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Managing Wildfire Risk in Mosaic Landscapes: A Case Study of the Upper Gata River Catchment in Sierra de Gata, Spain

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Abstract: Fire prevention and suppression approaches that exclusively rely on silvicultural measures and containment infrastructure have become increasingly ineffective in stopping the spread of wildfires. As agroforestry landscape mosaics consisting of a mix of different land cover and use types are considered less prone to fire than forests, approaches that support the involvement of rural people in agriculture and forestry activities have been proposed. However, it is unknown whether, in the current socio-economic context, these land-use interventions will nudge fire-prone landscapes towards more fire-resistant ones. We report on a case study of the Gata river catchment in Sierra de Gata, Spain, which is a fire-prone area that has been a pilot site for Mosaico-Extremadura, an innovative participatory fire-risk-mitigation strategy. Our purpose is to assess the efficacy of project interventions as "productive fuel breaks" and their potential for protecting high-risk areas. Interventions were effective in reducing the flame length and the rate of spread, and almost 40% of the intervention area was in sub-catchments with high risk. Therefore, they can function as productive fuel breaks and, if located strategically, contribute to mitigating wildfire risk. For these reasons, and in view of other economic and social benefits, collaborative approaches for land management are highly recommended.

Keywords: wildfire; risk assessment; agroforestry; landscape approach; forest management; firebreak

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1. Introduction

For millennia, humans have successfully controlled and used fire in our efforts to domesticate forest ecosystems and landscapes for our own benefit [1,2]. Fire is an effective and efficient tool used to enhance soil fertility, to alter the structure and the composition of forest vegetation, and to clear land of unwanted woody biomass [3,4]. Thus, fire regimes (i.e., the size, frequency, intensity, and impact of fire) [5] have been generally controlled in domesticated forest ecosystems. However, in the 21st century, the tide is turning: wildfires—unplanned, uncontrolled, and unmanageable fires—are becoming more pervasive, ravaging forests and causing economic losses, environmental damage, and the loss of human lives. Forests now seem to be not human-dominated [6] but fire-ruled ecosystems.

1.1. New Forests, Shifting Fire Regimes

Novel fire regimes are the consequence of a broad set of strongly interlinked climatic, socio-economic, and environmental processes, occurring at a range of spatial and temporal scales [7–9]. Globally rising temperatures, and longer and more frequent droughts resulting from climate change have made forest biomass more predisposed to burning and changed the seasonality and duration of fire seasons in temperate and boreal regions [10]. Wildfires have thus become more frequent and devastating in certain parts of the world, particularly in regions with fire-prone forests, such as the African savanna and Mediterranean climates, and even in the Amazon [11–15].

In Europe, dead and live forest fuel is nowadays more abundant, after more than a century of forest expansion and biomass buildup [16,17]. In mountainous areas, forests have become denser and natural regeneration has encroached onto formerly cropped and pastured land following the abandonment of traditional activities such as occupational burning, grazing, fuelwood collection, and wood harvesting [18,19]. During the 20th century, afforestation and the establishment of industrial tree plantations were energetically pursued, in some countries on a massive scale. Between 1940 and 1984 in Spain, 3,678,522 ha of land (nearly 14% of the current forest land) were afforested, most of it with flammable pine and eucalypt species [20]. Between 1990 and 2015, in Europe, the area covered by forests and woodlands increased by 90,000 square km—an area roughly the size of Portugal [21]. Today, about 42% of Europe's land area (182 million ha) is covered by forests [22]. Biomass fuel has been steadily accumulating in these new forests, after decades of fire suppression and increased tree mortality, due to climate change and pest outbreaks [23–26]. The increase in litter, fine fuels, large dead woody fuels, and fuel ladders due to infestations have increased the probability of burning [27,28].

The wildfire problem has been recently aggravated by new developments and demographic trends that have made forest areas more vulnerable to fires. Expansion of the road network, growing urbanization, and tourism and recreational activities in forests have increased the probability of ignitions and made fire prevention and suppression less tractable [29–31]. Forest roads may be key for controlling wildfires as they act as firebreaks and facilitate access for firefighting crews for fire suppression activities [32–34]. However, forest roads may also be a conduit for accidental ignitions, especially in those areas with natural or seminatural forests [35–38]. Growing urbanization in forested landscapes that were formerly managed for agrosilvopastoral activities is transforming the forest–agriculture interface (FAI) into an unmanaged wildland–urban interface (WUI) more exposed to wildfires, due to the increased potential for human ignitions [39]. Spain, which has experienced an average of 12,500 forest fires per year over the past decade, has 1.1 million ha of WUI areas, which represents more than 4% of the total forested land [40].

1.2. Socio-Economic and Environmental Impacts of Wildfires

Wildfires have serious economic, social, and environmental impacts. Wildfires cause direct and indirect economic losses in forestry, agriculture, tourism, infrastructure, private property, and public health [41]. Between 2000 and 2017, in the European Union (EU), wildfires are estimated to have caused EUR 54 billion of economic losses—roughly EUR 3 billion a year on average [42]. As the WUI grows, fire suppression expenditures are at an all-time high, with Greece, France, Italy, Portugal, and Spain together dedicating EUR 2500 million each year to fire detection and suppression [43]. In the past decade, in southern Spain, the suppression costs of large forest fires increased by between 65.67% and 86.73%, and the economic susceptibility to fire of timber and non-timber assets is expected to increase sharply in the near future [44]. Despite increasing firefighting capacity, wildfire fatalities remain a significant problem in Mediterranean Europe and elsewhere. Between 1945 and 2016, wildfires caused 865 fatalities in Greece, Sardinia (Italy), Spain, and Portugal, with a major rise in fatalities in the late 1970s in these four countries [44]. Between 1980 and 2010, in Spain, wildfires caused 241 deaths, of which 169 were firefighters [45]. In 2017, in Portugal, wildfires burned a record area of about 500,000 hectares and caused more than 120 fatalities [46].

Growing scientific evidence suggests that wildfires are making a significant contribution to the volume of greenhouse gases (GHG) in the atmosphere, transforming forests from carbon sinks into sources [47]. Unprecedented wildfires in the Arctic Circle during the summer of 2019 emitted 50 megatons of carbon dioxide per month. When biomass burning from small fires was accounted for, carbon emissions increased by 35% (from 1.9 Pg C yr^{-1} to 2.5 Pg C yr^{-1}) on a global scale [48]. Wildfires combined with a warmer climate might also increase nitrous oxide emissions [49].

Wildfires are also a major threat to biodiversity and may cause irreversible damage to conservation areas and other natural areas [50,51], although their impact on biodiversity is not yet fully understood.

1.3. Approaches to Fire Risk Management

For much of the 20th century, approaches to fire risk management have exclusively rested on fire prevention and suppression. Silvicultural practices such as thinning, pruning, and undergrowth clearing are conducted to reduce the load and break the horizontal and vertical continuity of fuel. Fuel breaks (i.e., a wide strip or patch of land cleared of flammable fuels) are established at strategic locations such as roadsides and mountain ridge lines, to stop or reduce the spread of fire and assist firefighting crews in containing and controlling fires. In addition, look-out towers are set up on mountain and hill tops to guarantee early detection and location of ignitions for rapid intervention and suppression by professional firefighting crews.

Prevention and suppression approaches can effectively reduce fire activity, especially when sufficient firefighting resources are available, including aerial resources [52]. However, novel fire regimes are creating high-intensity wildfires that are well above extinction capacity as a result of increasing risk factors, making silvicultural measures and fire-containment infrastructure increasingly ineffective in stopping the spread of fire. A study on the effectiveness of fuel breaks in the control of large fires in California showed that fires stopped at fuel breaks only 46% of the time when firefighters had safe access for performing suppression activities [53]. Firefighters reported that the primary reasons for fires crossing the fuel breaks were scarce technical and human resources, especially when fires were large, unpredictable fire behavior due to wind shifts, and lack of maintenance of the fuel breaks [53]. In extreme weather conditions, strong winds result in simultaneous, multiple ignitions due to long-distance ember cast, making fire behavior unpredictable due to swift shifts, precluding ground engagement and aerial support [54]. In the megafire of 1998, in Yellowstone, burning embers were carried 1 to 2 miles ahead of the main fire front, initiating new spot fires, and large, natural firebreaks such as the Grand Canyon of Yellowstone did not impede the spread of late-season fires [55].

By the late 20th century, recognizing the complexity of forests and of the fire problem, countries that for many years had invested in large and expensive fire detection and suppression strategies, such as the US, Canada, and Australia, began to develop integrated fire management (IFM) approaches. Based on the recognition of fire as an ecological process with both positive and negative effects, and on knowledge about its behavior and characteristics, IFM aimed to make judicious use of fire as a tool for land management and risk reduction [56]. These approaches basically consist of two strategies: (a) managed wildfires (i.e., allowing wildfires to burn under certain conditions) and (b) prescribed burning, with or without mechanical fuel treatments [57].

IFM approaches are effective in decreasing wildfire ignitions, controlling the behavior and severity of wildfires, and reducing suppression resources requirements [57]. Patch-burning techniques that recreate mosaics of burnt and unburnt areas to decrease the probability of large wildfires have been successfully used in Brazilian and Australian savannas [58,59]. However, IFM approaches also present important technical, managerial, and logistical issues. Prescribed burning requires large crews of qualified professionals and technicians, and substantial resources for planning and implementation. Where fuel recovery is rapid, frequent burning at 2-year intervals may be necessary, which may make prescribed burning unfeasible for forestry agencies with limited budgets and staffing [60]. Negative public opinion, especially in the proximity of residential developments or urban areas, environmental laws regulating air quality and smoke, and risk-averse forestry agencies and policies are also major impediments to the widespread use of prescribed burning [61]. Nevertheless, IFM is probably the most cost-efficient option for forest restoration and fuel management in extensive, unpopulated forest areas such as those of the western US and other countries.

IFM has been recently proposed as a conceptual framework for a new directive aiming to address the wildfire issue in Europe [62]. However, in the domesticated forest landscapes of Europe, with their small-sized forests finely intermixed with small-scale agricultural systems, heritage landscapes, protected areas, and numerous rural and urban forest users with different demands, values and perceptions, the use of fire as a land management tool may be undesirable and its benefits may not be fully appreciated. In these situations, in which people still play a significant role in land use and depend on forests for resources and as habitat, forests are better viewed as socio-ecological systems (SES) [63,64]. Viewing forests as SES requires consideration of their diverse sets of coevolving natural and social subsystems and their multiple feedbacks, time lags, and cross-scale interactions [64]. In SES, stakeholders and social networks are considered to play a large role in determining the system behavior and the responses of agents and ecosystem services to the system dynamics [65]. Therefore, in SES, social disturbances can be as important as ecological ones [66], and fire is a social process, an element that is "culturally framed and transmitted", as much as a natural phenomenon [67] (p. 142), and wildfire is more than a technical problem, it is a wicked one [68].

Wicked problems are characterized by complexity, a high degree of scientific uncertainty, and deep disagreement on values, perceptions, and attitudes among key stakeholders [69]. In this type of problem, it is much more complex and challenging to identify the specific drivers causing the problem and the best solutions. Thus, wicked problems have no single correct formulation and no clear-cut, optimal solution but only more or less useful ones (i.e., feasible and acceptable solutions) [70,71]. Approaches for addressing wicked problems include adaptive management, multisector decision-making, institutions that enable management to span administrative boundaries, markets that incorporate natural capital, and collaborative processes to engage diverse stakeholders and address inequalities [64,72].

In the past few decades, novel collaborative approaches and strategies have been proposed and tested in different geographical contexts to address the breadth of environmental challenges that fall into the category of wicked problems, including forest management planning, biodiversity conservation, landscape restoration, and wildfire [73–77]. Collaboration is the process whereby public agencies, stakeholders, and citizens pool resources and work together to solve problems, resolve conflict, develop a shared vision, or manage a common resource [74,78]. Intrinsic to collaboration is the concept of stakeholder participation, a fundamental element of all aspects of collaborative management from resource assessment and problem diagnosis to planning, monitoring, and evaluation [79].

Stakeholder participation presents a number of potential advantages in addressing the challenges posed by wildfires: (i) it helps to develop a comprehensive understanding of the problem and leads to higher-quality decisions due to the consideration of multiple perspectives; (ii) it enables interventions and technologies to be better adapted to local sociocultural and environmental conditions, and enhances their rate of adoption and diffusion among target groups; (iii) it increases the knowledge that stakeholders have about each other's actions, which may lead to the development of trust; (iv) it greatly facilitates collaboration and monitoring of stakeholder behavior; (v) it helps to increase public trust in decisions made by public administrations; (vi) it may make research more robust by providing quality information, and it aids stakeholder empowerment through the co-generation of knowledge with researchers and by increasing participants' capacity to use this knowledge; (vii) it promotes social learning through the development of new relationships, by building on existing ones and by helping participants to appreciate the legitimacy of each other's views; and (viii) it reduces the likelihood of marginalizing those on the periphery of the decision-making context or society [79,80]. Therefore, public institutions responsible for wildfire governance may benefit from adopting collaborative approaches, underpinned by strong participatory processes, to manage wildfire risk [81,82].

Over the past several decades, in Spain, various forms of collaboration among diverse stakeholders (i.e., farmers, shepherds, forestry technicians, and public institutions) have showcased the potential of collaborative approaches for addressing the problem of wildfires. As early as 1986, the Regional Government of La Rioja started the Plan for Shrub Clearing (PSC), which supported farmers to conduct shrub clearing, followed by livestock grazing, to control fires and improve the management of abandoned land in mountain areas. A recent assessment of the program concluded that it has been successful in reducing the number of fires and burned areas, as well as generating improved ecosystem services from mosaic landscapes [83]. Another successful program is RAPCA, the Andalusian network of grazed fuel breaks. This is an example of an effective payments for environmental services (PES) program. RAPCA currently involves 220 local shepherds who, with their flocks, control biomass in almost 6000 ha of fuel breaks in public forests [84]. Similarly, in the region of Valencia in eastern Spain, a program which ran between 1996 and 2009 provided financial support to farmers who concentrated their livestock on firebreaks for a minimum of 130 days per year, with a minimum stocking rate of either one cow, three goats, or five sheep per hectare. Under this system, 3680 ha of firebreaks were grazed in 2009, with the collaboration of 62 farmers. Unfortunately, this scheme has since been cancelled due to a shortage of funds [85].

A number of studies have also described, developed, and tested strategies, methods, and tools to facilitate collaborative fire risk mitigation in Spain, including: (a) methods for incorporating local knowledge and social values regarding landscape in both suppression and prevention planning [86]; (b) social learning and associationism among forest owners, as a strategy for cooperation and for creating socio-ecological structures that are less vulnerable to fire [87]; (c) priority maps as a tool for the implementation of diverse management options, restoring cultural fire regimes, facilitating safe and efficient fire response, and creating fire-adapted communities [88]; (d) social, institutional, and ecological transformations (e.g., reorganizing the fire department, or changing the philosophy of fire prevention services and dominant social values) required to reshape the wildfire management system and learning to coexist with fire [73]; (e) methods of scenario analysis that take into account social values, organizational resilience, and landscape resilience to help build resilient emergency response systems, explain risks to society, and involve citizens in the decision-making process [89]; and (f) a method for the participatory integration of qualitative local stakeholders' knowledge with expert GIS fire simulations [90].

Sierra de Gata is a mountainous area in southwestern Spain prone to anthropogenic fires. Since 2016, after a wildfire that burned through nearly 8000 ha and forced the evacuation of three villages in the upper catchment of the Gata river, a collaborative land management approach, the Mosaico-Extremadura (Mosaico) project, has been implemented to restore a mosaic landscape less prone to fire [91]. The project rests on the idea that agricultural and forestry activities (i.e., tree-crop plantations, livestock grazing, forest product harvesting) implemented by farmers will function as effective and productive firebreaks that will reduce fire risk [92]. Evidence that agroforestry landscapes with a diversity of land uses are less prone to wildfire than forests, shrublands, or grasslands, and that grazing in woodlands and firebreaks is effective in preventing fuel accumulation support this idea [93–98], although other studies have reached different conclusions [99,100].

The purpose of this study is to determine the potential of the interventions promoted by the Mosaico project to nudge the studied agroforestry landscape towards a less fire-prone one. We consider the following questions. (1) Do Mosaico-promoted interventions function as effective firebreaks? If so, to what extent do interventions influence fire behavior and methods of extinction attack? (2) Are the interventions implemented in high-risk areas?

To answer these questions, a relative fire risk index was developed and simulations of the fire behavior in the existing landscape in 2010 were conducted under two scenarios: before and after the implementation of Mosaico. We selected 2010 as the reference year before the wildfire of 2015, as it was the year for which the most complete set of data for the area studied was available.

2. Materials and Methods

2.1. Study Site

The study was conducted in the upper catchment of the Gata river, located in the Sierra de Gata mountain range, Cáceres province, southwestern Spain. The catchment has a total area of 15,092.4 ha, which can be subdivided into 3 different zones: the sub-catchment of the Acebo river (zone A, 7910 ha), the sub-catchment of the Gata river (zone B, 6987 ha), and a small sub-catchment to the south of the confluence of these rivers up to the tail end of the Gata reservoir (sub-catchment C, 194 ha) (Figure 1).

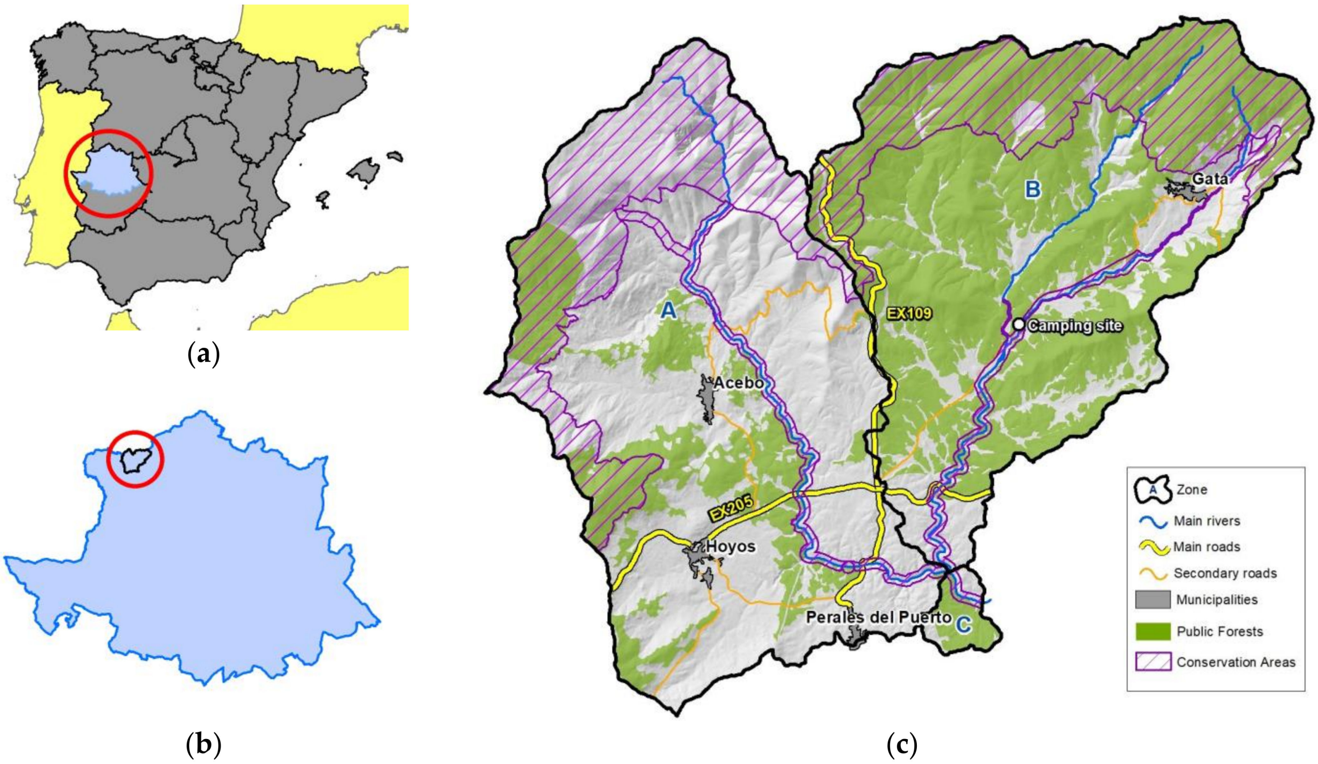


Figure 1. Overview of the study site: (**a**) Cáceres province; (**b**) location within Cáceres province; (**c**) study site.

In 2020, the study site comprised a total population of 3226 in four municipalities: Acebo (557), Gata (851), Hoyos (880), and Perales del Puerto (938) [101]. There is also a popular camping site, open all year round. The studied area is traversed by the roads EX109 in a north–south direction and EX205 in an east–west direction.

The site has a mountainous topography, with 40% of the area having slopes greater than 30% and 10% of the area with slopes above 50%, and an elevation ranging from 339 to 1487 m above sea level. It is characterized by a temperate, subhumid, Mediterranean climate with an average annual rainfall of 1336 mm for the period 1961–2003 (weather station "Hoyos-3536", 6°43′ W 40°10′ N). Average temperatures for the coldest and hottest month are 6.2 °C and 23.1 °C, respectively.

Around 85% of the study area is classified as forest land and 13.5% as agricultural land, while the rest includes urban areas and water bodies. Zone A of the study site was the most affected by the wildfire of 2015, which destroyed around 4800 ha of *Pinus pinaster* forests, shrubs, and grasslands. Nowadays natural regeneration of pine trees and shrubs is rapidly revegetating the affected area. Unburned forest areas are found in zone B of the study site (around 3800 ha of *Pinus pinaster*) and in the southwestern quadrant of zone A (1400 ha of *Quercus pyrenaica*). The rest of the forest area is composed of scrublands (700 ha), patches of shrubs and bare soil (550 ha), and pasture lands (about 70 ha) [102]. Olive plantations predominate on the agricultural land, followed by vineyards and fruit-tree orchards [103].

There are 11 public forests in the study site, covering more than 7500 ha. A total of 4300 ha of the catchment are within the designated conservation area ZEC ES4320037 "Sierra de Gata", and 583 ha of the gallery forest of the two major water courses are designated as conservation area ZEC ES4320076 "Rivera de Gata y Acebo".

Sierra de Gata is prone to forest fires. Important fires at the study site in the last 20 years occurred in 2003 (Acebo, 2360 ha), 2005 (Gata, 732 ha), 2012 (Gata, 677 ha), and the latest and largest in 2015 (Acebo, 7832 ha) [104].

2.2. Description of the Mosaico-Extremadura (Mosaico) Project

The Mosaico project was conceptualized in response to a wildfire that, in the summer of 2015, burned nearly 8000 ha, mostly in the municipalities of Acebo, Hoyos, and Perales del Puerto. The project aims to establish, through a collaborative land management approach, a mosaic landscape consisting of a mix of different land cover and use types, which is more resistant to wildfires than forests.

The project rests on two assumptions: (a) that the involvement of rural people in agricultural and forestry activities is key to fire prevention, as interspersed patches of crops, pastures, and grazed shrubs and forests break up the continuity of hazardous fuels across the landscape and provide safer zones from which fires can be suppressed; and (b) that the economic opportunities generated from farming and forestry activities in mosaic landscapes contribute to reducing the ignition of fire as a form of protest against restrictive and punitive policies common in centrally managed forest landscapes.

The project staff is composed of forestry, agriculture, and livestock technicians who have the following main responsibilities: (1) to provide advice and technical assistance to local stakeholders (farmers, shepherds, landowners, entrepreneurs, and NGOs) on the development of agricultural, livestock, or forestry projects (hereafter interventions) such as commercial tree-crop plantations, forest grazing, and resin tapping; (2) to facilitate dialog and collaboration between promoters of interventions and the regional Forest Service; and (3) to organize training and information dissemination activities [91].

2.3. Data Collection

In October 2020, the project database, containing detailed information on the interventions (proponents, location, type, area, management plan, etc.) and their implementation status was consulted for data collection. Interventions deemed unfeasible due to legal (e.g., requiring change from forest to agricultural land use), financial, or other important impediments were not considered. Thus, a total of 23 interventions were selected for the study, covering a total of 732 ha (i.e., 5% of the study area), of which almost 76% corresponded to forestry, 18% to livestock, and only 7% to agricultural interventions. Nearly 52% of the intervention area is public land and 48% is private land. A complete list of the interventions selected is given in Table 1.

Of the nine agricultural interventions, seven were in zone A, covering 96% of the total area under this intervention type. Zone B of the study site contained only eight interventions, most of them of the forestry type, but these covered 60% of the total area under interventions (439.1 ha) (Figure 2).

Table 1. Characteristics of interventions selected for the study.

Type of Intervention	Id Intervention	Ownership Status	Description	Area (Ha)
Agricultural	A-01	Private	Chestnut plantation	2.0
	A-02	Private	Almond and olive-tree plantation	8.4
	A-03	Private	Fruit plantation with apiculture	1.1
	A-04	Private	Olive-tree plantation	0.9
	A-05	Private	Red berries plantation	0.4
	A-06	Private	Mixed olive and chestnut plantation	0.8
	A-07	Private	Chestnut, almond, and pistachio plantation	26.3
	A-08	Private	Chestnut plantation	1.3
	A-09	Private	Almond and pistachio plantation	8.4
Forestry	F-01	Private	Thinning and mechanical clearing in chestnut forest	89.2
	F-02	Private	Thinning and mechanical clearing in forest farm	138.7
	F-03	Private	Mechanical clearing and tree planting	14.0
	F-04	Private	Thinning and mechanical clearing in chestnut forest	1.3
	F-05	Public	Thinning and mechanical clearing in resin tapping areas	172.7
	F-06	Public	Thinning and mechanical clearing in public forest	108.9
	F-07	Private	Thinning and mechanical clearing in chestnut forest	3.5
	F-08	Public	Mechanical clearing and tree planting in public forest	24.8
Livestock	L-01	Private	Sheep grazing in agroforestry farm	12.3
	L-02	Private	Sheep grazing in shrub lands	42.5
	L-03	Public	Targeted grazing in public forest land (BOCA)	28.0
	L-04	Public	Targeted grazing in public forest land (BOCA)	14.0
	L-05	Public	Targeted grazing in public forest land (BOCA)	31.2
	L-06	Private	Goat grazing in agroforestry farm	1.3

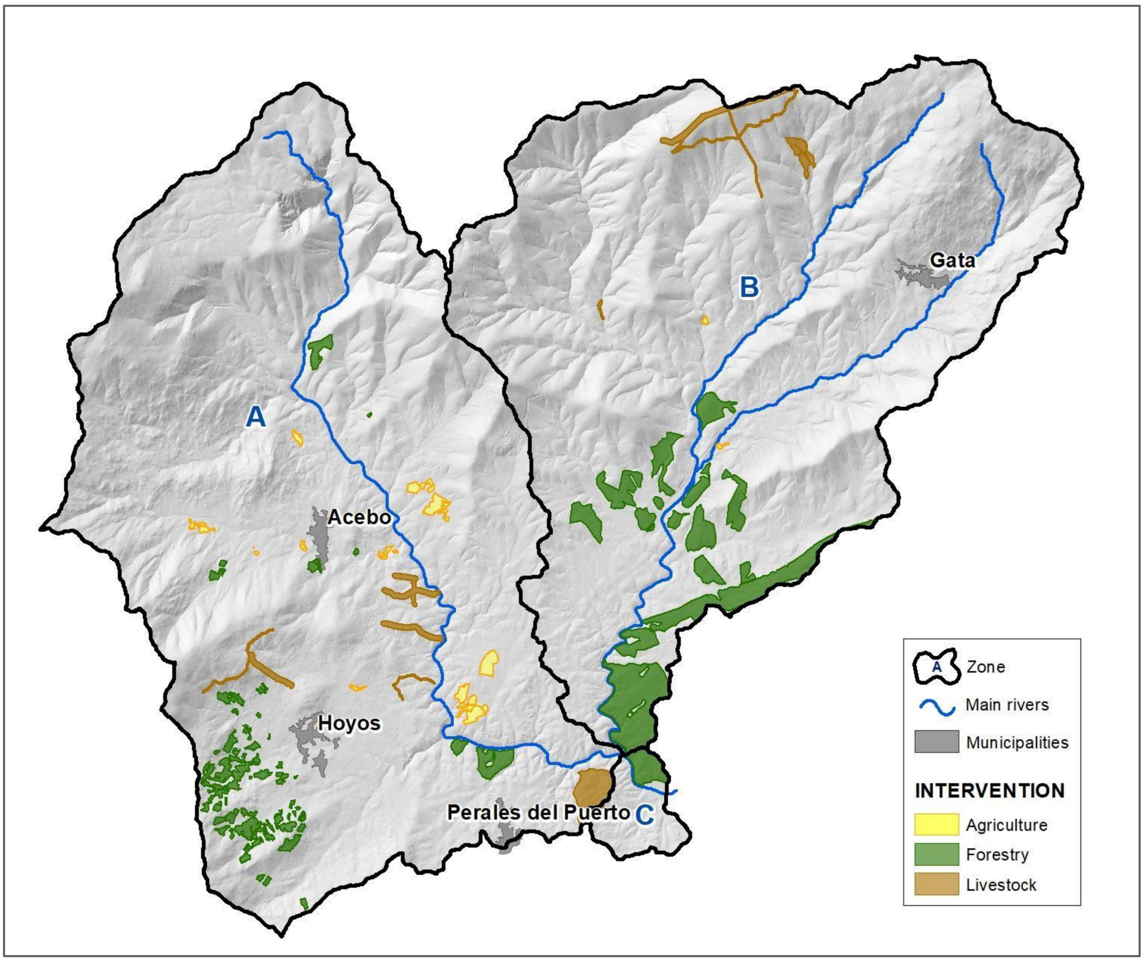


Figure 2. Overview of interventions at the study site.

2.4. Data Analysis

Simulations of surface fire behavior with and without interventions were performed using FlamMap software [105]. Flame length and rate of spread, which are both variables directly related to fire extinction capacity, were calculated under two climatic scenarios: L10 and VL35 (Table 2). Fuel moisture conditions were based on [106].

Table 2. Conditions of climatic scenarios used in the simulations of fire behavior.

Parameter	Climatic Scenario	
	L10	**VL35**
1 h dead fuel moisture content (%)	6	3
10 h dead fuel moisture content (%)	7	4
100 h dead fuel moisture content (%)	8	5
Live herbaceous fuel moisture content (%)	60	30
Live woody fuel moisture content (%)	90	60
Wind direction	blowing uphill	blowing uphill
Wind direction (level cells) (azimuth, °)	0	0
Wind speed (km/h)	10	35

Simulations were performed with a 5 m cell size. For the simulation before the implementation of interventions, the following datasets were used: (a) the digital terrain model (MDT05) supplied by CNIG [107] for altitude, slope, and aspect; (b) the fuel model map for 2010 provided by the regional government of Extremadura for fuel models, as described in Table 3, adjusted through photointerpretation of a 2010 PNOA orthophotograph [108]; (c) a map generated by photointerpretation of the same image as that used for the fuel models, for tree canopy cover. For the simulation after implementation of interventions we considered that altitude, slope, and orientation did not change. Projections of fuel models and tree canopy cover were made based on each of the interventions considered.

Table 3. Rothermel and Albini fuel models [109,110], as adapted in MFE25 [102].

Group	Fuel Model	Description
Grass	1	Dried, short grass with complete ground cover. Scattered woody plants may be found on 1/3 of the area or less. Fuel load (dry matter): 1–2 t/ha.
	2	Dried, short grass with complete ground cover. Scattered woody plants convering from 1/3 to 2/3 of the area. Fire spread is still governed by herbaceous fuels. Fuel load (dry matter): 5–10 t/ha.
	3	Thick, dense, dried and tall grass (>1 m). Scattered woody plants may be present. Fuel load (dry matter): 4–6 t/ha.
Shrubs	4	Mature shrubs or dense plantations of young trees, with a height greater than 2 m. Fire spread through the canopy layer. Fuel load (dry matter): 25–35 t/ha.
	5	Dense, live, short shrubs (<1 m). Fire spread through leaf litter and grass layer. Fuel load (dry matter): 5–8 t/ha.
	6	Similar to model 5, but with more flammable species, or logging slash and taller plants. Fire spread in conditions of moderate to strong wind. Fuel load (dry matter): 10–15 t/ha.
	7	Highly flammable shrubs, 0.5 to 2 m high, as an understory layer in conifer forest. Fuel load (dry matter): 10–15 t/ha.

Table 3. *Cont.*

Group	Fuel Model	Description
Timber litter	8	Dense forest, without shrub understory. Fire spread through thick leaf litter. Fuel load (dry matter): 10–12 t/ha.
	9	Similar to model 8, with less thick leaf litter of long needles or large leaves of broadleaves. Fuel load (dry matter): 7–9 t/ha.
	10	Forests with large quantities of dead biomass and fallen, dead trees due to perturbations (windstorm, pests, etc.). Fuel load (dry matter): 30–35 t/ha.
Logging slash	11	Open forest, intensively thinned. Pruning and thinning debris. Scattered debris from pruning and thinning, with resprouting herbaceous plants. Fuel load (dry matter): 25–30 t/ha.
	12	Biomass debris more abundant than trees. Ground completely covered by pruning and thinning debris. Fuel load (dry matter): 50–80 t/ha.
	13	Ground completely covered by large amounts of heavy and thick biomass debris. Fuel load (dry matter): 100–150 t/ha.

The impact of interventions on the fire extinction capacity was assessed through the definition of four "extinction classes" based on the thresholds of flame length and rate of spread presented in Table 4, evaluating in each cell whether the implementation of interventions was able to improve the "extinction class", and if so, by how many degrees.

Table 4. Extinction classes and thresholds for flame length and rate of spread (based on [111,112]).

Extinction Class	Fire Behavior and Control Method	Thresholds	
		Flame Length (m)	Rate of Spread (m/min)
1	Low spread rate and flame length; hand tools	<1.2	<0.5
2	Moderate spread rate and flame length; heavy equipment	1.2–2.4	0.5–2
3	Crown fires (serious control problems)	2.4–3.4	2–33
4	Crown fires and spotting Control methods ineffective	>3.4	>33

Then, avaluation matrix that ordered all possible "extinction class changes" according to their "degree of improvement in extinction" was created, establishing as more favorable those that managed to cross thresholds allowing fire to be effectively attacked (classes 1 and 2). The value "6" was assigned to the most desirable transition and "1" to the least desirable. The same reasoning was applied to changes that deteriorated the conditions for fire attack, creating a symmetric matrix with positive and negative values, as shown in Table 5.

Table 5. Degrees of improvement assigned to changes in extinction class.

		Extinction Class after Intervention			
		1	**2**	**3**	**4**
Extinction class before Intervention	**1**	0	−2	−5	−6
	2	2	0	−3	−4
	3	5	3	0	−1
	4	6	4	1	0

As flame length and rate of spread may result in different "extinction class changes", a reclassification matrix was generated by applying the relative weight coefficients of 0.66 to flame length and 0.33 to rate of spread. Thus, more importance was assigned to flame length due to the higher influence of this factor on suppression work, in line with [112–114] (Table 6).

Table 6. Final reclassification matrix of degrees of improvement in extinction, considering flame length and rate of spread.

		Degrees of Improvement in Extinction Rate of Spread Weight Coefficient: 0.33)						
		0	**1**	**2**	**3**	**4**	**5**	**6**
Degrees of Improvement in Extinction (Flame Length Weight Coefficient: 0.66)	**0**	0	0	1	1	1	2	2
	1	1	1	1	2	2	2	3
	2	1	2	2	2	3	3	3
	3	2	2	3	3	3	4	4
	4	3	3	3	4	4	4	5
	5	3	4	4	4	5	5	5
	6	4	4	5	5	5	6	6

2.5. Risk Analysis

The location of interventions promoted by Mosaico in relation to fire risk at the study site was analyzed through the definition of a relative "risk index", considering the "sub-catchment" as the spatial unit of analysis. A total of 14 sub-catchments were finally considered, after aggregating the 116 sub-units found in [115]. The relative risk index was calculated for each of the 14 sub-catchments and a comparative analysis conducted.

The risk index was generated from a combination of the concepts of "hazard" and "vulnerability", in line with [116]. Hazard and vulnerability components were calculated after an exhaustive literature review of the Spanish regional fire emergency preparedness and response plans and related legislation [112–114,117]. Each of these components was successively disaggregated into new components, as shown in Table 7. The values of each component were generated at the sub-catchment level and categorized into five classes that were used to compose the upper level through weight matrices. To reclassify the ranges of values, the natural breaks method was used [118].

"Fire behavior" was simulated with FlamMap for climatic scenarios L10 and VL35, assigning a higher weight coefficient to the latter, considering that extreme weather conditions due to climate change are increasingly more frequent. In both scenarios "fire behavior" was assessed by combining "flame length" and "rate of spread", using the same thresholds and weight coefficients as in Tables 4 and 6.

"Historical occurrence" was estimated based on an analysis of fire occurrence for the period 2000–2010 [104]. In line with [119,120], a continuous spatial distribution was generated through the kernel density algorithm, which was later applied to the sub-catchment level through its aggregated value per unit area.

"Risk elements" considered four categories: forest–agriculture interface (FAI), wildland–urban interface (WUI), road network, and infrastructure and legal activities in forest land (industrial activities, dump sites, recreational areas, and camping sites). The contribution to the "ignition probability" was calculated by considering an area of influence of a strip 8 m and 100 m wide for roads and the other elements, respectively. Then, in each sub-catchment, the relative area covered by these strips was estimated without considering overlapping.

"Economic value of ecosystem services" was calculated from 13 raster layers provided by the VANE project [121]. Each raster contained the value of an ecosystem service expressed in terms of flow (EUR ha^{-1} yr^{-1}). The ecosystem services considered were timber, fuelwood, cork, mushrooms, crops, livestock, fish and game, tourism and recreation, erosion control, carbon sequestration, and biodiversity. The total value of the ecosystem services in each pixel was obtained by aggregating the value of each raster. Then, the total value in each sub-catchment level was estimated through its aggregated value per unit area.

The "conservation areas" component was calculated by estimating the proportion of the total area legally protected by the Nature 2000 network in each sub-catchment, based on the dataset provided by the regional government of Extremadura [122].

Table 7. Risk index by components and weight coefficients (Wc).

Level 1	Wc	Level 2	Wc	Level 3	Wc	Level 4	Wc	Level 5	Wc	Level 6
Risk	0.5	Hazard	0.6	Fire behavior	0.7	Fire behavior in climatic scenario VL35	0.65	Flame length	Simulation	Elevation
										Slope
										Aspect
							0.35	Rate of spread		Fuel model
										Canopy cover
										Fuel moisture conditions, wind
					0.3	Fire behavior in climatic scenario L10	0.65	Flame length	Simulation	Elevation
										Slope
										Aspect
							0.35	Rate of spread		Fuel model
										Canopy cover
										Fuel moisture conditions, wind
			0.4	Ignition probability	0.7	Historical occurence	-	Kernel density of point of ignition (in a period)	-	-
					0.3	Risk elements	Agregation	Forest–agriculture interface (100 m)	-	-
								Wildland–urban interface (100 m)		
								Infrastructure and legal activities in forest land (100 m)		
								Road network (8 m)		
	0.5	Vulnerability	0.5	Value	0.7	Economic value of ecosystem services	Agregation	Timber	-	-
								Fuelwood		
								Cork		
								Mushrooms		
								Crops		
								Livestock		
								Big game		
								Small game		
								Fish		
								Tourism and recreation		
								Erosion control		
								Carbon sequestration		
								Biodiversity		
					0.3	Conservation areas	-	Legally protected conservation areas		
			0.5	Fragility	0.7	Human population	-	Population density (radius 1 km^2)	-	-
					0.3	Biophysical environment	0.6	Regeneration capacity	0.85	Biotic regeneration capacity
									0.15	Site quality index
							0.4	Potential erosion	-	-

The "human population" component was estimated by using a continuous distribution of population density with a radius of 1 km. Data were obtained from the census statistics for 2010 [101] and digital maps associated with the 2011 Population and Housing Census INE [123]. The final value of each sub-catchment was calculated through its aggregated value per unit area.

The "biophysical environment" component was composed of the level-5 factors "regeneration capacity" and "potential erosion". "Regeneration capacity" was estimated based on a "site quality index" and the "biotic regeneration capacity", according to [112]. Vegetation units found in the Spanish Forest Map MFE50 [124] were reclassified into five land-cover types with decreasing biotic regeneration capacity: (i) agriculture; (ii) pasture; (iii) shrubs; (iv) adapted trees; and (v) non-adapted trees. Chestnut (*Castanea sativa*) plantations, *Populus* sp. and young afforestation of *Pinus pinaster* were considered to be non-adapted. “Biotic regeneration capacity” was weighted according to a “site quality index” based on the potential productivity cartography classes found in [125]. “Potential erosion” was estimated using the revised universal soil loss equation (RUSLE) [126]. For consistency with the method used for calculating the “economic value of ecosystem services” component, a dataset for “potential erosion” expressed in t ha^{-1} yr^{-1} was obtained from the VANE project [121]. The final value of each sub-catchment was calculated through its aggregated value per unit area.

3. Results

3.1. Efficacy of the Interventions as “Productive Firebreaks”

A change in fuel model type occurred in nearly 90% of the 732 ha covered by the project interventions. By far the largest change was from fuel model type 7 (trees with understory shrubs) to type 9 (forest with a thin litter layer), occurring in 300 ha (41.1% of the total area under intervention), mostly due to forestry interventions (Table 8). A breakdown of the fuel model changes by intervention can be found in Table S1 (Supplementary Materials).

Table 8. Changes in fuel model after implementation of project interventions (% of area changed).

Intervention Type	Area (ha)	Change in Fuel Model (from X→Y) after Intervention (% of Area Changed)											
		7→1	6→1	5→1	4→1	2→1	7→9	6→9	5→9	4→9	2→9	4→5	No Change
Agriculture	49.6	19.8	3.9	31.0	0.0	38.1	0.9	0.0	0.0	0.0	0.0	0.0	6.3
Forestry	553.1	6.9	0.4	0.6	1.0	2.9	53.7	0.3	11.2	3.7	8.5	0.8	9.8
Livestock	129.3	14.7	29.0	10.2	19.5	8.3	2.5	0.7	0.0	0.0	0.0	0.0	15.0
Total	732.0	9.2	5.7	4.4	4.2	6.3	41.1	0.4	8.5	2.8	6.4	0.6	10.5

Agricultural interventions promoted land cover that corresponded to fuel model type 1 (managed perennial woody crops with pasture underneath). Livestock interventions also aimed at promoting fuel model type 1, in this case corresponding to grazed pasture. Agricultural interventions changed 18.9 ha of fuel model type 2 (unmanaged tree-crop plantations), 15.3 ha of type 5 (dense, young shrubs) and almost 10 ha of type 7 (trees with understory shrubs). Those agricultural interventions implemented in areas with fuel model type 7 entailed a change in land use from forest to agricultural land. The largest impact of livestock interventions was in areas with the most hazardous vegetation, i.e., fuel model type 6 (shrubs older, taller, or drier than in type 5) (37.5 ha) and areas with fuel model type 4 (dense shrubs or young trees with a height greater than 2 m) (25.2 ha). No change in fuel model was considered to occur when livestock interventions were proposed in areas that were already grazed or in existing fuel breaks (19.3 ha or 15% of the area under livestock interventions).

Forestry interventions promoted a managed forest with a thin litter layer (fuel model 9). The most frequent starting situation was from model 7. In this case, there was no change in the fuel model in 9.8% of the area under forestry intervention, due to the execution of shrub clearings in the reference year (2010) in the areas designated for resin harvesting. Similarly, small areas were considered in which there would be a change from model 4 to model 5, corresponding to bushes that would not yet have become wooded forest, but which the resin tappers would be required to clear regularly.

The implementation of project interventions resulted in improved extinction capacity (efficacy level 1 to 6) in 84% and 76% of the simulated area for scenarios L10 and VL35, respectively. The largest changes (efficacy level classes 4 and 5) were observed in climatic scenario L10, particularly for livestock interventions (Table 9). A breakdown by intervention of the efficacy levels as firebreaks is found in Table S2 (Supplementary Materials).

Table 9. Efficacy level of interventions as firebreaks in two climatic scenarios.

Climatic Scenario	Intervention Type	Efficacy Level as Firebreaks (Area %)								
		No Fire	−1	0	1	2	3	4	5	6
L10	Agriculture	0.2	0.0	42.0	56.9	0.9	0.0	0.0	0.0	0.0
	Forestry	0.8	0.0	11.7	15.5	66.5	1.0	1.0	3.5	0.0
	Livestock	11.5	0.0	10.0	56.3	2.7	1.4	18.2	0.0	0.0
	Total	2.7	0.0	13.4	25.5	50.8	1.0	4.0	2.7	0.0
VL35	Agriculture	0.2	19.7	37.0	42.3	0.9	0.0	0.0	0.0	0.0
	Forestry	0.8	6.8	10.1	4.8	52.1	25.3	0.0	0.0	0.0
	Livestock	11.5	13.3	10.6	61.4	0.6	2.6	0.0	0.0	0.0
	Total	2.7	8.9	12.0	17.3	39.6	19.6	0.0	0.0	0.0

Forestry interventions were the most effective as firebreaks, with an improvement in extinction capacity in 87% and 82% of the simulated area for L10 and VL35, respectively, followed by livestock (78% for L10 and 65% for VL35) and agricultural interventions (58% for L10 and 43% for VL35).

Contrary to intuition, agricultural interventions resulted in less-effective firebreaks than forestry or livestock interventions. This can be explained by the fact that all agricultural interventions implemented consisted of managed woody crop plantations with pasture as soil cover (without harrowing), which corresponded to fuel model 1, characterized by a low flame length but a high rate of spread. For livestock interventions, which also targeted fuel model 1, this counterintuitive effect was attenuated, as fuel models in 2010 were more hazardous in terms of flame length.

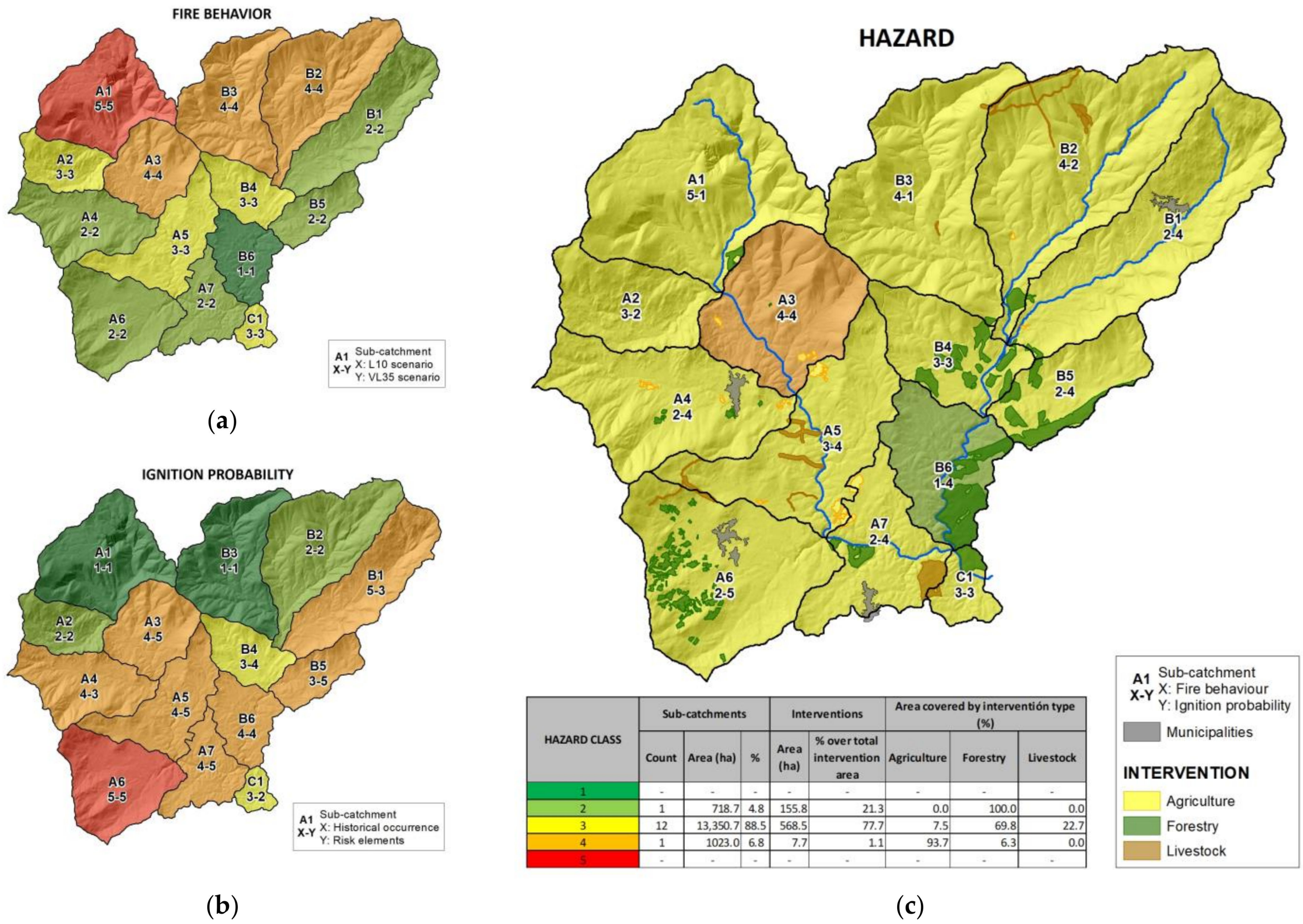
In scenario VL35, some interventions reduced the fire extinction capacity (efficacy level "−1"). This is because in this scenario, transitions to fuel model 1 notably increased the rate of spread, which prevailed over the reduction in flame length. In this study, this occurred only for changes in the rate of spread from extinction class 3 to 4. Nevertheless, a comparison between the two climatic scenarios also showed positive shifts, as efficacy classes 3 or higher changed from 7% in L10 to nearly 20% in VL35. This change was mainly due to forestry interventions where efficacy class 3 changed from 1% in L10 to 25% in L35.

3.2. Risk Analysis

"Hazard", the first component of the "risk index", is composed of "fire behavior" and "ignition probability". Simulations of "fire behavior" showed a similar distribution under both climatic scenarios, as in each pixel, the extinction class in L10 increased by one in VL35. Therefore, for the purpose of sub-catchment prioritization, both scenarios behaved similarly. Sub-catchments that showed the most undesirable levels of "fire behavior" were A1, A3, B2, and B3, due to a combination of large areas of fuel model 4 aggravated by steep slopes (Figure 3a). Around 62% of the total intervention area was in "fire behavior" classes 1 and 2. Only 10 ha (1.1% of the total intervention area) was in "fire behavior" class 5.

With regards to "ignition probability", an analysis of the subcomponents "historical occurrence" and "risk elements" showed coherent results. Sub-catchments with the largest proportion of urban and agricultural land presented a higher "ignition probability" due to human activity. Generally, the probability of ignition decreased towards the upper sub-catchments, with very low values in sub-catchments A1, A2, B2, and B3. Sub-catchment A6 presented the highest probability of ignition, due to the risks posed by the considerable agricultural activity in this sub-catchment and by the municipality of Hoyos (Figure 3b). More than 89% of the total area under intervention was in areas with the highest "ignition probability" (classes 3, 4, and 5). Most of these interventions belonged to the forestry type.

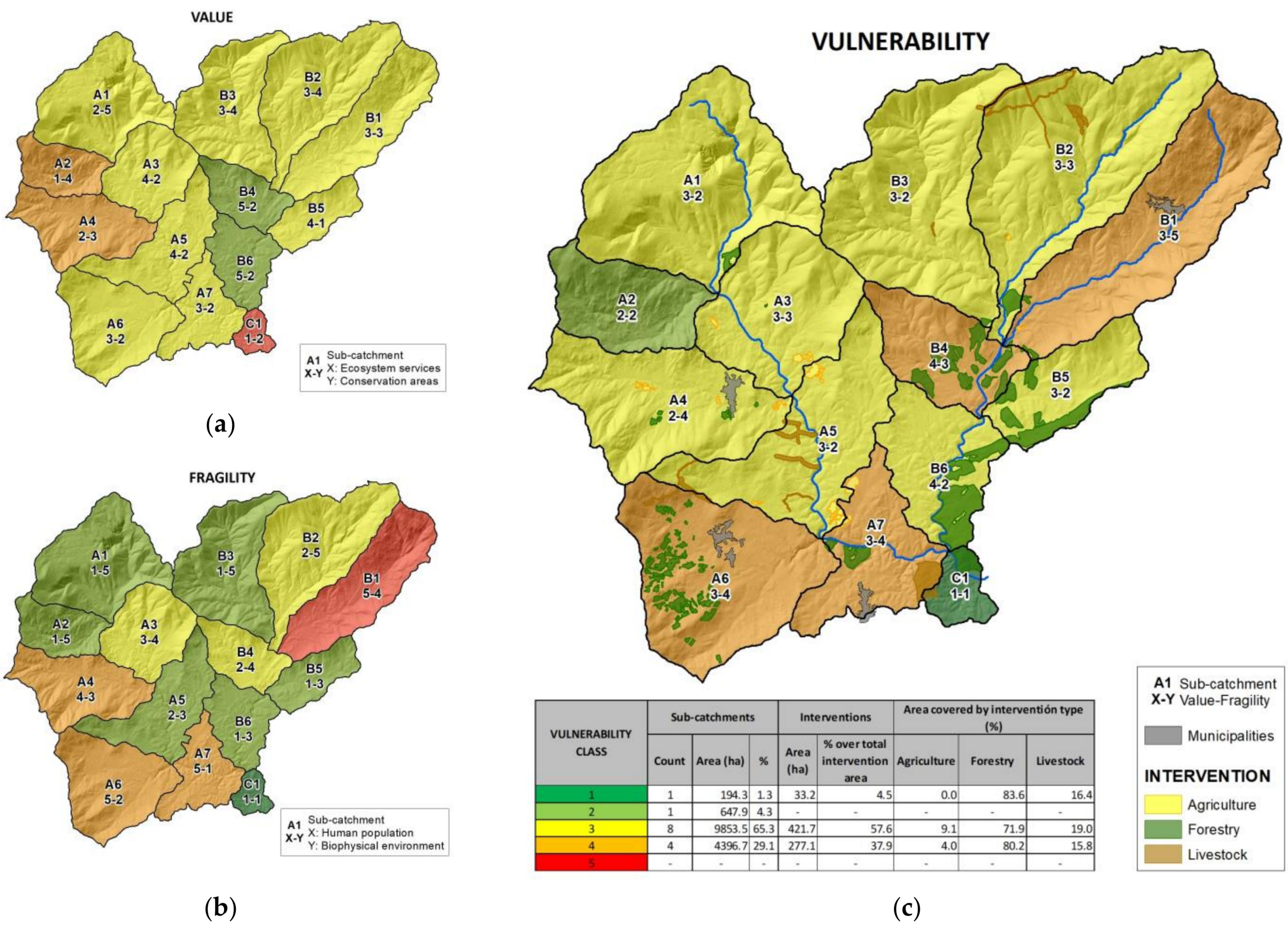
Weighting "fire behavior" and "ignition probability" by the coefficients 0.6 and 0.4, respectively, compressed the final "hazard" range from five possible classes to three. Sub-catchment A3 was the most hazardous as it presented a combination of factors of 4-4, but only 1% of the intervention area was within this sub-catchment. The rest of the sub-catchments, except for B6, showed an intermediate hazard level since high values of "fire behavior" in the upper sub-catchments were offset by a low "ignition probability" (Figure 3c).



HAZARD CLASS	Sub-catchments			Interventions		Area covered by intervention type (%)		
	Count	Area (ha)	%	Area (ha)	% over total intervention area	Agriculture	Forestry	Livestock
1	-	-	-	-	-	-	-	-
2	1	718.7	4.8	155.8	21.3	0.0	100.0	0.0
3	12	13,350.7	88.5	568.5	77.7	7.5	69.8	22.7
4	1	1023.0	6.8	7.7	1.1	93.7	6.3	0.0
5	-	-	-	-	-	-	-	-

Figure 3. Hazard and its components by sub-catchment at the study site: (**a**) fire behavior; (**b**) ignition probability; (**c**) hazard.

"Vulnerability", the second component of the risk index, is composed of "value" and "fragility". The "value" component had the highest values in sub-catchments B4 and B6, characterized by a high "economic value of ecosystem services", mainly due to high timber production and CO_2 sequestration (Figure 4a). All interventions in these sub-catchments were of the forestry type, covering 35% of the intervention area. Most of the remaining sub-catchments were in class 3 for "value", due to different combinations of the aggregated value of their ecosystem services (generally higher in lower areas of the catchment) and the low weight assigned for "conservation areas" (concentrated in the headwaters sub-catchments).



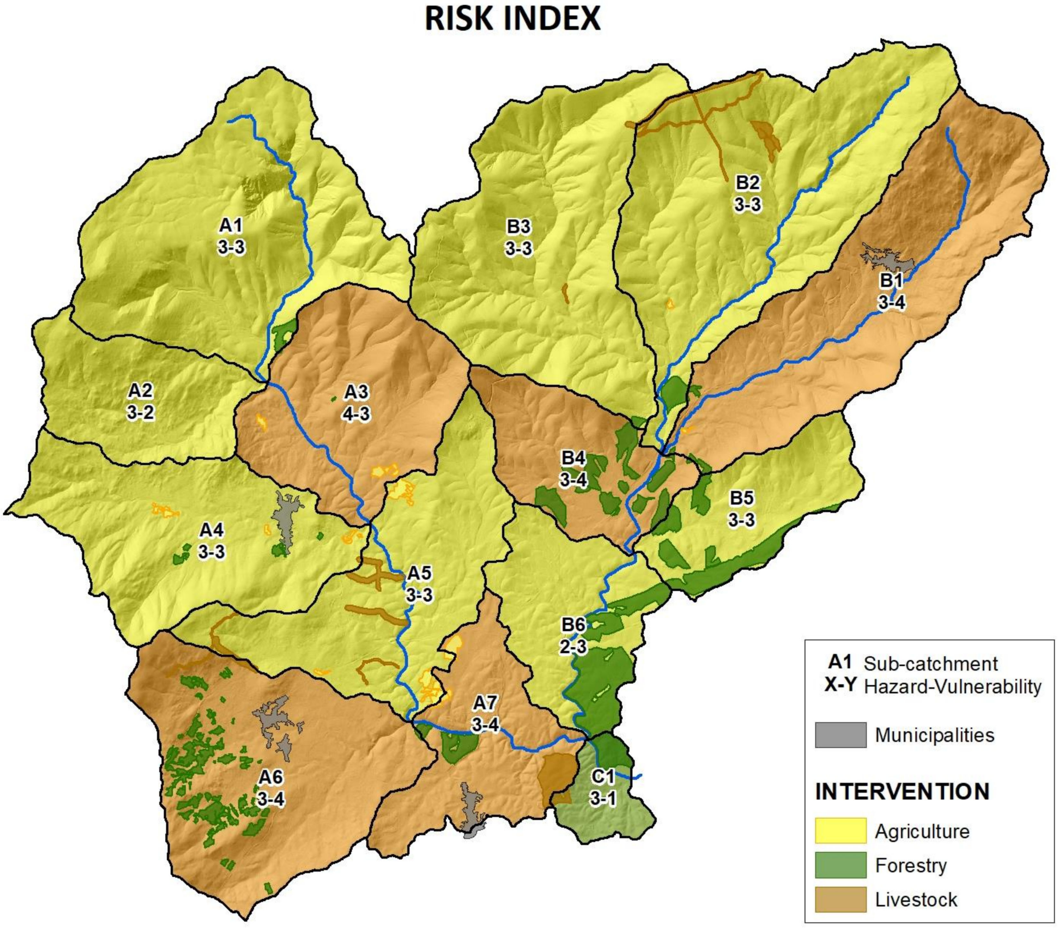
VULNERABILITY CLASS	Sub-catchments			Interventions		Area covered by intervention type (%)		
	Count	Area (ha)	%	Area (ha)	% over total intervention area	Agriculture	Forestry	Livestock
1	1	194.3	1.3	33.2	4.5	0.0	83.6	16.4
2	1	647.9	4.3	-	-	-	-	-
3	8	9853.5	65.3	421.7	57.6	9.1	71.9	19.0
4	4	4396.7	29.1	277.1	37.9	4.0	80.2	15.8
5	-	-	-	-	-	-	-	-

Figure 4. Vulnerability and its components by sub-catchment at the study site: (**a**) value; (**b**) fragility; (**c**) vulnerability.

Since the model used to estimate "fragility" weighted "population" more heavily against "environmental factors", the most fragile sub-catchments were those that included urban centers (B1, A4, A3, and A7). These sub-catchments contained 26% of the intervention area. Sub-catchments adjacent to these (A3, B2, and B4) combined low population density values (proximity to the villages of Acebo and Gata, and the campsite, respectively) with a fragile biophysical environment (high "potential erosion" and vegetation with low "regeneration capacity") (Figure 4b).

The combination of "value" and "fragility" using the same weight factors resulted in four classes for "vulnerability". The most vulnerable sub-catchments were A6, A7, and B1, which contain the municipalities of Hoyos, Perales del Puerto, and Gata, respectively, and B4, which is a territory with high "value" and intermediate "fragility". The least vulnerable sub-catchments were C1 (1-1) and A2 (2-2). The rest of the studied territory showed a similar level of intermediate "vulnerability" (Figure 4c).

Final integration of the "hazard" and "vulnerability" components into a relative "risk index" reduced the five possible classes to three (classes 2, 3, and 4), eliminating both extremes. Sub-catchments with the highest "risk index" in the study site were: A3, located to the north of the municipality of Acebo and including some residential houses spread over the area and a recreational area; A6, which contains the municipality of Hoyos; A7, containing the municipality of Perales del Puerto; B1, containing the municipality of Gata; and B4, with a 3-4 combination of "hazard" and "vulnerability" classes. The small-sized sub-catchment C1 was in the lowest risk class, while the rest of the territory presented an intermediate level of risk (Figure 5). The complete set of class values for all "risk index" components in each sub-catchment can be found in Table S3 (Supplementary Materials).



RISK CLASS	Sub-catchment		Interventions				Area covered by intervention type (%)		
	Count	Area (ha)	Count	Area (ha)	Proportion over sub-catchment (%)	% over total intervention area	Agriculture	Forestry	Livestock
1	-	-	-	-	-	-	-	-	-
2	1	194.3	2	33.2	17.1	4.5	0.0	83.6	16.4
3	8	9478.4	15	414.0	4.4	56.6	7.6	73.1	19.4
4	5	5419.7	13	284.8	5.3	38.9	6.4	78.2	15.3
5	-	-	-	-	-	-	-	-	-

Figure 5. Final risk index by sub-catchment at the study site.

Most interventions (15 out of 23) were located in sub-catchments with fire risk class 3, covering 414 ha (4.4% of the total area of these sub-catchments). Forestry covered 73%, livestock 19%, and agriculture 8% of the total intervention area. The largest intervention areas were found in sub-catchments B5 and B6, where forestry interventions covered around 20% of the sub-catchment's area. Around 15.7% of the total intervention area (110 ha) was found in sub-catchments A5 and B2, in which agricultural and livestock interventions predominated. Sub-catchments with the highest fire risk index (class 4) included 13 interventions, covering 285 ha and 5.3% of the total area of these sub-catchments. Forestry covered 78%, livestock 15%, and agriculture 6% of the total intervention area. The largest intervention areas were found in sub-catchments A6, A7, and B4. Forestry interventions covered 15.4% of the area of sub-catchment B4. Finally, sub-catchment C1 was the only one with risk class 2, with two interventions (forestry and livestock) covering 33 ha (17% of the area of the sub-catchment) (Table 10).

Table 10. Intervention type, area, and relative risk class in each of the sub-catchments of the study site.

Risk Class	Sub-Catchment		Intervention in Sub-Catchment				Area Covered by Intervention Type (%)		
	ID	Area (ha)	Count	Area (ha)	Proportion over Sub-Catchment (%)	Proportion over Total Intervention Area (%)	Agriculture	Forestry	Livestock
2	C1	194.3	2	33.2	17.1	4.5	0.0	83.6	16.4
3	A1	1683.1	1	9.9	0.6	1.4	0.0	100.0	0.0
	A2	647.9	0	0.0	0.0	0.0	0.0	0.0	0.0
	A4	1201.3	5	14.5	1.2	2.0	46.4	49.1	4.5
	A5	1179.3	9	58.4	5.0	8.0	40.5	0.0	59.5
	B2	1873.6	4	56.4	3.0	7.7	1.6	30.7	67.7
	B3	1599.5	3	9.6	0.6	1.3	0.0	31.3	68.7
	B5	575.2	2	109.4	19.0	14.9	0.0	100.0	0.0
	B6	718.7	3	155.8	21.7	21.3	0.0	100.0	0.0
4	A3	1023.0	5	7.7	0.8	1.1	93.7	6.3	0.0
	A6	1437.1	4	98.0	6.8	13.4	0.0	93.3	6.7
	A7	739.3	4	71.2	9.6	9.7	14.4	33.5	52.1
	B1	1572.3	2	8.1	0.5	1.1	9.9	90.1	0.0
	B4	647.9	1	99.7	15.4	13.6	0.0	100.0	0.0
-	Total	15,092.4	23	732.0	4.9	100.0	6.8	75.6	17.7

4. Discussion

We studied the efficacy of agricultural, livestock, and forestry interventions promoted by the Mosaico project as productive firebreaks and analyzed their locations in relation to fire risk at the study site. To this end, we conducted and compared simulations of fire behavior before and after implementation of the project interventions and developed a relative risk index to identify sub-catchments of the study site with the highest fire risk.

Overall, the results showed that interventions can function as effective firebreaks that can facilitate fire attack and suppression work. We also found that most interventions were located in areas with medium and high relative risk indexes (classes 3 and 4).

Forestry interventions (i.e., tree thinning and shrub clearing in areas devoted to resin tapping) always had a positive impact, due to the type of targeted fuel model (forest with a thin litter layer), reducing both flame length and rate of spread and thus contributing to improving attack and suppression work. The efficacy of agricultural and livestock interventions as firebreaks depended on the fuel model on which they were implemented. This is due to the behavior of fire in the final fuel model 1 (i.e., managed woody crop with pasture underneath and grazed pasture for agricultural and livestock interventions, respectively), which is characterized by a low flame length, but a high rate of spread. In this regard, livestock initiatives (i.e., extensive grazing) would be more effective, as they could be implemented on steep slopes, which usually present the most unfavorable fuel models. However, whether this occurs depends on the type of livestock (goats are the most adapted to grazing in difficult and thickly vegetated terrain) and the various incentives for grazing in these areas, including troughs and folds.

A logical corollary of our results is that resin tapping, and its associated silvicultural practices (F-05 in Table 1), should ideally be implemented in all suitable areas (i.e., accessible areas with mature trees) in order to maximize the benefits from this type of productive fuel break. Resin tapping has recently become an attractive forest-based livelihood option due to the high market demand, stable price of resin, and its promotion by the Forest Service Office and local governments. Around 30 resin tappers have been established in Sierra de Gata since its promotion began in 2015 [127], and the potential for further expansion is high. Favorable climate and soils make the nearly 15,000 ha of high pine forest found in Sierra de Gata and in the neighboring Las Hurdes among the most productive in the region, with conservative estimates of an average production between 2.4 to 3.1 kg of resin annually per tree [128]. However, according to resin tappers involved in Mosaico, two important problems should be addressed to make it more attractive. First, more intensive thinning should be conducted at tapping sites to improve productivity, since the current tree density, at an average of 400 to 500 trees per hectare, is still above the optimum range of 200 to 300 required for maximum resin production [128]. Second, the Forest Service Office could facilitate arrangements to involve resin tappers in silvicultural activities and other forest management work during the 4-month-long lean period in winter [126].

Another corollary is that livestock and agricultural interventions will be most effective when practiced in targeted locations. Unlike the traditional extensive grazing in public forests, targeting grazing in fuel breaks and in designated areas can be a cost-effective management option to reduce biomass fuel and the costs of mechanical clearing, while producing meat and milk as by-products and giving shepherds due recognition as land stewards [83–85,95]. Based on a successful targeted grazing approach in the neighboring region of Andalusia [93], Mosaico project staff and some shepherds at the study site conceptualized, proposed, and implemented a targeted grazing approach named BOCA on the fuel break network at the study site (interventions L-03, L-04, and L-05 in Table 1). Stimulated by this initiative, the Regional Forest Service has begun to roll out targeted grazing to other forests areas of the Extremadura region [129]. Targeted grazing, however, makes exacting demands on all stakeholders involved as it requires: (a) high-level, long-term political and institutional commitment; (b) appropriate, long-term financial incentives and infrastructure support, mainly troughs and folds; (c) adaptive management and continuous learning (e.g., to determine adequate stocking rates); (d) a sound monitoring plan that is strictly implemented; and (e) commitment from forest managers, as contentions will inevitably arise [84].

Agricultural interventions (i.e., commercial tree-crop plantations) will be most effective as firebreaks when established on abandoned agricultural land currently covered with thick, flammable vegetation. There is an important impediment, however, to the rehabilitation of abandoned agricultural land in Spain, and particularly in the Extremadura region. The National Forest Law [130] and The Agrarian Law of Extremadura [131] consider as forest all agricultural land that has remained uncultivated for at least ten years and that contains forest trees or shrubs with a diameter at the base of 15 cm or larger. Thus, the rehabilitation of an existing tree plantation or the establishment of a new one requires the approval of a formal request for land-use reclassification from forest to agricultural land, a lengthy, bureaucratic process that is unlikely to succeed. This precludes the rehabilitation of the many abandoned olive and almond groves, chestnut orchards, and vineyards encroached by forest vegetation.

Aware of this problem, the government of Galicia, a region in northwestern Spain prone to wildfires, enacted a pioneering law for the rehabilitation of abandoned agricultural land in 2021 [132]. Key to this law is the explicit recognition of agroforestry as a productive land-use system with the capacity to act as a fuel break and the consideration of private and public actors as agents of change. If successful, this law may help to rehabilitate agroforestry mosaic landscapes resilient to fire while helping to mitigate the abandonment of rural areas across the region. The spirit of this law could also provide a new stance regarding the long-standing issue of the "enclavados", traditional small-scale plantations of agricultural trees within public forests, which are so common throughout Spain. Considered as anomalies that complicate forest management [133] (p. 106), forest administration offices across the country have been trying to address this "problem" by either purchasing these lands or exchanging them for others in more accessible agricultural areas [130]. Our study shows that, rather than a nuisance, small-scale tree plantations within a forest matrix can be seen as effective and productive fuel breaks that, in the event of a fire, will facilitate attack and suppression work.

Although most interventions were found in areas with a medium or high relative risk index, we cannot state that Mosaico has had a positive impact on fire risk in the territory. This is because the interventions did not respond to prior planning, but to the objectives, interests, and capacity of the owners to implement the interventions. Ideally, interventions should be located in the sub-catchments with the highest risk, and within them, at "strategic management points" (SMPs), i.e., locations where the modification of fuel or the establishment of infrastructure allow for maximum risk mitigation [73,134,135]. Instead, all Mosaico interventions, except targeted grazing, were implemented on agricultural land owned by the proponents, or in forest areas designated for resin tapping by the regional Forest Service Office. Therefore, studies to identify SMPs are essential to help prioritize areas for the strategic location of fire prevention interventions. The identification of SMPs can help forest managers and other stakeholders make better, more informed decisions about the location of fire prevention interventions, although with an important caveat: SMPs may not be suitable for the location of fire prevention agricultural or forestry interventions managed by rural people, and, not all interventions may be feasible at SMPs.

As well as facilitating a network of productive fuel breaks, the Mosaico project has produced some less tangible but equally important social benefits and outcomes that show the important advantages of collaborative approaches for fire risk management. Mosaico has shown that: (1) when given the chance, rural people take a proactive rather than reactive responsibility for reducing the vulnerability of the landscape and mitigating risk; (2) if given a voice, stakeholders can positively influence decisions taken on projects, actions, and activities in public and private forests; (3) by securing long-term commitment and funding from the regional government, stakeholders, including municipal governments, become more dedicated and committed to a project and more willing to invest their own financial resources in project interventions; (4) collaboration has helped to increase the capacity of stakeholders to accomplish work and leverage other funding in support of the approach; and (5) by being involved in participatory action research, stakeholders have provided valuable feedback to scientists, technicians, and local policymakers [91].

Collaborative approaches to address the wicked problem of wildfire are not a panacea. Stakeholder participation requires time, long-term political commitment and institutional support, sufficient resources, and fundamental changes in public institutions including a new working culture moving away from "the expert knows best" culture and a new perspective on the role of rural people as land managers [79,136]. However, as the Mosaico project demonstrates, collaborative strategies can effectively mobilize rural people and other stakeholders for the co-creation, in partnership with forest administrators, of agroforestry landscapes more resilient to fire. More importantly, by promoting economic activity and restoring strong links between the rural population and its surroundings, collaborative approaches such as Mosaico have the potential for mitigating, or even reversing, the abandonment of rural areas and contributing to more sustainable land and forest management.

Further Research

In both simulated climate scenarios, positive and negative trends in extinction capacity were observed. Future studies should consider more scenarios, as more frequent extreme weather conditions are increasing the severity of fires [10–12].

Methodologies based on extinction thresholds complement other methods which are based only on the variability of certain fire parameters, without considering their relationships with extinction capacity. It would be advisable to perform a sensitivity analysis of the thresholds used. It would also be desirable to model crown fires when more recent, open, national LiDAR data become available.

This study should be complemented by others evaluating the influence of Mosaico on the rest of the level-3 components of the risk index developed ("ignition probability", "value", and "fragility"). It is also important to study the influence of interventions on extinction work, including improvement in accessibility and the passability of forest roads, and greater availability of water points.

5. Conclusions

This study reinforces the premise on which the Mosaico project is based: that rural people's activities such as tree farming, extensive livestock grazing, and forest product harvesting within forests and forest margins create areas that can effectively function as productive firebreaks. Furthermore, forest landscapes could become more fire-resistant if these activities were strategically located in areas with high fire-risk-mitigation potential. The co-creation of these fire-resistant landscapes, however, requires the participation of rural people in land-use planning, appropriate incentives for land management, policy support, and the removal of some regulatory and administrative barriers that hinder collaboration. For these reasons, and for the sake of the potential contribution to the economic and social development of disadvantaged rural areas, public administrations should decisively support the adoption of collaborative strategies for land management.

Supplementary Materials: The following supporting information can be downloaded at: https://www.mdpi.com/article/10.3390/land11040465/s1, Table S1: fuel model changes by intervention; Table S2: efficacy level as firebreaks by intervention; Table S3: class values for risk index components in each sub-catchment.

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