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Soil erosion modelling of burned and mulched soils following a Mediterranean forest wildfire

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16 **Soil erosion modelling of burned and mulched soils following a Mediterranean forest**  
17 **wildfire**

18  
19 Erosion prediction by WEPP, MMF and USLE-M models in burned and mulched forests

20  
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33  
34 **ABSTRACT**

35 Soil erosion modelling applied to burned forests in different global regions can be unreliable due  
36 to a lack of verification data. Here, we evaluated the following three erosion models: (1) Water  
37 Erosion Prediction Project (WEPP), (2) Morgan-Morgan-Finney (MMF) and (3) Universal Soil  
38 Loss Equation-Modified (USLE-M). Using field plots that were either untreated or mulched with  
39 straw, this study involved observations of soil loss at the event scale at a burned pine forest in  
40 Central Eastern Spain. The erosion predictions of the three models were analysed for goodness-  
41 of-fit. Optimisation of the MMF model with a new procedure to estimate the C-factor resulted in  
42 a satisfactory erosion prediction capacity in burned plots with or without the mulching treatment.  
43 The WEPP model underestimated erosion in the unburned areas and largely overestimated the  
44 soil loss in burned areas. The accuracy of soil loss estimation by the USLE-M model was also  
45 poor. Calibration of the Curve Numbers and C-factors did not improve the USLE-M model  
46 estimation. Therefore, we conclude that an optimised MMF model was the most accurate way to  
47 estimate soil loss and recommend this approach for in Mediterranean burned forests with or  
48 without post-fire mulching. This study gives land managers insight about the choice of the most  
49 suitable model for erosion predictions in burned forests.

50

51 **KEYWORDS:** soil loss; post-fire management; calibration; erosion model; plot scale; event  
52 scale.

53

## 54 1. INTRODUCTION

55

56 Models are essential tools to simulate the complex processes in disturbed soils, and to predict the  
57 soil response to the hydrological input in a cost-effective and time-efficient way (Filianoti et al.,  
58 2020). Many models are capable of predicting soil erosion in different climatic and  
59 geomorphological conditions (Bezák et al., 2021; Borrelli et al., 2021). In comparison to the  
60 most complex physically-based algorithms, empirical models are easy and quick to apply at  
61 specific sites, and their prediction capability is often satisfactory (Aksoy & Kavvas, 2005; Lucas-  
62 Borja, Bombino, et al., 2020). Many empirical models, such as the USLE (Universal Soil Loss  
63 Equation) equation, have been used as erosion component of many catchment-scale erosion  
64 models (e.g., AnnAGNPS and SWAT models). In contrast, where input parameters can be  
65 directly measured in the field or are available for a specific environment, the physically-based  
66 models theoretically provide more accurate predictions of erosion (Zema, Lucas-Borja, et al.,  
67 2020; Zema, Nunes, et al., 2020).

68 Wildfire is a common natural disturbance in Mediterranean ecosystems, and usually enhances  
69 soil erosion, which may increase ecosystem degradation to unsustainable rates. To limit soil  
70 degradation in the Mediterranean forests, a proper control of erosion is necessary. On this regard,  
71 erosion models, commonly used in agricultural areas (Borrelli et al., 2018; Panagos &  
72 Katsoyiannis, 2019), have been also extended to burned areas (Lopes et al., 2021; Vieira et al.,  
73 2014). According to Lopes et al. (2021), published literature on post-fire erosion modelling is not  
74 homogeneously distributed worldwide. While post-fire prediction models are commonly used in  
75 the United States, research is still far from being exhaustive in other regions (Hosseini et al.,  
76 2018). In the Mediterranean burned areas, where the hydrological processes are site-specific  
77 (Shakesby, 2011; Wagenbrenner et al., 2021), the available erosion models may find limitations  
78 in their applicability, especially when the soils are subjected to post-fire management. Therefore,  
79 these models require localised optimisation, since these hydrological tools have been developed  
80 in other environmental contexts and are not designed to be applied specifically in burned areas  
81 (Vieira et al., 2018a).

82 Erosion models include simple empirical models, such as the USLE-family models, to semi-  
83 empirical models (e.g., the Morgan–Morgan–Finney model, MMF), physically-based models

84 (for instance, the Water Erosion Prediction Project, WEPP) or artificial neural networks (Aksoy  
85 & Kavvas, 2005; Merritt et al., 2003; Zema, Lucas-Borja, et al., 2020; Zema, Nunes, et al.,  
86 2020). Regarding the USLE-family models, only the RUSLE version has been used in wildfire-  
87 affected soils (e.g., Fernández et al., 2010; Karamesouti et al., 2016; Larsen & MacDonald,  
88 2007; Vieira et al., 2018b). To the authors' best knowledge, the event-scale USLE-family model  
89 (USLE-M) has been applied only by Carrà et al. (2021) in Southern Italy treated with prescribed  
90 fire and fern mulching. Erosion predictions using WEPP were carried out by (Covert et al., 2005;  
91 Fernández & Vega, 2018; Larsen & MacDonald, 2007; Soto & Díaz-Fierros, 1998) in  
92 Mediterranean forests burned by wildfires. The MMF model has been verified in burned areas  
93 with or without post-fire treatments under different Mediterranean climates in North-Western  
94 and Central-Eastern Spain, and in Central Portugal (Fernández et al., 2010; Vieira et al., 2014,  
95 2018b; Zema, Nunes, et al., 2020). The latter authors suggested an adaptation of this model to  
96 uses on the event scale in burned forests, mulched or untreated.

97 This study evaluated the prediction capability of the USLE-M (coupled to the Curve Number  
98 method to estimate the runoff coefficient), MMF and WEPP models in a Mediterranean forest  
99 burned by a wildfire and then treated with soil mulching in comparison to unburned areas. The  
100 investigation was carried out at the plot and event scales throughout one year in a pine stand of  
101 Castilla-La Mancha (Central Eastern Spain). This study is the first application of the USLE-M  
102 and WEPP models to wildfire-affected soils subjected to post-fire mulching in Mediterranean  
103 forests as well as applying the MMF model in a Mediterranean pine forest on untreated or  
104 mulched soils. The erosion prediction capacity of the latter model was successfully evaluated  
105 (Zema, Nunes, et al., 2020), but without a further validation in other sites with similar  
106 environmental characteristics. Moreover, a new procedure to estimate the C-factor based on the  
107 ground cover as input parameter for the MMF model was proposed and evaluated in this  
108 modelling exercise. This study aims at providing insight about the choice of the most suitable  
109 model for erosion predictions in burned forests to land managers and hydrologists.

110

## 111 **2. MATERIALS AND METHODS**

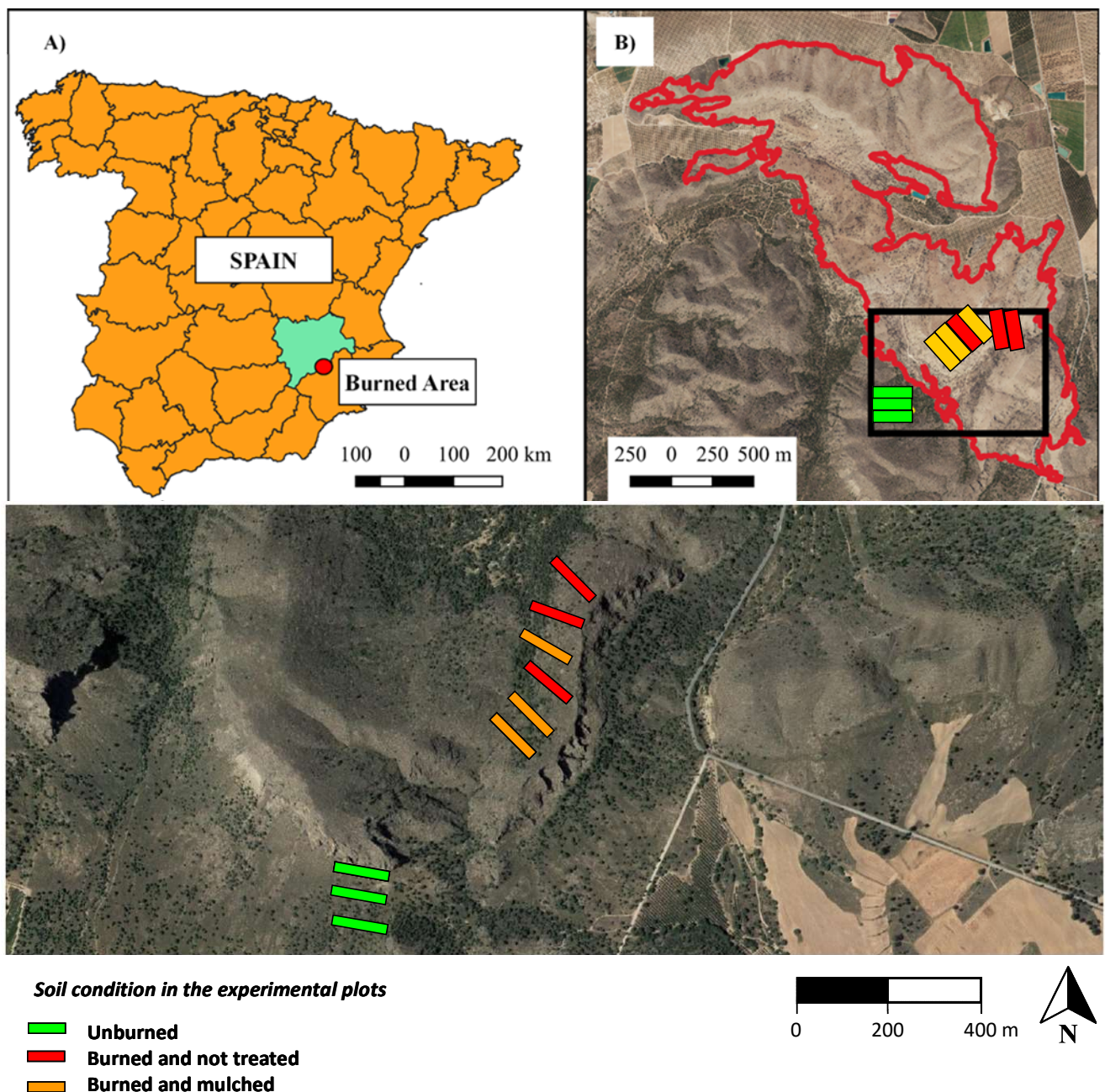
112

### 113 **2.1. Study area**

114

115 The study area is located between the municipalities of Liétor and Agramón (geographical  
116 coordinates 38°25'19"N, -1°38'15"E, province of Albacete, Castilla-La Mancha, Central Eastern  
117 Spain) (Figure 1a). The elevation ranges between 400 and 700 m above the mean seal level.

118 The area is framed in a semi-arid climate Mediterranean, BSk according to the Köppen-Geiger  
119 classification (Kottek et al., 2006), where the average annual precipitation and temperature are  
120 320 mm and 16 °C, respectively. In the first year after the fire, a total precipitation of 400 mm  
121 was recorded. The parental materials that make up this area are limestones, such as dolomites,  
122 marls, and clay. The soils are shallow (depth lower than 0.3 m), with low organic matter content,  
123 the main types being Inceptisols and Aridisols (USDA, 1999). The soil texture of the study area  
124 is presented in Table 1. The vegetation in the area consists of a *Pinus halepensis* forest with a  
125 shrub and herbaceous companion layers dominated by *Macrochloa tenacissima* and *Salvia*  
126 *Rosmarinus*, and other minor species, such as *Pistacia lentiscus*, *Quercus coccifera*, *Rhamnus*  
127 *lycioides*, *Thymus vulgaris* and *Cistus clusii*.



128 Figure 1 - Geographical location of the study area (A) and of the burned area within the wildfire  
129

130 perimeter (B) and aerial map of the experimental plots (C) (Agramón, Castilla-La Mancha,  
131 Spain).

132

Table 1 – Soil properties and surface cover of the experimental plots (Agramón, Castilla-La Mancha, Spain) after the wildfire in 2020.

Soil condition	Plot #	Topographic characteristics		Soil properties				
		Slope (%)	Aspect	Sand content (%)	Silt content (%)	Clay content (%)	Organic matter content (%)	Soil cover (%)
Unburned	1	47.8	East	28.7	58.8	14.8	2.40	75.6
	2	47.1	East	24.3	58.1	14.3	2.45	74.3
	3	46.9	East	25.9	61.2	13.9	2.10	75.0
Burned and not treated	1	40.8	South-East	31.5	55.7	12.2	2.01	30.2
	2	41.4	South	33.6	54.2	13.9	2.20	28.6
	3	41.5	South-East	29.9	56.5	12.2	2.17	31.2
Burned and mulched	1	40.1	South-East	33.4	49.2	19.2	2.90	44.1
	2	39.8	South-East	29.1	53.2	17.8	2.86	42.3
	3	40.9	South-East	28.1	51.6	18.6	2.88	39.7

## 2.2. Experimental design

On 27<sup>th</sup> July 2020, a wildfire burned approximately 275 ha of forestland in the study area. The soil burn severity, estimated according to Vega et al. (2013), was high. The tree mortality in the study area was 100%.

After the wildfire, nine plots, each one of 0.5 ha (width of 25 m and length of 200 m), were equipped to collect the sediments (Figure 1). The soil slope was between 39.8% and 47.8%, while the aspect south, east or south-east. The soil texture was homogenous, with contents of sand, silt and clay in the range 24.3-33.6%, 49.2-61.2% and 12.2-19.2%, and of organic matter between 2.01% and 2.90% (Table 1). Of the nine plots, three were identified in an unburned area that was adjacent to the burned forest. Other six plots were selected in the burned area, of which three were not treated and other three were subjected to post-fire soil mulching one month after the fire. The mulch material was barley straw, which was manually spread on the plots at a rate of 0.25 kg/m<sup>2</sup> (dry weight) and thickness of 3 cm (Lucas-Borja et al., 2018; Lucas-Borja et al., 2020). This mulch rate was based on a previous study by (Vega et al., 2014), who proposed this dose to achieve a soil cover of 80% for burned plots installed in Northern Spain. The soil cover varied from 28.6% (burned and untreated plots) to 75.6% (burned and mulched plots) (Table 1). At the outlet of each plot, sediment traps were installed since 1<sup>st</sup> September 2020 until 31<sup>st</sup> December 2021 (16 months) and soil loss was measured in this observation period. The accumulated sediment at each sediment trap was removed after each rainfall event, weighted in the field, and oven-dried in laboratory to calculate dry weight. A weather station, located in Liétor (geographical coordinates 38°32'27"N; 1°57'17"W), about 12 km far from the experimental area, provided the precipitation and temperature records. Rainfall depths were available at 15-min intervals, while the air temperatures were on a daily scale.

## 2.3. Experimental observations of soil erosion

Throughout the observation period, 13 “erosive” events (rainfall >13 mm) according to Wischmeier & Smith (1978) were monitored. Of these events, only seven (all with rainfall depth over 30 mm) produced erosion, while were recorded and used for modelling purposes. In more detail, the rainfall depth was in the range 43.3 - 115.2 mm, the maximum intensity in 30 minutes ( $I_{30}$ ) was between 9.2 and 64.8 mm/h (Table 2 and Figure 2), and the maximum rainfall erosivity ( $EI_{30}$ ) was 298 MJ mm/ha h (Figure 2). In the unburned plots, the soil loss was between 0.0005 to 0.0009 tons/ha. The soil loss measured in the burned and not treated plots was in the range

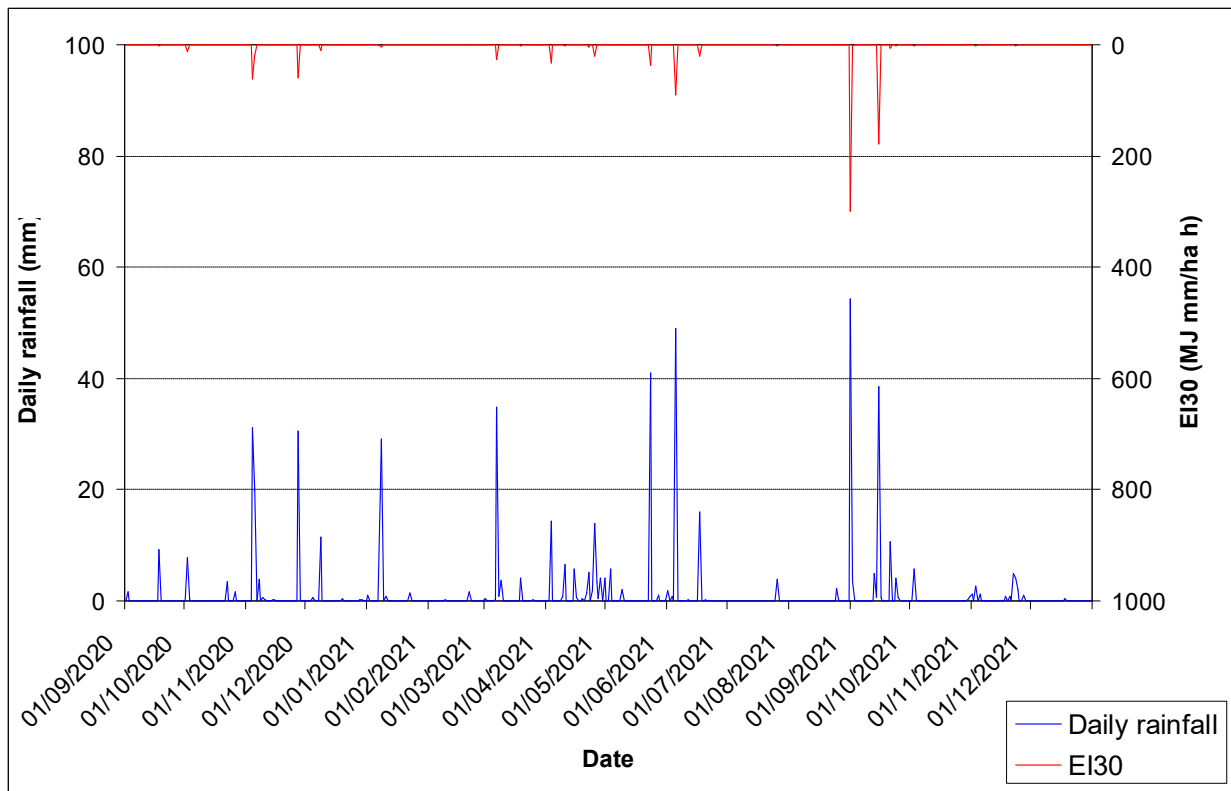
169 0.96 to 1.63 tons/ha, while the values measured in the burned and mulched plots varied from  
170 0.16 to 0.29 kg/m<sup>2</sup> (Table 2).

171  
172 Table 2 – Rainfall characteristics and soil loss observations at the experimental site (Agramón,  
173 Castilla-La Mancha, Spain).

174

Event date	Rainfall		Observed soil loss (tons/ha)		
	depth (mm)	max 30-min intensity (mm/h)	Unburned soil	Burned and not treated soil	Burned and mulched soil
10 Nov 2020	79.8	33.0	6.85 x 10 <sup>-4</sup>	1.227	0.165
16 Dec 2020	43.3	17.6	6.38 x 10 <sup>-4</sup>	1.600	0.184
22 Mar 2021	104.9	9.2	7.91 x 10 <sup>-4</sup>	1.463	0.198
4 Jun 2021	115.2	17.2	6.13 x 10 <sup>-4</sup>	1.631	0.228
5 Jul 2021	65.4	44.0	8.93 x 10 <sup>-4</sup>	1.399	0.193
14 Sep 2021	64.0	64.8	5.41 x 10 <sup>-4</sup>	1.338	0.292
10 Oct 2021	60.8	38.6	7.31 x 10 <sup>-4</sup>	0.959	0.176

175



176  
 177 Figure 2 – Rainfall depth and erosivity ( $EI_{30}$ ) at the experimental site (Agramón, Castilla-La  
 178 Mancha, Spain).

179  
 180  
 181 The hydrological dataset used for model is made of a sample of seven observations of soil loss,  
 182 which, however, allows the evaluation of the model’s prediction capacity for two reasons. First,  
 183 wildfire opens in the burnt area a so-called “window of disturbance” (Prosser & Williams, 1998;  
 184 Zavala et al., 2014), when the soil is bare due to the vegetation burning and many of its physico-  
 185 chemical properties undergo noticeable changes (Moody et al., 2013; Zema, 2021). Both effects  
 186 result in a strong alteration in the hydrological response of severely-burned soils, which however  
 187 is only temporary. Although the duration of this window of disturbance may be variable  
 188 depending on soil, weather and fire characteristics, the pre-fire conditions generally recover after  
 189 one year, which is the duration of the observation period of this study. The measurements of soil  
 190 erosion in the experimental area were made in this window of disturbance. Second, in the semi-  
 191 arid environments (such as the study area), annual erosion is substantially due to few but intense  
 192 rainstorms (the so-called “erosive events”, according to Wischmeier & Smith (1978), that is  
 193 events with rainfall depth over 13 mm). The events of our dataset are all those events that  
 194 overcome this threshold and produced erosion, while the other smaller events were not  
 195 considered.

196

## 197 **2.4. Short description of the WEPP, MMF, and USLE-M models**

198

199 The "Water Erosion Prediction Project" (WEPP, Flanagan & Nearing, 1995) is a physically-  
200 based, distributed parameters, continuous simulation model, which was developed by USDA-  
201 Natural Resources Conservation Service in late '1980s. The model is used to predict the spatial  
202 and temporal distribution of net soil loss and deposition for a wide range of time periods and  
203 spatial scales. The WEPP hillslope version (Foster & Lane, 1987) was used in this study to  
204 predict soil erosion along a single slope profile. More details about the equations governing the  
205 erosion simulations by the WEPP model and the input parameters are reported in Flanagan &  
206 Nearing (1995) and Foster & Lane (1987).

207 The "Morgan-Morgan-Finney" model (MMF, Morgan, 2001; Morgan et al., 1984) was  
208 developed in early '1980s. The model was revised in 2001 by the same authors, in order to  
209 improve the accuracy of erosion predictions and to propose guidelines for the optimal choice of  
210 the input parameters. The equations for calculating the hydrological variables are reported in the  
211 original papers by Morgan (2001) and Morgan et al. (1984).

212 The "Universal Soil Loss Equation" (USLE) was first developed in USA (Wischmeier & Smith,  
213 1978), in order to predict soil loss in small agricultural catchments. The mean annual soil loss is  
214 calculated as the product of six input parameters (the so-called "USLE-factors"), linked to  
215 climate, soil cover and properties, topography, and human activities (R, K, L, S, C and P). In  
216 1998, Kinnell & Risse (1998) proposed the USLE-M model (USLE modified), which assumes  
217 that the sediment concentration in the runoff is governed by the event rainfall erosivity index ( $R_e$ ,  
218 Wischmeier & Smith, 1978) per unit quantity of rain ( $P_e$ , mm). The equations to calculate the  
219 soil loss  $Y$  and the input factors are reported in the study by Kinnell & Risse (1998).

220

## 221 **2.5. Model implementation**

222

### 223 *2.5.1. General information*

224

225 The climate data needed by the models was collected at the weather station of Liétor. Three  
226 treatments were simulated: (i) unburned soil (control); (ii) burned and not treated soil; and (iii)  
227 burned and mulched soil.

228

### 229 2.5.2. *WEPP model*

230

231 The climate data needed as input by WEPP were the storm depth (mm), duration (h) and  
232 maximum intensity (mm/h), and the percent duration to peak intensity. These values were  
233 derived from rainfall records at the gauging station of Liétor for each modelled event.

234 A concave profile with three slopes was assumed for each plot. The value of the soil albedo  
235 parameter was calculated by Baumer's equation and set to 0.23 (Flanagan & Nearing, 1995).

236 According to the guidelines in the WEPP manual, an initial saturation level of 75% was assumed  
237 (Table 1SM).

238 Three conditions were input according to the WEPP management files into the software  
239 interface: (i) "shrub perennial" for the unburned plots; (ii) "25% ground cover - high severity"  
240 for the burned and not treated plots; (iii) "fallow initial condition - wheat residues" for the burned  
241 and mulched soils. For the three management conditions, some input parameters of the model's  
242 default database were updated to the experimental conditions.

243

### 244 2.5.3. *MMF model*

245

246 MMF was run at the event scale and parameterised using the default input data (hereafter  
247 "default model") or adjusted values for post-fire conditions ("calibrated model") and for each  
248 event. For runs using the default model, the values of the input parameters suggested in the  
249 original studies of Morgan et al. (1984) and Morgan (2001) were adopted. For the erosion  
250 predictions using the calibrated model, MMF was run simulating the post-fire conditions,  
251 according to the adaptations suggested by Fernández et al. (2010), Vieira et al. (2014) and Zema,  
252 Nunes, et al. (2020), where more details can be found. To summarise, the input parameters  
253 needed by MMF were measured in-field or derived from the guidelines of Morgan (2001),  
254 Morgan et al. (1984) and Morgan & Duzant (2008) with the corrections according to Vieira et al.  
255 (2014) and Nunes et al. (2016), or estimated from the literature (Doorenbos & Kassam, 1979;  
256 Fernández et al., 2010; Vieira et al., 2014; Wischmeier & Smith, 1978; Zema, Nunes, et al.,  
257 2020).

258

#### 259 *2.5.4. USLE-M model*

260

261 The USLE-M model was initially run with default parameters, estimated from the model  
262 guidelines. In this study, the runoff coefficient of the USLE-M equation was calculated as the  
263 ratio between the runoff volume, and the rainfall depth (both in mm), while the erosivity factor  
264 was estimated as the product of the kinetic energy and the maximum intensity in 30 minutes for  
265 each rainfall, following

266 Due to the lack of measured runoff volumes, needed by the USLE-M model, the runoff volume  
267 was estimated by the Soil Conservation Service-Curve Number (SCS-CN) method (SCS, 1985)  
268 (as explained below). In more detail, the rainfall depth of each modelled storm event was  
269 estimated by aggregating the sub-hourly records collected at the rain gauging station. The  
270 Antecedent Moisture Content (AMC) was estimated by aggregating the 5-day rainfall depths  
271 prior to each event. The soil hydrological group was identified as “B” (“soils having a  
272 moderate infiltration rate when thoroughly wet”). The default values of CN were assumed  
273 following the standard procedure by the USDA Soil Conservation Service (SCS, 1985).

274 The K-factor was calculated according to the procedure proposed by Kinnel et al. (1998), on its  
275 turn based on the nomograph of Wischmeier & Smith (1978). The C-factor was calculated using  
276 the empirical equation based on canopy cover and aboveground biomass proposed by Bombino  
277 et al. (2002). The P-factor was always set to one.

278

## 279 **2.6. Model calibration**

280

### 281 *2.6.1. WEPP model*

282

283 Since WEPP is a physically-based model with input parameters that should be measured in the  
284 field (Aksoy & Kavvas, 2005; Merritt et al., 2003), the model was not deliberately calibrated in  
285 this study. This choice also allows a model performance assessment in data-poor environments,  
286 that is in the case that no data for calibration/validation is available.

287

### 288 *2.6.2. MMF model*

289

290 The MMF model, commonly used for erosion predictions at the yearly or multi-yearly scale, was  
291 implemented at the event scale. Since Zema, Nunes, et al. (2020) have tested the model

292 prediction capacity after rainfall events in burned forest soils for Mediterranean conditions, we  
 293 adopted these adaptations for the MMF calibration were (Tables 3 and 2SM): (i) input of  
 294 seasonal values of the moisture content at field capacity (MS), corrected by changes in soil water  
 295 repellency (SWR) (except for unburned plots), ratio between the actual ( $E_t$ ) to potential ( $E_0$ )  
 296 evapo-transpiration ( $E_t/E_0$ ), and ground cover (GC), according to Vieira et al. (2014); (ii) model  
 297 running at the event scale rather to simulate seasonal or annual soil loss, since, in the  
 298 Mediterranean climate, soil erosion is mainly determined by few but intense rainfall events (e.g.,  
 299 Fortugno et al., 2017); (iii) setup of the MS input parameter to 0.28 for sandy loam soils  
 300 (Morgan, 2001) with the correction suggested by the “SM-SWR” modelling approach of Vieira  
 301 et al. (2014) and Nunes et al. (2016), to take into account the effect of soil water repellency on  
 302 soil wetting (from 0.8 for extreme repellency to 1.1 under wettable conditions (Vieira et al.,  
 303 2014); (iv) modification of effective hydrological depth (EHD) suggested by Hosseini et al.  
 304 (2018) and Vieira et al. (2014), considering two soil layers, of which the deeper layer was not or  
 305 scarcely influenced by the fire effects (50% of the original depth), while the topsoil was affected  
 306 by the high burn severity and the post-fire treatment.

307 The C-factor is one of the few empirical factors of the model, which, for this reason, was  
 308 calibrated. This factor was estimated differently from the previous study by Zema, Nunes, et al.  
 309 (2020), in order to take into account its seasonal variability, due to growth of the herbaceous  
 310 vegetation by regeneration in burned areas, and by seasonal natural cycle in unburned plots.  
 311 More specifically, the following equations were adopted for estimating the C-factor:

312

$$313 \quad C_{corr} = C \cdot FC \quad (1)$$

314

$$315 \quad FC = \frac{I}{\frac{GC - \overline{GC}}{\overline{GC}} + I} \quad (2)$$

316

317 where  $C_{corr}$  and  $C$  are the corrected and original C-factors, respectively,  $FC$  is the correction  
 318 factor,  $GC$  is the ground cover (in percent on the total plot area) at the time of the modelled  
 319 event, and  $\overline{GC}$  is the value of  $GC$  averaged throughout the observation period. Of course,  $FC$  is  
 320 undefined and thus can not be calculated, when the soil is totally bare (in this case  $GC = 0$ )  
 321 (Table 3 and 2SM).

322

323

324 Table 3 – Input parameters of the MMF model to predict soil loss at the experimental site  
 325 (Agramón, Castilla-La Mancha, Spain).

326

Factor	Soil condition					
	Min	Max	Min	Max	Min	Max
	<i>Unburned</i>		<i>Burned and Not Treated</i>		<i>Burned and Mulched</i>	
	Uncalibrated model					
<i>MS</i>	0.2	0.2	0.4	0.4	0.4	0.4
<i>EHD</i>	0.1	0.1	0.1	0.1	0.1	0.1
<i>A</i>	0	0	0	0	0.3	0.3
<i>Et/E<sub>0</sub></i>	1	1	0.1	0.1	0.9	0.9
<i>C</i>	0.003	0	1	1	0	0
<i>CC</i>	0.7	0.7	0	0	1	1
<i>GC</i>	0.4	0.6	0	0	1	1
<i>PH</i>	0	0	0	0	0.5	0.5
	Calibrated model					
<i>MS</i>	0.35	0.35	0.28	0.28	0.28	0.28
<i>EHD</i>	0.2	0.2	0.09	0.09	0.12	0.12
<i>A</i>	0	0	0	0	0.06	0.06
<i>E<sub>t</sub>/E<sub>0</sub></i>	0.80	0.70	0.05	0.05	0.50	0.45
<i>C</i>	0.135	0.156	0.120	0.558	0.029	0.034
<i>CC</i>	0.7	0.7	0.	0	0.05	0.05
<i>GC</i>	0.45	0.39	0.19	0.07	0.56	0.47
<i>PH</i>	0.5	0.5	0	0	0.6	0.6

327 Notes: MS = moisture content at field capacity; EHD = effective hydrological depth; A = vegetation cover; E<sub>t</sub>/E<sub>0</sub> =  
 328 ratio of actual and potential evapotranspiration; C = cover management factor; CC = percent canopy cover; GC =  
 329 ground cover; PH = plant height to the ground surface.

330

### 331 2.6.3. USLE-M model

332

333 For USLE-M calibration, the most sensitive input parameters (CN for the SCS-CN model and  
 334 the C-factor for the USLE-M) were chosen (Carra et al., 2021). First, constant CNs and C-factors  
 335 for all the modelled events were input, then these parameters were increased for the first two

336 rainfall events for the burned catchments, in order to simulate the variable hydrological response  
337 of soils throughout the observation period. This choice agrees with several studies (e.g., Carra et  
338 al., 2021; Cawson et al., 2012; Lucas-Borja, Bombino, et al., 2020; Vieira et al., 2015). To avoid  
339 separate calibration of two factors, the effects of the mulching practice were included in the C-  
340 factor, and the P-factor was set to one (Table 4 and 3SM).

341 For MMF and USLE-M models, the calibration was carried out manually by a trial-and-error  
342 procedure until the maximum coefficient of efficiency (see section 2.6) and the minimum error  
343 between the mean values of the observations and simulations of soil loss were achieved. Due to  
344 the lack of runoff observations, only the erosive sub-model of MMF was optimised. However,  
345 the MMF adjustments of the hydrological sub-model previously proposed by Zema, Nunes, et al.  
346 (2020), who found good runoff estimations after sub-model optimisation under the same  
347 environmental conditions, were embedded in the optimised model.

348 Table 4 – Input parameters of the USLE-M model to predict soil loss at the experimental site (Agramón, Castilla-La Mancha, Spain).

349

Model	Input parameter	Measuring unit	Soil conditions					
			Unburned		Burned and not treated		Burned and mulched	
			Default	Calibrated	Default	Calibrated	Default	Calibrated
SCS-CN	CN	-	39	7	90 (79)	13	65 (39)	8
USLE-M	C-factor	-	0.0001	0.0002	0.006	0.526	0.0002	0.085

350 Notes: CN = curve number; C-factor = cover management factor; the values related to the first two modelled events are reported in brackets.

351

## 352 **2.7. Model evaluation**

353

354 The erosion predictions by the three models were analysed for “goodness-of-fit” with the  
355 corresponding observations adopting qualitative and quantitative procedures. The qualitative  
356 approach consisted of the visual comparison of the observed and the corresponding soil loss in  
357 scatterplots. The quantitative procedure used the indicators that are commonly adopted in the  
358 literature (e.g., Willmott, 1982; Legates and McCabe, 1999; Loague and Green, 1991; Zema et  
359 al., 2017; 2018). More specifically, we used: (i) the main statistics (i.e., the maximum, minimum,  
360 mean and standard deviation of both the observed and simulated values); (ii) the coefficient of  
361 determination ( $r^2$ ); (iii) the coefficient of efficiency (NSE, Nash & Sutcliffe, 1970); (iv) the Root  
362 Mean Square Error (RMSE); and (v) the percent bias (PBIAS). The equations to calculate these  
363 indicators are reported in the studies by Krause et al. (2005), Moriasi et al. (2007), Van Liew &  
364 Garbrecht (2003) and Zema et al. (2012). The acceptance or optimal values are as follows (i)  $r^2$  is  
365 in the range 0 (no agreement between observed and predicted data) to 1 (perfect agreement),  
366 being acceptable when over 0.5 (Santhi et al., 2001; Van Liew et al., 2003; Vieira et al., 2018b);  
367 (ii) NSE varies between  $-\infty$  and 1, and the model accuracy is "good" if  $E \geq 0.75$ , "satisfactory" if  
368  $0.36 \leq E \leq 0.75$  and "unsatisfactory" if  $E \leq 0.36$  (Van Liew et al., 2003); (iii) RMSE is optimal  
369 when it approaches to 0 (Fernández et al., 2010), and the predictions are “good”, when RMSE is  
370 lower than half the observed standard deviation (Singh et al., 2005); (iv) PBIAS, which is also  
371 reported as "coefficient of residual mass", is positive, when a model underestimates the  
372 observation, and negative in the case of overestimation (Gupta et al., 1999); a model with  
373 CRM/PBIAS below 55% is considered fair for erosion predictions (Moriasi et al., 2007).

374

## 375 **3. RESULTS**

376

### 377 **3.1. WEPP model**

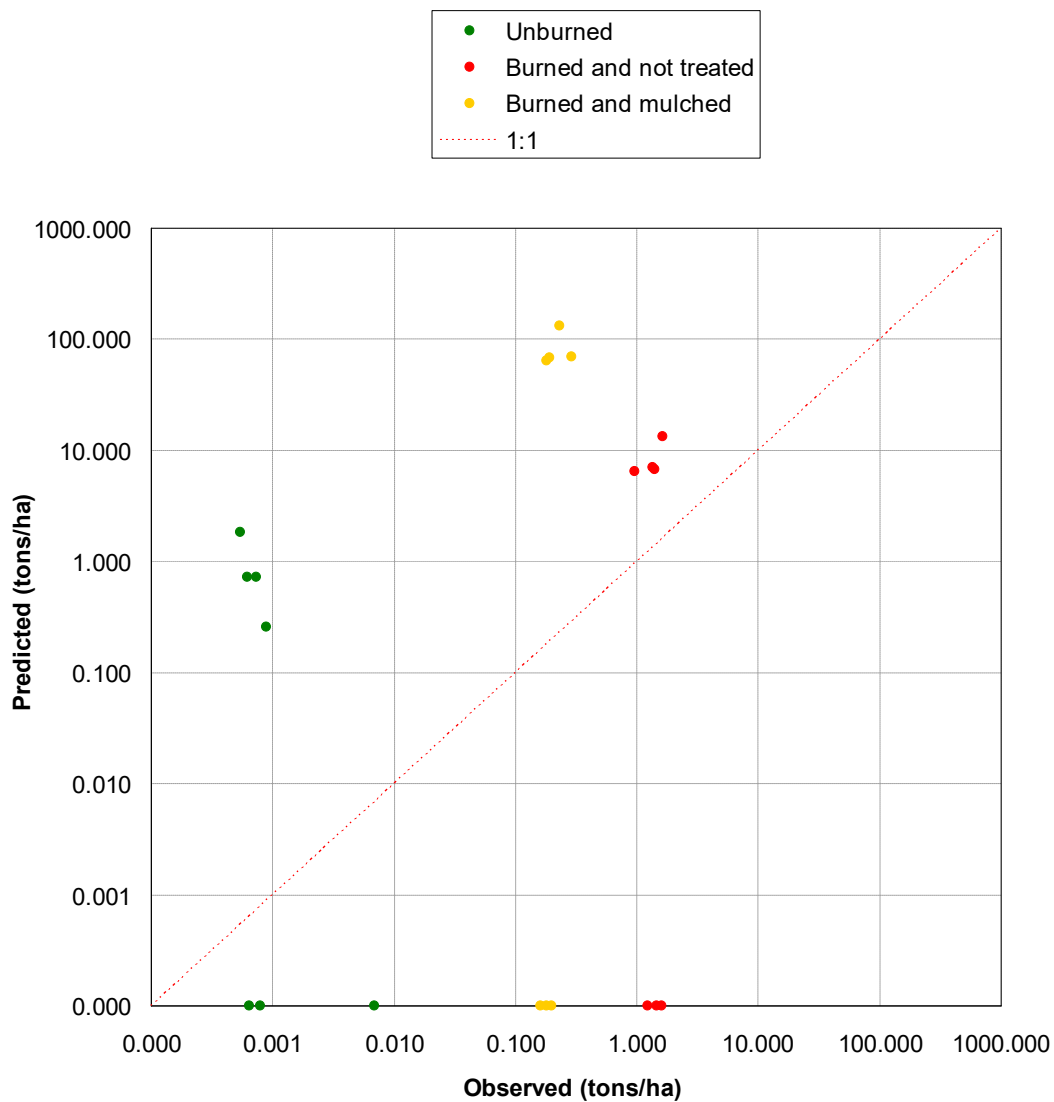
378

379 The WEPP model gave poor predictions of soil loss under all the modelled soil conditions, as  
380 shown by the large scattering of "observations vs. predictions" pairs around the line of perfect  
381 agreement (Figure 3). This low prediction accuracy is confirmed by the poor values of the  
382 evaluation indicators. In more detail, the difference between the mean observed and predicted  
383 soil loss was from 1237% (burned plots) to +23233% (burned and mulched soils). The  
384 coefficient of determination was very low (from 0.01 in burned and not treated plots to 0.35 in

385 unburned soils), and the NSE values were even negative ( $< -12815$ ). Moreover, while the soil  
386 loss was largely underestimated in all soil conditions (see the negative PBIAS). The RMSE was  
387 from 80% (unburned soils) to several orders of magnitudes (burned plots with or without  
388 mulching) higher compared to half the standard deviation of the observed values, and thus  
389 unsatisfactory (Table 5).

390

391



392

393 Figure 3 – Scatterplots of observed vs. predict soil loss using the WEPP model at the  
394 experimental site (Agramón, Castilla La Mancha, Spain).

395

396 Table 5 – Indexes used to evaluate the erosion prediction capacity of the WEPP, MMF and USLE-M models at the experimental site (Agramón,  
 397 Castilla-La Mancha, Spain).

398

Soil loss	Mean (tons/ha)	Minimum (tons/ha)	Maximum (tons/ha)	Standard deviation (tons/ha)	r <sup>2</sup>	E	RMSE (tons/ha)	PBIAS
	<b>WEPP model</b>							
<i>Unburned soil</i>								
Observed	0.001	0.001	0.001	0.000	0.35	-53764953	0.80	-722
Predicted	0.506	0.000	1.835	0.669				
<i>Burned and not treated soil</i>								
Observed	1.374	0.959	1.631	0.231	0.00	-12815	24.2	-12.4
Predicted	18.368	0.000	47.612	18.682				
<i>Burned and mulched soil</i>								
Observed	0.205	0.165	0.292	0.043	0.24	-2764390	66.7	-232
Predicted	47.882	0.000	132.530	50.364				
<b>Uncalibrated MMF model</b>								
<i>Unburned soil</i>								
Observed	0.001	0.001	0.001	0.000	0.00	-35648	0.00	-23.05
Predicted	0.017	0.003	0.040	0.014				
<i>Burned and not treated soil</i>								
Observed	1.374	0.959	1.631	0.231	0.16	-1922	0.94	-5.52

Predicted	8.964	2.516	18.931	6.060				
	<i>Burned and mulched soil</i>							
Observed	0.205	0.165	0.292	0.043	0.01	-26.18	0.02	1.00
Predicted	0.000	0.000	0.000	0.000				
	<b>Calibrated MMF model</b>							
	<i>Unburned soil</i>							
Observed	0.001	0.001	0.001	0.000	0.01	-84.6	0.00	-0.13
Predicted	0.017	0.003	0.040	0.014				
	<i>Burned and not treated soil</i>							
Observed	1.374	0.959	1.631	0.231	0.94	0.75	0.01	0.06
Predicted	8.964	2.516	18.931	6.060				
	<i>Burned and mulched soil</i>							
Observed	0.205	0.165	0.292	0.043	0.86	0.43	0.00	-0.13
Predicted	0.000	0.000	0.000	0.000				
	<b>Uncalibrated USLE-M model</b>							
	<i>Unburned soil</i>							
Observed	0.001	0.001	0.001	0.000	0.08	-27.4	0.00	0.80
Predicted	0.000	0.000	0.000	0.000				
	<i>Burned and not treated soil</i>							
Observed	1.374	0.959	1.631	0.231	0.00	-40.3	1.38	0.99
Predicted	0.015	0.001	0.030	0.011				
	<i>Burned and mulched soil</i>							

Observed	0.205	0.165	0.292	0.043	0.02	-26.1	0.21	1.00
Predicted	0.000	0.000	0.001	0.000				
<b>Calibrated USLE-M model</b>								
<i>Unburned soil</i>								
Observed	0.001	0.001	0.001	0.000	0.08	-15.8	0.00	0.42
Predicted	0.000	0.000	0.001	0.000				
<i>Burned and not treated soil</i>								
Observed	1.374	0.959	1.631	0.231	0.03	-11.8	0.77	0.24
Predicted	1.038	0.040	1.881	0.749				
<i>Burned and mulched soil</i>								
Observed	0.205	0.165	0.292	0.043	0.28	-10.3	0.13	0.25
Predicted	0.154	0.000	0.338	0.152				

399 Notes:  $r^2$  = coefficient of determination; NSE = coefficient of efficiency of Nash and Sutcliffe; RMSE =

400 Root Mean Square Error; PBIAS = Coefficient of Residual Mass.

401

### 402 **3.2. MMF model**

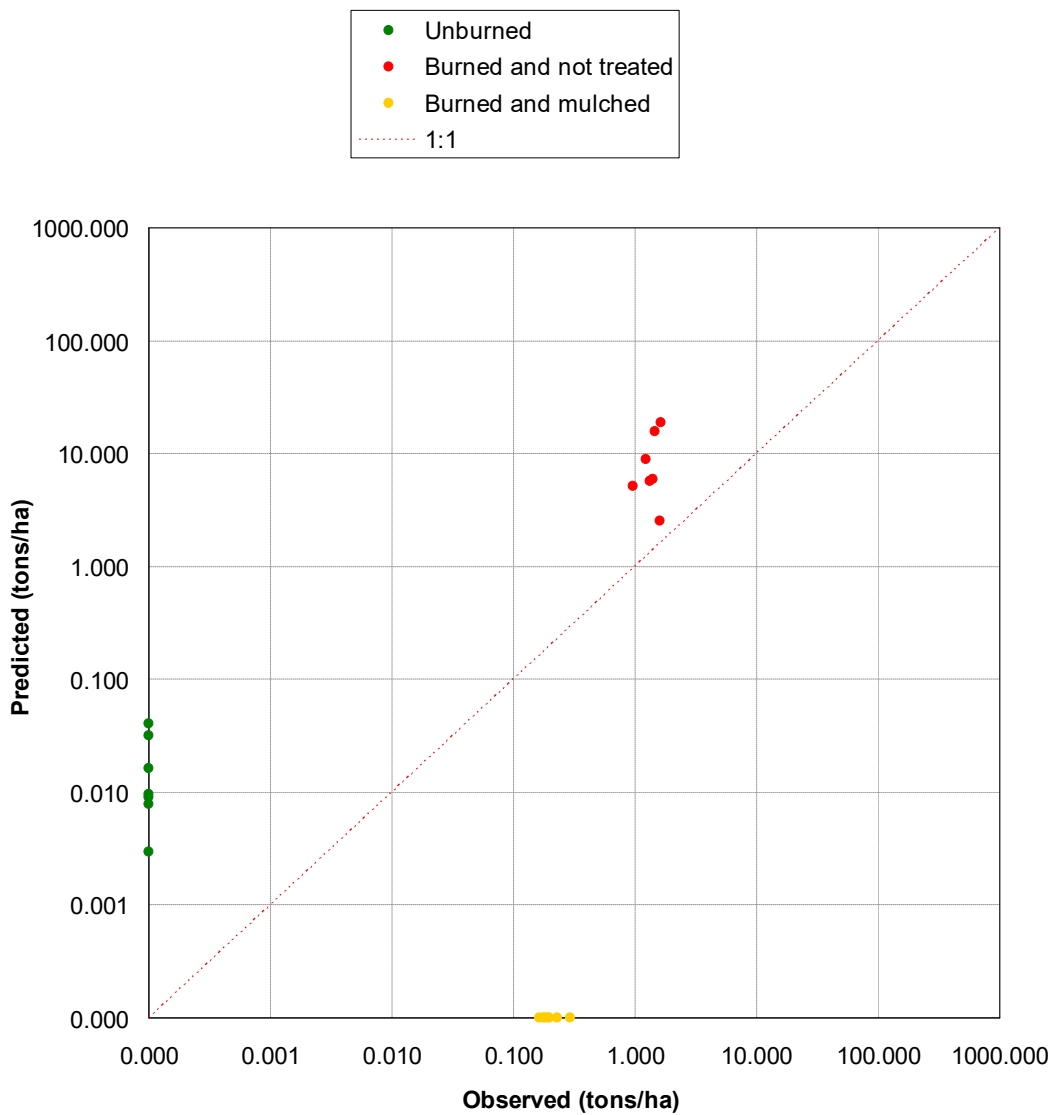
403

404 As noticed for WEPP, also the MMF model running with default parameters generally showed a  
405 large inaccuracy in predicting the soil loss. This is visually shown in the relevant scatterplot  
406 under all the soil conditions (Figure 4a) and confirmed by the poor evaluation indicators. The  
407 differences between the mean, maximum and minimum values of the predictions and  
408 observations were very large ( $> 100\%$  for the mean, minimum, and maximum values). The  
409 values of  $r^2$  were very low ( $< 0.16$ ), and negative NSE were achieved ( $< -26.2$ ). These values,  
410 coupled to the very large RMSEs ( $> 0.02$  tons/ha), show the inaccuracy of the default MMF  
411 model in predicting the erosion under all the modelled soil conditions (Table 5). This poor  
412 prediction capacity is due to the large over-estimation (in unburned, PBIAS = -23.1, and burned  
413 plots, PBIAS = -5.52) or under-estimation (in burned and mulched soils, PBIAS of 1) of the  
414 observations.

415 Due to these unsatisfactory performances, MMF was optimised as reported in the section 2.5.2.  
416 The model adaptations to the post-fire conditions greatly improved the erosion prediction  
417 capacity of MMF, except in the unburned soils. Under this condition, the values of  $r^2$  were close  
418 to zero, and NSE was largely negative (-84.6). The differences between the predictions and the  
419 corresponding observations were low for the mean values (-12.6%), but high for the other  
420 statistics ( $> 92.5\%$ ), although the model tendency to over-prediction was low (PBIAS = -0.13).  
421 In burned conditions, the large scattering of the "observations vs. predictions" pairs, which was  
422 large in the unburned soils, was reduced, and these pairs were close to the line of perfect  
423 agreement (Figure 4b). The improvement of the model performance is confirmed by the  
424 quantitative analysis. More specifically, the differences between the mean observed and  
425 predicted soil loss were from 5.1% (burned and not treated soils) to 12.9% (burned and mulched  
426 plots), while the maximum values differed by 1.41% (burned and not treated soils) to 6%  
427 (unburned plots). The  $r^2$  values were between 0.86 (burned and mulched soils) and 0.94 (burned  
428 and not treated conditions), and the RMSE values were lower than half the observed standard  
429 deviations (except under burned and mulched soil conditions). The soil loss prediction capacity  
430 can be considered good in the unburned, and burned and not treated soil conditions (NSE =  
431 0.75), and satisfactory for the burned and mulched plots (NSE = 0.43). The large over- or under-  
432 estimation, previously noticed for the default model, disappeared, and the observed soil loss was  
433 only slightly under-predicted in burned and not treated soils (PBIAS of 0.06) or over-predicted  
434 (burned and mulched plots, PBIAS = -0.13) by the optimised MMF model (Table 5).

435

436



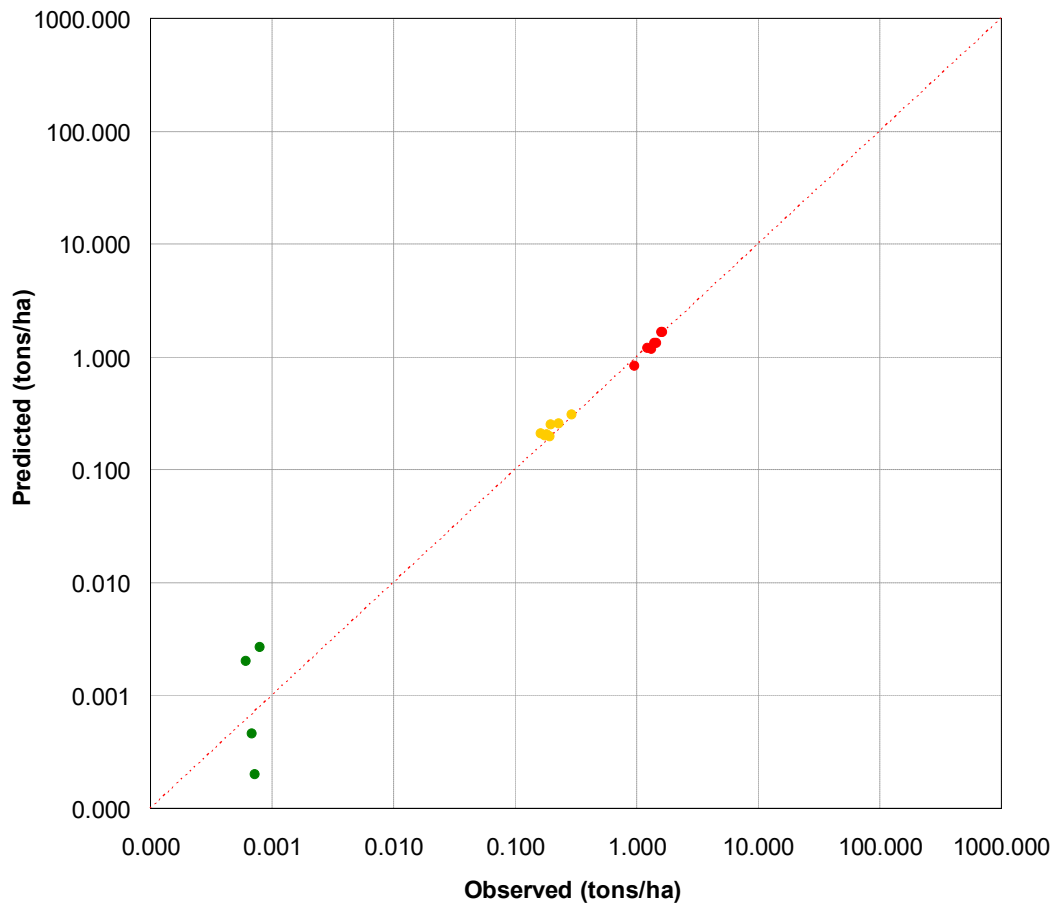
437

438

439

440

(a)



(b)

441  
 442  
 443 Figure 4 – Scatterplots of observed vs. predict soil loss using the MMF model (default, a, and  
 444 calibrated, b) model at the experimental site (Agramón, Castilla-La Mancha, Spain).

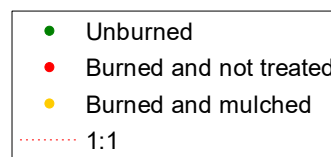
445  
 446 **3.3. USLE model**

447  
 448 The USLE model, running with default input parameters, showed a poor prediction capacity of  
 449 erosion under all soil conditions. The visual evaluation highlights a large scattering of data  
 450 around the line of perfect agreement (Figure 5a), and also the evaluation indicators confirm this  
 451 poor performance. In more detail, the differences in the main statistics between the observations  
 452 and the corresponding predictions were large (up to 100% for the mean and maximum values).  
 453 The values of  $r^2$  did not exceed 0.08 (unburned plots), NSE was always negative ( $< -26.1$ ), and  
 454 RMSE was always unsatisfactory (not lower than 10-fold half the observed standard deviation).  
 455 This poor model performance is due to the large under-estimation (for all soil conditions, PBIAS  
 456  $> 0.80$ ) of the observations (Table 5).

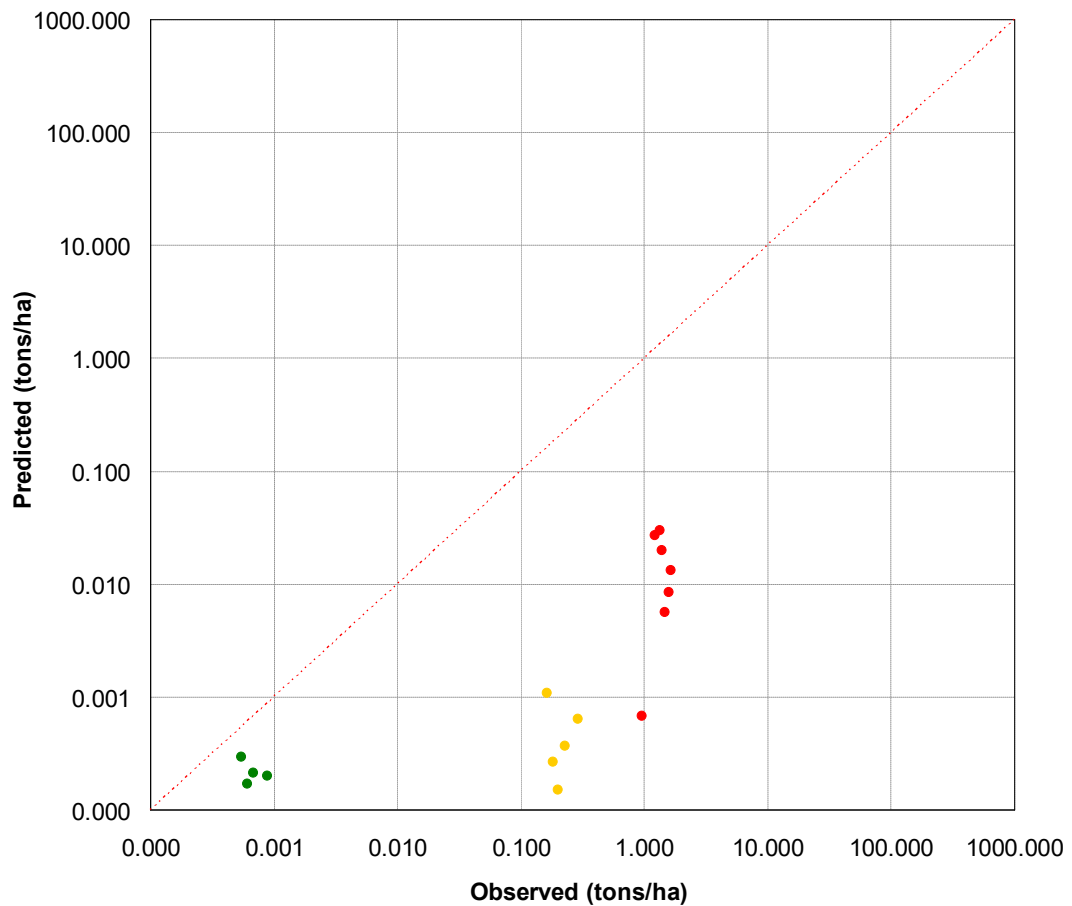
457 The USLE-M calibration through the setup of the values of CNs and C factors improved the  
 458 model performance, although its erosion prediction capacity was still unsatisfactory for all the

459 modelled soil conditions (Figure 5b). More specifically, the large differences between the main  
460 statistics of the observed and predicted soil loss decreased compared to the default model runs  
461 (errors in the mean values lower than -24.4%). The values of  $r^2$  were not higher than 0.28 for the  
462 burned and mulched plots, and close to zero for the other soil conditions. NSE was never  
463 satisfactory, since always negative ( $< -10.3$ ). The large under-estimation shown by the default  
464 model was reduced by the calibration, and the PBIAS (lower than 0.24) was always significant  
465 (Table 5).

466



467



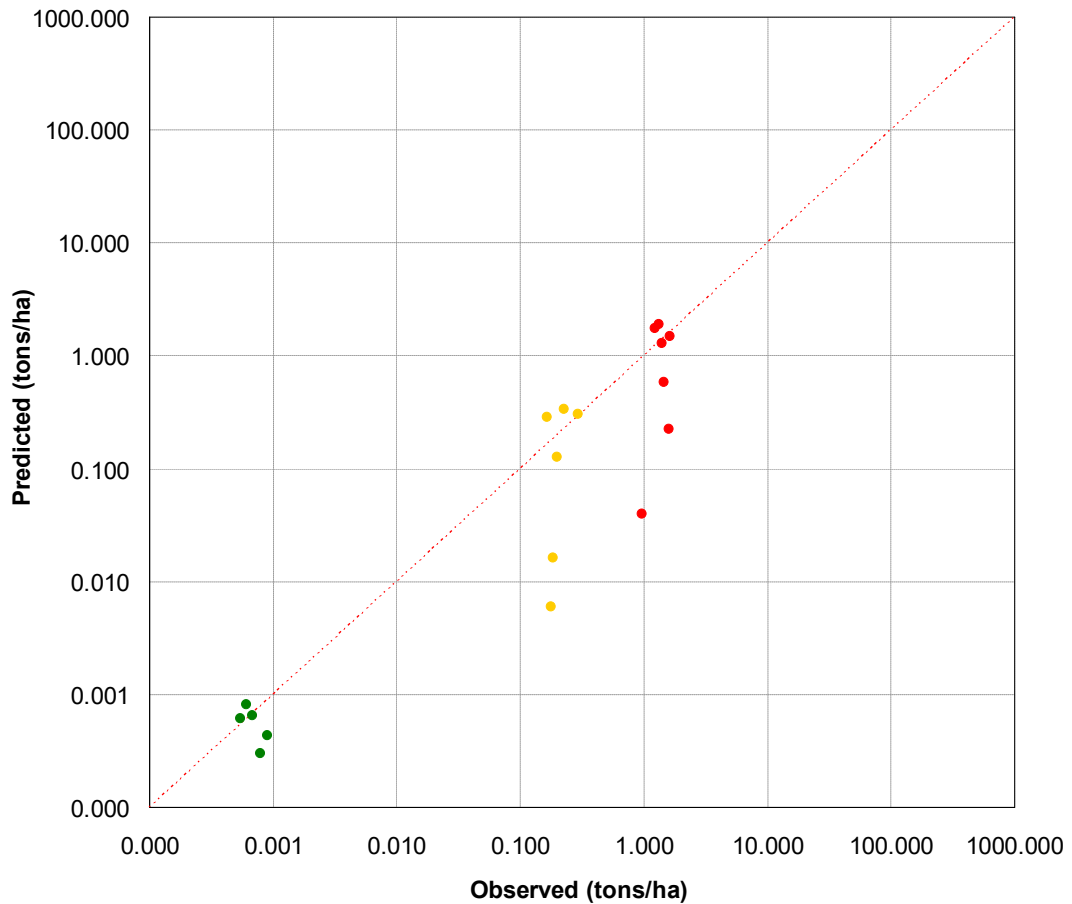
468

469

470

471

(a)



(b)

Figure 5 – Scatterplots of observed vs. predict soil loss using the USLE-M (default, a, and calibrated, b) model at the experimental site (Agramón, Castilla-La Mancha, Spain).

## 4. Discussion

### 4.1. WEPP model

The WEPP model gave poor predictions of soil loss in both unburned and burned areas (with and without the post-fire treatment). WEPP underestimated the erosion in the unburned plots, while, in contrast, the soil loss was largely overestimated in burned soils also by several orders of magnitude. The reason of this high inaccuracy could be many. First, the lack of calibration did not balance errors related to model parameterisation and hydrological simulations. The WEPP model requires dozens of input parameters that should be measured in the field. Since these measurements were not available at the experimental site, the modellers were forced to adopt literature values (e.g., for  $K_e$ ,  $K_i$ ,  $K_r$ , and  $\tau_c$ ) with evident low accuracy at reproducing the observed erosion rates. This problem limits the WEPP applicability in poor-data environments, where the erosion predictions may be generally inaccurate. Second, low erosion rates, as observed at the experimental site, are generally predicted with unsatisfactory reliability by WEPP (e.g., Grønsten & Lundekvam, 2006; Konz et al., 2009; Licciardello et al., 2006; Soto & Díaz-Fierros, 1998), when no runoff is simulated (as in many of our modelled events). This model considers only Hortonian processes and not runoff due to saturated flow (Soto & Díaz-Fierros, 1998), the latter mechanism being common for low-intensity rainfall (Beven & Kirkby, 1979). Third, the overland erosion being zero, it is evident that WEPP also failed at simulating the rainsplash erosion, which may be dominant at the experimental site (Díaz et al., 2022; Lucas-Borja

495 et al., 2022). Fourth, due to the lack of observations about the soil water content at the experimental plots, a constant  
496 initial saturation level was input to the model, while the real values were presumably variable, depending on the  
497 antecedent precipitation, thus affecting the runoff response simulated by the model. Also other authors found a large  
498 inaccuracy of WEPP in erosion predictions under burned conditions, as Soto & Díaz-Fierros (1998) and Fernández  
499 & Vega (2018), who applied WEPP in North-Western Spain, and by (Larsen & MacDonald, 2007), who modelled  
500 erosion in Colorado (USA). The latter authors confirmed that the model tends to over-predict sediment yields under  
501 1 ton/ha-yr.  
502

#### 503 **4.2. MMF model**

504

505 The erosion predictions of the default (uncalibrated) MMF were inaccurate for all modelled soil  
506 conditions, since the model noticeably over- or under-estimated the soil loss observations. We  
507 should remind that MMF has a strong empirical base and was developed using mostly smaller  
508 scale erosion plot data derived from agricultural fields (Vieira et al., 2018b). Therefore, the  
509 extrapolations to other land uses, such as the burned forestlands, have been made using  
510 corrections of the internal algorithms based on an empirical approach. In the burned and mulched  
511 plots, these poor performances were due to the inadequate estimation of the sediment transport  
512 capacity, since the model did not produce soil loss (which, in MMF, is the lower value between  
513 the sediment detachment and the transport capacity). The model, running with a too high default  
514 C-factor, simulated a total shadowing effect of the mulch material against rainfall erosivity,  
515 which resulted in a soil particle detachment equal to zero. This explanation agrees with the  
516 findings of Zema, Nunes, et al. (2020) and Vieira et al. (2014), who also found MMF failures in  
517 reproducing the sediment transport capacity. For the other soil conditions (unburned, and burned  
518 and not treated soils), the reasons of the low prediction capacity are not so clear as for burned  
519 and mulched conditions. A reason may be the inadequate incorporation of the fire effects in the  
520 soil parameters, which are variable over time, while all parameters were left constant throughout  
521 the modelling period. Wildfire, especially when at high severity, such as in our experiment,  
522 noticeably changes some physical properties of soil, and these changes are not easy to be  
523 reproduced in empirical models without recurring to correction factors (e.g., the C factor). Also  
524 Fernández et al. (2010) highlighted the role played by the C and P factors on reliable predictions  
525 of erosion by MMF, which often may lead to poor results, when these parameters are not fully  
526 established in the model input procedure.

527 This model inaccuracy in reproducing the soil loss in the experimental conditions required  
528 adjustments in MMF. After calibration, the prediction capacity of MMF noticeably improved  
529 compared to the default model, and the soil loss predictions were closer to the corresponding  
530 observations in burned soil (with or without mulching). However, the model prediction capacity  
531 was still poor in unburned plots. This model inability may be due to the null prediction of the

532 sediment transport capacity, which was too low to convey the sediment detached downstream. In  
533 other words, the soil loss due to rainsplash erosion and particle detachment by overland flow is  
534 simulated by MMF, but the process of sediment entrapment in the flow is not realistically  
535 reproduced. In contrast, the model prediction capacity was acceptable in the mulched plots and  
536 satisfactory in the burned and not treated soils. We ascribed the reasons for this model accuracy  
537 to the adaptations of the model to the burned conditions using the modifications of the internal  
538 algorithms and correction factors reported in methodology. More specifically, the incorporation  
539 of SWR effects, the modification of EHD, both following seasonal patterns, and the temporal  
540 downscaling to rainstorm events were successful to reproduce the variable dynamics of post-fire  
541 hydrology in the experimental soil. In contrast, the input of calibrated C-factors under post-fire  
542 management was less effective at modelling erosion in burned and mulched areas. A slight  
543 tendency to under-estimation or over-estimation of the observed soil loss remained. This model  
544 behaviour has been commonly detected in many MMF applications in burned areas (Fernández  
545 et al., 2010; Hosseini et al., 2018; Vieira et al., 2014, 2018b; Zema, Nunes, et al., 2020). It is  
546 worth to notice that the optimisation of the C-factor, which was weighted by the vegetal cover,  
547 improved the good prediction capacity of erosion under the same environmental conditions  
548 reported by Zema, Nunes, et al. (2020). The need of C-factor tuning is in agreement with several  
549 studies, which have demonstrated how accurate methodologies to estimate the C-factor are the  
550 key to reproduce the fire effects on soils (e.g., Larsen & MacDonald, 2007; Vieira et al., 2014).  
551 The comparison of our results with other MMF modelling experiences in Mediterranean  
552 conditions shows that the changes suggested by the previous studies (Hosseini et al., 2018;  
553 Vieira et al., 2014, 2018b; Zema, Nunes, et al., 2020), coupled to the C-factor tuning proposed in  
554 this investigation, successfully improved the MMF ability to model soil hydrology under  
555 variable conditions at the event scale. The model efficiency calculated in our study is similar to  
556 the values found by Vieira et al. (2014), who applied MMF in North-Central Portugal, and by  
557 Zema, Nunes, et al. (2020) in Central-Eastern Spain. Lower model efficiency (in the range 0.54-  
558 0.74) were reported by Vieira et al. (2014) again in North-Central Portugal, and by Fernández et  
559 al. (2010) and Hosseini et al. (2018) in North-Western Spain and North-Central Portugal,  
560 respectively. However, it should be highlighted that, while our study applied MMF at the event  
561 scale, all the other model experiences were carried out at aggregate temporal scales (from  
562 seasonal to annual periods). Many studies have shown that the erosion models better perform in  
563 predicting the average soil loss rather than erosion rates for specific events (Fernández et al.,  
564 2010; Larsen & MacDonald, 2007).

### 566 *4.3. USLE model*

567

568 The erosion predictions using the USLE-M model, running with default and calibrated input  
569 parameters, were poor. This low performance of the model is common for many erosion models,  
570 which generally over-estimate and under-estimate the lower and higher soil losses, respectively  
571 (Kinnell, 2003; Larsen & MacDonald, 2007; Nearing, 2000). In our opinion, the most important  
572 reason for this inaccuracy was the lack of observed values of the runoff coefficients. Their  
573 estimations required the use of the SCS-CN method in our study leading to a worse performance  
574 of the USLE-M model compared to MMF (which internally simulates surface runoff). The  
575 increases in the CNs of fire-affected areas are suggested by several authors (e.g., Carra et al.,  
576 2021; Lucas-Borja, Bombino, et al., 2020; Papathanasiou et al., 2015; Soulis, 2018), in order to  
577 simulate the effects of soil water repellency (left by wildfire) and complete removal of  
578 vegetation (due to burning) on soil hydrological properties in the “window-of-disturbance”.

579 In accordance with Vieira et al. (2015), another limitation of the USLE-family model is its great  
580 dependence on empirical parameters, such as the C and P-factors, to estimate the soil losses. The  
581 C-factor values, in spite of calibration, were not able to simulate the effect of the vegetation  
582 cover (in unburned and burned but untreated plots) and of the mulch layer against rain splash  
583 erosion and overland flow, thus contributing to the unsatisfactory model performances. Also the  
584 increase in soil erodibility (expressed by the K-factor) due to the wildfire-induced changes in  
585 physical parameters of soil and the lack of parameters simulating the effects of SWR may have  
586 played a significant role in this model inaccuracy (Fernández et al., 2010). We also agree with  
587 the latter authors about the inefficient use of an inadequate kinetic energy equation of rainfall for  
588 the Mediterranean climate by the USLE-family models, which could have affected the  
589 simulation of the real rainfall erosivity in the experimental conditions.

590 The poor prediction capacity of USLE-M indicates that the literature values that were adopted  
591 for the input parameters were not suitable for simulating erosion in unburned and burned (with or  
592 without mulching) plots. The USLE-M model inaccuracy in reproducing erosion in the unburned  
593 plots agrees with the findings by (Carra et al., 2021), who also achieved poor predictions of soil  
594 losses in unburned pine stands of Southern Italy. Comparisons of our results with other  
595 modelling experiences using USLE-family models in burned forests show satisfactory and  
596 inaccurate erosion predictions in Portugal (Vieira et al., 2018b) and in North-Western Spain  
597 (Fernández et al., 2010 and Fernández & Vega, 2016; 2018) in the short-term using the RUSLE  
598 model. The latter authors demonstrated that neither the model calibration by modifying the soil  
599 erosivity and erodibility, and the C-factors or letting the model account for the high organic

600 matter content of soil significantly improved the model accuracy. In Greece, (Karamesouti et al.,  
601 2016) reported a large overestimation of erosion using RUSLE, due to inadequate C-factors.

602

#### 603 *4.4. Comparison of the three models and future research needs*

604

605 The use of erosive models to predict soil loss under the three soil conditions gave contrasting  
606 results. A comparison of the erosion prediction capacity shown by the three models showed that  
607 the MMF model performs better and WEPP is the less accurate model in the experimental  
608 conditions. Therefore, MMF is the most accurate model for erosion predictions in the  
609 experimental environment, and furthermore offers simplicity of use low demand of input  
610 parameters. This derives from the availability of input parameters that successfully reproduce the  
611 effects of SWR and the post-fire variability of vegetation cover on the hydrological and erosive  
612 response of burned soils. Therefore, the MMF model appears as a valuable tool to predict both  
613 surface runoff (although this hydrological variable was not directly evaluated in this study) and  
614 soil loss in the Mediterranean burned forests, although being of less common use compared to  
615 the other two models. As such, the model may be used to prioritise the forest areas for post-  
616 management actions, in order to control the hydrogeological hazard and the risk of  
617 contamination of downstream water bodies. Furthermore, its process-based nature allows MMF  
618 to easily handle hydrological scenarios outside its calibration range, making the model  
619 particularly suitable for research purposes and scenario analysis (Beven, 2011; Vieira et al.,  
620 2014). In contrast, the USLE-M model must be further improved for erosion simulations in  
621 burned conditions, since the suggested values of the CN and C parameters are not able to  
622 reproduce the changes in soil hydrology due to wildfire. The use of the WEPP model is not  
623 advised in data-poor environments, such as the experimental site, since the scarce availability of  
624 input data affects the erosion prediction accuracy. Moreover, the use of MMF and WEPP models  
625 is more convenient compared to USLE-M, since the latter model does not predict surface runoff,  
626 which may be an essential information in areas that are prone to flooding hazard.

627 Overall, the comparison of these models to the same dataset, as done in this study, represents an  
628 added value (Larsen and MacDonald, [2007](#); Vieira et al., [2018](#)), since this allows the  
629 determination of the structural uncertainty in modelling predictions (Lopes et al., 2021). This is  
630 in agreement with Alewell et al. ([2019](#)) and Batista et al. ([2019](#)), who consider that there is no  
631 optimal model for worldwide applications, but the accuracy of the predictions of a model is  
632 mainly due to the quality of the input parameters and the calibration process (Lopes et al., 2021).  
633 Therefore, comparative studies as the current work are essential for land managers and

634 hydrologists, who must choose the most suitable prediction model for specific environmental  
635 conditions.

636

#### 637 ***4.5. Future research needs***

638

639 The verification of the three models in only one study area represents a limitation of our study,  
640 and this should be considered for future uses in other environmental contexts. More research is  
641 therefore needed to ensure the model transferability to other environmental contexts.

642 Widening the spatial scale and reducing the temporal scale supports a better understanding of  
643 post-fire impacts from on-site processes (plot or hillslope) to off-site impacts (catchment scale)  
644 (Lopes et al., 2021) on one side as well as the improvements of model predictions at the event  
645 scale. The latter concept is particularly true in those environments, where the hydrological and  
646 erosive response is produced by a low number of small but intense rainfall events (such as in the  
647 semi-arid conditions).

648 Moreover, there is the need for larger field datasets, which should allow a simultaneous model  
649 calibration and validation, which leads to a more robust prediction capacity of the tested models  
650 for MMF and to the possibility to identify the weakness in the applicability of the other two  
651 models.

652 Finally, we also suggest enlarging the modelling exercises about erosion in burned areas treated  
653 with post-fire management techniques, considering that the relevant research is not exhaustive  
654 and the number of studies that analyse the post-fire actions through erosion modelling is limited  
655 (Zema 2021).

656

## 657 **5. Conclusions**

658

659 In forest soils of Central Eastern Spain, burned by a wildfire and then mulched using straw, the  
660 WEPP model noticeably underestimated the erosion in the unburned areas, while, in contrast, the  
661 soil loss was largely overestimated in the burned soils also by several orders of magnitude. This  
662 large inaccuracy was presumably due to lack of model calibration. The uncalibrated MMF model  
663 noticeably over- or under-estimated the soil loss observations for all modeled soil conditions,  
664 when running with default input parameters. However, the optimisation of this model with a new  
665 procedure to estimate the C-factor, resulted in a satisfactory erosion prediction capacity, in  
666 burned plots with or without the mulching treatment. Calibration failed to improve the MMF  
667 model simulations in unburned soils. The performances of the USLE-M model were poor before

668 and after calibration CNs and C-factors under all simulated soil management conditions. We  
669 conclude that the most accurate prediction model is MMF to estimate the soil loss in  
670 Mediterranean burned forests with or without post-fire mulching.

671

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673

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679

## 680 **Conflict of interest statement**

681

682 All authors declare no conflict of interest.

683

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685

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885

886

**SUPPLEMENTARY MATERIAL**

887

888

Table 1SM – Full list of input parameters of the WEPP model to predict soil loss at the experimental site (Agramón, Castilla-La Mancha, Spain).

889

Management	Slope			Soil							
	Position	Length (m)	Slope (%)	Albedo	Initial saturation level (%)	Depth (mm)	Sand (%)	Clay (%)	OM (%)	CEC (meq/100g)	Rock (%)
<i>Unburned</i>											
Shrub perennial	<i>Upper</i>	12.9	68.3	0.23	75	1000	26.3	14.3	2.88	9.9	50
	<i>Middle</i>	103.5	51								
	<i>Lower</i>	12.9	22.3								
<i>Burned and not treated</i>											
25% ground cover - high severity	<i>Upper</i>	13.6	23.7	0.23	75	1000	31.7	12.8	2.13	9.9	50
	<i>Middle</i>	108.8	41								
	<i>Lower</i>	13.6	21.3								
<i>Burned and mulched</i>											
Fallow initial condition - wheat residues	<i>Upper</i>	15.4	48.7	0.23	75	1000	30.2	18.5	2.38	9.9	50
	<i>Middle</i>	123.5	40.3								
	<i>Lower</i>	15.4	19								

890

Notes: OM = organic matter; CEC = cation exchange capacity.

891 Table 2SM – Full list of input parameters of the MMF model to predict soil loss at the experimental site (Agramón, Castilla-La Mancha, Spain).

892

Factor	Soil condition					
	Min	Max	Min	Max	Min	Max
	<i>Unburned</i>		<i>Burned and Not Treated</i>		<i>Burned and Mulched</i>	
	Uncalibrated model					
<i>R</i>	43.3	115.2	43.3	115.2	43.3	115.2
<i>R<sub>n</sub></i>	1	1	1	1	1	1
<i>I</i>	10	10	10	10	10	10
<i>MS</i>	0.2	0.2	0.4	0.4	0.4	0.4
<i>BD</i>	1.3	1.3	1.3	1.3	1.3	1.3
<i>EHD</i>	0.1	0.1	0.1	0.1	0.1	0.1
<i>K</i>	0.9	0.9	0.9	0.9	0.9	0.9
<i>COH</i>	2.0	2.0	2.0	2.0	2.0	2.0
<i>S</i>	27	27	22	22	22	22
<i>A</i>	0	0	0	0	0.3	0.3
<i>Et/E<sub>0</sub></i>	1	1	0.1	0.1	0.9	0.9
<i>C</i>	0.003	0	1	1	0	0
<i>CC</i>	0.7	0.7	0	0	1	1
<i>GC</i>	0.4	0.6	0	0	1	1
<i>PH</i>	0	0	0	0	0.5	0.5

	Calibrated model					
<i>R</i>	43.3	115.2	43.3	115.2	43.3	115.2
<i>R<sub>n</sub></i>	1	1	1	1	1.0	1.0
<i>I</i>	10	10	10	10	10.0	10.0
<i>MS</i>	0.35	0.35	0.28	0.28	0.28	0.28
<i>BD</i>	1.3	1.3	1.3	1.3	1.3	1.3
<i>EHD</i>	0.2	0.2	0.09	0.09	0.12	0.12
<i>K</i>	0.9	0.9	0.9	0.9	0.9	0.9
<i>COH</i>	2	2	2	2	2	2
<i>S</i>	27	27	22	22	22	22
<i>A</i>	0.0	0.0	0.0	0.0	0.06	0.06
<i>E<sub>t</sub>/E<sub>0</sub></i>	0.80	0.70	0.05	0.05	0.50	0.45
<i>C</i>	0.135	0.156	0.120	0.558	0.029	0.034
<i>CC</i>	0.7	0.7	0	0	0.05	0.05
<i>GC</i>	0.45	0.39	0.19	0.07	0.56	0.47
<i>PH</i>	0.5	0.5	0	0	0.6	0.6

893 Notes: *R* = rainfall depth; *R<sub>n</sub>* = rainy days; *I* = intensity of erosive rains; *MS* = moisture content at field capacity; *BD* = bulk density of the topsoil; *EHD* = effective hydrological  
894 depth; *K* = detachability index; *COH* = cohesion of surface soil; *S* = slope steepness; *A* = vegetation cover; *E<sub>t</sub>/E<sub>0</sub>* = ratio of actual and potential evapotranspiration; *C* = cover  
895 management factor; *CC* = percent canopy cover; *GC* = ground cover; *PH* = plant height to the ground surface.

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897 Table 3SM – Full list of input parameters of the USLE-M model to predict soil loss at the experimental site (Agramón, Castilla-La Mancha, Spain).

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Model	Input	Measuring unit	Soil conditions
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	parameter			Unburned		Burned and not treated		Burned and mulched		
				Default	Calibrated	Default	Calibrated	Default	Calibrated	
SCS-CN	CN	-		39	7	90 (79)	13	65 (39)	8	
	$\lambda$	-		0.2						
USLE-M	$Q_r$	-	max	0.375		0.694		0.570		
			min	0.176		0.409		0.001		
	$R_e$ -factor	MJ mm/ ha h	max	302						
			min	7						
	LS-factor			60.51						
	$K_{UM}$ -factor	tons h/MJ mm		0.002		0.001		0.002		
	C-factor	-		0.0001	0.0002	0.006	0.526	0.0002	0.085	
	P-factor	-		1						

899 Notes: CN = curve number;  $\lambda$  = rainfall depth coefficient;  $R_e$ -factor = rainfall erosivity factor;  $K_{UM}$ -factor = soil erodibility factor; C-factor = cover  
900 management factor; P-factor = conservation practice factor; the values related to the first two modelled events are reported in brackets.