval (maximum and minimum conditions) was provided for individual pixels, error propagation was calculated considering:

- Unburned – rainfall erosivity ranges as provided by Panagos et al. (2017);
- Post-fire periods – cumulative effect between rainfall erosivity ranges and C factor confidence intervals as determined by Vieira et al. (2015) and estimated for land cover (Fig. 2);
- Mitigation treatments – cumulative effect between rainfall erosivity ranges, C factor confidence intervals, and mitigation treatments efficiency ranges as determined by Girona-García et al. (2021).

Post-fire soil erosion field measurements were compared against the predicted ones by severity classification. The model validation was made by comparing soil losses measured during the first post-fire year in 9 groups of studies located in the Iberian Peninsula (111 field observations). These groups were aggregated by proximity within a radius of 25 km for NW Spain and Portugal, and 100 km for 4 studies located NE and SE Spain (Supplementary Fig. 1) due to the lack of 2017 fires in the surroundings of the field studies. To circumvent the fact that locations of the individual studies didn't match the 2017's burned area, the soil erosion rates median and standard deviation were extracted for all burned areas within the defined buffers (Supplementary Fig. 1).

## 2.6. Spatial data processing

The AOI and wildfire date, which corresponded to the 2017 burned areas of the EU and UK, was obtained by the European Forest Fire Information System (EFFIS, San-Miguel-Ayanz et al., 2012). The inclusion of UK was motivated by the fact this former Member State was integrated in the previous soil erosion assessments (Panagos et al., 2020), and therefore was possible keep the same soil erosion figures for comparability.

Within the AOI the burn severity of the 2679 individual fires was determined with the difference of normalized burn ratio (dNBR, Holsinger et al., 2021). Two satellite imageries were tested for the dNBR estimation as images from Sentinel-2 and Landsat 8. As the Landsat 8 images covered more area of the burned areas, these were used for the dNBR estimation. In particular, Landsat 8 level 2, collection 2, and tier 1 images were used (LANDSAT/LC08/C02/T1\_L2 in Google Earth Engine (GEE) platform). The area with the dNBR estimation for each fire was calculated, as well as the percentage of the area of each fire with dNBR estimation. The total burned area in 2017 in Europe was 1,251,513 ha and for 94.15% of the areas the dNBR was estimated. The dNBR was then converted to Low, Moderate and High burn severity classes, according to Key and Benson (2006), as the best estimation possible at the EU scale.

Given the heterogeneity of the land cover types that can be affected by wildfires, the reference dataset for land cover fractions used was the Dynamic Land Cover map at 100 m resolution (CGLS-LC100, Buchhorn et al., 2020), which allowed to estimate the individual contributions of soil erosion for forest, shrub, and grassland for individual pixels, considering the pre-fire land cover characteristics of 2015. Cropland was excluded from this analysis due to the fact that the burn severity anomaly factor used does not include the cumulative impact of fire and soil management operations on soil erosion. This procedure resulted in a reduction of our target area to 771,375 ha.

The ground cover changes were assessed with the Fraction of Vegetation Cover (FCOVER, Fuster et al., 2017), which provides a measure of the fraction of ground covered by green vegetation. The change in vegetation cover was determined annually (2014–2020) for the

reference date of 31 November, whereas for 2021 the closest available reference date available was 31 August. The selection of such reference date was motivated by the fact that most of the Member States fire seasons were finished by the autumn, and still was possible to include initial ground cover reductions caused by the wildfire for the first post fire year estimations. To ensure a constant area comparison for the entire study period, pixels in which FCOVER indices was not available, and were not possible to be replaced with values within one month range were also removed from the estimations, reducing thus the study area to the final value of 633,429.5 ha.

Regarding the choice of the reference sites for illustrating local post-fire impacts, Portugal, Italy, and Greece were selected because such MS were among the ones with the highest burned area for 2017, but also due to the author's local knowledge of the sites. The area selected to represent Portugal concerns a cluster of six burned areas with high proximity, burned between 17 June and October 15, 2017 in central Portugal. While for Italy and Greece, this selection occurred among the ones with greatest burned area available, resulting in isolated wildfire events.

## 3. Results

### 3.1. Soil erosion following the 2017 wildfires

During 2017, 2679 wildfires occurred within the EU space, affecting approximately 1.2 million ha of land, which is below the 1.4 million ha 10-years average (GWIS, 2022). The wildfire occurrence was the greatest within the Mediterranean basin, affecting Member States (MS) such as Portugal, Spain, Italy and Greece the most.

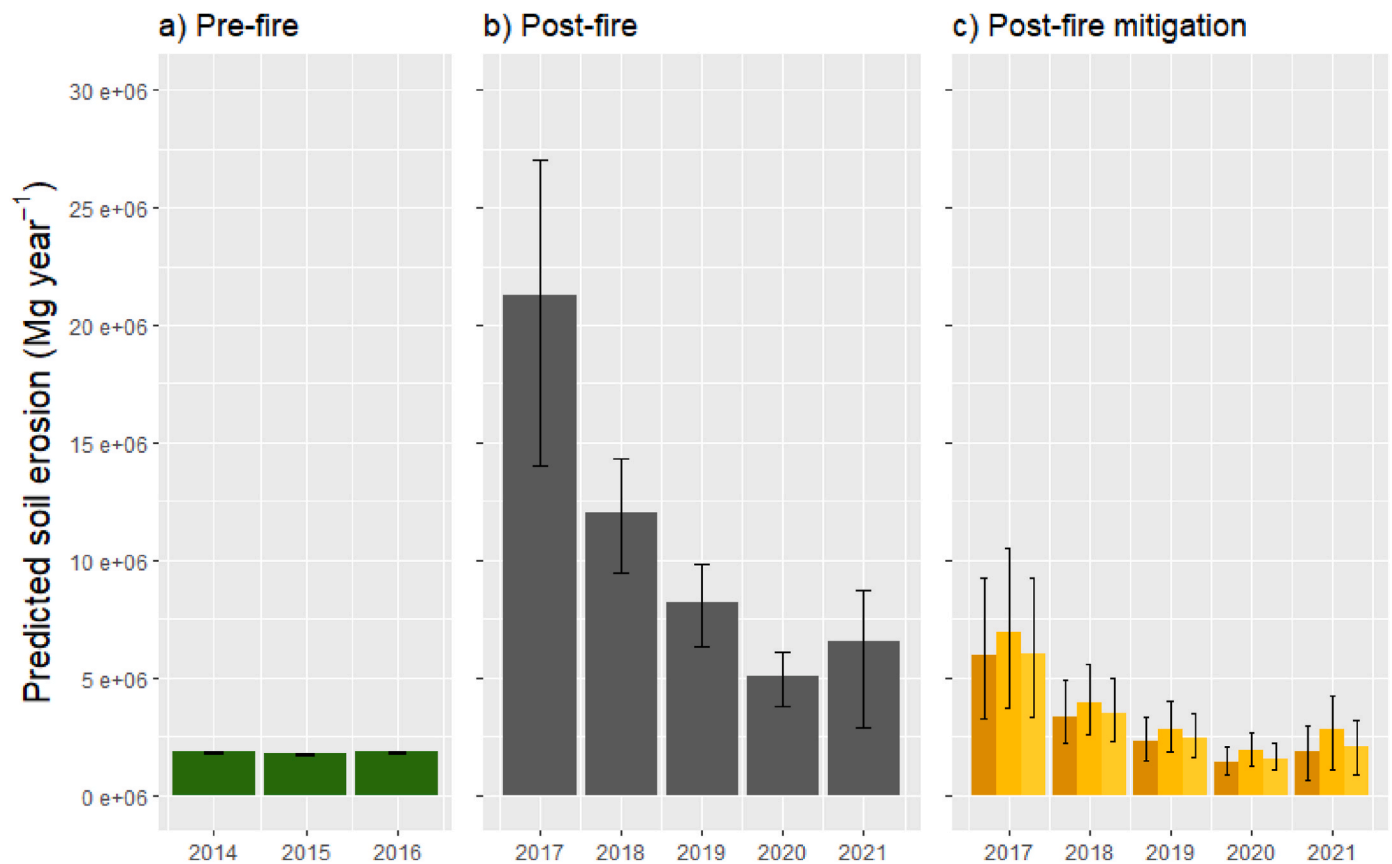
Soil erosion was possible to be assessed in 51% of the total burned area (633,429.5 ha), which corresponds to the land cover of forest, grassland and shrubland, burned at low (17%), moderate (51%), and high (32%) burn severities, referred as AOI. In what concerns the pre-fire conditions (2014–2016), our results agree with the assessment made for 2016 (Panagos et al., 2020), resulting in a mean 1.8 million Mg (Fig. 3a) of soil losses against the 2.2 million Mg for the same AOI. Regarding the impact of wildfires, our results evidence an 11.8 fold increase on the annual soil erosion rates generated when comparing the pre-fire conditions ( $2.8 \pm 0.5 \text{ Mg ha}^{-1} \text{ y}^{-1}$ ) with the first post-fire year ( $33.5 \pm 10.30 \text{ Mg ha}^{-1} \text{ y}^{-1}$ ), corresponding to an increase to  $21.2 \pm 8.5$  million Mg (Fig. 3b) of soil being displaced in a single year.

The impact of this single wildfire year represents an addition of 2% to the reference annual soil losses estimated at EU level (Panagos et al., 2020) for the first post-fire year despite representing only 0.2% of the previous area ( $633.4 \times 10^3 \text{ ha}$  vs  $394.1 \times 10^6 \text{ ha}$ ). Over the study period, our results suggest that a single wildfire year will continue contributing with at least 1% additional erosion to the annual reference, through the entire 5-year period (Fig. 3).

The estimated soil losses after the wildfires follow the classical window of disturbance model (Shakesby and Doerr, 2006), expressing a decrease of soil erosion with time since fire (Fig. 3b). As expected, the maximum soil losses were predicted for the first post-fire year (2017) summing up  $21.2 \pm 8.5$  million Mg, while the minimum amounts were predicted for 2020, summing up  $5.0 \pm 2.3$  million Mg. Despite the mean ground cover increase from 2020 to 2021, as derived by the Fraction of Vegetation Cover (FCOVER), soil erosion estimates unexpectedly increased from the 4th to the 5th post-fire year. Nonetheless, in both cases such erosion figures are still 3–4 fold greater than the pre-fire conditions (Fig. 3).

Regarding the distribution of these impacts over the EU and UK, our results suggest that post-fire soil erosion rates are also higher within the Mediterranean basin, namely in MS such as Portugal, Spain, Italy and Greece (Fig. 4), which have been identified as fire-prone regions and a frequent target of fire-induced land degradation (Shakesby, 2011).

Due to the lack of field studies assessing post-fire soil erosion at EU scale, it was only possible to compare the results of the current study



**Fig. 3.** Total soil erosion estimates for a) pre-fire conditions (green), b) post-fire conditions (grey), and c) post-fire scenarios with full mitigation (dark yellow), burn severity risk mitigation (yellow), and soil erosion risk based mitigation (light yellow) schemes (see 2.4) for the AOI. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

within the areas burned in the Iberian Peninsula (Supplementary Fig. 1). The validation of the mean soil erosion estimations results with field data at various scales (0.25–6533 m<sup>2</sup>), generally revealed a potential overestimation of the soil losses for the first post-fire year. This overestimation is more evident in sites burned with moderate and low severity (Supplementary Fig. 2), but still within the magnitude of uncertainty ( $\pm 10.30 \text{ Mg ha}^{-1} \text{ y}^{-1}$ ) obtained for the first post-fire year.

### 3.2. Post-fire recovery and mitigation

The ground cover derived by the FCOVER shows a steady recovery of the vegetation until 2021 (Fig. 5a), in which the 5th post-fire year revealed very similar profile to the pre-fire conditions and mean ground cover values (pre = 0.43 vs. 2021 = 0.45, Fig. 5b).

However, when analysing FCOVER evolution for individual pixels (Fig. 5c), it was possible to observe that by 2021, 46% of the burned area was still not considered fully recovered by presenting a Recovery Rate (RCOVER) index below 1. Moreover, in 2021 it was possible to observe an important increase (to 0.7%) of the 'No Recovery' class which was substantially lower by 2020 (0.1%), thus explaining the increase of the soil erosion estimation for 2021 (Fig. 3b).

When simulating the application of the most efficient post-fire mitigation measures (cover treatments) immediately after the wildfire (Fig. 3c), it was possible to estimate a reduction of soil erosion below background levels by the 4th post-fire year with the full mitigation scenario (i.e. total burned area treated). While for the burn severity risk scenario (i.e. only high and moderate burn severity treated) such efficiency resulted in insufficient protection to keep the soil erosion rates below the pre-fire conditions, with mean erosion 1.1 to 1.6 times higher than the background levels, between 2020 and 2021 despite the high

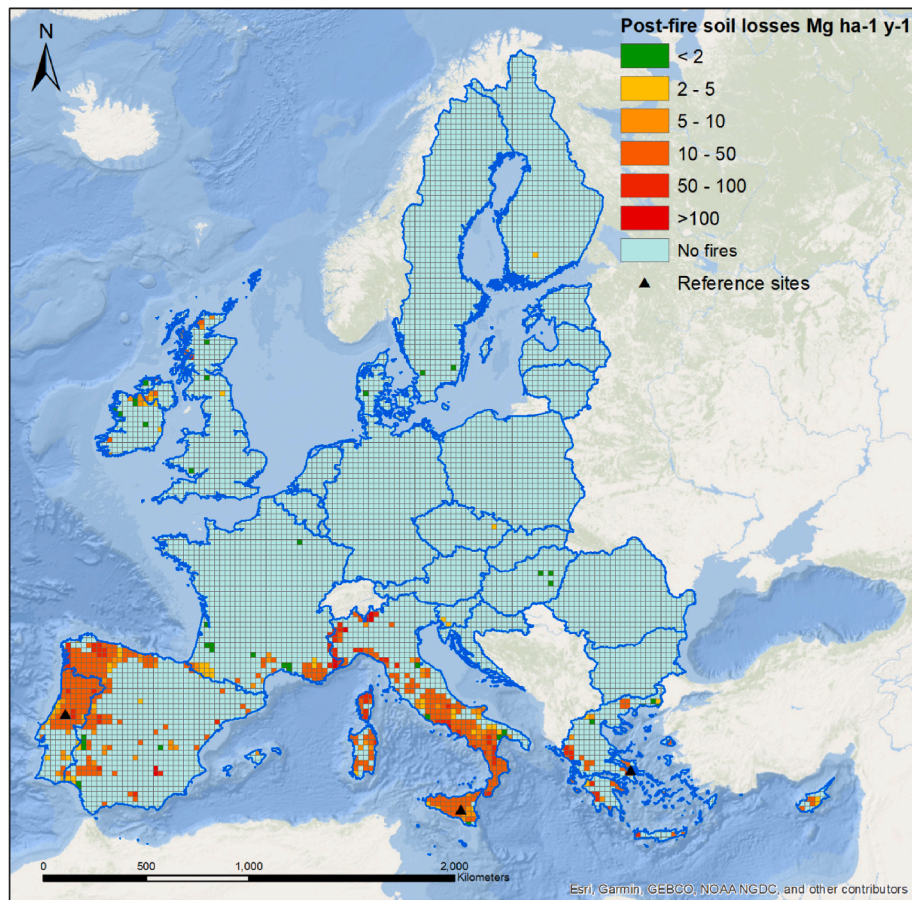
coverage of the treatment area (83%) Notwithstanding the lower efficiency the high risk scenario, it would still allow the prevention of 34.6 million Mg of displaced soil. When considering a soil erosion risk scenario (i.e. only erosion rates above  $2 \text{ Mg ha}^{-1} \text{ y}^{-1}$  treated), the efficiency was approximated to the full mitigation scenario with an expected recovery by the 4th post-fire year, with a reduction of the treatment area to 91%.

### 3.3. Reference sites

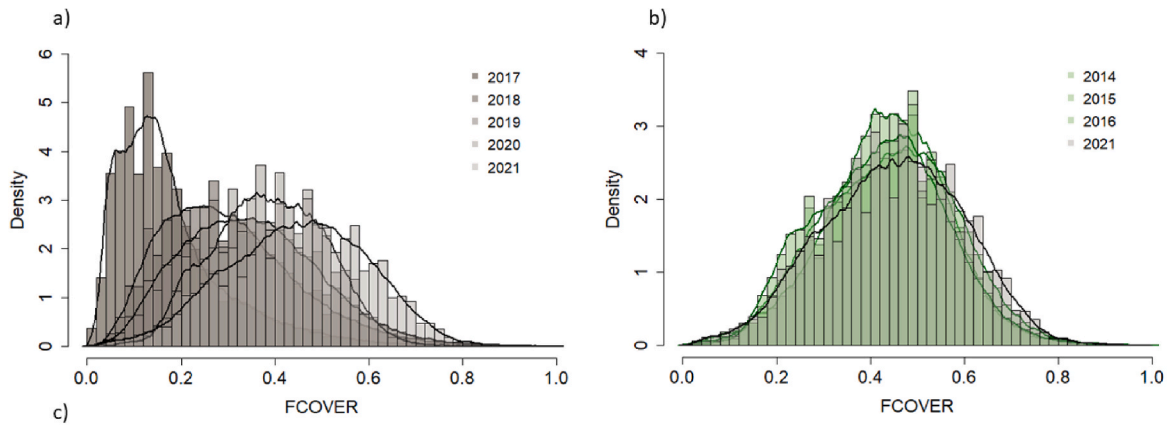
The 2017 wildfires affected a wide range of MS, which on their turn affected various land cover types, soil characteristics, and weather conditions. In order to illustrate such variability, three reference sites were chosen: i) Central Portugal, ii) Attica for Greece, and iii) Enna for Italy (Fig. 6).

The wildfires in Central Portugal (Fig. 6a) concerned approximately  $190 \times 10^3 \text{ ha}$  of land, becoming an iconic reference for wildfire research such as fire behaviour (Ribeiro et al., 2020), impacts on water quality (Basso et al., 2021), or remote sensing (Brown et al., 2018). Our estimations resulted in more than  $7.1 \times 10^6 \text{ Mg}$  of soil loss ( $37.4 \text{ Mg ha}^{-1}$ ) during the first post-fire year. This rate of soil loss decreased steadily with time since fire, however without achieving full recovery by the end of the assessment period ( $\text{RCOVER} \geq 1.0$  2021 = 62%).

In the case of the Attica burned area ( $2.3 \times 10^3 \text{ ha}$ , Fig. 6b), soil erosion estimations rounded the  $0.8 \times 10^6 \text{ Mg}$  of soil loss, resulting in very similar soil loss rates as Central Portugal for the first year ( $36.2 \text{ Mg ha}^{-1}$ ) despite the reduction of scale. Nevertheless, not only the recovery of the area was not progressively improving with time as the former example, as it also never reached total recovery ( $\text{RCOVER} \geq 1.0$  2020 = 58%,  $\text{RCOVER} \geq 1.0$  2021 = 38%) due to several forest interventions that

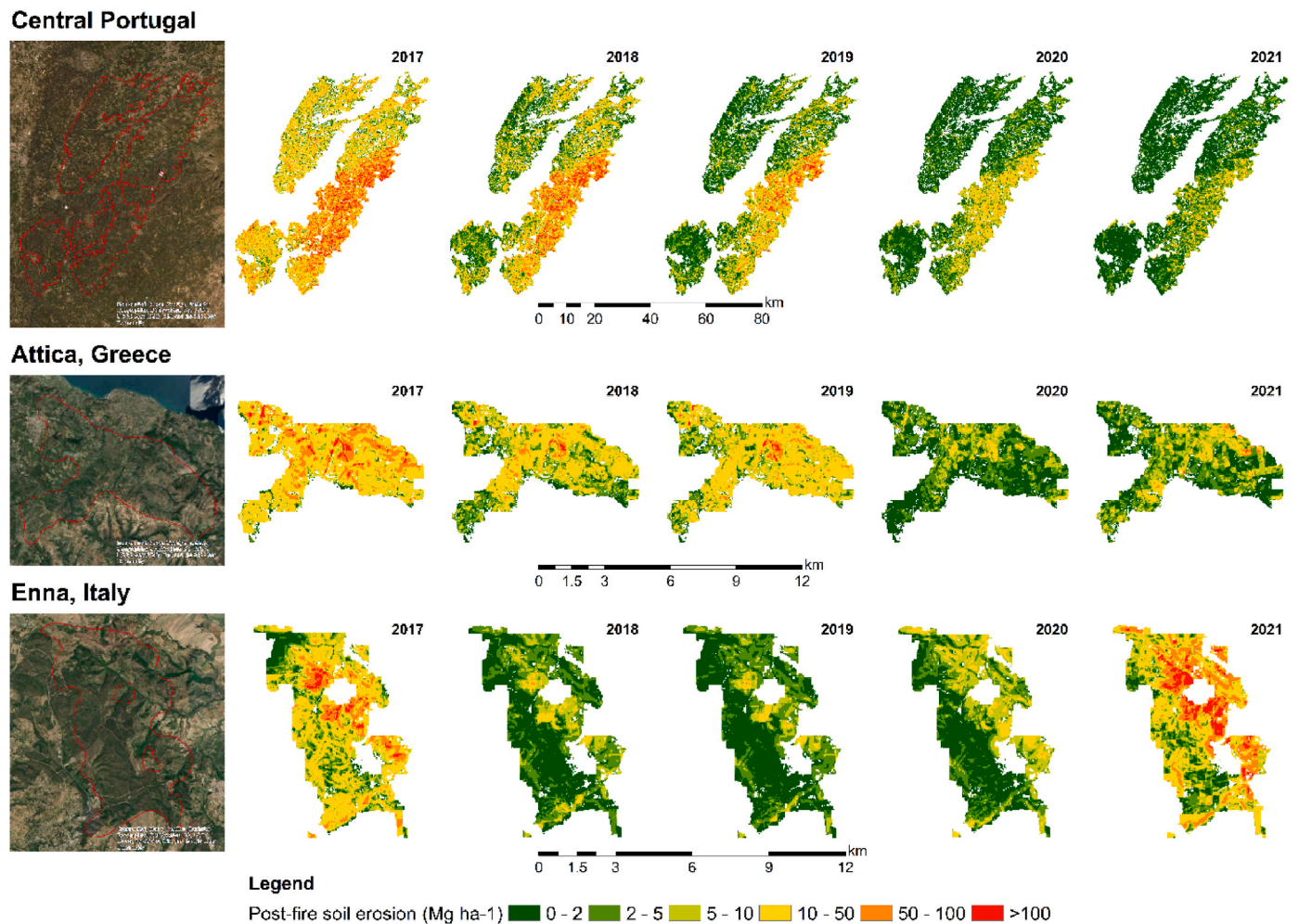


**Fig. 4.** Post-fire soil erosion estimations averaged by 25 × 25 km grid for EU and UK, for the first year following the 2017 wildfires, and location of the reference sites Pedrógão Grande for Portugal, Attica for Greece, and Enna for Italy.



RCOVER classes distribution (%)				
Year	No Recovery	Ongoing Recovery	Near Recovery	Recovered
2017	41.6	36.8	16.1	5.6
2018	4.9	34.4	46.2	14.6
2019	1.4	18.0	60.6	20.0
2020	0.1	4.5	67.6	27.8
2021	0.7	6.0	39.2	54.0

**Fig. 5.** Overlapping FCOVER pixel distribution for: a) post-fire recovery period (2017–2021), and b) pre-fire (2014–2016) and 5th post-fire (2021) year, and c) RCOVER indices distribution for the post-fire periods according to stipulated recovery classes (see 4.4).



**Fig. 6.** Fire perimeter and soil erosion estimates for 2017–2021, for a) Central Portugal, b) Attica, and c) Enna burned areas (see 4.6). Note: General location of reference sites depicted in Fig. 4.

followed such as logging and removal of the dead trees.

For the Enna burned area ( $2.3 \times 10^3$  ha, Fig. 6c), soil erosion estimations rounded the  $0.5 \times 10^6$  Mg of soil loss for the first post fire year, resulting in lower soil loss rates when compared to the previous examples ( $23.3 \text{ Mg ha}^{-1}$ ). In this case, and despite the fast recovery of the burned area for the two years that followed the wildfire ( $\text{RCOVER} \geq 1.0$  2019 = 45%), vegetation cover was highly affected by another wildfire occurring in the summer of 2021 ( $\text{RCOVER} \geq 1.0$  2021 = 2%).

These three examples evidenced a slow recovery of the ground cover during the study period, highlighting additional factors affecting the vegetation recovery, such as post-fire operations or additional wildfires, besides the usual conditionants such as climate dynamics, and lack of mitigation and restoration actions. Should be noted that post-fire operations were not systematically quantified thought the study period and AOI, due to the difficulties in assessing the timing of these interventions as also their nature (e.g. logging, clearcutting, mulching).

## 4. Discussion

### 4.1. Uncertainties

The main sources of uncertainties identified in this assessment can be represented by two groups: i) the parametric uncertainties, mostly associated to individual model inputs, and ii) the ones resulting from the modelling approach, associated to methodological decisions and structural model uncertainties:

- Parametric uncertainties – The current assessment is highly dependent on the quality of vegetation indices for the estimation of soil erosion and post-fire recovery assessment, possibly leaving out the recovery of soil properties (Basso et al., 2022). C factor estimations relies on the capacity of remotely-sensed data to capture burn severity heterogeneity and ground cover recovery. Differences between local and remotely-sensed burn severity classification have been highlighted previously (Sobrino et al., 2019; Robichaud et al., 2007), but is still the best approach for this scale. Post-fire mitigation schemes might not be totally represented by treatment efficiency within P factor, due to the likely increase of soil cover following its application.
- Model approach – Model validation is limited to the field data availability and data comparability across scales (Parente et al., 2022; Lopes et al., 2021). Local variability plays a significant role in the post-fire soil erosion rates, such as topography, rainfall dynamics, vegetation recovery, and local burn severity. Model estimations at this scale are frequently limited to the coarse spatial and temporal resolution of model inputs, such as the use of long term average rainfall and annual time-step (Lopes et al., 2021). For that same reason, it was not possible to include post-fire land management operations for individual burned areas.

An extended assessment of the uncertainties associated to this study can be found in the (Supplementary 2).

Future work should also address such uncertainties, namely in the

collection of post-fire soil erosion data for a more robust model calibration and validation, especially in MS such as Italy or Greece that present significant wildfire occurrence. Additionally, a better understanding how the burn severity assessed in the field is comparable with the remote sensed data (Parks et al., 2014) could reduce modelling uncertainties and model validation. Moreover, the uncertainties derived from spatial scale (catchment or plot vs. pixel), from climate variability (extreme vs. average), and from post-fire management operations are present in this assessment, and future studies will be necessary to refine the current results.

#### 4.2. Sustainability of the EU fire-affected soils

As already evidenced by other studies (Boschetti et al., 2021), wildfires consist in an environmental problem with a global dimension, and at EU scale wildfire occurrence is well documented (San-Miguel-Ayanz et al., 2012). However, the estimation of the indirect impacts of wildfires, such as increased soil erosion at wider scales, are still lacking (Parente et al., 2022). Therefore, results suggesting an immediate increase in 2% of soil erosion (19.4 million Mg) on the first post-fire year, and a 1% annual increase over a 5-years period (44.0 million Mg) despite concerning only 0.2% of the area, are very relevant if the EU wants to meet the Sustainable Development Goals commitments. In addition, several points of concern should be raised:

- Five years after the wildfires a significant portion of the burned area (46%) still evidences a lower vegetation cover when compared to pre-fire, highlighting the long-lasting impacts of wildfires (Moody et al., 2013; Shakesby, 2011), and a low recovery rate. Either because the post-fire conditions (e.g. burn severity, weather conditions) did not allow a better recovery performance, or because the post-fire actions taken were not efficient enough, but also due to additional disturbance caused by post-fire logging activities or subsequent wildfires, which likely led to an underestimation of the total soil erosion;
- Despite some uncertainties, the fact that the burned area extent for 2017 stands below the 10-years burned area average, and this assessment concerns only forest, shrubland, and grassland, representing 51% of the total, the soil losses might be largely underestimated. This fact is specifically relevant when the effects of wildfires are combined with agricultural soil management operations such as tillage, which likely increase soil erosion rates (Vieira et al., 2016; Shakesby et al., 2002), but also because this study didn't consider the cumulative effect of additional wildfires occurring beyond 2017 (such as the exceptional wildfires occurring in Sweden in 2018, Greece in 2021, or Spain 2022);
- The fact that fire-prone areas are projected to expand in Europe combined with more extreme rainfall events (Bednar-Friedl et al., 2022), will likely increase these figures on-site, but also extend these impacts to other ecosystems such water bodies, or human-infrastructures downstream the burned areas (Basso et al., 2022; Robinne et al., 2021);
- The immediate implementation of mitigation measures has a high potential to prevent wildfires impacts, with results suggesting a 63–77% reduction of soil erosion in the first post-fire year alone. However, mitigation measures remain largely unapplied in the EU-space (Girona-García et al., 2021).

This work highlights limited actions towards the prevention of additional impacts from wildfires within EU soils. The potential reasons are either, that few actions were taken, or such actions did not result in a significant recovery increase. Highlighting a lack of perception of the ecosystems services provided by these soils, or reduced investment in the recovery of burned area (Lopes et al., 2022) from the decision-makers. This aspect is particularly relevant not only when preventing additional soil losses *per se*, but also when the transport of

sediments (soil and ash) from recently burned areas affect important downstream values-at-risk such as water uptake points (Basso et al., 2021, 2022; Neris et al., 2021; Robinne et al., 2021; Nunes et al., 2018b), or trigger destructive debris flows events that may damage infrastructures downstream of the burned area (Santi and Rengers, 2020; Rengers et al., 2016), which likely result in additional economic burden. Moreover, and as previously emphasised, wildfire occurrence is not exclusive from the European territory, and corresponds to a global environmental problem (Boschetti et al., 2021), therefore the application of such methodology should also be considered at Global scale.

#### 4.3. Post-fire management

Given the assessment performed in this study, together with the future climate change impact, it is important to highlight that further attention should be given to the recovery of burned European soils. Despite the fact that most of the affected areas from this study are considered fire-prone regions, such as the Mediterranean basin (Shakesby, 2011), other studies suggested that fire prone regions within the EU will likely increase (Bednar-Friedl et al., 2022). Being therefore essential, to implement actions to limit current and future wildfires, but also post-fire impacts that might follow.

The application of post-fire mitigation measures should be the norm. The results from this study reveal a significant potential reduction on soil erosion rates following the application of the most efficient mitigation treatments available, i.e. the cover treatments (Girona-García et al., 2021). Notwithstanding, it should be highlighted that the assessment of post-fire mitigation needs and measures should be done at local scale, in which higher priority should be given to areas burned at higher severities, but also if the burned area is located upstream to important values-at-risk (e.g. reservoirs, water uptake points, bridges). For that reason, the usage of spatially explicit modelling approach as the one described in this study can be of much use for decision-makers. Scenarios as the ones presented in this modelling exercise clearly, can help identifying the areas with higher risk, but also determine the expected effectiveness of the mitigation treatment, and therefore make a more efficient use of funds and resources to tackle post-fire impacts (Parente et al., 2022).

Such suggestions are in line with the EU Soil Strategy (European Commission, 2021a), the EU Forest Strategy (European Commission, 2021b), and the EU Biodiversity Strategy (European Commission, 2021c) which aim to restore and improve the EU soils and the ecosystem services provided by them, but also improving climate adaptation and forest resilience, among other ambitions. Ultimately, such strategies will set the scene for a change in the EU soil health, and support the necessary restoration efforts to tackle long term land degradation caused by multiple threats, including wildfires (Lopes et al., 2022).

#### 4.4. Data availability

The maps for the post-fire soil erosion following the 2017 wildfires will be available in the European Soil Data Centre (ESDAC) (Panagos et al., 2012).

### 5. Conclusions

This is the first study that assesses soil erosion in post-fire conditions at the European scale following the 2017 wildfires. By adapting the RUSLE model to consider post-fire impacts in function of burn severity, and vegetation recovery for forests, grasslands, and shrubland, the Pan-European Soil Erosion assessment was updated with an additional source of soil erosion. Moreover, this work also assesses the current post-fire erosion and vegetation recovery dynamics, and the potential mitigation of such impacts if appropriate forest management actions take place. The principal findings of this modelling exercise with respect to the 2017 EU burned area are as follows:

- a sharp increase in the soil erosion estimations was found after wildfire (11.8-fold), representing an addition of 2% (19.4 million Mg) to the reference soil erosion for EU, for the first post-fire year alone, despite represents only 0.2% of the area;
- a full recovery of the soil erosion rates to the background levels was not observed within the study period, as also evidenced by the 46% of the burned area with vegetation cover lower than the pre-fire;
- post-fire natural recovery was affected by post-fire land management operations, and high fire recurrence;
- post-fire mitigation has the potential to reduce post-fire soil erosion impacts by 63–77% in the first year, and allow a reduction in soil erosion to the background levels by the 4th post fire year.

The results of this assessment evidence that not sufficient attention has been given to the mitigation of wildfire impacts given the importance of forest soils for the ecosystems services provision. Moreover, the new climate demands evidence the urgent improvement of EU's forest resilience and adaptation to climate change, in order also to halt land degradation in forest soils.

### CRedit author statement

Vieira DCS: Conceptualization; Data curation; Formal analysis; Investigation; Methodology; Validation; Visualization; Writing – original draft; Writing – review & editing. Borrelli P: Conceptualization; Investigation; Methodology; Visualization; Writing – original draft; Writing – review & editing. Jahanianfard D: Data curation; Formal analysis; Methodology; Writing – original draft; Writing – review & editing. Benali A: Data curation; Formal analysis; Methodology; Writing – original draft; Writing – review & editing. Scarpa S: Data curation; Formal analysis; Resources; Validation; Visualization. Panagos P: Conceptualization; Data curation; Funding acquisition; Investigation; Methodology; Project administration; Resources; Writing – original draft; Writing – review & editing.

### Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

### Data availability

Data will be made available on request.

### Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.envres.2022.114936>.

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