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The effect of broadleaf forests in wildfire mitigation in the WUI – A simulation study

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ABSTRACT

The increasing occurrence of large wildfires in Southern European countries calls for the adoption of more effective measures of fire prevention, to protect people and infrastructures, namely in the Wildland-Urban Interface (WUI). Previous research suggests that broadleaf forests could mitigate the effects of these wildfires due to their lower flammability. However, few attempts have been made to investigate the possibility of using broadleaf-based green firebreaks to protect infrastructures, as an alternative to current less ecologically and economically sustainable fuel reduction techniques. Here, we assess the relevance of a broadleaves fuel model in reducing fire hazard in the WUI, by analyzing a set of six Industrial Zones (IZs) in Central Portugal, severely affected by wildfires during the catastrophic fire season of 2017. We developed alternative scenarios for the spatial simulation of fire behavior, using a factorial combination of weather conditions (standard and extreme), buffer around each IZ (distances of 100 m and 500 m) and land cover (current and broadleaves fuel model). The simulations were grounded on real-world data obtained from reconstructed fire front isochrones and fieldwork. Our results suggest that replacing the flammable vegetation present in the WUI with broadleaf forests could reduce fireline intensity by up to five times, even under extreme weather conditions. This reduction occurs subtly as the broadleaf cover interface is expanded from 100 m to 500 m. Our results show that fires that exceed the suppression capacity in pine and eucalypt forests (>4 m flame length) can be effectively suppressed in broadleaf forests under extreme fire weather (1.4 m flame length) and easily suppressed in broadleaf forests in standard weather (0.8 m flame length). Due to significant changes in land use and extreme weather events, future large wildfires could occur again in Portugal. In this regard, our results corroborate the urgent need to discuss forest management in the country, which has already proven to be insufficient to prevent fire disasters in the WUI.

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1. Introduction

Wildfires are ecologically necessary to many ecosystems but become problematic when they occur within the Wildland Urban Interface (WUI), where human development intermixes with wildlands [1–4]. Fires in WUI are a worldwide socioeconomic problem, becoming increasingly frequent, especially in North America [4–7], Australia [8,9], and, the Mediterranean basin [10–14]. Particularly in Portugal mainland, the devastating season of mega-fires that occurred in 2017, has brought economic losses estimated at EUR 1.7 billion [15,16] and 112 human fatalities [17,18]. A year later, 99 lives were lost in Greece in a single wildfire [19]. After these fatalities and due to the higher occurrence of wildfire disasters in the WUI areas, the European Commission [20] and other countries over the world [4] are joining efforts to devise strategies based on fire-smart policies. These policies attempt to address the ever-increasing challenges of wildfire protection in face of climate change [20–22]. In general terms, these policies recommend mitigation activities designed to treat the WUI, through fuel management and development of fire-resistant landscapes [21,23,24], as well as promoting improvements at the property level, that is, in WUI structures [4,25,26].

In North America [27] and in the Mediterranean [28], particularly in Portugal [22,24], strategies have already been established for the development of multifunctional landscapes capable of mitigating fire hazard within the scope of fire-smart forest management. As well, there is already evidence that wildfires are weakly to moderately selective in relation to different types of forests [11,29–31]. Small remnants of broadleaf native forests in the northern region of Portugal, for example, demonstrated a strong edge effect on fires in non-extreme weather conditions, with a high probability of self-extinction [32]. This was also the case of Mata da Margaraça, a protected broadleaf forest that was affected by large fires in 1987 and 2017 [33] and its biological legacy has remained almost intact. In 2017, Mata da Margaraça withstood two high intensity fires, with a decrease in fire severity from the edges towards the interior [18]. These study cases suggest that broadleaf forests may decisively mitigate fire potential in Portugal [32,34,35] through fire-smart fuel management based on conversion to less flammable forest types [36] and on the development of resilient landscapes to fire in the European context. However, the positive effect of broadleaf forests can vary depending on fire intensity [30] and on the extent of this forest type [34] and is unknown in WUI areas in Portugal.

On the other hand, the advantages of broadleaf forests lie far beyond the specific goal of fire prevention, because of the known benefits in terms of nature conservation, landscape, and recreation, for example. However, in Portugal mainland, these forests represent less than 10% of forest cover [37], and their long-term persistence is limited by different factors that affect tree recruitment, including biophysical conditions, seed availability, land use intensity, and disturbance regimes [38]. It is also important to point out that the possibility of expanding broadleaf forests is in line with the current challenges of the Portuguese government on the need to change the profile of reforestation throughout the country to meet sustainability agendas [39,40] and with the main targets set out in the Decade for Ecosystem Restoration (2021–2030) launched by the United Nations [41].

However, to our knowledge, the effect of broadleaf forests has never been scientifically tested as a means to specifically reduce fire hazard nearby infrastructures. We propose to contribute to fill this knowledge gap by seeking to test the following hypothesis: What is the potential mitigation effect of a broadleaf forest cover on wildfire behavior in WUI under extreme weather conditions? To answer this question, we developed spatial simulation fire behavior scenarios, using the industrial-forest interface of six Industrial Zones (IZs) that were extremely affected by the 2017 fires, in Central Portugal. According to the Portuguese legislation [42], a buffer zone of 100 m around buildings (including industrial and housing settlements) must be fuel-reduced, aiming fire hazard mitigation. The result of this effort is ephemeral, since vegetation grows fast, leading to the need of recurrent treatments [43,44]. Moreover, given the high fuel treatments costs current regulations are poorly enforced [45].

Here we use real-world data from the 2017 fires [17,18] in Portugal to support fire simulations around IZs, based on fuel models representative of alternative forest cover scenarios. These scenarios include the use of broadleaves as a sustainable alternative to the current obligation of recurrent fuel reduction operations. The previously mentioned case of the Mata da Margaraça forest came up as an opportunity to study this effect. Thus, using field data, reconstructed fire isochrones and fire behavior modeling, the objectives of this paper are: a) to estimate the behavior of a 2017 wildfire in a mature broadleaf forest (Mata da Margaraça), compared to the surrounding landscape; b) to estimate the behavior of five 2017 wildfires that affected six IZs in Central Portugal, contrasted to a scenario where the original land cover and corresponding fuel models of the WUI interface areas, is replaced by a fuel model typically associated to broadleaf forest stands occurring in mainland Portugal and, c) to use these data to model the effect of land cover (fuel models), buffer lengths, and weather conditions, on standard fire behavior metrics.

2. Data and methods

2.1. Study areas

The study areas include six Industrial Zones (IZs) located in Central Portugal: Mira (MI), Tocha (TO), Oliveira do Hospital (OH), Oliveira de Frades (OF), Freixo-Mortágua (MO) and Graça-Pedrógão Grande (PG) (Fig. 1). The IZs have an average area of 45.3 ± 39.2 ha interfacing mainly with maritime-pine and eucalypt stands, shrubland, and invasive woody species (Fig. 1). The IZs are located at altitudes ranging from 20 m to 500 m (Table 1). Average annual temperatures are between 13 °C and 15 °C and rainfall varies between 880 mm and 1200 mm.

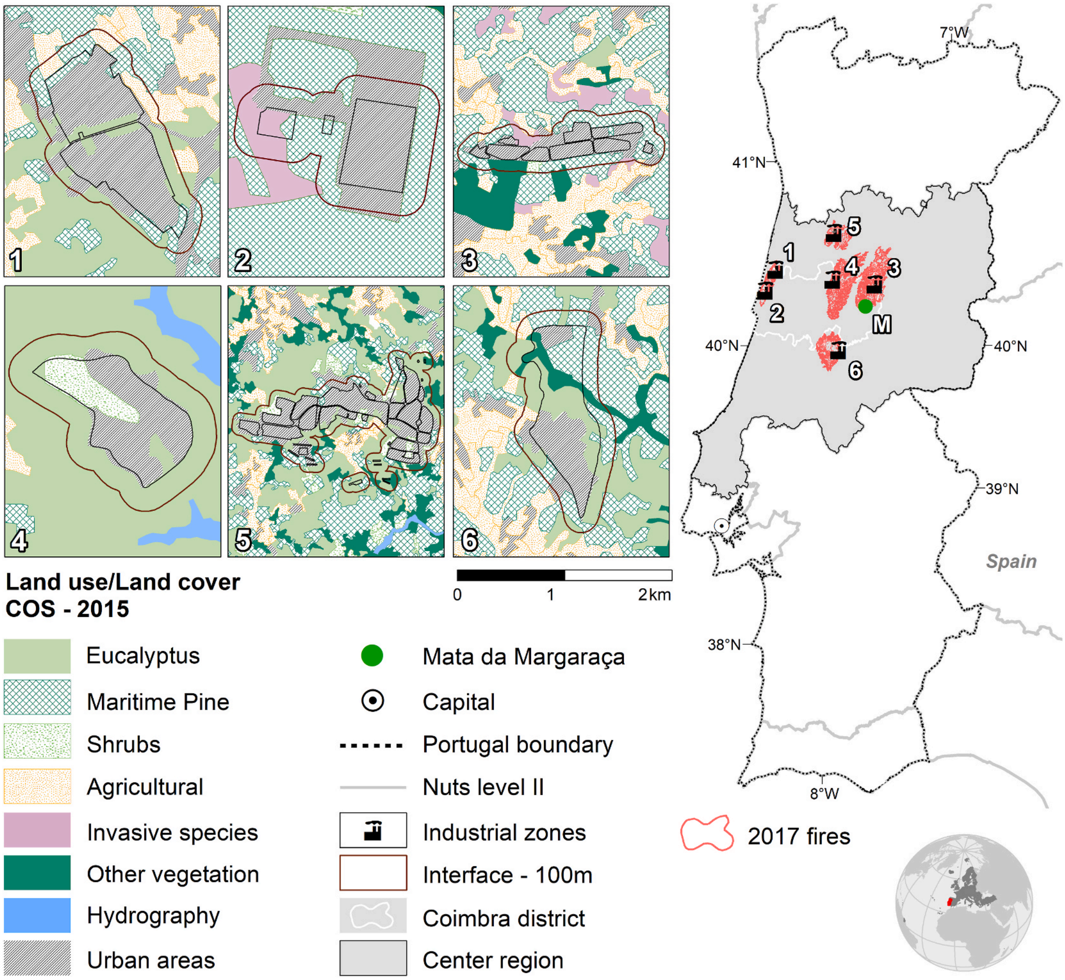

Fig. 1. Industrial Zones and land use/land cover from the 2015 Land Cover Map (*COS*-Carta de Ocupação do Solo) by the General Directorate of the Territory [46]. 1: Mira, 2: Tocha, 3: Oliveira do Hospital, 4: Freixo-Mortágua, 5: Oliveira de Frades and 6: Graça-Pedrógão Grande. The five mega-fires are represented on the right as red spots. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

All IZs were affected by the 2017 wildfires (Table 1), for which we produced a series of fire behavior simulations under different scenarios. With that purpose, we collected field data in Mata da Margaraça (MM), a native mature broadleaf forest that was also impacted by fire and is located in Central Portugal (Fig. 1). MM was used in this study as a reference broadleaf forest of the center region, mainly in terms of fuel and canopy characteristics. It comprises a mix of Atlantic and Mediterranean woody species, such as *Quercus robur, Castanea sativa, Quercus suber, Viburnum tinus, Ilex aquifolium, Laurus nobilis* and *Prunus lusitanica* [52]. MM covers an area of 68 ha (Table 1) on a slope with N-NW aspect and maintains high humidity values almost all year round, due to the sub-humid pre-atlantic climate and mild temperatures, even in the dry season [52,53].

Table 1
Characterization of the location (climatic annual averages) of the six Industrial Zones: Mira (MI), Tocha (TO), Oliveira do Hospital (OH), Oliveira de Frades (OF), Freixo-Mortágua (MO), Graça-Pedrógão Grande (PG) and of the MM (Mata da Margaraça). *100 m interface; $^{+}$Represents the fire that reached Mata da Margaraça area; [1]Burned area on the interface; [2]Guerreiro et al. [17,18].

Study area	Area	Interface Area*	Altitude	Rainfall	Temp	Burned area[1]	2017 fires[2]	Reference
	ha	ha	m	mm	°C	ha		
MI	56	96	20	880	15	22	Quiaios	[47]
TO	12	37	20	880	15	12	Quiaios	[47]
OH	23	67	450	1000	14	25	Arganil	[48]
MO	19	41	160	990	15	22	Vilarinho	[49]
OF	127	272	500	1200	13	134	Vouzela	[50]
PG	36	70	450	800	14	61	Pedrógão	[51]
MM	*68*	-	*600*	*1100*	*12*	*33*$^{+}$	*Arganil*	[52]

2.2. Fire behavior simulations

2.2.1. General aspects

Fire simulations were performed using the FARSITE 4 software [54]. FARSITE delivers spatially-explicit information of fire growths and burned area across time and the corresponding fire behavior metrics, namely rate of fire spread (ROS), heat per unit area (HPA), flame length (FML), and fireline intensity (FLI), using the Rothermel's surface fire spread model as the core algorithm [55]. The simulations ran on a composite 25 m x 25 m raster file including different data layers characterizing topography (elevation, slope angle, slope aspect), and surface and canopy fuels (section 2.2.3). The simulations were carried out for the reference broadleaf forest (Mata da Margaraça) and for six IZs including a 2-km buffer surrounding these areas. We used a simulation time step of 1 h (1h00) with a 25-m perimeter resolution. Fig. 2 presents a scheme of the simulation process. Real-world data were used in the reconstructed isochrones of the 2017 fires, in the development of the broadleaves fuel model and in the simulation scenarios (section 2.2.2).

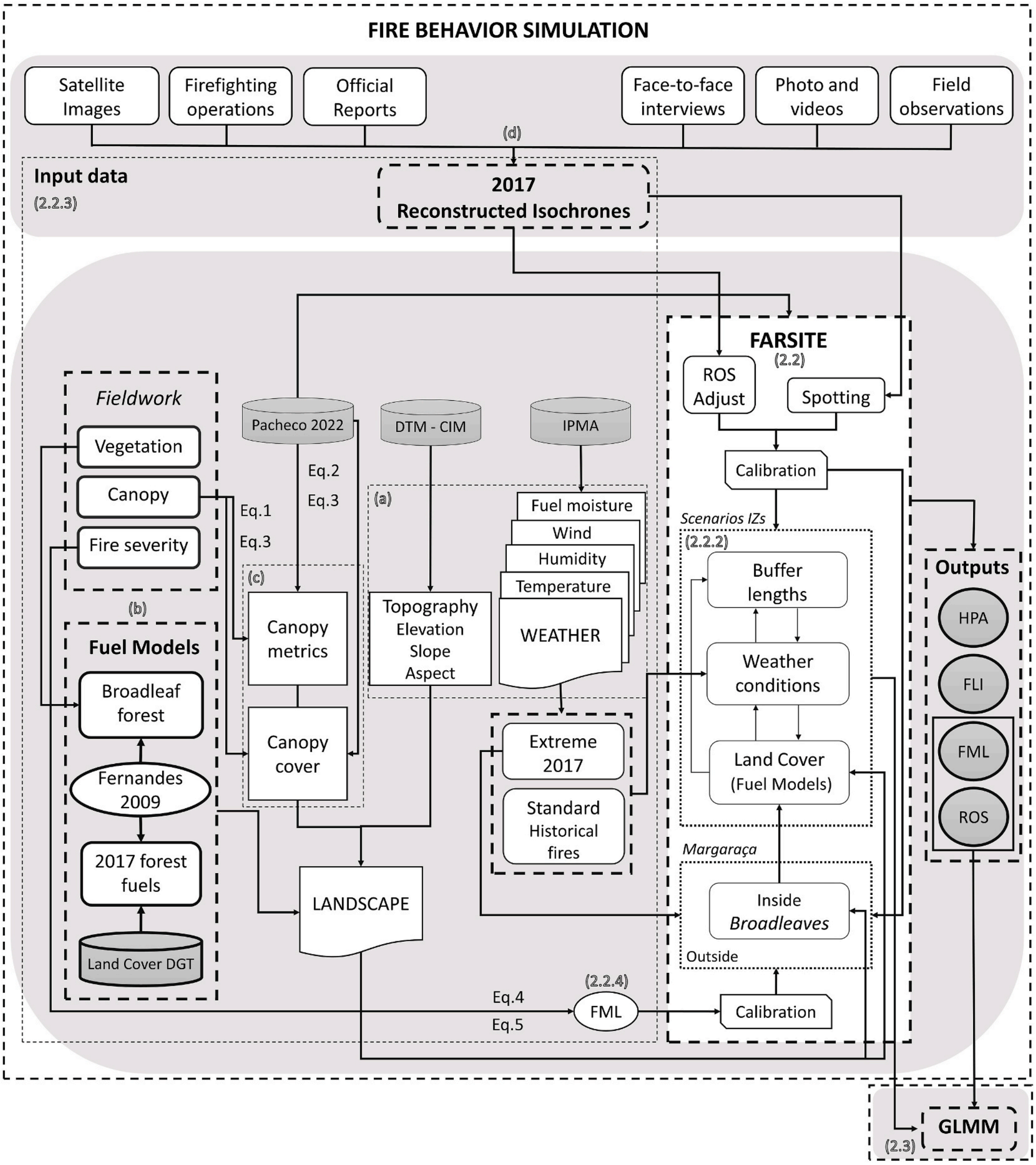

Fig. 2. Simulation flowchart, including the Data Sources: DGT - General Directorate of the Territory [46]; DTM-CIM - Digital terrain model - DTM [56] and IPMA - Portuguese Institute for Sea and Atmosphere [57]; Fire Behavior Metrics: ROS - Rate of Spread (m min^{-1}); FLI - Fireline Intensity (kW m^{-1}); FML - Flame Length (m) and HPA - Heat per Unit Area (kJ m^{-2}); Statistical Analysis: GLMM (Generalized Linear Mixed Models) and fire behavior simulations in FARSITE software: basically, include data collection (*Fieldwork*), selection of Fuel Models, other Input data and data used for calibration of model parameters (2017 Reconstructed Isochrones and Fire Severity). The FARSITE frame includes the simulation structure in Mata da Margaraça, the simulation scenarios for the IZs, and the calibrated parameters (e.g., ROS Adjust and Spotting). The outputs are fire behavior metrics. Among these, FML and ROS were used in the GLMM statistical model. The alphanumeric indications (e.g., (2.2); (a) (b), etc.); are the methodology sections to which the flowchart refers.

2.2.2. Scenarios

Our simulation scenarios for IZs include different fuel models, buffer lengths and weather conditions (FARSITE frame - Fig. 2). We ran the simulations for two buffer lengths around the IZs, respectively based on distances of 100 m (required by legislation), and of 500 m, for two weather scenarios: the extreme conditions observed in 2017 and standard conditions for large fires in Portugal. Standard conditions are the median of meteorological variables (temperature, relative humidity, and wind speed) of the large-fire (>500ha) days of the last ten years (2011–2021), excluding 2017 [57]. In this period, we identified 130 large wildfires using the Portuguese Fire Atlas provided by the Institute for the Conservation of Nature and Forests [33]. We developed four specific scenarios, responding to the four combinations of weather and buffer lengths, for each of the six IZs covered by the fuel models present in 2017, thus resulting in 24 simulations.

We produced new landscapes for FARSITE simulations by replacing the 2017 fuel models of the IZs by a fuel model representative of a mature broadleaf forest of the center region of Portugal to test the effects of this forest type on fire behavior. These replacements were made for the same buffer lengths (100 m and 500 m) and weather scenarios and resulted in 24 additional simulations for the six IZs.

2.2.3. Input data

The input data (Fig. 2) include data from official sources, namely topography, weather, and wind; data from the literature (fuel models and canopy characteristics); calculations based on previously available methods, e.g., fine fuel moisture content and slope; primary fieldwork data respecting to the reference broadleaves forest; and reconstructed isochrones of the 2017 wildfire used for calibrating the simulations.

a) Topography, weather, and wind

The three topographic data layers were extracted from a digital elevation model (DEM [56]). Slope angle and slope aspect were computed following the Horn's method [58]. Weather and wind data (Table S1) were provided by the IPMA-Portuguese Institute for Sea and Atmosphere [57]. From these data, we also estimated the initial fuel moisture content using the Fine Fuel Moisture Code (FFMC) of the Canadian Fire Weather Index (FWI) system, according to the equations of Van Wagner [59]. Through the Nelson [60] model, FARSITE adjusts the initial fuel moisture throughout the simulation according to local terrain and the canopy cover conditions.

b) Fuel models

We adopted the set of 18 fuel models developed by Fernandes et al. [61], applicable to the whole Portuguese mainland (Table S2). The development of fuel maps was based on the land cover map of 2015 (COS [46]). We converted these land cover classes into the most likely fuel models, following the guidelines provided by Fernandes et al. [61] and integrating our own knowledge of local conditions. The broadleaves fuel model used in the development of fuel scenarios around the IZs was also extracted from Fernandes et al. [61] but adjusted to field data collected in Mata da Margaraça, our reference broadleaf forest for which we simulated the behavior of an extreme 2017 fire that allowed comparison with other forest types.

Given the low severity of the 2017 fire at the core of the MM, which highlights the effect of broadleaves, we assumed that fuels in 2021 had sufficiently recovered to 2017 pre-wildfire levels. Moreover, we wanted to replicate as close as possible the fuel characteristics of a mature broadleaf forest typical of the center region, to be tested in the IZs buffers. We established a representative sampling transect with 12 sampling points, each consecutive point separated by 50 m. The transect had an East-West orientation (the same direction as the 2017 fire) and was established in the center of a small dense and fire-resistant forest patch. We measured litter depth at each point and characterized the vegetation layers (Table S3). Subsequently, the fuel model guidelines of Fernandes et al. [61] were used to select the fuel model at each point, based on shrub cover and depth, litter cover and depth, and tree layer composition. Along the transect, we identified the M-CAD fuel model (deciduous hardwood litter with understory) and the F-FOL model (deciduous hardwood litter). Thus, the final fuel model adopted (M-M broadleaves) resulted from a weighted average of the parameters of the two models (fuel loads, fuel depth, heat content, moisture of extinction, surface-area-to-volume ratio, etc.) based on the relative frequency of each model along the transect, in this case, 66% of M-CAD and 34% of F-FOL (Table S4). Fernandes et al. [61] fuel models were assigned to the vegetation types outside MM (Table S4).

The simulations in the IZs include the fuel models [61], which cover the buffer lengths of 100 m and 500 m. The buffers represent the areas used to establish the fuel scenarios in the IZs. Table 2 shows the percentage of each fuel model in the buffers.

The 2-km landscape used for the simulations considers all spatial arrangements (also as fuel models) existing both around Mata da Margaraça (Table S4) and the IZs (Table S5). These arrangements include, for example, agricultural areas that can function as fuel-breaks and influence fire progression. However, all spatial arrangements located in the 100 m and 500 m buffers around the IZs (Table 2) were replaced by broadleaf forests, in the specific scenarios in which we tested the M-M model to simulate fire behavior. We did the full replacement in these scenarios because the spatial heterogeneity around the IZs is low (Fig. 1), the interfaces being 90% covered by pine and eucalypt forests on average [46].

Table 2
Relative proportion (%) of each fuel model around the six IZs (100 m and 500 m buffers). The Industrial Zones (IZs) are identified as: Mira (MI), Tocha (TO), Oliveira do Hospital (OH), Oliveira de Frades (OF), Freixo-Mortágua (MO) and Graça-Pedrógrão Grande (PG). F-models (litter predominance), M-models (litter and vegetation) and V-models (vegetation predominance); M-CAD: Deciduous forest with litter and understory; M-EUC: Eucalyptus litter and understory; M-PIN: Pine with litter and understory; V-Ha: Tall herbs; V-Hb: Short herbs; V-MAa: Shrub spp. With dead fuel, high; V-MH: Herb-shrub matrix; V-MMa: Shrub spp. Without dead fuel, high; V-MMb: Shrub spp. Without dead fuel, low.

Industrial Zones		MI		TO		OH		MO		OF		PG	
Buffer (m)		100	500	100	500	100	500	100	500	100	500	100	500
Fuel Models (%)	M-CAD	–	–	–	–	–	1	–	–	–	–	–	–
	M-EUC	28	33	–	–	–	1	43	76	23	33	13	15
	M-PIN	19	28	32	69	38	36	–	1	10	17	19	29
	V-Ha	–	–	–	–	–	–	–	1	–	1	–	6
	V-Hb	16	16	–	1	6	32	–	5	4	17	10	19
	V-MAa	–	–	8	13	2	9	–	–	–	–	–	–
	V-MH	–	–	–	1	1	1	–	–	3	3	–	–
	V-MMa	–	–	–	–	–	–	–	–	–	–	–	–
	V-MMb	1	2	–	–	1	1	–	–	–	1	5	4
Non-Combustible (%)		36	21	60	16	52	19	57	17	60	28	53	27

c) Canopy characteristics

Canopy characteristics were assessed through fieldwork and using empirical relationships from the literature (Fig. 2). Considering the main forest cover of the study areas we gathered information on broadleaf, pine (mostly maritime pine, *Pinus pinaster*), and eucalypt (*Eucalyptus globulus*) trees. The variables Canopy Cover (CC), Tree Height (TH) and Crown Base Height (CBH) for broadleaf forest were obtained through fieldwork in the MM. Data for pine and eucalypt were directly obtained from fieldwork in 30 stands distributed in Central Portugal, by Pacheco et al. [44] between June and September 2021. To measure CC in MM, we took a photo at each transect point, using a digital camera with a fish-eye lens installed in a smartphone, facing the sky. The images were processed in ImageJ [62] to extract the percentage of tree crowns. Using a hypsometer (VERTEX 5), we measured the TH and CBH of the four nearest trees from each point along the sampled transect, i.e., n = 48 trees. Diameter at Breast Height (DBH) was measured in the same trees with a forest compass (Table S3). The same procedures were adopted in pine and eucalypt stands by Pacheco et al. [44]. Subsequently, we used these data to estimate canopy fuel load, assumed to equate to foliage weight, using biomass equations adopted by the national forest inventory [37].

For oaks and other broadleaves, the equation is:

$$\mathrm{wi} = 0.00399\ \mathrm{c}^{1.88754}\mathrm{h} \tag{1}$$

where, **wi** is leaf biomass (kg), **h** is total tree height (m); and **c** is tree circumference measured at 1.30 m.

For pine and eucalyptus, the equation is:

$$\mathrm{wi} = \beta_0\ \mathrm{d}^{\beta 1}\left(\frac{\mathrm{h}}{\mathrm{d}}\right)^{\beta 2} \tag{2}$$

where, **wi** is leaf biomass (kg), **d** is DBH (cm); **h** is total tree height (m); $\boldsymbol{\beta_x}$ are allometric ratio coefficients that vary according to the species and component of the tree [63–65]. Using the estimated leaf biomass, we subsequently calculated Crown Bulk Density (CBD, kg m^{-3}) for broadleaves, pine and eucalypt according to the equation:

$$\mathrm{CBD} = (\mathrm{wi}^{*}\mathrm{dt}) \,/\, c\mathrm{h} \tag{3}$$

where, ***c*h** is canopy height and dt is tree density. Table 3 shows the canopy metrics variables.

Table 3
Canopy fuel metrics used in fire behavior simulation: CC (Canopy Cover); TH (Tree Height); CBH (Crown Base Height) and CBD (Crown Bulk Density). See Table 2 legend for details about the fuel models.

Fuel Model	CC (% - average)	Canopy Fuel Metrics		
		TH (m)	CBH (m)	CBD (kg m^{-3})
M-M	90	20.7	9.5	0.09
M-CAD	85	15.6	8.4	0.05
M-PIN	65	12.9	5.2	0.20
M-EUC	65	12.3	6.8	0.23

Table 4
Rate of spread (ROS) adjustments and fire times used in the fire behavior simulations. +Fire times refer to the fire front that reached the 2-km buffer around IZs and MM; *All fires occurred in October 2017, except for the Pedrógão fire that occurred in June 2017.

Study areas	ROS (m min^{-1})	Fire Start	Fire End	Spotting rate	2017 fires*
	Adj	hour (day)+	hour (day)+	%	
Mira	1	21h (15)	00h (17)	5	Quiaios
Tocha	3	19h (15)	03h (16)	5	Quiaios
Oliveira do Hospital	4	17h (15)	04h (16)	5	Arganil
Freixo-Mortágua	8	19h (15)	09h (16)	5	Vilarinho
Oliveira de Frades	8	22h (15)	11h (16)	5	Vouzela
Graça-Pedrógrão Grande	20	20h (17)	03h (18)	5	Pedrógão
Mata da Margaraça	*3*	*03h (15)*	*00h (15)*	*5*	*Arganil*

d) 2017 reconstructed isochrones

To calibrate the fire simulations, we used the reconstructed 1-h fire isochrones [17,18], allowing reproducing fire spread over the landscape surrounding each IZ. This reconstruction used several data sources that allowed identifying the fire arrival time and active firelines to georeferenced locations (Fig. 2). The primary data sources were satellite images (MODIS, VIIRS, Delmos 1, and PROBA V) and fire hotspots from FIRMS allowing the definition of partially burned areas, active firelines, and fire spread direction. Other relevant data sources were used, such as: official reports, face-to-face interviews, photos and videos, field observations, and firefighting operations reports.

Table S6 summarizes the data that was used for reconstructing the isochrones of the five fires analyzed in our study. A brief description of how these fires were reconstructed and their characteristics can also be seen in the supplementary material (pages 7–10). We also added information about their effects on infrastructures and buildings, which was collected following technical visits that were carried out in 2020 to the six IZs [66]. These fires (Fig S1) burned an average of 38,838 ± 16,214 ha, attaining maximum intensities of 30,000–90,000 kW m^{-1} and maximum rates of the spread of 4.5–8.8 km h^{-1} [17,18]. We used the wildfire isochrones to (i) define the start and end of fires in each simulation; (ii) adjust rate of spread (ROS) to reproduce the fire behavior as closely as possible to the events that occurred in 2017 using the adjustment functionality of FARSITE, and (iii) define the spotting rate (Table 4).

As the 2017 fires produced intense spotting [18], we applied a spotting rate of 5% in FARSITE. We identified this rate based on a continuous calibration exercise of the fire model in the Industrial Zone of Mira, which contained observed evidence of fire spotting, through camera records. We compared the fire hour recorded by the camera with the location of the reconstructed isochrones. As the fires of October 15 had similar behavior, since they were all driven by hurricane Ophelia [18], we adopted the same spotting rate to simulate the fires in the broadleaf forest (MM) and in the other IZs. Due to the extremely warm and dry weather conditions of 2017, the June 17 fires were similar in terms of intensity and spread magnitudes. Therefore, due to similar environmental conditions, we assumed the same spotting rate for the Zona Industrial da Graça in Pedrógão Grande. Although we adopted the same initial spotting rate for all study areas, FARSITE adjusts the spotting distance according to local topography and land cover [67].

2.2.4. Fire severity

In the fieldwork (Fig. 2) we also collected data about local fire severity to estimate flame length and fireline intensity inside the MM. These data were also further used to calibrate the simulation in the reference fuel model (M-M) used in the scenarios. For this, we measured the Stem Char Height, which was still apparent in 2021. On average, we found a Stem Char Height ranging from 0.9 m to 2.0 m (Table S3). We first estimated fireline intensity using the average of the Stem Char Height Max and Min, according to Weber et al. [68]:

$$Ib = \exp\left(2.64 + \frac{hc}{0.36}\right) \tag{4}$$

where, *Ib* is fireline intensity and *hc* is Stem Char Height. We subsequently calculated flame length *L* (equals FML-FARSITE), according to Byram [69]:

$$L = 0.0775\ Ib^{0.46} \tag{5}$$

We used a Mann-Whitney-Wilcoxon test to verify the statistical significance of the differences between flame length observations and flame length simulations.

2.3. Statistical analysis

We tested the effects of forest composition (2017 cover vs. replacement by broadleaves), weather conditions (extreme and standard weather) and buffer lengths (100 m vs. 500 m) and their interactions, on two basic fire behavior metrics (Flame Length - FML and Rate of Spread - ROS) obtained from the FARSITE simulations, using Generalized Linear Mixed Models (GLMM). For this, we extracted the average values of these variables, within the buffer lengths of 100 m and 500 m, for each weather scenario and forest composition. Flame length and Rate of Spread were selected because they are standard fire-behavior metrics and express fire suppression difficulty for fire management organizations [70,71]. We used a Gamma error distribution, a log link function, and the IZ as a random effect. The GLMM was implemented with the R package lme4 [72]. The variance explained by the model was estimated using the R package

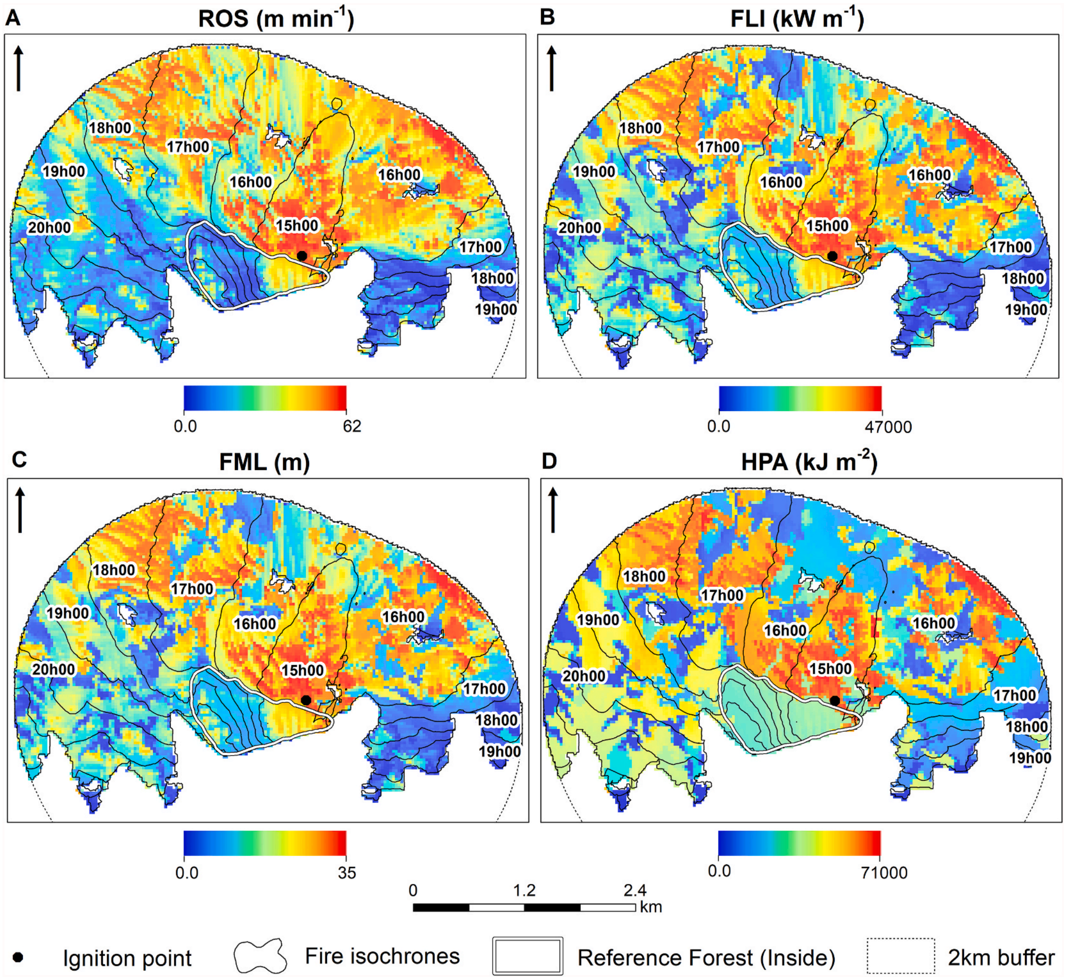

Fig. 3. Simulated isochrones inside and outside the reference forest (MM) and simulated fire behavior metrics for the right flank of the Arganil Fire. ROS: Rate of Spread (m min^{-1}), FLI: Fireline Intensity (kW m^{-1}), FML: Flame Length (m), and HPA: Heat per Unit Area (kJ m^{-2}).

performance [73]. Using the R package car [74], we calculated variance inflation factors (VIF) to determine the final set of potential predictors. We adopted VIF >4 [75] as the threshold for discarding correlated variables. Therefore, we removed the variable buffer length and the interactions between variables since they were not significant in ANOVA tests. Model selection was based on the lowest AIC obtained for competing models [76].

3. Results

3.1. Fire behavior in the broadleaves reference forest

From measurements of fire severity (stem char), an average flame length (FML) of 1.6 ± 0.9 m was estimated for the fire which reached the interior of the reference forest (MM) corresponding to an average fireline intensity of 1,030 ± 1,350 kW m^{-1} (Table S7). Fig. 3-C shows a drastic reduction of the average flame length of 4.3 ± 7.6 m when the simulated fire isochrones advance towards the interior of the forest and decrease to 1.7 ± 3.3 m, similarly to field-based estimates. The Mann-Whitney-Wilcoxon test indicates no significant differences between observed and simulated flame length (Table S8).

Within the reference forest, ROS was 3 times lower than in the forest patches outside this area. Fireline intensity (Fig. 3-B), similarly to field data (Table S8), was almost four times lower, reducing from 4,094 ± 9,066 kW m^{-1} to 1,035 ± 3,783 kW m^{-1}. Despite the low ROS inside the forest, 2.2 ± 5.2 m min^{-1}, a significant increase at its edges is noticeable, with ROS reaching 35.0 m min^{-1} (Fig. 3). This increase was also observed in the other fire behavior descriptors. In general, the simulated isochrones coincide with the reconstructed isochrones (Fig S2). Both indicate a propagation in the N-NW direction, with high ROS in the first hours of the fire (36.4 m min^{-1} on average), between 3:00 p.m. and 6:00 p.m.

3.2. Fire behavior scenarios nearby the industrial zones

We identified important differences between fire behavior in the IZs buffer areas using the fuel models representing the 2017 land

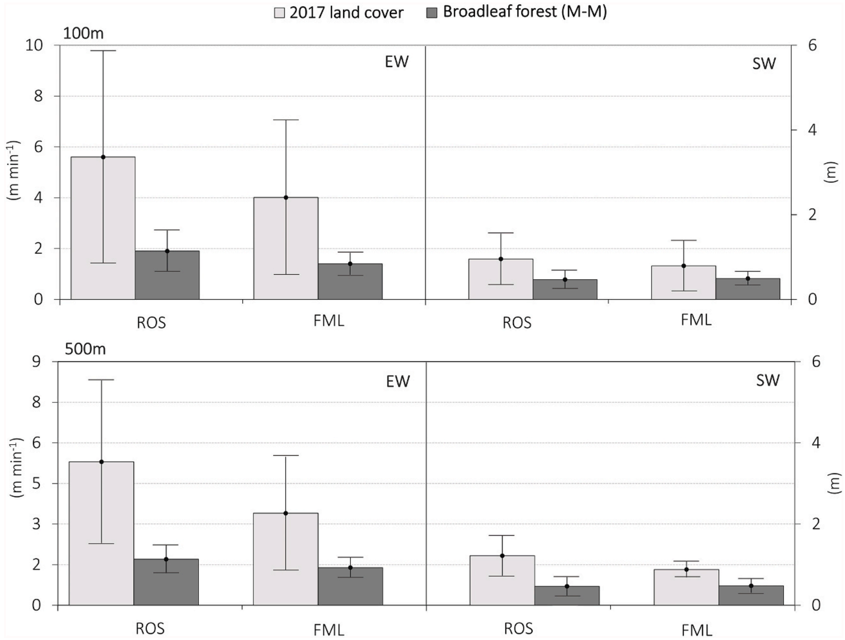

Fig. 4. Mean and standard deviation for rate of spread - ROS (m min^{-1}) and flame length - FML (m) for the six IZs. Includes existing vegetation (2017 land cover - LC) vs. broadleaf forest (M–M) in 100 m and 500 m buffers lengths and in extreme - EW (left) and standard - SW (right) weather conditions.

cover (LC) and the broadleaf scenarios (M-M). This applies to both weather conditions (extreme weather-EW and standard weather-SW) (Fig. 4). Under EW conditions, the average fire rate of spread in all IZs was 3 times lower (2.0 ± 0.83 m min^{-1}) in M-M scenario than in LC (5.5 ± 4.2 m min^{-1}), regardless of buffer lengths (100 m or 500 m). ROS in the 500 m buffer is only 12% lower than in 100 m. In this respect, the reduction in flame length (Fig. 4) was also small. For all IZs, FML decreases by 7% on average when M-M expands from 100 m to 500 m in EW conditions, and 13% in SW. Average ROS in SW conditions is substantially reduced compared to EW conditions. This reduction is up to 3.4 times lower in LC (1.6 ± 0.5 m min^{-1}) and 2.4 times lower in M-M (0.6 ± 0.3 m min^{-1}). Likewise, FML decreases about 2.8 times in LC between EW and SW conditions, whereas in M-M decreases 1.8 times. In LC, values range from 4 m in EW to 1.3 m in SW. In M-M, the average FML is 1.4 m in EW, while in SW, the average FML decreases to 0.8 m.

Despite not being included in our GLMM model, Fig S3 indicates that fireline intensity (FLI) decreases considerably between LC and M-M. In LC, mean FLI reaches 3,399 ± 3,827 kW m^{-1} and can decrease to 608 ± 342 kW m^{-1} in M-M, even under extreme weather conditions. In addition, FLI is reduced by 35% when M-M expands from 100 m to 500 m considering all IZs. The average FLI reduction is even higher from EW to SW, about 5.8 times in LC (560 ± 207 kW m^{-1}) and 3 times in M-M (226 ± 121 kW m^{-1}). The drastic reductions in fire behavior in the IZs of Tocha and Graça-Pedrógão Grande when interfaces are occupied by M-M must also be highlighted. Simulated data indicate that at Tocha the Quiaios fire reached 33,000 kW m^{-1} at the LC interface with a ROS of 33.4 m min^{-1} (Figs S9 and S10). Under EW conditions in the M-M scenario, ROS and FLI decrease respectively by factors of three and nine in

Table 5
GLMM models for ROS (m min^{-1}) and FML (m).

	Estimate	Std. Error	t value	Pr (>|z|)
ROS *(rate of spread)*				
(Intercept)	1.5615	0.1713	9.116	<2e-16 ***
<u>*Land use*</u>				
Broadleaf forest	−0.9615	0.1253	−7.672	1.70e-14 ***
<u>*Weather*</u>				
Standard weather	−0.9788	0.1317	−7.435	1.04e-13 ***
FML *(flame length)*				
(Intercept)	7.7894	0.2326	33.491	<2e-16 ***
<u>*Land use*</u>				
Broadleaf forest	−1.3386	0.1727	−7.751	9.11e-15 ***
<u>*Weather*</u>				
Standard weather	−1.2884	0.1864	−6.912	4.78e-12 ***

Significance codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 '' 1

ROS	Conditional R^2: 0.715	Marginal R^2: 0.655
FML	Conditional R^2: 0.706	Marginal R^2: 0.632

this IZ. In Graça-Pedrógão Grande the fire reached a maximum FLI of 28,300 kW m^{-1} and a ROS of 34.0 m min^{-1} in the 100 m buffer, regardless of cover type (Figs S15 and S16). However, mean values decrease strongly when LC is replaced by M-M for both weather scenarios. The data indicate an average ROS of 11.6 ± 15.0 m min^{-1} in LC and 2.4 ± 3.3 m min^{-1} in M-M. FLI ranged between 805 ± 2,010 kW m^{-1} in M-M and 8,602 ± 10,950 kW m^{-1} in LC.

The GLMM models for ROS and FML explained a fairly high proportion of variance (marginal R^2 values of 0.65 and 0.63, respectively). The very similar conditional R^2 values compared to the marginal R^2 show high consistency among the six IZs. Among the four variables tested, only land cover and fire-weather had a statistically significant effect. As for land cover, the broadleaf forest had negative coefficient both for ROS and FML. A similar negative coefficient estimate was obtained for weather and for both dependent variables (Table 5). See Table S9 for the other results of GLMM models.

Figs S4 to S15 display simulated versus reconstructed isochrones in all IZs and descriptive statistics for all fire behavior metrics (ROS, FLI, FML and HPA). These figures show close similarity between the simulated and reconstructed isochrones, especially in the first hours of the fire. Furthermore, for scenarios that include broadleaf forest cover, an abrupt approximation between isochrones is easily identifiable due to the decrease in ROS when reaching the 100 m buffer zone. This proximity is even higher in the 500 m buffer zone.

4. Discussion

4.1. The potential hazard-reduction role of dense broadleaf forests

The results of our fire simulations indicate that the replacement of current land covers by patches of dense broadleaved trees could substantially change fire behavior in industrial-forest interfaces. Our results also indicate that the fire behavior metrics vary subtly as the buffer zone covered by broadleaved trees increases from 100 m to 500 m, which is consistent with the fact that fire behavior modeling outputs respond gradually to changing conditions. However, we found that these changes, based on the introduction of less flammable fuel complexes in the WUI become more effective under standard fire-weather conditions in mainland Portugal (historical fire-weather conditions) than under severe meteorological conditions. This is line with other findings about the higher selectivity of fire under less severe weather conditions (e.g. Nunes et al. [30]).

Our results demonstrate that fire behavior metrics in our reference forest stand (Mata da Margaraça) were significantly lower than those outside, where fuels are more flammable. Although these results consider previous fires, it is relevant to note some aspects. The first is that fire growth near our reference forest was limited by a fire that occurred a week before (October 06, 2017). Fire spread downslope into the reference forest, which, as shown by the reconstructed isochrones, was encountered by one of the wildfire flanks during its initial growth phase and not by the wildfire head, where rate of spread, fireline intensity, and energy release are maximum [77,78]. Another important point is that this forest is located in an area that favors high fuel moisture, for being at the bottom of a north-facing slope in a mountain area. Despite these dampening aspects, the reference forest shares similar terrain and stand structure features with other documented interceptions of broadleaf deciduous forests by large and intense wildfires in Portugal. In this respect, fire severity mitigation in mature broadleaved stands in northern Portugal has already been reported by Fernandes et al. [34] and Proença et al. [79], always in midslope to valley bottom topographic positions and mainly in the moister aspects (North and West), which comprised respectively 87% and 85% of the sampled locations.

Similar results to the reference forest, under extreme fire conditions, were obtained in the fire simulations carried out across the six IZs when replacing 2017 land cover with the land cover of the reference forest to test differences in fire behavior. These results reflect the structure of the surface fuel complex associated with the broadleaves fuel model and the denser canopy cover of this forest type. Fuel load, for example, is lower in this fuel model and litter is more compacted in a shallower fuel bed compared to the fuel models representing eucalypt and pine stands. The dense canopy cover, in turn, increases shading, allowing lower flammability conditions inside the forest [34,35]. This increases dead fuel moisture content by decreasing solar radiation and ambient temperature and increasing relative humidity [60]. These conditions also limit the development of shade-intolerant heliophytic vegetation in the forest understory reducing the presence of flammable shrubs and fuel strata connectivity [80]. The denser canopy also decreases in-stand wind speed, thus directly decreasing fire spread and intensity, and contributing to decrease evaporation from the surface fuel layer by reducing ventilation by the wind [81,82] and fine fuel moisture loss [83,84].

Several authors concluded that forest type has a marginal effect on fire selection ratios, fire size or fire behavior when compared to the surface fuel complex both at landscape and stand scales, and particularly during fire-weather conditions conducive to large fires (e.g. Barros and Pereira [85] and Fernandes et al. [86]). This may be a combined outcome of higher fuel structure variability within forest type than between forest types [87,88], with fire-resistant stand structures too sparse and random to affect fire growth [89]. However, our results show that dense-cover forest stands of broadleaves could reduce fire behavior, increasing the suppression capacity relatively to the common land use around IZ. These forest patches can interrupt the general horizontal continuity of the flammable surface fuel complexes in the landscape and contribute to increased bottom-up influences, which exert higher local control over fire behavior [86]. In this respect, Silva et al. [31] argue that forest stands with dense canopy can provide local-scale microclimate and fuel conditions that are less conducive to fire spread. Nunes et al. [90] also concluded that dense canopy forest stands of Iberian pines and deciduous oaks have a lower probability to burn. However, the relative contributions of vegetation and fuel characteristics at a landscape scale, fire weather conditions and the distribution of fire ignitions is under debate and appears to be context dependent even within a given fire-prone area [91].

Our estimates of fire behavior in dense canopy forest stands of broadleaved trees are similar to those obtained by Pinto & Fernandes [35]. Despite the higher fuel load and litter depth of the M-CAD broadleaf fuel model used by these authors, the average estimates of FLM and ROS obtained for days of severe fire weather were 1.5 m and 2.0 m min^{-1}, respectively. Our estimates for the flammable forests of maritime pine also resemble the assessments of extreme fires and crown fire potentials by Botequim et al. [92] and by Fernandes et al. [93]. Botequim et al. [92] estimates a maximum ROS of 42.6 m min^{-1} and fireline intensity (FLI) of 30,876 kW m^{-1} in pine stands in the central-west region of mainland Portugal. For the same parameters, the average estimates of ROS and FLI obtained by Fernandes et al. [93] were 44.2 m min^{-1} and 33,927 kW m^{-1}, respectively, in pine stands aged between 20 and 40 years in the northern and central regions. Our results also confirm the fire proneness of maritime pine forests [31,35,86,88,93]. The conversion to broadleaves in the IZs of Tocha and Graça-Pedrógão Grande, mostly occupied by maritime pine, showed a much greater reduction in the rate of spread (2.5 times higher), compared to the other IZs mostly covered by eucalyptus [31].

4.2. Management implications

Challenges to reduce fire damages within the WUI will increase due to climate change [94]. This requires the adoption of well-adjusted public policies promoting fire-smart landscapes and sustainable fire management approaches. Our results show remarkable gains in suppression capacity through changes in shading conditions, canopy cover, forest composition and the corresponding fuel models, when shifting from maritime pine and eucalypt stands with understory, to high-density broadleaved stands with low understory fuel loads. By framing our results within the classification that establishes fire suppression capacity as a function of fire behavior thresholds [95], we identified that wildfires exceeding the suppression capacity in poorly managed forest stands of maritime pine and eucalypt (FML>4 m) can be effectively suppressed in high-density broadleaf forest stands, under extreme fire-weather conditions (FML = 1.4 m) and easily suppressed in this same forest type under standard weather (FML = 0.8 m).

In this context, the ecological restoration treatments based on the use of broadleaf forests appear to be more promising to balance fire prevention and suppression efforts allowing for a well-suited landscape management aiming to mitigate fire damage around IZs [96]. This is one of the main challenges that Portugal faces today, in reducing the effects of extreme wildfires [97]. The increasing difficulty in fire suppression, in a large part of mainland Portugal, clearly demonstrates that the current prevention policy, based on fuel reduction, despite the good local effects [98], is limited on other scales and operationally expensive [99]. In addition to the territory's biophysical conditions that prevent its generalization, the landscape changes induced by the increasing agricultural abandonment and the decrease in the land use intensity in most of the large forest patches resulting from the past afforestation, increased fuel loads at patch and landscape scales [96] demanding recurrent fuel reduction treatments [43,44]. In this respect, Aparício et al. [98] found that current fuel management is insufficient to reduce the spread of large fires, unless this policy is supported by additional strategies and integrated with ecological restoration treatments to reduce the intensity and size of wildfires.

The current fire prevention policy also establishes a width of 100 m for fuel breaks. This severely limits any further action to protect the forestry-industrial interface against mega-fires, which commonly result in burned material being sent over long distances. In order, to protect the IZs, the fire management, therefore, must consider wider fuel breaks, contrary to current legislation, and include reforestation actions with fire-resistant tree species, capable of preventing the projection of embers [98,100,101]. Our results indicate a reduction trend in fire behavior for wider management buffers covered by broadleaves, in most IZs (e.g., from 100 m to 500 m). Among the evaluated metrics, fire intensity stands out for its considerable reduction in wider management areas, specifically, where the pine fuel model was replaced by broadleaves. Our study, therefore, supports the need to extrapolate land use with broadleaf forests from small WUI areas to wider areas. In addition to being a strategy to reduce spotting rates due to potential to mitigate canopy fires, this forest type has numerous ecological benefits [102].

Despite the advantages of conversion to broadleaves to mitigate fire hazard, little is known about the cost-benefit ratio of this fuel management option [36] in comparison to other more common fuel management strategies (e.g., prescribed fire). Many authors have addressed the cost-effectiveness of different fuel management methods for protecting WUI areas [103–106] and other assets [104,107–109] but this type of assessment has deserved little attention so far in regard to the conversion to less flammable forests. Nonetheless, changes in land cover to less flammable forest types and the maintenance of these forest patches in the Euro-Mediterranean context can be fostered by policies framed within the EU Green Deal [40], such as the EU strategies for the Bioeconomy [110], Biodiversity [39], Adaptation to Climate Change [111], Forest [112] and Soil [113]. Financing instruments or schemes designed to enhance the provision of ecosystem services (e.g., Pais et al. [24] and/or the generation of carbon credits (e.g., Alcasena et al. [114] could leverage changes in line with the objectives of the aforementioned strategies. Some authors argue that incentive policies or socio-environmental funds (e.g., REDD+ [115] REACT-EU [116]) instituted to guarantee the maintenance of ecosystem services are more promising to achieve sustainable and resilient landscapes [117]. However, forest cover changes for fire risk management can also be strongly encouraged by programs that promote Rural Development (RDPs) [118] and by European investments for research and innovation (e.g., Horizon 2020 (2014–2017) and LIFE programs [20]).

In this context, additional research initiatives are needed to support these policies and stakeholders in Portugal. For this, we recommend, for example, cost-effectiveness assessments that compare forest conversion to broadleaves forest stands versus the traditional management based on fuel reduction. We also consider important, an economic assessment of a going-forward versus a going-backward (fuel reduction) policy in terms of the ecological succession. Going backward is the most common solution because it provides fast results, but vegetation grows again, whereas going forward towards a mature forest takes time but may be more sustainable in the long-term since shade prevents the regrowth of surface vegetation, particularly in Mediterranean-type regions [119]. Although these approaches are outside the scope of this work, we reinforce the importance of these studies in the future.

We also emphasize that although our analyzes are limited to small samples of WUI areas in Portugal, the methodology proposed here can serve as a reference to assess fire hazard and to design fuel management programs in other countries with similar vegetation and climate, mainly in southern Europe [28,120]. Our findings are supported by the extant evidence about the ability of deciduous forests to break the continuity of flammable fuels and fire spread in the landscape and thus their potential for reducing fire behavior by creating strategically-located patches [27,121]. Our study, therefore, contributes to overcoming a knowledge gap on the effectiveness of fire-smart mitigation strategies at the WUI level and to the development of a new fire management paradigm in the Mediterranean region. In this region, traditional fire exclusion policies still prevail, contributing, together with agricultural abandonment and the decrease of land use intensity in forest patches, to the accumulation of flammable fuels at landscape and patch scales and, consequently, to the increase of conditions conducive to extreme fires that threaten people and goods in the near future [24,122,123].

4.3. Limitations and assumptions

We acknowledge some limitations in our fire behavior simulation and assumptions.

(I) Although FARSITE is used extensively for both research and practical purposes, the software may have estimated fire rates of spread (ROS) with little accuracy. This is because FARSITE has limited ability to track embers in complex terrain, associated with significant spotting, and under extreme wind situations [67], and the used data spatial resolution (25 m x 25 m) also prevents the manifestation of finer effects that can affect the rates of spread [54]. Our simulations were also limited by the lack of more accurate data. In particular, the weather data did not precisely reflect the actual wind speed (hurricane Ophelia), neither the changes in micrometeorology that occur under different types of forest [35]. On the other hand, our data on fuel models and canopy cover did not entirely reflect the exact land cover conditions that existed in 2017 around the six IZ. However, we partially overcame these limitations by calibrating the fire simulations using the reconstructed isochrones from the 2017 fires. This calibration resulted from a direct comparison between the reconstructed and simulated output data in terms of fire direction, distances between isochrones and the extent of fire. From this comparison, it was possible to best reproduce the behavior of the 2017 fires by adjusting the rate of spread (adjustment from FARSITE) and the spotting rate.

(II) Our study is based on fuel models and, therefore, does not address the current and future ecological limitations for the establishment of broadleaf forests, the mechanisms of competition between species and the degree of human intervention required in forest restoration actions to achieve the intended forest/fuel structure. These restoration actions have important economic costs [124] that are not addressed in our study.

(III) We also recognize that it is not realistic to think that we can simply transpose our reference forest (Mata da Margaraça) covered by a dense canopy cover forest of broadleaved trees into the buffer zones around the six IZs, located in areas with very different soil, topography and climate characteristics. In fact, as previously reported, the Mata da Margaraça forest is located in an area that favors high fuel moisture, for being at the bottom a north-facing slope in a mountain area. However, our main premise is based on the fact that native broadleaves, particularly *Quercus* spp., would have good conditions to be established in all six IZs according to the Portuguese biogeographic units [125] and potential vegetation map [126]. In some cases, we may consider the use of silvicultural techniques and operations (e.g., individual protection of seedlings and saplings) that allow overcoming the limiting factors for tree growth. This approach would have unaffordable costs in other land use contexts, but may be admissible in small-scale areas that promote additional functions valued by the users of urban forest parks, such as recreation and climate amenity.

(IV) Although the WUI areas (100 m and 500 m buffers) of the six IZs are predominantly covered by pine and eucalyptus forest stands (between 70% and 99%), there were areas with other land uses not so appropriate for conversion such as the case of agricultural land. By replacing this mosaic of land uses with a single land use made up of broadleaf forests we did not take into consideration this limitation. However, selecting the land uses to be replaced by broadleaf forests and leaving out the agricultural land, would very likely result in an even higher reduction of fire behavior because agriculture has in general a very low fire proneness [30,127].

Despite the above limitations, we believe our results still provide valuable information for fire mitigation in WUI areas. Our simulations were carefully calibrated to reproduce the fires of 2017 and show an alternative land use that appears to be more advantageous to reduce the spread of fire, compared to the land uses existing around the IZs in 2017.

5. Conclusion

Our results suggest that dense canopy cover patches of broadleaved trees have the potential to mitigate fire risk and contribute to preventing disasters in WUI areas in Portugal, even under extreme weather situations. The broadleaf forest model and cover we tested, were capable of significantly reducing fire behavior magnitude in six industrial zones in Central Portugal, suggesting positive effects and a potential mitigation approach. In this context, current issues surrounding the expansion of flammable forests (eucalypts and pines), as well as funding limitations for efficient fuel management, are crucial to understanding the risks that forest producers will face under climate change. Thus, our work suggests new prevention actions that involve the use of broadleaf forests, as green belts around infrastructures, accompanied by clear definitions of the direct and indirect benefits of this type of management. This requires, for example, the creation of economic incentives that can stimulate the interest of land managers about this forest type. However, we understand that decision-making in this regard can be better supported by assessments that compare the cost-benefit and feasibility of a forest restoration project that includes the use of broadleaf species versus the current legislation that promotes the traditional management of fuel reduction around infrastructures, namely the IZs. Although this approach is outside the scope of this work, our results reinforce the importance of this research in the future.

Overall, our work demonstrates the mitigation potential of broadleaf forests, which in the future may help to avoid disasters in one of the countries that suffer the most from wildfires in the world. For the moment, little has been done in terms of forest legislation, for the strategic adoption of this option in the Portuguese territory. Given the impossibility of changing extreme weather conditions and given the complexity and uncertainty surrounding the occurrence and severity of mega-fires, we fear that the catastrophic year of 2017 will be repeated in the future if new solutions are not found for preventing fires.

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.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ijdrr.2023.103788.

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