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Wildfire risk management in southern European landscapes: Towards a long-term comprehensive strategy

Fermín J. Alcasena Urdíroz

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Ph.D. Thesis

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Towards a long-term comprehensive strategy

Fermín J. Alcasena Urdíroz

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Servei Meteorològic de Catalunya

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Contents

Foreword

Wildfires represent a devastating natural hazard causing every year significant losses worldwide. In Mediterranean areas current land management and fire response policies are failing to protect our landscapes, property and human lives from large uncharacteristic events despite increasing expenditures in fire suppression resources. Responsible factors have been widely discussed, but few efforts have been conducted to provide a long-term comprehensive solution to better coexist with fire while accounting for existing environmental constraints and finite budgets to manage vast areas.

The Doctoral School of the University of Lleida has as one of its principal objectives the promotion of high-quality third cycle public education and research training while increasing the interconnection networks between pioneering international institutions and the society, to ultimately contribute in the resolution of the most significant challenges facing the world today. Thus, the generation of applied knowledge is the desired outcome to translate the results from research and innovation into policy making and decision processes. Annually, the University of Lleida provides grants and financial aid as research scholarships to young teaching and research staff, and this doctoral Thesis was funded by a July 2014 to July 2017 University of Lleida fellowship. Also, this Ph.D. was complemented with a 6-month research stay at Oregon State University to integrate the latest advances in wildfire science and enhance the quality of the dissertation.

This Thesis implements the newest developments in large fire modeling across complex terrain and diverse fire regimes and integrates a quantitative assessment framework for exposure analysis, wildfire transmission and risk assessment. The real value of this scientific work relies on the replicability and transferability of the management-oriented cross-scale results for prioritizing and optimizing wildfire risk management efforts in any fire-prone southern European Union Mediterranean region and elsewhere.

The dissertation is organized in 5 chapters corresponding to the content of the following scientific papers published in peer review journals:

- **Alcasena FJ**, Salis M, Vega-Garcia C (2015) A fire modeling approach to assess wildfire exposure of valued resources in central Navarra, Spain. *European Journal of Forest Research* **135**, 87-107. Doi: 10.1007/s10342-015-0919-6

- **Alcasena FJ**, Salis M, Nauslar NJ, Aguinaga AE, Vega-Garcia C (2016) Quantifying economic losses from wildfires in black pine afforestation of northern Spain. *Forest Policy and Economics* **73**, 153-167. Doi: 10.1016/j.forpol.2016.09.005

- **Alcasena FJ**, Salis M, Ager AA, Castell R, Vega-Garcia C (2017) Assessing wildland fire risk transmission to communities in northern Spain. *Forests* **8**, 27. Doi: 10.3390/f8020030

- **Alcasena FJ**, Ager AA, Salis M, Day MA, Vega-Garcia C (2018) Optimizing prescribed fire allocation for managing wildfire risk in central Catalonia. *Science of the Total Environment* **4**, 872-885. Doi: 10.1016/j.scitotenv.2017.11.297

- **Alcasena FJ,** Ager AA, Bailey JD, Pineda N, Vega-Garcia C (2018) Towards a comprehensive wildfire management strategy for Mediterranean areas: Framework development and implementation in Catalonia, Spain. *Journal of Environmental Management*. 231, 303-320. Doi: 10.1016/j.jenvman.2018.10.027

- **Alcasena FJ,** Evers C, Vega-Garcia C (2018) The Wildland-Urban Interface raster dataset of Catalonia. *Data in Brief* **17**, 124-128. Doi: 10.1016/j.dib.2017.12.066

- **Alcasena FJ,** Ager AA, Salis M, Day MA, Vega-Garcia C (2018) Wildfire spread, hazard and exposure metric raster grids for central Catalonia. *Data in Brief* **17**, 1-5. Doi: 10.1016/j.dib.2017.12.069

- **Alcasena, FJ**, Vega-García C, Ager, AA, Salis, M, Nauslar N, Mendizabal FJ, Castell R. (2019) Metodología de evaluación del riesgo de incendios forestales y priorización de tratamientos multifuncionales en paisajes mediterráneos. *Cuadernos de Investigación Geográfica* **45**. Doi: 10.18172/cig.3716

Significant findings and preliminary results were advanced as oral communications in international congresses, including:

- **Alcasena FJ,** Costafreda-Aumedes S, Monfort-Bagüe I, Vega-Garcia C. Fine-scale fire risk mapping in forest stands of central Navarre (Spain). II International Conference on Fire Behavior and Risk. 26-29 May 2015. Alghero (Italy). Oral communication.

- **Alcasena FJ,** Ager AA, Salis M, Vega-Garcia C. Optimizing the use of prescribed fire for managing wildland fuels in Catalonia. International Congress on Prescribed Fires. 1-3 February 2017. Barcelona (Spain). Oral Communication.

- **Alcasena FJ**, Ager AA, Bailey JD, Vega C.Wildfire risk management on large Mediterranean landscapes. IV International Congress on Risks. 23 – 26 May 2017. Coimbra (Portugal). Oral Communication.

Summary

Few large and destructive fires account for most negative impacts on socioeconomic and natural values in Mediterranean areas. As a result of an increasing amount of biomass accumulation on the previously fine-grained cultural landscapes, these uncharacteristic events occurring under extreme weather conditions are resistant to suppression efforts due to massive showering embers, overwhelming fire intensities, and very high spread rates. Moreover, increasing wildland-urban interface areas represent a conditioning factor demanding protection and substantially increasing emergency management complexity. Ignition prevention and fire suppression policies alone result ineffective to mitigate losses from contemporary fires.

In this Thesis I implemented a multiple-scale analytical framework to inform the decision-making of a wildfire risk management strategy aiming at creating fire resilient landscapes, restoring the cultural fire regime, supporting safe and efficient fire suppression, and creating fire-adapted communities (USDA Forest Service 2014). By decomposing wildfire risk into the main causative factors at scales related to management capabilities for the different agents, from the individual homeowners to Regional Governments, this dissertation attempts to provide a comprehensive solution to achieve those core goals on the mid-term in southern European Union regions. A fire simulation modeling approach was implemented to obtain the required risk causative factors or exposure metrics. Fire spread and behavior in large areas were modeled accounting for variable fire regimes in terms of seasonality, large fire number, and spatial distribution. Expert-defined susceptibility relations or mortality models were then used to assess fire effects as potential economic losses to values at risk. Moreover, we used a transmission analysis to delineate community firesheds and assess fire exchange among neighboring municipalities. Fuels management is the main wildfire risk mitigation strategy at the landscape scale, and spatial optimization models were used to help in strategic landscape treatment design and explore collocation opportunities under budgetary restrictions.

Results were provided at appropriate operational scales to inform different wildfire management strategies. Exposure profiles and risk assessment at fine scales for individual housing structures and timber stand forest values attempt to promote homeowners' involvement and demand forest managers' good practices aiming at mitigating losses from fires ignited on the same site (treatment units) and the neighboring lands. Management efforts within Planning Areas articulated as collaborative planning projects among various socioeconomic agents include landscape fuel treatments on strategic locations reducing overall wildfire likelihood and fire intensity, landscape planning to exclude hazardous areas for the urban development, community preparedness reducing social vulnerability, and municipality ordinances to reduce housing vulnerability. Treatment joint-production represents an opportunity in multi-functional Mediterranean forest ecosystems to arrange complex solutions. Regional scale policy-making prioritizes at municipality level the different management strategies such as ignition prevention programs, suppression resource pre-positioning, assignation of subsidies for fuel treatments, and law enforcement for managing fuels in wildland-urban interface communities at highest risk. The different papers were developed in various Mediterranean areas to highlight the applicability of the framework elsewhere.

Keywords: large fire risk, fire transmission, multi-scale analysis, cohesive strategy, Mediterranean areas.

INTRODUCTION

Wildfires in the Mediterranean

Humans and fire regimes have been co-evolving in Mediterranean ecosystems for almost a millennium, and both are still the principal agents that model the ecosphere (Moritz *et al.* 2014; Waters *et al.* 2016). Lightning fires during water shortage periods have constantly reshaped landscapes by conditioning the adaptive strategies and changing vegetation structure, composition, and spatial distribution according to fire size, frequency, and severity (Pausas *et al.* 2008). On the other hand, the use of fire as a tool by humans is mostly limited to recent history when fire began to be used to clear undesired vegetation (Bowman *et al.* 2009; Seijo *et al.* 2015). At the same time, humans replaced native herbivores by domestic livestock and transformed the landscape into a mosaic of complex cultural systems by cultivating fertile soils surrounding communities, maintaining grasslands in the shallow marginal areas excluded for agriculture, and managing forests for multi-functional purposes. For the latter, wooded pastures represented the most singular case in many areas, where local livestock breeds fed on pasture, and intensively trimmed low-density and open-canopy old growth forestlands that provided acorn, firewood, and cork (Arosa *et al.* 2017; Garrido *et al.* 2017). These and other Mediterranean ecosystems have represented an excellent example of sustainably managed cultural landscapes over several hundred years, and this is the reason why, currently, we hardly know the particulars of plant species compositions, structure, cover, growth dynamics, and spatial distribution of previous reference conditions. Paleo-ecological data facilitate the identification of existing species and fire activity over the past hundred years (Leys *et al.* 2014). The very few natural undisturbed habitats and primary forests are constrained to remote areas or mortmain lands where human intervention has been minimal (Sabatini *et al.* 2018).

In recent Mediterranean history, the interactions between anthropic management and fire resulted in mixed dynamics combining seasonal lightning fires in mountainous areas with human-caused fires. Cultural fires included seasonal burns related to pastoralism on slopes during spring, and the rainfed crop waste elimination on arable lands and pile burning in forestlands during fall. Apart from the changes in fire activity triggered during historic socioeconomic events occurring at particular periods (e.g., the ecclesiastical and public land seizure known as the Spanish disentailment, the disappearance of *La Mesta* seasonal livestock migration, and the arrival of the grape phylloxera plague from America), a low-severity high-frequency fire regime was overall dominant in the pre-industrial era. Using dendroecological, paleontological and historical archive data, we can start reconstructing the fire activity by the beginning of the XIXth century. While the former data are obtained respectively from fire-scars in remaining old growth stands and charcoal accumulation rates in peat bogs (Fulé *et al.* 2008; Christopoulou *et al.* 2013; Camarero *et al.* 2018), the latter corresponds to court judgments and fire damage claims (Montiel 2013). In the post-industrial era, from around the middle of the XXth century, the fires rapidly evolved from fuel-limited short-events to weather-driven catastrophic events (Pausas and Fernández-Muñoz 2012; Seijo and Gray 2012).

Currently, in southern EU countries (Portugal, Spain, France, Italy, and Greece) some 49,000 fires, mostly caused by humans (> 85%), annually burn about 448,000 ha (1980 to 2015 period) (San-Miguel-Ayanz *et al.* 2013; San-Miguel-Ayanz *et al.* 2017). The most damaging fires usually concentrate during multiple-fire extreme-weather episodes, spread for long distances (> 10 km), and expose natural resources and human communities far beyond ignition locations (Athanasiou and Xanthopoulos 2010; Alcasena *et al.* 2015). Fire occurrence spatiotemporal patterns and motivations also changed over time and nowadays most fires concentrate close to urban development areas, communication corridors and infrastructures (Gonzalez-Olabarria *et al.* 2012; Costafreda-Aumedes *et al.* 2016). In just a human generation, the typical patterns with high numbers of sparsely distributed ignitions across the landscape associated to the traditional fire use shifted towards arson or negligence fire ignition hot-spots in traveled roads and highly populated areas (Moreno *et al.* 2014; Vilar *et al.* 2016). Similarly, fire danger meteorological indexes (e.g., FWI) currently make better predictors of fire initiation than the timing for conducting farming practices (e.g., crop harvesting)(Ager *et al.* 2014). The contrast between changing patterns emerges when comparing the rural mountainous areas where extensive farming is still relevant to the peri-urban areas experiencing rapid growth (Martínez *et al.* 2009; Gonzalez-Olabarria *et al.* 2015). In the midterm, recent studies forecast more frequent heat waves and reduced summer precipitation, which would lead into increasingly longer wildfire seasons and higher wildfire likelihood on Mediterranean landscapes (Cardil *et al.* 2014; Lozano *et al.* 2017; Spinoni *et al.* 2017).

Large destructive contemporary fires

The fact is that a small number (<15%) of large events (> 100 ha) account for the bulk of the burned area. Despite being rare events, extreme fires escaping from initial attack grow to a massive size and result devastating. These stand-replacing uncharacteristic fires burn large portions in forest ecosystems and result in very substantial economic and environmental losses (e.g., destruction of low-intensity fire-adapted forest ecosystems), in addition to the extraordinary expenditures required for firefighting and post-fire restoration work (Prats *et al.* 2014; Mallinis *et al.* 2017). Likewise, these severe fires substantially contribute to carbon emissions and global warming (Domingo *et al.* 2018) and, more importantly, pose a real threat to firefighters since most causalities relate to entrapments during these episodes (Cardil *et al.* 2017). Conversely, wildfire managers and scientist are calling attention to the positive effects of low-intensity fires and the need for selectively directing efforts to preventing, fighting and mitigating losses from the large destructive fires. Indeed, small fires spreading under mild weather conditions positively contribute to the conservation of fire-dependent old-growth natural ecosystems, maintain the discontinuities within large forested areas as required by many animal species of particular interest, and reduce future fire severity and spread rates (Finney *et al.* 2005; Regos *et al.* 2014).

Major factors triggering uncharacteristic fires

Despite changing climatic conditions in southern EU fire-prone Mediterranean areas and existing regional differences in terms of vegetation, topography, land tenures and historical management, we can identify some triggering factors in common that explain evolution of fire regimes in barely fifty years, and why do contemporary fires result so devastating for property and natural values (Moreira *et al.* 2011; Koutsias *et al.* 2012; Fernandes *et al.* 2014).

An increasing fuel buildup in cultural landscapes

Mediterranean landscapes are evolving from intensively managed, and highly fragmented multi-functional intermingled smallholdings to an enormous-patch-size mosaic of a dense regenerate young forest continuum only disrupted by the cultivated plains (Poyatos *et al.* 2003; Cervera *et al.* 2016). While arable properties in open lands are being transformed into intensive agricultural systems providing food commodities to the population living in the cities and intensive livestock breeding corporations, the lack of management in forestlands lead into a very high-hazard fuel buildup continuum. Various factors including a drastic decrease in the use of firewood, the cessation of extensive livestock farming systems, and the abandonment of marginal agricultural lands (i.e., those excluded from land consolidation or reparcelling projects) represent the fundamental factors. Moreover, active conifer afforestation policies aiming at facilitating the following establishment of oak species in shallow and poor marginal soils did not get required transitioning treatments towards a mixed forest and more fire resilient structures due to decreasing investment trends in non-productive forests.

The implementation of a fire exclusion policy

In parallel, by efficiently suppressing all lightning ignitions and banning the traditional use of the fire, this fuel accumulation process and the fast transition to highly flammable landscapes was accelerated. The positive contribution of fire in preserving cultural landscapes was ignored for decades, and strong wildfire prevention campaigns were directed to present fire as a natural hazard responsible for high losses on property and forest values. These campaigns initiated on the early sixties and advocated for preventing all fire ignitions. Forest managers, shepherds, and farmers replaced the cultural fire use by the more expensive mechanical treatments that were limited to lowest-slope open lands and had a minor impact in reducing landscape fuels. Likewise, improving firefighting resources by including aerial means and firefighting engines on the first response facilitated a rapid response and allowed direct attack with water, which increased initial-attack success drastically and contention capacity on escaped fires. This full suppression policy has been forcefully implemented until the present in most fire-prone Mediterranean areas for decades, except for some firefighting units specialized in the tactical fire use. As a result of this wildfire management policy escalating investments have been devoted to preventing ignitions and suppressing all fires at any cost, but the large-fire trend and burned areas have not declined significantly despite reducing the numbers of human sources (San-Miguel-Ayanz *et al.* 2013; Jiménez-Ruano *et al.* 2017).

Extreme fires are resistant to suppression efforts

Massive showering embers that surpass fuel discontinuities (spotting distance > 2 km), high spread rates (> 2 km^{-h}), high fire intensities (flame length > 3m) and crown fires in heading and flanking fires overwhelm firefighting capacity (Castellnou and Miralles 2009; Costa *et al.* 2011). Moreover, recent mega-fires of 2017 (Pedrógão Grande fire in Portugal and Las Maquinas in Chile) and the 2018 fires of Attica in Greece showed an unprecedented extreme behavior that burned vast areas and caused dozens of fatalities (de la Barrera *et al.* 2018). While better training, tactical fire use, and high resource availability make first responders highly effective in controlling most fire ignitions, the very few that escape initial attack and grow under extreme fire-weather conditions become resistant to suppression efforts. Then, firefighting becomes opportunistic on large fires and containment efforts concentrate on the backing and flanking fire spread areas on strategic locations when mild fire-weather conditions allow it (Finney *et al.* 2009). Nonetheless, protecting human communities is the main priority during the extended attack, and vast resources are directed to protect human communities at the expense of fire front contention efforts. The cultural landscapes with a dense road infrastructure network, cultivated agricultural lands, and pasture mosaics with low shrublands present best opportunities for an aggressive fire suppression policy.

A rapidly expanding wildland-urban interface

The rapidly expanding areas between unmanaged forests and human development comprise most human fatalities and losses to property. This area is widely known as the wildland-urban interface (WUI), and it has been characterized into different classes (interface, intermix and dispersed low density) according to housing structure density and the surrounding hazardous vegetation types (Radeloff *et al.* 2005; Martinuzzi *et al.* 2015). In turn, the area surrounding individual structures (30-60 m) where fuels closely intermingle with structures and largely determine structure ignition and loss is known as the home ignition zone (HIZ) (Cohen 2008; Calkin *et al.* 2014). Major factors explaining the WUI area increase on Mediterranean areas during the last decades include increased fuel loadings from an expansion of shrub and forest vegetation into abandoned agricultural lands surrounding rural communities, suburban sprawl over metropolitan agricultural belts of big cities, and new housing construction within wilderness areas looking for landscape amenities in coastal areas.

Fuel load reduction in multifunctional cultural landscapes

Fuel treatments represent the most widely used measures to reduce wildfire hazard on treated areas and overall likelihood in large fire-prone landscapes. However, sparsely distributed treatments are inefficient to restrict large fire spread, and most landscape managers are doomed to mitigate wildfire risk implementing fuel reduction programs with very tight budgets. Treatment optimization represents a complex issue, and apart from the spatial arrangements depending on the treatment goal (Ager *et al.* 2013), program efficiency is conditioned to the treated area (percentage or fraction of the landscape) and treatment intensity (removed biomass on treated units) (Finney 2007; Salis *et al.* 2016b). Besides, most fuel management projects mostly rely on government subsidies due to the limited revenue from thinning in Mediterranean forest ecosystems, and budgetary constraints are currently seriously conditioning its viability. In order to find potential opportunities with other treatments pursuing different objectives such as game or protected species habitat restoration and community protection, recent studies explored joint-production benefits using a Pareto efficient curve for the objective trade-off analysis (Vogler *et al.* 2015). This approach is especially interesting for multi-functional Mediterranean cultural landscapes providing for many ecosystem services and products such as firewood, pastures, cork, game, and edible mushrooms. Here, the implementation of prescribed fire as a tool to manage hazardous fuels in extensive areas with non-commercial timber interests while preserving a fire-resistant forest structure presents a challenging opportunity for landscape managers (Fernandes *et al.* 2013).

Towards a long-term solution to coexist with fire

Large destructive events continue to grow in size and evidence the need for a new long-term solution to cope with fires in Mediterranean areas. The short-term fire exclusion policy based on human ignition prevention and fire suppression was decisive in reducing small fire number trends but mostly ineffective in containing uncharacteristic events. This *fire paradox* has been widely discussed in previous works and researchers emphasized the need for a long-term sustainable strategy where forest fuel reductions with prescribed fires could play a crucial role, rather than escalating economic investments to support the increasingly aggressive suppression policy (Rego *et al.* 2007; Silva *et al.* 2010; Bovio *et al.* 2017; Curt and Frejaville 2017).

However, limited efforts have been conducted to articulate and facilitate the implementation of these findings on managing vast landscapes encompassing various fire regimes (i.e., fire frequency and large fire size distributions). In other words, we lack standard procedures to make the wildfire risk mitigation measures effective in vast areas. In this Thesis, I partitioned the problem in four core goals, which parallel efforts in the US where similar concerns challenge to land managers and the research community (USDA Forest Service 2015): creating fire resilient landscapes, restoring the natural/cultural fire regime, supporting a safe efficient fire response, and promoting fire-adapted communities. In turn, each goal relies on specific management options including fuels management projects, lightning fire monitoring, cultural fire use, suppression resource pre-positioning, and community action projects.

OBJECTIVES

The main objective of this Thesis was to provide a wildfire risk management-oriented framework for Mediterranean multi-functional cultural landscapes while developing a cross-scale fuels reduction strategy to mitigate the negative impacts from the large events currently driving fire-regimes in southern European regions. Ultimately, this Thesis attempts to replace the widely enforced fire exclusion policy by a sustainable long-term comprehensive solution that may facilitate better coexistence with fire in Mediterranean areas.

Respective specific objectives for the different chapters of the dissertation published as separate scientific papers include the following:

- to assess wildfire hazard and likelihood risk causative factors,
- to quantify expected economic losses to values at risk,
- to scale the risk to residential communities in the wildland-urban interface,
- to optimize prescribed fire fuel treatments and explore trade-offs among competing objectives in multi-functional forest systems,
- and, to generate a comprehensive set of priority maps for wildfire management in a large fire-prone representative Mediterranean region.

METHODOLOGY

General framework

This Thesis is organized in 5 chapters, which escalate from the detailed assessment at individual value level or treatment unit (housing structure or forest stand) to the policy-making maps of vast areas (Regional level) (Fig. 1). Each chapter develops one of the five specific objectives. Some of the chapters contain more than one peer-reviewed paper to show the applicability of the framework on different Mediterranean areas. Likewise, the results and required geospatial data in the analyses required for some chapters were published as database papers. The landscape fragmentation at different scales in different chapters is related to the management capabilities of the different agents and depends mainly on land ownership, administrative boundaries, and large fire spread distance.

I first used wildfire simulation modeling to assess wildfire exposure to valued forest resources and assets (Alcasena *et al.* 2016b). Fire effects were then estimated to quantify expected losses over the primary values at risk while considering the economic market values (Alcasena *et al.* 2016a; Alcasena *et al.* 2017). Next, modeled large fire perimeters were used to delineate the community firesheds (Alcasena *et al.* 2017; Alcasena *et al.* 2019b), and explore fire exchange among the different administrative units or planning areas (i.e., municipalities) on vast landscapes (Alcasena *et al.* 2019a). The modeling results were combined with historical fire occurrence data to prescribe the most suitable management options at the municipality level. On the one hand, I identified the fire-prone blocks where fuel treatments (e.g., thinning, prescribed fires and mastication) can concentrate on effective intensities to restrict likely occurring large fires. On these high-priority areas, I also implemented a downscaling treatment optimization model for designing the fuel treatment mosaics as required by wildfire managers, while exploring the co-location opportunities among multiple forest management objectives (Alcasena *et al.* 2018a, 2018b). On the other hand, I analyzed the anthropic ignition prevention priorities and lightning re-introduction possibilities considering the potential impacts on neighboring human communities from escaped fires. In parallel to this, I used wildfire hazard and fire transmission to communities as a proxy to better understand the current possibilities for aggressive wildfire response. Lastly, I generated a wildland-urban interface (WUI) map (Alcasena *et al.* 2018c), that coupled with exposure profiles might help inform community protection plan implementation.

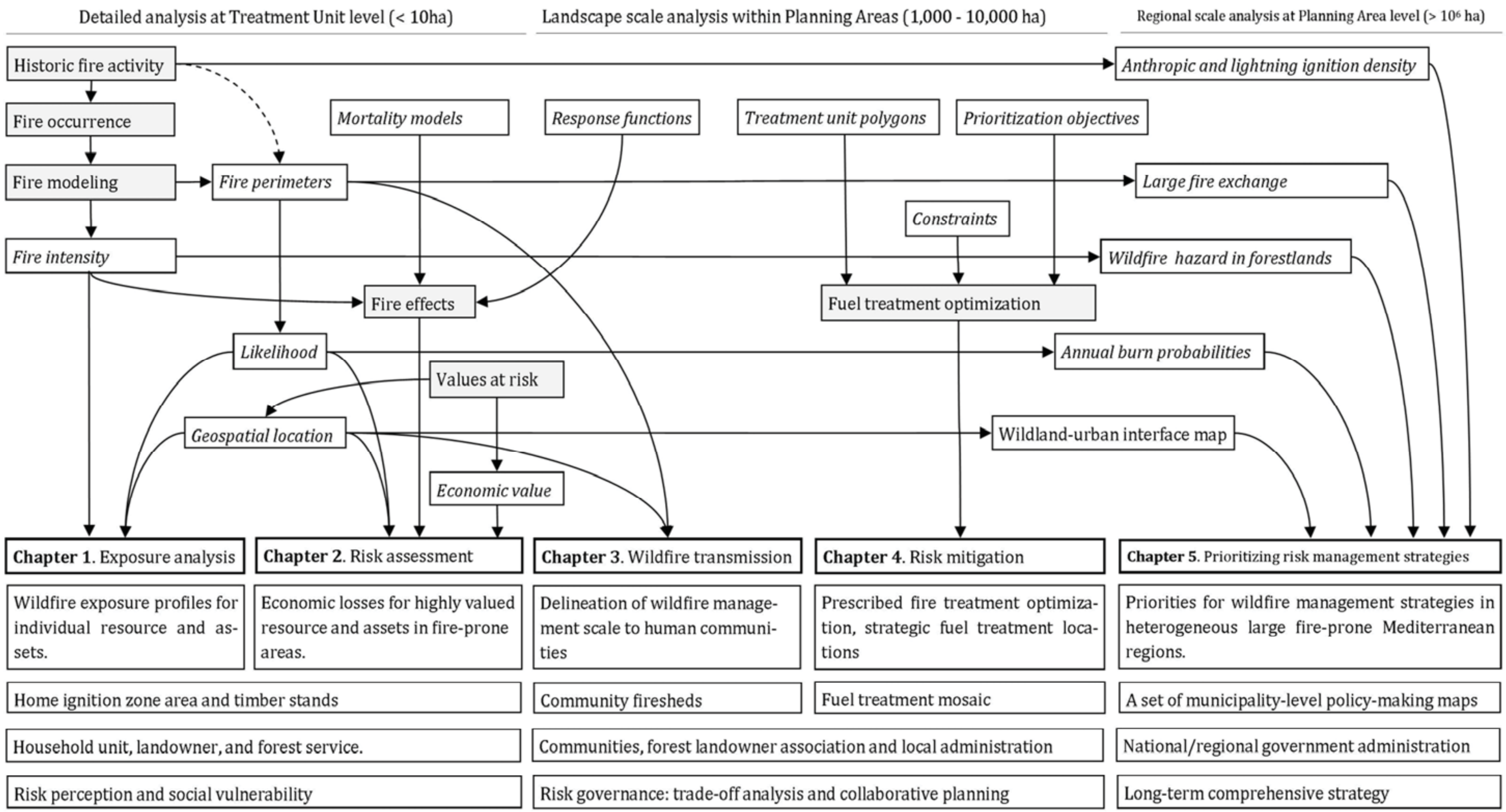

Figure 1. Streamlined cross-scale methodological framework conducted in the Thesis for exposure and risk assessment, community fireshed delineation, fuel treatment optimization and wildfire risk management strategy prioritization in large fire-prone landscapes.

Historical fire regime

Understanding fire activity is essential to calibrate fire spread models and replicate historical fire activity. For that purpose, we require fire occurrence data providing ignition location, date, cause, and fire size data for a 20-30 year series (ADCIF 2012). Longer time series could provide biased results since human activities cause most fires, and evolving socioeconomic factors trigger important changes over time that affect spatiotemporal patterns (Salis *et al.* 2014). While ignition location is used to model fire occurrence, the date and fire size are used to define the wildfire season, large fire numbers and size distributions within the fire modeling domain. In fire-prone areas, small fires are excluded from the analysis because they do not contribute significantly to the overall burned area (<15 %). The large-fire size threshold is usually between 100 and 1,000 ha and varies according to the total contribution to the burned area. Wildfire season is the period concentrating most large fire activity during the year (>90% accumulative burned area) and defines the timeframe to characterize most common fire-weather scenarios occurring during extreme events.

On vast landscapes, the subdivision in different fire-regime macro-areas might result in a pre-requisite when we observe substantial differences among large fire seasonality, size distribution, and perimeter shapes in different areas as a result of changing weather, topography, and fuels. While actual large fire number and size distributions provide the reference burn pattern baseline to calibrate the fire spread model, the mean annual burned area provides an empiric wildfire likelihood estimate to adjust the modeled outputs to the wildfire season or years in study areas. In the areas presenting recurrent large fires and long series with accurate fire perimeters, these data can be used to estimate the wildfire likelihood empirically. Nonetheless, fire modeling can more accurately capture all potential scenarios when increasing annual fire modeling replicates and can consider the post-fire changes in landscape fuels conditioning fire spread and intensity.

Fire occurrence

Fire occurrence models use observed fire ignition locations and geospatially variables to predict ignition probability. Fire occurrence models can be generated with different methods including logistic regression or artificial neural networks, and ignition probability grids can be then used to display ignition coordinates as required for fire spread and behavior modeling (Alcasena *et al.* 2016a; Alcasena *et al.* 2017). In Mediterranean areas most fires are caused by humans and variables related to anthropic activities such as distance to urban development, distance to a communication infrastructure (roads, forest tracks, and railways), distance to power-lines, land-use land cover classes and population density usually explain these models to a great extent (Rodrigues and De la Riva 2014). The geospatial variable dataset to construct the model is extracted using actual fire ignition coordinates and the same number of random non-fire observations. Fire occurrence models can be then used to generate fine-resolution ignition probability grids (< 50 m) encompassing the fire modeling domain, where a probability value is estimated at pixel-level from local geospatial variables. Logistic regression is useful to predict the presence or absence of a fire ignition probabilistically from a mixture of predictive variables that can be either continuous or categorical (Hosmer and Lemeshow 2000). On the other hand, the artificial neural network models are robust pattern detectors which can approximate mathematical relationships with non-normal distributions and spatially correlated variables where other statistical models could cause multicollinearity (Costafreda-Aumedes *et al.* 2017).

Wildfire simulation modeling

Fire modeling can be used to predict fire spread and behavior at a wide range of scales and resolutions. Required input data include ignition locations, landscape gridded geospatial information, wildfire season fire-weather scenarios, and the modeling settings that better replicate historical fire size perimeter distributions. Results are provided as fire perimeters or continuous cover fire intensity or likelihood grids.

Input data

The geospatial information assembled in the landscape file (LCP) contains the surface fuel, canopy metric (canopy height, canopy cover, canopy base height, and canopy bulk density) and terrain (aspect, elevation, and slope) grids (Finney 2006). In the absence of customized surface fuel data, it is possible to assign standard fuel models (Anderson 1982; Scott and Burgan 2005) to land cover class or habitat classes considering the information concerning grass, shrub and tree cover species, composition, density, and heights. On the other hand, LiDAR technology can capture the changing gradients in tree cover and structural variable at very high resolutions to facilitate the generation of canopy metrics (González-Olabarria *et al.* 2012). Weather data include wind (wind speed and wind direction) and fuel moisture content for live and herbaceous components. Automatic-weather-station hourly records provide accurate information to characterize the wildfire season fire-weather scenarios (Bradshaw and McCormick 2000). Fuel moisture content can be estimated with physical models or directly derived from species-specific long series sampling campaigns assessing moisture content trends under prolonged drought periods (Nelson 2000; Castro *et al.* 2003; Pellizzaro *et al.* 2007). Fires resistant to suppression efforts concentrate on few extreme weather episodes, and these conditions (97th percentile) are frequently considered to characterize dominant scenarios. For high-resolution modeling studies, mass-consistent wind models can provide spatially-explicit wind speed and direction input grids using historical records in automatic weather stations and digital surface models (Forthofer *et al.* 2014a; Forthofer *et al.* 2014b).

Large fire spread and behavior modeling

Fire spread was modeled using the minimum travel time (MTT) two-dimensional fire growth algorithm as implemented in FlamMap program and FConstMTT, the command line version (Finney 2006). The MTT algorithm replicates fire growth as a wavefront based on the Huygens' principle and finds the straight-line shortest path between the nodes in a fire-front network by calculating travel times from each cell corner to every other cell corner on the landscape (Finney 2002). Then the MTT algorithm calculates the fire behavior on the flow-path segments. Rothermel's semi-empirical model is used to predict surface fire (Rothermel 1972), fire intensity (kW m^{-1}) is converted to flame length (m) (Byram 1959), and crown fire activity (surface fire, passive crown fire, and active crown fire) is determined from surface fire intensity and canopy characteristics (Van Wagner 1977; Forestry Canada Fire Danger Group 1992; Van Wagner 1993). MTT allows modeling thousands of fires at a broad range of scales, and many previous studies have used this algorithm to model large wildfires in complex terrain and heterogeneous landscapes across the US, Mediterranean areas and elsewhere for different purposes (Alcasena *et al.* 2015; Jahdi *et al.* 2016; Palaiologou *et al.* 2018). Fires are simulated as short events escaping from the initial attack and assuming constant fuel moisture, wind speed, and wind direction weather conditions while excluding the fire containment effects on the fire perimeters.

To calibrate the fire spread model and validate the fuel models, I replicated the actual large fire perimeter size, average size, and fire size distribution. To do this, I assumed extreme fire-weather scenarios and adjusted the duration settings, while selecting the appropriate fuel type input. On the very large areas (> 1 million ha), the landscape was first subdivided in different fire-regime homogeneous macro-areas considering the wildfire season duration, fire response capabilities, weather conditions, dominant vegetation types, and terrain. Additionally, different coefficients can be used to analyze the shape agreement and accuracy between simulated perimeters and historical fire perimeters (Conalgton and Green 2008; Legendre and Legendre 2012). Overall models capture well the heading fire spread escaping suppression capabilities, and some overestimation is usually expected and observed in backing fire spread areas in smaller fires where firefighting containment efforts might result effective to some extent (Salis *et al.* 2016b).

Modeling outputs

The fire model provides a fire perimeter polygon for every fire ignition. Then the burn probability output is obtained from the juxtaposition of the fire perimeters, and calculated as a proportion value from the number of times a pixel burns and the total number of modeled fires (Finney 2005; Ager *et al.* 2007; Salis *et al.* 2013; Alcasena *et al.* 2015). The result is spatially explicit and facilitates the identification of the areas with a higher chance of burning, given a fire ignites within the study area under certain weather conditions. In the large fire-prone landscapes, wildfire-season extreme-weather scenarios are frequently used to calculate conditional burn probabilities. The results can be annualized considering the relation between the annually burned area on average (historical fire records) and the burned area in the output. Wildfire hazard outputs include the flame length and crown fire activity. The conditional flame length calculates the fire intensity on a given pixel from all the modeled fires while accounting for fire spread direction (backing, flanking and heading). The crown fire activity relates to extreme fire behavior, and modeling outputs can be provided as the fire type (surface fire, passive crown fire, and active crown fire) or the burned crown fraction (percentage).

Values at risk

The identification and location of the values at risk in the landscape is essential for accurate wildfire exposure and risk assessment. Overall, residential structures, communication network infrastructures, industrial sites, and small farm holdings are the primary values at risk. Among all, the human communities that closely intermingle with forestlands in the area known as the wildland-urban interface (WUI) represent the main concern to wildfire managers. In the WUI the population exposed to the fire is unusually high, and the housing destruction can account for very substantial economic losses. We can differentiate various WUI classes based on structure density, vegetation type in the HIZ, and distance to potential ember emitting forestlands (Martinuzzi *et al.* 2015; Alcasena *et al.* 2018c). While the structure density is related to total potential losses, the fuel loading on the 30 to 60 m buffer HIZ area determines the structure loss to a large extent (Cohen 2008; Calkin *et al.* 2014). On the other hand, the timberlands have represented the most relevant forest value on many sub-Mediterranean and temperate areas, where stand-replacing extreme events burned large areas and caused substantial economic losses to rural communities (González-Olabarria *et al.* 2017).

Wildfire exposure

The wildfire exposure analysis is a preliminary step required for risk assessment, and assembles spatially-explicit wildfire likelihood (e.g., burn probability) and intensity modeling outputs on the valued asset geolocation (e.g., WUI maps) (Miller and Ager 2013; Alcasena *et al.* 2016b). Wildfire exposure does not describe fire effects, and therefore the consequences are unknown. On the other hand, wildfire hazard determines the potential for loss without considering the likelihood of the fire event, and it is usually represented with fire intensity modeling outputs (e.g., flame length or crown fire activity). Some exposure outcomes including the high-intensity burn probability (HIBP) and the wildfire hazard potential (WHP) integrate both likelihood and intensity outputs into a single metric (Dillon 2015; Lozano *et al.* 2017; Alcasena *et al.* 2019a).

Wildfire transmission

The fire transmission concept is used to attribute or link the exposed assets (e.g., number of structures) or burned areas to the fire source area, which may be ignition location or a polygon (Palaiologou *et al.* 2018; Ager *et al.* 2019). Specifically, in this Thesis, the transmission analysis was implemented to assess the transmission to structures, delineate the community firesheds, generate fire transmission grids, and assess the fire exchange between neighboring municipalities (Alcasena *et al.* 2017; Alcasena *et al.* 2018b; Alcasena *et al.* 2019a). Large fire perimeters were first intersected with individual housing unit locations to assess the fire transmission (number of exposed structures) and transmission rates (number of exposed structures per burned). The large-fire ignitions that exposed the bulk of structures within communities were then used to delineate the fireshed extent contour. Those ignitions were used to generate transmission smoothed grids with geospatial interpolation methods. Finally, I used the municipality polygons instead of ignition locations to assess the burned area fire exchange (incoming, outgoing and self-burning) between neighboring blocks.

Risk assessment

Wildfire risk is the expected *loss* or *benefit* to any valued resource and asset and integrates the wildfire exposure profiles (wildfire likelihood and intensity modeling outputs) with fire effects (Finney 2005). Results are quantitative and can be presented as the expected economic loss.

Fire effects

We can use either mortality models or response function to estimate fire effects in values at risk, which will essentially depend on the susceptibility relation to fire intensity levels (Miller and Ager 2013). Mortality models result especially useful and accurate for conifer species, where the damage to the roots, cambium and tree crown determine the fire effects (Peterson and Ryan 1986; Fernandes *et al.* 2008; Temiño-Villota *et al.* 2016). Alternatively, expert-judgment derived response functions result very valuable to approximate complex susceptibility relations such as structure loss (Alcasena *et al.* 2017) and facilitate integrating multiple risk outcomes from different values and assets on a single wildfire risk map (Thompson *et al.* 2011).

Expected economic losses

After assessing wildfire risk in terms of expected net value change (Scott *et al.* 2013), we can use market values for the different goods or services to calculate the economic loss. While the cadastral value provides a reasonable estimate for individual housing units (Alcasena *et al.* 2019b), the awarding price in timber auctions from public forests represent an excellent and very realistic reference value for the different tree species and wood products on treatment units or stands (Alcasena *et al.* 2016a).

Wildfire risk management

Fuels reduction, human ignition prevention, fire suppression, making valued assets less susceptible to fire, and reducing social vulnerability are the different management options aiming at mitigating wildfire risk (Calkin *et al.* 2011; Penman *et al.* 2015; Paveglio *et al.* 2016). Fuel reduction treatments (i.e., prescribed fires, tree thinning and surface fuel mastication) at effective intensities on strategic locations mitigate wildfire hazard on the treatment site, restrict fire spread, and reduce overall burn probability in the landscape. Likewise, lightning and cultural fires occurring under mild weather conditions also reduce fuels, help prevent uncharacteristic events, and preserve not only the fire-dependent natural ecosystems but also the cultural landscapes. In Mediterranean areas, human ignition prevention campaigns during wildfire season in highly developed areas and forestlands with a massive and constant influx of visitors result very effective to reduce ignitions caused by negligence. Aggressive fire suppression (an improved resource pre-positioning plus a rapid fire detection and first dispatch deployment) is a feasible management option on low hazard areas such as managed agricultural landscapes where reducing burned areas can drastically mitigate the number of exposed human assets. In order to make residential houses, and structures in general, less susceptible to fire, we can use fire-resistant materials in enclosures, windows, and roofs. Implementing community protection plans that specifically address the existing limitations for the most vulnerable population (children, elderly and disabled) can help prevent fatalities during extreme fire events.

Fuel treatment optimization in multifunctional landscapes

In this Thesis, I further developed the wildfire risk management option based on fuels reduction, while considering forest management complexity in multifunctional forest systems (Alcasena *et al.* 2018a). I implemented the Landscape Treatment Designer optimization model (Ager *et al.* 2017) to design the multifunctional fuel treatment mosaics in high-priority areas for fuel treatment implementation including community firesheds and high fire-activity blocks (Alcasena *et al.* 2018a; Alcasena *et al.* 2019b). Prescribed fire fuel treatments are the most cost-effective and widely extended treatment type for low-revenue fire-adapted structures such as the even-aged mature Mediterranean pine forests, and this model allowed considering stand-level mortality thresholds to prevent undesired effects on young and thick vegetation patches.

The model requires a treatment unit polygon file, where individual polygons are attributed to quantitative metrics that relate to the different management objectives (Alcasena *et al.* 2018b). We assume that the objective attainment of a project (treatment unit cluster) is correlated with the metric values on treated units. The model requires the weights for the objectives (project priorities) in the analysis. Moreover, the units can also be populated with treatment threshold metrics (e.g., wildfire hazard or tree mortality) that allow selecting stands meeting some specific conditions. Protected sites or the land tenures where treatments are not allowed can be excluded with a binary flag. Other settings include the treatment aggregate option and the treated area per project within planning areas. The treated area is usually a constraint due to the budgetary restrictions. The output is a polygon file where the program identifies a treatment unit mosaic for the optimal solution. The total attainment for the different objectives at the project level can be directly estimated, as a percentage, from the values on treated areas with respect to the total in the study area.

Trade-off analysis and spatial collocation opportunities among competing objectives

Pareto efficient production possibility frontier curves (PPFs) generated from objective attainment values result very valuable to explore the trade-offs among competing objectives, and assess the spatial collocation opportunities (Alcasena *et al.* 2018a; Alcasena *et al.* 2019b). This is because treating a particular forest stand we can target more than one objective (e.g., wildfire risk reduction, habitat restoration, and timber harvesting). Using the LTD maximized output set, I generated the PPF curves for various objectives; every point in the curve represents the maximal mix for two objectives as prioritized with specific weights. A convex hull respect to the origin indicates high co-location possibilities where the points in the curve indicate the set of maximal-mix optimal solutions. By contrast, a straight line (i.e., constant slope) denotes nonexistent maximization possibilities because treatments on a given unit only would achieve a unique objective. Obtaining revenue from thinning would increase the available budget for treatments (and treated areas), and this would shift the curve outwards and set a new PPF with higher attainment values for both objectives. Technological improvements can increase the attainment value for a given objective (e.g., increasing harvesting capabilities on steep slopes would increase timber extraction possibilities). While the points below the curve are inefficient solutions, the points above the curve are impossible to achieve.

Prioritizing wildfire risk management options in large areas

Vast Mediterranean landscapes present widely changing fire-regimes, complex development patterns or community archetypes, multiple land tenures with different restriction levels for management, and a finely grained mosaic with a myriad of smallholders. Prioritizing the different wildfire risk management options aimed at mitigating wildfire risk is a very challenging issue for landscape managers, but essential for policy-making. In this Thesis I developed a framework to generate a set of maps to prioritize the main wildfire risk management options aimed at creating fire resilient landscapes, restoring cultural fire regimes, facilitating safe efficient fire response, and creating fire-adapted communities (Alcasena *et al.* 2019a). I first used advanced simulation modeling methods to assess various wildfire exposure metrics across spatially changing fire-regime conditions, and these outputs were then combined with land use maps and historical fire occurrence data to prioritize different fuel and fire management options at the municipality level.

To *create fire-resilient landscapes*, I used fire transmission and fire intensity metrics to target fuel treatments on "large-fire source hazardous municipalities" and transform them into "low-intensity fire containers". Low percentage area dispersed and random treatments across vast landscapes are rarely able to contain large fire spread (Finney et al. 2007), and thus I identified large fire source blocks or municipalities (i.e., planning areas) to allow the concentration of treatments there at effective intensities. Since large fires spread for long distances and affect socioeconomic and natural values far beyond the ignition location (Ager et al. 2016), concentrating treatments on these priority municipalities would substantially mitigate wildfire risk not only on property and natural ecosystems contained in it, but also in the neighboring municipalities receiving outgoing fires. Also, I considered wildfire intensity on forest stands within municipalities (i.e., treatment units) as a criterion to identify hazardous areas and exclude managed lands where agricultural and herbaceous fuels are still dominant.

For the purpose of *restoring the cultural fire regime,* I combined undesired human ignition prevention and natural fire re-introduction management options, using fire transmission to the residential housing structures and fire occurrence (lightning and human fire ignition density historical records) metrics to assess management priorities. This strategy is aimed at preventing human ignitions that affect communities but facilitating lightning fires on fire-adapted natural ecosystems. Traditional fire use for pasture clearing and waste elimination is also a component in this strategy where these fires do not represent a threat to neighboring communities and natural values. As a first approximation to identify the areas where ignitions would potentially cause high potential losses and smoke concerns, I identified municipalities where ignited fires expose a high number of structures. While only the human ignitions can be prevented, integrating lightning ignitions to reduce fuels on forest ecosystems is only suitable in specific areas concentrating a substantial lightning fire activity.

The strategy to *emphasize a safe and efficient fire response* was mapped by identifying the municipalities with best opportunities for full suppression considering fire transmission rate to structures (i.e., number of exposed structures per burned area) and fire intensity modeling results (see methods). A quick response on initial attack and aggressive fire suppression play a key role in reducing large fire potential and protecting property during wildfires when extreme fire behavior does not overwhelm firefighting capabilities (Andrews *et al.* 2011; Syphard *et al.* 2014; Penman *et al.* 2015). I focused fire suppression in locations where firefighting activities conducted under operational safety (low predicted intensity) can reduce fire size from the potentially catastrophic events exposing large numbers of residential houses. By directing suppression efforts towards high transmission rate areas, we can discriminate between fires affecting very large portions in the landscape with little socioeconomic interest, and those smaller events impacting densely populated communities in the wildland-urban interface.

Lastly, to promote *fire-adapted human communities* I identified highly exposed communities where implementing defensible space in the home ignition zone should be a priority. We considered the number of residential structures in the WUI and the overall wildfire exposure as derived from simulation modeling to prioritize management. Transforming vulnerable communities into a fire-adapted secure areas can provide safe confinement for the local population (and firefighters), mitigates losses to property and creates an opportunity for a safe fire response. The home ignition problem is mainly an issue related to structure resistance and fuels management at the site, and highly exposed communities are usually the priority for urban planning and fuel treatment allocation (Calkin et al. 2014). Accordingly, we identified the municipalities with communities likely burning at high intensities where building ordinances are needed to exclude high exposure areas, promote fire-resistant structures and request homeowners' involvement in home ignition zone maintenance to reduce fuels (Cohen 2008; Penman et al. 2013). During extreme events, most human fatalities and losses to property occur in densely-populated WUI areas, and therefore we also considered the number of individual residential houses to prioritize management efforts (Haas et al. 2013).

CHAPTER 1 – Exposure analysis

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A fire modeling approach to assess wildfire exposure of valued resources in central Navarra, Spain

Fermín J. Alcasena[1,2] · Michele Salis[2,3] · Cristina Vega-García[1]

Abstract Wildfires are a growing threat to socioeconomic and natural resources in the wildland–rural–urban intermix in central Navarra (Spain), where recent fast-spreading and spotting short fire events have overwhelmed suppression capabilities. A fire simulation modeling approach based on the minimum travel time algorithm was used to analyze the wildfire exposure of highly valued resources and assets (HVRAs) in a 28,000 ha area. We replicated 30,000 fires at fine resolution (20 m), based on wildfire season and recent fire weather and moisture conditions, historical ignition patterns and spatially explicit canopy fuels derived from low-density airborne light detection and ranging (LiDAR). Detailed maps of simulated fire likelihood, fire intensity and fire size were used to assess spatial patterns of HVRA exposure to fire and to analyze large fire initiation and spread through source-sink ratio and fire potential index. Crown fire activity was estimated and used to identify potential spotting-emission hazardous stands. The results revealed considerable variation in fire risk causative factors among and within HVRAs. Exposure levels across HVRAs were mainly related to the combined effects of anthropic ignition locations, fuels, topography and weather conditions. We discuss the potential of fire management strategies such as prioritizing mitigation treatment and fire ignition prevention monitoring, informed by fine-scale geospatial quantitative risk assessment outcomes.

Keywords Wildfire risk · Wildfire simulation · Highly valued resources and assets · Mediterranean areas · Forest–rural–urban intermix

Introduction

Fuel load and continuity increased notably in southern Europe during the second half of the twentieth century due to fire exclusion policies, abandonment of marginal agricultural land, active conifer reforestation and reduced anthropic pressure on natural resources, mainly through firewood cutting and livestock grazing (Scarascia-Mugnozza et al. 2000; Loepfe et al. 2010; San-Roman-Sanz et al. 2013). Many mature forests are now dominated by a shrubby ladder fuel understory, and herbaceous pastures are being replaced by shrubby vegetation and young thicket forests (Lloret et al. 2002; Romero-Calcerrada and Perry 2004; Mouillot et al. 2005), which lead to more intense wildfires (Moreira et al. 2011). Changes in fuel load and continuity have been especially noticeable in areas around the northern rim of the Mediterranean basin, such as the pre-Pyrenees and the central Iberian Peninsula (Roura-Pascual et al. 2005; Vega-Garcı´a and Chuvieco 2006). In these areas, larger wildfires now threaten many rural–urban interfaces and ecosystems adapted to frequent and low-intensity fire regimes (Pausas et al. 2004; F u l ´e et al. 2008). For instance, the 1998 Solsones wildfire in the central Catalonian pre-Pyrenees burned more than 20,000 ha

✉ Fermín J. Alcasena ferminalcasena@eagrof.udl.cat

1 Agriculture and Forest Engineering Department (EAGROF), University of Lleida, Alcalde Rovira Roure 191, 25198 Lleida, Spain

2 IAFES Division, Euro-Mediterranean Center on Climate Change (CMCC), Via De Nicola 9, 07100 Sassari, Italy

3 Department of Science for Nature and Environmental Resources (DIPNET), University of Sassari, Via De Nicola 9, 07100 Sassari, Italy

where the previous largest fire events in that area burned few hundred hectares. Moreover, recent studies in the Mediterranean basin have highlighted that climate change projections suggest increasingly long and frequent heat weaves and greatly reduced summer precipitation (Gao and Giorgi 2008; Giorgi and Lionello 2008; Giannakopoulos et al. 2009; Arca et al. 2012), which are consistent with observed trends (Pal et al. 2004; Cardil et al. 2013a). This weather scenario will likely increase wildfire season duration and the frequency of weather conditions associ-ated with large fire events (IPCC 2014).

Annually, some 51,200 forest fires burn approximately 477,400 ha in southern European countries (from 1980 to 2013, in Portugal, Spain, France, Italy and Greece; Rodriguez-Aseretto et al. 2014) and over 85 % of fires are caused by anthropic activities (San-Miguel-Ayanz and Camia 2010). A small number of large catastrophic fire events are responsible for most of the burned area (Gan-teaume et al. 2013; San-Miguel-Ayanz et al. 2013) and greatest loss of highly valued resources and assets (HVRAs). These fires overwhelm fire suppression capa-bilities despite the fact that suppression resource levels and fire crew training are better than ever before (WWF 2006, Cardil et al. 2013b). In recent years, not only in the USA, Canada or Australia but also in the Mediterranean basin, extreme fire behavior events have exhibited fire-line intensities, spreading and massive spotting (Koo et al. 2010; Molina et al. 2010) that have made them resistant to suppression efforts until a change in weather (i.e., wind speed reduction and relative humidity increase) or in fuel load and continuity (Finney 2007; Werth et al. 2011). These events have challenged fire risk management activ-ities and policies and revealed the need to integrate fire risk mitigation into landscape management actions, through fire ignition prevention plans and strategies to reduce fuel load and continuity (Fernandes et al. 2013). Most previous attempts to implement landscape fire management and planning in the EU have led to coarse scales and static, non-quantitative assessment outcomes of limited utility for landscape fire managers (reviewed in Miller and Ager 2013). Nevertheless, recent studies based on quantitative fire modeling assessment frameworks have been developed for the southern EU at various scales (Kalabokidis et al. 2013; Salis et al. 2013; Mitsopoulos et al. 2014; Alcasena et al. 2015), as well as for the USA (Ager et al. 2014a; Thompson et al. 2012, 2015).

Fire modeling approaches that account for site-specific key drivers of wildfire spread can provide reliable burned area estimates, particularly where large wildfires are responsible for most of the burned area (Calkin et al. 2011; Miller and Ager 2013). In the Mediterranean areas, fire ignition location alone is a poor estimator of burned area, but fire spread modeling must account for historical igni-tion occurrence since anthropic activities are responsible for spatial–temporal ignition patterns (Bar-Massada et al. 2011; Ager et al. 2014b; Salis et al. 2014, 2015). Wildfires can be accurately and massively replicated using fire modeling programs such as FlamMap, FSim or Randig (Finney 2006), which are built on the computationally feasible and efficient minimum travel time (MTT) fire spread algorithm (Finney 2002). It has been extensively demonstrated in previous studies that MTT can accurately predict fire spread and replicate large fire boundaries for heterogeneous landscapes in the USA (Ager et al. 2010a, 2012) and the southern EU (Salis et al. 2013; Kalabokidis et al. 2013; Alcasena et al. 2015). MTT has been used for diverse purposes, such as endangered species habitat loss assessment (Ager et al. 2007), municipal watershed wild-fire exposure assessment (Scott et al. 2012), urban planning (Haas et al. 2013), measurement of the effects of fuel treatments on forest carbon (Ager et al. 2010b) and land-scape-level fuel treatment optimization (Finney 2007; Finney et al. 2007; Chung et al. 2013). Nevertheless, when used under different conditions to those in which the simulators were developed, accurate fire spread model calibration and input data validation are needed to generate reliable results (Arca et al. 2007; Salis 2008).

Advances in laser imaging detection and ranging (LiDAR) remote sensing technologies have facilitated the creation of high-resolution spatially explicit maps of canopy fuel metrics (i.e., canopy cover, canopy height, canopy base height and canopy bulk density; Scott and Reinhardt 2001), which improve these input metrics for wildfire behavior modeling (Andersen et al. 2005; Erdody and Moskal 2010; Garcı´a et al. 2011; Gonzalez-Olabarria et al. 2012a; Hermosilla et al. 2014). Other remote sensing technologies such as near-infrared aerial imagery have been also used for fuel model mapping (Fallowski et al. 2005; Arroyo et al. 2008), but only small-footprint dis-crete-return airborne LiDAR pulses can penetrate beneath the tree canopies to allow pixel-based reconstruction of three-dimensional forest structure characteristics for large regions. Canopy fuel metrics can be estimated from LiDAR point cloud statistically derived regression models and processed at broad scales using analytical tools (Mc Gaughey 2014). Previous studies have suggested that the use of LiDAR canopy fuel metrics as input data could allow for more realistic predictions of fire spread and intensity (Mutlu et al. 2008), particularly in crown fire modeling (Peterson and Nelson 2011). In most previous studies, however, canopy fuel metrics have been derived from low-resolution layers and expensive field surveys, where pixel data are spatially homogeneous within stands or fuel models. Current efforts to increase LiDAR data availability for large areas, such as the PNOA project (*Plan Nacional de Ortofotografi´a Ae´rea*; Ministerio de Fomento 2010), which provides low-density (0.5 first returns m^{-2}) airborne LiDAR data for the whole of Spain, permit the estimation of pixel-based canopy fuel characteristics (Gonza´lez-Ferreiro et al. 2014) and are invaluable resources for fire managers, as this study demonstrates.

Beyond fire modeling, fire risk is defined as the expected *loss* or *benefit* to any number of socioecological values affected by fire, and its calculation requires an understanding of the spatially explicit burning likelihood and the value change in resources from fire intensity (Finney 2005). Consequently, quantitative fire risk assessment must encompass three major elements: (1) estimation of the spatially explicit fire likelihood and intensity across the territory; (2) geospatial identification of the HVRAs that could undergo a change in value due to wildfire; and (3) estimation of this change in value change in response to fire (Thompson et al. 2011; Miller and Ager 2013; Scott et al. 2013). By contrast, wildfire exposure analysis requires the geospatial overlapping of the causative risk factors with the location of each HVRA, and although it does not explicitly consider the potential impacts of fire (Miller and Ager 2013), it is more than adequate for wildfire risk assessments (Ager et al. 2012; Salis et al. 2013; Kalabokidis et al. 2013) and mitigation planning (Ager et al. 2010a). Fire effects on HVRAs have been analyzed in integrated fire risk assessment frameworks through the use of expert appraisals of fire intensity versus net value change response functions (Calkin et al. 2011; Thompson et al. 2012), but the difficulty of distinguishing between low-likelihood high-hazard events and high-likelihood low-intensity events can critically undermine risk assessment and miti-gation planning.

The goal of this study was the methodological implementation of an improved fine-scale quantitative wildfire exposure assessment for HVRAs, as well as the identification of the likely areas of large fire initiation and spread, to better inform landscape-level fire management in a forest–rural–urban intermix area in central Navarra (Spain). Our approach integrates a pixel-based LiDAR canopy fuel characterization that uses the MTT algorithm to model burn probability (fire likelihood estimate; Ager et al. 2010a), conditional flame length (fire intensity esti-mate; Scott et al. 2013) and fire size (Calkin et al. 2011; Ager et al. 2012) fire risk causative factors affecting HVRAs. We also performed complementary analyses of likely large fire initiation areas using the fire potential index (Salis et al. 2013), large fire transmission using the source-sink ratio (Ager et al. 2012) and likely active torching and ember-emitting stands based on active crown fire probability.

Materials and methods

Study area

The study area is located in the northern-rim Mediterranean area of the Pamplona Basin (Autonomous Community of Navarra, Spain) and encompasses a rectangular frame of 28,000 ha (Fig. 1). The study area is limited by the regional capital Pamplona to the southwest, where most of the population lives (in Pamplona and the neighboring towns, with a total of ~250,000 inhabitants; www.navarra.es), while the central and northern parts present a very low population density and a highly scattered rural–urban intermix, characterized by sparsely distributed small villages with fewer than 150 inhabitants.

The orography consists of open and flat areas in the southern part in contrast with rough mountainous terrain in the northern part, ranging from 375 m in the south to highest peaks of 1100 m in the north (Fig. 1). Most watersheds present small watercourses that flow pre-dominantly to the south to converge at the Arga River. The climate is transitional Mediterranean in lower ele-vations to temperate in the mountains, with cool sum-mers and abundant precipitation, though with two dry months. The average annual rainfall of ~900 mm is evenly distributed from autumn to spring, and water shortages occur from July to mid-September. The mean annual temperature is ~12 °C, ranging from ~6 °C in the coldest month to ~21 °C in the warmest month, but easily peaks above 35 °C on summer days. The average wind speed in summer is moderate (21.6 ± 10.8 km h^{-1}), and the most frequent wind direction is NW–N, with southerly wind less frequent but also common (http://meteo.navarra.es).

The vegetation in the study area (Fig. 2) corresponds to *Roso arvensis-Quercetum humilis* phytosociological vegetation series (Peralta 2000). Pine forests occupy a sizeable proportion of the area (11.3 %), consisting of *P. nigra* ssp. *austriaca* Endl. afforestations in marginal agricultural lands and *P. sylvestris* L. natural forests in the northeastern mountains. Broadleaf Mediterranean oak woodlands (18.4 %) occupy south-facing slopes, while *Q. pubescens* Willd. are found at mid-high altitudes and *Q. ilex* L. at low altitudes. Some north-facing slopes in the northern mountains are occupied by mature *Fagus sylvatica* L. stands (5.4 %). Shrubby pastures are a patchy mosaic of *Juniperus communis* L., *Rosa canina* L., *Echinospartum horridum* (Vahl.) Rothm. and *Prunus spinosa* L. formations (10.6 %), which intermingle with, or are replaced by, *Genista scorpius* (L.) DC., *Thymus vulgaris* L. and *Quercus coccifera* L. in shallow soils and rocky areas (0.2 %).

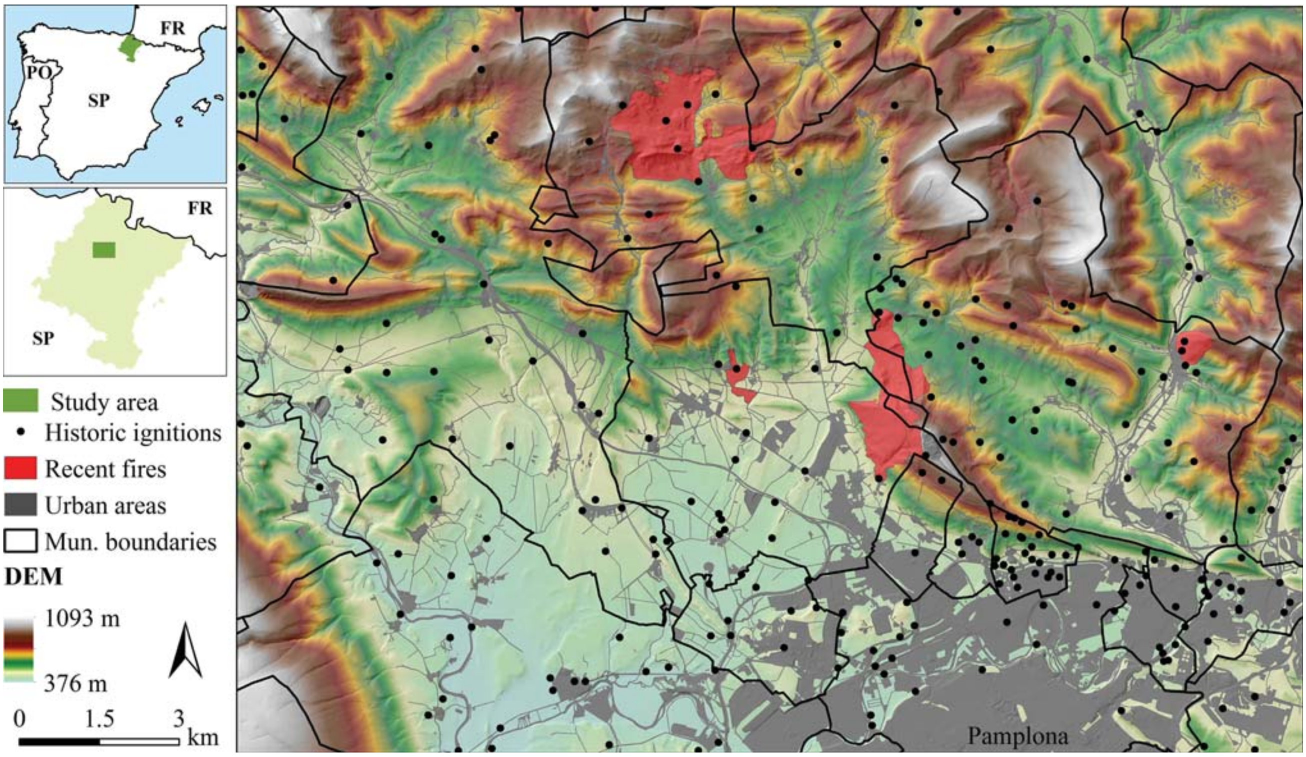

Fig. 1 Map of elevation, urban areas, historical ignitions (20 years, data for 1985–2012; EGIF, MARM 2012), recent fires (2000–2012; Government of Navarra pers. comm., 2012), and municipal boundaries in the study area (28,000 ha), located in central Navarra (northern Spain). The municipality of Pamplona is located in the southeastern of the study area

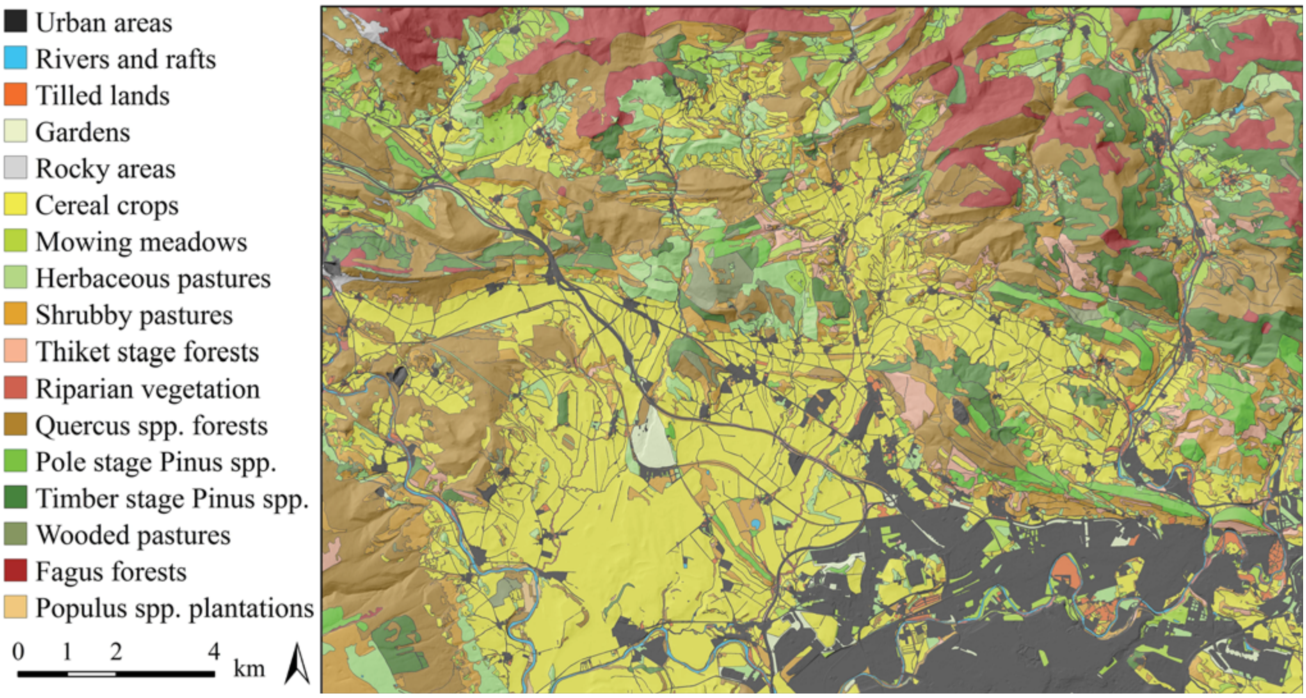

Fig. 2 Map of the vegetation types in the study area, derived from the SIGPAC 2014 (http://sigpac.navarra.es) and the 2012 Crops and Land Use Map (http://idena.navarra.es) themes for fuel model assignments

Riparian vegetation is restricted to the major watercourses where *Populus nigra* L., *Fraxinus angustifolia* Vahl. and *Salix atrocinerea* Brot. predominate, with a dense and closed shrubby *Rubus fruticosus* L. stratum (0.5 %). Natural herbaceous Meso-xerophytic pastures (e.g., *Brachypodium pinnatum* L., *Bromus erectus* Huds. and *Trifolium pratense* L.) cover transitional areas between forests and cultivated lands (6.3 %) and are usually managed as extensive livestock-grazing areas. Rainfed cereal crops (i.e., *Triticum aestivum* L., *Hordeum vulgare* L. and *Avena sativa* L.) occupy the valley bottom and areas suited to mechanization (26.1 %), whereas in the wetter northern areas, cereal crops are replaced by hay meadows (4.1 %; e.g., *Lolium perenne* L., *Agrostis capillaris* L. and *Arrhenatherum elatius* (L.) Beauv.). Urban developments, which are mainly concentrated in the southwestern part of the study area, occupy 13.7 % of the land. Land ownership is highly fragmented, and forest areas are mainly public and owned by the corresponding authorities. Local authorities are responsible for forest management under the supervision of the regional forest service.

Wildfire history

The study area is one of the most hazardous and fire-prone regions in the Autonomous Community of Navarra, both in terms of fire number and annual burned area (0.046 fires km^{-2} $year^{-1}$ and 0.2 % of surface burned $year^{-1}$ on average; MAGRAMA 2014), with simultaneous episodes that have overcome fire suppression capabilities and threatened HVRAs in recent years (e.g., Juslapen˜a and Izagaondoa wildfires in 2009). Small fires (< 10 ha) account for only 10 % of the overall burned area despite constituting 94 % of the fire number, whereas the less frequent large fires (> 100 ha, and 1.3 % in fire number) are responsible for 56 % of the burned area; no fires larger than 1000 ha have been reported in the database period (Spanish EGIF database 1985–2012, MAGRAMA 2014; Fig. 3a). The wildfire season usually falls between July and September and is responsible for 86 % of the burned area recorded over the study period; it is followed by a less severe late-winter/spring season (Fig. 3b). Large inter-an-nual variability in burned area has been also reported, due to differences in weather and fuel moisture conditions during wildfire seasons: The largest burned areas were reported in 1991, 2005 and 2009. Although the causes of many fires in the study area are still unknown, the main known causes of fire ignitions are (Fig. 3c): agro-pastoral burning (20 %), arson (12 %), engines and machines (3 %), railways (3 %), power lines (2 %) and lightning (2 %).

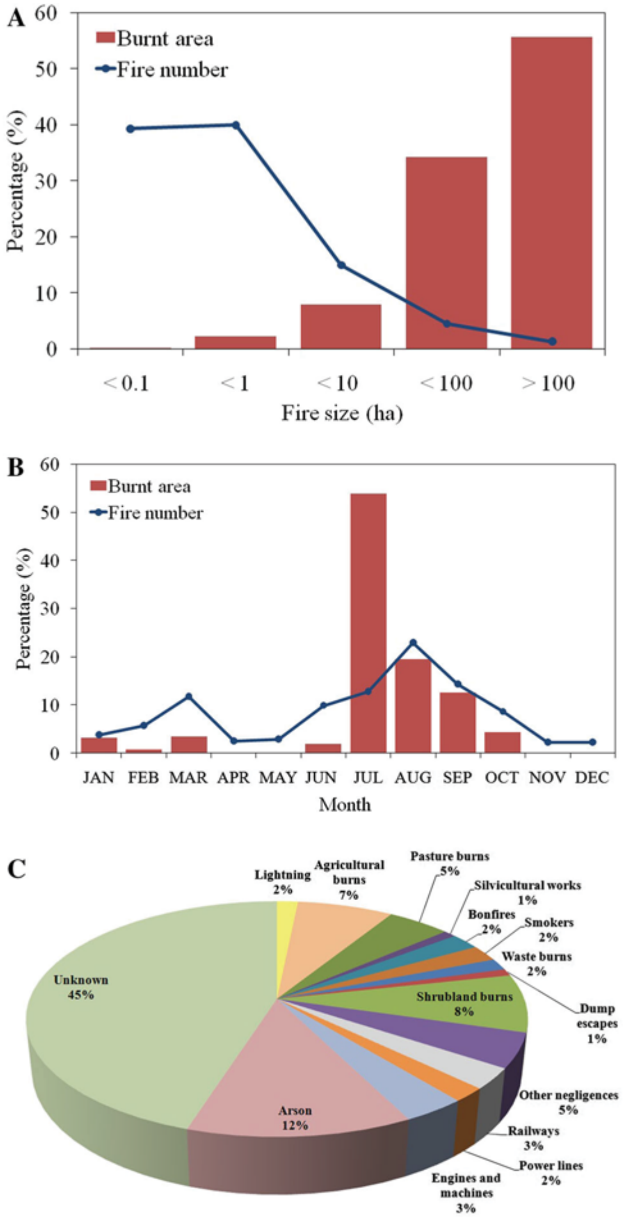

Fig. 3 Wildfire history. Burned area and fire number by fire size category (**a**), monthly frequency distribution (**b**) and fire ignition causes (**c**) in the study area in central Navarra (Spain), from the period 1985–2012 (MAGRAMA 2014)

Input data for fire spread and behavior modeling

Fuel model input data (i.e., surface fuels and canopy fuel characteristics) and topography (i.e., aspect, elevation and slope) input data layers were assembled in a 20-m-resolution landscape file (.LCP), as required by FlamMap (Finney 2006), using ArcFuels 10 (Ager et al. 2011). The surface fuel map was derived from the land use/land cover typologies of the 1:5000 scale Agricultural Plots GIS shapefiles (SIGPAC, http://sigpac.navarra.es; Gobierno de

Navarra 2014a), where herbaceous, shrubby and forest land cover formations are accurately delimited. Patches classed as forested in SIGPAC without further specification were classified in fuel types using the Government of Navarra's Crops and Land Use Map (*Mapa de Cultivos y Aprovechamientos*, http://idena.navarra.es; Gobierno de Navarra 2014b), in which forest types are classified according to tree stand species composition and developmental stage (Table 1; Fig. 2). Standard fuel models (Scott and Burgan 2005; Fernandes 2009) were assigned to the land use/land cover types (Table 1) to obtain the surface fuel map of the study area (Fig. 2). Spatially explicit canopy fuel characteristics (i.e., canopy cover, canopy bulk density, canopy base height and canopy height) were obtained at 20-m resolution from low-density (0.56 returns m^{-2}) airborne LIDAR data (Gobierno de Navarra 2014c), with models from other studies (Gonzalez-Olabarria et al. 2012a) using Fusion software (Mc Gaughey 2014). The LIDAR flight was carried out under the supervision of the Government of Navarra in 2011–2012 by TRACASA S.A. using a Leica ASL60 sensor with a pulse repetition rate of 97 kHz, a scan frequency of 37.5 Hz, a maximum scan angle of 40° and an average flying height of 3315 m (Gobierno de Navarra 2014c); the results were integrated into the PNOA project (Ministerio de Fomento 2010). Elevation, slope and aspect input data were obtained from 5-m-resolution digital elevation map from the National Geographic Institute (IGN, ign.es; Ministerio de Fomento 2010).

Wind speed and direction for fire modeling were determined from wildfire season data recorded at the Airport of Noain, at the standard height reference of 10 m (July–September, 1998–2013; AEMET pers. comm. 2014), at which wind data are considered representative for the study area and are not influenced by topography. Two dominant wind directions (43 % northwest and 21 % north; Fig. 4) were most frequent during the wildfire seasons, with south winds also recorded (15 %; Fig. 4). We set as a modeling reference the 97th percentile of wind speed for every wind direction during the wildfire season (Fig. 4). In order to obtain more realistic wind field input data to inform wildfire simulations, we used a mass-consistent model (WindNinja; Forthofer et al. 2014a, b) to generate 50 m resolution wind field grids, considering 12 wind speed and direction scenarios (Table 3). WindNinja com-putes spatially varying wind fields from elevation, a domain-mean initial wind speed and direction, and speci-fication of the dominant vegetation data in the area (Forthofer and Butler 2007).

The information on dead fuel moisture emulated the conditions of the Izagaondoa 2009 wildfire, which was the default choice for replicating extreme wildfire conditions in

Table 1 Vegetation types and respective fuel models and fuel moisture contents used for wildfire simulations. Dead fuel moisture contents where associated at recent extreme fire events and live woody fuel moisture content, as well as crown foliar moisture content (85 %) where derived from bibliography (Chuvieco et al. 2011). *na* not applicable

Vegetation type	Incidence (%)	Fuel model	1-h fuel (%)	10-h fuel (%)	100-h fuel (%)	Live herb. fuel (%)	Live woody fuel (%)
Urban areas and development	13.7	NB1 (Scott and Burgan 2005)	na	na	na	na	na
Rivers and rafts	0.7	NB8 (Scott and Burgan 2005)	na	na	na	na	na
Orchards, tilled lands	0.9	NB3 (Scott and Burgan 2005)	na	na	na	na	na
Gardens and golf courses	0.7	GR1 (Scott and Burgan 2005)	11	13	15	100	100
Rocky areas	0.2	GS1 (Scott and Burgan 2005)	6	8	10	50	55
Cereal crops	26.1	GR5 (Scott and Burgan 2005)	6	8	10	40	55
Mowing hay meadows and grazed pastures	4.1	GR2 (Scott and Burgan 2005)	6	8	10	40	55
Herbaceous pastures	6.3	GR4 (Scott and Burgan 2005)	6	8	10	40	55
Shrubby herbaceous pastures	10.3	SH6 (Scott and Burgan 2005)	6	8	10	50	60
Thicket-stage forests and shrublands	1.9	SH5 (Scott and Burgan 2005)	6	8	11	60	80
Riparian vegetation	0.5	SH8 (Scott and Burgan 2005)	6	8	11	60	85
Quercus spp. forests	18.4	TU3 (Scott and Burgan 2005)	7	9	11	60	80
Pole-stage *Pinus* spp. plantations	2.7	PCL (Fernandes 2009)	7	9	11	60	75
Timber-stage *Pinus* spp. plantations	6.7	SH3 (Scott and Burgan 2005)	7	9	11	60	75
Wooded pastures	0.9	GR3 (Scott and Burgan 2005)	7	9	11	60	80
Timber-stage *Populus* spp. plantations	0.1	SH3 (Scott and Burgan 2005)	8	11	14	60	80
Fagus spp. Forests	5.4	TL2 (Scott and Burgan 2005)	9	11	14	60	85

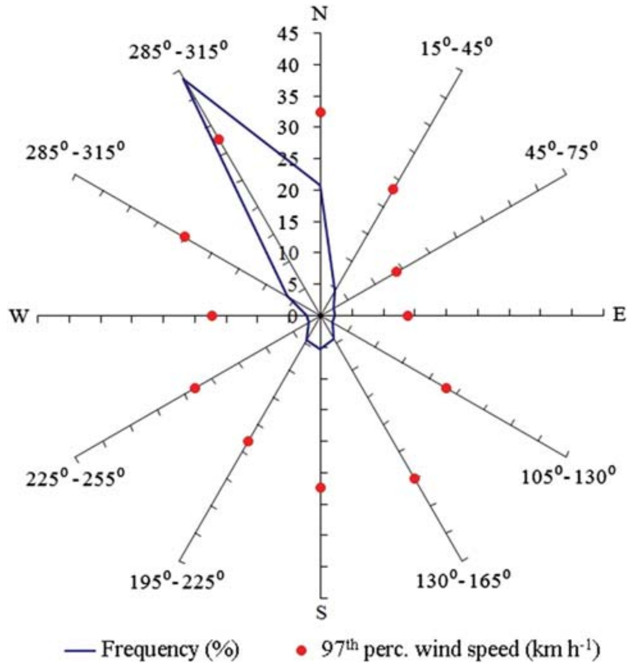

Fig. 4 Wind direction frequency and 97th percentile wind speed (km h^{-1}) rose for the July–September wildfire season (data from the period 1998–2013; AEMET pers. comm. 2014)

central Navarra (Fire Service of Navarra pers. comm. 2013). Live woody fuel moisture content for surface fire spread and foliar moisture content for crown fuels were also selected, in agreement with species and vegetation-complex data derived from sampling campaigns conducted in Spain in recent years (Chuvieco et al. 2011). We con-sidered the observed 97th percentile values of the annual fuel moisture records, to take into account the conditions most frequently associated with the peak wildfire season (Table 1).

In order to replicate the observed fire ignition spatial pattern (most ignitions occur close to main roads and urban developments; Fig. 1), we used the historical fire ignition coordinates (ADIF database 1985–2012; MAGRAMA 2014) for the study area to create an input file of 2500 historical ignition points for fire modeling. Initially, a kernel-smoothed point density grid was constructed from the observed ignition locations (Gonzalez-Olabarria et al. 2012b), with a bandwidth (search radius) of 1000 m; it was then divided by the number of years in the fire record to create an historical ignition probability (IP) grid, and the 2500 fire ignition points were drawn (Kalabokidis et al. 2013; Salis et al. 2013; Alcasena et al. 2015).

Wildfire simulations

Fires were simulated using the MTT fire spread algorithm (Finney 2002) as implemented in FlamMap 5 (Finney 2006), which requires geospatial input data on topography and fuels, as well as data on weather, fuel moisture content and fuel characteristics. The algorithm finds the straight-line shortest path between the nodes in a fire-front network, producing spatial data fields of arrival time (and other characteristics) recorded at discrete points (Finney 2006). Surface fire spread is predicted by the semiempirical Rothermel equation (Rothermel 1972), and crown fire initiation is evaluated according to Van Wagner (1977), as implemented by Scott and Reinhardt (2001). FlamMap assumes constant wind speed and direction within every pixel as defined in the wind grid, and constant fuel mois-ture content. It is therefore suitable for simulating short-duration fire events (Ager et al. 2011) like those recorded in the study area.

To calibrate the fire spread model and validate the standard fuel models assigned, we attempted to replicate the perimeter of the Izagaondoa wildfire, which started with the reactivation of a fire caused by lightning the previous day. It burned 873 ha (Gobierno de Navarra pers. comm. 2013) in 4.5 h of active spread on July 22, 2009 (Fire Service of Navarra, pers. comm. 2013). Fire crews worked mostly on the fire flanks and rear, but no data were available to account for the influence of suppression efforts on the final burned area. The fire developed under strong southern winds of over 40 km h^{-1}, low atmospheric rela-tive humidity (under 15 %) and with several spotting fires, leading to an average rate of spread above 20 m min^{-1} (Fire Service of Navarra, pers. comm. 2013). Multiple simulations were run (10 simulations at 10 m resolution) to analyze the agreement between observed and simulated perimeters, since the predicted burned area changes from simulation to simulation due to the stochastic behavior caused by spot fires in any run (Cochrane et al. 2012), although all input data and parameters were kept constant (Fig. 5). Simulation overestimation of backing and flanking fire spread areas was expected and observed, since sup-pression efforts that contained the fire spread in these areas were not considered in the model (in any case, containment activities have a very limited influence on heading fire spread during extreme fire events). The average simulation accuracy for the burned area, as measured by the Sorensen coefficient, was 0.50 ± 0.07 (Legendre and Legendre 1998), yielding an overall accuracy of 0.80 ± 0.07 (Con-algton and Green 1999).

Twelve simulations were run, considering a set of wind scenarios (different wind directions and wind speeds), constant fuel moisture content and the 2500 historical fire ignitions (Table 3). Overall, 30,000 fires were simulated at 20-m resolution, with a 0.10 spot probability and a spread duration of 8 h. Simulated fires were large enough to burn the pixels more than 100 times on average and over 97 %of the burnable area at least once. The simulation outputs were burn probability (BP), conditional flame length

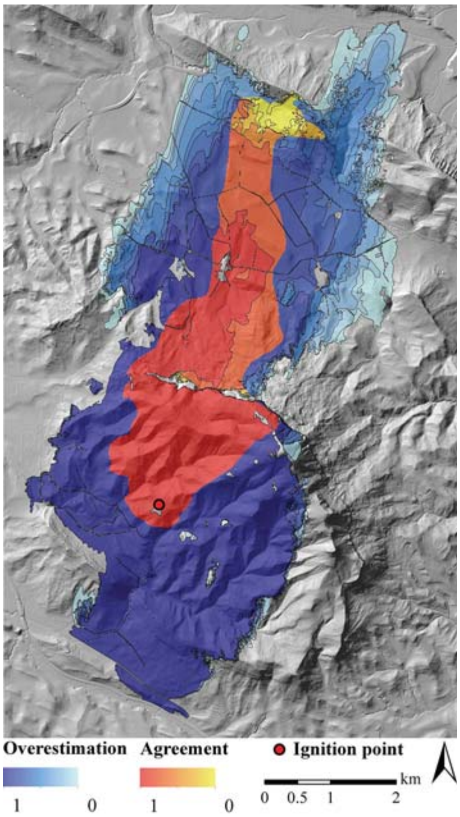

Fig. 5 Degree of agreement (0 to 1, warm colors) and overestimation (0 to 1, cool colors) in the Izagaondoa fire (on July 22, 2009, 873 ha burned) replication considering the observed final perimeter (Gobierno de Navarra pers. comm. 2014). The real fire showed important spotting distances reaching 300 m. The FlamMap MTT simulation did not take into account the attempted suppression of backfire spread or flanking. The fire was simulated at 10 m resolution, using 50 m resolution gridded wind fields

(CFL), fire size (FS) and crown fire activity (CFA). Burn probability defines the number of times a pixel burns as a proportion of the total number of fires and is defined as follows:

$$BP_{xy} = \frac{F_{xy}}{n_{xy}} \quad (1)$$

where F_{xy} is the number of times the pixel xy burns and n_{xy} is the number of simulated fires. In other words, the burn probability for a given pixel is an estimate of the likelihood that the pixel will burn given a single fire ignition in the study area and the assumed fuel moisture and weather conditions (Ager et al. 2010a; Salis et al. 2013).

Wildfire intensity depends on the direction from which the fire reaches a pixel relative to the major direction of fire spread (i.e., heading, flanking or backing fire) and on slope and aspect (Finney 2002). FlamMap converts fire-line intensity (FI, in W m^{-1}) to flame length (FL, in m) using Byram's (1959) equation:

$$FL = 0.0775 \times (FI)^{0.46} \quad (2)$$

Flame length distribution and BP were used to calculate the CFL for each pixel in the study area:

$$CFL = \sum_{i=l}^{20} \frac{BP_i}{BP} (FL_i) \quad (3)$$

where FL_i is the flame length (m; Eq. 2) midpoint of the ith category and BP is the burn probability (Eq. 1). For each pixel, FlamMap generates a frequency distribution of FL values (ranging from 0 to 10 m) that are divided into twenty 0.5-m fire intensity ranges. The CFL is the proba-bility-weighted FL assigned to a fire and is a measure of wildfire hazard (Ager et al. 2010a). Flame length is a consistent fire property that embeds severity and spread rate (Scott 2006). We also analyzed FS outputs, which provide the coordinates of the ignition points and burned area (ha) of each fire.

The fire potential index (FPI; Salis et al. 2013) was used to identify the areas with a greater likelihood of an ignition that could lead to a large fire, since almost all ignitions (98 % in the study area; Fig. 3) are caused by anthropic activities. The FPI was calculated using the FS and the historical ignition locations:

$$FPI = FS \times IP \quad (4)$$

where FS is the average fire size for all fires that originated from a given pixel and IP is the historical ignition probability. The FPI combines the historical ignition point probability with simulation outputs of FS to measure the expected annual burned area for a given pixel under the assumed weather and fuel moisture conditions (Salis et al. 2013).

In addition, we used the source-sink ratio (SSR; Ager et al. 2012) to measure wildfire transmission through the landscape, calculated as:

$$SSR = \log\left(\frac{FS}{BP}\right) \quad (5)$$

where FS is the average fire size for all fires that originated from a given pixel and BP (Eq. 1) is the burn probability. The SSR measures a pixel's wildfire contribution to the surrounding landscape relative to the frequency with which it is burned by fires originated elsewhere or ignited in the pixel. If an ignition occurs, pixels with high BP values that do not generate large fires behave as wildfire sinks, whereas pixels with low BP but large FS behave as wildfire sources (Ager et al. 2012).

Crown fire activity was also modeled with FlamMap for all cells of the landscape containing a forest stand. To determine crown fire activity, the surface fire-line intensity is compared with the intensity threshold that is critical to involving the overlying crown fuels. Crown fire typology (i.e., passive or active; Van Wagner 1977) is then determined from the rate of spread threshold for the current fire spread direction (Rothermel 1972). Using active crown fire and BP output grids, we identified those stands where active crown fire is likely to occur in the large fire event, through the active crown fire probability (ACP), calculated as:

$$\mathrm{ACP} = \mathrm{ACF} \times \mathrm{BP} \tag{6}$$

where ACF is the active crown fire-type occurrence (a binary value, 0 absence or 1 presence) of a pixel and BP (Eq. 1) is the related burn probability. Active crown fire probability can give crucial information about which stands are potentially responsible for ember emissions that could lead to spot fires, as well as high fire intensity areas where HVRAs could suffer severe fire effects.

Highly valued resources and assets

Wildfire simulation outcomes must be coupled with geospatial identification of the HVRAs whose value may be affected, in order to analyze the differences in wildfire exposure between HVRAs of different types and within patches or structures (Calkin et al. 2011). HVRAs are key social, economic and ecological resources which are exposed to wildfire effects (Thompson et al. 2011; Scott et al. 2013). In the current study, we focused our analysis on four major HVRA typologies (Table 2): urban development, infrastructure, natural habitats (92/43/CEE Directive, http://ec.europa.eu; European Community 1992) and forest resource values. Each type is broken down into several classes according to human presence, economic value and ecological value, using geospatial data themes (Table 2). We obtained the HVRA data themes from IDENA (http://idena.navarra.es; Gobierno de Navarra 2014b) and IGN (www.ign.es; Ministerio de Fomento 2010).

Graphical and statistical analyses

The 12 simulation output results for of BP, CFL, FS and CFA were combined weighting from the modeled wind scenario frequency (Table 3) to create the maps for the whole study area. The FS maps were obtained by filling the spaces between the smoothed data for simulated fire ignitions with a nearest-neighbor interpolation procedure (Ager et al. 2010a, 2012). The fire potential index (Salis et al. 2013), source-sink ratio (Ager et al. 2012) and ACP were derived from the combined maps of fire risk causative factors, IP and ACF.

Box plots of the main descriptive statistics were built to analyze the variations among HVRA classes (Table 2) for the modeled causative risk factors (BP, CFL and FS). To graphically assess the differences in wildfire exposure among and within HVRA units through scatter plots, we calculated the average causative risk factor values considering a 60 m buffer home ignition zone (HIZ; Cohen 2008) for the building structure classes and within feature patches for the land use/land cover and habitat designation classes (Table 2).

Additionally, we calculated the average BP and active crowning surface in forest stands to identify and compare the patches most likely to burn and emit embers in the event of an extreme wildfire in the study area. For these analyses, we used the standard fuel models for forested land cover (Table 4) and the current land registry property boundaries at 1:5000 scale (https://catastro.navarra.es; Gobierno de Navarra 2014b) in order to account for the implicit constraint of forest land ownership in the study area and its effect on the selection and implementation of wildfire management policies.

Results

Spatial variation of fire likelihood, fire intensity and fire size in the study area

Burn probability values produced a highly variable spatial pattern in the study area that ranged from a low of 8.0×10^{-4} to a high of 0.197 (Fig. 6c). The areas with the highest values ($\mathrm{BP} > 0.15$) were located on the northern and northeastern edges of the highly urbanized periphery of Pamplona (Fig. 2), corresponding mainly to cereal crops, *Pinus nigra* afforestations and Mediterranean oak forests. The highest average BP in the study area by land use/land cover was obtained for cereal crops, pole-stage afforestations and thicket-stage forests and shrublands, with values of 0.8×10^{-1}, 0.7×10^{-1} and 0.65×10^{-1}, respectively (Table 3). Sharp transitions have been observed on the border marked by the Arakil River (southwestern part of the study area), which created a large barrier, as well as in the most important infrastructure border lines (e.g., the north–south roadway in the eastern part), which contained the fires originated in the central part of the study area. Nevertheless, some areas with a high concentration of ignitions on the south-facing slopes of the San Cristobal mountain (the closest mountain to the southeastern urban area; Fig. 1) also contributed to the high BP values in the central area, even in southern-wind-driven fire scenarios (15 % frequency; Fig. 4). The lowest BP values were observed in the northern–northeastern beech forests (avg. $\mathrm{BP} = 9.9 \times 10^{-3}$; Table 3), where very few ignitions occurred and few fires arrived from elsewhere, and in gardens (average $\mathrm{BP} = 0.95 \times 10^{-2}$; Table 3) in urban areas where only a very small number of ignitions can burn

Table 2 Description of the highly valued resources and assets (HVRAs) considered in the study. The HVRAs were grouped into four types according to human presence, economic value and ecological value (http://idena.navarra.es; www.ign.es)

HVRAs types	Classes (abbreviation)	Number of sites or patches	Average size (ha)	Total area (ha)
Urban development	Residential housing (RH)	2572	0.036	93.201
	Industrial buildings (IN)	186	0.380	70.693
	Livestock farm buildings (FA)	122	0.030	3.605
	Churches and hermitages (CH)	33	0.036	1.168
	Cemeteries (CE)	53	0.243	12.859
	Sports areas (SA)	30	0.160	4.800
	Petrol stations (PS)	2	0.033	0.066
Infrastructure	Power lines (PL)	925	0.020	18.670
	Communication sites (CS)	8	0.001	0.008
	Water treatment plants (WT)	3	3.448	10.463
Natural habitats (Directive 92/43/CEE)	Oro-Mediterranean heaths with gorse (MH)	20	39.901	789.02
	Xerothermophilous scrub on rock slopes (XS)	3	51.137	153.431
	J. communis scrub on calcareous grasslands (JS)	2	4.268	8.535
	Seminatural dry grasslands (SG)	17	29.818	506.910
	Pseudo-steppe with grasses (PG)	3	84.222	252.666
	Mediterranean and thermophilous scree (TS)	1	1.456	1.456
	Chasmophytic vegetation on rocky slopes (CV)	2	6.860	13.719
	Medio-European limestone beech forests (MB)	22	54.658	1202.472
	Alluvial forests with *A. glutinosa* and *F. excelsior* (AF)	10	6.264	62.645
	S. alba and *P. alba* galleries (GA)	3	0.724	2.173
	Mediterranean sclerophyllous forests (SQ)	22	55.056	1211.23
Forest values	*Pinus* spp. commercial timber plantations (PI)	590	5.650	3333.370
	Firewood forests (FF)	776	8.553	6636.843
	Livestock-grazing natural pastures (GP)	976	4.049	3952.153
	Populus spp. plantations (PO)	10	2.418	24.180

individual plots. The urban areas in the southwestern part of the study area (Fig. 2) correspond to non-burnable fuels (e.g., paved areas) and therefore did not support fire spread.

Fire intensity values produced a complex mosaic pattern, ranging from 0.04 to 9.68 m (Fig. 6b). High-fuel-load models located in steep-sloping area showed the highest intensities (e.g., 90th percentile CFL = 7.33 in thicket-stage forests and shrublands; Table 3; Fig. 6b). On average, thicket-stage forests and shrublands, shrubby herbaceous pastures and cereal crops showed the highest intensities, with CFLs of 4.01, 2.63 and 2.53 m, respectively (Table 3). Nonetheless, riparian vegetation, shrubby pastures and pole-stage *Pinus* spp. afforestations also burned locally at high intensities (90th percentile CFL > 3 m; Table 3). The lowest intensities were observed in gardens, wooded pastures and broadleaf litter-type fuel models (i.e., beech forests) located on north-facing slopes in the northern part of the study area, with average intensities lower than 0.4 m (Table 3). The sharpest transitions were observed in sudden continuity changes from high to low fuel load models, as well as in areas where the alignment of slope and winds that drives heading fire spread is disrupted (e.g., the top of mountain edges). Overall, CFL values were consistent with the observed intensities of recent fire events in the study area (i.e., Juslapeña 2009 wildfire). Fire intensity was not affected by apparent spatial changes in non-burnable surface fuel continuity, whereas burn probability was affected by the spatial pattern of major water courses and communication infrastructure (Fig. 6b vs. a).

Fire size values revealed a clearly identifiable area in the northern-central part of the study area with the largest FS potential, where ignited fires covered an area of more than 3200 ha (Fig. 6c). There is also a large area with high FS

Table 3 Description of wildfire simulation parameters and associated values. The 12 wind scenario wind grids were generated with WindNinja from historical weather data for the wildfire season (Fig. 4)

Input data	Description												
Number of scenarios	12 scenarios												
Wind scenarios	Frequency (%)	5	3	2	2	4	5	4	2	2	6	44	21
	Direction (°)	30	60	90	120	150	180	210	240	270	300	330	360
	97th percentile speed (km h^{-1})	32	23	14	14	23	30	27	23	23	17	25	32
Fire ignitions per scenario	2500 historically based ignition points												
Surface fuels	Fuel model (Scott and Burgan 2005; Fernandes 2009) assignment derived from SIGPAC 2014 and Crops and Land Use Map 2012												
Crown fuel metrics	Derived from 0.56 point m^{-2} LIDAR point cloud (Gonzalez-Olabarria et al. 2012a)												
Dead and live fuel moisture	Izagaondoa 2009 wildfire conditions and observed 97th percentile moisture content (Chuvieco et al. 2011)												

Table 4 Fire simulation average and 90th percentile values for the different vegetation types in the study area (Fig. 1) for diverse fire risk causative factors (Fig. 6a–c). Non-burnable vegetation types have been excluded

Vegetation type	90th percentile BP (m)	Mean BP (m)	90th percentile CFL (m)	Mean CFL (m)	90th percentile FS (ha)	Mean FS (ha)
Gardens	0.0359	0.0095	0.253	0.163	1315	360
Rocky areas	0.0210	0.0135	1.306	0.930	3057	1905
Cereal crops	0.1506	0.0800	2.972	2.526	3495	1966
Mowing hay meadows and grazed pastures	0.0430	0.0228	1.381	1.005	3609	1843
Herbaceous pastures	0.0900	0.0426	2.977	2.053	4167	1993
Shrubby herbaceous pastures	0.1334	0.0582	3.968	2.630	3484	1798
Thicket-stage forests and shrublands	0.1191	0.0649	7.326	4.014	3579	1613
Riparian vegetation	0.1496	0.0450	3.431	1.983	2732	1171
Quercus spp. forests	0.0702	0.0341	2.062	1.497	3191	1931
Pole-stage *Pinus* spp. plantations	0.1621	0.0700	3.146	2.406	2435	1232
Timber-stage *Pinus* spp. plantations	0.1166	0.0433	1.433	1.023	3006	1693
Wooded pastures	0.0830	0.0428	0.524	0.395	2839	1632
Fagus spp. forests	0.0272	0.0099	0.509	0.249	3224	1780
Timber-stage *Populus* spp. plantations	0.0739	0.0345	1.215	0.791	1448	728

values in the western part, where ignited fires surpassed 1000 ha. By contrast, the southern and southwestern parts produced hardly any large fires of more than 1000 ha. Cereal crops and herbaceous pastures were the vegetation types with the highest average FS values, with almost 2000 ha (Table 3); gardens and *Populus* spp. plantations had the lowest FS results (<1000 ha; Table 3). Thicket-stage forests and shrublands did not show high average FS values, although locally ignited fires surpassed 3500 ha FS (90th percentile CFL; Table 3). Analysis of FS by distri-bution frequency (Fig. 7) showed that the bulk of fires ignited from the historical ignition pattern (>60 % fires) burned between 750 and 6000 ha, while small fires (0–175 ha FS class) account for only <5 % of fires.

Fire potential index and source-sink ratio

The source-sink ratio map (Fig. 8a) was used to identify the sink areas (low SSR) in the northern part of the study area, which were mainly low-spreading broadleaf forests (predominantly beech forests), where fires encroach from neighboring areas, and wildfire sources (high SSR), which were mainly housing-urban development borders (generally with higher values in the northern boundaries) in the

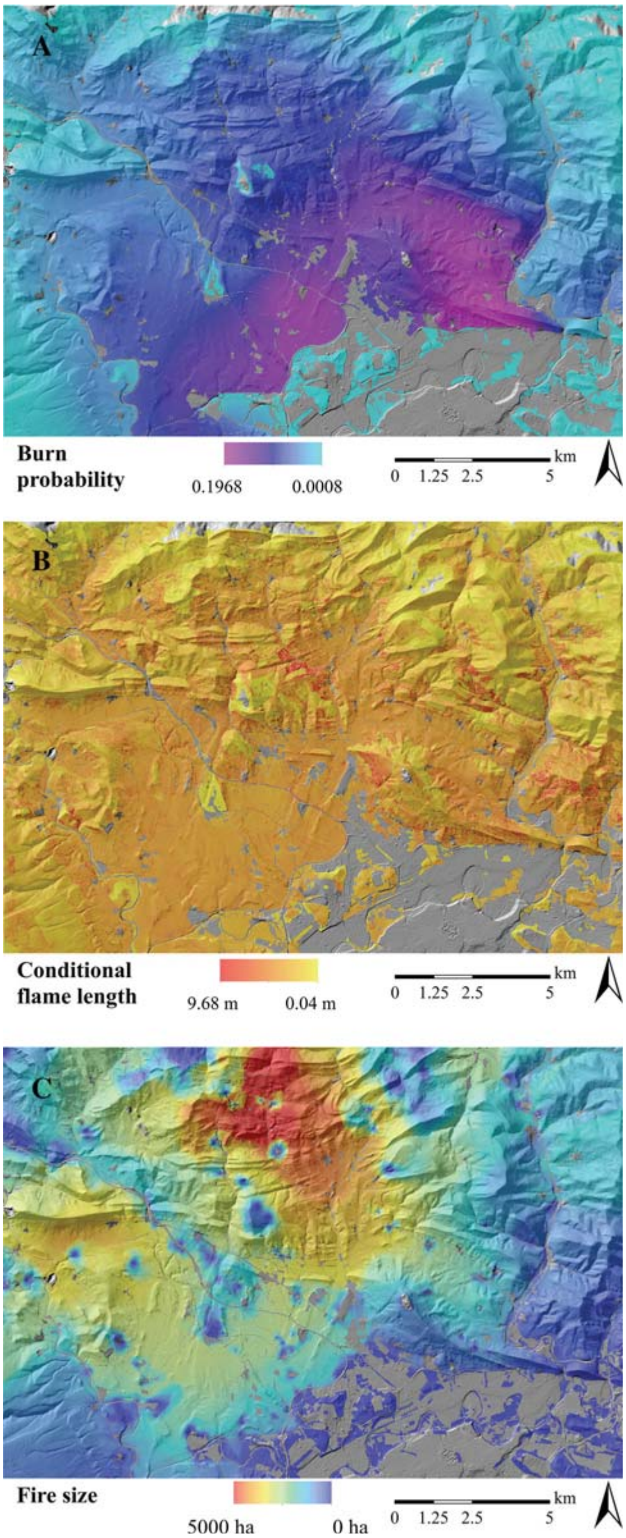

Fig. 6 Fine resolution (20 × 20 m) maps of burn probability (**a**), conditional flame length (**b**) and kernel-smoothed fire size (**c**) for the study area. Non-burnable areas (paved and urban development, see Fig. 2) occupy large zones of the southwestern part of the study area

southern part of the study area and some forests in the central part with moderate FS and high burn probability values (Fig. 6a, c). In the case of the FPI, we clearly identified five major areas with the highest values, where the probability of an ignition leading into a large fire is very high with respect to the other areas (Fig. 8b). These areas were mainly located in the highest observed ignition point areas that also presented moderate-to-high FS values (FS > 2000 ha; Fig. 6c). By contrast, mountainous areas on the eastern side of the study area showed the lowest FPI values, due to the lack of fire ignitions and low FS.

Crown fire activity

Only a small proportion of the forested areas showed no torching (CFA; Fig. 9a); these were beech forests, managed old-grown *Pinus nigra* stands and grazed wooded pastures (Fig. 2). Nonetheless, although forest stands generally showed evidence of at least passive crown fires or isolated torching, our analysis identified active CFA patches in the study area (Fig. 9a), where fire spread (i.e., faster rates of spread than surface fire, as well as spotting) and intensity can easily overwhelm firefighting suppression capabilities (Andrews et al. 2011). The forest vegetation types presenting the highest incidence of CFA (Table 4) were pole-stage *Pinus nigra* afforestations (13.6 %), followed by Mediterranean oak forests (7.7 %) and timber-stage *Pinus nigra* afforestations (7.5 %). The results by fuel model type were in agreement with the observed CFA in recent fires (e.g., Juslapeña and Izagaondoa 2009).

In order to locate the stands that currently present the highest probabilities of ember emission under extreme wildfire conditions, we compiled the ACP map (Fig. 9b).

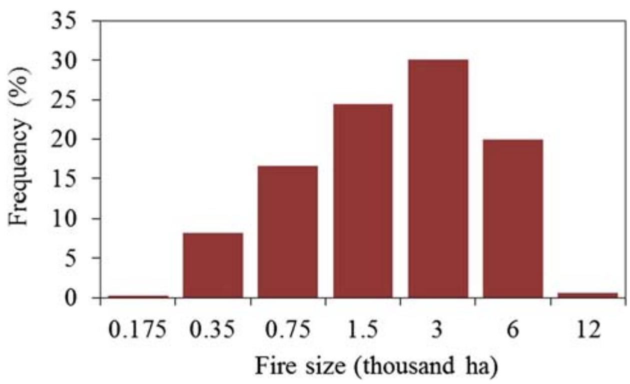

Fig. 7 Frequency distribution of fire sizes in the study area from the simulation of 30,000 fires combining the 12 scenarios and using historical ignition patterns. Maximum fire size was 7265 ha

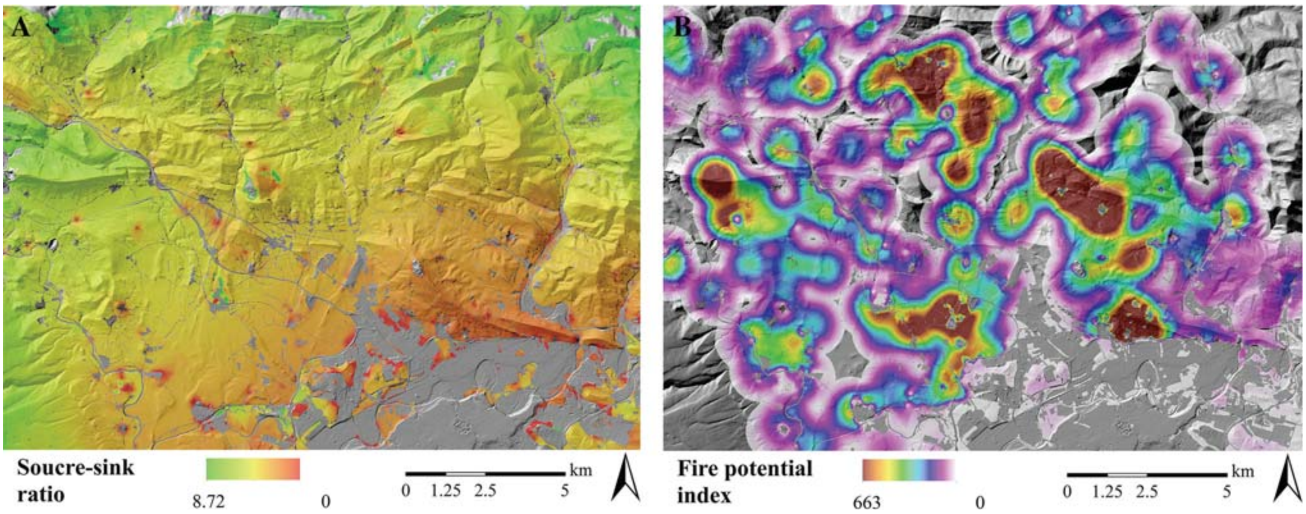

Fig. 8 Map of source-sink ratio (**b**) and fire potential index (**a**) for the study area. Source-sink ratio (SSR) is a logarithm of the ratio between fire size (Fig. 6c) and burn probability (Fig. 6a), while the fire potential index (FPI) was calculated from the historical ignition point density grid and the fire size map (Fig. 6c)

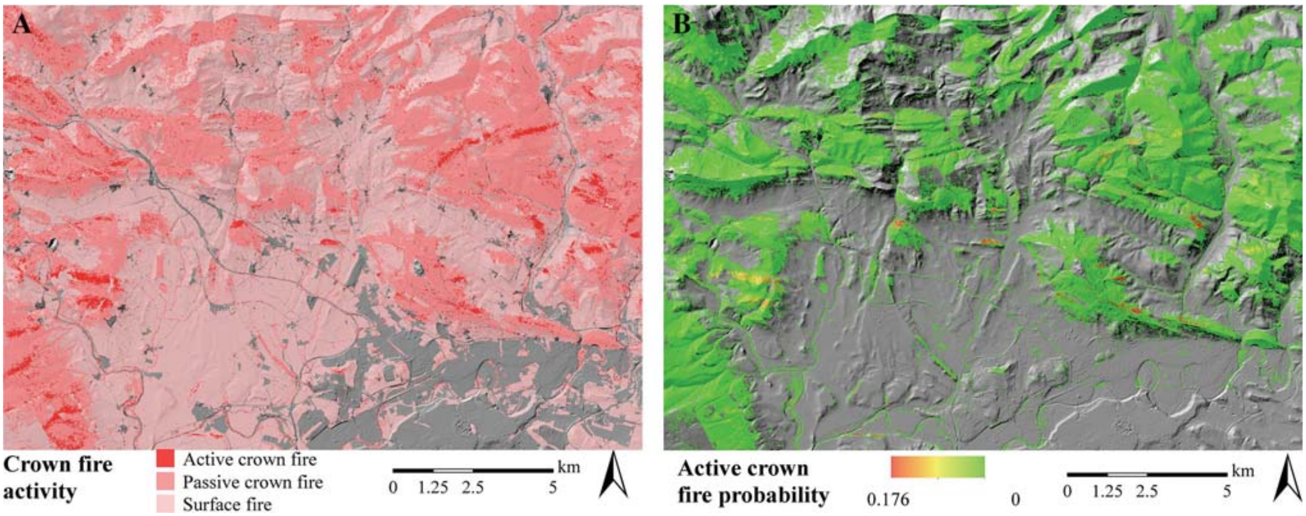

Fig. 9 Crown fire activity (**a**) and active crowning probability (**b**) maps. The crown fire activity map shows the type of fire (surface fire, passive crown fire or active crown fire) and was used in combination with the burn probability map (Fig. 6a) to identify the forest stands that, in case of a fire, would present crown fire activity and potential ember emission, as well as high post-fire mortality

The highest ACP values were recorded in small, unmanaged, closed and dense forest patches (<6 ha; Fig. 10) on north-facing mountain edges in the central part of the study area (Fig. 9b). Larger areas presenting CFA, mostly Mediterranean oak forests (>6 ha; Fig. 11), are located in the southwestern part of the study area (Fig. 9a), where BP values are four to five times lower than in the central part (Fig. 6a). In terms of ACP dif-ference among active crown fire values, the highest values for patches in the central part of the study area with ACP > 0.10 decrease smoothly to ACP < 0.04 in the peripheral (southwestern and northeastern) forest stands.

Fire exposure variation among HVRA classes and within patches

Box plots showing the dispersal of BP, CFL and FS output values among the HVRA classes illustrate a large range of variability (Fig. 11). Average BP values for the 27 HVRA classes ranged from a minimum of BP $= 0.2 \cdot 10^{-2}$ for the few authorized dumpsites (AD) in isolated areas to a maximum of BP $= 0.7 \times 10^{-1}$ for the livestock farms (FA) in valley bottom open areas surrounded by fast-fire-spreading light fuels. The BP results highlighted several structures/patches of some HVRA classes (power lines (PL), rural housing (RH), grazed pastures (GP) and

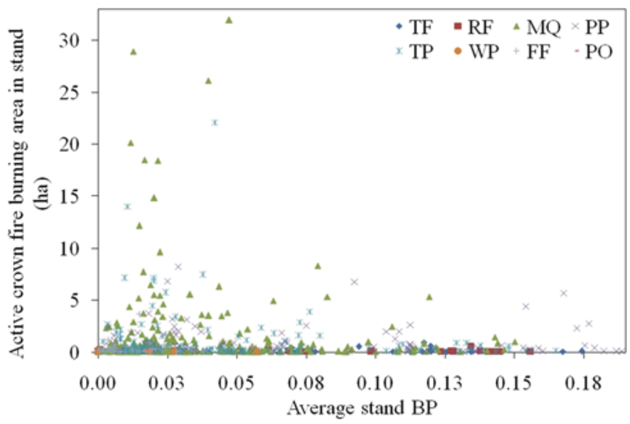

Fig. 10 Scatter plot of the stand patches (from forest-type fuel models) burning with active crown fire versus the average burn probability. TF: thicket-stage forests; RF: riparian forest; MQ: Mediterranean *Quercus* spp. forests; PP: pole-stage *Pinus* spp. plantations; TP: timber-stage *Pinus* spp. plantations; WP: wooded pastures; FF: *Fagus sylvatica* forests; PO: *Populus* spp. plantations

firewood forests (FF), in particular; Fig. 11) with very high fire likelihood values (dots) above the 90th percentile (upper whisker limits in the respective HVRA box plots), and even above the 97th percentile for all HRVAs ($BP > 0.155$, Fig. 11). With regard to the fire intensity outputs summarized through the CFL box plots (Fig. 11), commercial pine timber afforestations (PI) showed the most hazardous conditions, with 90th percentile FL values of over 6.5 m for stands predominantly burning under fast-fire-front-spreading conditions. Other HVRAs (i.e., FF, RH and PL) also showed high intensity values above the 97th percentile for all HRVAs ($CFL = 3.75$ m, Fig. 11) in certain structures/patches covered by high-fuel-load areas. Industrial buildings (IN), medio-European beech forests (MB) and mining sites (MS) presented the lowest average values, below the 25th percentile for all HVRAs ($CFL = 1.07$ m, Fig. 11b). Mediterranean thermophilous scree (TS), sclerophyllous forests (SQ) and *J. communis* scrub on calcareous grasslands (JS) showed the highest average FS values ($FS > 2400$ ha; Fig. 11), which were almost twice the average value for all HVRAs ($FS = 1290$ ha), although several other HVRAs such as livestock farms (FA), churches and hermitages (CH) and seminatural dry grasslands (SG) also broadly surpassed the average HVRA value (Fig. 11). Some other classes like rural housing (RH) and firewood forests (FF) exhibited extreme outlying values within classes (dots), above the 97th percentile for all HVRAs ($FS = 3776$ ha, Fig. 11).

Scatter plots of the average BP, CFL and FS values for individual HVRA patches revealed important variations among and within the different classes (Fig. 12). In general terms, most HVRA classes showed a decreasing point cloud concentration pattern from the lowest average BP and CFL values ($BP < 0.05$ and $CFL < 2$ m) to moderate–high BP ($0.10 < BP < 0.15$) and moderate average CFL (2 m $< CFL < 4$ m) values (e.g., FA, FF, GP, PL or RH; Fig. 12). However, it is difficult to describe any pattern for those HVRAs with a small number of patches or sites (<15, Table 2; e.g., AF, CS or CV, Fig. 12). With regard to FS, for HVRAs with many patches (>500; RH, PL, PI, FF and GP; Table 2), the highest values ($FS > 3200$ ha; Fig. 12) are clustered at moderate–high BP ($0.04 < BP < 0.12$; Fig. 12). The *Pinus* spp. commercial afforestation (PI; Fig. 12) showed the most scattered point cloud, and many patches presented high overall wildfire exposure, with values above the 97th percentile for BP or CFL ($BP > 0.155$; $CFL > 3.75$ m; Figs. 11 and 12) and also above the 97th percentile for FS ($FS > 3776$ ha; Figs. 11, 12).

Discussion

We used a fire spread simulation approach to analyze HVRA wildfire exposure in a forest–rural–urban intermix area located in northern Spain. Although the modeling outcomes are taken from a relatively small area (28,000 ha), there are many Mediterranean northern-rim regions, especially in Europe (e.g., all of the pre-Pyrenees and inland mountainous areas in Spain), with similar landscape configuration in terms of topography and vege-tation, weather conditions and anthropic activities. Other studies in Mediterranean northern-rim areas (e.g., in southern France and central-northern Italy) with different land cover and landscape management practices could help us to better understand local wildfire exposure variation according to HVRA classes. In this study, we also analyzed indexes related to large fire initiation and fire spread within the study area, which, in combination with the assessment of HVRA fire exposure, can help fire managers to address fire risk management and policy making in a more informed way. Until now, only very few studies in southern Europe have considered large fire spread for a realistic fire likelihood estimation (Salis et al. 2013; Kalabokidis et al. 2013), yet large fires are known to be responsible for most of the burned area in Mediterranean climate areas. Simi-larly, since fire intensity is strongly related to spread direction (e.g., heading, flanking and backing), large fire modeling is important in intensity estimation weighted by burning probability and, by extension, in fire hazard esti-mates (Miller and Ager 2013).

Fire occurrence analysis in Mediterranean areas is a prerequisite for fire modeling, since most fires are associ-ated with anthropic activities (Martı´nez et al. 2009; Padilla and Vega-Garcı´a 2011; Ager et al. 2014b; e.g., lightning caused just ~2 % fires in the study area) and exhibit

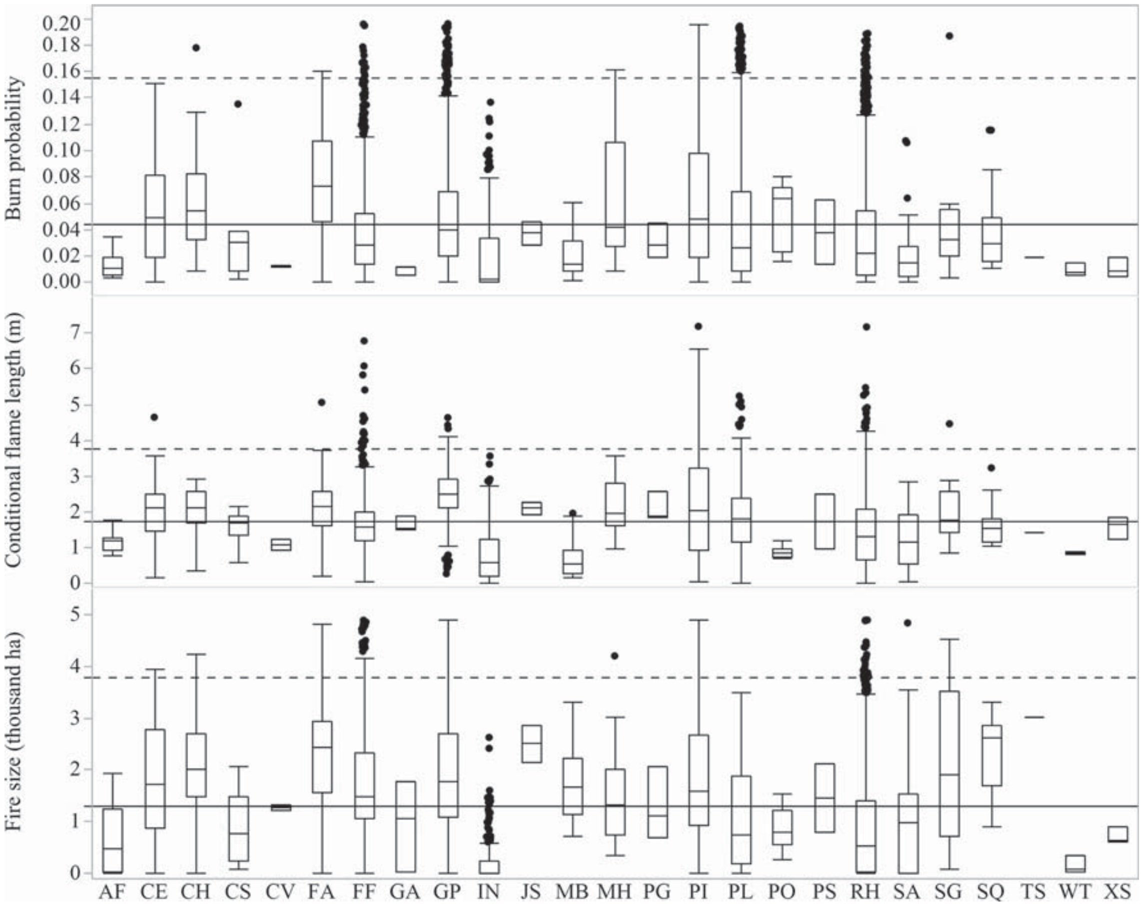

Fig. 11 Box plots of burn probability (BP), conditional flame length (CFL) and fire size (FS) for the highly valued resources and assets in the study area (see Table 2 for abbreviations). The *box* indicates the first/third quartiles, the *whiskers* indicate the 10th/90th percentiles, the *black line within the box* is the median, and the *dots* correspond to values below the 10th percentile or above the 90th percentile. The *horizontal continuous lines* indicate the average value (BP = 0.0437; CFL = 1.74 m; FS = 1.29 × 10^3 ha) and the discontinuous lines the 97th percentile value (BP = 0.155; CFL = 3.75 m; FS = 3.77 × 10^3 ha)

spatial–temporal ignition patterns that must be taken into account for accurate fire modeling (Bar-Massada et al. 2011; Salis et al. 2014, 2015). The anthropic causes of fire ignition are often unknown (45 % in the study area; Fig. 3c); how-ever, it is expected they keep the same proportionality found in known cause fires. Further research to integrate spatial–temporal wildfire occurrence and causality models with fire spread and behavior simulation approaches would lead to a better understanding of spatial burn patterns (Bar-Massada et al. 2011) and historical changes in fire likelihood in the Mediterranean basin. More research is also needed to assess the potential for preventing human-caused fires and reducing the probability of landscape burning (e.g., efforts focused on reducing fires from well-known causes such as grain-har-vesting machinery).

Fire modeling input data and the spatial identification of HVRAs are becoming considerably more accurate, reducing the uncertainty and possible sources of error in fire modeling and fire risk assessment. High-resolution data on local topography (5 m) and spatially explicit canopy characteristics derived from low-density airborne LiDAR (0.5 first returns m^{-2}), coupled with accurate land use/land cover information (e.g., 1:5000 SIGPAC map), have enabled landscape information input data to be characterized at fine scales (20 m). In addition, the EGIF fire database (MAGRAMA 2014) now contains complete records for more than 20 years covering most of Spain, providing extensive information on spatial ignition patterns and the associated fire causality. With regard to local wind speed and direction input data, surface wind fields can now be generated at fine resolutions (e.g., < 150 m) through models (WindNinja; Forthofer et al. 2014a, b) using weather station records, which can be used to increase the accuracy of fire modeling outcomes. Once a pixel-based causative factor maps have been compiled, geospatial HVRA data (e.g., from the land registry, IDENA and IGN) provide sufficient detail to assess wildfire exposure at the level of individual structures.

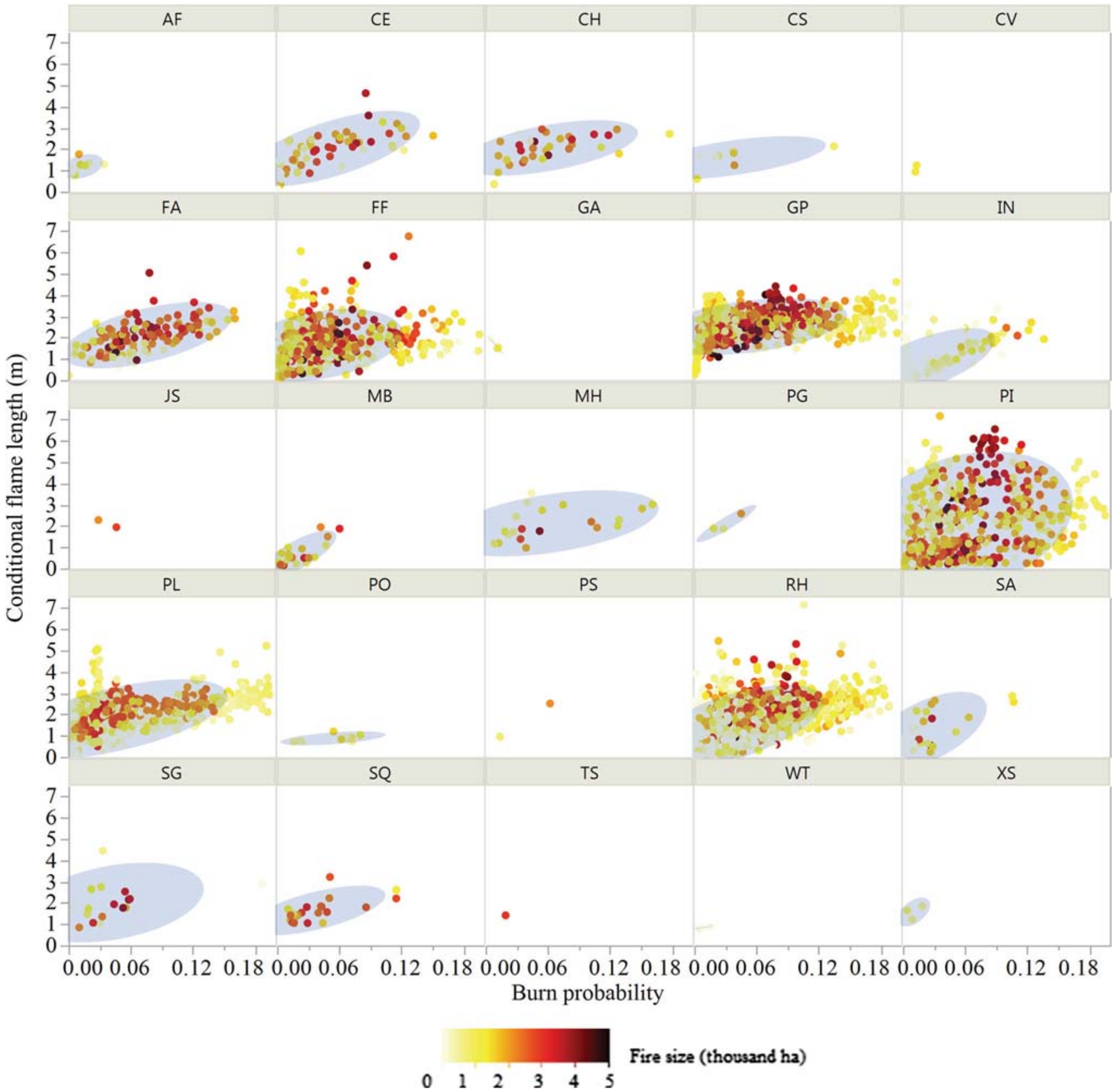

Fig. 12 HVRA average conditional flame length versus average burn probability scatter plots. Each point represents a patch/site for a HVRA (see Table 2 for abbreviations) and is colored according to the average fire size. The shaded area shows the bivariate normal density ellipse containing 90 % of the patches

Fire spread and behavior modeling outcomes permitted to identify areas and HVRAs (Fig. 12) where there is need to implement and prioritize fuel treatments and mitigate expected potential losses from large wildfires. The awareness of the role played by efficient fire management programs in central Navarra increased after the 2009 forest–rural intermix fires, which promoted the undertaking of strategically placed fuel treatments. Those treatments were spatially located based on expert criteria (i.e., ravine junctions and crest junctions; Costa et al. 2011) usually in conifer afforestation and consisted in the underburn after commercial thinning. In broadleaf natural forests, the treatments mostly consisted in the suppressed and dominated tree firewood cuts. Nonetheless, the expert criteria could be conditioned by the lack of experience (few observed large fires that in the future might be ignited elsewhere or spread under different weather conditions), and the limited budgets and personnel do not allow implementing the desired fuel treatments for the entire landscape. Within this context, our methodology accounts for the most likely environmental conditions that can lead to large wildfires in the study area, and takes into account historically based ignited pattern. Moreover, our approach allows to quantify and map fine-scale fire likelihood (Fig. 6a), intensity (Fig. 6b), large fire sources (Fig. 8a) and ember-emitting forest stands (Fig. 8b), and thus to transfer to land managers more awareness and knowledge about fire behavior and exposure nearby resources and assets.

The results suggest that BP outputs for our study area were strongly influenced by the frequent NW-N wind direction and the fast-burning fuel models that played a key role in surface fire spread, as shown in the areas with the highest BP values. This was primarily related to the spread of several large fires from the northern parts of the study area through cereal crops and herbaceous fuel types. Continuous non-burnable features (i.e., motorways, railways and rivers) in flat areas mainly covered by light fuels were sufficient to contain surface fire spread, as shown in sharp BP transitions. Nevertheless, we should not overlook the influence of spotting on large fire propagation and, by extension, on BP, as recent extreme fire events in the study area (e.g., Iza-gaondoa 2009 fire with 300 m spotting distance; Bomberos de Navarra pers. comm.) have shown that the heading fire intensity may be sufficient to overcome the non-burnable barriers to surface fire spread. The highest intensities (CFL $>$ 2.4 m; Fig. 6b) were found in steep-sloped areas when aligned with the dominant wind direction in high-fuel-load models (i.e., thicket-stage forests and shrublands). The shrubby pastures that often surround urban areas must be considered a potential source of damage to HVRAs, since their average burning intensities (CFL $>$ 2.5 m; Table 3) exceed the direct attack capabilities of firefighting crews (Andrews et al. 2011). In the study area, FS results revealed that three-quarters of fires would spread in excess of 750 ha (Fig. 7) under extreme weather conditions in the absence of suppression efforts; fortunately, no fire of this size has been observed to date. There are three possible explanations: (1) the rapid-response first attack, due to the proximity of automatic dispatch crews; (2) the efficient fire containment by ground crews and machinery, facilitated by the high road density and the ease of access to agricultural and forest lands (Fig. 5); and (3) the very limited number of fire ignitions under extreme weather conditions until now. Even so, FPI results revealed areas with large fire potential (Fig. 8a) where ignition prevention efforts need to be intensified, as they are the ignition sites of the most recent large fire events (Fig. 1). The largest fire source areas relative to burning frequency are mainly forested north-facing slopes in the northern parts of the study area (high SSR; Fig. 8b), where fire impacts are caused by fires ignited in the vicinity. Conversely, the most relevant sink areas (low SSR; Fig. 8b) are located on the boundaries of urban areas in the central part of the study area. The SSR revealed fire-prone forested areas in the mountains of the central part of the study area (SSR $<$ 2; e.g., the whole of the San Cristobal mountain), which is consistent with the observed fire events in these areas that spread from fires ignited in urban areas and roads.

Major wildfire exposure differences were observed between HVRAs, as shown in the scatter plots (Fig. 12). This information could be very useful for landscape managers in prioritizing fuel treatments for hazardous vegetation surrounding high relative importance structures, like residential housing (RH) and industrial buildings (IN)(Ager et al. 2012; Alcasena et al. 2015). Moreover, although further detailed studies would be needed, reduc-ing hazardous vegetation in housing vicinities—at least in the 60 m buffer HIZ (Cohen 2008)—would in theory create safe confinement areas in the event a wildfire, due to the low flammability of the materials used in the local constructions, the low probability of ember ignition (urban areas are surrounded by agricultural lands and far from active crown fire areas) and the improved fire suppression capabilities of ground crews. The current fire confinement capacity achieved through investment in linear non-burnable infrastructure (e.g., highways) for flat areas with herbaceous fuel models in the southern part of the study area suggests that further investment should be made part of the strategic fuel management containment strategy to facilitate fire suppression in these locations (Ager et al. 2013). Although major highways in the study area could became a good opportunities for fire suppression, they usually present strips with low vegetation and dense bushy barriers in the maintenance zone. In these cases, it would be advisable to widen the low-vegetation areas (i.e., maintaining short herbaceous grass vegetation), and to thin and prune as much as possible the bushy barriers, as well as to prefer low-flammability species and to remove the accumulated dead materials. The benefits of the management of fuels in the vicinity of major highways can be tested and quantified by applying fire spread modeling, to determine the best strategy as well as the potential to suppress wildfires.

We spatially identified in the ACP map (Fig. 9b), and even at stand level (Fig. 10), the areas in which mitigation should be prioritized to disable the spotting fires that easily overwhelmed extinction capabilities in past fires. Those areas are mainly located in hilly terrain and rough mountain windward edge crests, where dominant winds and slope are aligned with the heading fire major runs (Costa et al. 2011): Here the transition from surface to active crown fires is fast under extreme weather conditions. The initiation of the crown-to-crown transmission can be avoided elevating the canopy base height or disrupting crown continuity within stands. The typical forest stands usually correspond to overstocked pole-stage *Pinus nigra* afforestation (Table 5), characterized by a very low canopy

Table 5 Summary of the forest vegetation types (Fig. 2) in the study area and expected area burned by active crown fires (ACF; Fig. 9a)

Forest vegetation types (abbreviation)	Number of stands	Average stand area (ha)	Total area (ha)	Percentage of active crown fires (%)
Thicket-stage forest (TF)	436	1.2	527.9	1.0
Riparian forest (RF)	246	0.6	145.2	3.0
Mediterranean *Quercus* ssp. forest (MQ)	1786	2.9	5163.1	7.7
Pole-stage *Pinus nigra* plantations (PP)	230	3.3	758.0	13.6
Timber-stage *Pinus nigra* plantations (TP)	232	8.1	1878.6	7.5
Wooded pastures (WP)	269	1.0	260.2	0.1
Fagus sylvatica forests (FF)	67	22.7	1523.1	1.1
Populus ssp. plantations (PO)	14	1.7	24.2	3.0

base height (dead branches at ground level and shrubby laddered fuels in the understory), high canopy bulk density (afforestation with 2500 trees ha^{-1}) and high canopy cover. Overall, mitigation measures would therefore combine heavy weight thinning, pruning up to 2–2.5 m (at a maximum height of one-third of tree height), slash and laddered fuel underburn, and beef livestock extensive grazing to control the growth of heliophilous shrubs (e.g., *Rubus* sp. and *Rosa canina*) in the understory. The effectiveness of the abovementioned fire risk mitigation strategies can be evaluated and quantified using fire spread modeling, in order to identify the best compromise among risk reduction, costs and environmental constraints. Work is in progress to assess, applying a burn probability approach, the trade-offs among competing fuel management strategies for fire risk mitigation purposes within the study area as well as in other Mediterranean ecosystems.

Conclusions

We presented a consistent methodological framework for exposure analysis that could be adopted as the preliminary step in fire risk mapping and mitigation for land managers and policy makers. In this case study, we followed a stochastic fire modeling approach based on a robust quantitative geospatial assessment framework, broadly used and accepted in the USA but not yet in Europe. The outputs have the potential to address the real requirements of landscape managers working with restricted budgets, who need reliable fine-scale analysis to prioritize mitiga-tion measures, prevent and monitor fires caused by anthropic activities and define policies. Further research into the effects of fire on HVRAs coupled with the use of likelihood and intensity maps would allow a better under-standing of the expected losses or benefits associated with wildfire events.

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Quantifying economic losses from wildfires in black pine afforestations of northern Spain

Fermín J. Alcasena [a,*], Michele Salis [b], Nicholas J. Nauslar [c], A. Eduardo Aguinaga [d,1], Cristina Vega-García [a,e]

[a] Agriculture and Forest Engineering Department (EAGROF), University of Lleida, Alcalde Rovira Roure 191, 25198 Lleida, Spain
[b] IAFES Division, Euro-Mediterranean Center on Climate Change (CMCC), Via De Nicola 9, Sassari 07100, Italy
[c] Division of Atmospheric Sciences, Desert Research Institute, 2215 Raggio Parkway Reno, 89512 Nevada, USA
[d] Guarderío Forestal, Departamento de Desarrollo Rural y Administración Local, Gobierno de Navarra, Calle González Tablas 9, 31005 Pamplona, Navarra, Spain
[e] Forest Science Center of Catalonia (CTFC), Pujada del Seminari s/n, 25280 Solsona, Catalonia, Spain

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A B S T R A C T

We implemented a fire risk assessment framework that combines spatially-explicit burn probabilities, post-fire mortality models and public auction timber prices, to estimate expected economic losses from wildfires in 155 black pine stands covering about 450 ha in the Juslapeña Valley of central Navarra, northern Spain. A logit fire occurrence model was generated from observed historic fires to provide required fire ignition input data. Wildfire likelihood and intensity were estimated by modeling 50,000 fires with the minimum travel time algorithm (MTT) at 30 m resolution under 97th percentile fire weather conditions. Post-fire tree mortality due to burning fire intensity at different successional stages ranged from 0.67% in the latest stages to 9.22% in the earliest. Stands showed a wide range of potential economic losses, and intermediate successional stage stands presented the highest values, with about 124 € ha^{-1} on average. A fire risk map of the target areas was provided for forest management and risk mitigation purposes at the individual stand level. The approach proposed in this work has a wide potential for decision support, policy making and risk mitigation in southern European commercial conifer forests where large wildfires are the main natural hazard.

1. Introduction

Large wildfires in the last decades have threatened black pine forests (*Pinus nigra* Arn.) in southern Europe, and currently represent a major concern among valued resources on forest lands (Espelta et al., 2003; Ordóñez et al., 2006; Christopoulou et al., 2014). In the landscapes of central Navarra (northern Spain), black pine afforestations are natural-ized and cover around 28,200 ha in the whole region (Fig. 1; www. idena.navarra.es; Gobierno de Navarra, 2012). The first relevant affores-tation efforts in Navarra were initiated by the regional Forest Service in the early 1920s, with the bulk of planted areas in the 1930s and 1960s (peaks over 1000 ha year^{-1} in 1933 and 1962; Diputación Foral de Navarra, 1982). Although this subspecies is native to the Austrian Alps, its mid XIXth century afforestations across southern France adapted well to heavy and compact soils, and this prompted later the use of this species in central Navarra with homologous edafoclimatic conditions. Those forests were planted in mid slope small plots of mar-ginal agricultural lands, mountain pastures, degraded shrubland forma-tions, and defective-coverage broadleaf forests with little evolved clayey soils, and the good adaptation of this species achieved not only erosion and soil protection goals, but also a good timber production potential (Roselló et al., 1990).

* Corresponding author.
E-mail address: ferminalcasena@eagrof.udl.cat (F.J. Alcasena).
[1] Retired.

Currently black pine afforestations provide important timber-based income for rural communities but landscape managers lack manage-ment-meaningful studies assessing potential economic losses from large wildfires. Moreover, these forests also provide other services, espe-cially in open woodland structures, such as grazing for extensive breed-ing (Valderrábano and Torrano, 2000; Torrano and Valderrábano, 2005; Casasús et al., 2007) with 550 kg pasture dry matter ha^{-1} year^{-1} (Mangado et al., 2014) and edible marketed ectomycorrizal fungi pro-ductions (e.g., *Lactarius deliciosus*) with 0.23 (0.01–0.66) kg fresh weight ha^{-1} year^{-1} (Martínez de Aragón et al., 2007). Black pine forests also represent important habitats of hunting species in the early successional stages (e.g., *Sus scrofa* and *Scolopax rusticola*), and for species of natural interest (e.g., woodpeckers *Dendrocopos major*) in the few naturalized oldest forests that resemble endemic sub-Mediterranean black pine priority habitats (Directive 92/43/ECC). The multifunctionality and economic income provided by the black pine forests makes them one of the most important natural valued resources, not only in the rural-urban and forested central belt of Navarra, but also in many mountainous forest ecosystems of southern Europe.

Among the most frequent natural disturbances in black pine (*Pinus nigra* Arn.) afforestations in southern Europe, wildfires cause most dam-ages, while other natural hazards such as heavy snows and wind blows

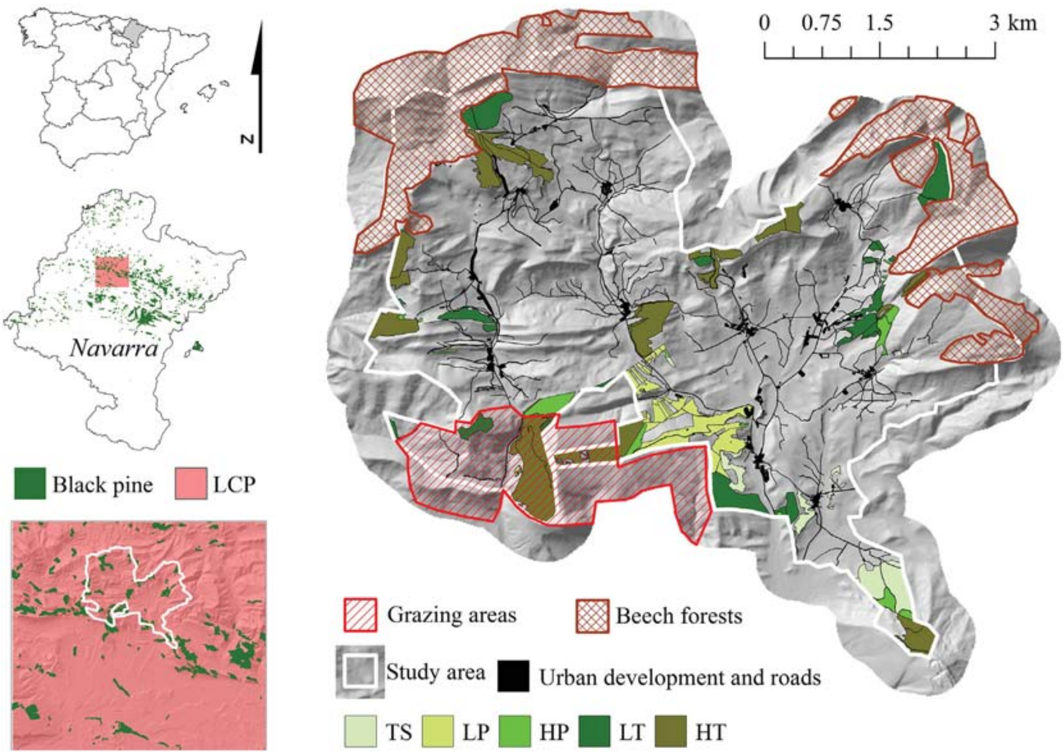

Fig. 1. Map of the Juslapeña Valley (Navarra, northern Spain) study area showing the black pine target stands for fire risk assessment (see Table 2 for abbreviations). We framed a 36,000 ha landscape file (LCP) containing the study area for wildfire modeling. Black pine afforestation polygons correspond to the minimum forest management units and are mostly owned by the municipalities (96%).

have a limited impact on managed stands. Pest incidence (e.g., *Thaumetopoea pityocampa* pine processionary caterpillar) is usually low and hardly ever kills the whole stand. Black pine has a thick bark and good self-pruning characteristics that confer resistance and good adaptation to frequent low-moderate fire intensity wildfire regimes (Fulé et al., 2008; Touchan et al., 2012; Christopoulou et al., 2013). How-ever, in preferential distribution areas of this species, changes in natural fire regime to low-frequency and intense crown fire events trigger stand-replacement dynamics in favor of re-sprouting broadleaves, since post-fire regeneration is poor or non-existent due to the lack of se-rotinous cones in the black pine (Martín-Alarcón and Coll, 2016). There-fore, post-fire restoration usually entails substantial economic investments in burned productive stands (>2500 € ha $^{-1}$) t o finance the required mechanized afforestation, new forest road opening or maintenance, and fencing to protect tree seedlings from livestock. Le-gally, in public lands the regional Forest Service must guarantee post-fire restoration in forested areas when natural regeneration is unfeasi-ble and causality is unknown or natural (art. 42 LF 13/1990). Inasmuch as most afforestations in Navarra are in public lands, both the regional Forest Service and rural municipalities are particularly interested in protecting those forests from large catastrophic fires. Nonetheless, cur-rently wildfire managers do not have landscape planning resources that could anticipate potential economic losses, and decision-making in fire risk mitigation strategies (i.e., fuel treatment prescription, prioritization treatments among stands and mitigation effects assessment) is usually based on expert criteria.

At stand level, given a fire occurs, post-fire tree mortality can be modeled as a probability from morphological descriptors (e.g., bark thickness and tree size) and post-fire damage measurements (e.g., tis-sue and crown damage) using logistic regression analysis (Peterson and Ryan, 1986; Ryan and Reinhardt, 1988; Catry et al., 2010; Fernandes et al., 2012; Temiño-Villota et al., 2015). Other approaches have predicted stand damage and tree survival with mortality models based on tree and stand variables (e.g., basal area and diameter at breast height) gathered from forest inventory data in burned areas, and topo-graphic variables (e.g., slope of the terrain) correlated with fire hazard (González-Olabarria et al., 2007). Interdependences between fire inten-sity and damage measurement (i.e., crown scorch height or stem-bark char height with fire intensity; Van Wagner, 1973; Weber et al., 1987; Alexander and Cruz, 2012) allow the estimation of expected tree mor-tality for a range of intensities at different growth stages (Fernandes et al., 2008). The fact is that different fire spreading direction (i.e., heading, backing or flanking) at which the fire front encounters a burning fuel, fire weather and terrain slope substantially affect fire intensity and res-idence time, and the combinations of those factors widely vary in the territory. Therefore, the use of simulators to model spatially-explicit fire behavior under the most feasible fire weather conditions is neces-sary to determine potential fire effects on the vegetation.

Most difficulties arise at the time of implementing fire modeling ap-proaches that quantitatively assess wildfire risk at landscape scale to provide management-meaningful outcomes to forest managers. Fire risk is the expectation of loss (or benefit) from unplanned fire over socio-economic and natural resources, and combines burning probabil-ity with fire effects at associated fire intensities (Finney, 2005; Calkin et al., 2011b; Miller and Ager, 2013). Spatially explicit burning probability can be accurately modeled at a broad range of scales, from project to na-tional scale, using fire spread and behavior simulators (Scott et al., 2012; Salis et al., 2013; Thompson et al., 2013; Alcasena et al., 2015a; Ager et al., 2016). Wildfire effects can be estimated for any highly valued re-source within the landscape (e.g., commercial forests, endangered spe-cies habitat, rural-urban interface and infrastructures) in response to different fire intensities (previously obtained from fire modeling) using expert-defined response functions (RFs; Thompson et al., 2011; Calkin et al., 2011a), modeling post-fire stand level conditions in endan-gered habitats (Ager et al., 2007) and using tree species- and size-specif-ic loss functions (Ager et al., 2010; Fernandes et al., 2008).

When fire risk is analyzed for multiple valued resources with too complex fire effect economic assessment (e.g., landscape value and social use of the forest), relative importance weighting assignation is possible and allows for comparison and overlaying for an integrated assessment (Scott et al., 2013). This quantitative fire risk assessment framework has been successfully implemented in western United States (US) fire-prone areas to prioritize protection among highly valued resources (Scott and Helmbrecht, 2010; Thompson et al., 2012; Helmbrecht et al., 2013). However, few previous efforts in southern Europe have integrated fire probability with expected consequences in wildfire risk assessment (González-Olabarria and Pukkala, 2011; Chuvieco et al., 2014; Rodríguez y Silva and González-Cabán, 2010), and only few exceptions accounted for spatially explicit burning probabilities and fire intensities (Mitsopoulos et al., 2015) even though large fires are responsible for most of the burned area in Mediterranean landscapes (San-Miguel-Ayanz and Camia, 2010). In wildfire exposure assessment spatially-explicit fire modeling of likelihood and intensity outputs are analyzed in relation to valued resource locations in maps, but there is no susceptibility analysis (Miller and Ager, 2013). Therefore, differences in resource potential losses with the same wildfire exposure-level but different response to fire (and different relative importance or economic value) are ignored. When fire effect quantification has been found uncertain or exceedingly difficult to estimate, both due to the lack of sound expert criteria and pixel level accurate information (e.g., individual structure-specific loss function from ignitability information and pixel-level geospatial information), wildfire exposure analysis could enable fire managers to prioritize mitigation efforts in fire-prone areas (Ager et al., 2010; Salis et al., 2013; Kalabokidis et al., 2013; Alcasena et al., 2015b).

In the Mediterranean areas fire ignitions present complex spatial patterns usually associated to anthropogenic variables (i.e., most fire ignitions are caused by humans) like urban development, wildland urban interface and main roads, and other spatial features such as land use land covers and topographic features (Padilla and Vega-García, 2011). The most broadly used statistical approaches that better capture those patterns correspond to logistic regression (Vega-García et al., 1995; Syphard et al., 2008), artificial neural network analyses (Ruiz-Mirazo et al., 2012) and geostatistic procedures (Gonzalez-Olabarria et al., 2012a; Koutsias et al., 2014). Some fire spread modeling studies in that sense highlighted the key role that fire ignitions play on fire likelihood estimates, and used fire occurrence preferential areas from observed historical fires to display fire ignition input data location (Carmel et al., 2009; Bar-Massada et al., 2011; Salis et al., 2015). Notwithstanding fire occurrence might portray well fire likelihood where fires are small (<100 ha), such as per-humid climate mountainous areas in the Alps or Pyrenees, fire spread modeling is necessary in Mediterranean areas to estimate burning probability over target valued resources from fires ignited elsewhere and arriving from large distances (Miller and Ager, 2013).

In this fire modeling approach, we combined fine resolution and spatially-explicit fire likelihood and intensity modeling outputs with tree mortality expectations at different burning intensities. We accounted for a fire occurrence model to generate ignition input data, and wildfire season meteorological records to characterize the typical extreme fire events in the study area. Fires accounting for most of the burned area occur under extreme weather conditions and the 97th percentile has been widely used as a statistic reference value for these extreme fire weather conditions (Ager et al., 2010; Salis et al., 2013; Scott et al., 2013). The potential economic loss was estimated at pixel-level from dead trees, considering the timber price from public auctions at the respective stand successional stages, and the timber volume at stand level gathered from forest inventories. Our specific objectives in this study are to spatialize and assess (*i*) wildfire exposure (wildfire likelihood and intensity), (*ii*) wildfire risk (conditional tree loss, *cL*, and expected tree loss, *eL*) and (*iii*) potential timber economic losses (expected economic loss, e*EL*) in black pine (*Pinus nigra* Arn.) stands of Juslapeña valley (central Navarra, Spain). We discuss the potential implications in using this geospatial information for forest management. The implementation of this quantitative assessment methodological framework could assist southern European forest managers in mitigating wildfire economic losses inconifer timberlands.

2. Material and methods

2.1. Study area

The Juslapeña Valley study area is located 12 km north of the capital city of Pamplona, in central Navarra (northern Spain) (Fig. 1). It covers around 3430 ha and is limited by the municipalities of Berrioplano and Iza to the south, Ezkabarte to the west, and Atez to the north. The elevation ranges between 460 m in the open valley bottoms and 944 m in the highest peak. The climate is humid-Mediterranean with 1100 mm average annual rainfall, 2–3 months of rainfall shortage between July and August, 12 °C mean annual temperature and 32 °C mean of maximum temperatures in the warmest month (www.meteo.navarra.es). The 548 inhabitants are distributed in 12 small villages located at mid slopes within a rural-urban mosaic of dryland cereal crops and herbaceous pastures. Mountainous areas and hilly terrains are covered by Mediterranean broadleaf natural forests in south facing slopes (*Quercus pubescens* Willd. and *Quercus ilex* L.), beech forests in the highest elevation north-facing slopes (*Fagus sylvatica* L.) and sparse black pine (*Pinus nigra* subsp. *nigra*) afforestation (Table 1).

In the study area, 155 black pine stands cover about 450 ha (Table 2). The bulk of black pine afforestation was conducted between 1920 and 1930, and the most area burned occurred in the last decade, when the burned area surpassed the afforested land area (Forest Service pers. comm.; Fig. 2). Stands were planted at high densities (around 2500 trees ha^{-1}) in small (<10 ha) and sparse plots, and showed average growth rates over 4 m^3 ha^{-1} $year^{-1}$ in good site index areas (Eraso et al., 1996). These forests have been intensively managed under the supervision of the regional Forest Service through a uniform shelterwood system (a sequence of 3–4 early and heavy weight thinnings from below in 15–20 year lags, up to final cut at rotation age of 80 years) with artificial regeneration (natural regeneration is usually scarce and does not guarantee the persistence of this species) for commercial timber production purposes (e.g., paper pulp, poles, wooden formwork, wooden pallets, packaging in general and furniture) that aims to maximize the timber volume obtained from thick logs in the last cut (DBH >35 cm).

Historically, and up to the late 1960s, fire was used systematically for pasture clearing and cereal waste elimination with fires burning very small areas (<10 ha). Nowadays, the use of the fire is restricted to certain periods during the year and forbidden without express authorization from the regional Forest Service (art. 6 and 7 OF 195/2014). Livestock extensive grazing reduction and limited forest management in firewood woodlands during the last half century has led to significant increases in fuel load and continuity in forested areas, thus facilitating intense and erratic wildfire events where fires historically were not considered a threat. In the study area three large fires (>100 ha) burned about 95% of the total area, and the largest fire events occurred in the last decade, in 2005 and 2009, burning respectively 174 and 344 ha (1985 to 2012 database; MAGRAMA, 2014).

2.2. Fuels and topography

We framed a wildfire modeling input landscape file (LCP) of 36,000 ha, containing the Juslapeña valley study area (3430 ha). All the required landscape input data for wildfire modeling were assembled together into a 30 m resolution grid with ArcFuels (Ager et al., 2011). Topographic data (i.e., aspect (azimuth), elevation (m) and slope (degrees)) were obtained from a 5 m resolution digital elevation model (www.ign.es), and a surface fuel model grid was build assigning standard fuel models (Scott and Burgan, 2005; Fernandes, 2009) to a 1:5000 map of land use/land cover typologies (www.sigpacnavarra.es;

Table 1
Main vegetation types, coverage and fuel model assignments for wildfire simulation in the study area (Mapa de cultivos y aprovechamientos 2012, http://idena.navarra.es; Gobierno de Navarra, 2012). Most thicket-stage forests and wooded pastures correspond to *Pinus nigra* afforestations.

Vegetation type	Surface (ha)	Incidence (%)	Fuel model
Urban areas and development	94.9	2.8	NB1 (Scott and Burgan, 2005)
Rivers and rafts	10.2	0.3	NB8 (Scott and Burgan, 2005)
Orchards, tilled lands	16.2	0.5	NB3 (Scott and Burgan, 2005)
Cereal crops	901.2	26.3	GR5 (Scott and Burgan, 2005)
Mowing hay meadows and grazed pastures	148.2	4.3	GR2 (Scott and Burgan, 2005)
Herbaceous pastures	376.7	11.0	GR4 (Scott and Burgan, 2005)
Shrubby herbaceous pastures	503.1	14.7	SH6 (Scott and Burgan, 2005)
Thicket-stage forests and shrublands	87.8	2.6	SH5 (Scott and Burgan, 2005)
Riparian vegetation	6.0	0.2	SH8 (Scott and Burgan, 2005)
Quercus spp. forests	649.6	18.9	TU3 (Scott and Burgan, 2005)
Pole-stage *Pinus* spp. forests	42.1	1.2	PCL (Fernandes, 2009)
Timber-stage *Pinus nigra* forests	246.9	7.2	SH3 (Scott and Burgan, 2005)
Wooded pastures	61.7	1.8	GR3 (Scott and Burgan, 2005)
Fagus sylvatica forests	286.5	8.3	TL2 (Scott and Burgan, 2005)

idena.navarra.es; Alcasena et al., 2015b). Land use/land cover maps have been widely used in EU and elsewhere when there is not a standardized fuel dataset available like in the US (Arca et al., 2007; Salis et al., 2013, 2014; Jahdi et al., 2016). Canopy metrics (i.e., canopy base height (m), canopy height (m), canopy bulk density (kg m^{-3}) and canopy cover (percent)) were generated from low-density (0.56 returns m^{-2}) airbone LiDAR (ign.es) using Fusion software (Mc Gaughey, 2014) and available models for the target tree species in the study area (*Pinus nigra* models from Gonzalez-Olabarria et al., 2012b).

2.3. Fire occurrence

A multiple-regression logistic model generated from observed fire ignitions was used to replicate the spatial location in fire ignition input data because fires in the study area are mostly caused by humans (lightning fire ignition cause <2%) and do not present a random spatial distribution. Logistic regression is useful to predict the presence or absence of a fire ignition probabilistically from a mixture of predictive variables that can be either continuous or categorical (Hosmer and Lemeshow, 1989). The logit occurrence model uses the following function:

$$IP = 1/(1 + e^{-z}) \qquad (1)$$

where *IP* is the probability of occurrence of the fire ignition, and *z* is obtained from a linear combination of the independent predictive variables estimated from a maximum likelihood fitting:

$$z = b_0 + b_1x_1 + b_2x_2 + \ldots + b_nx_n \qquad (2)$$

where b_0 is the constant and b_n is the weighing factor of the variable x_n. The *z* values can be interpreted as a function of the probability of occurrence, and *IP* converts *z* values in a continuous probability function that ranges from 0 to 1.

Table 2
Black pine stands in the study area. Stand physical polygons were gathered from 1:5000 regional cadaster map (https://catastro.navarra.es; Gobierno de Navarra, 2014), which were identified using the land use land cover map (http://idena.navarra.es; Gobierno de Navarra, 2012).

Successional stage (abbreviation)	Total area (ha)	Average stand area (ha)	Number of stands
Thicket-stage (TS)	44.82	3.0	15
Low pole-stage (LP)	65.57	1.9	34
High pole-stage (HP)	30.44	0.6	48
Low timber-stage (LT)	108.67	4.2	26
High timber-stage (HT)	198.72	6.2	32

We first reviewed the fire database and located fires that lacked fire ignition point coordinates but gave explicit description of ignition location, gathering together in the complete fire dataset 200 ignitions in the period from 1985 to 2013 (MAGRAMA, 2014), and then the same number of points from non-ignition pixels were randomly sampled within the fire modeling 36,000 ha landscape and fire occurrence area. Multiple anthropogenic and biophysical variables were assigned to the 400 points (Table 4), considering significant independent variables from other studies in the Mediterranean area and geospatial data availability at fine resolution (Catry et al., 2009; Martínez et al., 2009; Vilar et al., 2010; Ager et al., 2014). Variables were entered into a multiple logistic regression model, and the final model was selected through a backwards stepwise elimination process using the Akaike Information Criterion (AIC; Venables and Ripley, 1999). The Area Under Curve (AUC) of the chosen model was 0.772 and its global accuracy 71%. Independent variables in the multiple-regression model indicated that ignitions were most likely to occur in highly populated municipalities, close to urban areas, close to rural tracks, close to power lines, close to railways and in some land covers (Table 4). We then generated a 30 m cell-size resolution fire ignition probability (IP) map for the whole wildfire modeling 36,000 ha landscape using the logistic regression model (Fig. 3), to then generate a set of spatially well-balanced 10,000 points over the study area (excluding non-burnable areas such as water bodies and urban development) according to the ignition probability grid (Stevens and Olsen, 2004). The 10,000 point file was later used as fire ignition coordinate dataset for the wildfire simulations.

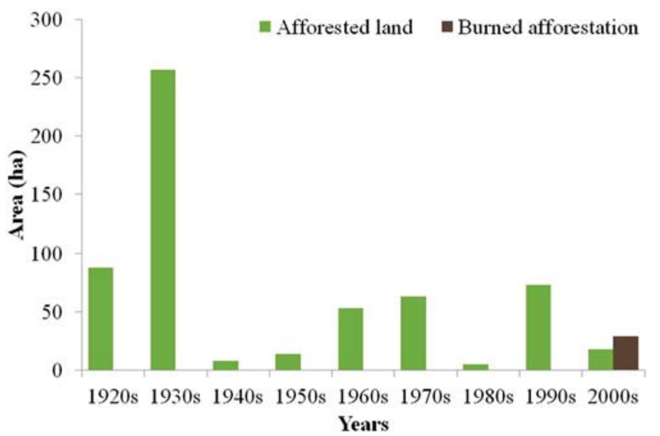

Fig. 2. Black pine afforested and burned lands in the study area (Forest Service pers. comm.). The bulk of afforestation efforts were carried at the first third of the XXth century in public lands before the civil war. No afforestation has been done after 2010.

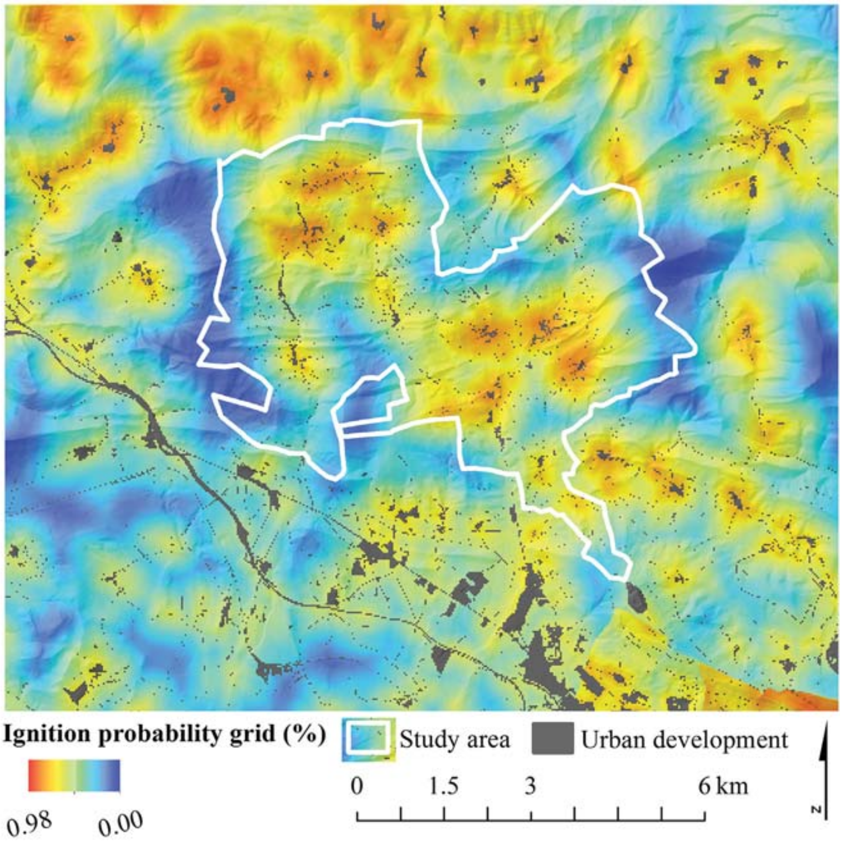

Fig. 3. Ignition probability grid (30 × 30 m) generated from the logit model (Table 4) and used to draw the spatially weighted ignition pattern input data for fire modeling. Most fire ignitions in the study area are anthropic (>90%; MAGRAMA 2013) and located close to urban development.

2.4. Fire weather

We used Fire Family Plus (Bradshaw and McCormick, 2000) to estimate the 97th percentile fuel moisture and wind speed for the most frequent wind directions (Table 5). The meteorological database contained 10 min data of precipitation, air temperature, relative humidity, solar radiation, and 10 m wind speed and wind direction data from the automatic weather station of Pamplona, the closest representative weather station with a long enough database to generate a consistent weather input data (1997 to 2014 data series; www.meteonavarra.es). We estimated the 97th percentile wind speed for the most frequent wind directions during wildfire season to generate 150 m resolution terrain-adapted wind grids throughout the study area using the mass-consistent WindNinja model (Forthofer et al., 2014a, 2014b). WindNinja computes spatially varying wind fields from elevation, a domain-mean initial wind speed and direction, and specification of the dominant vegetation data in the area (Forthofer and Butler, 2007). The fuel moisture content was estimated for the dead and live fractions of the fuel models from conditions exceeding the 97th percentile of the energy release component using as reference the fuel model G (ERC-G; Deeming et al., 1972; Nelson, 2000). Air temperature, relative humidity, solar radiation, and rainfall are weather inputs into the physical model that estimates fuel moisture content via equations describing heat and moisture transfer (Nelson, 2000).

2.5. Modeling wildfire likelihood and intensity

Wildfires were simulated using the minimum travel time (MTT) two-dimensional fire growth model (Finney, 2002) as implemented in FlamMap (Finney, 2006) simulator. Many previous studies have used this algorithm to model wildfires in heterogeneous landscapes of the southern EU and elsewhere for different purposes (Ager et al., 2007; Bar Massada et al., 2009; Kalabokidis et al., 2013; Salis et al., 2013; Mitsopoulos et al., 2015; Alcasena et al., 2015a, 2015b). From given input data, MTT finds the shortest path via a straight line by calculating travel times from each cell corner to every other cell corner on the landscape, and then calculates fire behavior on flowpath segments. Spotting can also be simulated in fire growth when passive or active crown fires occurred (Finney, 1998; Scott and Reinhardt, 2001). MTT allows the modeling of thousands of fires at a broad range of scales, assuming constant weather and fuel moisture conditions, and is appropriate for short fire event modeling (Ager et al., 2007), such as those in this case study. We performed five FlamMap runs (i.e., one run per wind direction) at 30 m cell-size resolution, using for each run a 10,000 fire ignition pattern drawn from the logit fire occurrence probability grid (Fig. 3). This way the 50,000 fires ensured individual burnable pixels burning once at least and more than one hundred times on average. The fire modeling landscape frame of 36,000 ha was large enough to capture the arrival of large fires ignited in the boundaries of the 3430 ha study area to the 155 black pine stands, and avoid the potential edge effects on modeling outputs and risk assessment. The burn period was set at 6 h and spot fire probability at 10%, consistent with active fire spread observed during the largest wildfires in the study area (i.e., Juslapeña wildfire in 2009).

Modeled outputs consisted of pixel-level overall burn probability and flame length probability. Burn probability defines the number of times a pixel burns as a proportion of the total number of fires, and is defined as follows:

$$\mathrm{BP}_{xy} = \mathrm{F}_{xy}/\mathrm{n}_{xy} \quad (3)$$

where F is the number of times the pixel xy burns and n_{xy} is the number of simulated fires per run (10,000 fires in this case study) at assumed fuel moisture and weather conditions (97th percentile in this case).

Fire intensity (Byram, 1959) is predicted by the MTT algorithm (Finney, 2002) and is converted to flame length as:

$$FL = 0.0775 \times I^{0.46} \quad (4)$$

where *FL* is flame length (m) and *I* fireline intensity (kW m^{-1}). The backing, heading and flanking fire spread flame length distribution generated from the multiple fires burning each pixel was used to calculate the flame length probability (FLP). The flame length probability (FLP) output file is the probability of flame length occurring at *i* categories of fire intensity levels (FIL_i), given that at least one of the simulated fires has burned the pixel. In this study FILs are expressed as 0.5 m flame length 20 categories, for FIL_1-FIL_{19} and $FIL_{20} > 9.5$ m.

2.6. Wildfire effects on black pine stands

Post-fire stand mortality response functions (RFs) to fire intensity ranges were generated as a function of flame length FILs (Byram, 1959) for the main black pine successional stages (Peterson and Ryan, 1986). The aim was to incorporate mortality expectations with pixel-level landscape scale fire modeling outputs and into the wildfire risk assessment framework (Finney, 2005; Calkin et al., 2011a). Response functions translated fire effects (considering only negative effects in this particular case study) into tree mortality for the midpoint of the 20 bin 0.5 m flame length-fire intensity levels (FIL_i), since fire intensity is a consistent fire metric that accounts for the main important fire characteristics (i.e., fire severity and rate of spread; Scott et al., 2013). We gathered the morphometric data (i.e., bark thickness, tree height and crown base height) required by the tree mortality model at the different successional stages from the 4th National Forest Inventory (MAGRAMA, 2010) and forest inventory data provided by the regional Forest Service (Gobierno de Navarra pers comm.). Due to the intensive and equal management in all the commercial forests, morphometric characteristics tend to be very homogeneous within the even-aged stands of the same successional stage. Generated tree mortality response functions have non-linear relations with increasing flame length and present relevant differences among the successional stages (Table 5).

2.7. Risk assessment

We assessed fire risk in terms of tree mortality (%) with procedures that consider two different assumptions: (*i*) given a fire occurs (all pixels have the same probability to get burned, BP = 1) as conditional tree loss (*cL*), and (*ii*) considering wildfire likelihood (spatially-explicit BP outputs) as expected tree loss (*eL*). We calculated pixel-level conditional tree mortality, from the custom RFs that described the impact of the flame length distribution generated from multiple fires as:

$$cL_j = \sum_{i=1}^{20} p(FIL_i) * RF_{ji} \quad (5)$$

where cL_j is the conditional tree loss (%) at successional stage *j*, $p(FIL_i)$ is the probability of the *i*-th category *FIL*, and the RF_{ji} is the response function of *j* successional stage at the *i*-th *FIL* (Table 5). Combining burn probability with conditional tree loss, we estimated a pixel-level probabilistic expectation of tree loss (Finney, 2005) as:

$$eL_j = \sum_{i=1}^{20} BP * p(FIL_i) * RF_{ji} \quad (6)$$

where eL_j is the expected loss as mortality (%) of black pines at *j*-th successional stage (i.e., thicket-stage, low pole-stage, high pole-stage, low timber-stage and high timber-stage) and *BP* is the burn probability, from Eq. (3). In this case study, the valued forest resources (i.e., black pine stands at different successional stages) do not spatially overlap and there are no benefits from fires.

2.8. Quantifying economic loss

We quantified the expected economic loss at pixel-level as:

$$eEL_{jx} = eL_j * V_x * P_j \quad (7)$$

where eEL_{jx} is the expected black pine timber economic loss (€ ha^{-1}) in a given forest stand *x* at a *j*-th successional stage, *V* is the standing timber volume (m^3 ha^{-1}) at the stand *x*, and P_j is the timber price (€ m^{-3}) at the successional stage *j*. In Eq. [7] eL_j is a positive fraction of unity value (among −1 and 0, since no positive fire effects are expected). Stand level timber volume data (*V*) was gathered from the 4th National Forest Inventory (MAGRAMA, 2010) and provided by the Forest Service (Pers comm.; Table 3). The timber volume on stand mainly depends on the site index and the previous management (number of thinnings). We considered successional-stage-specific prices (*P*) achieved in latest timber auctions as the reference to monetize the potential loss (Table 3). In the study area the timber is sold, on stand, in public timber auctions (96% of black pine forests are public) to private logging companies. Prices primarily depend on stemwood dimension and potential uses (i.e., paper pulp, poles, packaging and furniture). Stands present very similar conditions in terms of accessibility and machinability, and technical constrains established by the Forest Service with respect to the timber harvesting and slash (limbs and tops) treatment are the same for all stands within the study area.

2.9. Analyses

We built a black pine afforestation stand boundary map gathering the stand-polygons from the 1:5000 cadaster map (www.catastro.navarra.es; Gobierno de Navarra, 2014). Forest stand polygons correspond to the minimum management units and contain even-aged stand structures. Stand polygon spatial feature data was used to spatially locate individual stands or patches in the study area, analyze fire modeling outcomes and asses wildfire exposure and risk zonal statistics. Stand polygons have 3 ha on average, and all the 155 stands polygons cover around 448 ha (Table 2).

First we assessed wildfire exposure, combining respectively burn probability (BP) and flame length probability (FLP) fire modeling outputs, considering the observed frequency for the wind direction scenarios (Table 5). We then mapped BP for the whole study area, and we analyzed pixel-level BP and FLP among and within black pine successional stages in box-plots. Then, we assessed wildfire risk for the target polygons or stands. We calculated pixel-level expected tree loss (eL) and conditional tree loss (cL) as a mortality percentage value, combining BP and FLP with the custom response functions (RFs, Table 6) with ArcFuels (Ager et al., 2011). ArcFuels is a streamlined wildfire risk assessment tool which creates a trans-scale (stand to large landscape) interface to apply fire behavior models (e.g., FlamMap) to support forest management, wildfire behavior modeling, and wildfire risk assessments (Vaillant et al., 2013). We spatialized eL and cL in maps for the study area, and analyzed pixel-level differences among and within black

Table 3
Timber sale prices from public forest black pine auctions (Juslapeña Valley Municipality pers. comm.). The price, on stand, majorly depends on the log size and characteristics that allow for the final destination (use) of the timber. We used the average price on stand and stand timber volume to estimate the economic loss from wildfires.

Successional stage (abbreviation)	Use	Average price (€ m^{-3})	Average volume (m^3 ha^{-1})
Thicket-stage (TS)	–	–	57
Low pole-stage (LP)	Paper pulp	2.75	136
High pole-stage (HP)	Paper pulp and poles	12.50	172
Low timber-stage (LT)	Pallet and packaging	19.50	203
High timber-stage (HT)	Packaging and furniture	26.00	231

Table 4
Multivariate logistic regression model with 6 variables used to generate the fire ignition probability grid (Fig. 3; http://idena.navarra.es). Variables are ordered by decreasing importance.

Variables	Coefficient	Standard error	Chi-square	Degrees of freedom	P value
Distance to urban (m)	−0.0014	0.0003	16.32	1	<0.0001
Distance to power lines (m)	0.0002	<0.0001	16.00	1	<0.0001
Population density (p km^{-2})	0.0002	<0.0001	14.62	1	0.0001
Distance to railways (m)	0.0001	0.0002	14.10	1	0.0002
Distance to rural roads (m)	−0.0012	0.0004	9.55	1	0.0020
Land cover (class)				7	
Broadleaf forest	−0.9057	0.3751	5.83	1	0.0158
Shrublands	0.9032	0.4300	4.41	1	0.0357
Agricultural lands	0.5445	0.3328	2.68	1	0.1019
Herbaceous pastures	−0.3871	0.4550	0.72	1	0.3949
Conifer forest	−0.3586	0.4862	0.54	1	0.4608
Rocky areas	−0.4158	1.4664	0.08	1	0.7768
Urban development	−0.0991	0.3631	0.07	1	0.7848
Constant	−0.3250	0.3864	0.71	1	0.4003

pine successional stages into box-plots. Finally, we quantified potential timber economic loss. We monetized potential economic losses combining pixel-level eL, timber stock on stand and timber price for the respective successional stages, to map the results for the target polygons in the study area, and analyze the differences among and within black pine successional stages in box-plots. Differences between the individual stand patches were depicted graphically in a bubble-plot, considering the average values within stands.

3. Results

3.1. Wildfire exposure

Fire simulation outputs showed a clear spatially explicit BP pattern related to fuel types and dominant wind direction frequency distribution (Fig. 4). The southern and southwestern parts of the study area showed the highest BP values (BP > 0.1). These values decrease gradually moving northwards to BP < 0.03 in the large fire sink forests of the north and northwestern mountainous rim areas, which are mainly covered by timber-stage beech forests (Fig. 1). Nonetheless, some timber-stage black pine stands in the central and southwestern parts of the study area also presented some low BP patches, which correspond to low fuel load grazed forests (Fig. 1). Valley bottom areas, mainly located in the central part of the study area and covered by cereal crops and herbaceous pasture open areas, showed relatively high values (values over the average BP = 0.0583 obtained for the whole study area) and behave like fire transmitters (Ager et al., 2012; Alcasena et al., 2015b; Mitsopoulos et al., 2015), especially from fires initiated in the southern part of the study area. Fine-scale modeling results showed large differences among the fire occurrence grids (Fig. 3) and BP (Fig. 4). Although the eastern and northwestern boundaries of the study area presented both low occurrence and low burn probability, the central and southern areas showed very different patterns, especially areas distant from urban development where fire occurrence is low (IP < 0.35) and the burn probability is high (BP > 0.08). Box-plots showed a wide pixel-level variation in BP among the different successional stages (Fig. 5). The earliest successional stages (i.e., thicket-stage and low pole-stage forests) showed the highest pixel-level burn probability values, and values for latest successional stages (i.e., high pole-stage, low timber-stage and high timber-stage) showed much wider value range and lower values in most cases (i.e., 1st quartile value in late stages < 3rd quartile value in early stages).

Burning fire intensity levels showed a large pixel-level variation among successional stages. In general we observed a decreasing burning fire intensity probability from the bulk of FILs in early stages among FIL_4-FIL_9, to FIL_1-FIL_3 in the latest stages (Fig. 6). Although we found pixels burning in all fire intensity levels, only few pixels burn with high intensity (>FIL_{12}; Fig. 6). Nonetheless, in several cases and especially in thicket-stage and low pole-stage successional stages, wildfire would cause substantial tree mortality since burning fire intensity levels are usually higher than FIL_6 (2.5–3 m flame length), in which the mortality is total. Overall, FIL outputs showed the highest values in the low-pole untreated and dense overstocked stands, where high surface fire intensities in those laddered fuels would easily trigger active crown fires under extreme fire weather, as already observed in recent largest fires occurred in central Navarra (i.e., Izagaondoa and Juslapeña 2009 historic fires wildfires). In the timber-stage forests, burning fire intensity levels were lower than FIL_4 (1–1.5 m flame length) in most pixels, and at those values individual trees on mature forest stands would probably resist wildfires (Table 6). The low FIL values obtained in the timber stage forests are largely due to short grass type fuels found in many of those forests, where extensive livestock grazing prevents from hazardous fuel buildup.

3.2. Wildfire risk on black pine stands

Conditional tree loss (cL) exhibited considerable spatial variation within the study area, ranging from <15% on average in southeastern and northern mature forests, to near 75% in the central part of the study area where thicket-stage and low pole-stage stands dominate

Table 5
Fire modeling input parameters corresponding to extreme fire weather. We considered the most frequent wind directions their respective 97th percentile wind speeds. Fuel moisture content was derived from 97th percentile ERC fuel moisture content (Bradshaw and McCormick, 2000; Nelson, 2000). Historical weather data were gathered from the meteorological station of Pamplona (1997 to 2014; http://meteo.navarra.es). Fuel models correspond to Scott and Burgan (2005) and Fernandes (2009).

Wind scenario			Fuel moisture content (%)			
Direction (°)	Speed (km h^{-1})	Probability	Fuel loading category	Fuel model type		
				GS1, GR5, GR2, GR4, SH6, SH5	TU3, PCL, SH3, GR3	GR1, SH3, TL2, SH8
67.5	32 (km h^{-1})	0.43	1-h	4	6	8
337.5	35 (km h^{-1})	0.28	10-h	5	7	9
45.0	19 (km h^{-1})	0.17	100-h	8	9	12
180.0	31 (km h^{-1})	0.06	Live herbaceous	20	45	70
22.5	23 (km h^{-1})	0.06	Live woody	60	85	100

Table 6
Response functions for 0.5 m 20 flame length interval categories, fire intensity levels (FILs), in *Pinus nigra* afforestation successional stages. Values indicate the mortality (%) at the midpoint intensity, calculated as proposed by Peterson and Ryan (1986). Intensities > FIL_6 cause total mortality in all the successional stages.

Successional stage (abbreviation)	Expected loss, as tree mortality (%) within stands at FIL(m) class midpoint					
	FIL_1	FIL_2	FIL_3	FIL_4	FIL_5	FIL_6
	0–0.5	0.5–1	1–1.5	1.5–2	2–2.5	2.5–3
Thicket-stage (TS)	0	97.4	100	100	100	100
Low pole-stage (LP)	0	0	98.7	100	100	100
High pole-stage (HP)	0	0	4.6	100	100	100
Low timber stage (LT)	0	0	0	71.5	100	100
High timber-stage (HT)	0	0	0	1.1	9.6	100

(Fig. 1 and Fig. 7). The average pixel value among all stands was 34.38%. The highest conditional tree loss values in the latest successional stage stands were obtained in the very steep slope pixels and northern boundary stand neighboring with high fuel load fuel models (dots over 90th percentile, Fig. 8). As expected, in agreement with the RFs (Table 6), we found an increasing fire resistance trend from the earliest to the latest successional stages (Fig. 8), ~ 90% on average for the thicket-stage and low pole-stage, and 10% in the high timber-stage. In the study area, this observed trend is due not only to the stand morphological characteristics, but also to low fire intensity in many grass-type understory at wooded pastures (GR3; Table 1).

The expected tree loss (eL) pixel-level map illustrates a large range of variability in the study area (Fig. 9). Box plots depicting the expected tree loss showed a large variability (Fig. 10). Many areas with high conditional tree loss (>80%) coincided with high burn probability (>0.1), which occurred in central and especially southern part of the Juslapeña Valley. In general, although the late successional stages showed high values in a few cases (dots over the average 2.5% eL), early stages resulted in higher loss expectations (>8%). Largest pixel-level variability within successional stages was found for the high pole-stage, where the difference between the 1st and 3rd quantile values was higher than 5%. As with burn probability and conditional tree loss, within individual stand polygons the values for pixel-level tree loss for high pole stage decrease inwards, they presented a fire-sink behavior.

3.3. Potential timber economic loss

Pixel-level expected economic loss (eEL) was likewise highly variable throughout the study area (Fig. 11). The highest expected economic loss was found in the high pole-stage stands of the central part of the study area, and not in the early successional stages due to the low or nil commercial interest (Table 3). Southeastern stands, for instance, showed a high burn probability and conditional tree loss (Fig. 4 y 6), but have a very low economic loss expectation from large fires (Fig. 11) because they correspond to thicket stage forests. The timber stage forests, by contrast, have a high timber economic value (Table 3) but they showed a low conditional tree loss (<5%; Fig. 12), and only a few stand boundary pixels (values over 90th percentile; Fig. 12) burning with high intensity and likelihood show high economic loss expectations. Among successional stages, high pole-stage stands obtained the highest average values (124 € ha^{-1}), >2.5 times the low timber-stage stands. Low pole-stage stands and low timber stage stands presented similar average eEL (~39 € ha^{-1}), although low timber-stage stands had 5.5 times lower average cL than pole-stage, but timber price is 7 times higher (Table 3).

The bubble-plot of average BP, cL and eEL showed that, in general, high BP and especially high cL were positively correlated with high eEL values (i.e., high bubble size; Fig. 13). However, expected economic loss vary widely for the same fire hazard levels either because of timber price, timber stock on stand, successional stage fire resistance or

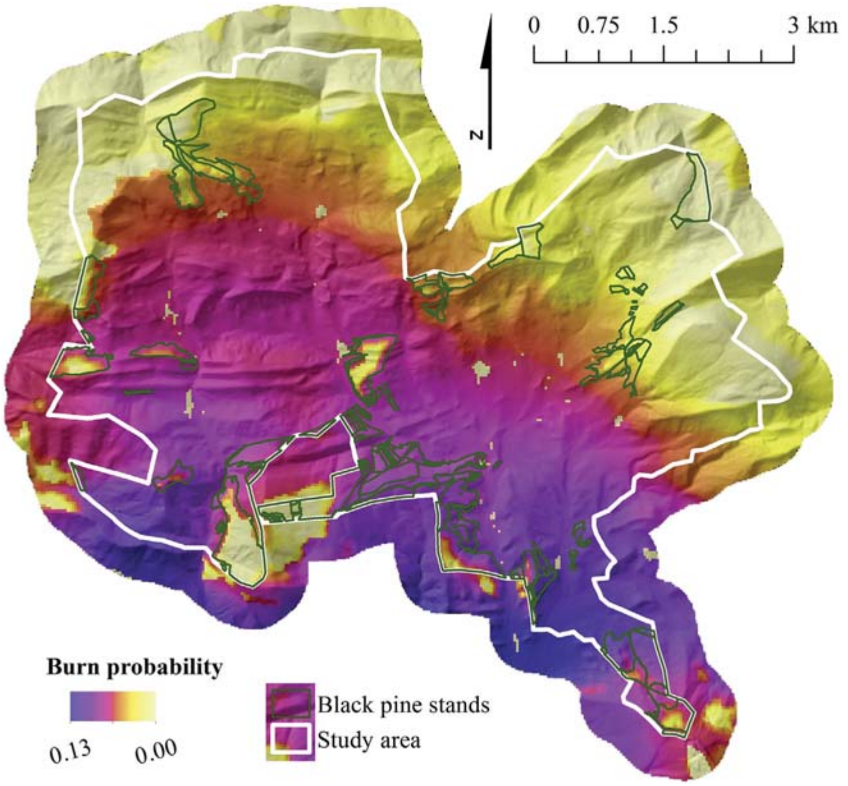

Fig. 4. Map of burn probability (BP; 30 × 30 m cellsize) across the study area. This map was generated combining the BP output maps from the fire modeling scenarios considering the wind direction probability (Table 5). BP allows to identify the areas that in case of a large fire event will most likely burn. Values range from a high near 0.01 in the southern part, except to low spread-rate grazed stands, to BP < 0.03 in northern and northeastern mountainous fire-sink beech forests.

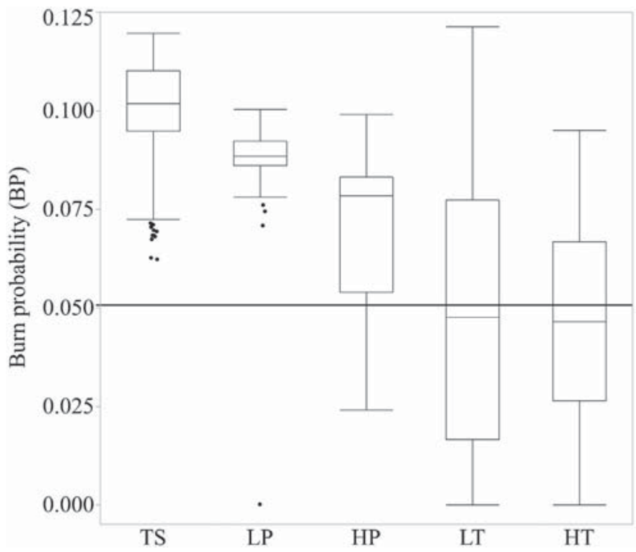

Fig. 5. Pixel-level burn probability box-plots for the black pine successional stages (see Table 2 for abbreviations). The box indicate the 1st/3rd quartiles, the whiskers indicate 10th/90th percentiles, the black line within the box is the median, and the dots indicate values below 10th percentile or above the 90th percentile. The horizontal continuous line indicates all pixel average value (BP = 0.0506).

burning probability. Stand designations corresponding to ticket-stage and low pole-stage showed high average burn probability (avg. BP > 0.05) and conditional tree loss (avg. cL > 75%). This finding was expected since the first afforestation (i.e., now corresponding to timber-stage forests, grazed in many cases) were completed in the highest elevation marginal lands, and the latest (i.e., now thicket-stage and low pole-stage, excluded from grazing the last 25 years) in abandoned extensive grazing areas closed to fast fire spreading agricultural lands. However, there is a evident influence of the successional stage on the average eEL, since highest values (>100 € ha^{-1}) were not necessarily found over 97th percentile average BP and average cL conditions (Fig. 13).

4. Discussion

The current study presents a methodology for quantifying wildfire risk from large fires in timber commercial stands of *Pinus nigra* subsp. *nigra* located in northern Spain, but it can be applied elsewhere. For this purpose, in this fire modeling approach we quantitatively assessed the expected economic losses (€ ha^{-1}) at high resolution (30 m) and among and within different black pine stands at different successional stages. We also integrated a logit model to generate spatially balanced fire ignition points and replicate the most likely spatial scenarios of fire occurrence. Fire weather is a decisive conditioning factor and this fire modeling approach was carried out using extreme conditions (i.e., 97th percentile), since those conditions contribute to large fire activity, which is primarily responsible for the burned area in the region. Wildfires burning under mild fire weather conditions commonly represent a limited threat to valued resources, and local fire crews result very efficient in off-season fire suppression. The use of black pine successional-stage-specific response functions coupled with the most probable burning fire intensities in all pixels (the fire model considers the distribution of heading, flanking, and backing flame lengths and their respective frequencies) provided a robust quantitative indicators in terms of tree mortality. As far as we know, this work is the first in Mediterranean Europe that accounts for large fire spread and fire effects at different burning fire intensities, and results should be viewed as a reference values to reduce uncertainty and support decision-making. Various sources of error in the models and data are possible (e.g., using the same species model to characterize the canopy metrics for the whole study area from LiDAR), as well as modeling output bias since we only accounted for extreme fire weather conditions that slightly overestimate wildfire likelihood and hazard with respect to intermediate and mild conditions (Thompson et al., 2015).

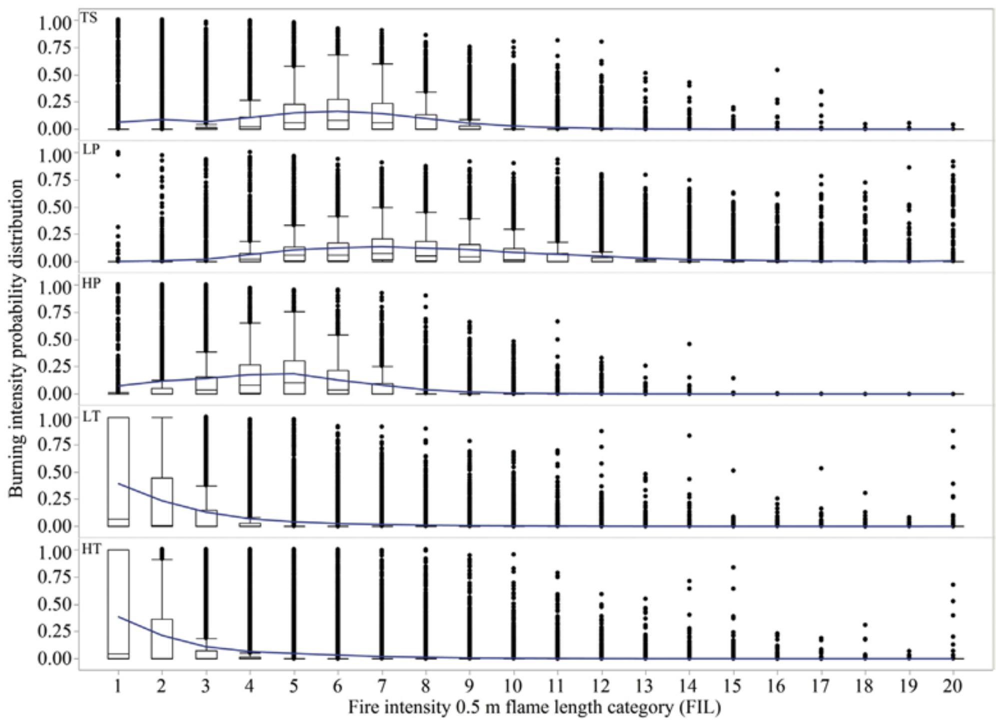

Fig. 6. Pixel-level burning intensity probability distribution by 0.5 m 20 flame length categories (FILs), for the black pine successional stages (see Table 2 for abbreviations). The box indicate the 1st/3rd quartiles, the whiskers indicate 10th/90th percentiles, the black line within the box is the median, and the dots indicate values below 10th percentile or above the 90th percentile. The blue line indicates the trend on average value of FILs at the different successional stages. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

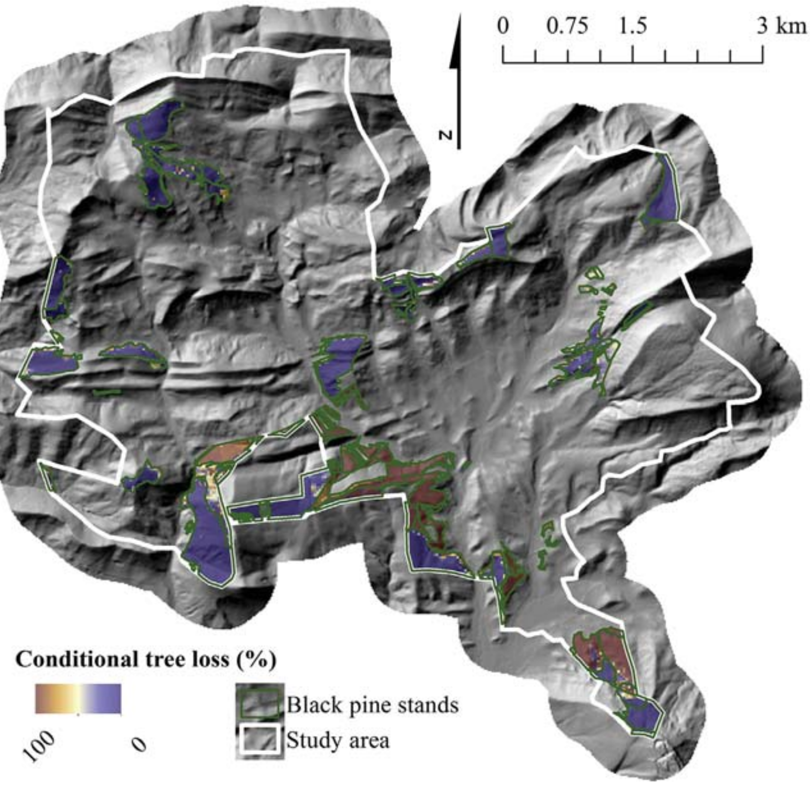

Fig. 7. Map of conditional tree loss (cL; 30 × 30 m cellsize), given a fire occurs, shown as post-fire mortality (%) for black pine afforestation in the study area. Only negative effects are considered from fire, which are estimated with fire modeling intensity outputs and successional stages specific mortality models (Table 6).

The risk analysis in this study is based on the problem-fire escaping from first attack. Small fires burning under mild conditions do not represent a real threat to conifer afforestation due to limited spreading distances and small burned areas, apart from a lower severity. In the study area, fires larger than 100 ha are rare events burning under extreme fire weather conditions (i.e., 97[th] percentile fire weather). Only 3 fires larger than 100 ha have been observed in the fire modeling landscape frame (the largest event was the Julsapeña fire, 334 ha) 1985–2013 period occurrence data (MAGRAMA, 2014). In fact, the 876 ha Izagaondoa fire was the largest for the same fire regime-fuelscape in extended fire occurrence area, and even in that case large fire frequency was not changing substantially. Moreover, suppression efforts successfully contained flanking-backing fire spread and almost all burned area corresponded to escaping heading fire (Alcasena et al., 2015b). Average fire size for the 50,000 modeled fires was 1653 ha (80% of fires were >100 ha), which is a good approximation to the potential extent of Izagaondoa fire (i.e., excluding suppression efforts). Similarly, suppression efforts together with the rarity of these events made the average burn probability (0.055) higher than the empirically estimated average value (0.002). Although the conditional burn probability (i.e., given a fire occurs under extreme fire weather conditions within the modeling landscape) is not comparable with other assessment areas elsewhere and might overestimate potential economic losses for a cost-effective mitigation analysis, it provides a reliable quantitative assessment to managers in treatment prioritization within the study area (Ager et al., 2007).

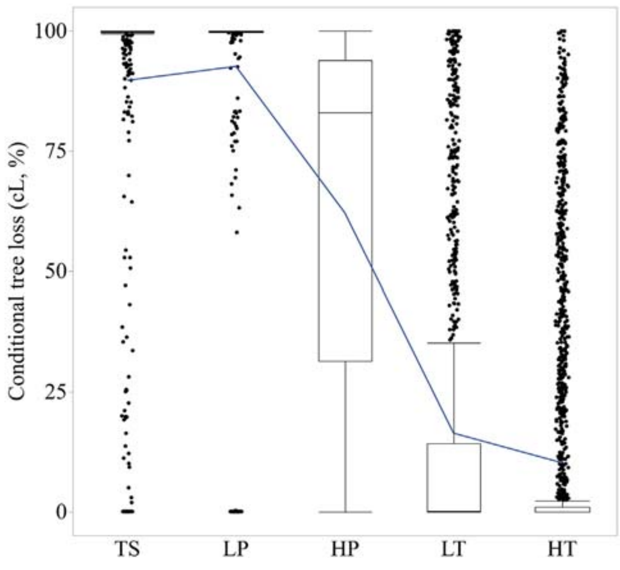

Fig. 8. Pixel-level conditional tree loss (cL, in %) box-plot for the black pine successional stages (see Table 2 for abbreviations). The box indicate the 1[st]/3[rd] quartiles, the whiskers indicate 10[th]/90[th] percentiles, the black line within the box is the median and the dots indicate values below 10[th] percentile or above the 90[th] percentile. The blue line indicates the average cL in the black pine successional stages, which was 89.73%, 92.65%, 62.19%, 16.34% and 10.14% respectively for the TS, LP, HP, LT and HT successional stages. The average value for all pixels is 34,38% (not shown). (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

Even though the results are specific to this case study, we would expect similar results in most of the 28,200 ha afforested with black pine in Navarra, as well as in many black pine forests in the pre-Pyrenees of Aragon and Catalonia in Spain, where fire weather conditions and fuel configurations in the landscape do not differ much from our study area. However, natural black pine forests in other autonomous regions (*Pinus nigra* subsp. *salzmannii*) often present uneven-aged structures and difficulties for the commercial exploitation (González-Olabarria and Pukkala, 2011). Therefore, the quantification of economic losses would require adapted RFs to these structures and different timber prices on stand that would be presumably lower, due to the higher costs in exploitation and higher distance to the main sawmills and cellulose industries of northern Spain.

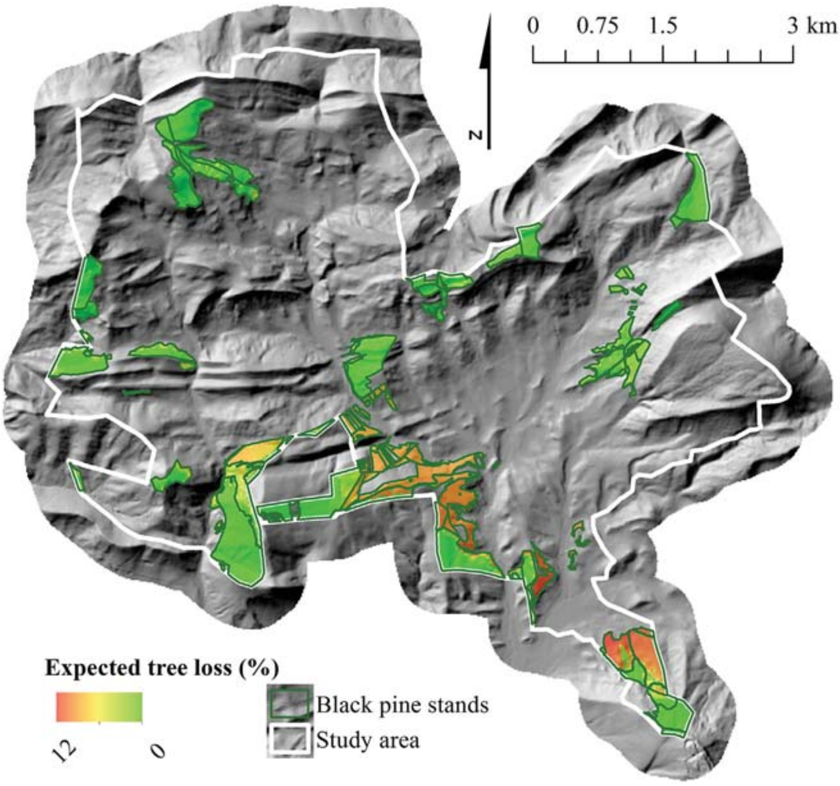

Fig. 9. Map of expected fire effects (eL); 30 × 30 cellsize), considering the fire modeling BP (Fig. 4), shown as post-fire mortality (%) for black pine afforestation in the study area. Expected tree loss was estimated from historically based fires, which were modeled under 97th percentile fire weather conditions, and accounting for successional stages specific response functions (Table 6).

Outputs in this study showed that black pine stands might experience a range of different fire behaviors, and their forest structure can enhance or diminish fire spread-rate and intensity. While young and dense afforestations can accelerate fires and become ember emitting areas, managed late successional afforestations can pose an obstacle to large fire spread. Most substantial changes occur after first thinning at pole-stage when the tree crown continuity is disrupted and there is not a high fine fuel load that could speed up fire spread rate (the previously closed canopy cover did not allow understory growth). Additionally, results showed that if the stand is integrated into extensive grazing areas we obtain a more fire resistant structure (Fig. 7), which also entails a fire spread slowing area. These open stands of large trees with small amounts of ground fuels are less susceptible to suffer damage. Conversely, in the absence of any other understory management strategy, fuel load buildup, being mainly dominated by heliophilous shrublands in the first stage and later by *Quercus* ssp. undesired regeneration (if the goal is conifer timber production), promotes high fire intensities (>2.5 m flame length; Table 6) that supposes a real threat for the old grown dominant trees.

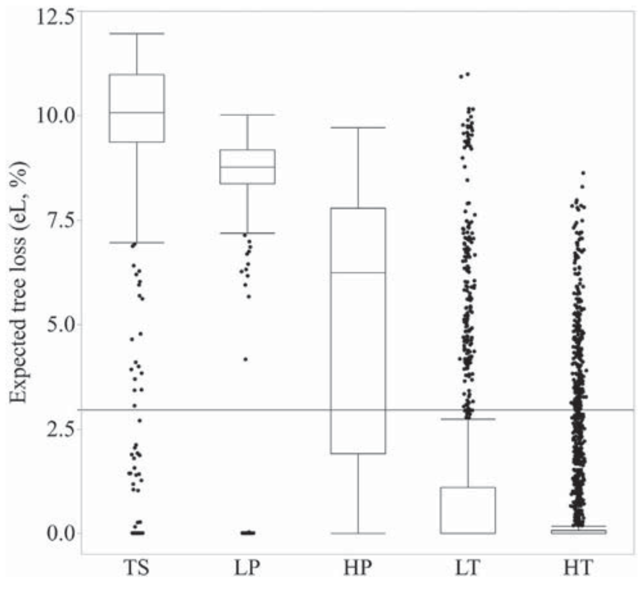

Fig. 10. Pixel-level expected tree loss (eL, in %) box-plot for the black pine successional stages (see Table 2 for abbreviations). The box indicate the 1st/3rd quartiles, the whiskers indicate 10th/90th percentiles, the black line within the box is the median and the dots indicate values below 10th percentile or above the 90th percentile. Average conditional tree in the black pine successional stages was 9.22%, 8.22%, 4.73%, 1.20% and 0.67% respectively for the TS, LP, HP, LT and HT successional stages (not indicated). The horizontal continuous line indicates all pixel average value (eL = 2.95%).

The implementation of fuel treatments is the most effective and extended strategy to mitigate fire risk (Cochrane et al., 2013; Hudak et al., 2011; Murphy et al., 2010). Although underburning is not a broadly used fuel treatment strategy in central Navarra, recently implemented prescribed burns by Bomberos de Navarra with the support of the regional Forest Service in the study area indicate that this could be a good risk mitigation strategy when is implemented by trained personnel. According to the successional stage specific response functions for black pine (Table 6) and observed flame lengths on field experiences in the study area, results suggest that using prescribed fire could commence after high-pole stage, within a low-mortality flame length threshold (<1.5 m of flame length) that cause assumable mortality (<10%). Prior to start using prescribed fires as extensive treatment in all commercial afforestations (not only in strategically located points that would significantly reduce large fire spread), further studies are required to know how the eventual dead tissue in the logs would affect to the final timber quality on thick logs that present good technologic aptitudes for furniture. One possible combination in sequential treatments for risk mitigation in target areas would be a heavy-weight thinning

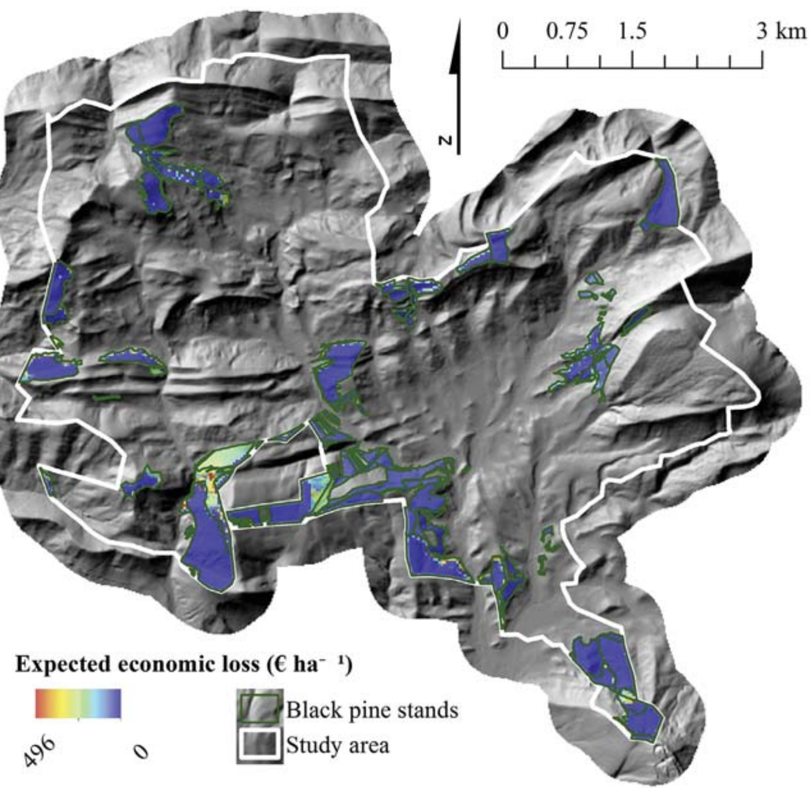

Fig. 11. Map of timber economic loss (eEL; 30 × 30 m) from large fires in the black pine afforestation, shown as expected economic loss (€ ha^{-1}). We accounted from forest inventory data (IFN 4; Forest Service pers. comm) and current timber prices gathered in public auctions (Ayuntamiento de Juslapeña pers. comm.) to economically evaluate expected fire impacts (Fig. 9).

from below (i.e., at 25–30 years after planting with extractions of 30–40% basal area), prescribed burn (1 year after thinning, to reduce piled slash) and grazing (1 year after prescribed burn, inclusion of the stand in extensive pasture management units). A possible fire regime restoration that would increase fire resistance of black pine would be too complex to achieve and not feasible to carry out (like in most southern EU landscapes), because the departure from former conditions (previous to any anthropic impact in modeling of the landscapes) is unknown (Moritz et al., 2014) and housing and agricultural lands intermingle with forests in Mediterranean cultural landscapes. Likewise, in the mid-term future we would expect substantial differences in the forest and fire environment (Muñoz et al., 2016), and therefore the implemented framework should be readapted in order to account for different biophysical and socio-economic scenarios when providing outcomes for use in decision-making.

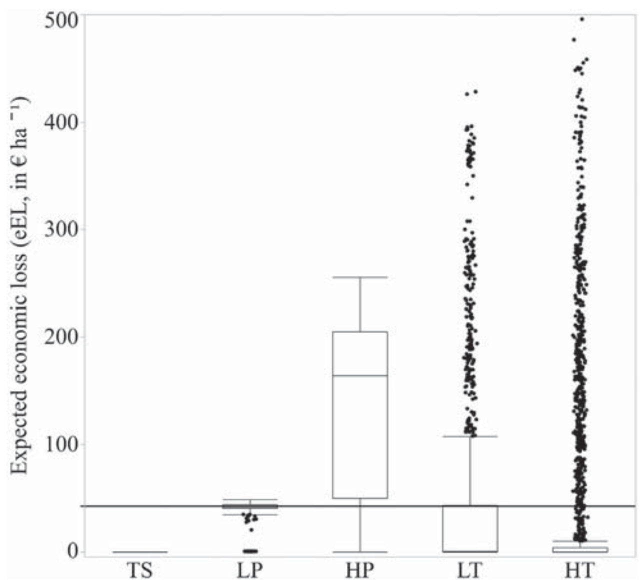

Fig. 12. Pixel-level expected timber economic loss (eEL, in € ha^{-1}) for the black pine successional stages (see Table 2 for abbreviations) considering the timber existences on stand and the latest prices from public auctions. The box indicate the 1st/3rd quartiles, the whiskers indicate 10th/90th percentiles, the black line within the box is the median and the dots indicate values below 10th percentile or above the 90th percentile. Average conditional tree in the black pine successional stages was 0 € ha^{-1}, 39.48 € ha^{-1}, 124.27 € ha^{-1}, 46.55 € ha^{-1} and 38.50 € ha^{-1} respectively for the TS, LP, HP, LT and HT successional stages (not indicated). The horizontal continuous line indicates all pixel average value (eEL = 42.41 € ha^{-1}).

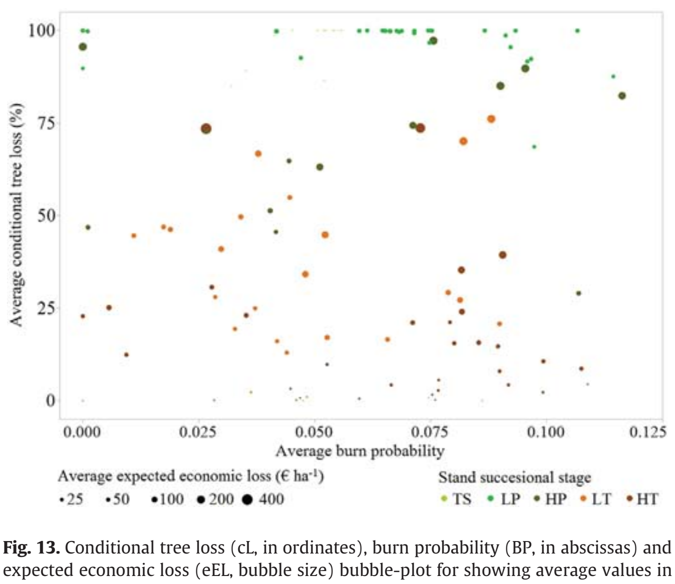

Fig. 13. Conditional tree loss (cL, in ordinates), burn probability (BP, in abscissas) and expected economic loss (eEL, bubble size) bubble-plot for showing average values in individual black pine polygons or stands (see Table 2 for abbreviations).

The fire risk analysis and the expected economic loss assessment at this scale are useful resources to inform forest and fire management al local scale, and this study highlighted that accounting for stand-level differences in the timber price and timber stock is crucial to assess fire risk accurately. The small size of management units together with a highly fragmented land ownership pose nowadays relevant constrains for management. Although most forests are declared public and managed under the supervision of the Forest Service, they are owned by different small rural municipalities (they manage their own public lands separately from the neighboring) and require a detailed scale of analysis. Landscape managers need to spatially identify individual stands (management units) with high potential economic loss in fine scale maps and in order to prioritize treatments. This is one of the major advantages of this framework versus expert-criteria based prioritization. Further studies should integrate more valued resources from forests (e.g., hunting, mushroom picking and social use) and throughout the landscape (e.g., housing and infrastructures, water yield). Then, assessing changes in fire risk mitigation from different spatial strategies (Ager et al., 2013; Salis et al., 2016) and analyzing tradeoffs among various management objectives (Vogler et al., 2015) could complement the current case study to better inform landscape managers over risk management plans.

5. Conclusions

We implemented a methodology that holds a great potential for wildfire managers to quantitatively assess potential economic losses in timberlands from large wildfires in northern rim Mediterranean areas of southern Europe and elsewhere. Results allow local managers prioritizing and prescribing individual stand-scale-specific risk mitigation strategies better adapted to growth stage and wildfire risk level. This study could be considered as a baseline to support decision-making at landscape scale in prioritizing mitigation efforts and a preliminary step in evaluating potential benefits from different fuel treatment strategies, where financial contribution of forest resources to rural communities continues to be of great importance.

Acknowledgments

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CHAPTER 3 – Wildfire risk transmission to communities

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Assessing Wildland Fire Risk Transmission to Communities in Northern Spain

Fermín J. Alcasena [1,*], Michele Salis [2], Alan A. Ager [3], Rafael Castell [4] and Cristina Vega-García [1,5]

[1] Agriculture and Forest Engineering Department (EAGROF), University of Lleida, Alcalde Rovira Roure 191, Lleida 25198, Spain; cvega@eagrof.udl.cat
[2] Euro-Mediterranean Center on Climate Change (CMCC), IAFES Division, Via Enrico De Nicola 9, Sassari 07100, Italy; michele.salis@cmcc.it
[3] USDA Forest Service, Rocky Mountain Research Station, 72510 Coyote Road, Pendleton, OR 97801, USA; aager@fs.fed.us
[4] Bomberos de Navarra, Agencia Navarra de Emergencias (ANE), Calle Aoiz 35 bis 3°, Pamplona 31004, Spain; rcastels@navarra.es
[5] Forest Sciences Centre of Catalonia, Carretera de Sant Llorenç de Morunys km 2, Solsona 25280, Spain
* Correspondence: ferminalcasena@eagrof.udl.cat; Tel.: +3-497-370-2673

Abstract: We assessed potential economic losses and transmission to residential houses from wildland fires in a rural area of central Navarra (Spain). Expected losses were quantified at the individual structure level (n = 306) in 14 rural communities by combining fire model predictions of burn probability and fire intensity with susceptibility functions derived from expert judgement. Fire exposure was estimated by simulating 50,000 fire events that replicated extreme (97th percentile) historical fire weather conditions. Spatial ignition probabilities were used in the simulations to account for non-random ignitions, and were estimated from a fire occurrence model generated with an artificial neural network. The results showed that ignition probability explained most of spatial variation in risk, with economic value of structures having only a minor effect. Average expected loss to residential houses from a single wildfire event in the study area was 7955€, and ranged from a low of 740 to the high of 28,725€. Major fire flow-paths were analyzed to understand fire transmission from surrounding municipalities and showed that incoming fires from the north exhibited strong pathways into the core of the study area, and fires spreading from the south had the highest likelihood of reaching target residential structures from the longest distances (>5 km). Community firesheds revealed the scale of risk to communities and extended well beyond administrative boundaries. The results provided a quantitative risk assessment that can be used by insurance companies and local landscape managers to prioritize and allocate investments to treat wildland fuels and identify clusters of high expected loss within communities. The methodological framework can be extended to other fire-prone southern European Union countries where communities are threatened by large wildland fires.

Keywords: wildland urban interface; wildfire simulation modeling; wildfire risk transmission; community fireshed

1. Introduction

Most wildfires that cause human fatalities and losses to property occur in the rapidly expanding interface areas between wildlands and human development [1,2]. This area where residential and other infrastructures intermingle with flammable vegetation is widely known as wildland–urban interface (WUI) or rural–urban interface (RUI) [3,4]. While the former definition is mainly used for predominantly wildland vegetation areas surrounding developed areas, the latter is most commonly used in Mediterranean landscapes where fuels have been influenced by human activities for millennia [5,6]. In areas lacking sharp transitions between development and wildlands, where structures are surrounded by hazardous fuels, the term intermix has been used to describe the juxtaposition of fuels and dwellings [7]. In all cases, a number of factors have contributed to wildfire losses in developed areas (hereafter WUI), including urban expansion, increased fuel loadings from expansion of shrub and forest vegetation into abandoned agricultural lands, and suburban sprawl over metropolitan agricultural belts [8–10]. Likewise, fire suppression policies have contributed to a buildup of fuels in and around developed areas, resulting in higher hazard within developed areas and structure ignition [11]. Wildland fires in the WUI are a growing concern at global scales due to escalating losses to life and property [12,13], and have become a priority for wildfire management policies in many fire-prone areas [14].

Previous efforts on WUI wildfire risk characterization in Mediterranean landscapes have emphasized the importance of flammable vegetation surrounding communities [15], since fuel loadings are directly related to fire intensity and structure loss [16]. Aggregation of dwellings (isolated, grouped and urban center) combined with vegetation types or land covers have been proposed as a WUI classification system to inform risk and vulnerability assessments [17,18]. Other studies have focused on ignition likelihood to measure wildfire risk [19–21]. The vast majority of fires in the Mediterranean basin are caused by humans [22–24], and most fire-occurrence modeling studies include explanatory variables to describe human activities, such as population density, accessibility (e.g., distance to roads, distance to railways, distance to forest tracks) and human activities [25,26]. However, neither of these previous approaches account for the likelihood of loss from large fires (e.g., 5000–50,000 ha) that ignite at some distant location and spread to urban development. Thus, low fire ignition probability close to a WUI area does not necessarily translate to low burn probability, and vice versa. Moreover, fire intensity can substantially vary depending on fire weather and fire front spreading direction [27,28].

To better account for the spatial scale of wildfire risk to human communities, a growing number of researchers have employed wildfire simulation methods [29,30]. Both burn probability and fire intensity in the home ignition zone (HIZ, the immediate 30–60 m-buffer area around dwellings) [11] can be estimated by simulating a large number of fires (e.g., 10^4–10^5) to assess wildfire exposure from large fires [31,32]. These estimates can be then used in risk assessments to quantify the potential socioeconomic impacts, including expected net value change on residential structures [27,28,33]. While it is generally agreed that higher wildfire exposure results in larger losses in the WUI, variability in structure susceptibility and economic valuation can substantially affect risk estimates at the scale of individual dwellings. For instance, high overall exposure levels can be mitigated by construction materials and structure design [34]. These differences in construction can be incorporated into risk assessments using different susceptibility relationships [35,36]. Simulation studies can also be used to understand the scale of risk to communities to help identify responsible landowners [37,38]. For instance, using wildfire transmission analysis, fire effects on valued resources can be traced back to the ignition location [39], and landscape planning to reduce hazardous fuels can then target these areas for fuel treatments [40].

In this paper, we assess potential wildfire economic losses and transmission to residential houses in the rural communities of Juslapeña Valley, northern Spain. We used simulation modeling to map the source of wildfire exposure to communities and estimated the expected financial loss at the scale of individual structures. The simulation modeling incorporated a fine scale ignition probability grid developed from historical fire locations. Simulation outputs were used to estimate a number of exposure metrics, including burning probability and fire intensity. We estimated expected loss in the community using wildfire exposure metrics combined with a structure susceptibility function. The later was generated by a panel of local experts using an interactive structured communication technique. We also conducted a transmission analysis to delineate community firesheds and understand the source of wildfire exposure to communities. The methods provide a number of new ways to examine wildfire exposure to communities that can inform wildfire protection and improve fire resiliency in rural–urban interface areas in the Mediterranean region.

2. Material and Methods

2.1. Study Area

The study area is located in the Juslapeña Valley, central Navarra (Spain), 18 km north of the city of Pamplona (Figure 1A). The Juslapeña Valley is a 31.63 km^2 municipality with 548 inhabitants dispersed among 14 small rural villages or councils (minimum administrative division). The climate is transitional Mediterranean with annual rainfall around 1000 mm, a water shortage period from July to September corresponding to the wildfire season, and average maximum temperatures over 30 °C in the warmest month (meteo.navarra.es). The landscape is a mosaic of dryland cereal crops covering the valley bottom, mesoxerophytic pastures with shrubby edgings on marginal agricultural lands (*Genista scorpius* L., *Juniperus communis* L., *Buxus sempervirens* L., and *Prunus spinosa* L.), downy oak (*Quercus pubescens* Mill.) forests on south facing slopes (replaced by *Quercus ilex* L. in shallow soil foothills), beech (*Fagus sylvatica* L.) forests on high elevation north facing slopes, and scattered stands of black pine (*Pinus nigra* Arn.) [41]. Land management is largely conditioned by ownership. Most forests and natural herbaceous pastures are council common lands and agricultural fields are owned by local inhabitants. Community housing is located at mid-slopes, usually surrounded by agricultural lands and orchards at the front southern side, and forested lands arrive closer at the back (Figure 1B). We focused our analysis on residential houses (n = 306 structures), and we did not consider other structures or constructions such as agricultural warehouses. In the study area, there are no industrial sites or sport-recreational facilities. The largest observed wildfires are characterized as fast-spreading one-day summer events with less than 1000 ha burned (e.g., Juslapeña Fire in 2009 and San Cristobal Fire in 2001). Most fires are caused by humans, while lightning represents only 5% of ignitions (1985 to 2013 fire records; mapama.gob.es).

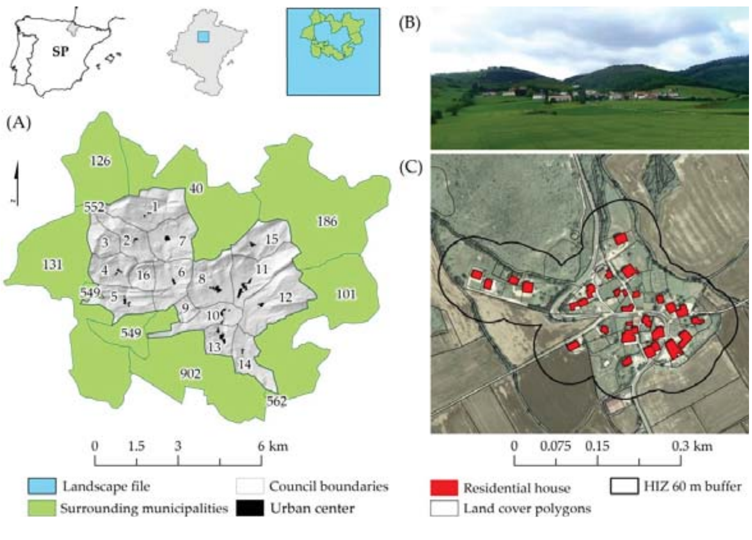

Figure 1. Location of the Juslapeña Valley (3163 ha) in central Navarra (Spain) (**A**). The numbers refer to the regional cadaster council polygon (1 to 16) and municipality code (**B**). The 36,000-ha wildfire modeling domain framed by the landscape file (LCP) encompassing the study area had a wider extension to the south to account for incoming fires from the fire-prone areas of central Navarra. Land covers in the cultural landscapes present sharp edges in vegetation (**B** urban center of the council No. 8). Detailed cartography and cadaster polygons (scale 1/5000) were used to generate surface fuel maps (sigpac.navarra.es) and locate residential houses (catastro.navarra.es) (**C**). The HIZ is the 60-m buffer around structures [11], and was calculated for each residential house to conduct this study. In the figure (**C**) we show the external HIZ contour of the residential houses in urban center of the council No. 8.

2.2. Wildfire Simulation

We gathered multiple datasets and geospatial inputs for this modeling approach (Figure 2). We simulated wildfire spread and behavior (fire size, burned area polygons, flame length probabilities, and conditional burn probability) within a 36,000-ha fire modeling domain. Overall, we conducted separated simulations for the most frequent extreme weather conditions of the wildfire season, thus obtaining different sets of modeling outputs. All the output raster grids were obtained at modeling resolution. Details are presented below in the following sections (Table 1).

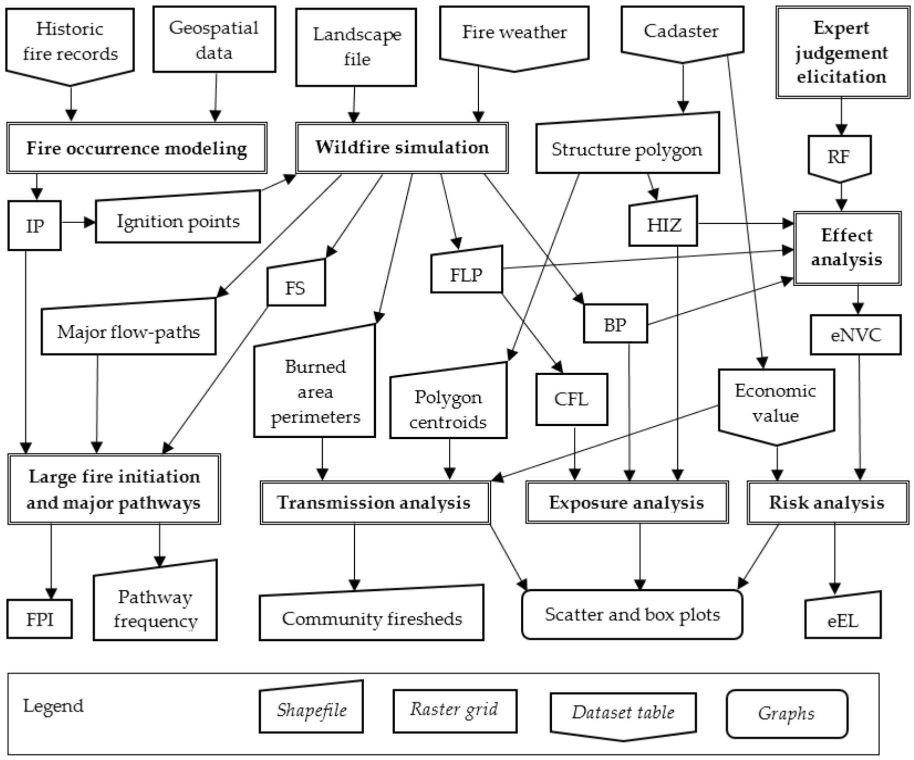

Figure 2. Wildfire simulation and analysis process summary flowchart. Wildfire simulation requires fire weather, landscape and fire ignition input data. Fire initiation, transmission, exposure and risk analysis use different fire modeling outputs. Exposure and risk analyses were conducted at individual structure HIZ level. Results were presented in maps or graphics. See Table 1 for the abbreviations.

Table 1. Summary table with the abbreviations used in this study for the main geospatial inputs, modeling outcomes and analysis results. The terms are described and contextualized for the use in this study. We provide further details in the following sections.

Name (Abbreviation)	Description and Use
Home ignition zone (HIZ)	Area surrounding structures within a 30–60 m-buffer [11]. HIZ was used to assess wildfire exposure and risk on the individual residential houses located in the study area.
Ignition probability (IP)	Fire occurrence probability grid (0–1) generated by artificial neural network analysis [26] of historical ignition locations. It was used to calculate FPI and generate the simulated fire ignition locations.
Fire size (FS)	Fire size (ha) resulting from each individual simulated wildfire. Fire size is output from simulations along with the ignition location. It was combined with IP to generate FPI.

Table 1. *Cont.*

Name (Abbreviation)	Description and Use
Fire potential index (FPI)	Is the grid generated with FS and IP, and it was used to identify large fire initiation areas [31]. The FPI provides spatially explicit valuable information to target anthropic fire ignition prevention priority areas on fire-prone landscapes.
Flame length probability (FLP)	Probability of a fire of a specific flame length given that a pixel burns under the simulated conditions. FLP is output in 0.5-m classes and sums to 1 for a given pixel. A distribution of flame lengths is generated for each pixel since fires can arrive as heading, flanking or backing fires.
Conditional flame length (CFL)	Probability-weighted flame length (m) calculated from the FLP output. CFL was summarized for the HIZ to estimate wildfire hazard and exposure to residential houses [35].
Burn probability (BP)	Number of times a pixel burns as a proportion of the total number of simulated fires (0–1). BP average values for each HIZ were used to estimate wildfire likelihood and assess wildfire exposure to residential houses [35].
Response function (RF)	The susceptibility of structures as a function of flame length represented by percent value loss (%) [42]. It was obtained from expert judgment [35].
Expected net value change (eNVC)	Expectation of *gain* or *loss* in values expressed on a percentage basis (%) [28]. Derived from combining burn probability, intensity, and susceptibility functions to estimate expected change on a percentage basis for structures [27]. Only expected losses were considered in the study.
Expected economic loss (eEL)	Expected loss expressed specifically in economic values (€) given a fire ignition and spread at assumed extreme fire weather conditions. Quantified as the product of the cadaster value of the structures and the average eNVC within the HIZ.

2.2.1. Landscape File and Fire Weather Input Data

We compiled the complete set of input data as required by the FlamMap fire simulator [43], including landscape file (LCP) and wildfire season extreme fire weather data. The LCP is a gridded frame containing the characteristics of the terrain, surface fuels and canopy fuel metrics. The terrain (aspect, slope and elevation) was derived from 5-m resolution digital terrain model raster data (ign.es). Standard fuel models [44,45] were assigned to 1/5000 scale land use land cover considering species composition, shrub cover and forest growth stage (idena.navarra.es and sigpac.navarra.es) (Figure 3). Canopy metrics (canopy height, canopy base height, canopy bulk density and canopy cover), were derived from low density LiDAR data (0.56 returns m^2; ign.es) using FUSION [31,46]. The surface fuel and canopy metric characterization and required raster grid generation were detailed in previous studies [47,48]. The LCP was assembled at 20-m resolution [49] and comprised a 36,000-ha fire modeling domain (Figure 1A). Extreme fire weather conditions were derived using Fire Family Plus [50] from the hourly records of the Pamplona automatic weather station (1999 to 2015 records; meteo.navarra.es), as the 97th percentile ERC-G fuel moisture content [51] and wildfire season dominant winds (Table 2). We generated five wind scenarios considering the most frequent wind directions (frequency >5% in weather records) during wildfire season and the respective 97th percentile wind speeds.

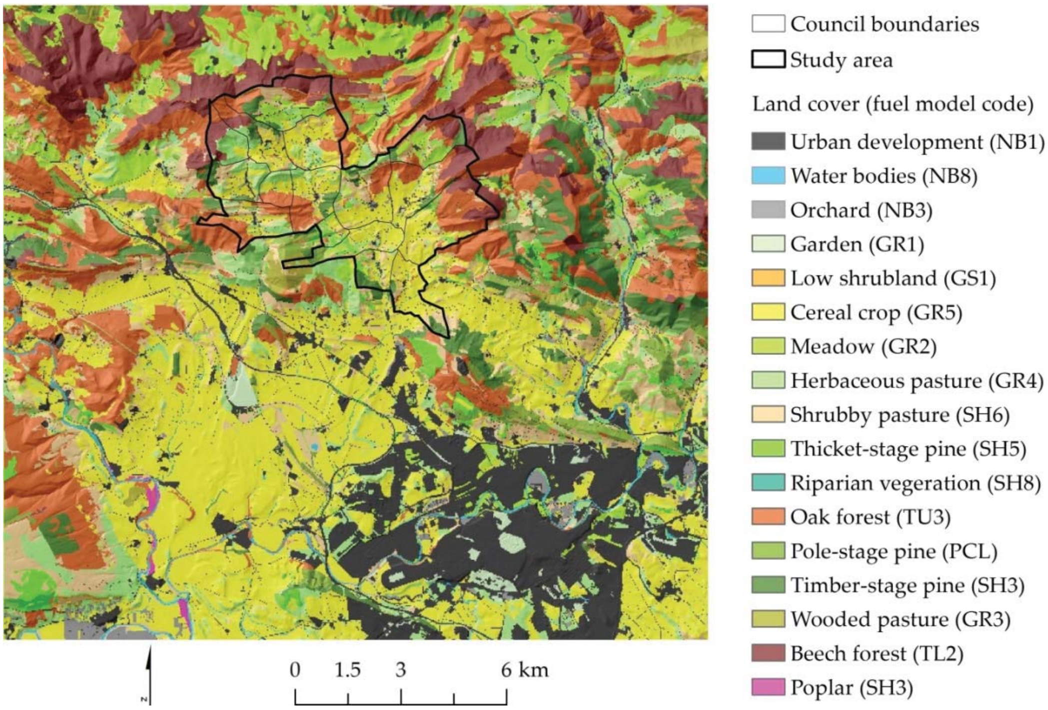

Figure 3. Land cover map (idena.navarra.es) and assigned fuel models [44,45] for the wildfire modeling. The large urban development areas in the southeast correspond to the capital city of Pamplona. Cereal crops occupy all the flat cultivated areas to the south and mountains in the northern part are covered by mosaics of different forest types. See fuel model parameter details in the references [44,45].

Table 2. Fire weather input data, corresponding to the historical 97th percentile conditions, used for wildfire simulations. We considered the most frequent wind directions (frequency >5%) during the last 17 wildfire seasons. Historical weather data were gathered from the meteorological station of Pamplona (meteo.navarra.es). We used standard fuel models for fire modeling, see references [44,45] for further details.

Wind Scenario			Fuel Moisture Content (%)			
				Fuel Model [44,45]		
Direction (°)	Speed ($km \cdot h^{-1}$)	Probability	Fuel Loading Category	GS1, GR5, GR2, GR4, SH6, SH5	TU3, PCL, SH3, GR3	GR1, SH3, TL2, SH8
67.5	32	0.43	1-h	4	6	8
337.5	35	0.28	10-h	5	7	9
45.0	19	0.17	100-h	8	9	12
180.0	31	0.06	Live herbaceous	20	45	70
22.5	23	0.06	Live woody	60	85	100

GS1 = low load, dry climate grass-shrub; GR2 = low load, dry climate grass; GR4 = moderate load, dry climate grass; GR5 = low load, humid climate grass; SH6 = low load, humid climate shrub; SH5 = high load, dry climate shrub; TU3 = moderate load, humid climate timber-grass shrub; PCL = closed and low litter pine stands; SH3 = moderate load, humid climate shrub; GR3 = low load, very coarse, humid climate grass; GR1 = short, sparse dry climate grass; SH3 = Moderate load, humid climate shrub; TL2 = low load broadleaf litter; SH8 = high load, humid climate shrub."

2.2.2. Fire Occurrence Modeling

We used artificial neural networks (ANNs) to construct a fire occurrence model, and ultimately to generate a 20-m resolution ignition probability grid encompassing the modeling domain. A 10,000-fire ignition point input file for wildfire simulation was then created from the ignition probability (IP) grid masked to burnable fuels. ANN models are robust pattern detectors which can approximate mathematical relationships with non-normal distributions and spatially correlated variables where other statistical models could cause multicollinearity [52,53], and have been successfully applied to fire occurrence prediction in previous work [26,54].

The historical fire ignitions within the fire modeling domain (200 ignitions in all, 1985 to 2013 fire records; mapama.gob.es) and the same number of random no-fire observations were matched to topography (elevation, aspect, slope), land cover class, population density, and accessibility (distance to roads, tracks, railways, urban areas and powerlines) 20-m resolution raster grids (ign.es; idena.navarra.es). Ten percent of the fire and no-fire observation variable dataset (40 cases) was set apart for validation purposes before model building. We selected feed-forward, multilayered, non-linear, fully connected, cascade-correlation networks [55], built using Neural Works Predict® v.3.30 software (NeuralWorks Predict®3.30, Serial Number NPSC30-70755, Carnegie, PA, USA) [56] with an adaptive gradient learning rule, a variant of the general algorithm of back-propagation [57], and a weight decay factor which inhibited complexity of the models [58]. The historic fire records of fire and no-fire observations for model building (90%) were further divided in two. One part was used for iterative training (70%, 252 cases) and the other part (30%, 108 cases) for early stopping, the periodic assessment of performance accuracy in order to avoid losing generalization capacity due to overtraining [59]. The cascade-correlation models followed a similar procedure to [60,61], in which the model architecture (number of nodes in the hidden layer) is optimized during training.

The best model found had an 8–6–1 (input–hidden–output) structure, and classification rates of 0.78–0.73–0.69 for training–test–validation datasets (Table 3). When selecting the best ANN classification model, we looked for the highest classification rate on observed and predicted fire/no-fire observations, balanced results between the three datasets and a parsimonious architecture. Variables in the model, by order of importance, were distance to forest tracks (three times input to the model), distance to urban areas (twice input to the model), distance to powerlines (twice input to the model), and population density (once input to the model). Finally, this best fire occurrence model was run at 20-m resolution pixel level to generate the ignition probability grid (IP; values ranging between 0 and 1; Figure 4).

Table 3. Classification table with the results for the best occurrence model. The model was generated with Neural Works Predict® v.3.30 software. This occurrence model was used to generate a 20-m resolution ignition probability grid (Figure 4). Geospatial variables associated with the historical fire ignitions (200 fire ignitions, 1985 to 2013 fire records; mapama.gob.es) (1) and a random sample with the same number of no-fire observations (0) were included within the fire modeling domain. A set of 40 cases (10%) was used for the validation of the model.

Classification Rate		Class	0	1	Total
Training	0.756	0	99	32	131
	0.802	1	24	97	121
	0.779	Total	123	129	252
Test	0.679	0	36	17	53
	0.782	1	12	43	55
	0.731	Total	48	60	108
Validation	0.765	0	13	4	17
	0.609	1	9	14	23
	0.687	Total	22	18	40

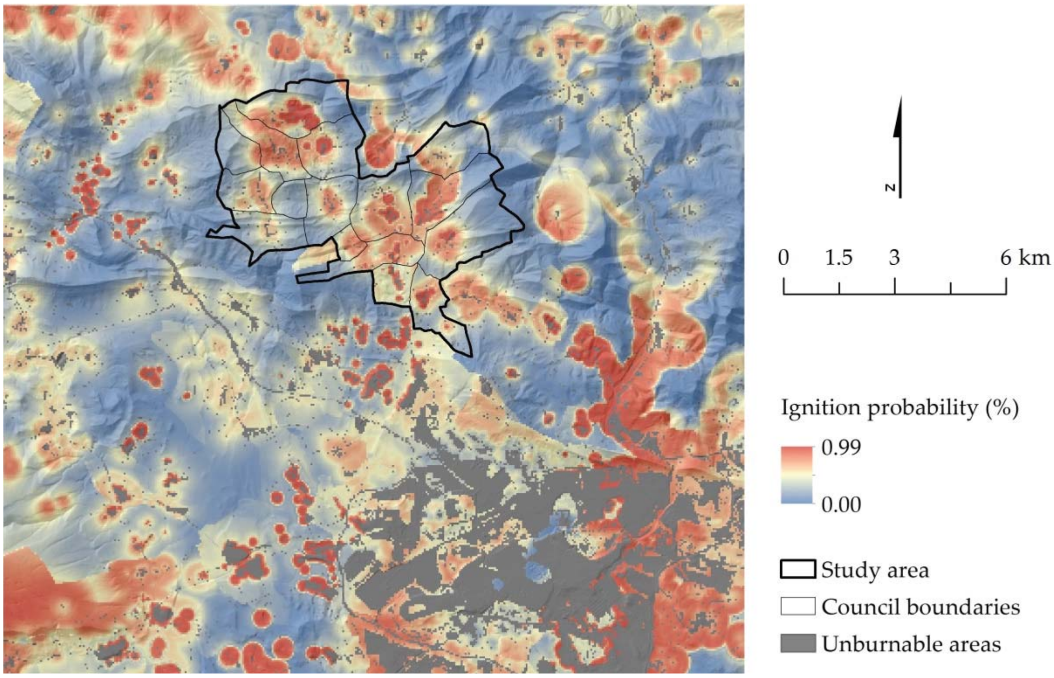

Figure 4. Ignition probability grid generated with an artificial neural network using the geospatial variables associated with the observed ignition data (1985 to 2013 historic fire records; mapama.gob.es). This fire occurrence grid was used to generate the 10,000 fire ignition input data masked to burnable fuels in the wildfire modeling domain. Unburnable areas ($IP = 0$) correspond to urban development, roads and water bodies.

2.2.3. Wildfire Spread and Behavior Simulation

We used FlamMap to simulate wildfires under conditions of constant fuel moisture, wind speed and wind direction [43]. We conducted five different weather scenarios at 20-m resolution, with 10,000 wildfires per scenario (Table 2). FlamMap uses the two-dimensional fire growth minimum travel time algorithm (MTT) [62], which has been widely used worldwide at a broad range of scales with multiple purposes [63–66]. The MTT algorithm replicates fire growth based on the Huygens' principle, where the growth and behavior of the fire edge is modeled as a vector or wavefront [62], and fire spread distance is predicted by the Rothermel's surface fire spread model [67]. Fire duration was set at 6 hour, in agreement with the active fire spread duration of the observed largest wildfire events in the study area (i.e., Juslapeña 2009). We did not consider barriers to fire spread or fire suppression efforts. Overall, modeled fires burned burnable pixels at least once and more than 100 times on average.

FlamMap outputs burn probability (BP) and flame length probability (FLP) grids, as well as a fire size (FS) text file and the fire perimeters (polygons). The burn probability (BP) is the number of times a pixel burns as a proportion of the total number of fires, and is defined as follows:

$$\mathrm{BP} = F/n \tag{1}$$

where F is the number of times a pixel burns and n is the number of simulated fires per run (n = 10,000 in this study). Specifically, the conditional burn probability in the study area is the BP given that a fire ignites within the fire modeling domain and spreads for 6 hours at assumed fuel moisture and weather conditions (97th percentile fire weather). Fire intensity [68] is first predicted by the MTT algorithm [62] and is converted into flame length as:

$$FL = 0.0775 \times I^{0.46} \tag{2}$$

where *FL* is flame length (m) and *I* fireline intensity ($kW \cdot m^{-1}$, kW = kilowatt). Then the program calculates a FLP regular point grid (at the fire simulation resolution) from the multiple burning fires at different flame lengths (i.e., backing, heading and flanking fire spread flame lengths). For every pixel in the FLP output, the probability of flame length is calculated at *i* categories of different fire intensity levels (FILs), given that at least one of the simulated fires has burned the pixel. In this study, FILs were obtained as twenty 0.5-m flame length categories (for FIL_1-FIL_{19} and FIL_{20} >9.5 m).

In the fire size (FS) text file output generated by FlamMap, the simulated burned area (ha) is attributed to each *xy* coordinate fire ignition. Moreover, we also obtained burned-area polygon shapefiles associated with each simulated fire and minimum travel time (MTT) major flow-paths polyline shapefiles for the five fire weather scenarios (Table 2). Travel pathways are straight lines that connect nodes and intersect cells to form segments for which fire behavior is calculated from the input data [43].

2.3. Expert Judgement of Structure Susceptibility

We used a response function (RF) to approximate structure susceptibility (potential losses) using fire intensity level model outputs [36]. To generate a customized RF for residential houses in the study area, we used the Delphi method [69]. The Delphi method is an iterative questionnaire process used to obtain a reliable consensus from a carefully selected expert panel, and it has been used in previous studies to determine wildfire causality from the personnel involved in fire suppression activities [70,71].

We conducted a face-to-face and anonymous two-round questionnaire process with the regional firefighting "Bomberos de Navarra" chiefs, focusing on the most experienced in WUI fire suppression in central Navarra. Fire intensity is the main causative factor of home loss given that a fire reaches a housing structure, and therefore in the questionnaire, potential value loss of structures (as a percentage) was associated to four different fire intensity class response functions (intensity levels of FIL_1, FIL_2-FIL_4, FIL_5-FIL_7, and FIL_8-FIL_{20}). The four fire intensity classes were selected considering previous studies and the capabilities of existing geospatial tools to integrate the fire modeling outputs with potential fire effects [36,49]. In the first round of the questionnaire process, the experts filled the questionnaire anonymously according to their own personal experience to reduce the effect of dominant individuals. Then in the second round, the questionnaire was repeated to the same experts, but included results from the first round (average values and deviation in the fire intensity classes) to meet a higher consensus and refine the final results. The obtained custom RF presented moderate to strong losses in housing structures as fire intensity increased (Table 4), similar to RFs obtained in other studies conducted in Mediterranean areas [35].

Table 4. Custom response function (RF) used to approximate fire effects in terms of value loss (%) on residential houses in the study area [42]. The fire modeling output fire intensity levels (FILs) were grouped into four classes for the geospatial risk assessment [49]. We used the Deplhi method to obtain the susceptibility function from an expert panel composed of the most experienced firefighter chiefs on wildland urban interface fire suppression [69]. The wildfire had negative impacts in structures at all fire intensities.

Valued asset	**Relative Net Value Change (%) at Different Fire Intensity Classes**			
	Low (FIL_1)	**Moderate (FIL_2–FIL_4)**	**High (FIL_5–FIL_7)**	**Very high (FIL_8–FIL_{20})**
	FL < 0.5 m	**0.5 m < FL < 2 m**	**2 m < FL <3.5 m**	**FL > 3.5 m**
Residential house	−10	−45	−75	−95

2.4. Residential House Economic Value

We used the official cadaster method described in the Navarra Foral Decree 334/2001 of November 26 to assess the economic value (V) of the individual housing structures in the study area. This Foral Decree approves the procedure for the economic assessment of immovable property in the Foral Community of Navarra throughout the implementation of the Comparison Method of the average market prices, with reference to Inheritance and Gift taxes, and over Property Transfer and Certified Legal Documents (text published in the Boletín Oficial de Navarra No. 155 of 21 December 2001, and the Boletín Oficial de Navarra No. 21 of 18 February 2002; lexnavarra.navarra.es). The method has been updated several times since its first publication, with the Foral Decree 39/2015 of 17 June being the last update. There are specific models to estimate the values for flats, single residential houses, and parking or storage rooms. We used the model for single houses, since most dwellings in the study area were well preserved rural houses or recently built constructions. The main parameters used by the model are the year of the information, type of individual house, location, cadastral category and conducted reforms, year of construction, constructed surface, and the ratio of constructed surface to urban development polygon surface. The residential houses with more than one cadastral sub-division (original building and dwelling expansion) were merged into a single unit. We used market prices from 2015 to obtain the most up-to-date values (Table 5).

Table 5. Summary table of the cadaster economic value (V) for the residential houses in the Juslapeña Valley. Council polygon cadaster codes No. 3 and No. 16 do not have residential houses (Figure 1A). The cadaster value was estimated for the year 2015, considering the model published in the Foral Decree 334/2001 of November 26 (lexnavarra.navarra.es).

Cadaster (Polygon No.)	Council Name	Residential Houses (No.)	Cadaster Economic Value (€)			
			Average	Median	Maximum	Minimum
1	Beorburu	11	108,825	109,767	151,432	52,124
2	Osacar	8	145,249	133,338	223,641	100,226
4	Osinaga	16	130,053	123,993	213,505	50,040
5	Aristregui	23	162,244	191,847	423,310	48,396
6	Larrayoz	17	156,733	138,493	269,403	77,480
7	Nuin	26	142,338	109,033	261,787	52,254
8	Marcalain	31	163,710	141,553	315,990	91,589
9	Iruzkun	1	132,192	132,192	132,192	132,192
10	Garciriain	12	139,194	143,079	199,405	74,682
11	Belzunce	48	168,233	135,984	483,043	64,657
12	Navaz	21	131,148	114,763	223,539	68,061
13	Ollacarizqueta	55	135,030	127,351	319,046	67,402
14	Unzu	16	154,063	174,413	218,129	88,067
15	Usi	21	111,784	108,407	167,685	71,229

2.5. Analysis

Wildfire simulation outputs were used to assess large fire initiation, transmission, exposure and risk to residential houses of rural communities within the study area (Figure 2). We combined five sets of fire simulation outputs (BP, FLP, FS, burned area perimeters and major flow paths), one for each scenario by weighting the relative scenario probability (Table 2).

2.5.1. Large Fire Initiation and Incoming Major Pathways

We estimated fire potential index (FPI) [31], and MTT major flow-paths to spatially analyze where large fires likely initiate and from which surrounding neighboring municipalities do these fires spread to reach the target residential houses. We calculated fire potential index (FPI) as:

$$\text{FPI} = FS \times IP \tag{3}$$

where the *FS* is the spatially smoothed fire size grid, and *IP* is the historical-based ignition probability grid generated with the ANN fire occurrence model (Figure 4) used to generate the fire ignition input file. We used a kriging geostatistical analysis method to generate a continuous distribution grid of FS from fire size data contained in the ignition location output point file. MTT flow-paths within surrounding municipality polygons (Figure 1A) were then overlaid and classified in three frequency classes (<33%, 33%–66% and >66%), considering the simulation scenario probability (Table 2), to identify preferential pathways entering to the Juslapeña Valley.

2.5.2. Transmission Analysis

We analyzed how incoming fires are shared among surrounding municipalities (Figure 1A) and mapped the potential impact of each independent fire on dwellings with transmission analysis. We only considered large fires (>100 ha) because small fires do not substantially contribute to total burned area. In the study area observed, large fires (>100 ha) burned about 95% of the total area (1985 to 2012 historic fire records). We quantified (*i*) the number and (*ii*) the economic value of residential houses within fire perimeters. Fire transmission in terms of the number of structures was quantified as:

$$TFS_{ij} = \sum S_j \tag{4}$$

where *TFS* measures the number of individual *S* affected structures in the *j*th municipality (study area) given a large fire (>100 ha) ignited in the *i*th surrounding municipality (Figure 1A) spreading under 97th percentile fire weather conditions for 6 hours. Correspondingly, the cadastral value of all affected structures contained inside the burned area from transmitted fires was quantified as:

$$TFV_{ij} = \sum V_j \tag{5}$$

where *TFV* measures the cadaster structure value sum of all houses affected and located in *j*, given a fire arriving from the *i*th polygon, and *V* is the individual structure cadaster value (€). *TFV* is not the expected economic loss of affected structures, but the value of all affected structures within the burned area polygons. We considered *j* as the municipality polygon corresponding to the study area (i.e., Juslapeña Valley) containing all the target residential houses, and *i* as the surrounding municipality polygons (Figure 1A). In total, we analyzed the transmission of 12,515 fires larger than 100 ha ignited from the six different municipality polygons surrounding the study area. Since we focused our analysis only on fires incoming from surrounding polygons, self-burning was not considered (i.e., $i \neq j$).

The TFS transmission results (i.e., the 10,000-fire ignition point file attributed with the number of structures intersected in the fire perimeter polygons) for the five different fire modeling simulation scenarios (Table 2) were separately spatialized into fireshed continuous grids using a 1000-m fixed radius and spherical semivariogram model kriging analysis statistical method. The area estimated within a fireshed is conditional on assumed fire weather and hence we estimated firesheds for each of the scenarios. We also developed contour plots using six different transmission levels to map the internal transmission gradient (0–50, 50–100, 100–150, 150–200, 200–250 and >250 structures).

2.5.3. Exposure Analysis

We analyzed individual residential house wildfire exposure in the home ignition zone (HIZ) (Figure 1C). The HIZ is the 60-m buffer immediately surrounding residential houses that determines structure ignition potential during extreme wildfire events [10]. Fire likelihood and intensity modeling outputs were considered as key causative wildfire risk factors for this analysis. Structure exposure assessment does not account for the fire effects. The geospatial location (polygons) for the individual residential house structures (n = 306) was obtained from cadaster shapefiles (1:5000 scale) of the Regional Government (catastro.navarra.es; Figure 1C).

Wildfire likelihood was estimated as conditional burn probability, and fire intensity as the conditional flame length. We used the pixel-level FIL distribution to calculate the conditional flame length (CFL) as:

$$CFL = \sum_{i=l}^{20} FLP_{\mathrm{i}} \times FL_{\mathrm{i}} \tag{6}$$

where FLP_i is the flame length probability of a fire at the ith flame length category, and FL_i is the flame length (m) midpoint of the ith category FIL. The CFL is the probability-weighted FL assigned to a fire, and is a measure of wildfire hazard [35]. We assessed exposure at individual residential houses from the average values (BP and CFL) within the HIZ.

2.5.4. Risk Analysis

We quantified the expected losses to individual residential houses combining wildfire likelihood and intensity modeling outcomes with expert judgement elicitation response functions [28]. RFs were used to approximate fire effects (losses) to different fire intensity classes. Then, fire effects and respective burning probabilities were considered to estimate the expected net value change [36]. Expected net value change is a risk-neutral measure in terms of *gain* or *loss* expressed on a percentage basis, and allows quantitative wildfire risk assessment for multiple valued resources and human assets [33]. In order to consider the variations between economic values of different houses and quantify economic losses at the individual structure level, we used the latest cadaster reference of economic values (V_i).

The probabilistic expectation of loss (eNVC) was estimated by combining the customized response function with fire intensity and conditional burning probabilities [28] at the pixel-level on the HIZ:

$$eNVC = \sum_{i=1}^{20} BP \times FLP_i \times RF_i \tag{7}$$

where $eNVC$ is the expected net value change neutral base measure in terms of gain or loss (%) [36], BP is the conditional burn probability, FLP_i is the flame length probability of the ith category FIL, and RF_i is the response function at the ith FIL (Table 4). We assigned the average value within the HIZ to the individual dwellings.

Losses at the individual structure level were monetized using the cadaster value as:

$$eEL_x = eNVC_x \times V_x \tag{8}$$

where eEL is the expected economic loss in the xth residential house (€) given that a fire ignites within the wildfire modeling domain and spreads under extreme fire weather conditions, $eNVC_x$ is the average expected net value change in the xth residential house HIZ, and V_x is the latest cadastral reference value of the xth residential house (€; catastro.navarra.es). Previously, $eNVC$ negative values (fires always produced losses) were transformed into a positive fraction of unity value (e.g., −5% to −0.05).

3. Results

3.1. Large Fire Initiation and Major Pathways

The source location of large fires as quantified by FPI was concentrated around the southwestern and central part of the northern councils (Figure 5). Fires ignited in the south of the study area resulted in larger fire size, and therefore higher FPI values than in the northern and eastern areas. In the northwestern and eastern forested remote areas, the fire ignition probabilities were very low and consequently FPI values were the lowest compared to other areas. Incoming fires exhibited two main paths, either from the northern central part or the southeastern open valleys (Figures 3 and 5). These results highlighted the effect of topography and fuel models in the major flow-paths, especially in the mountainous northern areas of the study area. Fires in the even-aged mature beech forests were largely impenetrable on the northern border within the municipality 126, and most incoming flow-paths were routed through municipality 40, where heading fire spread from the different scenarios' pathways coincided frequently (>66%). On the other hand, in the more fire-prone unmanaged oak and black pine stands, fires arrived from south facing slopes in some cases (e.g., 180° flow-paths). Overall, herbaceous type fuel models located in lowland valley bottom flat areas facilitated the spread of fire and were the preferential fire spread pathways into the study area.

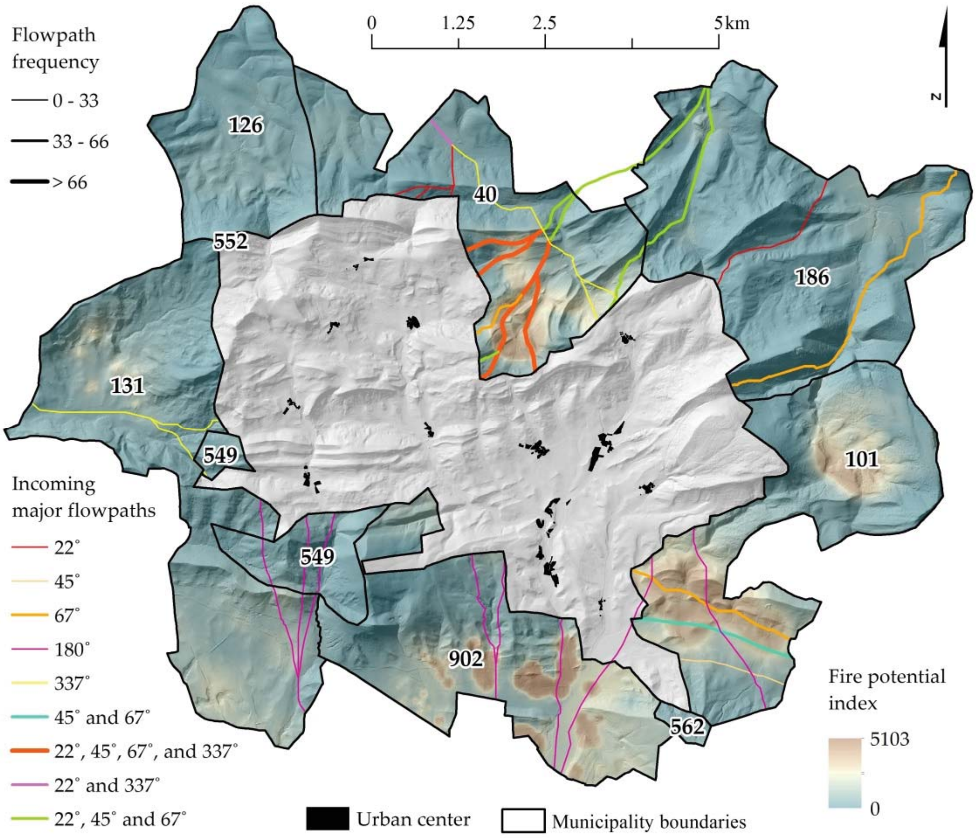

Figure 5. Fire potential index (FPI) and incoming major flow-paths from the surrounding municipalities. FPI was calculated by combining the fire size and ignition probability output grids, and was used to identify the areas where the ignition of a large fire is more likely [31]. Major flow-paths were obtained with the minimum travel time algorithm (MTT) [62] considering the five most recurrent fire weather scenarios (Table 2). The flow-path thickness indicates frequency and color indicates fire scenarios.

3.2. Transmission Analysis

Fires threatening the highest number of residential houses initiated in the southeast 101 and northern 40 municipalities, affecting on average 71 and 80 structures respectively (Figure 6A). The maximum number of structures affected was 188 from a fire ignited in municipality 40. Fires from municipalities 126 and 131 showed limited transmission capability with six or fewer structures burning on average. Although fires in eastern municipality 186 burned on average 15 structures, a few fires (2%) burned more than 100 structures. Due to the limited variability in cadaster economic values of structures within the study area (Table 5), both transmission boxplots depicted similar distributions (Figure 6A,B). Thus, both transmission metrics (TFS and TFV) provide equivalent results in the study area. Given the same response function for all structures (Table 4), economic losses of residential structures from large fires (>100 ha) ignited in surrounding municipalities are highly dependent on HIZ fire intensity and number of structures.

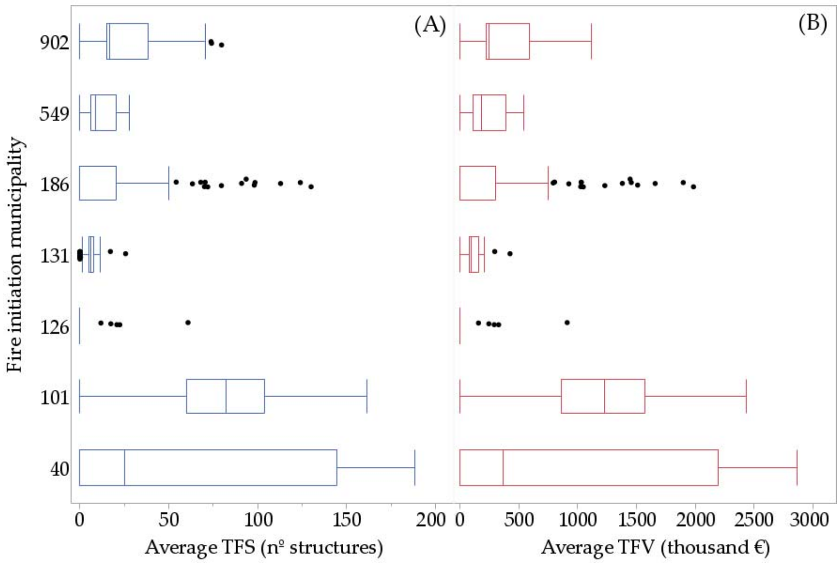

Figure 6. Box plots of average wildfire transmission into the study area from independent ignitions in surrounding municipalities (Figure 1A), in terms of (**A**) number of residential houses and (**B**) cadaster economic value of residential houses affected. For every fire ignition, the number of affected structures and the sum of their economic value was calculated combining the results obtained in the five modeling scenarios (Table 2). Boxes indicate the first/third quartiles, the whiskers indicate 10th/90th percentiles, the black line within the box is the median, and the dots indicate values below the 10th percentile or above the 90th percentile. The municipalities are identified with the cadaster code (Figure 5).

We found a wide variation in predicted community fireshed area for the different scenarios used in the fire simulation (Figure 7A–E). The southern wind direction scenario presented the largest firesheds and smooth gradients, expanding southwards more than 5 km from the study area boundary for the highest >250 structure transmission class. Fires arriving from the south burned through dryland cereal crops and represented the most extreme threat fire scenario to the residential houses in the study area (180°). Firesheds for northwestern to northeastern component wind directions presented the sharpest transitions gradients between transmission classes (337° and 67°). North facing timber litter fuel model beech stands on wind direction perpendicular orientations delineate the fireshed boundaries in northeastern and northwestern wind directions (Figures 3 and 7A,D,E). Highest TFS and TFV values were obtained for fires ignited inside the study area in the majority of cases. Fireshed extension in the north was limited to valley bottom herbaceous fuels on the central part for the scenarios that present similar wind direction and mountain ridge orientation (22° to 67°). Fireshed delineation results agree with the major flow-path results, and overall on larger areas over flow-path influence areas.

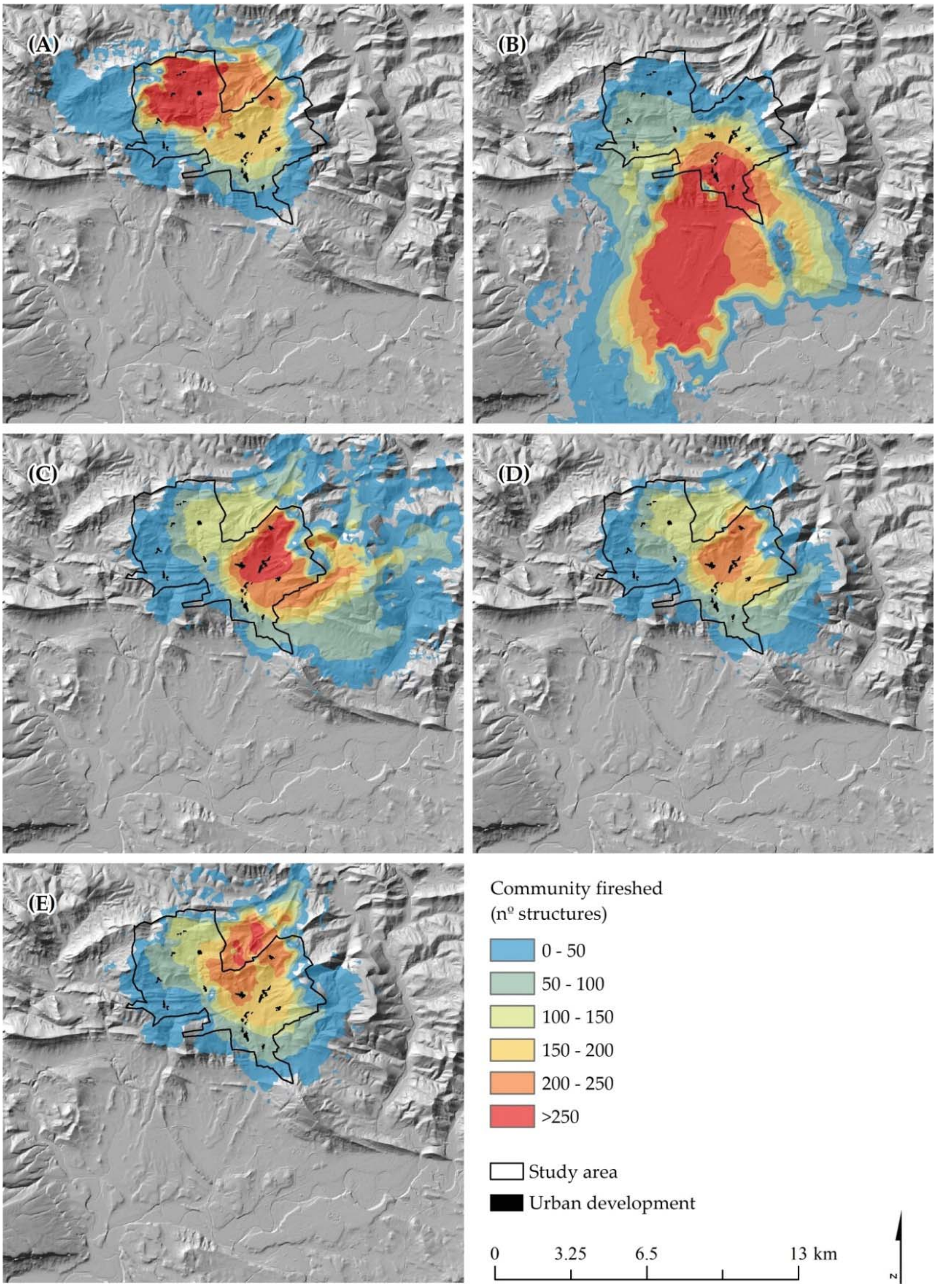

Figure 7. Community fireshed maps corresponding to the number of residential houses burned for the five wildfire scenarios. The letters from **A** to **E** indicate respectively the fire modeling wind direction scenarios of 337°, 180°, 67°, 45° and 22° (Table 2). Fireshed values were generated using a 1-km constant width radius spherical semivariogram model kriging analysis from the transmission values (TFS) assigned to fire ignition locations. Values indicate the number of structures affected by ignitions in a given pixel. Fires were simulated for 97th percentile fire weather conditions and 6-hour duration.

3.3. Exposure Analysis

The burn probability and conditional flame length wildfire modeling outputs showed complex spatial patterns in the study area (Figure 8A,B). As expected, the results highlighted important differences between the fire occurrence IP grid (Figure 4) and conditional burn probability in structure HIZ, since fire occurrence is closely associated with anthropic ignition sources but not necessarily burn probability (Figure 9). While the average IP is usually high on the HIZ (IP >0.8), the BP presents a wide range of values between 0.001 and 0.120 (Figure 8). Southern councils presented the highest ignition probability and burn probability values (e.g., councils No. 13 and No. 14; Figure 1A). Burn probability was higher than 0.1 in most southern areas, ten times higher than values in the northern part of the study area (BP < 0.001; Figure 8A). Highest values were associated in most cases to fast spreading surface fires in herbaceous type fuel models, such as rangelands and cereal crops (the Pamplona Basin northern rim extensive dryland agricultural landscape continuum) that dominate the valley bottom in the southern plain of the study area. On the other hand, the lowest values of the northern mountainous areas corresponded to beech and pine forests on north aspects, both characterized by low biomass understories. The smooth spatial gradients in burn probability were in contrast to the conditional flame length (CFL) (Figure 8B), where CFL highest values did not correspond with high burn probability (Figures 8B and 10A). Low CFL values (<1 m) were obtained in northern areas where the burn probability was the lowest, especially in the low fuel load, timber litter and closed canopy mature forest stands. Mosaics of fuel types, together with wind direction and slope, were the main drivers of fire intensity. High shrubs and dense forests on slopes aligned with the dominant winds (68° and 338° azimuth) showed the highest intensities (CFL > 6 m) .

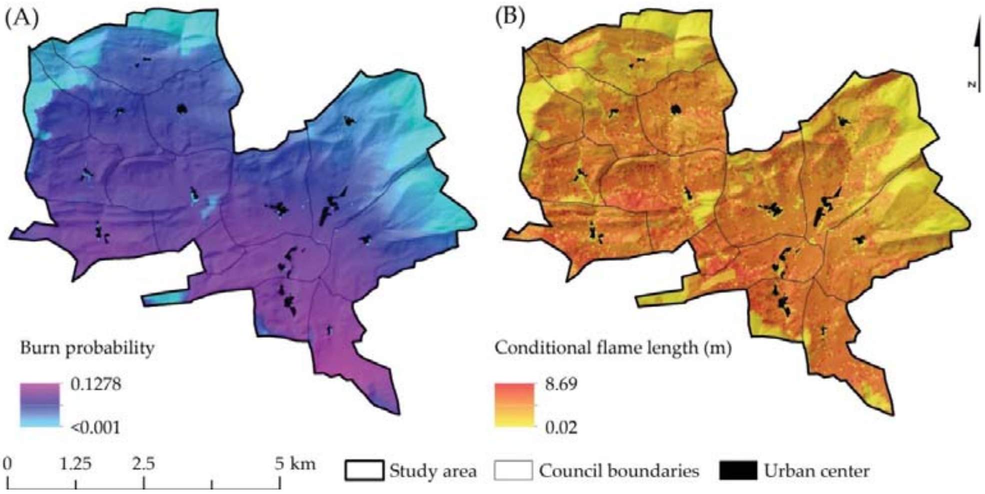

Figure 8. Conditional burn probability (**A**) and conditional flame length (**B**) output maps for the study area. Fires were modeled at 20-m resolution under 97th percentile fire weather conditions. The urban centers containing the bulk of residential structures are indicated with black polygons.

Average burn probability and conditional flame length for pixels within the 60-m circular buffer around individual residential houses varied widely among and within the different councils (Figure 10A,B). Overall, the bulk of houses had average conditional flame length values between 1.5 and 3 m, while the burn probability varied more widely, and was mostly concentrated between 0.4 and 0.11. Around some residential houses located in the central parts of the urban centers, where fuels consisted of managed gardens and orchards, the conditional flame length was the lowest (<0.25 m). Burn probability results showed much wider variations, especially between houses of different councils (Figure 10B). For instance, the residential houses in council No. 15 (located in the northeast, Figure 1A) presented on average four to five times lower burn probability (BP~0.02) compared to the most meridional council No. 13 (BP~0.10). Within the same urban center, residential houses exhibited variations among southern and northern locations, especially in the central parts of the study area (e.g., councils No. 5, No. 6 and No. 8), mainly because upslope spreading fires over cereal crops on the southern sides of urban centers present the fastest spread rates. Therefore, housing aggregation into compact urban centers and the relative structure position in the urban center had a strong effect on HIZ wildfire likelihood. In other words, wildfires were more likely to arrive and impact the southern side, and structures located there were exposed to higher BP. The highest overall exposure was experienced by residential houses nestled within forested and shrubby unmanaged areas with high fuel accumulation.

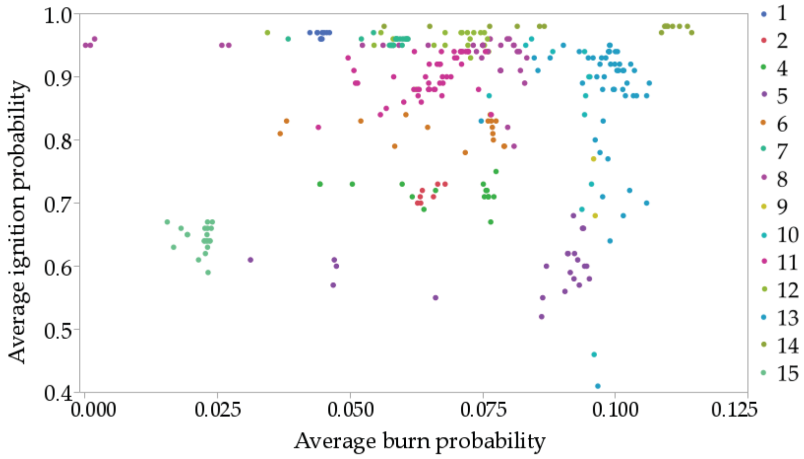

Figure 9. Scatter plot of ignition probability (IP) versus conditional burn probability (BP) for individual residential houses (n = 306 structures). Each dot is related to a different residential house, and values correspond to the mean value in the HIZ [11]. The bubble color indicates the council cadaster polygon (Figure 1A). While BP values showed a wide distribution, most IP values were concentrated above 0.85. Overall, results tended to present clustered aggregations with respect to the council.

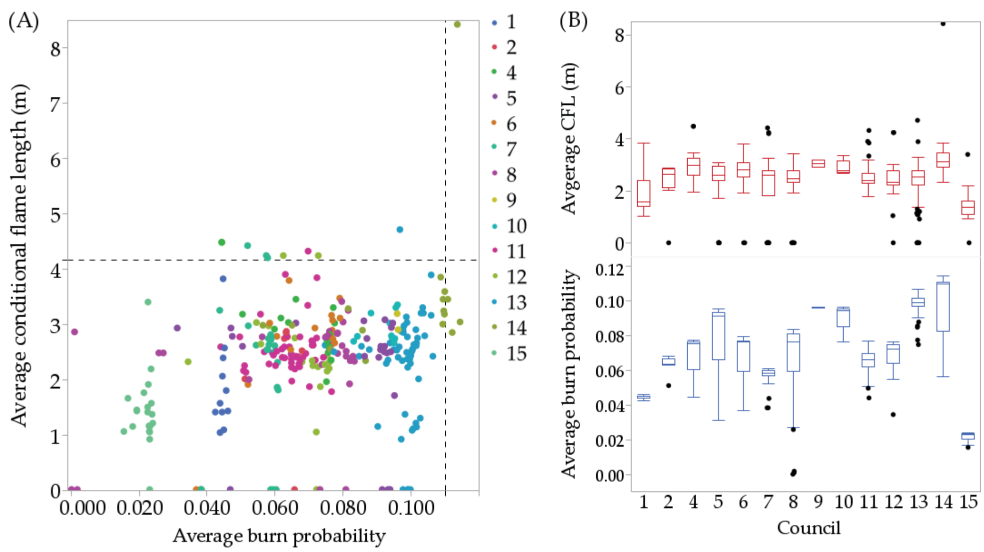

Figure 10. Individual residential house scatter (**A**) and box plots (**B**) for the different councils in the study area. Each point in the scatterplot indicates the average value of burn probability (BP) and conditional flame length (CFL) within the home ignition zone for a single structure. The bubble color indicates the council (Figure 1A), and the dotted lines the 97th percentile values of 0.11 for BP and 4.16 m for CFL. In the box plots, the boxes indicate the first/third quartiles, the whiskers indicate 10th/90th percentiles, the horizontal line within the box is the median, and the black dots indicate values below the 10th percentile or above the 90th percentile.

3.4. Expected Economic Loss

Expected economic loss for individual dwellings (eEL) ranged from a low of 740 to a high of 28,725€ within the study area (mean = 7955€), and also varied widely among the different councils (Figure 11, Table 6). The highest average values were obtained for the southern council No. 14 with 13,323€, followed by councils No. 5 and No. 10 with 12,976€ and 9715€ respectively. On the other hand, the lowest average eEL values were obtained in the low wildfire exposure northern councils No. 1 and No. 15, with 1429€ and 2803€ respectively. Overall, results depicted higher expected economic loss (eEL) for residential houses presenting lower expected net value change (eNVC), that ranged from −1.04% to −11.04%, with an average value of −5.23%. Except for a few cases, most residential houses have cadastral values between 110 and 180 thousand euros (Table 5), and therefore exposure metrics required for risk assessment translated similar patterns into risk outcomes (i.e., higher losses for higher overall exposure). Nonetheless, when the cadastral value varied substantially for the same eNVC (e.g., more than three times), wide differences were observed in terms of eEL. In those cases, the residential house cadastral value influenced the eEL result more than the eNVC (Figure 11).

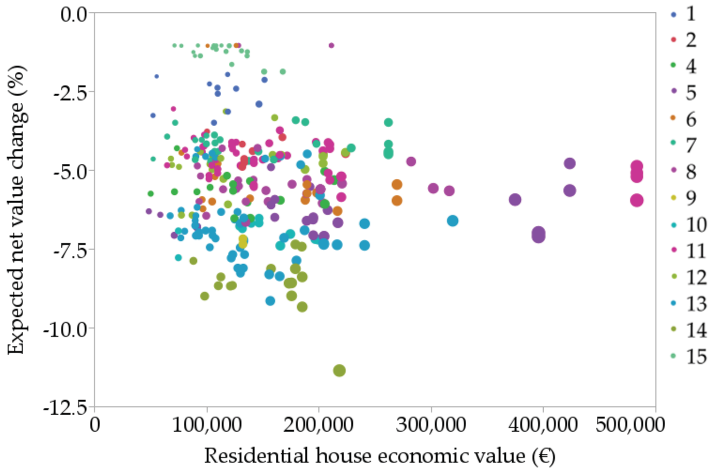

Figure 11. Wildfire risk bubble plot of residential houses in the study area. The expected net value change (eNVC) is the percentage variation with respect to the price of the residential houses. The bubble size indicates the expected economic loss (eEL), which ranged from a low of 3622€ to a high of 28,086€. The color indicates the council.

Table 6. Council level summary table with expected economic loss (eEL) results for the residential houses in the Juslapeña Valley (Figure 1A). Potential expected economic loss was obtained as a result of the implementation of the framework presented in this study (Figure 2). Council polygon cadaster codes No. 3 and No. 16 do not have residential houses (Figure 1A).

Cadaster (Polygon No.)	Council Name	Residential Houses (No.)	Expected Economic Loss (€)			
			Average	Median	Maximum	Minimum
1	Beorburu	11	2803	2826	4248	1120
2	Osacar	8	6378	6263	10,004	3776
4	Osinaga	16	7343	7188	12,459	2870
5	Aristregui	23	12,976	12,480	28,086	3051
6	Larrayoz	17	8347	8152	16,049	1062
7	Nuin	26	5930	4855	11,757	2425
8	Marcalain	31	8167	7502	17,855	1334
9	Iruzkun	1	9592	9592	9707	9478
10	Garciriain	12	9715	9443	14,366	5169
11	Belzunce	48	8044	6232	28,725	2143
12	Navaz	21	6157	5341	9909	3150
13	Ollacarizqueta	55	9105	8219	21,046	3960
14	Unzu	16	13,323	13,419	24,757	6933
15	Usi	21	1429	1222	3131	740

The map of eEL for dwellings in the study area showed how spatial location greatly influenced the result (Figure 12A). Highest economic losses (>10,000€) were located in the southern councils (i.e., No. 14, No. 5 and No. 13) and some individual houses in councils of the central part, while the lowest values (<1000€) were concentrated in the northeastern and northern councils (i.e., No. 1 and No. 15). The highest variation within residential houses of the same council were seen when the urban center tended to present a more scattered linear orientation following the communication corridors (e.g., council No. 11), and greater distances between the most distant houses (>1 km). Spatial patterns in eEL were similar to the gradient observed for the burn probability (Figure 8A), since fire hazard among residential houses (Figure 10A,B) within the study area did not show large differences. Thus, according to the results at individual residential structure level, eEL treatment priorities in the HIZ would be preferentially located in southern councils and structures occluded in hazardous forest fuels.

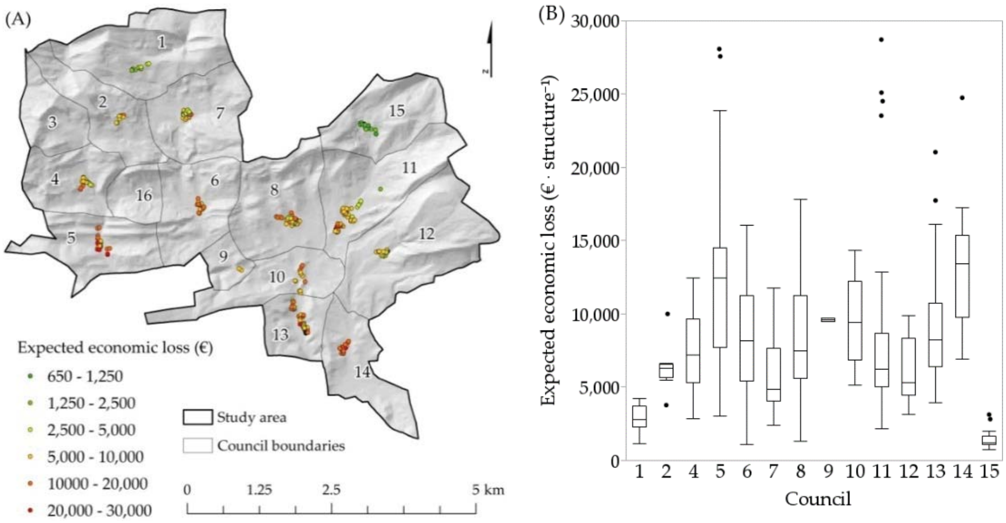

Figure 12. Map (**A**) and box-plots (**B**) of expected economic losses (eEL) for residential houses given that a fire occurs within the fire modeling domain under extreme fire weather conditions. Councils No. 3 and No. 16 do not have residential houses. In the map, every dot corresponds to a single residential house. In the boxplots, boxes indicate the first/third quartiles, the whiskers indicate 10th/90th percentiles, the black line within the box is the median, and the dots indicate values below the 10th percentile or above the 90th percentile (€ structure^{-1}).

4. Discussion

The integration of biophysical fire modeling with susceptibility relationships derived from expert judgement provides a method to calculate expected financial loss to communities from potential wildfire events. Our analysis also demonstrated the tracking of burned areas in the communities to ignition locations, thus providing a linkage between wildland fuels and risk to communities. The results provided useful insights that can inform ignition prevention fuel management programs for reducing risk to communities [72]. Transmission analysis allows the identification of sources of risk in terms of specific landowners within the study area [39,40]. The fireshed mapping defined the scale of risk to rural communities [73] and delineated the area within which fuel treatments could be prioritized to reduce large-fire impacts [37]. Coupling the fireshed with maps of exposure provides a wealth of information to inform the prioritization of wildfire management within the study area [35]. Our fire risk quantitative assessment results showed a very strong structure-level spatial gradient in economic loss within and among the 14 councils in the Juslapeña Valley study area, and provided findings that are potentially useful for insurance companies and local landscape managers.

We identified high probability paths of incoming fire in central north and south east valley bottom flat areas [62,65], which were mainly located in neighboring municipalities 101 and 40. These two areas accounted for the bulk of the transmission as measured by the highest number of residential structures affected. Historically, fires frequently have impacted populated areas after spreading large distances from their ignition location, well beyond community wildfire planning boundaries, underscoring the importance of analyzing firesheds to minimize scale mismatches [41] between the landscape planning and fire risk mitigation efforts [73]. In fact, 67% of large fires (>100 ha) ignited in the surrounding municipalities reached target communities, and each of these fires affected 56 structures on average. Also, we found that the 180° wind direction fire weather scenario in particular resulted in fires spreading the longest distance from ignitions outside the study area administrative boundary (>8 km). Thus, collaborative planning efforts need to involve neighboring administrations and landowners (Figure 7), and the significance of current land management in areas outside of target councils needs to be recognized for its potential to enhance wildfire risk. These practices include grazing, firewood collection in coppice oak forests and thinning in dense conifer plantations.

We contribute several new methods for exposure assessments within the Mediterranean region [47,72]. In particular, we used an ANN fire occurrence model to generate fire ignition input locations, and included an expert-defined response function for structure-scale assessment of potential economic losses. Although wildfire loss or benefit quantification is not possible for many socioeconomic values, a number of important services derived from forests can be represented with market pricing [48]. Specifically, 72% of structures have estimated values ranging from 100 to 250 thousand euros, with relatively few (6%) having values >250 thousand euros. Rather than economic value, we found that spatial patterns of wildfire likelihood were the major causative risk factor, and thus fire occurrence spatiotemporal patterns in Mediterranean environments are especially important for fire prevention. ANN performed well and facilitated the generation of a high-resolution ignition probability grid. Understanding how fire weather and geospatial variables associated with anthropic activities can explain fire occurrence has been conducted in previous works [74,75].

This study highlighted the importance of fire spread modeling for risk assessment in Mediterranean environments where large fires spread through mosaics of fuel type and administrative jurisdiction [32,76,77]. Urban interface classification based on housing density has been considered a key factor in structure loss and risk mitigation in some previous studies [19,78]. Indeed, scattered and occluded houses within wildlands usually present higher exposure levels from catastrophic fire events than densely populated urban development areas [79]. However, structures at the periphery of communities usually incur higher losses since they intercept heading fires and associated embers showers (Figure 13). Although most houses in the study area are built with fire-resistant designs and materials, and have cultivated orchards in the surroundings, exposure to ember showers makes them vulnerable to fire. Isolated dwellings in remote areas are hampered by poor access for ground-based suppression crews, a primary factor contributing to structure loss probability and human fatalities [34]. Urban areas with fire-resistant structures and managed fuels in the HIZ can facilitate fire suppression opportunity, and help relocate residents to save zones during catastrophic fires events. The more typical situation is where developed areas become a resource sink for most of the firefighting resources, creating the potential for entrapment and accidents during mass evacuation during extreme fire events [80].

Figure 13. Close-up view of a residential house level wildfire risk map (eEL) for Marcalain (council cadaster polygon No. 8) expanded from Figure 12, over the June-2015 aerial photograph (idena.navarra.es). Other structures such as farm stores, churches and water deposits were excluded from the analysis. Overall, structures located in the periphery were more exposed to wildfires and presented higher potential losses. On the north side of rural communities, closer to shrublands and forest areas, higher wildfire hazard can enhance potential losses.

Multiple management implications result from this study. First, the results provided useful insights to identify preferential areas for future urban development (e.g., high overall exposure area exclusion criteria) and to inform fire-resistant building design and material requirements. Integrating exposure from other natural hazards such as floods in river basin plains and rock falls or avalanches in mountainous areas is widely accepted as criteria for potential urban development, but fire risk is not accounted in most fire prone areas where catastrophic fires are frequent events (<30 years). In this regard, many southern EU governments concerned with WUI problems are now dictating specific public policies and municipal ordinances to promote community and homeowner involvement in hazardous fuel management. We present structure level risk assessment results that can contribute to risk reduction efforts by identifying where fuel treatment provides the highest benefits at the individual house level. Urban planning and fire managers have limited budgets to cover risk mitigation over thousands of scattered housing communities dispersed throughout fire prone landscapes, and quantitative risk assessment frameworks [28,33,66] can help prioritize planning and investments as well help design specific spatial strategies [81,82].

Reducing structure susceptibility to fire [34] in combination with fuel treatments, both in HIZ [83] and strategically located areas on the landscape [10,35], are the key to mitigating wildfire risk to communities. Fuel treatments reduce potential fire intensity and spread rates by reducing surface and canopy fuel loadings and include a wide range of activities (prescribed burns, low pruning and low fuel load hedges, disrupting tree crown continuity and removing combustible material adjacent to structures) [12,84,85]. Other measures such as the implementation of structure self-protection plans can alleviate extreme fire environments and improve suppression capabilities (e.g., water sprinklers and cannons). Apart from typical treatments in forest fuels and reducing structure susceptibility, other strategies that focus on reducing fire spread over herbaceous land cover could reduce the impacts of long-distance spreading fire events. For instance, we observed long-distance fire events originating in dryland croplands in the southern portion of the study area. By managing herbaceous fuels with extensive grazing in fenced pasture common lands [86,87], and using grass species with patchy growth habit on dryland hay meadows, wildfire spread and intensity could be reduced in these areas. However, implementation of supervised grazing after cereal harvesting that is needed to break fuel beds on the edges between mosaics of cultivated lands is nowadays complicated to implement [88]. Currently, the major risk mitigation effort in agricultural areas is the prevention of ignitions during cereal harvesting operations from equipment, and increasing capabilities for more rapid response to ignitions if they do happen [89].

We assume various sources of error in the models and input data, and results should be viewed as a local approximation of wildfire risk to residential houses in Juslapeña Valley given a large fire event in the study area. Modeling outcomes are conditioned to a specific configuration of extreme fire weather conditions, fuels, topography and the rural–urban interface spatial distribution of the study area. Although fuel models around structures did not differ much from the dominant types in the study area, elsewhere complex interface areas with trimming hedges among structures (e.g., cypress *Cupressus sempervirens* L. hedges) might require a more detailed fuel characterization or various different response functions depending on secondary variables in addition to fire intensity. Community firesheds should be interpreted as a dynamic boundary that changes with assumptions about fire weather, and with existing spatial patterns of fuels as influenced by land management practices. The latter includes forest management practices, grazing practices and agricultural production. Moreover, structure loss is a complex process [12], and is difficult to model at the landscape scale [35]. As in other previous studies, we adopted an expert-defined response function to approximate fire effects at different fire intensities while acknowledging the margin of error [36]. We also did not consider the potential effects of fire suppression that could affect our estimates of structure ignition, especially for low intensity fires with flame lengths <1.2 m [90]. We also understand that structure economic value (conditioned to market changes) might not always be the best way to quantify real risk, due to the lack of correlation among the economic value and the social impact of structure loss on inhabitants. Focusing exclusively on economic criteria when setting treatment priorities might bias results to favor protection of the wealthy neighborhoods at the expense of lower priced homes, although in our study the value of homes did not substantially influence the results.

Further research is needed to better understand not only large fire transmission into the study area, but also the dominant transmission patterns at wider scales (e.g., regional and national), to understand how the study area is integrated into larger scale fire transmission patterns [40]. Understanding major large fire movements would provide a wider perspective to identify the nodes or high priority areas in the landscape requiring investments in treatments. Identification of treatment polygons or stands in priority areas (or firesheds) can be facilitated with optimization models and trade-off analysis to maximize the reduction in risk to multiple values of interest, including structure loss, game species habitat improvement or conifer timber production [91]. The risk assessment in this study should be considered as a preliminary step for mitigation and it does not necessarily reveal the optimal treatment allocation, especially considering that treating fuels at locations far from the urban interface can substantially slow large fire arrival [35]. Analyzing multi-objective treatment strategies in rural–urban intermix fire-prone Mediterranean EU landscapes is challenging, although newer landscape planning tools that allow for integration of fire transmission have opened a wide range of new analytical approaches to analyze trade-offs between local hazard versus large-scale transmitted fire [81].

5. Conclusions

We implemented a fine scale wildfire risk assessment and transmission framework in rural communities of central Navarra (Northern Spain). Potential economic losses were quantified on individual residential houses considering exposure results [42], local expert-defined susceptibility functions, and dwellings cadaster economic values. With the transmission analysis we traced the origin and quantified the potential impacts of large wildfires [40]. Using major flow-paths [62] we identified preferential fire spreading path-ways entering to the study area. We demonstrate that wildfires ignited in neighboring municipalities far beyond human communities can cause substantial economic losses. This work increases the awareness and knowledge on wildfire risk assessment in Southern European fire-prone areas, and highlights the need of a collaborative planning and management among neighboring communities, different landowners and landscape managers to mitigate losses from wildfires.

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Author Contributions: F.J.A. was the leading author, responsible for the study design and GIS analysis; M.S. contributed in the study design and wrote risk assessment section; A.A.A. contributed in in the study design and wrote sections of the manuscript; R.C. developed the custom susceptibility function and contributed in the discussion of the results; C.V.G. generated the ANN model, contributed in the study design and wrote sections of the manuscript.

Abbreviations

meteo.navarra.es	Meteorología y climatología de Navarra. Nafarroako Gobernua/Gobierno de Navarra
mapama.gob.es	Ministerio de Agricultura y Pesca, Alimentación y Medio Ambiente. Gobierno de España.
sigpac.navarra.es	Sistema de Información Geográfica de Navarra para la Política Agraria Comunitaria. Nafarroako Gobernua/Gobierno de Navarra.
catastro.navarra.es	Servicio de la Riqueza Territorial. Nafarroako Gobernua/Gobierno de Navarra.
ign.es	Instituto Geográfico Nacional. Centro Nacional de Información Geográfica. Ministerio de Fomento. Gobierno de España.
idena.navarra.es	Infraestructura de Datos Espaciales de Navarra. Portal de acceso a la información geográfica de Navarra. Nafarroako Gobernua/Gobierno de Navarra.
lexnavarra.navarra.es	Derecho navarro. Nafarroako Gobernua/Gobierno de Navarra

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Optimizing prescribed fire allocation for managing fire risk in central Catalonia

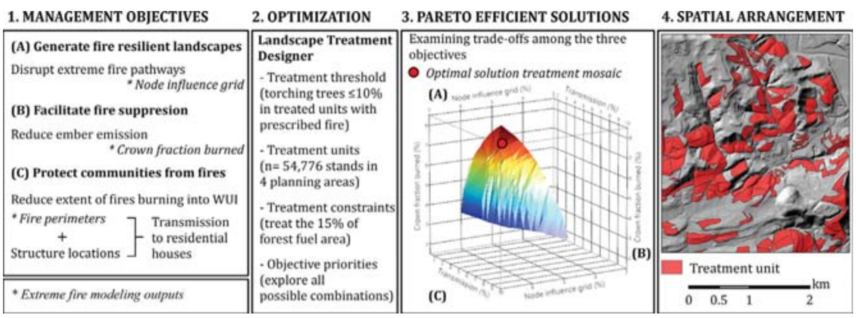
Fermín J. Alcasena [a,*], Alan A. Ager [b], Michele Salis [c,d], Michelle A. Day [e], Cristina Vega-Garcia [a,f]

[a] Agriculture and Forest Engineering Department (EAGROF), University of Lleida, Alcalde Rovira Roure 191, 25198 Lleida, Catalonia, Spain
[b] USDA Forest Service, Pacific Northwest Research Station, Western Wildland Environmental Threat Assessment Center, 3160 NE 3rd Street, Prineville, OR 97754, USA
[c] National Research Council, Institute of Biometeorology (CNR-IBIMET), Regione Baldinca, 07100 Sassari, Italy
[d] Euro-Mediterranean Center on Climate Change (CMCC), IAFES Division, Via Enrico De Nicola 9, 07100 Sassari, Italy
[e] Oregon State University, College of Forestry, Forest Ecosystems & Society, 321 Richardson Hall, Corvallis, OR 97331, USA
[f] Forest Sciences Centre of Catalonia, Carretera de Sant Llorenç de Morunys km 2, Solsona 25280, Catalonia, Spain

HIGHLIGHTS

- Prescribed fire treatment optimization for reducing wildfire risk is challenging.
- We designed a multi-objective treatment mosaic for a fire-prone Mediterranean area.
- We used an optimization program to explore trade-offs among competing objectives.
- Results can be used to evaluate ongoing projects and improve long-term efficiency.
- Spatial optimization can guide investments on large landscape management projects.

GRAPHICAL ABSTRACT



ABSTRACT

We used spatial optimization to allocate and prioritize prescribed fire treatments in the fire-prone Bages County, central Catalonia (northeastern Spain). The goal of this study was to identify suitable strategic locations on forest lands for fuel treatments in order to: 1) disrupt major fire movements, 2) reduce ember emissions, and 3) reduce the likelihood of large fires burning into residential communities. We first modeled fire spread, hazard and exposure metrics under historical extreme fire weather conditions, including node influence grid for surface fire pathways, crown fraction burned and fire transmission to residential structures. Then, we performed an optimization analysis on individual planning areas to identify production possibility frontiers for addressing fire exposure and explore alternative prescribed fire treatment configurations. The results revealed strong trade-offs among different fire exposure metrics, showed treatment mosaics that optimize the allocation of prescribed fire, and identified specific opportunities to achieve multiple objectives. Our methods can contribute to improving the efficiency of prescribed fire treatment investments and wildfire management programs aimed at creating fire resilient ecosystems, facilitating safe and efficient fire suppression, and safeguarding rural communities from catastrophic wildfires. The analysis framework can be used to optimally allocate prescribed fire in other fire-prone areas within the Mediterranean region and elsewhere.

Keywords:
Prescribed fires
Fire modeling
Treatment optimization
Production possibility frontiers
Mediterranean areas

* Corresponding author.
E-mail address: ferminalcasena@eagrof.udl.cat (F.J. Alcasena).

1. Introduction

Uncharacteristic large fire events in the Mediterranean basin during the last decades suggest a rapid evolution of a fuel-limited anthropogenic fire regime to a weather-driven post-industrial regime (Fernandes et al., 2016; Pausas and Fernández-Muñoz, 2012; Seijo and Gray, 2012). Increasing fuel connectivity and buildup are the main contributing factors to large fires, and result from fire suppression policies, rural exodus, lack of management, and extensive afforestation (Bovio et al., 2017; Curt et al., 2016; Poyatos et al., 2003). Mediterranean areas represent one of the most important fire activity hotspots worldwide (Moritz et al., 2014), and in southern European Union (EU) countries (Portugal, Spain, France, Italy and Greece) 48,640 fires burned 447,807 ha annually on average between 1980 and 2015 (San-Miguel-Ayanz et al., 2016). Relatively few large fires (<10%) associated with extreme fire weather conditions accounted for the bulk of burned area (>80%). These mega fires often occur in multiple-fire episodes, overwhelm suppression capabilities, emit spot-fires capable of breaching fuel breaks (>100 m), spread for long distances (>10 km) and impact many communities located in the wildland urban interface (Alcasena et al., 2016b; Castellnou and Miralles, 2009; San-Miguel-Ayanz et al., 2013). Furthermore stand replacing high severity events threaten remaining old growth forests and increase future fire hazard by promoting dense regeneration from serotinous conifer species ($>10^4$ tree saplings ha^{-1}), resprouting shrublands, and coppice stands (Pausas et al., 2008). Traditional wildfire management strategies based solely on fire suppression and ignition prevention programs have proven to be ineffective (Keane et al., 2008; Piñol et al., 2007), and managing fuels on fire-prone landscapes represents the most promising strategy capable of reversing the escalation of mega fire events and restoring fire resilient ecosystems (Hessburg et al., 2016; Reinhardt et al., 2008).

Prescribed fire is a widely used fuel treatment technique on large landscapes due to its low cost and high efficiency in reducing surface fuels, removing ladder fuels and increasing crown base height (Agee and Skinner, 2005; Casals et al., 2016; Fule, 2002). Fighting fire with fire represents an important paradigm shift after decades of suppression policy, and the positive effects in terms of fire risk reduction, especially in fire adapted ecosystems, have now been widely demonstrated (Arkle et al., 2012; Fernandes, 2015; North et al., 2012; Prichard and Kennedy, 2014; Vaillant et al., 2009). Despite existing administrative and legal constraints, operational limitations and lack of social acceptance, the use of prescribed fire by landscape managers to treat fuels is gaining importance in fire-prone southern European countries (Ascoli and Bovio, 2013; Molina-Terrén et al., 2016). In addition, prescribed fire can be used to restore habitats, maintain forest canopy openings, facilitate natural regeneration, clear logging debris, control pest and disease, and improve pastures in mountain areas (San Emeterio et al., 2016). In fact, until the mid-1950s in many southern EU countries fire was used systematically in rural areas for pasture and edge clearing, and agricultural waste elimination (Lázaro, 2010). However, conditions in some forest stands are not suitable for prescribed fire treatment due to the potential for fire escape, smoke impacts, negative effects on the topsoil and undesired effects on certain vegetation structures or species compositions and tree growth (Armas-Herrera et al., 2016; Valkó et al., 2014; Valor et al., 2015). For instance, mechanical treatments such as thinning and mastication or entire tree harvesting are required in high fuel load conditions or dense forest ecosystems with ladder fuels to reduce canopy bulk density and mitigate hazard prior to using fire to reduce fuels. Thus prescribed fire programs, especially on large, highly fragmented, and complex land tenure landscapes (i.e., $>10^5$ ha) require accurate stand-level information to properly plan fuel treatments.

Planning fuel treatments to reduce large fire spread is a complex problem and must consider how to efficiently treat landscapes in terms of spatial configuration and density of treatments. In addition, legislation regulating management in protected areas, as well as land ownership constraints, complicates treatment allocation. Treatment strategies must consider multiple objectives, causing the spatial configuration of fuel treatments to substantially differ from case to case (Ager et al., 2013; Oliveira et al., 2016; Schmidt et al., 2008; Stevens et al., 2016; Thompson et al., 2017). For instance, while treatments designed to reduce wildfire likelihood may be prioritized in areas likely to maximize reduction in spread rate (Finney, 2007), treatments designed to mitigate structure ignition in residential communities would prioritize treating hazardous fuels surrounding valued assets (Calkin et al., 2014; Cohen, 2000; Elia et al., 2014). In the former case, a fire modeling approach is required to model fire spread, and the latter will depend on the valued asset location and surrounding vegetation. Despite the high interest in developing multi-objective treatment prioritization guidelines to efficiently allocate investments, few studies have provided transferable results that could be used by landscape managers (Salis et al., 2016b; Scott et al., 2016). Previous studies assessed wildfire risk or exposure to highly valued resources and typically did not include assessment of alternative treatment designs and their effect on wildfire (Alcasena et al., 2016b; Argañaraz et al., 2017; Mitsopoulos et al., 2015; Salis et al., 2013; Thompson et al., 2015), but see also Collins et al. (2013) and Moghaddas et al. (2010). For instance, there has been little study of how fuel management activities including mechanical treatments in concert with prescribed fire can meet the divergent objectives of restoring fire adapted ecosystems versus protecting developed areas from wildfire impacts. Specifically, how does focusing on one fuel management objective result in trade-offs in others, and where are these opportunities to achieve multiple fire management objectives? Recent studies have explored these questions using production possibility frontiers (PPFs) to show trade-offs associated with a fixed amount of investment in fuel management (Ager et al., 2016b; Vogler et al., 2015). These analyses used PPFs to graphically represent Pareto efficient optimal resource allocations for competing objectives associated with a fuel treatment program (e.g. habitat restoration vs. wildfire risk mitigation). These PPFs can be used to identify the opportunity cost of a manager's decision to support one particular objective at the expense of the other.

In this study we experimented with new methods for allocating prescribed fire treatments on a large fire-prone landscape ($>10^5$ ha) in central Catalonia (northeastern Spain). Recent catastrophic fires in the study area have motivated managers and policymakers to re-examine fire policies including the development of a comprehensive and strategic fuel treatment program (Castellnou and Miralles, 2009; Costa et al., 2011). To help inform these policy discussions we conducted a case study that combined fire simulation and trade-off analyses to evaluate the compatibility of three prescribed fire management objectives that focused treatments to improve: 1) forest resiliency to fire, 2) effectiveness of fire suppression, and 3) protection of rural communities. We used optimization methods to examine both trade-offs among the objectives and priorities for sample planning areas. We discuss application of the methods to evaluate current and proposed fuel management programs as part of strategic policy development as well as field application by local fire managers.

2. Material and methods

2.1. Study area

The 0.13 million ha study area encompasses Bages County in central Catalonia (northeastern Spain) (Fig. 1A). Major communication corridors transverse the study area from north to south and east to west, apart from the secondary roads which present a radial distribution connecting the capital city of Manresa in the core of the study area with secondary urban centers. The orography ranges in elevation from 150 m in the central valley to >1,250 m in the highest mountains. The climate is predominantly Mediterranean with an average annual precipitation of 500–900 mm, with <15 mm falling in the driest month of July when the mean maximum temperatures exceed 30 °C. Conifer forests are dominated by Aleppo pine (*Pinus halepensis* Mill., 22% of the

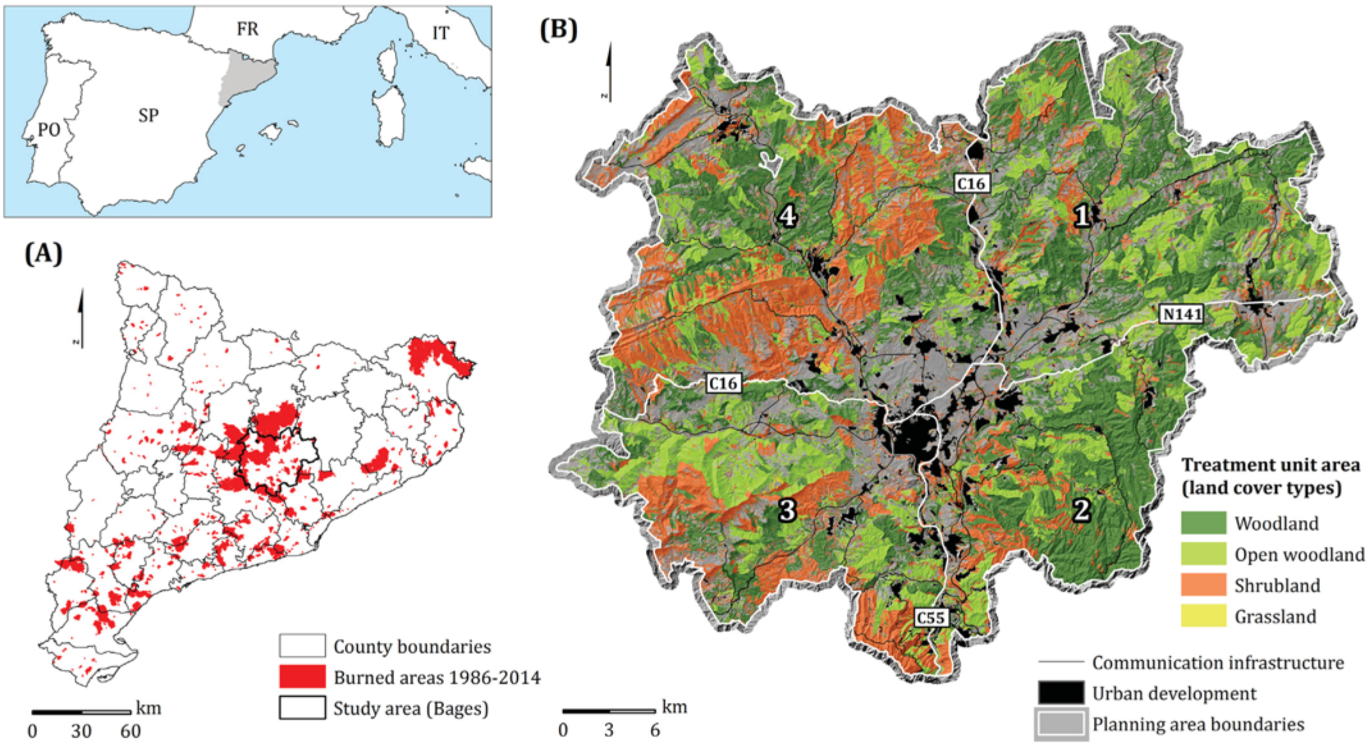


Fig. 1. (A) Location of the study area (Bages County, central Catalonia, northeastern Spain) and recent wildfire perimeters (interior.gencat.cat) and (B) planning area boundaries and treatment area by land cover type (agricultura.gencat.cat). Gray areas in (B) are areas ineligible for treatment.

study area) on south facing slopes and the lowest elevations, with black pine at the higher elevations (*P. nigra* Arn. subsp. *salzmannii*, 14%). Mediterranean pastures and low shrublands dominated by thyme (*Thymus vulgaris* L.), rosemary (*Rosmarinus officinalis* L.), cushion-heads (*Genista scorpius* L.) and kermes oak (*Quercus coccifera* L.) which have colonized abandoned agricultural lands, occupy a substantial portion of the landscape (14%). Overall, Mediterranean oaks (*Q. faginea* Lam, *Q. pubescens* Willd. and *Q. ilex* L.) have a limited presence as pure stands (<10%). Dryland cereal crops cover most valley bottoms (23%) and surround main city centers and urban development areas (8%). On average, about 1,000 ha (i.e., 0.77% of the study area) are burned annually by wildfires (period 1986 to 2015), mostly from human caused ignitions, and historical large fire episodes of 1986, 1994 and 1998 accounted for 86% of the cumulative burned area (MAAyMA, 2015). During the last 30 years large fire events (>100 ha) burned 22% of the study area (Fig. 1A), and here vigorously sprouting oaks and high density Aleppo pine forests replaced the dominant black pine stands (Retana et al., 2002; Rodrigo et al., 2004). Moreover, recent heavy snow and strong wind episodes (e.g., 2006 year) substantially increased coarse fuel loads on unmanaged forests with falling trees and broken branches, and wildfire events in the future will potentially show even greater wildfire hazard.

2.2. Residential housing at risk

The capital city of Manresa is located in the center of the study area and accounts for about 42% of the population (74,752 inhabitants). Nonetheless, several hundred dispersed rural houses and farms are spread across a rural urban interface characterized by very low housing density (i.e., <6.18 houses km^2) and high wildfire hazard. Only arable lands remain cultivated and residential structures closely intermingle with forest fuels in most cases. In addition, their often precarious maintenance increases fire susceptibility and makes structures vulnerable to ignite from showering embers, despite the fire resistant materials used on rural construction. In order to accurately identify all these individual structures, we used the structure polygon centroids from the BTN25 (IGN, 2016) to generate a point file with residential house locations. The 1:25000 scale BTN25 official geodatabase is a widely used spatial information resource for landscape and urban planning at the municipality level. In all, we identified 23,633 individual residential houses across the study area, excluding industrial structures, silos and agricultural machinery storage.

2.3. Planning areas and treatment units

We divided the study area into four planning areas (i.e., project scale blocks) considering major communication infrastructure (north to south C16 and C55 roads, and east to west C25 and N141 roads, Fig. 1B). The planning areas ranged in size from 25,140 to 43,470 ha (average = 32,480 ha). Treatment units (i.e., minimum management area for treatment implementation) were derived from the forest land SIGPAC2016 polygons (agricultura.gencat.cat). These polygons are used as reference in EU rural development and agricultural subsidy monitoring, and accurately delineate at a 1:5000 scale major land cover types (i.e., agricultural, grasslands, shrublands, open woodlands, forested, water bodies, urban areas and rocky outcrops) according to land ownership boundaries (Fig. 1B). We excluded agricultural and unburnable cover types, and then largest land cover units where further divided into polygons with a maximum area of 6 ha to homogenize the spatial resolution and better capture spatial gradients in treatment objective metrics across the landscape. We used forest tracks and natural breaks such as ravines, water divides and slope changes to split the large land cover units into smaller polygons. In total, we obtained 54,773 treatment polygons based on land cover with an average size of 1.67 ha.

2.4. Fire modeling

We used FlamMap for fire spread and behavior modeling (Finney, 2006). FlamMap has been widely used for landscape scale wildlife exposure and risk assessment in studies worldwide, including southern EU Mediterranean countries (Alcasena et al., 2016a; Elia et al., 2014; Jahdi et al., 2016; Mallinis et al., 2016). The landscape input data were constructed with topography, surface fuel and canopy metric grids (Ager et al., 2011). Using hourly weather records from a long series automatic weather station within the study area we characterized the most frequent wildfire season wind scenario (speed and direction), and derived the fuel moisture content (Bradshaw and McCormick, 2000). Fire modeling was conducted at 40 m resolution considering extreme weather conditions (97^{th} percentile) to obtain node influence grid (NIG), crown fraction burned (CFB) and individual fire perimeters (Alcasena et al., in press).

2.5. Wildfire management objectives

We explored three management objectives in this study: 1) increasing the resiliency of sub-Mediterranean forest ecosystems, 2) facilitating fire suppression, and 3) protecting wildland urban interface rural communities from catastrophic events. Currently these objectives represent the major concerns for fire managers and Civil Protection in Catalonia (Costa et al., 2011). Different spatially explicit metrics were assigned to each objective in order to later facilitate the spatial optimization analysis.

2.5.1. Promote fire resilient forest ecosystems

Currently most forests in the study area are high density or with ladder fuel structures, where stand replacing high severity events threaten forest ecosystems. Endemic sub-Mediterranean old growth black pine stands in the study area (i.e., Castelltallat mountain range) are protected Natura 2000 European Union (EU) sites (Council Directive 92/43/ECC of 21 May 1992) vulnerable to large and intense fire events. Treating forest fuels can reduce large catastrophic fire potential and burn probability on fire-prone landscapes, in addition to mitigating hazard on treated stands and reducing expected tree mortality (Ager et al., 2007). Accordingly, heading fire pathways on large landscapes represent strategic areas to locate fuel treatments, while the minimum treatment area and intensity in reducing fuels also represent very important factors to efficiently design prescribed fire projects (Finney, 2007). We used the node influence grid fire modeling output (NIG; Fig. 2A; Finney, 2006) as the reference metric to optimize fuel treatment efforts to increase resiliency in forest ecosystems. Overall the treatment units with highest values will be prioritized, while units with lowest values and limited

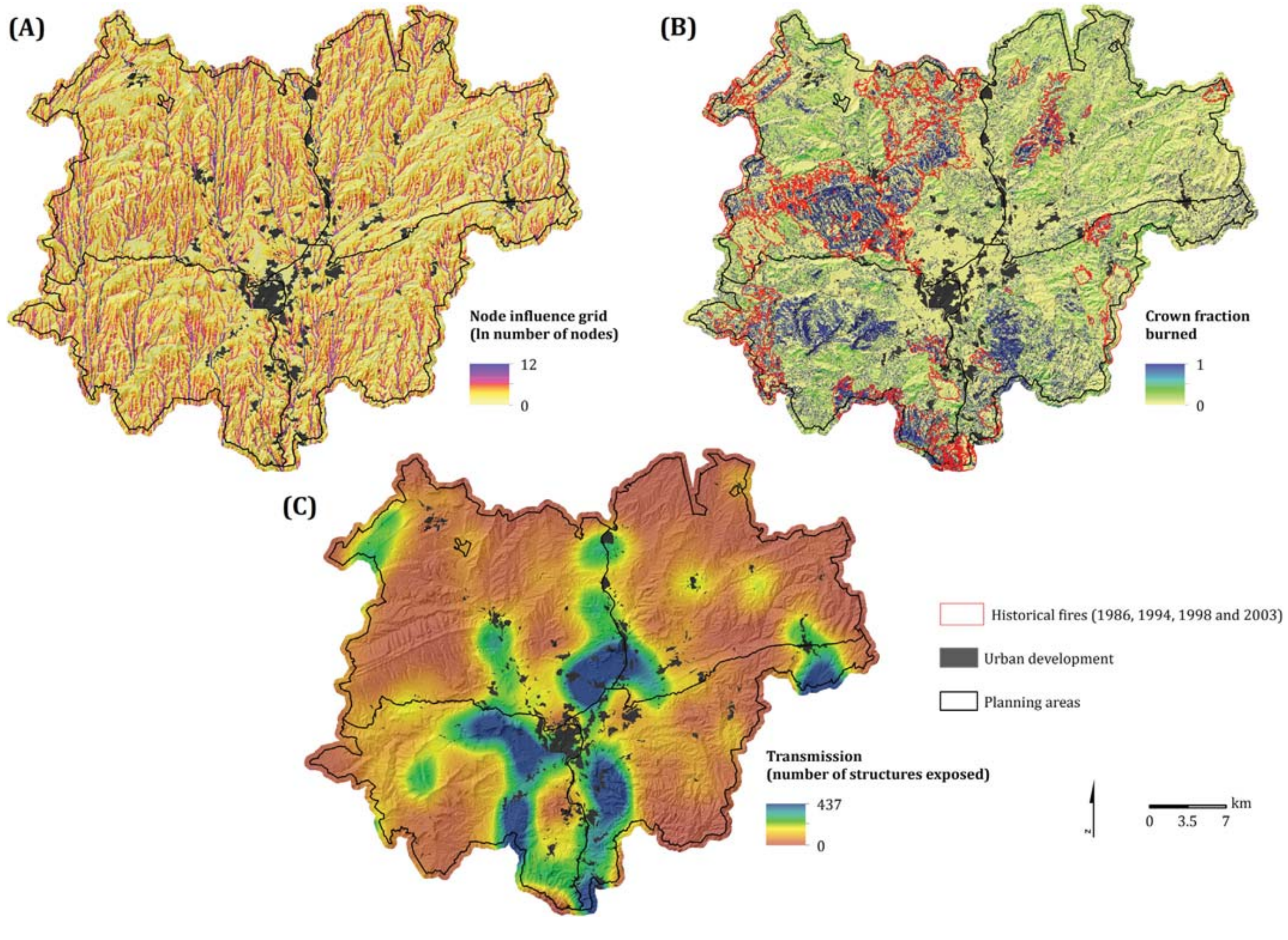


Fig. 2. Fire modeling outputs and exposure metrics corresponding to node influence grid (A), crown fraction burned (B) and wildfire transmission to residential structures (C) used to prioritize prescribed fire treatments in central Catalonia, northeastern Spain. We considered extreme fire weather conditions (97^{th} percentile) for fire modeling with FlamMap (Finney, 2006). See Alcasena et al., in press for further details on modeling outputs and exposure metrics.

influence on major fire propagation will be excluded for treatment allocation. Since the study area is subjected to severe fires and reducing fuels on the entire landscape is impossible, treating areas with high NIG is the most efficient way to reduce large fire spread and therefore increase landscape resiliency to fire.

2.5.2. Facilitate fire suppression

Ember emission represents one of the main factors overwhelming fire suppression capabilities on Mediterranean areas. Despite existing high fragmentation on landscapes with mosaics of cultivated lands and dense communication networks, spot-fires during plume-driven fires easily surpass surface fire strategic containment barriers. In fact, long spotting distances as much as 2 km have been recorded on historical large fire events in Catalonia (Costa et al., 2011). Reducing ember emission will substantially increase firefighter safety and efficiency during fire suppression, reducing entrapment possibilities and increasing fire-front containing success probability via backfires or black-line anchoring implementation from existing linear fuel discontinuities. We used the crown fraction burned (CFB; Fig. 2B) output to target likely ember emitting forest stands. Moreover, treating stands with highest CFB values (i.e., highest crown fire severity) will also increase future fire resistance on treated stands. We prioritize treatments on stands presenting highest average values and intermittent to continuous crown fire types.

2.5.3. Safeguard rural communities from large catastrophic fires

Protecting residential communities from catastrophic fires is the main priority for most civil protection agencies and wildfire managers, since long distance spreading fires can burn into multiple rural communities and affect multiple residential houses. Previous studies demonstrated how landscape fuel treatments can mitigate large fire arrival to residential areas, and in this study we used fire transmission to residential houses to target treatment units where ignited fires affect a high number of structures (Ager et al., 2016a; Ager et al., 2010). We define fire transmission (TR) as the number of structures exposed from fires ignited in a given location during typical blow-up events in the study area. For that purpose, we intersected fire modeling large fire perimeter outputs ($n = 6816$ fires > 100 ha) with residential house centroid locations ($n = 23{,}633$ structures) to assess fire transmission (Alcasena et al., 2017a). The number of exposed structures was assigned to fire ignition locations, and we used exponential kriging geostatistical methods (radius = 3000 m) to create a 40 m resolution smooth raster surface (Fig. 2C) in order to populate all treatment units with average values.

2.6. Spatial optimization analysis

In order to facilitate the treatment unit identification in the later optimization analysis, we used modeling metrics and exposure results to prioritize treatment allocation according to the different wildfire management objectives. We first populated the treatment unit polygons with average values, and then the percentage contribution with respect to the total of all treatment units (*pct*) was calculated to standardize reporting across all objectives, and assess the attainment degree of all treated units on a given project. We define the objective attainment as the percentage value contribution of a treatment unit or stand on achieving a given objective by implementing a fuel treatment on it, assuming a fulfillment degree proportional to the value on the treated unit with respect to the total in the planning area or study project. In other words, we quantified on every treatment unit the percentage value with respect to the cumulative values of all units (e.g., treating a unit with a value equal to 1 where the total value of all treatment units equals 1000 will have a $pct = 0.1\%$ for a given objective).

Then, we used the Landscape Treatment Designer (LTD) to optimize prescribed fire fuel treatment allocations in the study area (Ager et al., 2013). LTD has been used in forest restoration studies to analyze trade-offs among competing objectives and rank treatment priorities on planning areas for large western US landscapes (Ager et al., 2016b; Vogler et al., 2015). The program identifies the treatment units which maximize attainment levels for multiple objectives considering managers' priorities or weights for different objectives, limited resources for treatments (e.g., budgetary restrictions), implementation constraints (e.g., forest stands susceptible to high severity prescribed fire) and legislation (e.g., excluding protected areas). The optimization equation is the following:

$$Max \sum_{j=1}^{k} \left(Z_j \times \sum (W_i \times N_{ij}) \right) \tag{1}$$

subject to:

$$\sum_{j=1}^{k} (Z_j \times A_j) \leq C \tag{2}$$

where C is a global constraint on investment level per planning area (e.g., budgetary funds for treatments or treated equivalent area), Z is a vector of binary variables indicating whether the j-th stand is treated (i.e., Z = 1 treated and Z = 0 untreated), N_{ij} is the contribution (i.e., *pct* percent contribution to the total) to objective i in stand j if treated, and A is the treated area of the j-th treated stand. Since landscape managers can present different priorities, the maximization equation can integrate a W_i weighting coefficient to promote the i-th objective versus another.

In this study we assumed constant cost per treated ha with prescribed fire within the study area, and therefore polygon area represented the C constraint value for individual treatment units. We considered a 15% treatment area (13,684 ha) on forest lands, since lower treatment intensities have little or no influence on limiting large fire spread (Finney, 2007; Salis et al., 2016b). We are considering the use of prescribed fire as the treatment technique to reduce fuels, but not all forests in the study area are eligible for treatment due to dense ladder fuels on unmanaged timber-stage forests or very dense pole-stage post-fire regenerated stands (i.e., 1986, 1994, 1998 and 2003 fire cohorts). Fire caused mortality of trees requires crown consumption or substantial damage to cambium or roots, and we excluded forest stands with a crown fraction burned higher than 0.10 (i.e., >10% of torching trees on the overstory) for prescribed fire burn window conditions (Alcasena et al., in press). In order to accurately identify forested units we used LiDAR derived 20 m resolution canopy height data (ICGC, 2005) to discriminate between low vegetation and tree covered units considering a 3 m height break, and explore the alternative fuel treatment possibilities and potential revenue from stands excluded for prescribed fire treatments (see Appendix 1). In order to explore local managers' potential priorities or choices in the assignation of priorities for the treatment objectives (i.e., trade-offs between objectives), we ranged objective weights (W) from 0 to 5 in all integer combinations. First, for every weighting combination we obtained a solution with the respective attainment values for the three objectives. Planning area level production possibility frontier (PPF) three dimensional graphs were then generated from the representation of all the weighting scheme combination results using a separate axis (X, Y and Z) per objective.

3. Results

3.1. Fire modeling and exposure metrics

The node influence grid (NIG) showed a dominant wind oriented stripe-type spatial gradient, where the highest values were located over south-north oriented major fire pathways in line with the southern wind direction used for fire modeling (Fig. 2A). The average NIG within the study area was 3, and varied from a low of 0 to a high of 12. Treatment unit average NIG values presented similar ranges and distribution in all the planning areas (Fig. 3A). Although fire pathways were in most cases able to adjust spreading trajectories to valley bottom herbaceous fuels, the fires occasionally spread faster through forest fuels on steep slopes when the orientation of the valley bottom was perpendicular to dominant wind direction. While fuel discontinuities such as unburnable areas in urban development in the central part of the study area locally modified the fire trajectories, the fastest spreading pathways were located in shrublands where fire trajectories were generally straight (Fig. 1B). Nonetheless, average values on the treatment units classified by land cover type showed very similar distributions (Fig. 4).

Crown fraction burned (CFB) showed very interesting spatial patters across the study area (Fig. 2B). Large portions of the landscape in the central part of planning area 4 presented continuous crown fire (> 0.9) on the areas burned during 1994 and 1998 wildfire events. On the other hand, the patches burned on more recent wildfires (i.e., 2003, perimeters on the eastern and southeastern portion of planning area 2) indicated the forest has yet to recover and did not present any crown fire activity. On the eastern side and northeastern parts of the study area, the dominant intermittent crown fire level varied between 0.2 and 0.6, and the highest values were usually located on south facing slope mountain edges perpendicular to the dominant winds. In general, treatment unit average CFB values were slightly lower for planning areas 1 and 4 (Fig. 3B). The comparison of treatment unit average CFB values between extreme weather and prescribed fire conditions (Appendix 1) on areas burned within past fire events, indicated that young regenerating forests are especially prone to active crown fires (Fig. 5). While CFB was especially high for extreme fire weather within wildfires burned in 1998, differences between extreme fire weather and prescribed fire conditions within wildfires burned in 2003 were not marked. Currently most forest stands within 1998 wildfire perimeters presented CFB values above the prescribed fire treatment threshold and were therefore excluded from the treatment optimization analysis.

Fire transmission (TR) to residential houses (i.e., structures exposed to wildfire) located within the study area showed clustered patterns that where generally related to structure location, wind direction and fire size (Fig. 2C). Overall, the highest values (>350 residential structures) were concentrated in the central and southern portions of the study area, the location of the largest urban areas. In many cases, these areas corresponded to dryland cereal crop agricultural lands excluded as potential treatment units for the optimization analysis (Fig. 1B). Areas with the lowest TR were located in the northern and southwestern rough terrain forest lands, where rural communities are especially small in comparison with the larger cities in the central part. Treatment unit average TR value distributions varied between the planning areas and the bulk of values were higher on planning area number 3 (Fig. 3C). Overall, the largest TR values for individual fires surpassed 1000 structures but these did not necessarily correspond to the largest fires, and we did not find a very clear positive correlation between fire size and the number of residential houses exposed to wildfire (Fig. 6A to D). In

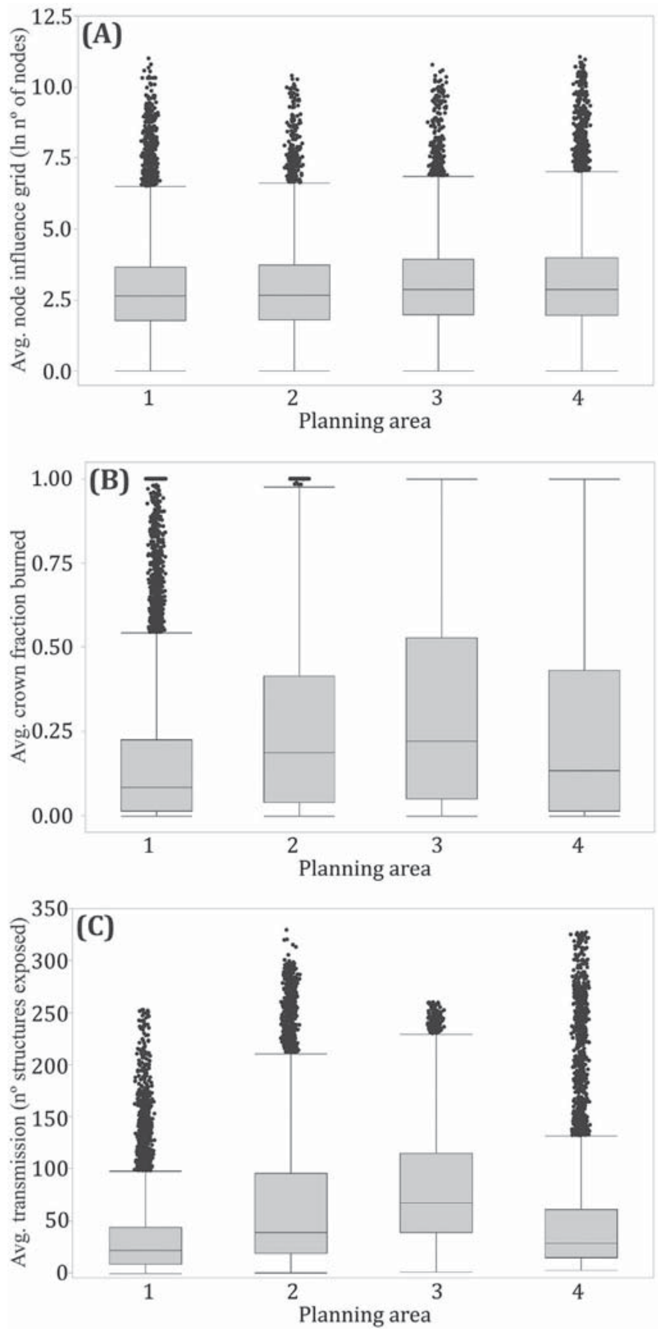


Fig. 3. Box-plots of fire modeling outputs and exposure metrics for treatment units within the four planning areas in the Bages County (central Catalonia, northeastern Spain). The boxes indicate the 1st/3rd quartiles, the whiskers indicate 10th/90th percentiles, the black line within the box is the median, and the dots indicate values below 10th percentile or above the 90th percentile. See methods for details on modeling outputs and exposure metrics.

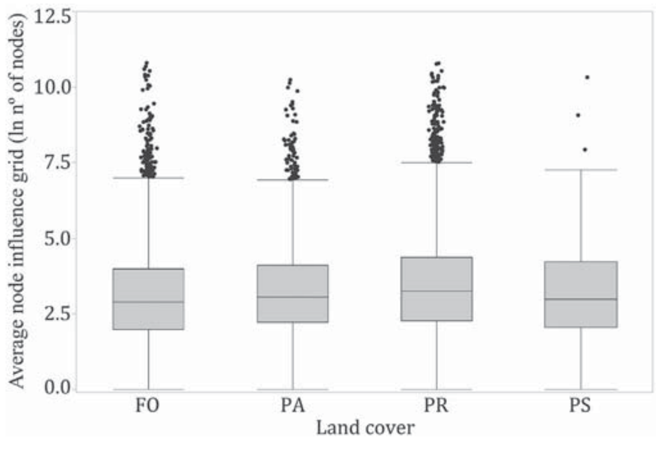


Fig. 4. Average node influence grid values for different land use-land cover types within Bages County (central Catalonia, northeastern Spain). Land cover data are from SIGPAC2016 (agricultura.gencat.cat). The boxes indicate the 1st/3rd quartiles, the whiskers indicate 10th/90th percentiles, the black line within the box is the median, and the dots indicate values below 10th percentile or above the 90th percentile. Abbreviations: FO: woodland, PA: open woodland, PR: shrublands, and PS: grasslands.

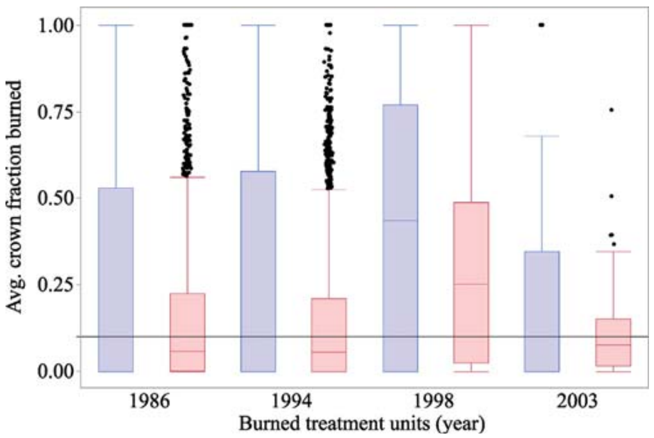


Fig. 5. Crown fraction burned (CFB) fire modeling results box-plots for treatment units located on previously burned areas (1986, 1994, 1998 and 2003) in Bages County (central Catalonia, northeastern Spain). The blue color corresponds to extreme fire weather modeling results (Fig. 2B) and the red refers to prescribed fire treatment weather conditions (Alcasena et al., in press). The boxes indicate the 1st/3rd quartiles, the whiskers indicate 10th/90th percentiles, the black line within the box is the median, and the dots indicate values below 10th percentile or above the 90th percentile. The horizontal line (CFB = 0.1) indicates where forest stands experience >10% of trees torching when implementing prescribed fires and thus were excluded from the treatment optimization analysis. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

fact, the largest fires (>12,000 ha) presented TR values below 500 structures (Fig. 6D), and the highest rates corresponded to fires <10,000 ha ignited in planning area 3 (Fig. 6C) with >8 structures ha^{-1}, although it is important to note the capital city of Manresa is located in planning area 3 (and contains 30% of the residential structures in the study area).

3.2. Production possibility frontiers

Attainment values with respect to the total within the entire study when each objective is optimized independently ranged between 19% and 33% (Fig. 7). Treatments located to prioritize the highest NIG units achieved the lowest value, and variation among the four planning areas was <3%. On the other hand, CFB and TR attainment values showed substantial variation, especially between planning areas 1 and 4. The highest planning area level attainment values corresponded to TR reaching very close to 10% in planning areas 2 and 4. The amount of attainment achieved per unit of treated area ranged from a low of 0.0010% ha^{-1} in planning area number 1 for CFB to a high of 0.0034% ha^{-1} in planning area 2 for TR.

We calculated production possibility frontiers (PPFs) for each of the four planning areas to explore the trade-offs among the different objectives and how they varied across the study area (Fig. 8). For every planning area, we graphically represented a PPF surface as a three dimensional projection to show the maximum possible attainment level for treatments constrained to 15% of the treatable landscape. Therefore, the surface represents the optimal scenarios where resources

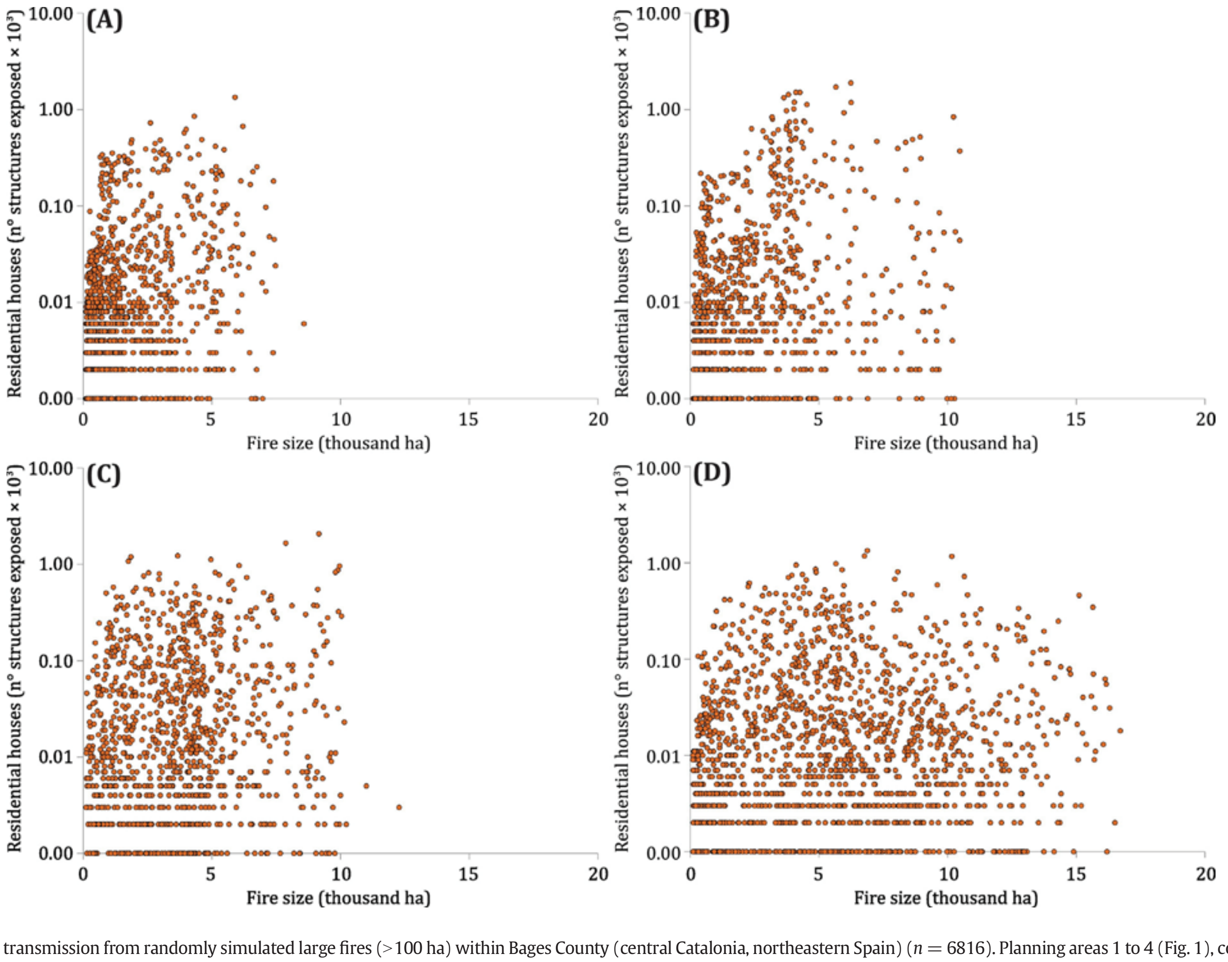


Fig. 6. Fire transmission from randomly simulated large fires (>100 ha) within Bages County (central Catalonia, northeastern Spain) ($n = 6816$). Planning areas 1 to 4 (Fig. 1), correspond respectively to the A to D scatterplots. We considered extreme fire weather conditions and 8 h fire spread duration to replicate historical catastrophic blow-up event patterns (e.g., Bages fire on 4th July 1994) with FlamMap (Finney, 2006). Note that planning area 3 (panel C) contains the capital city Manresa and 30% of the residential structures.

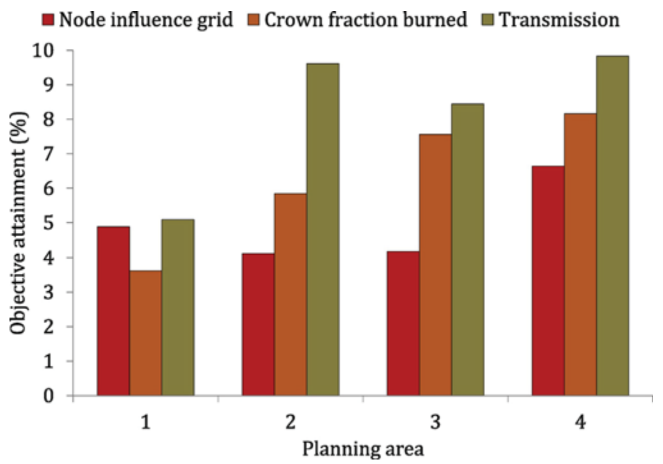


Fig. 7. Planning area attainment values on treated units in Bages County (central Catalonia, northeastern Spain) for each of the three metrics used to assess prescribed fire management objectives for the four planning areas, when each of the metrics is optimized independently. These correspond to optimization results from treating 15% of the burnable landscape within the study area, excluding forest stands where prescribed fire could cause undesired effects on the overstory. Node influence grid, crown fraction burned and transmission results (Fig. 2) were used to conduct the optimization analysis with the Landscape Treatment Designer (Ager et al., 2016b). See methods for more details on the fire model outputs and exposure metrics.

are invested most efficiently. PPF surfaces were concave to the origin and increasing attainment for a single objective was only possible by diverting resources (i.e., treated area) from another. Trade-offs presented an increasing opportunity cost when moving along a PPF surface from the maximum value of a one objective to increasing attainment of a second objective. Sharp trade-offs indicated high co-location possibilities, such as CFB with respect to TR in planning areas 2, 3, and 4 (Fig. 8B, C, D). On the other hand, TR with respect to NIG and CFB represented situations with the lowest joint-production among the objectives on treated units (Fig. 8C). Paradoxically, planning area 1 showed the highest trade-off between NIG and TR, but the lowest co-location between CFB and TR (Fig. 8A). The planning areas with concave PPF curves more distant to the origin (planning area 4; Fig. 8D) represented the highest joint-production potential, and thus the highest priority while implementing fuel treatment projects.

3.3. Treatment allocation spatial priorities

We generated the optimal multi-objective prescribed fire treatment allocation map (Fig. 9A) for the same priority setting in the three wildfire management objectives (i.e., same weights for all objectives in optimization, $W = 1, 1, 1$). These areas represent treatment units where all three of the metrics are optimized but may represent trade-offs between two particular metrics since obtaining the highest value for all the three objectives in one place was not possible. In the case of the three objectives having the same priority (local managers' choice), fuel treatments could be located in an optimal spatial design to promote fire resilient forest ecosystems, facilitate fire suppression and protect rural communities from large fires. This treatment unit selection mosaic is the solution where the joint production of all three metrics has the highest potential. Results revealed a fine grain, complex mosaic across the study area (Fig. 9A).

We also generated a combined map from independently optimized results to explore trade-off implications (i.e., managers' choice on the objective priority) in treatment unit selection for prescribed fire treatments (Fig. 9B). In other words, we overlaid the optimal mosaics for the different metrics to show treatment unit level potential spatial co-location and how treatment unit selection would change depending on the objective prioritized. As expected, TR results tended to cluster around the main populated areas where we find the highest number of residential structures. Despite most CFB units concentrated on 1998 burned areas in planning area number 4, overall the NIG and CFB showed a more complex widespread pattern across the landscape, especially for NIG. On the overlaid mosaic (Fig. 9B), the treatment units selected where the three metrics overlapped accounted for 2581 ha, and two of the three metrics overlapped in other 7774 ha.

4. Discussion

Rural Mediterranean landscapes have evolved since the mid-20th century from highly fragmented mosaics of small agricultural parcels interspersed with heavily grazed pastures and intensively managed forests, to relatively homogeneous dense vegetation with high fuel loadings (Moreira et al., 2011; Pausas and Fernández-Muñoz, 2012). Fuels fragmentation in the earlier conditions limited the spread of both agro-pastoral and lightning-caused fires, whereas under current conditions, fires spread unimpeded until contained by suppression forces. Relying on fire suppression as a primary strategy is increasingly being questioned in the Mediterranean region and elsewhere (Calkin et al., 2014; Thompson et al., 2017) as fire regularly overwhelms suppression activities and results in large scale human and ecological impacts (Cardil et al., 2017; San-Miguel-Ayanz et al., 2013; Xanthopoulos et al., 2009). Clearly, longer-term strategies to counter wildfire impacts must consider fuels management as a synergistic strategy to reduce fire spread and facilitate containment, particularly in the context of future climate change (Batllori et al., 2013; Bedia et al., 2014; Lozano et al., 2016; Turco et al., 2014). However, integrating the use of prescribed fire and other fuel management activities into current wildfire management on large landscapes poses many challenges for landscape managers. Competing landscape management objectives that may or may not be compatible with prescribed fire creates a complex spatial trade-off problem for managers that seek to identify optimal arrangements within economic and other constraints (Ager et al., 2017b).

In our study, we generated a wide range of potential treatment designs for a 0.13 million ha fire-prone area in central Catalonia, where prescribed fire treatments can potentially be used to re-create fuel mosaics that increase fire resiliency, facilitate fire suppression, and mitigate fire transmission into residential communities. Using large treatment units (>25 ha) average attainment values in the optimization can mask high differences within the polygons, and we used small and homogeneous grain treatment units (≤6 ha) to accurately capture existing sharp spatial gradients in objective metrics (Fig. 2B) and increased allocative efficiency. Our approach can be easily adapted to other fire-prone Mediterranean areas or elsewhere considering a range of treatment priorities, objectives or potential environmental constraints for fuel treatment implementation. Accordingly, we should point out that land ownership (i.e., private, public owned by municipalities and public owned by the regional government) is an important factor conditioning fuel treatment allocation not considered in this study but requiring special attention in project implementation. Nonetheless, we generated prescribed fire scenarios that could be fine-tuned by wildfire managers to consider local conditions (topography, safety planning, escape risk, smoke concerns close to residential areas) to develop appropriate treatment allocations. Prior to treatments, selected units could be easily aggregated into larger blocks according to available material and human resources.

Our use of production functions makes it possible to explore a wide range of efficiency analyses in the development of prescribed fire plans. For instance, increasing investment levels will shift PPFs (i.e., current maximum possible attainment level) outwards, and potentially change the shape of the trade-offs as well as the overall efficiency. Although we

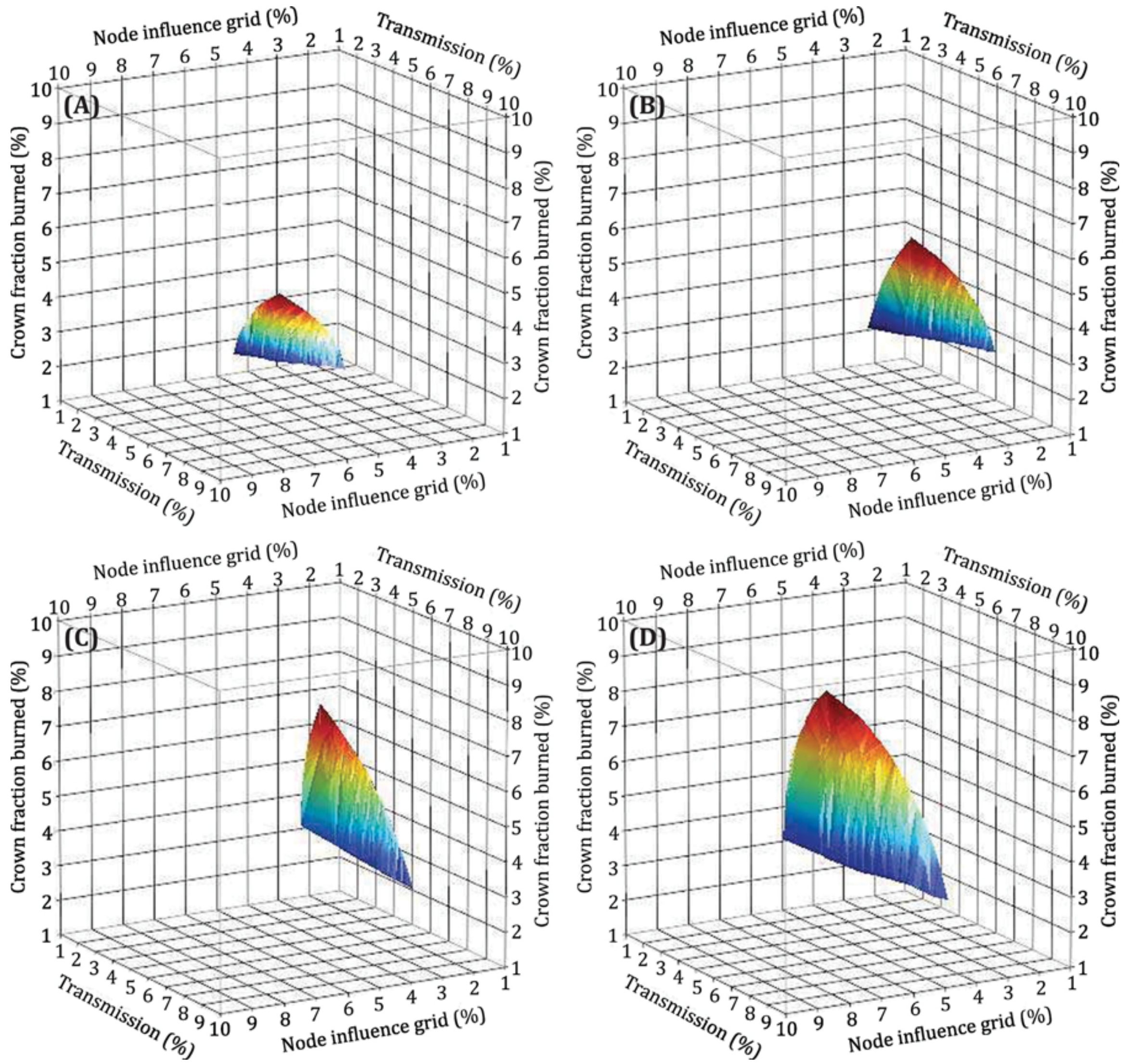


Fig. 8. Production possibility frontiers (PPF) of the three metrics used to assess prescribed fire management objectives for each of the planning areas. Planning areas 1 to 4 (Fig. 1) correspond respectively to panels A to D. The projected surface indicates the maximal-mix attainment within the study area on treated areas for the three metrics. Optimization results were obtained with the Landscape Treatment Designer (Ager et al., 2016a, 2016b) considering all integer weight combinations from 0 to 5 between the three metrics. Every point on the PPF has a corresponding treatment mosaic solution in the study area, where the optimization program identifies the individual treatment units for prescribed fire treatment location. The landscape was divided into 54,773 treatment units and we treated 15% of the burnable area. The convex PPF with respect to the origin indicates sharp trade-offs (e.g., high opportunity cost) when one particular goal is emphasized and the potential for efficient joint production. By contrast, a linear PPF indicates constant opportunity cost over all levels of production.

did not consider revenues on treated areas, we explored to what extent selling the timber stocks on excluded treatment units could increase available area for prescribed treatments. We found that a mere 1% has commercial value (from the 25,000 excluded ha), and revenues would only facilitate economic resources for treating another 0.5% in pre-commercial forest stands (see Appendix 1). Most of the excluded low pole-stage pre-commercial forests (17,258 ha) would require expensive mechanical treatments consisting of a systematic corridor opening with mastication treatments, plus manual lower for canopy pruning in tree covered strips in between (Navascuès et al., 2003). At this point, managers have two main options for these areas: utilizing existing subsidies to cover most of the treatment cost, or wait until the first commercial thinning at high pole-stage in 10–15 years despite the risk of an eventual crown fire. Indeed, the annual forest work subsidy call (co-founded with EU agroforestry and rural development 2014–2020 program) contemplates covering ≥75% of the total economic cost for risk mitigation thinning and mastication treatments on forest lands within natural sites of special interest ascribed to the certification system and presenting a management plan (e.g., Castelltallat mountain range natural site; Fig. 9). With regard to the second management option, rather than the marginal economic benefit from first commercial thinning (preferably as a heavy low-level thinning with entire tree extraction for biomass), changing stand structure into a low hazard forest to enable fire re-introduction in a few years should represent the main objective. Best conformation dominant trees (diameter at breast height > 20 cm) must remain after treatments and ladder fuels need to be eliminated from the understory to significantly mitigate wildfire hazard at strategic management points (SMP) (Madrigal et al., 2016; Ordóñez et al., 2005). All in all, target stands in SMPs should have a low tree density (150–200 trees ha^{-1}), single storied structure with a high canopy base (>5.5 m) and low fuel loads in the understory to withstand the most extreme events (Fernandes et al., 2015; Fulé et al., 2008).

Recent studies conducted in other fire-prone areas tested various optimization models to prioritize prescribed fire. Overall, these studies provide a number of methodological frameworks to solve the many challenges facing wildfire managers tasked with reducing wildfire risk. These challenges include identifying treatment spatial arrangement, treatment timing in long-term forest planning, suitability

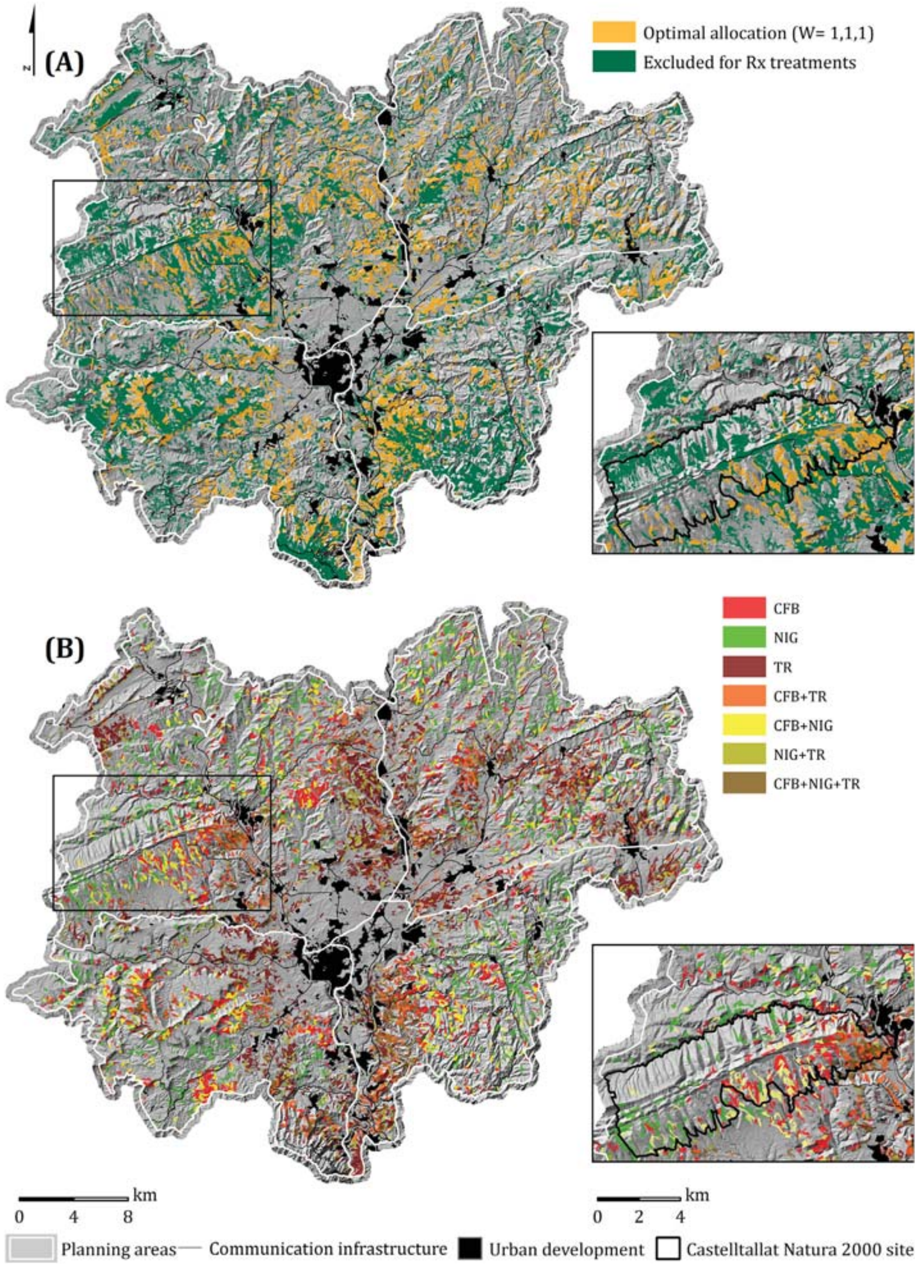


Fig. 9. (A) Optimal prescribed fire treatment locations in Bages County (central Catalonia, northeastern Spain) considering the same weights for all three metrics used to assess prescribed fire management objectives ($W = 1, 1, 1$). Implementing prescribed fire on densely regenerated young forest stands (e.g., *Pinus halepensis* cohorts with >10^3 trees ha^{-1} on 1998 Bages fire burned areas) could cause negative effects on the overstory (average crown fraction burned >0.1 or torching >10%), therefore these stands were excluded from the analysis. (B) We overlaid the treatment mosaic results when each metric was optimized independently (see attainments in Fig. 7) to explore areas where optimal solutions for a single metric overlap. The close up view corresponds to the Castelltallat mountain range Natura 2000 site of special interest and Sùria rural community. Abbreviations: CFB = crown fraction burned; TR = transmission; NIG = node influence grid; Rx = prescribed fire.

and combination with other treatments (thinning or mastication), and treatment integration into multi-functional forest management programs (González-Olabarria and Pukkala, 2011; Minas et al., 2014; Rachmawati et al., 2016; Vogler et al., 2015). In the current work we developed a multi-objective optimization approach to define optimal strategies and prioritize areas for implementing prescribed fire activities as part of larger fuel management programs. Previous optimization studies explored how treatment mosaics could be optimized to most efficiently disrupt large fire spread, and mitigate risk to communities (Chung et al., 2013; Rachmawati et al., 2015; Scott et al., 2016; Wei and Yehan, 2014; Wu et al., 2013). By contrast we explored how multiple fire management objectives can be achieved specifically with prescribed fire by identifying production possibilities (Fig. 9B). The methods are relatively simple compared to many other optimization models, thus facilitating wider implementation in a range of fire prone systems (Ager et al., 2017a). Large backlogs of prescribed fire treatments exist in many land management agencies, particularly in the western US, and tools to prioritize particular burn units to most efficiently achieve landscape resiliency objectives will become increasingly in demand. For instance, prioritizing prescribed fire to achieve desired landscape connectivity (Matsypura et al., 2017) could be performed with the methods we describe here.

In Catalonia, firefighters together with the Forest Service have been managing fuels since 1999, although on a limited basis, and the results from this study could be used to evaluate ongoing fuel treatment programs and provide insights into new project designs. For the former purpose, PPFs (Fig. 8) can facilitate multi-objective complex-solution project efficiency evaluation as informed by wildfire simulation and optimization. Nonetheless, quantitatively assessing the effectiveness for a specific solution (e.g., treatment mosaic on Fig. 9A) would require subsequent fire modeling considering the same fire weather conditions and treated landscape (Finney, 2007; Salis et al., 2016b). Our treatment plans (Fig. 9) could also be compared with existing management plans and historical wildfires to identify particular landscape features that could contribute to the design and refine the location of SMP for fuel treatments in Catalonia (Costa et al., 2011). For instance, recurrent long-distance spreading fire events burning under particular weather conditions provide interesting baseline information to characterize the most frequent synoptic scenarios associated with catastrophic events (Duane et al., 2015; Pereira et al., 2005; Rasilla et al., 2010), and the fire behavior that led to them (Duane et al., 2016; Salis et al., 2016a).

The development and persistence of vegetation and fuel mosaics on Mediterranean landscapes is influenced by a number of natural and anthropocentric disturbance factors that all must be integrated into strategic fuels planning. Fires can create fuel discontinuities and perpetuate grasslands or open woodlands that can limit the growth and severity of future fires. Post-fire afforestation activities can negate these benefits and perpetuate large continuous areas of hazardous fuels. At a minimum, commercial forestry activities need to consider fuel breaks to fragment the dense multi-storied forested landscapes that develop after afforestation activities. Livestock production can also facilitate fuels fragmentation and retard encroachment by highly flammable shrub vegetation (Elias and Tischew, 2016; Mena et al., 2016). Disturbances that create patches benefit game and protected species that prefer edge and open-habitats (De Cáceres et al., 2013). On the other hand, unburned patches play a key role in the regeneration ecology of low intensity fire-adapted non-serotinous conifer species (e.g., black pine *Pinus nigra*), obligate seeders that require mature stands to regenerate into openings created from severe fires (Martín-Alcón and Coll, 2016; Ordóñez et al., 2006). For instance, remaining old growth endemic black pine habitats after the 1994 and 1998 large fire episodes (e.g., stand-replacing fires burned 50% of Castelltallat mountain range endemic black pine habitat protected site; Fig. 9) are currently a conservation priority, where paradoxically restoring a low intensity cultural fire regime could help protect relict stands (Fulé et al., 2008). The combined effect of all of these factors must be integrated with fuel management plans such that landscape fuel mosaics that support low intensity fire can be created and maintained within economic and ecological constraints. The methods and tools described here can facilitate this process by providing the means to explore and identify spatial patterns of fuel management activities that promote the development of these landscape conditions.

5. Conclusions

Uncharacteristic fires during the last several decades are evidence of an ongoing transition towards an extreme weather-driven fire regime in Mediterranean landscapes. Increasing fuel loads and continuity represent the main factor responsible for these catastrophic events that overwhelm fire suppression capabilities as fires spread across unmanaged forest ecosystems and burn into developed areas. Managing forest fuels with prescribed fire has been demonstrated to be an efficient strategy to fragment fuels and reduce fire spread rates and severity. However, large scale strategic analyses to examine operational aspects of implementing prescribed fire are rare. We demonstrated an optimization framework to design strategically located treatment unit configurations that efficiently disrupt major fire movements, and reduce the potential of fires burning into developed areas. Reversing the current wildfire trends in Mediterranean areas and building fire resilient landscapes that sustain landscape production will require integrated strategies that consider the myriad land uses and disturbance processes that shape fuel mosaics and resulting fire behavior.

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Appendix A. Alternative fuel treatment possibilities on torching stands excluded for prescribed fire treatments

We explored the possibility of implementing alternative treatments and obtaining revenue from silvicultural treatments on treatment units excluded from prescribed fire implementation due to high torching probability values (CFB > 0.1 treatment threshold). We first estimated timber stocking in terms of basal area ($m^2\ ha^{-1}$) for the different treatment units using a LiDAR 20 m resolution grid available for Catalonia (icgc.cat). Then, we classified the forest stands considering the slope of the terrain (percentage) and the average dominant height (m). We considered three slope classes (<30%, 30–60%, >60%) in order to account for the potential limitations for tree harvesting and the implementation of fuel mastication treatments (Visser and Stampfer, 2015). Likewise, we considered three dominant tree height classes (h_0; <7.5 m, 7.5–10.5 m, >10.5 m) according to existing guidelines for intermediate site index, even-aged Aleppo pine stand management for increasing fire resistance and commercial timber production (Beltrán et al., 2011). Finally, we cross-tabulated slope and dominant height classes to estimate timber extractions, considering post-thinning basal areas of 19 and 20 $m^2\ ha^{-1}$ for high pole-stage ($h_0 = 7{,}5–10{,}5$ m) and timber stage ($h_0 > 10{,}5$) stands respectively.

Important portions of the landscape were excluded for prescribed fire implementation to prevent negative impacts on the overstory (Fig. 9A). In total, excluded treatment units (n > 15,000) accounted for the 19% of the forest land within the study area (about 25,000 ha). In order to address harvesting mechanization possibilities, characterize type of thinning products and quantify extracted timber amounts, we cross-classified these forest stands for management meaningful dominant height and slope intervals that allowed the characterization into nine major combinations (Fig. 1A). Low pole-stage stands ($h_0 < 7.5$ m) account for the bulk of excluded units (17,257 ha), here treatments do not have any commercial interest, and manual pre-commercial thinning plus mastication treatments are only feasible on the lowest slope class areas (<30%; 9534 ha) and would represent a substantial economic outlay (>1500 € ha^{-1}). High pole-stage stands ($h_0 = 7.5–10.5$ m) cover significant portions of the landscape (2682 ha) and might represent some interest for biomass and paper pulp in low slope (<30%) and overstock stands (basal area > 19 $m^2\ ha^{-1}$). Specifically, around 3300 m^3 would be extracted from thinning treatments on these areas, with an approximate market value of 1–3 € m^{-3} (biomass and or paper pulp destination). In addition, it should be noted that 2532 m^3 correspond to thinning products from only 155 ha where extractions are >10 $m^3\ ha^{-1}$ basal area. The timber stage forest stands have the highest commercial interest (>15 € m^{-3}), even in the intermediate slope areas, but only occupy 84 ha in total. Here, extractions from all stands (<60% slope) would vary depending on the thinning weight from the low of 1669 (for a 20 $m^2\ ha^{-1}$ basal area on stand after low thinning) to 5951 m^3 (for 10 $m^2\ ha^{-1}$ basal area on stand after a dissemination cut). Cliffs and steep slopes areas (>60%; 465 ha) were excluded for tinning treatments.

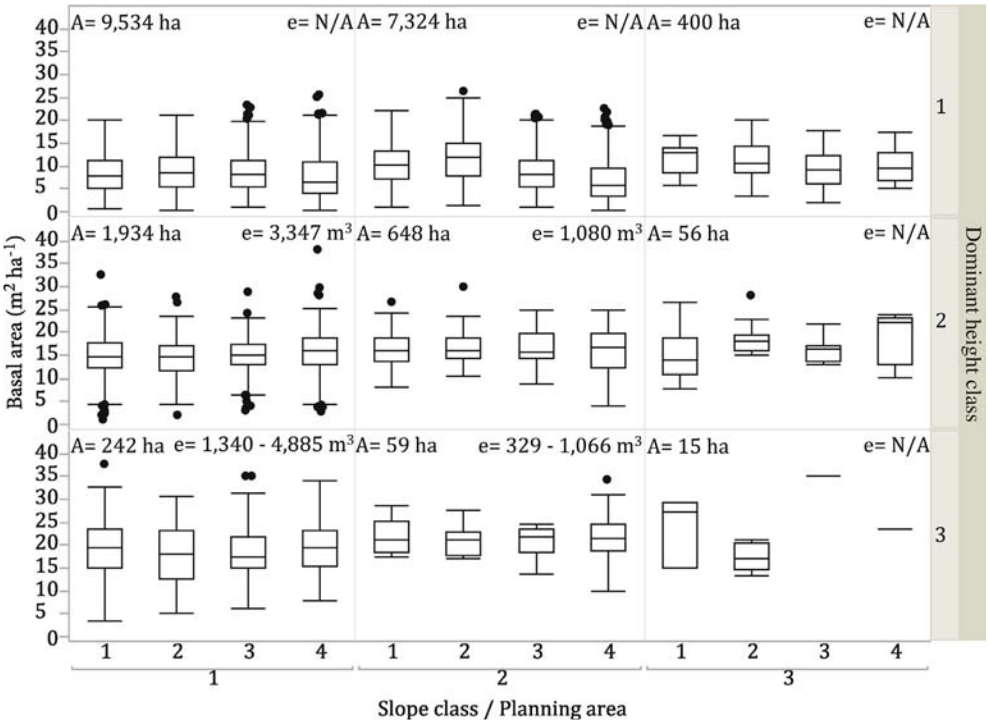

Fig. 1. Timber stocking on treatment units excluded for prescribed fire implementation (Fig. 7A). We explored thinning possibilities on these treatment units (n = 15,168) to identify areas where potential revenue from extractions (i.e., biomass, paper pulp and packaging timber) could partially or totally cover the cost required for stand preparation for complementary treatments (i.e., low-pruning, thinning and mastication or entire tree extraction). High resolution (20 m) LiDAR derived basal area (m^2 ha^{-1}) and stand height (m) grids were used to characterize forest stands (icgc.cat). We considered three slope classes (1 ≤ 30%, 2 = 30–60%; 3 ≥ 60%) (Visser and Stampfer, 2015) and dominant height classes (1 ≤ 7.5 m, 2 = 7.5–10.5 m; 3 ≥ 10.5 m) (Beltrán et al., 2011) to account for main technological factors on timber harvesting and facilitate the estimation of the type of materials and quantities obtained in thinning. The numbers in the boxes indicate the area (A; top-left) and extractions from thinning (e; top-right) within the study area, for the treatment units within all planning areas corresponding to the respective slope and height cross-classification.

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Wildfire spread, hazard and exposure metric raster grids for central Catalonia

Fermín J. Alcasena [a,*], Alan A. Ager [b], Michele Salis [c,d], Michelle A. Day [e], Cristina Vega-Garcia [a,f]

[a] *Agriculture and Forest Engineering Department (EAGROF), University of Lleida, Alcalde Rovira Roure 191, 25198 Lleida, Catalonia, Spain*
[b] *USDA Forest Service, Pacific Northwest Research Station, Western Wildland Environmental Threat Assessment Center, 3160 NE 3rd Street, Prineville, OR 97754, USA*
[c] *National Research Council, Institute of Biometeorology (CNR-IBIMET), Regione Baldinca, 07100 Sassari, Italy*
[d] *Euro-Mediterranean Center on Climate Change (CMCC), IAFES Division, Via Enrico De Nicola 9, 07100 Sassari, Italy*
[e] *Oregon State University, College of Forestry, Forest Ecosystems & Society, 321 Richardson Hall, Corvallis, OR 97331, USA*
[f] *Forest Sciences Centre of Catalonia, Carretera de Sant Llorenç de Morunys km 2, Solsona 25280, Catalonia, Spain*

A B S T R A C T

We provide 40 m resolution wildfire spread, hazard and exposure metric raster grids for the 0.13 million ha fire-prone Bages County in central Catalonia (northeastern Spain) corresponding to node influence grid (NIG), crown fraction burned (CFB) and fire transmission to residential houses (TR). Fire spread and behavior data (NIG, CFB and fire perimeters) were generated with fire simulation modeling considering wildfire season extreme fire weather conditions (97th percentile). Moreover, CFB was also generated for prescribed fire (Rx) mild weather conditions. The TR smoothed grid was obtained with a geospatial analysis considering large fire perimeters and individual residential structures located within the study area. We made these raster grids available to assist in the optimization of wildfire risk management plans within the study area and to help mitigate potential losses from catastrophic events.

Keywords:
Catalonia
Wildfire exposure
Fire transmission
Crown fire activity
Prescribed fires

E-mail address: ferminalcasena@eagrof.udl.cat (F.J. Alcasena).

Specifications Table

Subject area	Environmental sciences, forestry.
More specific subject area	Natural hazards
Type of data	Maps (×4)
How data was acquired	Fire simulation modeling and a geospatial analysis with a geographic information system (GIS).
Data format	Raster grids at 40 m resolution (.tif).
Experimental factors	Extreme fire weather conditions in terms of fuel moisture content and wind speed for the wildfire season dominant scenario (southern wind) were considered to model wildfire spread and behavior. We only considered residential houses within the study area for the transmission analysis, excluding industrial areas, farms and any other structures. Modeling output fire perimeters $<$ 100 ha were excluded from the transmission analysis.
Experimental features	We used FlamMap for wildfire spread and behavior modeling (Finney 2006) and geographic information system software to conduct the transmission and geospatial analysis (ArcMap version 10.1). ArcFuels was used to create ensemble landscape input data for fire modeling (Ager et al., 2011), and the Fire Family Plus program was used to process weather data (Bradshaw and McCormick, 2000).
Data source location	All the landscape file fire modeling input data (topography, surface fuels and canopy metrics) corresponded to the Bages County in central Catalonia (northeastern Spain) plus a 6 km buffer. We used hourly meteorological data (1998 to 2016 records) from the Castellnou de Bages automatic weather station (U4 station reference, Longitude 1.832°N and Latitude 41.842°E) to characterize the fire weather modeling scenario.
Data accessibility	The repository of the University of Lleida (**URL**): http://hdl.handle.net/10459.1/60357
Related research article	Alcasena FJ, Ager AA, Salis M, Day MA, Vega-Garcia C. Otimizing prescribed fire allocation for managing fire risk in central Catalonia. Sci Total Environ. 2018 4:872-885.

Value of the Data

- We provide spatially-explicit value grids for major wildfire risk causative factors in Bages County, central Catalonia (northeastern Spain).
- The raster grids provide quantitative value data to assist ongoing fuels management programs aiming at reducing wildfire risk efficiently.
- The node influence grid (NIG) identifies high fire activity cells or pixels on the landscape (strategic areas) where fuel treatments restrict large fire potential.
- The crown fraction burned (CFB) grid provides data on wildfire effects to the overstory, related to tree mortality and crown fire activity. We generated CFB grids fore extreme fire weather and prescribed fire conditions. The data provide valuable information to prescribe fuel treatments on forested areas.
- Fire transmission to residential houses (TR) provides quantitative exposure data for large fires spreading long distances across the landscape and defines the scale of risk to communities.

1. Data

The Node Influence Grid (NIG) is a raster output that quantifies for each pixel the number of nodes in fire-flow direction from that cell onwards exerting influence on the fire pathways, as the logarithm (ln) of the number of nodes burning as a result of burning through that particular node [2]. The higher the frequency, the stronger the influence is on fire pathways. The log-transform is required to facilitate the visualization of details, since the node counts can range over 4 or 5 orders of magnitude (e.g. from 1 to 10^4). Node influence is highly dependent on fire weather conditions (i.e., wind speed, wind direction and fuel moisture content) and the arrangement of fuel on the landscape. Details of the geospatial data:

- Node influence grid for extreme fire weather scenario (**NIG.tif**). Units= ln of number of nodes. Resolution= 40 m. Coordinate system= ETRS89 UTM 31N.

Crown Fraction Burned (CFB) indicates the degree of potential crown fuel consumption as a proportion of the total number of tree crowns (fraction between 0 and 1), and indicates the probable type of fire activity [1]. The fire types can range from surface fire (< 0.10) to continuous crown fire (> 0.90), while intermediate values represent a scaled value of intermittent crown fire. Crown fire activity is calculated independently of any neighboring cells, and does not consider fire front spreading direction in the calculations. Details of the geospatial data:

- Crown fraction burned raster grid for extreme fire weather scenario (**CFB.tif**). Units= fraction between 0 and 1. Resolution = 40 m. Coordinate system= ETRS89 UTM 31N.
- Crown fraction burned raster grid for prescribed fire conditions (**CFB_Rx.tif**). Units = fraction between 0 and 1. Resolution = 40 m. Coordinate system= ETRS89 UTM 31N.

Fire transmission (TR) quantifies the number of residential houses exposed from fires ignited in a particular location. Values ranged between 0 to 417 structures. The raster is a continuous cover smoothed grid generated with a geospatial analysis from values attributed to the large fire (> 100 ha) ignition locations. The result is highly dependent on fire spread duration, fire weather scenario, fuels arrangement and the location of valued assets on the landscape. Details of the geospatial data:

- Fire transmission to residential houses smoothed raster grid (**TR.tif**). Units = number of structures (residential houses) exposed. Resolution = 40 m. Coordinate system= ETRS89 UTM 31N.

2. Experimental design, materials and methods

We modeled with FlamMap [2] the 1) node influence grid (NIG); 2) crown fraction burned (CFB); and 3) fire perimeters required to assess fire transmission. The minimum travel time (MTT) algorithm [3] implemented in the program is used for fire growth modeling, and crown fire calculations are available with different methods [4,5]. The MTT algorithm calculates a two-dimensional fire growth by searching for the set of pathways with minimum fire spread times from the cell corners at an arbitrary resolution [3]. We used landscape data and wildfire season fire weather conditions for fire modeling.

The landscape file is a regular grid containing spatial data for terrain (aspect, elevation and slope), surface fuels and canopy metrics (canopy height, canopy base height, canopy bulk density and canopy cover). We generated a 40 m resolution landscape file of 252,000 ha encompassing Bages County plus a 6 km buffer using ArcFuels [6]. The fire modeling domain was larger than the study area to account for incoming fires from neighboring vicinities. Required terrain and canopy metric data were respectively derived from a 5 m resolution digital elevation model and 20 m resolution forest cover biophysical data grids generated from low density airborne LiDAR (icgc.cat). We obtained the surface fuel model grid from the assignation of standard fuel models [7] to the habitat cartography of Catalonia considering species composition, vegetation cover percentage, average shrub height and

Table 1
Fire weather data used in fire modeling in Bages County. Extreme fire weather conditions (97th percentile) were used for large fire event modeling in the study area. Weather scenarios were generated with Fire Family Plus [8] using as reference 1998 to 2016 historic hourly data records from U4-Castellnou de Bages automatic weather station (meteo.cat). We considered the dominant wildfire season southern wind direction (180°) for fire modeling [10].

Fire weather conditions	**Fuel moisture content** (%)					**Wind scenario**
	1-h	10-h	100-h	Woody	Herbaceous	Speed (km h^{-1})
Extreme (97th percentile)	7	8	11	60	20	24
Prescribed fire	12	13	15	100	60	10

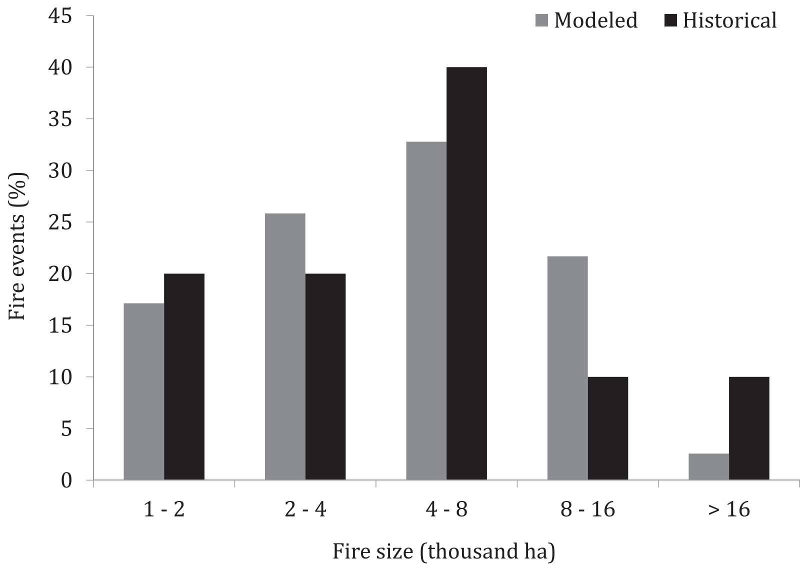

Fig. 1. Historical and modeled fire size distributions in Bages study area. We considered historical large fire events above a 1000 ha threshold [10] to replicate fire size distributions with fire simulation modeling. The fire modeling duration was set to 8 h and weather conditions corresponded to a southern wind direction and 97th percentile wind speed (Table 1). Historical and modeled average fire sizes were respectively 6,025 ha and 5,761 ha. For extreme fires burning for multiple days and spreading out of the study area we used the blow-up episodes burning inside the Bages County (i.e., > 10^4 ha 4th July 1994 Bages fire).

species biogeographic locations on habitat polygon attributes (2° edition 2008/2012 version; mediambient.gencat.cat).

We used extreme fire weather conditions to emulate historical blow-up events overwhelming fire suppression capabilities in the study area. Specifically, we considered historical wildfire season 97th percentile conditions in terms of winds and fuel moisture content. In the study area the wildfire season is concentrated in the month of July, when large fires (> 100 ha) account for 90% of burned area (1983 to 2014; mapama.gob.es). Hourly relative humidity, temperature, wind speed, wind direction, precipitation and solar radiation data from the Castellnou de Bages automatic weather station were used (1998 to 2016 records, U4 station reference; meteo.cat) to characterize extreme fire weather conditions with Fire Family Plus [8]. We obtained fuel moisture content data from 97th percentile ERC-G conditions [9], and 97th percentile wind speed corresponding to a predominantly southern direction (Table 1). Also, we simulated CFB for prescribed fire burn window conditions to assess the potential negative impacts of using fire to treat dense unmanaged forests. To carry out prescribed fire modeling we considered steady and persistent wind speed weather, and mild spring moisture content conditions. At these conditions, we can reduce surface and actively growing ladder fuels (i.e., shrubs and advanced regeneration), while deeper duff and higher soil moisture help to protect dominant tree root systems from fire damage.

Fire modeling was conducted at 40 m resolution under constant fire weather conditions, fuel moisture content and wind (Table 1). In total, we simulated 10,000 fires from random ignitions within the fire modeling domain, which provided the same number of fire perimeters (shapefile polygons) attributed with their respective fire ignition coordinates. The fire modeling duration of 8 h resulted in average fire size and distribution that resembled historical large fire events (Fig. 1).

To generate the TR grid, we first intersected the large fire perimeters (n=6,816 fires $>$100 ha) with structure location centroids within the study area (n=23,633 structures), to then assign the number of structures exposed to wildfire to ignition locations [11]. Small fires do not substantially contribute to the burned area and were excluded to assess fire transmission. Finally, we used exponential kriging geostatistical methods (radius= 3000 m) to create a 40 m resolution smoothed TR grid from ignitions attributed with the number of exposed structures.

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Transparency document. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.dib.2017.12.069.

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Metodoloía de evaluación del riesgo de incendios forestales y priorización de tratamienos multifuncionales en paisajes Mediterráneos

F.J. ALCASENA-URDÍROZ[1]*, C. VEGA-GARCÍA[1, 2], A.A. AGER[3], M. SALIS[4, 5], N.J. NAUSLAR[6, 7], F.J. MENDIZABAL[8], R. CASTELL[9]

[1]Departament d'Enginyeria Agroforestal, Escola Tècnica Superior d'Enginyeria Agrària, Universitat de Lleida, Alcalde Rovira Roure 191, 25198 Lleida, España.

[2]Centre Tecnològic Forestal de Catalunya, Carretera de Sant Llorenç de Morunys km 2, 25280 Solsona, España.

[3]USDA Forest Service, Rocky Mountain Research Station, 72510 Coyote Road Pendleton, 97801 Oregon, USA.

[4]Euro-Mediterranean Center on Climate Change, IAFES Division, Via Enrico De Nicola 9, I-07100 Sassari, Italia.

[5]National Research Council, Institute of Biometeorology, Regione Baldinca, 07100 Sassari, Italia.

[6]Cooperative Institute for Mesoscale Meteorological Studies, University of Oklahoma, Norman, 73019 Oklahoma, USA.

[7]NOAA/NWS/NCEP Storm Prediction Center, Norman, 73019 Oklahoma, USA.

[8]Servicio de Riqueza Territorial y Tributos Patrimoniales, Gobierno de Navarra, Carlos III nº 4, 31002 Pamplona, España.

[9]Bomberos de Navarra, Agencia Navarra de Emergencias, Calle Aoiz 35 bis 3º, 31004 Pamplona, España.

RESUMEN. *En las regiones mediterráneas, el efecto combinado del éxodo rural, la falta de gestión forestal y las políticas de supresión de incendios han contribuido notablemente en el aumento de la carga y continuidad de los combustibles forestales sobre extensas áreas. El resultado es una creciente incidencia de incendios forestales que supera la capacidad de extinción. Debido a una limitada disponibilidad de recursos económicos para la gestión del paisaje, resulta necesario priorizar la protección de los bienes con una expectativa de pérdida elevada y el tratamiento del combustible en puntos estratégicos para contener los incendios que impactan en núcleos urbanos. Este estudio se desarrolla en el Valle de Juslapeña (Navarra, España) para demostrar la priorización de actuaciones en la gestión de combustibles. En el área de estudio, los frecuentes y grandes incendios forestales han causado notables daños en el patrimonio forestal y los bienes de las comunidades rurales. Primero, se generó la cartografía de riesgo de incendios para los bienes de elevado valor, a continuación, se diseñó el mosaico óptimo de tratamientos dentro de la cuenca de exposición en base a la exposición de las masas arboladas y la transmisión a viviendas residenciales. A su vez, se identificaron los rodales capitalizados en existencias donde las extracciones podrían abastecer las necesidades de la población local o costear parcialmente el coste de los tratamientos. Según se observa, las mayores pérdidas se obtuvieron en las viviendas localizadas al sur del área de estudio debido a su elevada probabilidad de quema. Los incendios iniciados fuera del área de estudio también afectaron a las viviendas residenciales y, por tanto, la extensión de los planes de gestión de incendios debe ser ajustada considerando el origen y la escala del riesgo en los núcleos urbanos. La metodología que se presenta en este estudio puede ser adaptada a la gestión multifuncional de cualquier otra región mediterránea con un elevado riesgo de incendios.*

Forest fire risk assessment and multifunctional fuel treatment prioritization methods in Mediterranean landscapes

ABSTRACT. *In Mediterranean areas, the combined effects of the rural exodus, lack of forest management, and fire suppression policies have substantially contributed to increased forest fuel loadings and continuity over large areas. The result is a growing incidence of wildfires that exceed fire suppression capacity. Economic resources for landscape management are limited, and thus they must be prioritized towards the protection of valued assets where there is a high expectation of loss and the fuel treatments on strategic locations that restrict fires spreading into communities. We completed a case study in the Juslapeña Valley (Navarra, Spain) to demonstrate prioritization of fuel management activities. The study area has frequent and large forest fires that have caused significant damage to forest values and assets in rural communities. We first generated a wildfire risk map for valued assets, and then designed the optimal treatment mosaic within the community fireshed considering the wildfire exposure to forestlands and fire transmission to residential housing. We also identified overstocked stands where the timber or firewood production might supply the needs of local communities and partially cover the treatment cost. We found that the highest economic losses were obtained in residential houses located in the southern portion of the study area, mainly due to a higher burn probability. Fires ignited outside of the study area also exposed communities, and thus the extent considered in wildfire management plans needs to be adjusted to reflect the source and scale of risk to communities. The assessment framework presented in this study can be adapted to the multi-functional forest management in any fire-prone Mediterranean region elsewhere.*

Palabras clave: evaluación de riesgo, optimización espacial, tratamiento de combustibles, gestión multifuncional, cuencas de exposición.

Key words: risk assessment, spatial optimization, fuels treatment, multi-functional management, community fireshed.

*****Correspondencia: Fermín J. Alcasena Urdíroz, Departament d'Enginyeria Agroforestal, Universitat de Lleida, Alcalde Rovira Roure 191, 25198 Lleida, España. E-mail: ferminalcasena@eagrof.udl.cat

1. Introducción

El creciente incremento en la carga y continuidad de combustibles forestales en los ecosistemas europeos de clima templado y mediterráneo durante las últimas décadas ha facilitado la ocurrencia de grandes incendios forestales, constatando la rápida evolución del régimen de incendios a un nuevo régimen post-industrial sometido a las condiciones meteorológicas extremas (Pausas y Fernández-Muñoz, 2012; Seijo y Gray, 2012). Factores como el éxodo rural, la falta de gestión forestal y una política de supresión total de incendios son los principales factores que explican la creciente acumulación de biomasa en el territorio y la consecuente desaparición del paisaje cultural en mosaico donde la discontinuidad de combustibles limitaba la propagación del fuego (Cervera *et al.*, 2016; Poyatos *et al.*, 2003). En los países europeos meridionales en torno a 48.000 incendios forestales queman anualmente unas 448.000 hectáreas de media (1980 a 2015), situando a España como el país más afectado en esta región con el 45% del área quemada (San-Miguel-Ayanz *et al.*, 2017). A su vez, son los escasos (<10%) pero grandes incendios forestales (>100 ha) son los responsables de la mayor parte del área quemada (>60%) en el periodo estival y están asociados a condiciones meteorológicas de sequía acumulada, fuertes vientos y bajas humedades relativas (San-Miguel-Ayanz *et al.*, 2013). Estos incendios forestales recorren largas distancias (> 10 km), presentan comportamientos extremos que superan la capacidad de extinción (fuegos activos de copas con múltiples saltos de fuego), están asociados a episodios de simultaneidad, y frecuentemente impactan núcleos urbanos habitados localizados en la interfaz urbano forestal (Castellnou y Miralles, 2009). Además, la mayor parte del área quemada resulta severamente afectada y se requiere de costosos trabajos de restauración y monitoreo post-incendio encaminados a minimizar los efectos negativos en los bienes socioeconómicos y de interés natural (Moya *et al.*, 2014; Prats *et al.*, 2014). En cuanto a la causalidad se refiere, el origen de las igniciones es mayoritariamente antrópico y los incendios de rayo se concentran en cadenas montañosas donde difícilmente superan el 20% del total (Costafreda-Aumedes *et al.*, 2016; Rodrigues y De la Riva, 2014).

El *riesgo de incendios* se define como la expectativa de pérdida o beneficio en un bien o servicio que es afectado por el fuego, es espacialmente explícito y se puede evaluar cuantitativamente (Ager *et al.*, 2010; Finney, 2005; Scott *et al.*, 2013). Está integrado por la probabilidad de quema y sus consecuencias, siendo las consecuencias el resultado de la susceptibilidad del bien o servicio afectado por el fuego a una determinada intensidad. La *exposición* no contempla los efectos del fuego y se obtiene a partir de la intensidad y probabilidad de quema (Miller y Ager, 2013). Los *efectos del fuego* se pueden aproximar con funciones de susceptibilidad o modelos de mortalidad en el caso de especies arboladas (Fernandes *et al.*, 2012; Thompson *et al.*, 2011). La *transmisión* consiste en asignar a las coordenadas o polígono de origen del fuego los bienes o servicios expuestos (por ej., número de viviendas) dentro de su perímetro y nos permite delimitar el contorno y extensión de las *cuencas de exposición* (“firesheds”). La cuenca de exposición por tanto es el área de terreno donde las igniciones ocurridas en condiciones meteorológicas típicas asociadas al periodo de incendios originan fuegos que, en propagación libre, alcanzan un determinado bien de interés natural o elevado valor económico (por ej., núcleo urbano o hábitat de protección prioritaria) (Ager *et al.*, 2016a; Thompson *et al.*, 2013). Por otra parte, la *zona de ignición en viviendas* (“home ignition zone”) es el entorno próximo a las estructuras (buffer de 30-60 m) donde la intensidad del fuego determina en mayor medida los daños y su pérdida o destrucción (Cohen, 2008). Los *puntos estratégicos de gestión* (PEG) en el paisaje son aquellas parcelas o rodales a tratar preferentemente donde la reducción de combustibles facilita la labor de los medios de extinción (aplicación del fuego táctico) y contiene significativamente el potencial del gran incendio forestal (Ager *et al.*, 2011; Costa *et al.*, 2011). Su localización y extensión se puede determinar en base al criterio experto, a partir del estudio de incendios históricos (González-Olabarria *et al.*, 2019), o alternativamente mediante el empleo de simuladores y modelos de optimización espacial (Ager *et al.*, 2016b; Finney, 2007).

Los simuladores nos permiten predecir el comportamiento y la propagación de los grandes incendios forestales (Arca *et al.*, 2007; Jahdi *et al.*, 2016; Salis *et al.*, 2016a). A su vez, algoritmos altamente eficientes (Finney, 2002) nos permiten saturar el paisaje con miles de igniciones ($>10^4$ igniciones) y determinar en alta resolución (< 50 m de píxel) la intensidad del fuego y su probabilidad de quema (Alcasena *et al.*, 2015). Actualmente, el acceso a los datos requeridos para la simulación de incendios es cada vez mayor y los avances tecnológicos permiten caracterizar con precisión las variables geoespaciales del paisaje (topografía, modelos de combustible de superficie y parámetros de copas) en alta resolución (González-Olabarria *et al.*, 2012; Marino *et al.*, 2016). A partir de los registros en estaciones meteorológicas resulta posible determinar localmente cuáles son las condiciones meteorológicas extremas asociadas a los grandes incendios forestales y caracterizar así los escenarios más frecuentes durante el periodo estival (Bradshaw y McCormick, 2000; Duane y Brotons, 2018). Además, a partir de las igniciones históricas se pueden generar modelos de ocurrencia que nos permitan replicar el patrón de igniciones requerido como dato de entrada en las simulaciones (Alcasena *et al.*, 2017; Alcasena *et al.*, 2016a).

La mayoría de los estudios previos desarrollados en ambientes mediterráneos se limitan a cuantificar el riesgo y la exposición de incendios (Alcasena *et al.*, 2016b; Palaiologou *et al.*, 2018; Salis *et al.*, 2013), y muy pocos evalúan las diferencias entre posibles estrategias (configuraciones espaciales e intensidades de tratamientos) encaminadas a mitigar las pérdidas por incendios (Oliveira *et al.*, 2016; Salis *et al.*, 2016b). Debido a la escasez de recursos económicos disponibles para la ejecución de trabajos de prevención y la existencia de objetivos contrapuestos, la optimización espacial integra un complejo análisis que nos permite identificar una solución de compromiso con múltiples objetivos en las parcelas a tratar (Vogler *et al.*, 2015). Esto es posible ya que el tratamiento de combustibles (es decir, claras, quemas y desbroces) se pueden compatibilizar con labores de restauración de hábitats de especial interés, mejora de pastos y aprovechamientos forestales (Lasanta *et al.*, 2018). A tal efecto, resulta imprescindible proporcionar los resultados a escala de rodal (es decir, identificación de parcelas a tratar) para facilitar la transferencia de los resultados a los gestores del territorio responsables de la ejecución de los tratamientos (Alcasena *et al.*, 2018). En ocasiones, los trabajos de restauración implican importantes cortas de madera cuando se ejecutan en extensas masas arboladas y las extracciones de madera para la apertura de discontinuidades pueden aportar beneficios económicos que permitan cubrir al menos parcialmente el coste de los tratamientos (Ager *et al.*, 2016b).

Para mitigar el riesgo de incendios es necesario generar paisajes resistentes al fuego, aplicar medidas encaminadas a prevenir igniciones antrópicas, hacer un uso eficiente de los medios de extinción, reducir la susceptibilidad al fuego en los bienes de elevado valor y fomentar una percepción responsable del riesgo en la población local, especialmente en áreas de interfaz urbano forestal (Alcasena *et al.*, 2019; Paveglio *et al.*, 2016). Concretamente, es la gestión de combustibles la medida principal en lo que a la creación de núcleos urbanos y paisajes resistentes al fuego se refiere. A su vez, el tratamiento de combustibles es especialmente útil en entornos próximos a los bienes de interés y para reducir localmente la intensidad del fuego, o en puntos estratégicos del territorio para limitar la propagación de grandes incendios, reduciendo así la probabilidad de quema a gran escala de paisaje. El objetivo del presente estudio es (1) evaluar el riesgo de incendios (es decir, cuantificar la expectativa de pérdidas económicas en caso de gran incendio forestal) usando cartografía de alta resolución para los principales bienes (viviendas y repoblaciones de carácter productor) del Valle de Juslapeña (Navarra, España), e (2) identificar en el paisaje los rodales preferentes a tratar para mitigar la transmisión de incendios a núcleos urbanos, reducir las pérdidas potenciales en masas forestales y maximizar las extracciones en los tratamientos (leñas o madera), conformando con ello una metodología efectiva para la definición en el territorio estudiado de tratamientos multifuncionales y puntos estratégicos de gestión. Finalmente, en base a los resultados obtenidos, se proponen una serie de medidas que permitan la reducir el riesgo de incendios en el área de estudio.

2. Material y métodos

2.1. Esquema general

El esquema general empleado en este estudio integra dos procesos diferenciados, uno para evaluar el riesgo de incendios y otro encaminado a priorizar espacialmente el tratamiento de combustibles (Fig. 1). Ambos requieren resultados cuantitativos procedentes de la simulación de incendios forestales a escala de paisaje. El riesgo de incendios se determinó como la expectativa de pérdida económica en bienes de elevado valor (viviendas residenciales y pinares productores). En base a los objetivos fijados en la gestión forestal (y la priorización establecida entre objetivos) se diseñaron diferentes mosaicos de tratamientos multifuncionales en el área de planificación (cuenca de exposición) con un modelo de optimización, para finalmente analizar su correspondencia espacial. El trazo de línea discontinuo presenta un bucle de retorno que permitiría evaluar el efecto de los tratamientos estratégicos preventivos (es decir, soluciones multifuncionales) en la disminución de pérdidas económicas. La metodología empleada en este estudio puede adaptarse a las necesidades locales (objetivos de la gestión forestal) y aplicarse en cualquier otro lugar donde los incendios suponen un riesgo para los bienes y servicios.

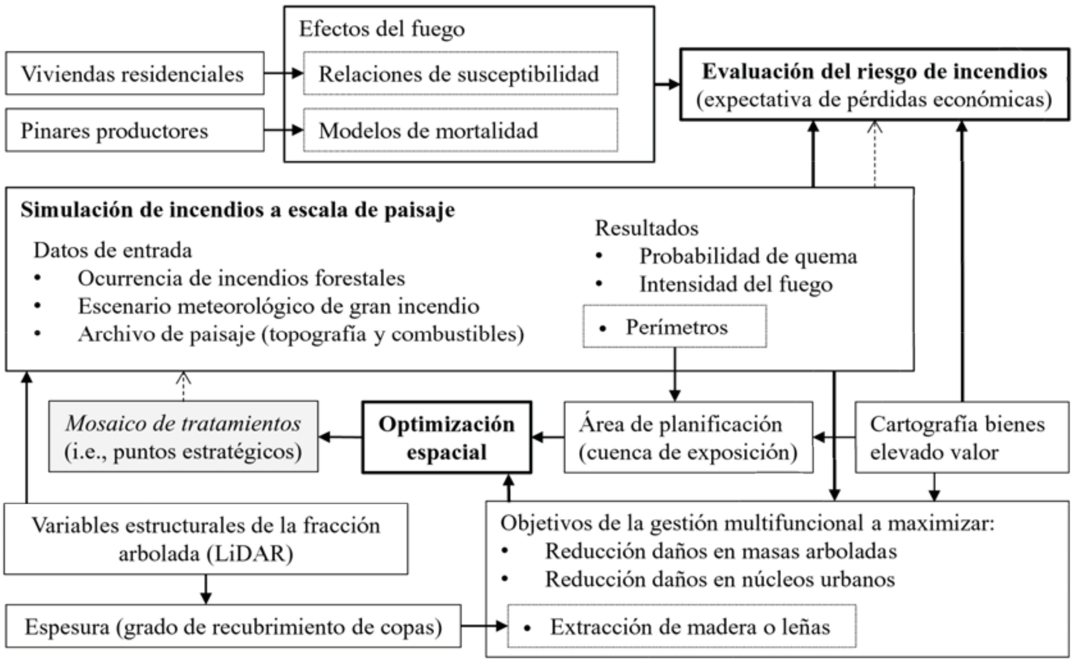

Figura 1. Esquema general de evaluación de riesgo y optimización espacial de tratamientos aplicado en el estudio. Los resultados de la simulación de incendios se emplean para cuantificar el riesgo de incendios, determinar el área de planificación y facilitar la priorización de tratamientos (es decir, métricas cuantitativas asociadas a los objetivos de gestión).

2.2. Área de estudio

El área de estudio se corresponde con el término municipal del Valle de Juslapeña (Navarra, España), tiene una extensión de 3163 ha y se encuentra situado en el límite norte de la Cuenca de Pamplona (Fig. 2A). La población es de 548 habitantes y se encuentra distribuida en 14 núcleos urbanos. El clima es mediterráneo transicional, con precipitaciones medias anuales en torno a los 1000 mm y un periodo de sequía estival de unos 3 meses, con temperaturas máximas diarias por encima de los 30º. Los campos de cultivo de cereal en secano ocupan los suelos arcillosos y profundos de los fondos de valle. Los pastos mesoxerófitos (*Bromus erectus* Huds. Y *Brachypodium pinnatum* L.) y las orlas de matorral (*Juniperus communis* L., *Prunus spinosa* L., *Buxus sempervirens* L. y *Genista scorpius* L.) se sitúan entre los campos de cultivo y zonas arboladas, llegando a ocupar importantes extensiones en el caso de algunos pastos comunales. Aunque el roble pubescente (*Quercus pubescens* Willd.) es la especie arbórea dominante en los bosques procedentes de regeneración natural, también existen múltiples repoblaciones de pino laricio de austria (*Pinus nigra* Arn. ssp. *nigra*). En el límite norte, son los hayedos (*Fagus sylvatica* L.) los que ocupan las exposiciones norte y cotas más elevadas. Los cambios en los usos del suelo durante los últimos 50 años evidencian el incremento de la superficie forestal arbolada, un incremento en la cobertura de matorrales y la reducción de pastizales o zonas abiertas (Fig. 2B y 2C). En los robledales se puede apreciar un claro incremento de la fracción de cabida cubierta así como su expansión en pastizales limítrofes. Las parcelas de cultivo se concentran ahora en los fondos de valle y gran parte de las antiguas roturas difícilmente mecanizables fueron repobladas con pino laricio. Únicamente los pastos comunales habilitados para su aprovechamiento en extensivo (es decir, que disponen de cierres ganaderos y abrevaderos) mantienen las zonas abiertas, en el resto de los casos el incremento de la cobertura de matorral y regenerado natural es muy notable. Estas nuevas masas en edad de monte bravo y latizal bajo son muy vulnerables a incendios debido a la elevada carga y continuidad de combustibles. En el área de estudio los incendios se concentran en verano y las causas principales son la quema de pastos y matorral, así como la quema de rastrojos de cereal (Fig. 3).

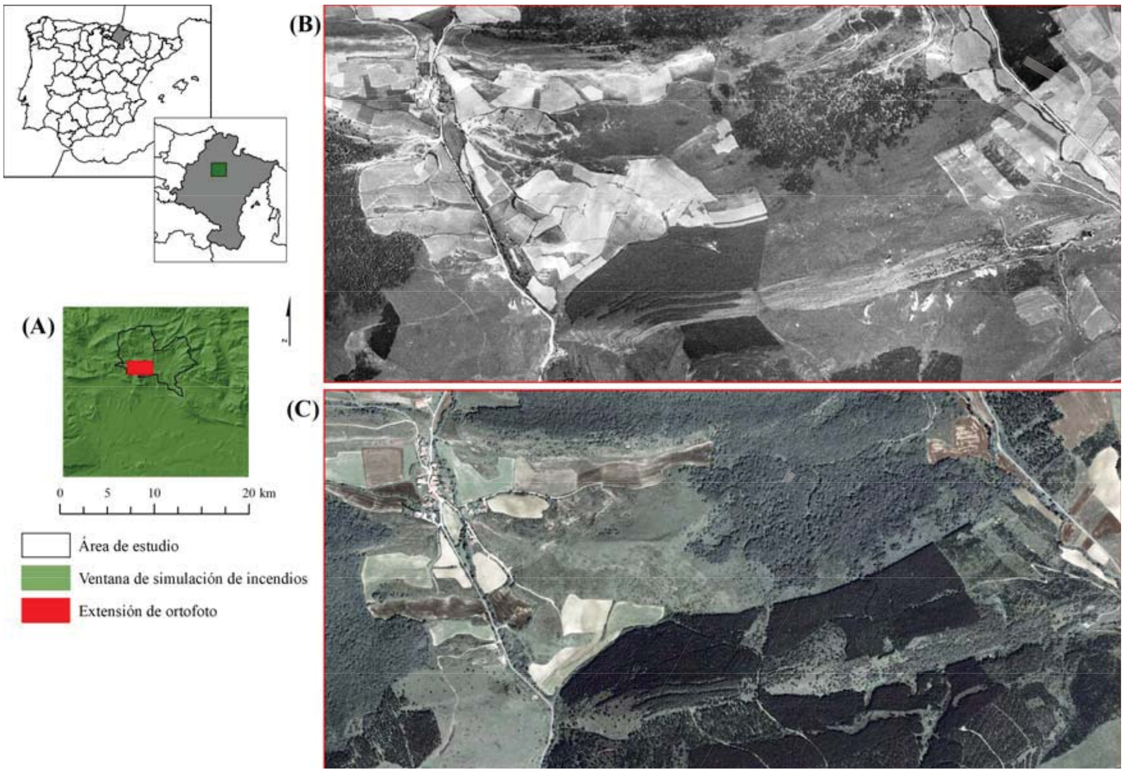

Figura 2. Cambios en el uso del suelo en el área de estudio (A). Las diferencias entre las ortofotos de 1956 (B) y 2016 (C) evidencian el rápido incremento en la carga y continuidad de los combustibles durante los últimos 50 años.

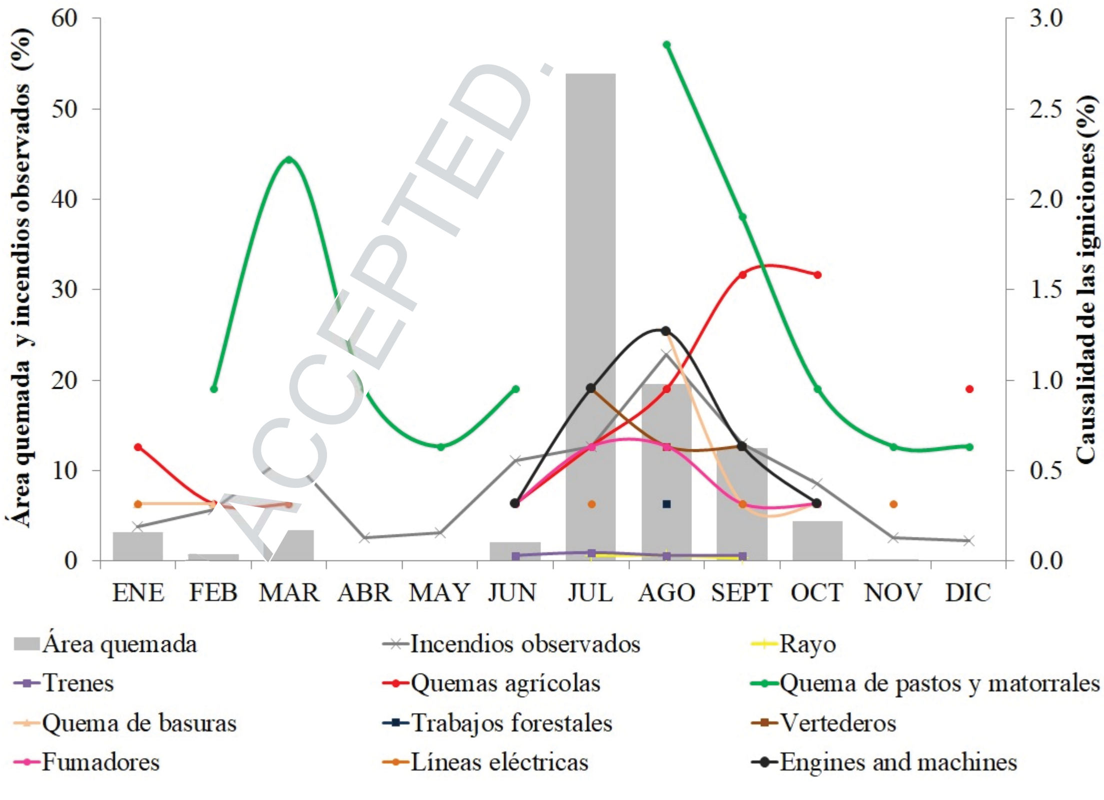

Figura 3. Estacionalidad y causalidad de incendios en el área de estudio. La mayor parte del área quemada se concentra durante el periodo estival y las quemas (pastos y agrícolas) son la causa más frecuente (1985 a 2013).

2.3. Ocurrencia de incendios

Los modelos de ocurrencia permiten predecir dónde es más probable que se inicie un incendio y generar un patrón de igniciones que posteriormente pueda ser empleado como dato de entrada en la simulación de incendios (listado de igniciones con sus coordenadas). Existen diferentes métodos para la generación de estos modelos, siendo los modelos de aprendizaje automático, la regresión logística y las redes neuronales los más empleados (Martínez *et al.*, 2009; Rodrigues y De la Riva, 2014; Vega-García *et al.*, 1995). Para ello, se emplean los incendios históricos (coordenadas) y una muestra de puntos (coordenadas) de no-igniciones, a partir de los cuáles se extraen del territorio una serie de variables de carácter geoespacial con las que se construye el modelo. Habitualmente, las variables asociadas a la actividad humana y sus transformaciones del paisaje son las que más contribuyen a explicar estos modelos, puesto que las actividades antrópicas son causantes de la mayor parte de las igniciones conocidas en ambientes Mediterráneos (Costafreda-Aumedes *et al.*, 2018). Para este estudio se consideró un área de ocurrencia de 36.000 ha abarcando el Valle de Juslapeña y los municipios más próximos de su entorno. Se contabilizaron un total de 200 igniciones (1985 a 2013) (MAAyMA, 2015), que además de otras 200 no-igniciones localizadas aleatoriamente, fueron empleadas para extraer una muestra de las variables geoespaciales correspondientes a usos de suelo, distancia a carreteras, distancia a núcleos urbanos, densidad de población y distancia a líneas eléctricas. Con el empleo de una red neuronal de correlación en cascada (Fahlman y Lebiere, 1990) se generó un modelo de ocurrencia (Alcasena *et al.*, 2017). Al aplicar el modelo en cada uno de los píxeles del área de ocurrencia, se generó un mapa de probabilidad de ignición (valores entre 0 y 1) a 30 m de resolución (Fig. 4).

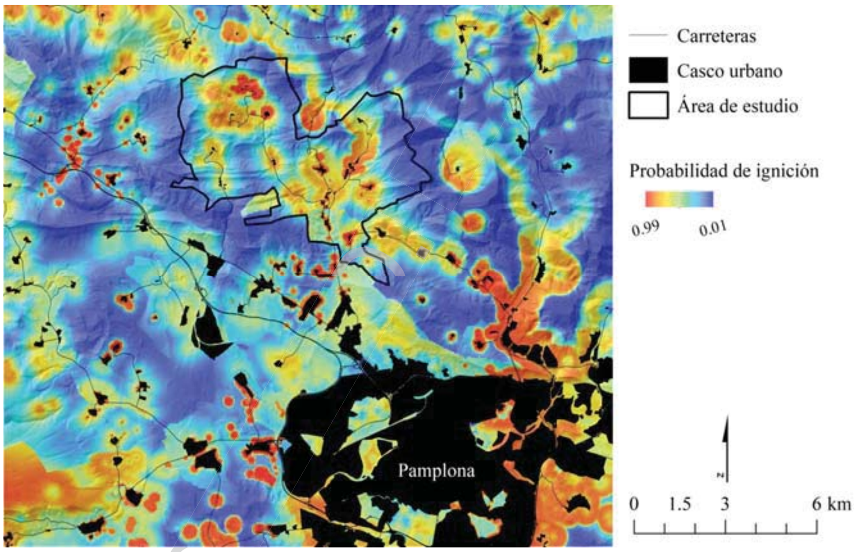

Figura 4. Mapa de probabilidad de ignición en alta resolución (30 m) generado con redes neuronales a partir de la localización de las igniciones históricas (Alcasena et al.*, 2017). El mapa fue empleado para generar un patrón con 10.000 puntos de ignición, dato de entrada para la simulación de incendios.*

2.4. Simulación de incendios

Para la simulación de los incendios se requiere la información geoespacial que caracteriza el paisaje, las condiciones meteorológicas típicas asociadas a eventos extremos (dirección del viento, velocidad del viento y contenido de humedad de los combustibles) y las coordenadas de las igniciones para los incendios que se desean simular. El archivo de paisaje es una retícula regular que contiene para cada pixel la información del terreno (pendiente, elevaciones y exposición), modelos de combustible y los parámetros de copas (altura de copas, densidad de copas, altura de la base de la copa y fracción de cabida cubierta). La información del terreno y los parámetros de copas se obtuvieron a partir de datos LiDAR (ign.es) y el mapa de modelos de combustible se generó mediante la asignación de modelos de combustible estándar (Fernandes, 2009; Scott y Burgan, 2005) a los diferentes usos de suelo, considerando como referencia los polígonos de SIGPAC 2017 (sigpac.navarra.es) y la información descriptiva (es decir, grado de cobertura, especies arbustivas y las clases naturales de edad del estrato arbolado) extraída del mapa de usos de suelo y aprovechamientos de Navarra (año 2012; idena.navarra.es). Para la generación del archivo de paisaje con ArcFuels (Ager *et al.*, 2011) se empleó la extensión del área de ocurrencia (36.000 ha; Fig. 2A) con el objeto de considerar la entrada de incendios iniciados fuera del área de estudio. Las condiciones meteorológicas extremas asociadas a los grandes incendios forestales se determinaron con Fire Family Plus (Bradshaw y McCormick, 2000), a partir de la serie histórica de datos horarios registrados en la estación automática de Pamplona durante los últimos 17 años (meteonavarra.es). Se consideró el percentil 97 como estadístico de referencia para fijar la velocidad del viento y el contenido de humedad (Nelson, 2000) en las simulaciones (Fig. 5). Se obtuvieron un total de cinco escenarios tipo, uno para cada dirección de viento (Alcasena *et al.*, 2017) (Tabla 1). A partir del mapa de ocurrencia se generó un patrón de 10.000 igniciones, listado de igniciones que fue empleado posteriormente como dato de entrada en la simulación de incendios.

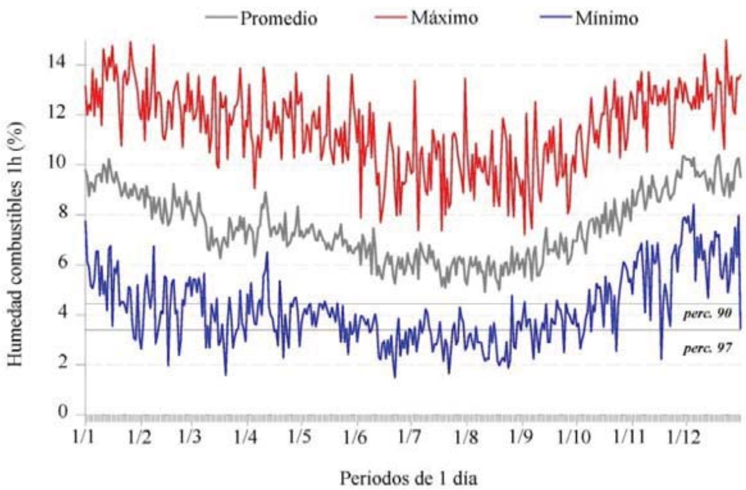

Figura 5. Evolución interanual en el contenido de humedad del combustible fino de 1h de tiempo de retardo en el área de estudio. Los valores fueron calculados (Nelson, 2000) a partir de los datos horarios de precipitación, humedad relativa, temperatura, vientos y radiación solar (1997 a 2015) registrados en la estación automática de la Universidad Pública de Navarra (Pamplona, Navarra). Para la simulación de incendios en el área de estudio se emplearon los valores extremos correspondientes al percentil 97.

Tabla 1. Condiciones meteorológicas asociadas a los grandes incendios forestales en el área de estudio (Alcasena et al.*, 2017). Los datos meteorológicos fueron obtenidos de la estación automática de Pamplona (meteo.navarra.es) y el contenido de humedad fue derivado a partir del percentil 97 del ERC-G (Nelson, 2000). La velocidad del viento corresponde respectivamente al percentil 97 de cada dirección. Se emplearon modelos de combustible estándar (Fernandes, 2009; Scott y Burgan, 2005), que fueron asignados a los diferentes usos de suelo (idena.navarra.es).*

Escenario de viento			**Contenido de humedad (%)**			
				Modelos de combustible		
Dirección (acimut)	**Velocidad (km h^{-1})**	**Probabilidad**	**Categoría de combustible**	GS1, GR5, GR2, GR4, SH6, SH5	TU3, PCL, SH3, GR3	GR1, SH3, TL2, SH8
67,5	32	0,43	1-h	4	6	8
337,5	35	0,28	10-h	5	7	9
45	19	0,17	100-h	8	9	12
180	31	0,06	Vivo herbáceo	20	45	70
22,5	23	0,06	Vivo leñoso	60	85	100

Se realizó una simulación con 10.000 igniciones por cada uno de los 5 escenarios (50.000 incendios en total) con el algoritmo “minimum travel time” (MTT) (Finney, 2002) implementado en el simulador FlamMap (Finney, 2006). Debido a que FlamMap MTT simula todos los incendios con las mismas condiciones meteorológicas (dirección de viento, velocidad de viento y contenido de humedad del combustible constante), se realizó una simulación para cada escenario de viento. El resultado final fue obtenido combinando los resultados de los diferentes escenarios, considerando la probabilidad de cada escenario (Tabla 1). Se fijó una duración en las simulaciones de 6 horas, duración del periodo de propagación activa observado en los grandes incendios históricos en el área de estudio (incendio de Juslapeña en 2009). Todos los píxeles se quemaron al menos una vez y más de 100 veces en promedio. Las simulaciones se realizaron a 30 m de resolución y en propagación libre, sin fijar barreras ni efectos de contención ya que la propagación del frente se considera resistente a los medios de extinción bajo condiciones meteorológicas extremas. Se obtuvieron resultados de salida de probabilidad de quema, intensidad del fuego y perímetros que se detallan a continuación.

La probabilidad de quema o "burn probability" (*BP*) es una trama regular con valores entre 0 y 1, obtenido a partir de la relación entre el número de veces que se quema cada píxel y el número de igniciones empleadas en la simulación. El resultado de BP obtenido en este estudio es un valor de probabilidad condicionado a la ocurrencia de un gran incendio bajo condiciones meteorológicas extremas previamente detalladas (Tabla 1). Además, FlamMap calcula en cada incendio la longitud de llama para cada pixel. Posteriormente, después de simular independientemente todos los incendios a partir de las igniciones indicadas, en cada píxel la intensidad se obtiene como un valor de probabilidad o "flame length probability" (FLP) para cada uno de los 20 niveles o "fire intensity levels" (FILs) de 0,5 m de longitud de llama, siendo para la clase 20 la longitud de llama > 9,5 m (FIL_1 = 0-0,5 hasta FIL_{20} > 9,5 m). De este modo, la intensidad en cada pixel considera la dirección en la propagación del frente (es decir, cabeza, cola y flanco). La suma de lodos los FLP en los 20 niveles es de 1 en cada pixel. Finalmente, también se obtuvieron los polígonos de los perímetros para cada uno de los incendios simulados. Cada perímetro tiene asignado el valor de la superficie quemada y las coordenadas de su ignición. Se obtuvieron un total de 50.000 perímetros, 10.000 por cada escenario. Para los análisis posteriores únicamente se emplearon los perímetros de los grandes incendios (> 100 ha) ya que estos son responsables de la mayor parte del área quemada.

2.5. Efectos del fuego

Los efectos del fuego se pueden cuantificar empleando relaciones de susceptibilidad derivadas del criterio experto o a partir de modelos de mortalidad en el caso de las especies arboladas. En ambos casos, los efectos del fuego (pérdidas o beneficios) se determinan para diferentes niveles de intensidad (principal factor causativo de riesgo) (Scott *et al.*, 2013). La aproximación de los efectos esperados a diferentes niveles de intensidad nos permite integrarlos con los resultados obtenidos en la simulación (probabilidad de quema e intensidad del fuego) y evaluar así el riesgo (Ager *et al.*, 2011). En este estudio se emplearon funciones de respuesta o "response functions" (RFs) (Thompson *et al.*, 2011) para determinar los efectos del fuego en viviendas residenciales. Las pérdidas en las viviendas se establecieron como un porcentaje de su valor total y se determinaron con el método Delphi (Dalkey y Helmer, 1963). El método Delphi es una técnica de comunicación estructurada desarrollada como un método sistemático e interactivo de predicción y basada en un panel de expertos (Lovreglio *et al.*, 2010; Meddour-Sahar *et al.*, 2013). Los resultados fueron obtenidos de encuestas realizadas a responsables de Bomberos de Navarra con experiencia en las labores de extinción de incendios de interfaz urbano forestal en el área de estudio, encuestas en las que se les solicitó que indicasen las pérdidas esperadas como un porcentaje del valor de las viviendas afectadas (de 0% sin daños, a -100% para la destrucción total de la estructura) a diferentes niveles de intensidad (Alcasena *et al.*, 2017) (Tabla 2). La inmensa mayoría de las estructuras en el área de estudio presentan muros de carga de piedra o ladrillo con revestimiento exterior, disponen de persianas (de madera, aluminio o plástico) y las cubiertas son de madera en la mayoría de los casos. Las encuestas se realizaron de modo anónimo en dos etapas, en la primera se asignaron los valores y en la segunda se afinaron los resultados. Por otro lado, con el objeto de determinar los efectos del fuego en las repoblaciones de pino laricio, se empleó un modelo de mortalidad genérico desarrollado para coníferas que considera variables como el espesor de la corteza y altura de copa (Fernandes *et al.*, 2008; Peterson y Ryan, 1986). A partir de datos de inventario disponibles por el Guarderío Forestal se calculó la mortalidad media esperada a nivel de rodal para diferentes niveles de intensidad (Alcasena *et al.*, 2016a) (Tabla 2).

Tabla 2. Funciones de respuesta (RF) para viviendas residenciales (pérdidas con respecto a su valor, en %) y mortalidad (valor medio, en %) post-incendio en repoblaciones de pino laricio para diferentes niveles de intensidad (Alcasena et al.*, 2017; Alcasena* et al.*, 2016a). Los niveles de intensidad (FIL) se especifican para rangos de 0,5 m de longitud de llama. Por encima de 2,5m de longitud de llama la mortalidad en los pinares es del 100%.*

<table>
<tr><th rowspan="3">Bien de elevado valor</th><th colspan="9">Efectos del fuego a diferentes niveles de intensidad (longitud de llama, en m)</th></tr>
<tr><th>FIL$_1$</th><th>FIL$_2$</th><th>FIL$_3$</th><th>FIL$_4$</th><th>FIL$_5$</th><th>FIL$_6$</th><th>FIL$_7$</th><th>FIL$_8$</th><th>FIL$_9$</th></tr>
<tr><th>0 – 0,5</th><th>0,5 – 1</th><th>1 – 1,5</th><th>1,5 – 2</th><th>2 – 2,5</th><th>2,5 – 3</th><th>3 – 3,5</th><th>3,5 - 4</th><th>>4,5</th></tr>
<tr><td>Viviendas</td><td>-10%</td><td>-45%</td><td>-45%</td><td>-45%</td><td>-75%</td><td>-75%</td><td>-75%</td><td>-95%</td><td>-100%</td></tr>
<tr><td>Monte bravo</td><td>0</td><td>-98%</td><td>-100%</td><td>-100%</td><td rowspan="5" colspan="5">-100%</td></tr>
<tr><td>Latizal bajo</td><td>0</td><td>0</td><td>-99%</td><td>-100%</td></tr>
<tr><td>Latizal alto</td><td>0</td><td>0</td><td>5</td><td>-100%</td></tr>
<tr><td>Fustal bajo</td><td>0</td><td>0</td><td>0</td><td>-72%</td></tr>
<tr><td>Fustal alto</td><td>0</td><td>0</td><td>0</td><td>-1%</td></tr>
</table>

2.6. Riesgo de incendios

El riesgo de incendios (Finney, 2005; Scott *et al.*, 2013) se cuantificó como la expectativa de pérdida económica (*eEL*) en caso de gran incendio, mediante el empleo de la siguiente ecuación:

$$eEL = V \times \sum_{i=1}^{20} BP \times FLP_i \times RF_i \qquad (1)$$

donde *BP* es la probabilidad de quema (resultado de simulación), FLP_i es la probabilidad de un fuego para el nivel de intensidad *i* (resultado de simulación), RF_i es la función de respuesta para el nivel de intensidad *i* y *V* es el valor económico del bien afectado por el fuego (es decir, repoblaciones y viviendas residenciales). Los valores económicos de la madera para las diferentes clases de edad (€ × m^3 con corteza en pie) se obtuvieron a partir de los precios medios de adjudicación definitiva alcanzados en las subastas realizadas en el área de estudio (Ayuntamiento de Juslapeña, com. pers. 2014). Se consideraron diferentes valores en base a las dimensiones de los productos y su destino (trituración, embalaje o sierra). Para las viviendas residenciales se emplearon los valores indicativos a efectos de Impuestos de Transmisiones Patrimoniales y Actos Jurídicos Documentados, obtenidos a partir de datos catastrales (catastro.navarra.es; Decreto Foral 334/2001 de 26 de noviembre). En el caso de las repoblaciones se generó una cartografía de riesgo con valores de pérdidas económicas esperadas en los rodales a 30 m de resolución, considerando las existencias (m^3 con corteza ha^{-1}) para cuantificar las pérdidas totales. Para las viviendas individuales se consideró el valor medio de *eEL* en todos los píxeles dentro de la zona de ignición en viviendas (buffer de 60 metros) (Cohen, 2008).

2.7. Optimización espacial de tratamientos

Para la optimización espacial de los tratamientos y la identificación de los polígonos o rodales a tratar preferentemente se empleó el programa Landscape Treatment Designer (LTD) (Ager *et al.*, 2016b). Este programa permite configurar el mosaico de tratamientos que maximiza la contribución del área tratada con respecto a uno o varios objetivos, para satisfacer, en medida de lo posible, las necesidades de los diferentes agentes sociales (es decir, soluciones multiobjetivo). En este estudio se establecieron tres objetivos para los tratamientos: (1) la reducción del riesgo de incendios en masas arboladas (pinares y robledales), (2) la reducción del riesgo de incendios en núcleos urbanos, y (3) la obtención de leñas o madera en el tratamiento de combustibles. A cada uno de los objetivos se le asocia un parámetro cuantitativo que permita captar el gradiente espacial existente en el territorio e identificar así los rodales o polígonos estratégicos con valores más elevados. En cada parcela o rodal a tratar se asume un grado de cumplimiento para cada objetivo proporcional al valor cuantitativo del parámetro asociado (es decir, mayor grado de cumplimiento de los objetivos en rodales con valores más elevados) con respecto al total en el área de estudio.

a) Parámetros cuantitativos asignados a los objetivos

Con la finalidad de identificar las masas arboladas (pinares y robledales) que presentan unas mayores pérdidas potenciales en caso de gran incendio forestal se empleó el parámetro de probabilidad de quema en alta intensidad o "high intensity burn probability" (*HIBP*) (Lozano *et al.*, 2017) , que fue calculado mediante la siguiente ecuación:

$$HIBP = \sum_{i=6}^{20} FLP_i \times BP \qquad (2)$$

El valor de HIBP se calculó para cada píxel a 30 m de resolución empleando los resultados de probabilidad e intensidad de las simulaciones. Por encima de 2,5 m de longitud de llama en fuegos de superficie (≥ $FLP_{6;}$ Tabla 2) la mortalidad es muy elevada o total.

Para la reducción del riesgo de incendios en la interfaz urbano-forestal, se empleó la medida de transmisión como referencia para identificar los polígonos o rodales en el paisaje que son el origen de incendios que alcanzan un gran número de viviendas en los núcleos urbanos del área de estudio. Los perímetros (resultados de simulación) se intersectaron con los centroides de las viviendas residenciales (catastro.navarra.es) y el número de viviendas intersectadas fue respectivamente asignado a las coordenadas de cada ignición. A continuación se calculó la transmisión (*TF*) para cada polígono de terreno forestal con la siguiente ecuación (Alcasena *et al.*, 2017):

$$TF_{ij} = \frac{RH_j}{N_i} \qquad (3)$$

donde *RH* son el número de viviendas residenciales afectadas en *j* (Valle de Juslapeña) y *N* es el número de igniciones dentro del polígono de terreno forestal *i* considerado en el análisis. En este análisis también se consideró la transmisión de incendios iniciados dentro del área de estudio (es decir, todos los *i* polígonos o rodales dentro de *j*).

Debido a que actualmente en el área de estudio no existen planes de ordenación ni datos de inventario detallados a nivel de rodal, son las medidas de espesura el criterio que habitualmente se considera para determinar la necesidad de ejecución de claras. A falta de datos de inventario detallados a nivel de rodal (en todas las masas, robledales y pinares) que nos pudiesen facilitar el cálculo de índices de espesura (por ej., Hart-Becking o Stand Density Index) se empleó el valor de la fracción de cabida cubierta (%) como una primera aproximación (FCC), ya que en última instancia el objetivo en el proceso de optimización es priorizar tratamientos y no cuantificar las extracciones. La fracción de cabida cubierta fue obtenida en alta resolución a partir de la nube de puntos LiDAR con FUSION (McGaughey, 2018). La altura de referencia considerada para diferenciar el estrato arbolado y el estrato arbustivo se fijó en 3 m. La FCC se obtuvo a partir de la relación entre el número de primeros retornos sobre los 3 m y el número total de primeros retornos en píxeles de 30 m de resolución.

b) Área de planificación y unidades de tratamiento

Para delimitar la extensión del área de planificación donde se pretende priorizar el tratamiento de combustibles a escala de paisaje, en este estudio se consideró la cuenca de exposición a incendios o "*fireshed*" de las viviendas residenciales en Valle de Juslapeña y no el límite administrativo. La delimitación de la cuenca de exposición se realizó a partir de una trama continua de transmisión, generada mediante una interpolación espacial a partir de los valores asignados a cada ignición. Concretamente, la trama se generó a 30 m de resolución con sistemas de información geográfica (SIG) mediante una interpolación bilineal y el límite de la cuenca se fijó en los píxeles con valores de transmisión igual a 0. A continuación, para delimitar los rodales (es decir, unidades de tratamiento de combustibles) se consideraron los polígonos de terreno forestal y pasto arbolado de SIGPAC 2017 (e: 1/5000; sigpac.navarra.es) situados dentro de la cuenca de exposición. No obstante, los polígonos con una superficie superior a 10 ha fueron divididos en rodales con una superficie máxima de 5 ha considerando pistas forestales, cursos de aguas superficiales, divisorias de aguas y cambios bruscos de pendiente. A continuación, a cada polígono dentro de la cuenca de afectación (n= 7218 polígonos, 9880 ha de superficie) se le asignó la suma del valor de todos los píxeles para cada objetivo (HIBP, FCC y TF). Para poder normalizar los valores de cada objetivo en cada polígono, a cada polígono se le asignó la contribución (%) del valor con respecto a la suma total en todos los polígonos (por ej., si la suma de HIBP en todos los polígonos fue de 3140, al polígono con un HIBP de 7,85 le corresponde el 0,25%).

c) Maximización de objetivos

Se empleó la siguiente ecuación para la maximización de objetivos y la identificación de los rodales a tratar (Ager *et al.*, 2016b):

$$Max \sum_{j=1}^{k} \left(Z_j \times \sum W_i N_{ij} \right) \tag{4}$$

condicionado a

$$\sum_{j=1}^{k} \left(Z_j A_j \right) \leq C \tag{5}$$

donde C es el factor limitante (es decir, área de terreno a tratar), Z es un vector binario para indicar si el rodal *j* se trata o no (por ej., Z_j=1 para rodales tratados y 0 para rodales no tratados), N_{ij} es la contribución para el objetivo i en el rodal j si es tratado, A es el área del rodal j si es tratado y W es un coeficiente para asignar pesos que permite enfatizar un objetivo respecto a otro. Cuando todos los objetivos tienen el mismo peso (W) se asigna el valor de 1 a todos ellos. El área a tratar se fijó en un 20% de la superficie forestal arbolada (A=1976 ha, ~10% de la cuenca de exposición). Estudios previos indican que las bajas intervenciones (área tratada <18-20%) no tienen efectos sustanciales en el comportamiento y la propagación de los grandes incendios (Ager *et al.*, 2013; Finney *et al.*, 2006). Para trazar las fronteras de posibilidad de producción a modo de proyección tridimensional (óptimo alcanzable para todas las combinaciones posibles en los tres objetivos) se emplearon los 415 puntos correspondientes a las combinaciones de pesos (W) entre 0 y 7 para incrementos de 1 unidad (LTD permite asignar pesos a objetivos en base a las preferencias de los gestores). Por ejemplo, un peso de 1 para FCC y 0 para HIBP y TF identifica los polígonos donde la suma de la contribución (%) en FCC sea máxima, independientemente de los valores de HIBP y TF. En las parcelas a tratar se contempla la combinación de varias técnicas (es decir, quemas prescritas, desbroces y claras) que deberán en cada caso adaptarse a los condicionantes socioeconómicos y topográficos existentes en las parcelas. En el proceso de optimización se excluyeron las parcelas con longitudes de llama inferiores a 1,2 m puesto que la gestión de combustibles predominantemente herbáceos se realiza mediante la ganadería extensiva en el área de estudio.

3. **Resultados**

3.1. Riesgo de incendios

La expectativa de pérdidas económicas en caso de gran incendio (eEL) presentó una gran variabilidad en el área de estudio (Fig. 6). Tal y como era de esperar, los daños en las viviendas residenciales fueron superiores a los daños en las repoblaciones de carácter productor. Las mayores pérdidas se obtuvieron en las viviendas de las entidades locales situadas al sur (> 8000 € vivienda^{-1}), siendo las pérdidas hasta más de ocho veces superiores con respecto a las localidades del norte (Fig. 6B). El valor medio en el área de estudio fue de 7.955 € vivienda^{-1}. La mayor probabilidad de quema (BP) obtenida al sur del área de estudio resultó el factor más decisivo en la expectativa de pérdida económica (eEL) ya que la variabilidad en las valoraciones económicas es reducida. La mayoría de viviendas (>75%) presentaban valores de entre los 100.000 y 200.000 €, rara vez se superan los 300.000 €. En las repoblaciones las pérdidas más elevadas se localizan en la parte central (> 500€ ha^{-1}; Fig. 6A), correspondiendo con los rodales en edad de latizal alto y fustal bajo capitalizados en existencias y con exposiciones elevadas (elevada probabilidad de quema y elevadas intensidades). Los pinares en edad de fustal alto localizados en áreas remotas presentaron pérdidas potenciales muy reducidas. En las repoblaciones, la variabilidad en las existencias y el valor de la madera resultaron factores decisivos que establecieron grandes diferencias entre los resultados de los diferentes rodales. El precio de la madera puede incrementarse hasta en más de ocho veces, puesto que oscila entre los < 3€ m^3 con corteza (trituración, pasta de papel) y los >25 € m^3 con corteza (madera de sierra).

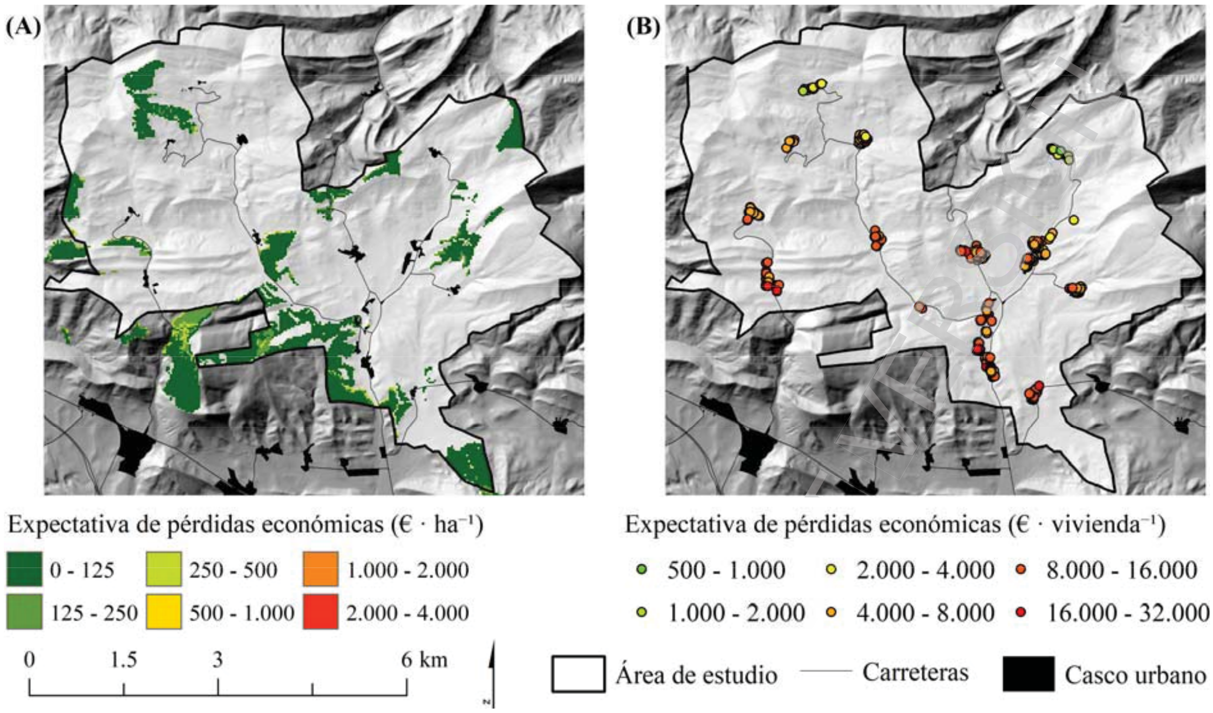

Figura 6. Mapa de riesgo para las repoblaciones de pino laricio (A) y viviendas (B) en el área de estudio. Para la valoración de las pérdidas en las repoblaciones se consideró la mortalidad esperada (%), existencias y los precios de la madera alcanzados en subastas. Las viviendas (puntos) se agrupan en núcleos urbanos. En el caso de las viviendas se consideraron funciones de respuesta (pérdidas, en %) y los valores derivados de catastro para las edificaciones. Los resultados que se presentan están condicionados a la ocurrencia de un gran incendio forestal.

3.2. Objetivos de los tratamientos

Para poder asignar los valores correspondientes a los tres objetivos en cada rodal, primero se calcularon los valores de HIBP y TF a 30 m de resolución en el paisaje de 36000 ha considerado en la simulación. El mapa de probabilidad de quema en alta intensidad (HIBP) alcanzó los valores más elevados en los fondos de valle de la parte central del área de estudio (Fig. 7A). En las montañas, exposiciones norte y cotas más elevadas los valores fueron bajos (<0,03) debido a la baja probabilidad de quema (BP) y bajas intensidades (FIL). Los resultados de HIBP fueron especialmente bajos en los hayedos limítrofes localizados en la zona norte, que representan una barrera a la propagación de los incendios. Los valores de transmisión (TF) fueron especialmente elevados para todo el fondo de valle en el área de estudio (>200 viviendas residenciales) (Fig. 7B). La cuenca de exposición a incendios (contorno delimitado por TF = 0; Fig. 7B) no se corresponde con el límite administrativo del Valle de Juslapeña y los incendios iniciados fuera del área de estudio (incluso a más de 5 km de distancia) pueden representar un grave riesgo. Cabe destacar el gran potencial destructor de los incendios iniciados sobre la llanura (campos de cereal) situada en los términos municipales al sur del área de estudio.

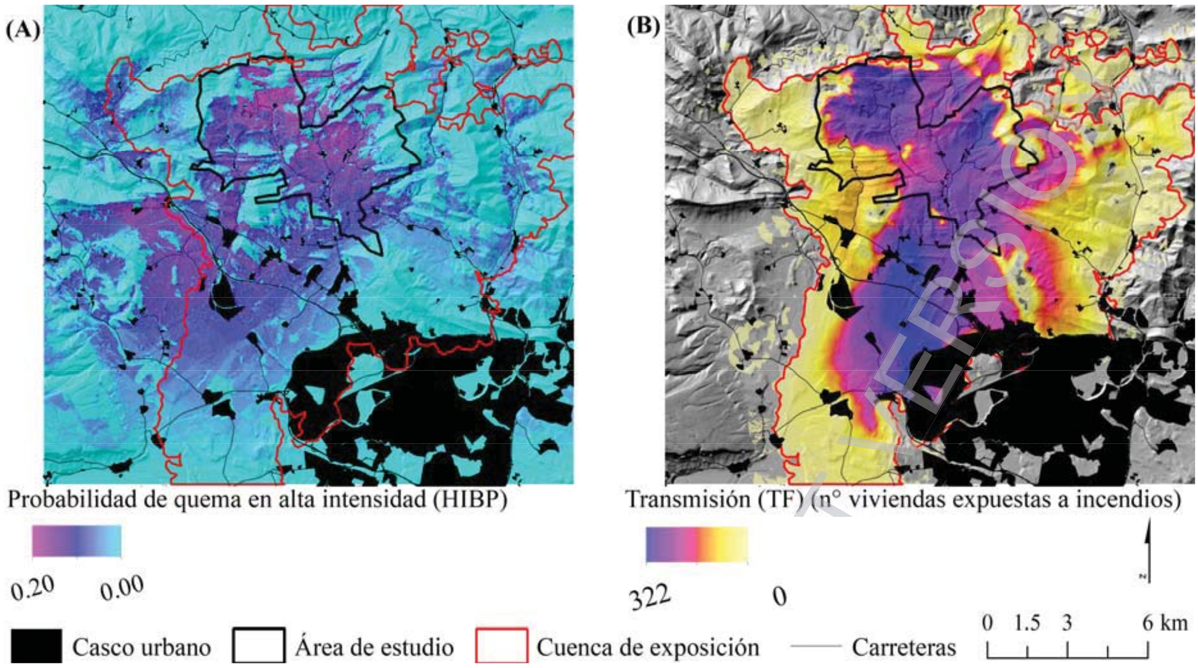

Figura 7. Mapa de probabilidad de quema en alta intensidad (HIBP; > 2,5 m de longitud de llama) (A) y transmisión de incendios a viviendas (número de viviendas expuestas a incendios) (B) en alta resolución (30 m). Los resultados fueron obtenidos para condiciones meteorológicas extremas (percentil 97) y una duración de 6 h en propagación libre. La HIBP está condicionada a la ocurrencia de un gran incendio forestal. El contorno exterior del mapa de transmisión delimita la cuenca de exposición a incendios o "fireshed" y fue empleada para acotar la superficie (polígonos) a considerar en la optimización espacial.

3.3. Frontera de posibilidades de producción

El proceso de optimización nos permitió maximizar la contribución de los polígonos tratados (20% de la superficie forestal dentro de la cuenca de exposición) para los tres objetivos (Fig. 8). Los resultados correspondientes al límite de las fronteras de posibilidades de producción fueron representados a modo de proyección tridimensional. La proyección representa el óptimo y la máxima contribución posible, las combinaciones por encima de la superficie son inalcanzables y las situadas bajo la proyección suponen combinaciones ineficientes. Las contribuciones de HIBP, FCC y TF presentaron variaciones entre 21-55 %, 15-33% y 15-39% respectivamente. El valor máximo alcanzable en cada objetivo se obtuvo cuando toda la superficie tratada se destinó a maximizar un único objetivo. Dependiendo de las prioridades establecidas por los gestores del territorio (es decir, pesos asignados a los objetivos) y las limitaciones económicas (necesidades de costear los tratamientos con las extracciones), la suma de la contribución de todos los polígonos tratados se localizaría en uno u otro punto de la proyección tridimensional. Para situaciones en las que las limitaciones económicas sean un factor determinante en el momento de ejecutar los trabajos, la obtención de beneficios puede ser decisiva y se buscará maximizar la contribución de FCC. En este caso la combinación de la suma en todas las contribuciones se encontrará localizada en la zona de colores más cálidos donde las masas tratadas presentan una mayor espesura y las extracciones en las claras serán mayores. Si por el contrario no existen grandes limitaciones económicas, los tratamientos buscarán maximizar HIBP y TF (zona de colores fríos).

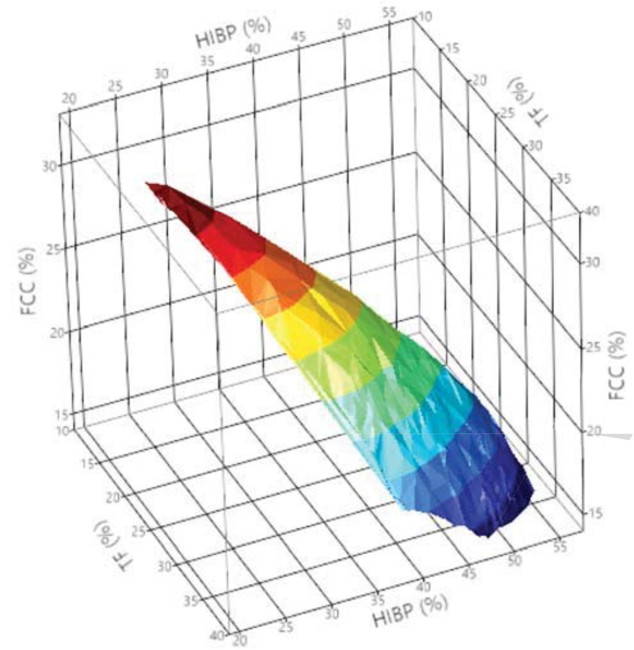

Figura 8. Fronteras de posibilidades de producción para la transmisión de incendios a viviendas residenciales (TF), probabilidad de quema en alta intensidad (HIBP) y fracción de cabida cubierta (FCC). La superficie tridimensional proyectada indica el máximo alcance posible (%) y óptimo para los tres objetivos al efectuar tratamientos en un 20 % de la superficie forestal arbolada. Las soluciones sobre la proyección son inalcanzables con los recursos disponibles y las situadas bajo la proyección son ineficientes.

3.4. Localización de los tratamientos

Se generaron mapas para la maximización de los diferentes objetivos por separado (empleando pesos de W = 1 para el objetivo a maximizar, y W = 0 para el resto), en los que se identificó el mosaico de rodales a tratar para cada uno de ellos (Fig. 9). En el caso de los tratamientos encaminados a reducir la transmisión de incendios a viviendas, la mayoría de los rodales seleccionados se concentraron dentro del área de estudio y en un entorno próximo. En cuanto a las masas arboladas con una elevada exposición a incendios, los rodales identificados se correspondieron principalmente con repoblaciones de pino laricio y robledales procedentes de regeneración natural en edad de latizal. Son masas próximas a los cultivos de cereal donde los incendios que se transmiten y propagan por el fondo de valle impactan con las zonas forestales arboladas. Los rodales con una mayor espesura donde a priori las extracciones de leñas y madera serían más elevadas, corresponden a todo tipo de masas arboladas dispersas dentro de la cuenca de afectación. Estas masas corresponden principalmente a pinares en los que han transcurrido más de 15 años desde la clara anterior y robledales o hayedos que presentan un difícil acceso y con un aprovechamiento de leñas para hogares complicado (es decir, elevada pendiente y ausencia de pistas forestales).

Con el objeto de conocer la correspondencia espacial de tratamientos para los tres objetivos maximizados por separado, se superpusieron los rodales estratégicos identificados en las tres soluciones. Además, se empleó el LTD para la identificación de los rodales a tratar cuando la prioridad (o peso) en los 3 objetivos es la misma (W = 1, 1, 1). Nótese que los rodales seleccionados a partir de la superposición de los tres objetivos maximizados por separado no se corresponden con la combinación de pesos W = 1, 1, 1 (trama cuadriculada) y sus contribuciones se situarían bajo el plano de la frontera de posibilidad de producción (Fig. 9). No obstante, la correspondencia espacial puede ser empleada para establecer el orden de ejecución en los tratamientos (es decir, ejecutar los tratamientos primero en rodales con mayor correspondencia de objetivos maximizados independientemente), ya que los proyectos de tratamiento de combustibles requieren varios años para completar su ejecución. En general, la mayoría de los rodales a tratar seleccionados en los tres objetivos se encuentran situados al límite norte ya que toda la parte sur de la cuenca de afectación se corresponde con campos de cultivo de cereal que han sido excluidos del análisis.

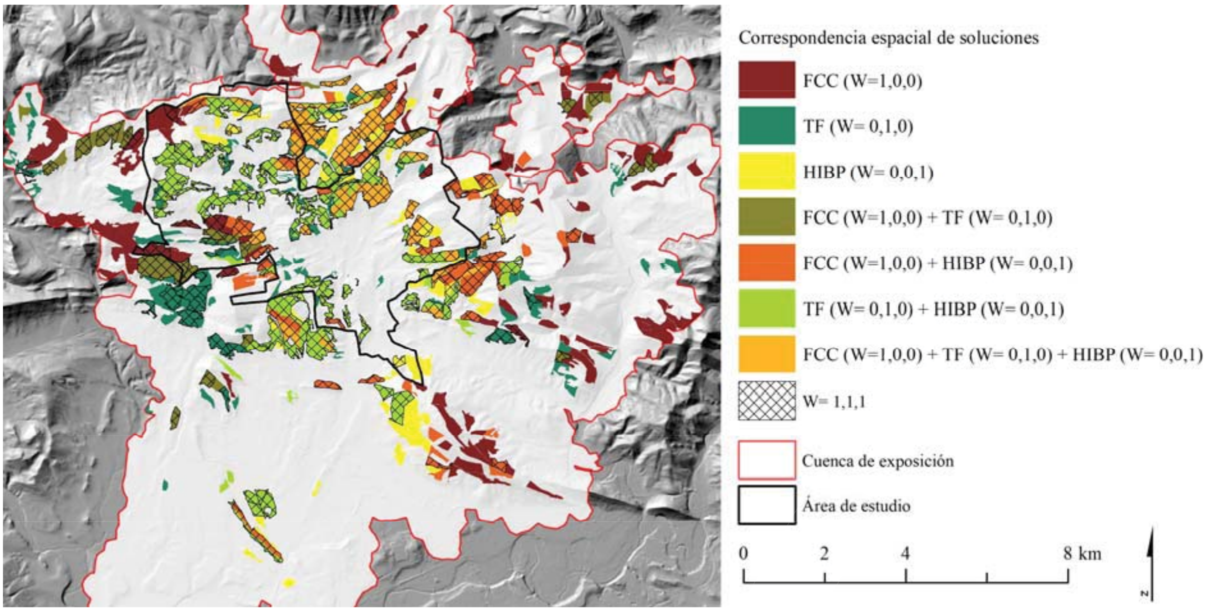

Figura 9. Correspondencia espacial de las parcelas o rodales a tratar, identificados a partir de la maximización de los tres objetivos por separado: la reducción de riesgo de incendios en viviendas (TF), mitigación de daños en masas arboladas con una elevada exposición (HIBP) y obtención de lañas o madera (FCC). Además, también se presenta la solución correspondiente al mismo peso o prioridad a todos los objetivos (W = 1,1,1), que se indica con una trama cuadriculada. Los resultados fueron obtenidos para tratamientos del 20% del terreno forestal dentro de la cuenca de exposición por incendios a núcleos urbanos. Únicamente se consideraron en los tratamientos los polígonos de terreno forestal (no pastizales) con intensidades medias superiores a 1,2 m de longitud de llama.

4. Discusión

La expansión de la superficie forestal arbolada durante las últimas seis décadas ha desencadenado un régimen de incendios forestales resistente a la extinción. Esta nueva superficie es un bosque no gestionado con un gran número de pies en las clases diamétricas inferiores, caracterizado por una elevada carga y continuidad de combustibles. En las regiones Mediterráneas, este cambio ha supuesto la desaparición del paisaje cultural en mosaico adaptado a un régimen de incendios recurrentes de baja severidad (Cervera *et al.*, 2016). A pesar de que multitud de estudios enfatizan la necesidad de reducir los combustibles forestales para combatir a los grandes incendios forestales (Bovio *et al.*, 2017; Fernandes, 2013; Madrigal *et al.*, 2016), los servicios de prevención no disponen de las herramientas necesarias para optimizar los recursos económicos disponibles.

Actualmente la localización de tratamientos preventivos se determina en base al criterio experto. Sin embargo, la ausencia de una metodología técnicamente consistente aceptada por los gestores del territorio dificulta el establecimiento de un procedimiento estandarizado que facilite su diseño e implementación. El criterio experto se basa en el estudio del comportamiento y la propagación de incendios históricos para determinar la localización de puntos estratégicos a escala de paisaje o macizo forestal, además de considerar la tipología de combustibles forestales (es decir, carga y estructura) en la prescripción del tipo de tratamiento más adecuado en cada caso (por ej., clara, quema, desbroce o poda). No obstante, en ocasiones, los tratamientos son sistemáticos y consisten en la apertura o mantenimiento de redes de cortafuegos que siguen principalmente divisorias de cuencas hidrográficas para segmentar el territorio en bloques o áreas de gestión. Las principales limitaciones del criterio experto son la imposibilidad de evaluar cuantitativamente el efecto de las diferentes alternativas en la reducción del riesgo de incendios (por ej., variación de la forma y superficie de parcelas tratadas, además del tipo de tratamiento empleado), así como la dificultad de establecer prioridades entre parcelas a tratar situadas en territorios extensos (>10.000 km^2) con un régimen de incendios cambiante. Nosotros proponemos una metodología que permite determinar dónde se concentran las pérdidas más elevadas, además de facilitar el diseño estratégico de tratamientos preventivos en paisajes multifuncionales. La disposición espacial de tratamientos contempla por tanto una solución integral con dos estrategias complementarias diferentes (Calkin *et al.*, 2014; Penman *et al.*, 2015): tratamientos junto a los bienes con elevadas expectativas de pérdida económica y tratamientos multifuncionales a escala de paisaje. Los tratamientos en el entorno próximo a viviendas residenciales y pinares de carácter productor disminuyen notablemente el riesgo en la parcela o rodal tratado y se debe a la reducción en la intensidad del fuego (es decir, longitud de llama). En el segundo caso, el mosaico de tratamientos estratégicos pretende reducir la transmisión a núcleos urbanos (es decir, probabilidad de quema) así como la exposición en masas arboladas, además de maximizar las extracciones de leña o madera.

Este estudio supone una primera aproximación que contribuye a esclarecer la complejidad existente en los procesos de optimización espacial. En líneas generales, la optimización en el tratamiento de combustibles se emplea para determinar dónde se deben localizar los tratamientos y cuánta superficie de terreno se requiere tratar, de modo tal que las inversiones realizadas en los tratamientos minimicen al máximo las perdidas asociadas a incendios forestales (Finney, 2007; Finney *et al.*, 2007). A su vez, la variabilidad en la intensidad de los tratamientos (es decir, cantidad de biomasa retirada y estructura en los combustibles tratados) y las diferencias existentes en el tamaño, forma y grado de agregación de las parcelas incrementa sustancialmente la complejidad del proceso (Arca *et al.*, 2015; Finney, 2004; Salis *et al.*, 2018; Salis *et al.*, 2016b; Scott *et al.*, 2016). Del mismo modo, la existencia de condicionantes económicos (presupuestos limitados), medioambientales (normativa restrictiva en hábitats protegidos) y de propiedad o tenencia de la tierra (público o privada) hacen que la solución sea distinta en cada caso, siendo difícil la obtención de conclusiones generales. Con el objeto de aportar una solución consistente para el área de estudio, en este artículo se identifica el conjunto de rodales (mosaico de parcelas) a tratar de manera preferente (es decir, parcelas con los valores más elevados por unidad de superficie asociados a uno o varios objetivos). Para facilitar el cálculo se establecieron una serie de consideraciones generales, así como la exclusión de cultivos agrícolas (tierra arable) y pastos, el establecimiento de una superficie máxima a tratar como único factor condicionante y el empleo de la delimitación de las parcelas o rodales en base a catastro actual. Además, el hecho de acotar el "área tratable" empleando la cuenca de exposición a viviendas nos permitió "concentrar los tratamientos" en un área menor y excluir así las áreas más remotas.

Los incendios no entienden de límites administrativos y la responsabilidad en la gestión de combustibles es compartida y debe descomponerse en varias escalas (Calkin *et al.*, 2011; Palaiologou *et al.*, 2018). En los núcleos urbanos, los propietarios particulares de viviendas (y parcelas) y la administración local deben coordinarse e implicarse en el mantenimiento de los combustibles en zonas de interfaz urbano-forestal con el objeto de generar comunidades adaptadas a incendios forestales. En el área de estudio, el arado de los campos de cereal próximos a los núcleos urbanos (hasta 60 m) inmediatamente después de la cosecha con un pase de cultivador, así como la conservación de zonas ajardinadas (es decir, franjas perimetrales) y huertas con una baja carga de combustibles reduciría significativamente la intensidad del frente del incendio en caso de impactar directamente con el núcleo urbano habitado (Alcasena *et al.*, 2015). En el entorno más próximo a las viviendas (<30 m) el empleo de especies poco inflamables y la retirada de todo el combustible muerto en setos y jardines (por ej., hojarasca y restos de podas, incluso la leña apilada junto a viviendas o tanques de propano) pueden resultar determinantes para evitar la ignición de las estructuras y frenar la transmisión del fuego entre viviendas vecinas (Cohen, 2008). Además, tal y como demuestra la discordancia entre la cuenca de exposición (es decir, área de planificación de tratamientos) y el límite administrativo (Fig. 7), la colaboración entre municipios vecinos también resulta necesaria ya que los grandes incendios originados fuera del área de estudio pueden llegar a causar graves daños en los núcleos urbanos situados dentro del área de estudio (Alcasena *et al.*, 2017). El desarrollo y la aplicación de normativa autonómica y ordenanzas municipales específicas pueden resultar de gran utilidad a tal efecto, requiriendo el cumplimiento de medidas preventivas.

Además de la gestión de combustibles en los núcleos urbanos, la reducción de la susceptibilidad al fuego en estructuras y la aplicación de medidas de autoprotección también contribuyen a reducir el riesgo. A pesar de que la ignición de las viviendas en esta tipología de interfaz urbano-forestal es principalmente causada por pavesas (es decir, el impacto directo del frente a elevadas intensidades contra la estructura es poco probable), el empleo de barnices o pinturas intumescentes en las estructuras y carpintería exterior, así como la instalación de persianas ignífugas aumentaría notablemente la resistencia de las estructuras al impacto directo del fuego (es decir, menores pérdidas en las FR para los mismos FIL). Además, las medidas de autoprotección consistentes en la habilitación de zonas seguras y la construcción de puntos de agua facilitan la operatividad y autonomía de los medios de extinción en los núcleos urbanos (Butler, 2014; Syphard *et al.*, 2014). A su vez, la habilitación de centros de reunión seguros para el confinamiento de personas vulnerables (niños, mayores y personas con movilidad reducida) puede resultar una medida acertada en núcleos rurales con mala comunicación. Ante la imposibilidad de confinamiento resultaría necesario identificar cuáles son las vías de evacuación preferente en caso de emergencia para evitar accidentes. En el área de estudio los incendios iniciados en el fondo de valle y que se propagan con viento de sur (escenario más frecuente) podrían causar el atrapamiento de personas durante su evacuación ya que únicamente existe una única carretera de salida o vía de escape al norte en todo el valle (Fig. 6B). Este problema se podría resolver fácilmente habilitando pistas alternativas que conecten los núcleos urbanos de este a oeste para permitir una evacuación segura.

Prácticamente la totalidad de la superficie forestal corresponde a comunales y es por tanto la Administración Forestal quien establece las condiciones técnicas en su aprovechamiento. En el caso de las claras de pinares productores, puede resultar oportuno requerir el tratamiento de restos (es decir, trituración de ramas y raberones) en las masas con elevada probabilidad de quema o próximas a núcleos urbanos. Esto puede solicitarse al maderista adjudicatario del aprovechamiento en el pliego de condiciones técnicas o realizarse después por cuenta propia una vez concluido el aprovechamiento forestal. Por su parte, la ejecución de quemas prescritas por parte de personal cualificado resulta una técnica adecuada para la reducción de combustibles en pinares a partir de edades de latizal alto y sin excesivas acumulaciones de restos. En los robledales con aprovechamiento de leñas para hogares las claras por lo bajo con un posterior apilado y troceado de restos supone el tratamiento más adecuado y extendido. En todo caso, es en los Planes Técnicos de Ordenación Forestal donde se deberían identificar los puntos estratégicos de gestión (PEG) así como las técnicas y prescripciones a seguir en cada tipo de masa. A tal efecto, el modelo de optimización que se presenta en este estudio supone una herramienta de gran utilidad para asistir en el diseño de los tratamientos capaz de generar la cartografía de detalle requerida por los gestores del territorio. Aunque nos hemos centrado en el tratamiento de combustibles forestales (masas con elevada carga y continuidad de combustibles), la gestión de pastos comunales con ganadería extensiva es una medida complementaria que ayuda a prevenir el crecimiento de matorral e incrementa la durabilidad de los tratamientos (Casasús *et al.*, 2007; Ruiz-Mirazo *et al.*, 2011) si los tratamientos se integran dentro de "las hierbas" (es decir, grandes recintos cercados de terreno comunal y habilitados con agua para el aprovechamiento ganadero) de los diferentes Concejos de Juslapeña. A pesar de que la longitud de llama en modelos de combustibles herbáceos no representa una gran limitación durante la extinción, la gestión con pastoreo puede ser determinante debido a que las velocidades de propagación en condiciones meteorológicas extremas (fuertes vientos y bajas humedades relativas) supera fácilmente la capacidad de extinción de los medios terrestres.

A pesar de que este estudio se centra en la gestión de combustibles, no se debe dejar de lado las medidas encaminadas a prevenir las igniciones antrópicas. En ambiente mediterráneo la mayoría de las igniciones son causadas por humanos y el desarrollo de programas de monitoreo y prevención de igniciones son también medidas prioritarias (Curt *et al.*, 2016; Gonzalez-Olabarria *et al.*, 2012). Nosotros hemos empleado un modelo de ocurrencia de incendios a partir del cual se pueden identificar fácilmente los lugares donde la probabilidad de ignición es especialmente elevada. Además, la identificación de las causas más frecuentes permite desarrollar protocolos con medidas preventivas específicas y que puedan ser aplicadas en los lugares que presentan una elevada probabilidad de ignición. Por ejemplo, las igniciones causadas por la maquinaria agrícola son uno de los casos más típicos en verano (cosechadoras y empacadoras). La aplicación de medidas preventivas así como la limpieza de restos vegetales en conductos y motores o la disponibilidad de equipos extintores o cubas que permitan una rápida respuesta en caso de ignición, pueden resultar decisivas para la extinción del incendio en un primer ataque (González, 2013).

Los paisajes culturales mediterráneos presentan a menudo un carácter multifuncional y el modelo de optimización empleado en este estudio permite la integración de varios objetivos y el establecimiento de prioridades o pesos en base a las necesidades determinadas en procesos de planificación (Ager *et al.*, 2016b; Alcasena *et al.*, 2018; Vogler *et al.*, 2015). En última instancia, la frontera de posibilidades de producción no es más que la curva (proyección tridimensional en nuestro caso; Fig. 8) obtenida a partir de las soluciones óptimas de todas las combinaciones de pesos posibles entre objetivos. A modo ilustrativo en este estudio presentamos el mosaico de tratamientos correspondiente a los escenarios que buscan maximizar un único objetivo, así como la solución intermedia que considera un mismo peso (misma prioridad) para los tres objetivos (Fig. 9). Las mayores sinergias se observan en aquellos rodales con una mayor correspondencia espacial. El empleo de métodos de evaluación multi-criterio podría facilitar en estudios futuros la determinación de los pesos a asignar en cada objetivo y obtener así la solución de consenso más conveniente para todos los agentes territoriales implicados en la gestión, uso y disfrute del territorio (Grošelj *et al.*, 2016; Uhde *et al.*, 2015).

5. Conclusiones

En el presente estudio se demuestra cómo se pude estimar económicamente el riesgo de incendios. Además, se aplica un procedimiento de optimización espacial que permite priorizar el tratamiento estratégico de combustibles a escala de paisaje. La metodología empleada permitiría evaluar cuantitativamente los efectos para cualquier tipo de configuración espacial de tratamientos. El área de planificación (cuenca de exposición) entorno a cualquier bien de elevado valor (por ej., comunidades localizadas en la interfaz urbano-forestal) viene determinada por el potencial de gran incendio forestal. El estudio de la correspondencia espacial de soluciones para objetivos diferentes permite determinar las oportunidades existentes en la gestión multifuncional de los paisajes Mediterráneos. Los resultados se trasladan a una cartografía de detalle que facilita su integración en los planes de gestión forestal.

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CHAPTER 5 – Prioritizing risk management strategies in large landscapes

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Towards a comprehensive wildfire management strategy for Mediterranean areas: Framework development and implementation in Catalonia, Spain

Fermín J. Alcasena[a,*], Alan A. Ager[b], John D. Bailey[c], Nicolau Pineda[d,e], Cristina Vega-García[a,f]

[a] Agriculture and Forest Engineering Department, University of Lleida, Alcalde Rovira Roure 191, 25198 Lleida, Spain
[b] USDA Forest Service, Rocky Mountain Research Station, Missoula Fire Sciences Laboratory, 5775 US Highway 10W, Missoula, MT 59808, USA
[c] Department of Forest Engineering, Resources and Management, College of Forestry, Oregon State University, 043 Peavy Hall, Corvallis, OR, 97331, USA
[d] Meteorological Service of Catalonia, Carrer Berlín 38–46, 08029 Barcelona, Spain
[e] Lightning Research Group, Technical University of Catalonia, Campus de Terrassa, Edifici TR1, Carrer Colom 1, Terrassa, 08222 Barcelona, Spain
[f] Forest Sciences Centre of Catalonia, Carretera de Sant Llorenç de Morunys km 2, Solsona, 25280 Lleida, Spain

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A B S T R A C T

Southern European countries rely largely on fire suppression and ignition prevention to manage a growing wildfire problem. We explored a more wholistic, long-term approach based on priority maps for the implementation of diverse management options aimed at creating fire resilient landscapes, restoring cultural fire regimes, facilitating safe and efficient fire response, and creating fire-adapted communities. To illustrate this new comprehensive strategy for fire-prone Mediterranean areas, we developed and implemented the framework in Catalonia (northeastern Spain). We first used advanced simulation modeling methods to assess various wildfire exposure metrics across spatially changing fire-regime conditions, and these outputs were then combined with land use maps and historical fire occurrence data to prioritize different fuel and fire management options at the municipality level. Priority sites for fuel management programs concentrated in the central and northeastern high-hazard forestlands. The suitable areas for reintroducing fires in natural ecosystems located in scattered municipalities with ample lightning ignitions and minimal human presence. Priority areas for ignition prevention programs were mapped to populated coastal municipalities and main transportation corridors. Landscapes where fire suppression is the principal long-term strategy concentrated in agricultural plains with a high density of ignitions. Localized programs to build defensible space and improve self-protection on communities could be emphasized in the coastal wildland-urban interface and inner intermix areas from Barcelona and Gerona. We discuss how the results of this study can facilitate collaborative landscape planning and identify the constraints that prevent a longer term and more effective solution to better coexist with fire in southern European regions.

1. Introduction

Wildfires continue to cause substantial losses to socio-economic and natural values in Mediterranean areas where human activities both drive fire regimes and simultaneously incur highest negative impacts (Díaz-Delgado et al., 2004; Martínez et al., 2009). In the southern EU countries (Portugal, Spain, France, Italy, and Greece) some 48,600 fires burn every year on average 447,800 ha (1980–2015), and a small number of large fires (< 15%) account for the bulk of burned area (San-Miguel-Ayanz et al., 2017). These fires spread for long distances (> 10 km), exhibit active crown fire that showers large amounts of embers into the wildland-urban interface (WUI) areas, and typically occur during simultaneous episodes associated to heat waves (Cardil et al., 2014; Castellnou and Miralles, 2009; San-Miguel-Ayanz et al., 2013). Currently, stand-replacing fires in unmanaged forest ecosystems, fatalities during extreme episodes, and increasing losses to human communities represent the major threats from large fires in southern European regions (Cardil et al., 2017; Costa et al., 2011). Such "mega-fires" are projected to increase due to climate change and increasing amounts and continuity of fuels (Barrera, 2011; Cardil et al., 2014; Kuemmerle et al., 2016; Moreira et al., 2011; Piñol et al., 1998).

The main strategy to reduce losses from large fires is suppression, which has been shown to be largely ineffective during extreme fire weather conditions, represent a significant financial outlay in countries like Spain (15–20 million € yr^{-1}), and result in human injuries and loss of life (56.3 injured yr^{-1} and 3.5 fatalities yr^{-1} on average from 1996 to

* Corresponding author.
E-mail address: ferminalcasena@eagrof.udl.cat (F.J. Alcasena).

2010) (ADCIF, 2012). These catastrophic events also burn through many land tenures, communities, and jurisdictional boundaries on landscapes highly fragmented in terms of ownership, fuels loadings, and land management objectives (Alcasena et al., 2017; Palaiologou et al., 2018). Thus, fires are increasingly becoming an issue that needs to be addressed collectively by the various organizations responsible for fire suppression, as well as rural inhabitants managing landscape fuels and institutions ruling territorial policies concerning wildfire (Acacio et al., 2010; Gazzard et al., 2016; Marino et al., 2014). Building a risk governance system in these Mediterranean cultural landscapes where human communities, multi-objective mosaics of pastures with forested lands and intensively managed numerous smallholdings intermingle has not progressed much beyond additional investments in suppression (Garrido et al., 2017; Oliveira et al., 2017; Senf et al., 2015). If no measures are taken, many rural communities and urban areas that depend on multifunctional forest systems for their livelihood will likely continue to face catastrophic wildfire events.

In the US, where similar concerns challenge to land managers in fire-prone areas, a new wildland fire cohesive strategy has been developed that partitions the problem into three objectives: fire-resilient landscapes, fire-adapted human communities, and a safe and efficient wildfire response (USDA Forest Service, 2014). Fire-resilient forest ecosystems have the adaptations needed to withstand and recover from fire with minimal restoration treatments, and are often characterized by low-density open stands of single-storied dominant trees with small patches of saplings and multiple discontinuities of surface, ladder and crown fuels (Fernandes et al., 2015; Hessburg et al., 2015). While mixed-severity lightning fires have been the main landscape disturbance agent maintaining low fuel loads and persistent openings in pre-settlement western US forests, the intensive anthropic management on Mediterranean cultural landscapes (i.e., agriculture, livestock, firewood and cultural use of fire) was historically responsible for preserving sharp-transition small-unit mosaics of low-fuel-load land covers (Cervera et al., 2016; Seijo et al., 2016). Although human communities in the Mediterranean have been historically less susceptible to losses from fire compared to the western US, the limited management surrounding rural communities and newly developed residential areas in the WUI has substantially reduced the capacity for firefighters and local residents to defend these communities during large-scale events (Costa et al., 2011; Sirca et al., 2017; Viedma et al., 2015).

There has been a minimal discussion in the literature on a broader, integrated approach to the fire problem for southern European Union (EU) countries, and fire exclusion and ignition prevention programs continue to be the main pillars of wildfire management (Corona et al., 2015; Fernandes, 2013; Silva et al., 2010). Nonetheless, a fire exclusion policy in fire-adapted ecosystems is not a viable long-term policy as demonstrated by the current situation in the EU countries and elsewhere (Otero and Nielsen, 2017; Seijo and Gray, 2012). Developing a broader mix of fire management objectives that are tailored to particular landscapes based on fire regimes, human values, and land use could potentially highlight where alternative and integrated strategies provide a long-term solution to better coexist with fire (Moritz et al., 2014). For instance, suppression efforts should be prioritized on areas where these interventions can efficiently prevent property loss without exposing firefighters to entrapment in hazardous environments (Cardil et al., 2017). On the other hand, fuel management should be prioritized where potential property loss is high (Alcasena et al., 2015; Salis et al., 2013). Where that is not the case, management could be directed at the re-introduction of fire in fire-dependent ecosystems, using unplanned fire as a means to manage fuels on protected natural sites and regulating traditional fire uses for pasture clearing and conservation when feasible (Barnett et al., 2016; Coughlan, 2015; Regos et al., 2014).

The current fire policy in European countries is failing to protect human communities and natural values from devastating events and this study proposes a new approach and long-term solution to deal with the growing large-fire problem in Mediterranean cultural landscapes. To explore how the current approach to wildfire could be broadened to consider other fire management strategies we combined outputs from simulation modeling with land use patterns, valued assets, and historical ignition data to map specific fire and fuel management goals including fire resiliency in forests, restoration of the cultural fire regime, safe and efficient fire response, and creating fire-adapted human communities. To illustrate our framework we implemented this study in Catalonia (northeastern Spain), a fire-prone Mediterranean region where extreme events caused very substantial losses during the last decades. Our study presents an innovative methodological framework to model historical fire size distributions and burn patterns on diverse fire-regime macro-areas while accounting for spatially changing weather scenarios across the study area. The maps obtained in this paper can be used to advance discussions about alternative management strategies and help resolve fire-related socioecological conflicts. Specifically, the results can be also used to locally prioritize specific management options as part of the landscape and urban planning within the study area. This study can represent the baseline for the development of a broader wildfire management strategy encompassing the entire fire-prone southern European regions.

2. Material and methods

2.1. *Overview of the proposed wildland fire management strategy*

We identified four wildfire management primary goals for southern European regions, while considering the US cohesive strategy (USDA Forest Service, 2015) as a referent, that each rely on specific management options (Fig. 1). We used these goals to prioritize and rank management options at municipality level (i.e., administrative division units) according to spatially-explicit quantitative metrics. The results were a set of maps that can be used to prioritize local fuels management projects, ignition prevention programs, suppression resource pre-positioning, community action projects or any other public or private risk mitigation initiatives. Human communities refer to development areas containing most residential housing structures within municipalities. See Appendix A in the supplementary material for further details about each goal, respective management options and the metrics used to assess priorities.

2.2. *Study area*

The study area was located in the northeastern extremity of the Iberian Peninsula and encompassed the 32,113 km^2 autonomous community of Catalonia (northeastern Spain). Catalonia is administratively divided into 948 municipalities, which are jurisdictionally aggregated into 42 counties and 4 provinces. Most of the 7.5 million inhabitants (> 90%) concentrate in the highly-developed metropolitan area of Barcelona and a few cities close to the coastline. The climate is predominantly Mediterranean with increasing rainfall on pre-littoral mountain ranges (precipitation > 500 mm yr^{-1}) and milder winters closer to the coastline to the east (average temperatures for January > 7 °C). The transition to high-mountain climate (precipitation > 750 mm yr^{-1} and average temperatures for January < 3 °C) is associated with the altitudinal gradient moving northwards to the Pyrenees mountain range above the 1500 m. Irrigated agricultural lands, mosaics of shrublands (*Solsona vermiculata* L.) and herbaceous xerophytic vegetation edges cover the central depression of Lleida's plain below 450 m. Increasing elevations and rough reliefs to the north confine cultivated plots to valley bottoms, with forested areas dominated by Mediterranean oaks (e.g., *Quercus ilex* L.) and low shrublands on slopes (*Lavandula angustifolia* Mill., *Rosmarinus officinalis* L. and *Quercus coccifera* L.). These shrublands and forests are gradually replaced by tall-shrubland species (*Buxus sempervirens* L. and *Juniperus communis* L.), mid-mountain oak (*Quercus pubescens* Willd.) and conifer species (*Pinus nigra* Arn. and *Pinus sylvestris* L.) first on north-facing

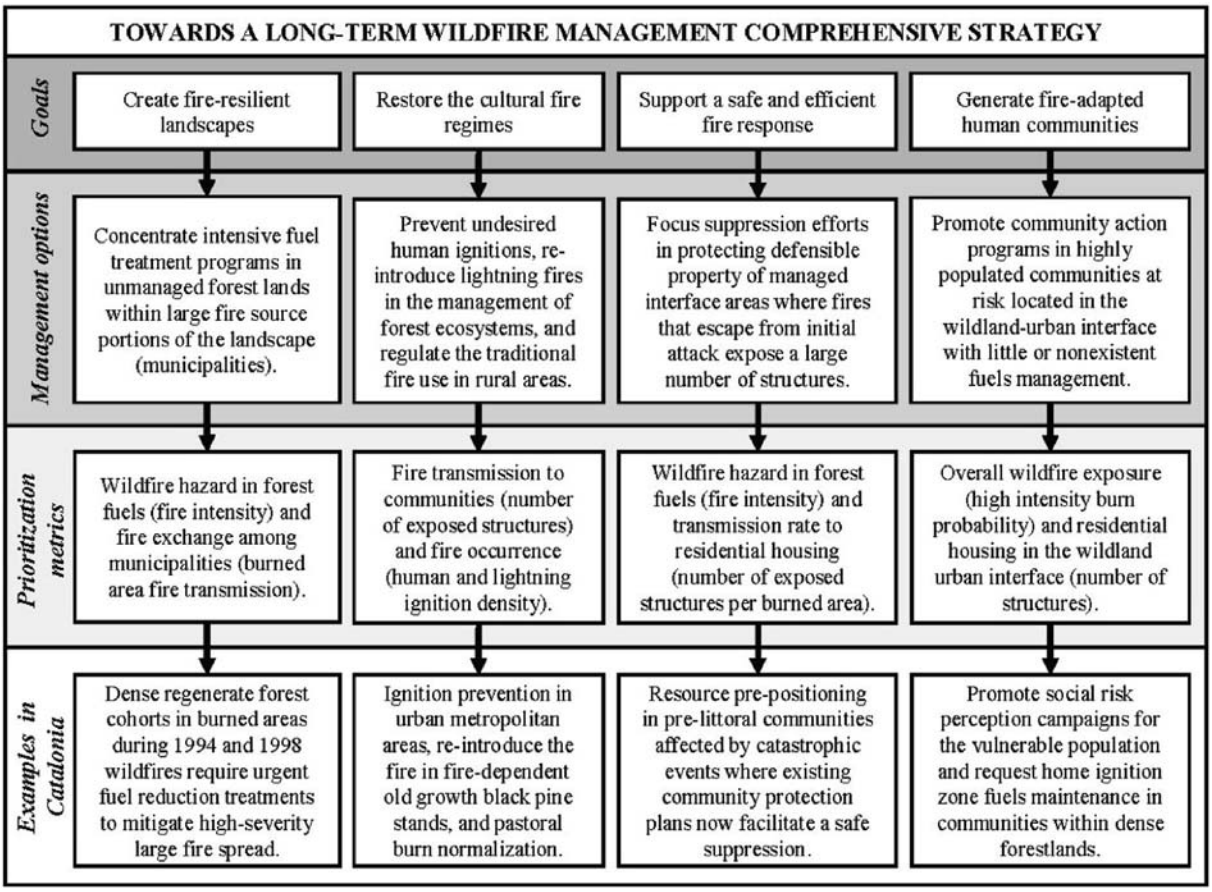

Fig. 1. General framework of the wildfire management comprehensive strategy baseline proposed for the southern European fire-prone regions. We identified four major objectives and, respectively, the most feasible management options. Wildfire occurrence, hazard, exposure, and large fire transmission metrics were used to rank priorities in the different management options at the municipality level. To illustrate the potential applicability of the framework we present some examples for its implementation in Catalonia (northeastern Spain).

slopes, and then all across on higher elevations (*Pinus uncinata* Ram.). The presence of broadleaved forests (*Fagus sylvatica* L.) and fir woods (*Abies alba* Mill.) is very limited. Mosaics of rocky outcrops, low shrublands (*Genista balansae* Boiss.) and pastures cover the high mountain tops above the 1400 m. On the pre-littoral mountain ranges, the Mediterranean maquis (*Pistacia lentiscus* L. and *Arbutus unedo* L.) appear in combinations with densely regenerated young Aleppo pine cohorts (*Pinus halepensis* Mill.). Silicicolous shrublands (*Cistus* ssp. and *Erica* ssp.) are frequently found in coastal lowlands sometimes with presence of stone pine (*Pinus pinea* L.). Cork oak (*Quercus suber* L.) is confined to the northeastern lowlands of the study area. Protected natural sites of special interest occupy about one-third of the study area and occasionally can represent a wildfire management constraint for the implementation of fuel reduction programs (Appendix B).

2.3. Historical fire activity

Catalonia is one of the largest fire-prone areas in the Mediterranean basin and encompasses a wide variety of landscapes, vegetation types, physiographic gradients, climates, and fire ignition patterns. On average some 650 fires burn about 11.5 thousand ha yr^{-1}, from which a low number (< 2%) of large fires (> 100 ha) account for more than the 88% of the burned area, and a few extreme events (> 1.000 ha fire of 1986, 1994, 1998, 2003 and 2012) concentrate the bulk (> 65%) of the burned area. Most fire ignitions (> 90%) are caused by humans (1983–2014) (MAAyMA, 2015). Lightning activity is concentrated from June to August, and most natural fires start from cloud-to-ground flashes between 12:00 and 18:00 UTC (Pineda et al., 2014).

The climatic factors in the study area controlling large fire weather conditions are associated with spatial and temporal atmospheric circulation patterns presenting substantial region-wide differences (Duane and Brotons, 2018; Rasilla et al., 2010). Therefore, we divided the study area into five zones that capture changing fire activity gradients across Catalonia, coincidental with major fire regime macro-areas (Fig. 2a; Appendix C): the Pyrenees, pre-Pyrenees, Western plain, Northern coast, and the Mediterranean coast. The delimitation of the fire regime areas was based on climatic and physiographical zone land divisions of Catalonia (Bolòs, 1975) using municipality boundary polygons. Analyzing fire activity separately on these areas facilitated the segmentation of the study area into blocks with a different wildfire season duration and very particular burn patterns associated with the local weather conditions. The wildfire season was considered as the annual period concentrating 90% of the burned area from fires > 100 ha (Fig. 2b; Table 1). Apart from the typical summer wildfire season corresponding to the Mediterranean dry period, the Pyrenees also have a secondary winter fire season (Costafreda-Aumedes et al., 2018). We can observe wide differences in fire activity between the macro-areas in terms of large fire number and mean annual burn probability (fire database from 1983 to 2014) (Table 1). For instance, large fire number and mean annual burn probability in the northern coast are, respectively, 5 and 20 times higher than in the Pyrenees.

2.4. Wildfire modeling

We used input data for fire modeling corresponding to the landscape grid (topography, surface fuels, and forest canopy metrics), fire weather

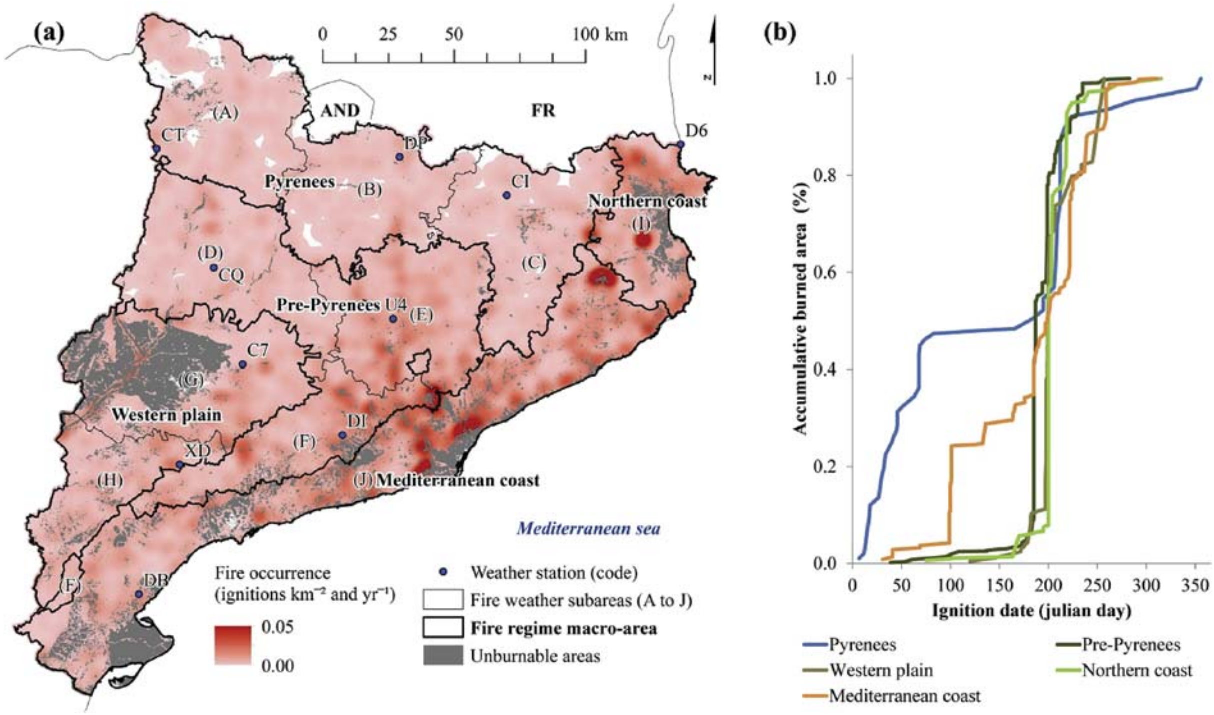

Fig. 2. Spatial extent (a) and historic fire activity (b) for the major fire regime macro-areas in Catalonia (northeastern Spain; Table 1). The fire occurrence grid (a) was generated with kernel geostatistical methods using historical ignition locations from 1998 to 2014. The fire regime macro-areas were further divided into 10 fire-weather subareas to consider the changing conditions on local wind scenarios (a; A to J; Table 2).

conditions (wildfire season wind and fuel moisture content scenarios), and an ignition probability grid derived from historical ignition locations. Topography, surface fuel, and canopy metric raster grids were assembled into the landscape file at 150-m resolution. In order to account for incoming fires, especially from the western side, and avoid edge effects on modeling outputs, the landscape file was extended with a 10 km buffer encompassing a total fire modeling domain area of 3.84 million ha. Topographic data grids (elevation, aspect, and slope) were generated from the 25-m resolution digital terrain model (ign.es), canopy metrics (canopy height, canopy cover, canopy base height, and canopy bulk density) were obtained from LiDAR-derived 20-m resolution woodland biophysical variable grids for Catalonia (ICGC, 2016), and surface fuels were obtained by assigning standard fuel models (Scott and Burgan, 2005) to the 1:5000-scale land use land-cover polygons (GENCAT, 2016). For the fuel model assignment to the different land cover polygons, we considered the vegetation characteristics such as species composition, cover, thickness, and shrubs and herbaceous fuels heights detailed in the 2012 map of habitats of Catalonia (GENCAT, 2012).

The fire modeling domain was divided in 10 subareas in order to capture the fire-weather variability across Catalonia (see A to J subareas in Fig. 2a). Some fire regime macro-areas were internally subdivided due to differences in the local wind scenarios. For every subarea, we identified a representative automatic weather station with a long data series. We used hourly temperature, rainfall, wind speed, wind direction, relative humidity, and solar radiation records to characterize the wildfire season weather conditions using Fire Family Plus (Bradshaw and McCormick, 2000). Specifically, we considered extreme weather reference conditions (i.e., 97th percentile) in terms of wind speed for most frequent wind directions and ERC-G fuel moisture content (Nelson, 2000) to obtain the fire modeling weather scenarios (Table 2). Containment efforts are very effective under mild weather conditions and thus most of the area is burned by a few extreme fires overwhelming suppression capabilities (Castellnou and Miralles, 2009; Finney, 2005).

In Catalonia, most historical fire ignitions are geospatially related to urban development and transportation corridors in highly-populated sites, and concentrate in high-density hot-spots with a sharp transition to non-ignition poor access remote areas (Costafreda-Aumedes et al., 2016; Gonzalez-Olabarria et al., 2015). In order to capture this pattern in the fire occurrence input grid required to display the ignitions within the fire modeling domain, we used fixed kernel density methods with a 2000 m bandwidth to generate a 150-m resolution ignition probability grid (Fig. 2a) considering all fire ignition coordinates for the 1998-2014 period (Gonzalez-Olabarria et al., 2012).

We used the FConstMTT command line version of FlamMap to

Table 1
Wildfire history on the main fire regime macro-areas of Catalonia (northeastern Spain; Fig. 2a). We considered a 100 ha large fire threshold to calculate the large fire frequency and define the wildfire season from the historical fire activity chart (Fig. 2b). The mean annual burn probability in Catalonia is 0.0036.

Fire regime macro-area	Wildfire season	Area (ha)	Large fire number (per 10^6 ha and yr^{-1})	Burned area (ha yr^{-1})	Mean annual burn probability
Pyrenees	Jan 7 to March 9, and July 13 to October 12	881,035	1.5	488	0.0006
Pre-Pyrenees	Jun 26 to August 27	837,776	5.1	5,234	0.0062
Western plain	July 10 to September 13	418,626	2.8	1,261	0.0030
Northern coast	Jun 18 to August 11	157,226	7.8	1,911	0.0122
Mediterranean coast	April 8 to September 10	531,950	5.6	2,604	0.0049

Table 2
Fire-weather modeling input scenarios in the different subareas (Fig. 2a). Every fire was modeled under constant extreme fuel moisture and wind conditions (97th percentile), those corresponding to the ignition location fire weather subarea. We modeled a burned area equivalent to 10,000 wildfire seasons at 150-m resolution while considering the historical fire occurrence grid (Fig. 2a) to display the fire ignitions within the modeling domain. The fire duration scenario was determined from fire size distribution replicates (Appendix D).

Fire weather subarea	Weather station (Code)	97th perc. fuel moisture content (%)					97th perc. wind speed ($km\ h^{-1}$) and frequency (%)								Duration (min) and probability (%)
		1 h	10 h	100 h	LH	LW	45º	90º	135º	180º	225º	270º	315º	360º	
Central (A)	El Pont de Suert (CT)	9	10	14	25	60	–	–	8 (15)	10 (50)	6 (5)	11 (5)	14 (15)	8 (10)	150 min (100)
Oriental meridional (B)	Das (DP)	9	10	14	25	60	23 (15)	16 (20)	11 (10)	14 (15)	14 (25)	17 (15)	–	–	
Transverse mountain system (C)	Sant Pau de Segúries (CI)	10	11	18	30	60	9 (15)	6 (30)	7 (20)	6 (10)	11 (25)	–	–	–	
Meridional central (D)	Vilanova de Meià (CQ)	8	8	11	20	60	–	–	6 (10)	8 (55)	9 (20)	7 (5)	10 (10)	–	145 min (70), 270 min (20) and 840 min (10)
Oriental (E)	Castellnou de Bages (U4)	8	9	13	20	60	–	–	7 (5)	10 (65)	11 (20)	13 (10)	–	–	
Pre-coastal mountain range (F)	Font-rubí (DI)	7	8	10	20	50	–	–	9 (10)	15 (65)	15 (25)	–	–	–	
Plain of Lleida (G)	Tàrrega (C7)	7	8	10	20	50	–	–	16 (20)	14 (20)	15 (20)	19 (40)		–	105min (90) and 215 min (10)
Southern plain (H)	Ulldemolins (XD)	6	7	9	15	50	–	–	13 (35)	19 (20)	9 (10)	13 (25)	20 (10)	–	
L'Empordà coastal plain (I)	Portbou (D6)	6	7	9	20	60	32 (15)	–	31 (20)	35 (25)	37 (5)	–	–	44 (35)	70min (70), 85 min (20) and 540 min (10)
Coastal range (J)	El Perello (DB)	7	8	11	20	50	–	15 (10)	12 (25)	19 (30)	18 (20)	–	34 (10)	28 (5)	85 min (70), 250 min (20) and 500 min (10)

model wildfire spread and behavior with the minimum travel time (MTT) algorithm (Finney, 2006). The MTT algorithm calculates a two-dimensional fire growth by searching for the set of pathways with minimum fire spread times from the cell corners at an arbitrary resolution set by the user (Finney, 2002). The algorithm has been widely used in previous studies assessing wildfire exposure and transmission in complex terrains worldwide (Jahdi et al., 2016; Kalabokidis et al., 2016; Oliveira et al., 2016; Palaiologou et al., 2018; Salis et al., 2013). Fire spread is predicted using Rothermel's surface fire spread model (Rothermel, 1972), fire intensity ($kW\ m^{-1}$) is converted to flame length (FL) using Byram's equation (Byram, 1959), and crown fire initiation is predicted according to Scott and Reinhardt (2001).

In order to calibrate the surface fire spread model, we replicated historical large fire size (> 100 ha) distribution in every macro-area separately (Appendix D). In each case, we obtained the fire spread duration that better replicated the historical fire size distribution under extreme weather conditions (Table 2). Fire ignitions were first distributed within the modeling domain according to the ignition probability grid, and then every fire was independently modeled considering the weather scenario (Table 2) in the ignition location subarea (Fig. 2a). During fire modeling, weather conditions were held constant, and fire suppression efforts were not considered due to their limited containment capabilities during extreme fire events. In total 160,000 fires were simulated at 150 m resolution, which accounted for an accumulated burned area equivalent to some 10,000 seasons. Modeled fires saturated the study area and burned each pixel more than 30 times on average. We obtained conditional burn probability (BP), fire intensity, fire size, and fire perimeter polygon outputs from fire modeling. Conditional BP is a pixel-level wildfire likelihood estimate obtained from the proportion of fires that burned each pixel given a fire occurs under extreme weather conditions within the modeling domain. FConstMTT generates the fire intensity result as flame length probability (FLP) where pixel-level outputs are expressed for 20 bin 0.5 m fire-intensity levels (FIL_1 to FIL_{20}, $FIL_{20} \geq 9.5$ m). The fire size (FS) output assigned a value (ha) to every fire ignition coordinates on a fire list file.

2.5. Analyses

We used fire modeling outputs (i.e., burn probability, fire intensity, fire size, and fire perimeters) and valued asset geospatial locations to assess wildfire hazard, overall exposure, and fire transmission. In addition, historic fire ignition data were used not only to generate the ignition probability grid required in fire modeling (Fig. 2a) but also to assess anthropogenic and lightning ignition density. Results were provided at the municipality level these because administrative boundaries delineate reference planning areas for landscape and urban planning, and represent the smallest division with management competencies. This allows transferring the core findings from this study to stakeholder and landscape managers dealing with policy-making and strategic planning. All results were annualized considering historical fire activity and normalized for a 10^3 ha area to facilitate the comparison between variable size and distant planning areas.

2.5.1. Historical fire ignitions and the cultural use of the fire

Fire records were used to calculate ignition densities for human-caused and lightning fires at the municipality level considering data from the last 32 years (1983–2014) (MAAyMA, 2015). Although early records before 1998 did not have ignition location coordinates, fire ignitions after 1983 were attributed to the municipality. The former 40 fire causes recognized in the national fire database were first grouped into natural (NAT) and 16 more major anthropic (ANT) classes. Then we calculated all anthropic (ANT) and lightning fire (NAT) densities as the number of ignitions yr^{-1} per 10^3 ha municipality area. We also calculated the incidence of the principal human causes associated with the traditional fire use. Major fire ignition causes related to the traditional use include grassland or shrub burns to improve pasture quality, silvicultural or pile burnings to eliminate thinning residue, agricultural edge property burning for multiple purposes (e.g., weed and pest control), and post-harvesting agricultural waste burnings.

2.5.2. *Wildfire hazard*

We used fire modeling outputs that describe flame length probability classes for each pixel to calculate conditional flame length (CFL):

$$\mathrm{CFL} = \sum_{i=1}^{20} \mathrm{FLP_i} \times \mathrm{FL_i} \tag{1}$$

where *CFL* is the conditional flame length (m), FLP_i is the flame length probability of a fire at the *i*-th flame length category, and FL_i is the flame length (m) midpoint of the *i*-th category fire intensity level (*FIL*). The CFL is the probability-weighted fire intensity accounting for all the possible fire front spreading directions at a given pixel (i.e., heading, flanking and backing) and is an estimate of wildfire hazard. Hazard refers to the potential for loss given a fire event, allows for the interpretation of fire suppression capabilities and facilitates the estimation of conditional losses on natural values (e.g., tree mortality and habitat loss) (Alcasena et al., 2016a; Andrews et al., 2011; Miller and Ager, 2013). At low intensities (< 1.2 m of flame length), fire can easily be contained by ground crews and those areas do not usually represent a priority in fuel treatment implementation. Intermediate fire intensity levels (1.2–2.5 m of flame length) are too intense for direct attack and can cause a significant mortality on young forests. On these areas treatments such as prescribed fires and mastication are frequently used to reduce fuels. High fire intensities (> 2.5 m of flame length) overwhelm fire suppression capabilities and easily torch dense unmanaged forests and cause massive mortalities. Here, thinning is usually required in addition to the surface fuel treatments to eliminate laddered structures and tree crown continuity.

2.5.3. *Overall wildfire exposure*

We used flame length probability and burn probability outputs to assess wildfire exposure as the high-intensity burn probability (HIBP) as follows (Lozano et al., 2017):

$$\mathrm{HIBP} = \sum_{i=6}^{20} \mathrm{FLP_i} \cdot \mathrm{BP} \tag{2}$$

where *HIBP* is the pixel level high-intensity burn probability, *FLP* is the flame length probability of a fire at the *i*-th flame length category above 2.5 m of flame length threshold, and *BP* is the conditional burn probability modeling output. Therefore, integrates both likelihood and intensity results in a unique exposure metric. Although exposure itself does not reflect fire effects, flame lengths above 2.5 m produce stand-replacing effects in conifer forests and high losses on residential houses (Alcasena et al., 2017). In this study, we used HIBP to assess exposure in the different wildand-urban interface, intermix, and disperse rural communities across Catalonia (Alcasena et al., 2018a).

2.5.4. *Large fire transmission*

We used a fire transmission analysis to assess the fire exchange across Catalonia and identify risk-source municipalities (i.e., planning areas). To assess burned area fire transmission, we used the following equation (Ager et al., 2014):

$$T_{ij} = \frac{BA_j}{N_i} \tag{3}$$

where T_{ij} measures the average fire transmission in terms of the *BA* burned area (ha) from large fires (> 100 ha) ignited in the *i*-th municipality and burning into the *j*-th neighboring municipality (i.e., $j = i$ for self-burning). Therefore, the study area was considered as a continuous cover polygon mosaic where the ignition location was assigned at the municipality in the origin and the fire exchange was estimated on every municipality (n = 948, with an average area of ∼3400 ha) in terms of self-burning (SB), incoming fire (T_IN) and the outgoing fire (T_OUT) burned area (mean annual ha yr^{-1}, per a normalized municipality area of 10^3 ha). Thus, for the entire fire modeling domain area encompassing Catalonia and the expanding 10 km buffer, ΣT_IN = ΣT_OUT.

In addition, we calculated fire transmission to structures (number of exposed structures yr^{-1}) separately for residential housing (T_RES) and industrial structures (T_IND) at the municipality level using equation [3], where *BA* burned area (ha) was replaced with the number of exposed structures in all municipalities (*SN*). To assess the transmission to structures, we intersected large fire perimeter outputs with structure geospatial locations (Appendix E) and then assigned the number of intersected structures to the ignition location (Alcasena et al., 2017, 2018c). Then, using the transmission to structures, we calculated the rates (TR) per burned area to have a better estimate of potential losses per burned ha. The latter represents a better metric to prioritize first attack and ground force pre-positioning because smaller fires burning several structures on the wildland-urban interface represent a higher priority with respect to the very large fires burning uninhabited remote areas.

2.6. *Management priorities*

Results from these efforts were presented in a set of priority maps in order to transfer our findings into straightforward meaningful outcomes for the implementation of a wildfire management strategy. We first designated a pair of metrics to prioritize each wildfire management option (Table 3). In particular, transmission and hazard metrics were used to prioritize fuels management in forest lands, fire occurrence and transmission to communities were used to target human ignition prevention areas, wildfire exposure and the number of structures on the WUI were considered to identify the communities requiring a protection plan, and transmission rates in combination with wildfire hazard were used to identify best opportunities for a safe efficient response. Then, we cross-tabulated the values from the two factors to set four priority levels: I-high, II-moderate, III-low and IV-very low (Table 4). Except for wildfire hazard, we considered quartile values to set the four categories in each metric. Fire-intensity classes associated with fire behavior were used for the interpretation of wildfire hazard: 0–1.2 m; 1.2–2.4 m; 2.4–3.4 m; > 3.4 m (Andrews et al., 2011). While the highest intensity categories present a higher priority for fuel treatment location, lower intensities represent a better opportunity for fire containment. Finally, we also generated a set of scatter-plots to explore the variation in average values among high-priority municipalities (Appendix F).

3. Results

3.1. *Historical fire ignitions and the traditional fire use*

Our results showed different annual ignition density results in terms of spatial patterns and density values for anthropic (ANT) and lightning (NAT) fires (Fig. 3a and b). While ANT ignitions concentrated at densities above 0.26 ignitions yr^{-1} per 10^3 ha in coastal municipalities and metropolitan areas of Barcelona, municipalities in central Catalonia reached the highest NAT density values (> 0.12 ignitions yr^{-1} per 10^3 ha). Anthropic ignitions showed clustered spatial patterns in areas where the human activity is especially intense (e.g. close to the communication corridors and highly populated urban areas). On the other hand, the spatial patterns of natural fire ignitions were associated with spatiotemporal atmospheric conditions, altitudinal gradients and lightning strike densities (Pineda and Rigo, 2017).

Overall, ANT ignitions resulted in much higher densities than NAT (i.e., on average ANT values were six times higher than NAT), and only very few municipalities in central Catalonia presented NAT > ANT. Among human-caused fires, ignitions related to the cultural fire use

Table 3
Assignment of metric pairs (i.e, fire occurrence, hazard, exposure, and transmission results) to the different objectives of the wildfire management strategy (Fig. 1). These metrics were cross-tabulated to obtain 4 priority classes (Table 2). The final results were presented at the municipality level (n = 948) in a set of spatial priority maps.

Goal	Management options	Prioritization metrics	Priority map
Create fire resilient landscapes	Fuel treatments	We used fire transmission (T_OUT quartiles; Fig. 5a) and wildfire hazard (CFL levels; Fig. 4a) to assess the priority classes. The classes were ranked from the highest transmission and hazard values to the lowest. Protected areas (Appendix B) were overlaid on the map to delineate areas with potential treatment constraints.	Fig. 7
Restore the cultural fire regime	Human ignition prevention	Annual anthropic fire ignition density (ANT quartiles; Fig. 3a) and transmission to residential houses (T_RES quartiles; Fig. 6a) were used to assess the priority classes. The classes were ranked from the highest ignition density and transmission to the lowest.	Fig. 8
	Natural fire re-introduction	Lightning ignition density (NAT quartiles; Fig. 5b) and transmission to residential houses (T_RES quartiles; Fig. 6a) were used to assess the classes. The classes were ranked from the highest lightning fire ignition densities to the lowest and from the lowest transmission values to the highest.	Fig. 9
Support a safe and efficient response	Fire suppression	Fire transmission rates to residential houses (TR_RES; Fig. 6c) and wildfire hazard (CFL levels; Fig. 4a) was used to set the classes. The classes were ranked from the highest transmission rates to the lowest and from the lowest hazard levels to the highest.	Fig. 10
Generate fire-adapted communities	Community action	High overall exposure levels (annual HIBP quartiles;Fig. 4b) and number of residential houses on the wildland-urban interface (Alcasena et al., 2018a) were used to set the classes. The classes were ranked from the highest exposure values and the highest number of structures to the lowest.	Fig. 11

Table 4
Metric pair cross-tabulation on the generation of management priority classes. First, we used quartile values to set 4 classes on the metrics, except for wildfire hazard where we considered interpretation charts (Andrews et al., 2011). The metric pairs were then cross-tabulated to generate 4 priority classes. These priorities were ranked from I (highest) to IV (lowest) and depicted on result maps using the color ramp of this table.

Priority class	Very low	Low	Moderate	High
Very low	IV	IV	IV	III
Low	IV	IV	III	II
Moderate	IV	III	II	I
High	III	II	I	I

required a separate consideration from those of accidental or arson origin (Fig. 3c and d). From all ANT ignitions, 29% were attributed to the cultural use, which locally represented the most important cause in some northern portions of the landscape. Fire was systematically used in the past for pasture and shrub clearing in the conservation of extensive grazing mountainous areas of the northwestern Pyrenees, and this was reflected in the results (> 0.2614 ignitions yr^{-1} per 10^3 ha). Likewise, using fire in agricultural post-harvesting waste elimination or edge clearing represented a widely extended practice, and our results highlighted this fact in many areas dominated by dryland herbaceous crops. Very similar site-specific spatial patterns for the main ignition causes were also observed in previous studies conducted in Catalonia (Gonzalez-Olabarria et al., 2015).

3.2. *Wildfire hazard and exposure*

Fire intensity in terms of conditional flame length (CFL, Fig. 4a) showed widely variable results across Catalonia, which were mainly related to the dominant vegetation types and fire season extreme weather conditions. Highest CFL values (> 2.4 m) concentrated in transition areas between open plains and Mediterranean shrubby or forest type vegetation edges (i.e., northeastern L'Empordà coastal plain and western plain of Lleida), except for some valleys in central Catalonia and some conifer forests on pre-littoral mountain portions where fast spreading heading fires were frequently impacting unmanaged forested lands. Conversely, high elevation mountainous areas (> 1,500 m) showed the lowest values due to milder weather conditions during fire season and multiple fuel discontinuities with low load patches on mosaics with rocky outcrops. Here, CFL values in temperate broadleaved forests and high-elevation conifer forests were overall very low (< 1.2 m). Agricultural irrigation lands and densely developed areas represented unburnable barriers to fire spread and showed the sharpest transitions in CFL.

Areas with high overall exposure values, as represented by the annual high-intensity burn probability (aHIBP; Fig. 4b) concentrated in the valleys of central Catalonia (i.e., Anoia, Barberà basin, Bages and southern Berguedà), where a mosaic of dense and laddered conifer forests with dryland agricultural patches dominated the landscape. In fact, one of the most devastating historical fire episodes in 1994 burned some 46,000 ha there within a week (GENCAT, 2014). Northeastern areas of Alt Empordà also had high values (> 0.06 aHIBP), where the frequency of historical high-intensity (> 2.4 m of flame length) fire is among the highest of Catalonia. For instance, "La Jonquera" large fire event on 2012 burned about 13,000 ha at flame lengths above 3 m with spread rates > 5 km h^{-1}. In all these areas wildfire risk is high since substantial losses can be expected to most valued resources at these intensities (Alcasena et al., 2017). Overall, highest HIBP values concentrated in open land to forested fuel transition areas because substantial numbers of fires ignited close to urban development areas and spread towards forested lands. Predictably, all the mountainous areas of the Pyrenees showed the lowest values (< 0.02 aHIBP), where forest fuels are only partially cured during wildfire season and fire spreading is limited to short distance upslope (< 5 km) heading runs.

3.3. *Fire exchange between municipalities and transmission to communities*

Fire exchange between municipalities in terms of the burned area revealed a high spatial variability (Fig. 5) that was related to historical ignition patterns, complex fire weather conditions, and dominant vegetation types. In total, all outgoing fires (T_OUT; Fig. 5a) represented the same amount as (T_IN; Fig. 5b) incoming fires, and varied from the low of 0 to the high of 46.21 ha yr^{-1} per 10^3 ha municipality area. Self-burning (SB; Fig. 5c) ranged between 0 to the high of 13.19 ha yr^{-1} per 10^3 ha municipality area (SB; Fig. 5a). On average, the 37% of the burned area in the municipalities (i.e., > 4000 ha yr^{-1}) corresponded to fires ignited in the vicinities (i.e., SB × 1.5 = T_IN, being T_IN = T_OUT). This is not a surprising result since the average municipality area (3400 ha) is 2.5 times smaller than the largest historic fire size in the macro areas with the highest activity. While some municipalities were net recipients of fire (T_OUT < T_IN), others resulted in net contributors (T_OUT > T_IN) (Fig. 5d). The net exchange map allowed for the interpretation of dominant fire flow directions across Catalonia, as evidenced in transitions among neighboring blocks from high fire contributors to high recipients (Fig. 5d). Locally, prevailing wind direction scenarios drove these gradients and resulted in clear

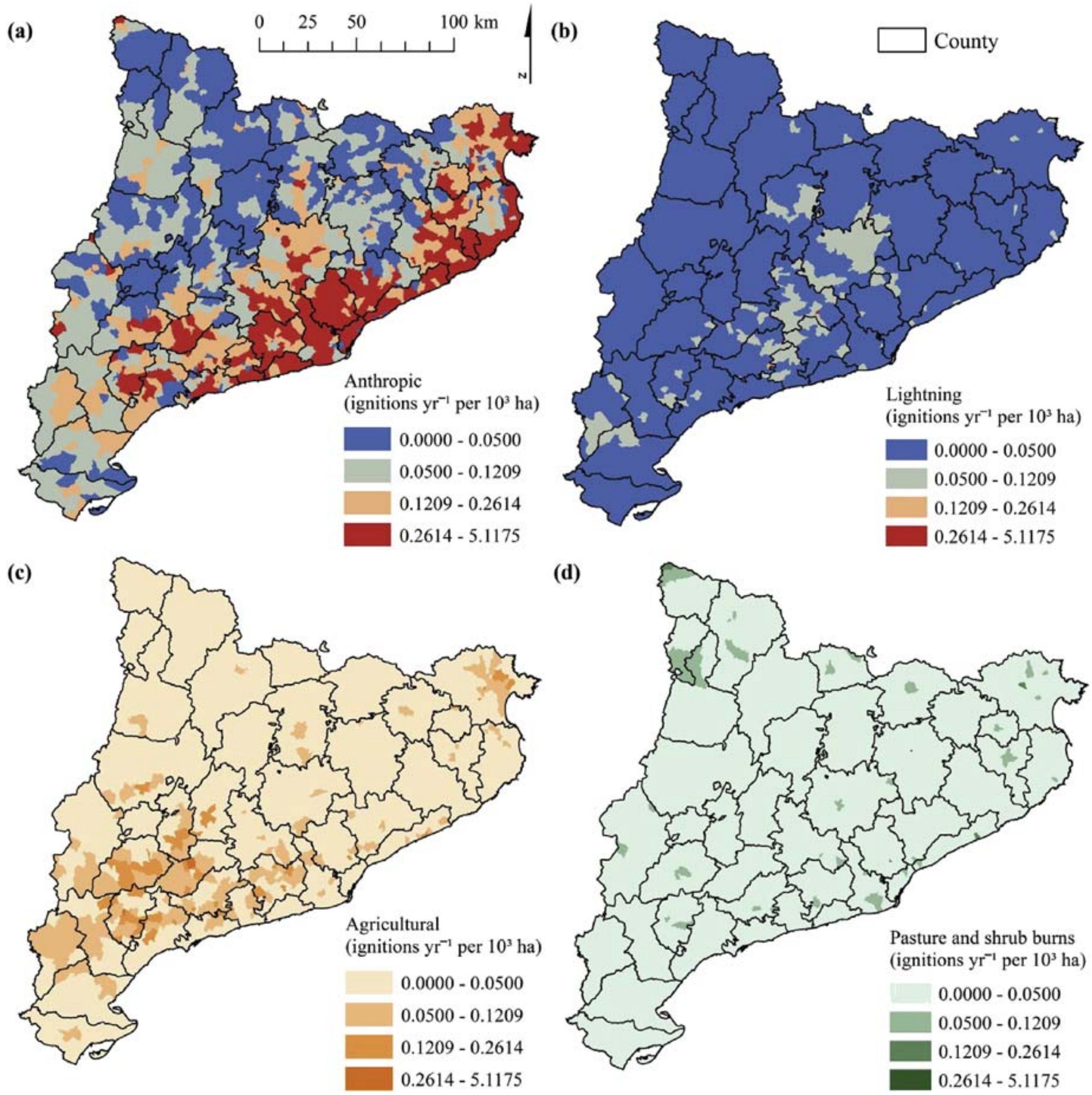

Fig. 3. Municipality level anthropic (a) and lightning (a) fire ignition densities in Catalonia for the period 1983–2014. The boundaries delineate the County level administrative division. We considered the anthropic ignition density quartile value intervals to set the classes. The cultural fire use is mainly associated to agricultural waste and edge cleaning (c), and pasture or shrub clearing (d).

trends where herbaceous fuel types covered relevant portions of the landscape. The lowest fire exchange occurred in northern mountainous areas and irrigation lands of the southwest, where SB, T_OUT, and T_IN hardy surpassed 1 ha yr^{-1} per 10^3 ha municipality area.

While high burned-area transmission was observed in Central and northeast Catalonia, transmission to structures (Fig. 6a and b) was substantially higher in coastal areas due to the very high concentration of residential houses in the wildland-urban interface (Alcasena et al., 2018a). Intensive-breeding farms were considered as industrial structures in the analysis and explained why the highest values located in certain central municipalities (Fig. 6b), while the bulk of industrial assets concentrate in highly-developed unburnable metropolitan areas of Barcelona and Tarragona. On average, transmission to housing and industrial sites at the municipality level resulted respectively in 0.45 and 0.07 structures yr^{-1} in Catalonia. Housing transmission rates (exposed structures ha^{-1}) revealed different patterns on the blocks where fires < 1.000 ha affected a high number of structures (Fig. 6c and d). This was the case for some Pyrenean municipalities where large fires (> 100 ha) were rare events and fire transmission were low (< 0.20 structures yr^{-1}; Fig. 6a), but transmission rates were high (> 0.15 structures ha^{-1}; Fig. 6c) because major fire runs affected valley bottoms and lower slopes where most structures concentrate.

3.4. Spatial prioritization

Municipality blocks with a high priority for fuels management concentrated in northeastern and several areas of central Catalonia, and represented 13% of the land in the study area (Fig. 7). Here, environmental protection land designations occupy 5.6% (23,339 ha; Appendix B) and this might represent a constraint for fuels management in some portions of the region. More specifically, treatments were not allowed on 2,822 ha (e.g., The National Park of Aigüestortes y Estany de Sant Maurici), interventions are restricted to habitat restoration on 423 ha, and fuel treatments are conditioned to traditional uses on another 2,360 ha. The highest concentrations of protected lands (i.e., Integral Natural Reserves) were located in northwestern Catalonia where fuels management priority was very low. On the other hand, active management land designations on high priority areas covered 17,735 ha

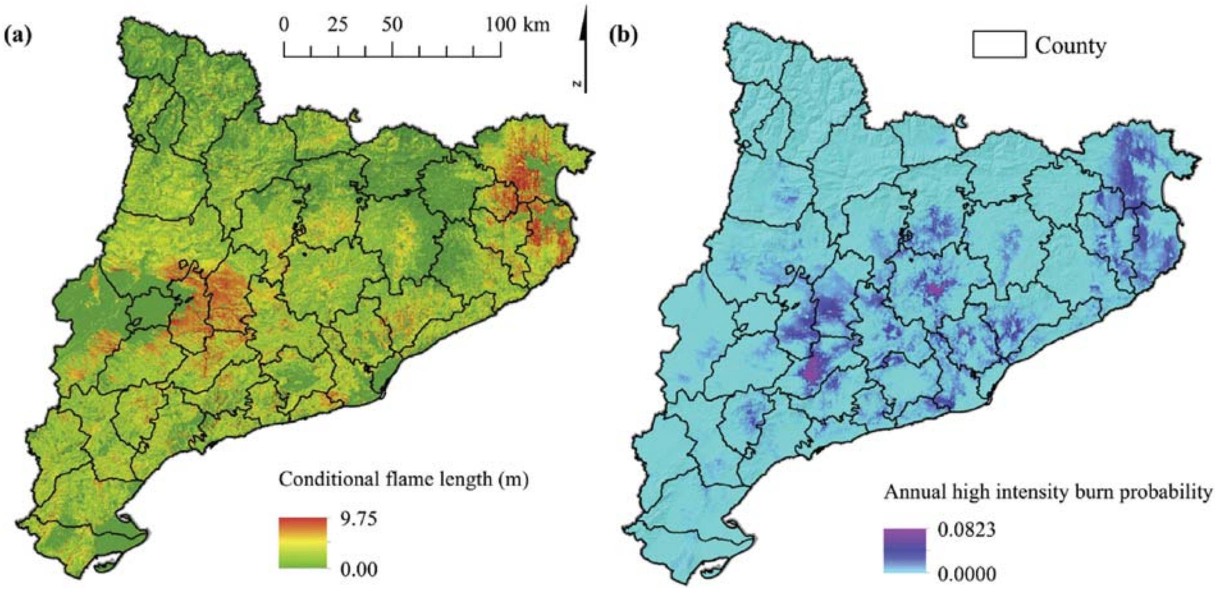

Fig. 4. Conditional flame length (CFL; a) and annual high intensity burn probability (HIBP; b) maps of Catalonia. We used them respectively as hazard and exposure metrics. The maps were generated at 150 m resolution from fire simulation modeling.

where the strong spatial collocation between timber production and risk mitigation objectives would facilitate the implementation of fuel treatment projects (Ager et al., 2017). In the municipalities with high fuels management spatial priority (n = 192 municipalities), the average wildfire hazard varied between 2.4 and 4.9 m CFL, and outgoing fire transmission varied between 3.9 and 42.66 ha yr^{-1} per 10^3 ha municipality area (Appendix F). Overall, the average CFL was higher in the municipalities with the lowest transmission values (< 15 ha yr^{-1} per 10^3 ha municipality area).

We identified the suitable areas for the cultural fire regime restoration in separate maps for anthropic fire ignition prevention (Fig. 8) and lightning fire reintroduction (Fig. 9). Coastal and metropolitan areas showed the highest priority for anthropic fire ignition prevention due to high ignition densities and transmission values to residential houses. Indeed, the bulk of municipalities in these areas had densities of 0.26–1.00 ignitions yr^{-1} per 10^3 ha and transmission values of 0.2–2.5 structures yr^{-1} (Appendix F). Conversely, high priority municipalities for natural fire re-introduction were located in the Pyrenees and remote mountainous areas of central Catalonia (Fig. 9). Instead of using high transmission values to residential houses, we used the lowest values to identify areas with high lightning ignition densities (> 0.016 ignitions yr^{-1} per 10^3 ha) but a low potential of exposing human communities (< 0.06 exposed structures yr^{-1}). Lightning ignitions densities rarely exceeded values of 0.1 ignitions yr^{-1} per 10^3 ha (Appendix F). Somehow, ignition prevention and fire reintroduction strategies showed antagonistic spatial gradients and presented a complementary basis to discern when the contribution of unplanned fire might be damaging or beneficial.

Most appropriate areas to promote safe and efficient fire response were located in some coastal municipalities and open mountainous valleys where fires affecting residential houses were surrounded by predominantly herbaceous vegetation (Fig. 10). In those areas, the numbers of exposed housing structures were high and average fire intensity was below firefighting capabilities. Many of the municipalities that showed high transmission rates were excluded as a high priority due to very high CFL values (> 3.4 m). Despite the wide variation on the average CFL, which ranged from the low of 0.75 m to the high of 2.4 m (maximum value for an effective fire suppression), most transmission rate values concentrated between 0.2 and 0.4 residential houses (Appendix F). In total, 218 municipalities covering 641,605 ha were classified as highly suitable for a full suppression strategy.

Community action can especially contribute to mitigating losses on human communities of the metropolitan area of Barcelona and densely populated municipalities of the northwest (Fig. 11). In fact, among the top 25 municipalities presenting the highest number of residential houses in the WUI, 64% were located in Barcelona and 36% in Gerona (Table 5). Although some human communities also presented high exposure values in central Catalonia, the number of residential structures there in the wildland-urban interface was much lower (< 100 structures) (Alcasena et al., 2018a) and therefore those areas were excluded from the high priority class. The bulk of human communities had less than 1500 residential houses and annual HIBP < 0.015 (Appendix F), and overall the exposure was lower as the number of residential houses in the wildland-urban interface increased. Community action represented the strategy with the highest number of municipalities in the high priority class (n = 219), which covered the 22% of the land in Catalonia.

4. Discussion and conclusions

This study advances ideas and a reference framework for a cohesive strategy founded on core themes of fire-resilient landscapes, cultural fire regime restoration, safe and efficient fire response, and fire-adapted human communities. These concepts parallel efforts in the US (USDA Forest Service, 2014), and can help facilitate a broader fire management strategy in fire-prone southern European regions (Appendix A). The cultural landscapes in the Mediterranean basin represent one of the most intensively-managed areas worldwide, where humans have been driving fire regimes for millennia (Seijo and Gray, 2012). However, losses from uncharacteristic high-severity fires and increasing suppression costs during the last decades emphasized the need for a new and wider comprehensive strategy beyond the fire exclusion policy (i.e. ignition prevention and firefighting) (Bovio et al., 2017; Curt and Frejaville, 2017). Our strategy used simulation modeling outputs to decompose the wildfire risk in a sequence of the major causative factors: (a) fire ignition source municipalities (Fig. 3), (b) large fire exchange among municipalities (Fig. 5), (c) forestland wildfire hazard in dominant fire trajectories (Fig. 4a), and (d) a pixel-based overall exposure on densely developed communities (Fig. 4b). Form these outputs we generated a consistent set of spatial priority maps for Catalonia (northeastern Spain) (Figs. 7–11) concerning specific strategies (Fig. 1). These strategies provide broad range of solutions for addressing the

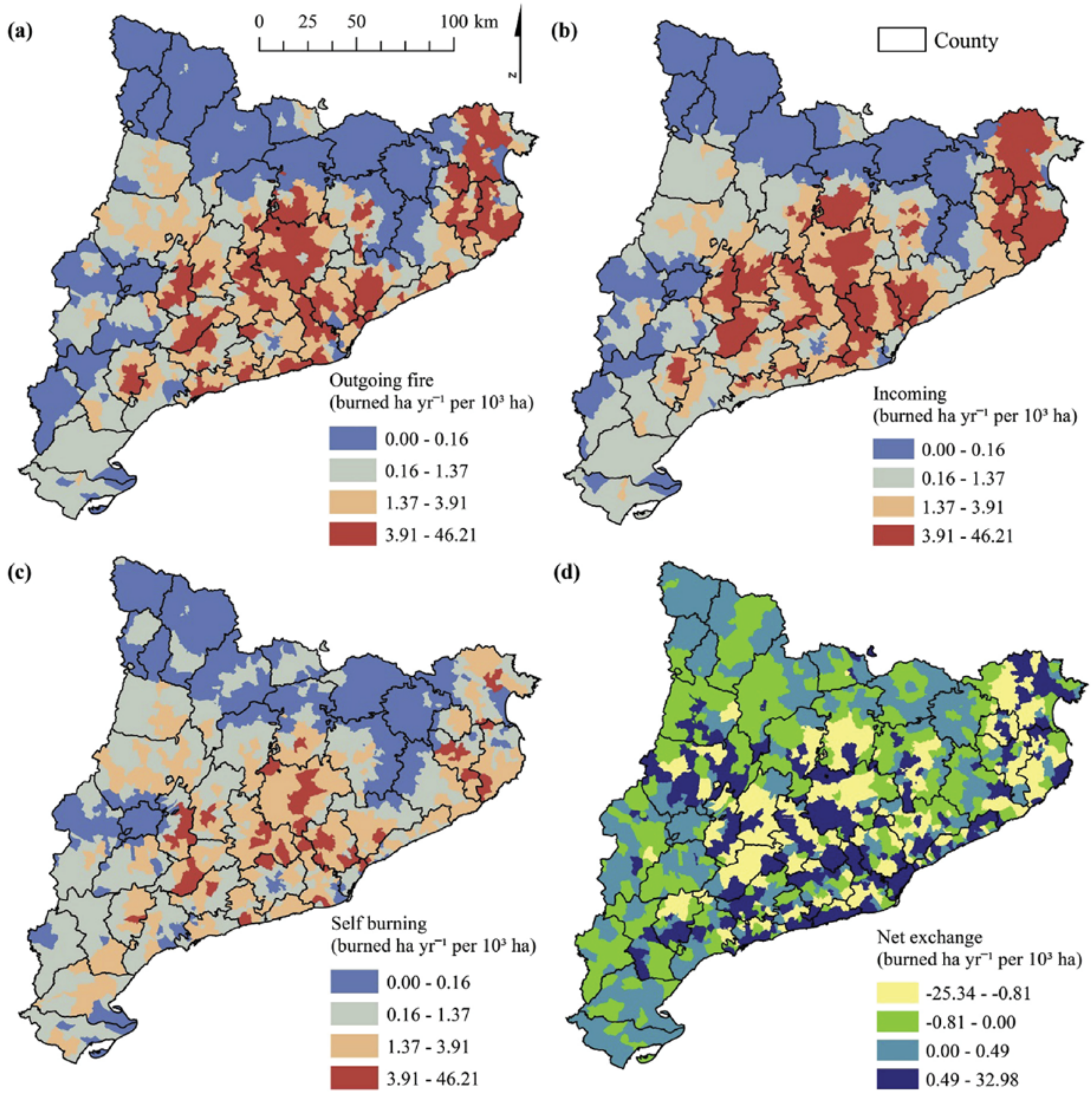

Fig. 5. Fire exchange across Catalonia at the municipality level (n = 948) in terms of incoming (T_IN; a), outgoing (T_OUT; b), self-burning (SB; b) and net exchange (Net exchange = T_OUT − T_IN; d) in terms of burned ha yr^{-1} per a normalized municipality area of 10^3 ha. On average wildfires in Catalonia burn about 11.5 thousand ha yr^{-1} (n = 650 fires yr^{-1}; 1983 to 2014), from which the 37% of the burned area came from fires initiated on the neighboring municipalities (ΣT_IN = ΣT_OUT = ~1.5 × ΣSB).

uncharacteristic fire problem that leverage the institutional capabilities to prevent ignitions, disrupt major fire movements or promote fire-adaptation strategies both in forest ecosystems and in human communities. The results also highlight the need for collaborative planning among neighboring communities at scales beyond jurisdictional boundaries since municipalities were highly interconnected by cross-boundary fire networks and local management actions can affect neighbors (Ager et al., 2016; Alcasena et al., 2017; Scott et al., 2016). Wildfire and landscape managers of Catalonia can benefit from these results to prioritize budgetary allocations in prevention and mitigation programs, in addition to urban planning and policy making.

There are only a few studies that have applied simulation modeling to examine wildfire risk and exposure at large scales in the fire-prone southern European regions (Oliveira et al., 2016; Palaiologou et al., 2018; Salis et al., 2013). Most previous fire modeling assessed wildfire exposure, and risk on smaller study areas and attempted to provide management prescriptions to local fire managers (Alcasena et al., 2016b; Elia et al., 2016; González-Olabarria et al., 2012; Molina et al., 2017). We added substantially more detail in our simulations compared to previous studies in order to capture the fire weather, ignition pattern, and fuel moisture changing gradients across the study area. The result was a 150-m resolution set of maps of key risk causative factors that previously have not been available for the fire prevention and mitigation purposes in Catalonia. Specifically, we generated consistent hazard, exposure and transmission quantitative results which facilitated comparisons and spatial prioritization between very distant areas (> 100 km) within the study area (Figs. 4 and 5). Previous studies in Catalonia characterized the dominant spread patterns from historic fire perimeters and principal synoptic fire weather conditions, to then prioritize fuel treatment allocation on strategic management points (ridges, ravines, changes in slope and buffering road infrastructure) according to expert criteria (Costa et al., 2011; Duane et al., 2015, 2016).

Extreme fires impacting populated communities represent a major concern in Mediterranean areas and many previous efforts accurately mapped the WUI types considering urban development structure

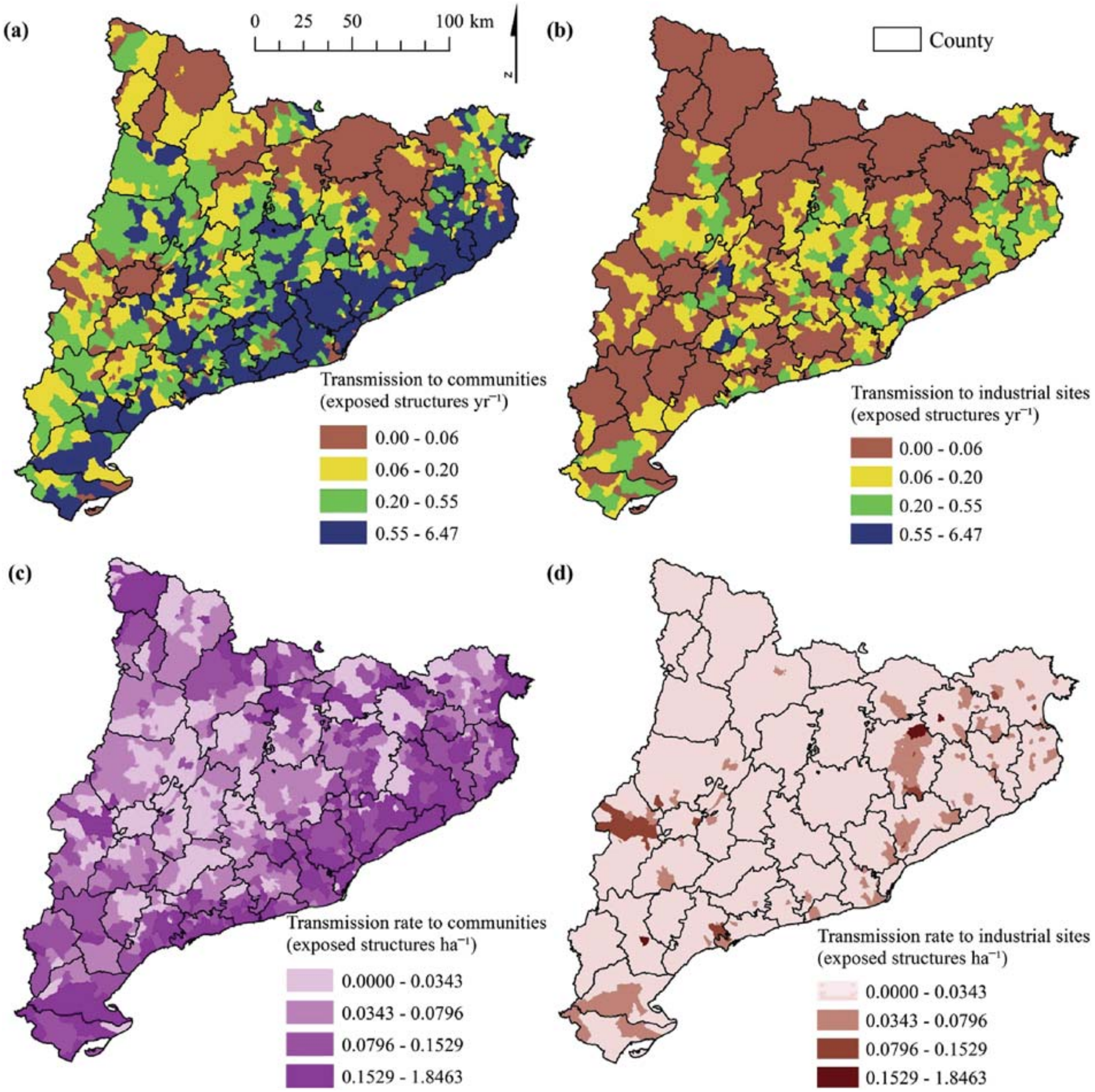

Fig. 6. Annual fire transmission and transmission rate per burned area (ha) for structures in communities (a and c) and industrial sites (b and d) at the municipality level. The analysis was conducted intersecting modeling output large fire perimeters (> 100 ha) with structure centroid locations (Appendix E).

aggregation degree, fuel types, and fire occurrence in the vicinities (Badia et al., 2011; Herrero-Corral et al., 2012; Lampin-Maillet et al., 2010; Madrigal et al., 2013; Pellizzaro et al., 2012; Sirca et al., 2017). Nonetheless, these studies did not assess wildfire exposure to large and catastrophic events, and only a few considered the potential fire effects (Alcasena et al., 2017; Mitsopoulos et al., 2015). On the other hand, previous studies in the US widely used fire modeling to estimate wildfire likelihood on populated areas, and assumed structure loss given a fire reaches a residential house (Bar Massada et al., 2009; Haas et al., 2013), or alternatively integrated structure susceptibility relations to assess the effects (Thompson et al., 2011, 2013). In our study, we first identified the number of individual structures in the wildland-urban interface (Alcasena et al., 2018a) to then use the annual high intensity (> 2.4 m flame length) burn probability as the structure potential for loss metric. We considered that fire suppression efforts can efficiently protect residential houses exposed to low intensities (i.e., in Catalonia major fire spread duration is limited to few days and structure loss is usually associated at high intensities on the home ignition zone).

Forest fuel management priority maps (Fig. 7) identified the municipalities where treatments (i.e., prescribed fire, thinning and mastication) on strategic locations above certain intensities (> 15–20% of the area) can effectively slow large fire spread and mitigate risk (Finney, 2007; Salis et al., 2016, 2018). In the context of current budgetary constraints treating the entire study area at effective intensities is impossible, and thus we proposed treating hazardous fuels (CFL > 2.4 m) on the specific high transmission planning areas that contributed the most to the burned area in neighboring municipalities. Wildfire management strategies based on fire suppression in these areas would have a low probability of success during extreme events (Andrews et al., 2011). Municipalities with lowest intensity values (CFL < 1.2 m) were classified as a very low priority for treatment implementation regardless of a high fire transmission, since managing flashy herbaceous fuels would require other options such as livestock grazing (Casasús et al., 2007; Riedel et al., 2013).

In order to restore the cultural fire regime, we identified the priority areas for anthropic ignition prevention program implementation and

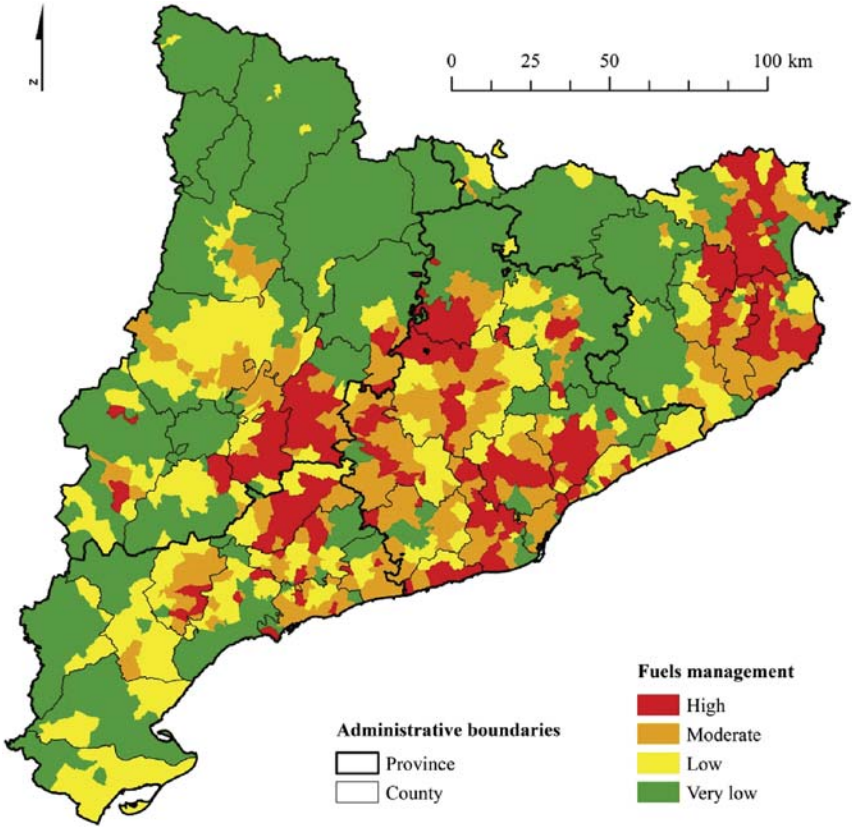

Fig. 7. Spatial prioritization map for fuel reduction programs in Catalonia. We cross-tabulated wildfire hazard on forest fuels (CFL levels; Fig. 4a) and burned area transmission (T_OUT quartiles; Fig. 5a) to prioritize fuel treatment program implementation. The highest priorities located on central and northeastern portions of the study area. In some planning areas, the protected lands might present a constraint in fuels management program implementation (Appendix B).

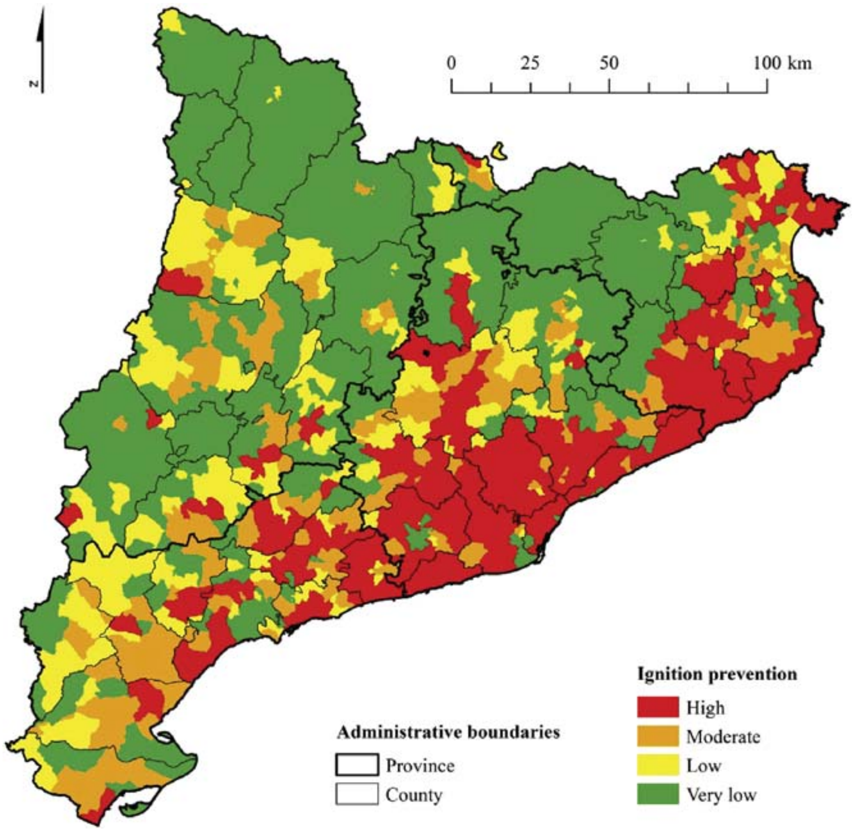

Fig. 8. Spatial prioritization map for human ignition prevention in Catalonia. We cross tabulated anthropic fire ignition densities (ANT quartiles; Fig. 3a) and transmission to residential houses (TF_RES quartiles; Fig. 6a) to prioritize ignition prevention program implementation. Coastal and metropolitan areas of Barcelona showed the highest priority.

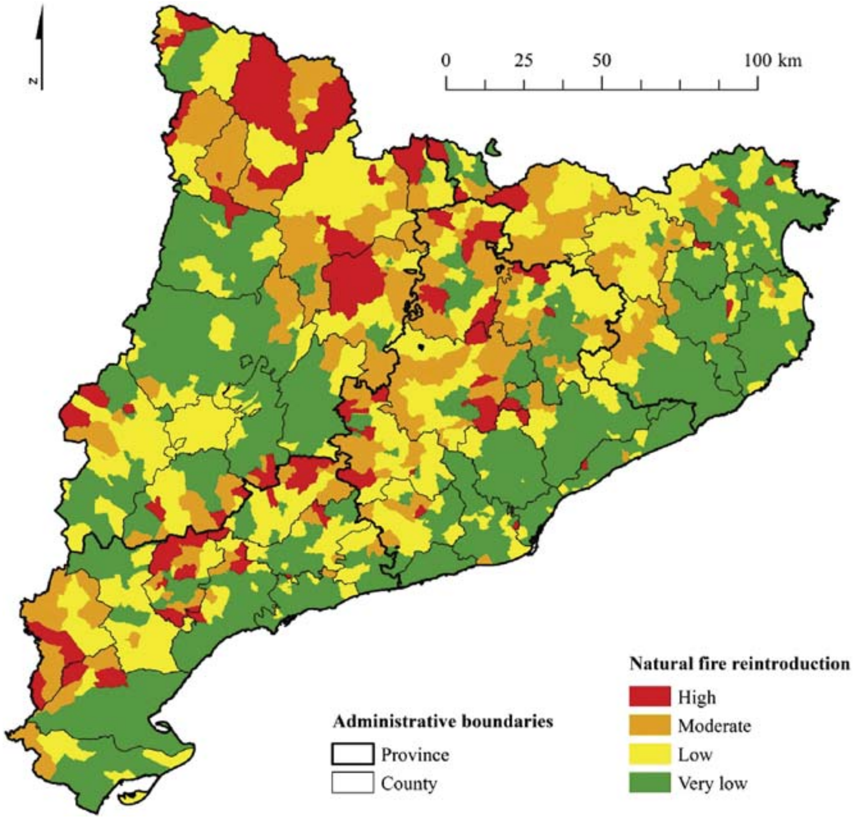

Fig. 9. Spatial prioritization map for natural fire re-introduction in forest ecosystems. We cross tabulated lightning fire ignition densities (NAT quartiles; Fig. 3b) and transmission to residential houses (TF_RES quartiles; Fig. 6a) to identify the most suitable areas for unplanned fire reintroduction. The municipalities with a highest potential located on remote mountainous areas were lightning fire reintroduction would not pose a risk to communities.

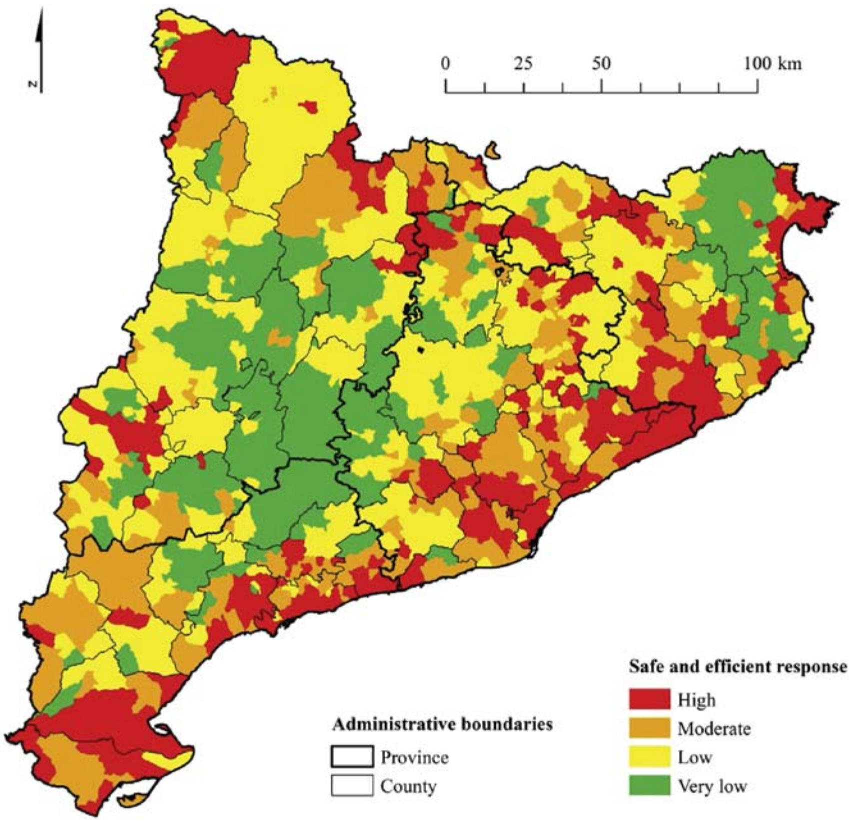

Fig. 10. Spatial prioritization map for a safe and efficient response in Catalonia. We cross-tabulated wildfire hazard on forest fuels (CFL levels; Fig. 4a) and transmission rate to communities (TR_RES; Fig. 6c) to identify the most suitable areas for an aggressive full suppression policy. Wildland-urban interface areas surrounded by managed fuels, predominantly agricultural plains and narrow valleys of the Pyrenees presented the highest priority.

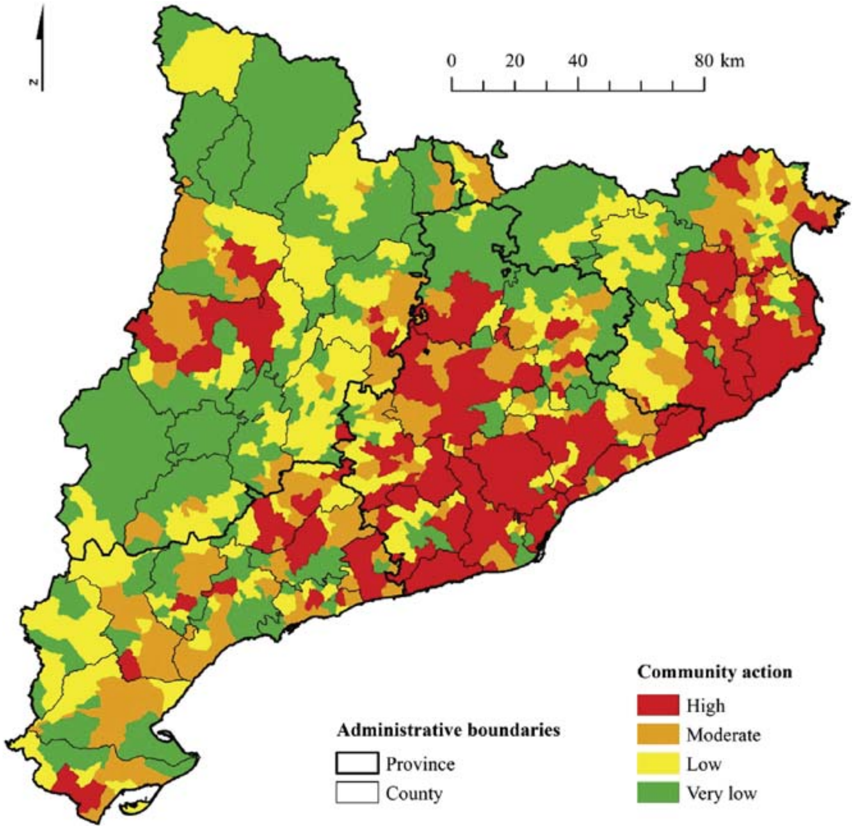

Fig. 11. Spatial prioritization map for community action program implementation in Catalonia. We cross-tabulated wildfire exposure values on the home ignition zone (annual HIBP quartiles; Fig. 4b) and the number dwellings on the WUI (quartiles; Alcasena et al., 2018) to identify the municipalities requiring a community action plan. The top priorities located on populated littoral and pre-littoral areas of Barcelona and Girona (Table 5).

Table 5
Community action (Fig. 11) priority municipalities (n = 25) in Catalonia, ranked by the highest number of structures on the wildland-urban interface (Alcasena et al., 2018a). The number of structures and the average overall exposure is detailed by wildland-urban interface classes. Largest human communities with the highest overall exposure values (annual HIBP > 0.0045) located on friction areas between the urban development and forestedlands of Barcelona and Gerona.

Municipality	Province	WUI class						Total	
		Disperse rural		Intermix		Interface			
		structures	avg. aHIBP	structures	avg. aHIBP	structures	avg. aHIBP	structures	avg. aHIBP
Rubí	Barcelona	1	0.0004	410	0.0053	3089	0.0044	3500	0.0046
Lliçà d'Amunt	Barcelona	1	0.0009	272	0.0040	2612	0.0055	2885	0.0051
Terrasa	Barcelona	7	0.0047	409	0.0090	2056	0.0113	2472	0.0106
Calonge	Girona	8	0.0047	469	0.0047	1944	0.0089	2421	0.0074
Mançanet de la Selva	Gerona	14	0.0008	415	0.0034	1992	0.0061	2421	0.0051
Begur	Gerona	4	0.0054	269	0.0050	1935	0.0050	2208	0.0050
Girona	Gerona	12	0.0081	322	0.0084	1833	0.0073	2167	0.0076
Cervelló	Barcelona	5	0.0031	170	0.0036	1957	0.0057	2132	0.0052
Palafrugell	Gerona	7	0.0068	261	0.0068	1855	0.0093	2123	0.0087
Santa Cristina d'Aro	Gerona	23	0.0050	359	0.0087	1682	0.0085	2064	0.0084
Caldes de Montbui	Barcelona	12	0.0022	316	0.0047	1507	0.0108	1835	0.0088
Santa Eulàlia de Ronçana	Barcelona	1	0.0047	158	0.0047	1653	0.0067	1812	0.0063
Castell-Platja d'Aro	Gerona	5	0.0084	385	0.0067	1316	0.0077	1706	0.0073
l'Amatlla del Vallès	Barcelona	1	0.0134	210	0.0055	1396	0.0065	1607	0.0062
Pals	Gerona	6	0.0079	187	0.0080	1335	0.0099	1528	0.0095
Castellbisbal	Barcelona	9	0.0029	172	0.0057	1335	0.0057	1516	0.0056
Esparreguera	Barcelona	5	0.0108	132	0.0081	1357	0.0071	1494	0.0075
Llagostera	Gerona	18	0.0028	194	0.0041	1280	0.0079	1492	0.0070
Palau-solità i Plegamans	Barcelona	0	0.0000	69	0.0051	1294	0.0073	1363	0.0071
Vilanova del Vallès	Barcelona	2	0.0065	208	0.0055	1067	0.0082	1277	0.0074
Sentmenat	Barcelona	8	0.0034	209	0.0036	984	0.0097	1201	0.0080
Olesa de Montserrat	Barcelona	6	0.0065	141	0.0065	1021	0.0072	1168	0.0070
Montcada i Reixac	Barcelona	2	0.0049	283	0.0071	878	0.0092	1163	0.0085
Sant Vicenç dels Horts	Barcelona	0	0.0000	194	0.0051	957	0.0045	1151	0.0047
la Roca del Vallès	Barcelona	7	0.0104	159	0.0037	962	0.0098	1128	0.0083

the suitable municipalities for fire re-introduction in forest systems (Figs. 8 and 9). Previous studies also showed that most ignitions in Catalonia were caused by humans, and suggested the implementation of ignition-cause-specific prevention measures on high ignition density areas to mitigate wildfire risk (Gonzalez-Olabarria et al., 2012, 2015). For instance, temporary bans to recreational uses on protected areas and public forests (Appendix B) during wildfire season could help reduce the number of unintended human ignitions. Since not all ignitions can potentially pose a threat to communities, we also considered fire transmission to residential houses (Fig. 6c). This way, we directed the implementation of prevention measures on areas with a high anthropic fire ignition density and a high transmission to communities. On the other hand, municipalities with high lightning ignition densities and low transmission to residential houses were a priori identified for re-introducing managed fire in the forest ecosystems, in parallel with pasture burning. Some sub-Mediterranean forest ecosystems are well adapted to low-intensity frequent fires and lightning fires could positively contribute to maintaining a fire-resilient forest structure with a minimal human intervention. For instance, endemic black pine old growth forests in central areas and pre-Pyrenees of Catalonia represent a good example of well adapted species to frequent surface fires (Fulé et al., 2008; Tíscar and Lucas-Borja, 2016). However, fire exclusion policies, poor forest-management practices (i.e., diametric cuts by just thinning the largest and the suitable trees for electric poles), and depletion of livestock transformed those forests into laddered fuel dense structures where high severity stand-replacing fires caused very substantial losses on past events (Martín-Alcón and Coll, 2016; Ordóñez et al., 2005). Thus, previous mechanical treatments and prescribed fires might be required to favor the resistance of remaining dominant seed trees before re-introducing the lightning fires.

We also identified the priority areas where fires spreading under extreme weather conditions might present some opportunities to safely and efficiently protect property (Fig. 10). In this study we considered wildfire hazard and transmission rate to residential housing metrics to rank priorities and other important factors that may compromise suppression efforts (e.g., rate of spread and spotting) were excluded from this first approach. Since burned area fire transmission might not always represent an effective exposure metric to communities, we considered the transmission rate to structures to demonstrate that a high transmission in terms of the burned area does not necessarily connote high potential for loss. This is the case for central Catalonia where fire transmission to neighboring communities is high (> 3.91 ha yr^{-1}; Fig. 6a), but the number of structures on the WUI is much lower than in coastal areas (Alcasena et al., 2018a), and therefore transmission rates are overall much lower (< 0.08 structures ha^{-1}; Fig. 6c and d). Specifically, our results could be used to strengthen ground crew and terrestrial resource allocation on high priority areas during wildfire season. Similarly, the development of an efficient transportation system and the increasing water pond density on these areas would allow a rapid response and a more effective aircraft work by reducing the time between discharges (Rodríguez y Silva et al., 2014). Even if aggressive full suppression alone is not the most effective way to mitigate structure loss in most fire-prone areas, it can exceptionally represent the main strategy for the municipalities located on intensively managed agricultural plains.

Lastly, we identified priority municipalities in Catalonia for the promotion of community action programs aimed at preventing wildfire disasters in the WUI (Fig. 11). In addition to annual high-intensity burn probability, we also considered the number of residential structures on the WUI matrix as a criterion to prioritize interventions on municipalities presenting a large number of dwellings surrounded by forest lands. In fact, more than 25 municipalities in Catalonia had more than 1 thousand residential houses on the WUI (Table 4). Community action measures on priority municipalities should consider treating fuels on the home ignition zone, using fire-resistant design and ignitable materials on structures, and reducing social vulnerability (Calkin et al., 2014; Paveglio et al., 2015; Penman et al., 2015). Currently the existing legislation in Catalonia requires homeowner and communities to manage fuels on the WUI (i.e., fuels treatment and maintenance in parcels within communities and a 25 m buffer), and our results could be used to technically justify and support the implementation of auto-protection plans on priority areas. Beyond ownership, local authorities can use ordinances on urban planning to exclude hazardous forestlands and prioritize the development in the safest areas within municipalities.

There are many socioeconomic and legal constraints to implement many of the management activities discussed in this research. Land ownership and environmental protection can especially constrain the implementation of fuel treatments programs. In Catalonia very large portions in forestlands are private, and public forests mostly concentrate in the northwestern side, where treatment priority is overall low (Appendix B; Fig. 7). While landowner risk perception on private properties is crucial for risk mitigation, management on public lands is strongly conditioned by social demands for multiple competing objectives (Fischer et al., 2014; Olsen et al., 2017; Paveglio et al., 2016). These include intensive timber management to supply local mills, pastures for extensive livestock breeding, leisure areas for urban citizens, and environmental conservation. Specific legislation regulates the protection degree (with direct implications in manageability) and establishes management restrictions to protect sensitive species habitat and the natural sites of special interest from negative human impacts. On the other hand, the spatial co-location between ongoing habitat restoration works and required risk mitigation treatments can provide an opportunity for conducting fuel treatment programs on many protected areas (Ager et al., 2017).

While economic and operational constraints can limit the extent and reach to required minimum treatment intensities on fire-prone landscapes (North et al., 2015), potential timber revenues from thinning can help overcome budgetary constraints on temperate forest ecosystems (Ager et al., 2017). However, economic opportunities from timber production are limited or nonexistent in Mediterranean landscapes, and thus subsidies continue to be necessary to implement large-scale fuel treatment programs. Accordingly, the annual budget to subsidize forest work attempts to compensate costs in fuel treatments. In particular, fuel treatment with a total cost up to a maximum base of 2000 € ha^{-1} is subsidized on public and private lands with the 75–100%, depending on the protected area designation (Appendix B) while requiring a forest management plan approved by the Forest Administration. Dense Aleppo pine regenerate cohorts ($> 10{,}000$ trees ha^{-1}) in central Catalonia from 1994 to 1998 large fire events ($> 20{,}000$ ha) represent a clear example of high priority areas where noncommercial treatments are required to convert hazardous forests into fire resilient landscapes (Verkaik and Espelta, 2006). In the longer term, the promotion of a circular economy where rural communities provide high-quality bio-based products to closer customers on densely populated urban areas may represent a promising solution to obtain required revenues for preserving fire resilient cultural landscapes (Lindner and Suominen, 2017; Verkerk et al., 2018).

Future efforts should be directed to downscaling within high priority planning areas or municipalities. On the one hand, fuel management programs would require an optimization analysis to design a cost-effective stand-level treatment mosaic while considering all the previous economic and environmental constraints (Alcasena et al., 2018b). Similarly, highly exposed communities should develop their own protection and management plans from higher-resolution and structure-level exposure and risk estimates (Alcasena et al., 2017). Concerning a fire response aimed at reintroducing lightning fires into natural acosystems, our approach represented a preliminary step and a more detailed study is required to accurately delineate the extent of the areas on remote municipalities areas where lightning fires pose a minimal risk to property and could positively contribute to fire-adapted ecosystem conservation (Barnett et al., 2016; Riley et al., 2018). Assessing fire containment probability at high-resolution on suitable municipalities for a full suppression policy would help identify the strategic locations where opportunistic firefighting efforts would likely result effective in controlling fires (O'Connor et al., 2017). Additionally, exploring the landscape management complexity using algebraic and topological methods, the analysis of fire transmission networks, and the implementation of human community clustering techniques would result useful in future research to complement our geospatial priority maps and help develop the most convenient fire policy at the municipality level (Evers et al., In press; Palaiologou et al., 2018; Papadimitriou, 2012, 2013).

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jenvman.2018.10.027.

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The wildland-urban interface raster dataset of Catalonia

Fermín J. Alcasena [a,*], Cody R. Evers [b], Cristina Vega-Garcia [a,c]

[a] Agriculture and Forest Engineering Department (EAGROF), University of Lleida, Alcalde Rovira Roure 191, 25198 Lleida, Catalonia, Spain
[b] Portland State University, Department of Environmental Science and Management, PO Box 751, Portland, OR 97207, USA
[c] Forest Sciences Centre of Catalonia, Carretera de Sant Llorenç de Morunys km 2, Solsona 25280, Catalonia, Spain

A B S T R A C T

We provide the wildland urban interface (WUI) map of the autonomous community of Catalonia (Northeastern Spain). The map encompasses an area of some 3.21 million ha and is presented as a 150-m resolution raster dataset. Individual housing location, structure density and vegetation cover data were used to spatially assess in detail the interface, intermix and dispersed rural WUI communities with a geographical information system. Most WUI areas concentrate in the coastal belt where suburban sprawl has occurred nearby or within unmanaged forests. This geospatial information data provides an approximation of residential housing potential for loss given a wildfire, and represents a valuable contribution to assist landscape and urban planning in the region.

Keywords:
Wildland-urban interface
Wildfire risk
Urban planning
Human communities
Catalonia

Specifications Table

Subject area	Environmental sciences, forestry, urban planning
More specific subject area	Natural hazards

* Corresponding author.
E-mail address: ferminalcasena@eagrof.udl.cat (F.J. Alcasena).

Type of data	Geospatial data
How data was acquired	*Does not apply*
Data format	Raster file (*.tif)
Experimental factors	*Does not apply*
Experimental features	We used a geographical information system (GIS) analysis to reclassify the residential housing at pixel level into different classes considering structure density and the surrounding vegetation.
Data source location	Autonomous community of Catalonia (Spain).
Data accessibility	The public repository of the University of Lleida: http://hdl.handle.net/10459.1/60480
Related research article	Martinuzzi, Sebastán; Stewart, Susan I.; Helmers, David P.; Mockrin, Miranda H.; Hammer, Roger B.; Radeloff, Volker C. 2015. The 2010 wildland-urban interface of the conterminous United States. Research Map NRS-8. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 124 p. [includes pull-out map]. https://doi.org/10.2737/NRS-RMAP-8.

Value of the Data

- Locations of valued assets within WUI can help prioritize risk mitigation activities at fine scales, including fuel treatments, ignition prevention programs, and evacuation or self-protection plans.
- These geospatial information data can be used to promote fire adapted communities when used in combination with fire modeling results and studies addressing social vulnerability.
- WUI maps can inform wildfire risk management in densely populated communities where large numbers of residential houses are exposed to recurrent wildfire risk.
- The WUI raster dataset can assist urban planning and policy making at a wide range of scales, from local to regional.

1. Data

The raster dataset of this article includes a detailed assessment (150-m resolution) of the wildland-urban interface for the 3.21 million ha autonomous community of Catalonia (Northeastern Spain) (Fig. 1). The WUI is the area where residential structures intermingle with hazardous vegetation and where most housing losses and human fatalities are concentrated in catastrophic wildfire events [1–4]. This WUI raster map contains non-vegetated low housing density, non-vegetated high housing density, vegetated (no housing), dispersed rural housing, intermix housing and interface housing classes [5].

Dispersed rural, intermix and interface community classes respectively occupy 0.61% (19,559 ha), 2.96% (94,955 ha) and 7.16% (229,952 ha) (Fig. 2A) of Catalonia. Interface WUI occupies the widest areas in the coastal belt, and intermix WUI is more typical in central Catalonia and the Pre-Pyrenees region to the north. In the southwestern plain of Lleida, both interface and intermix WUI areas are limited due to large areas of irrigated agricultural lands. Here, only residential houses constructed on the transition edges between irrigation and dryland or forest patches are classified as WUI. Although the majority of residential house structures ($>60\%$) are located in the interface WUI ($n=517{,}571$ structures), intermix ($n=$ 93,113 structures) and disperse rural classes ($n=8{,}693$ structures) still account for a substantial number of structures at risk to wildfire (Fig. 2B).

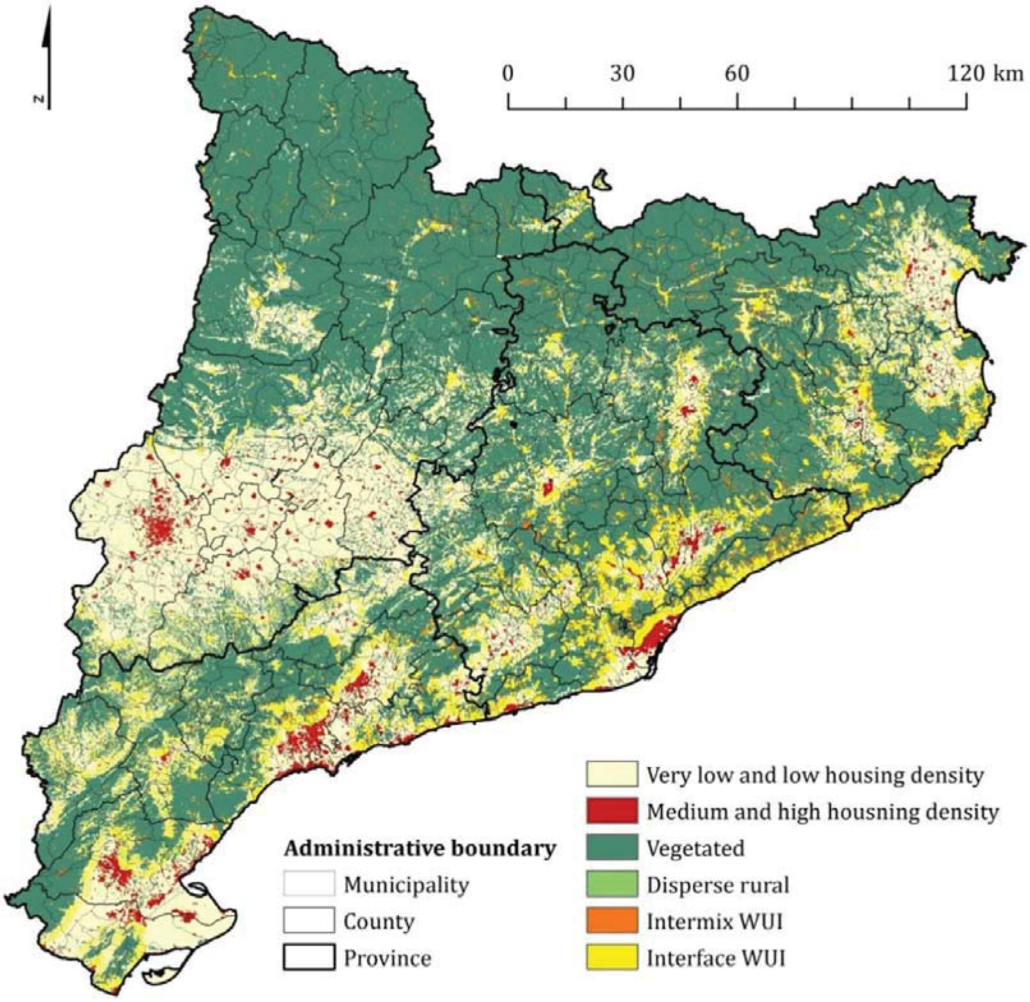

Fig. 1. The wildland-urban interface (WUI) map of Catalonia (Northeastern Spain). The residential houses built in wilderness areas and inland traditional rural farming communities are typical examples of intermix WUI. Interface WUI communities are located primarily in highly populated metropolitan areas along the Mediterranean coast. See geospatial analysis section for further details about each WUI class.

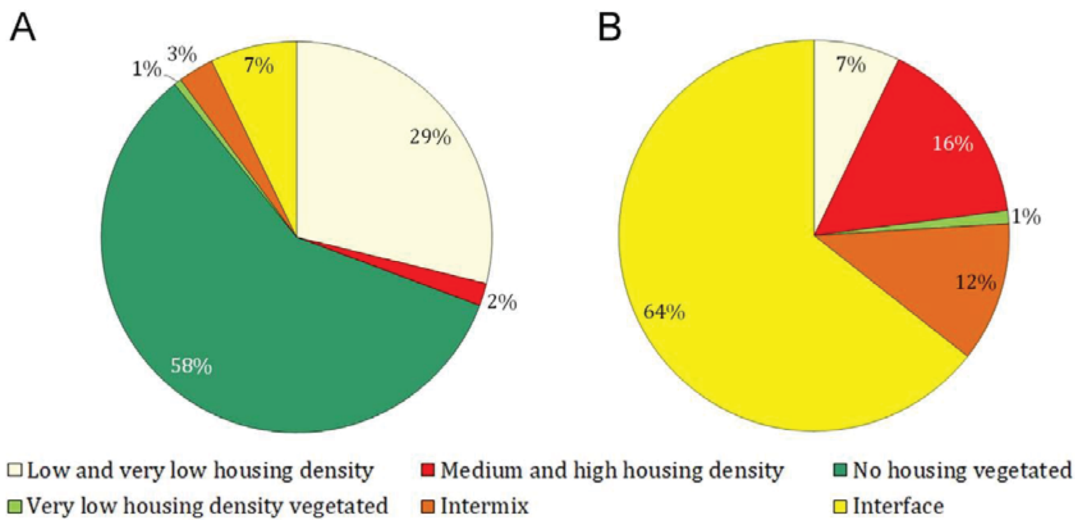

Fig. 2. Proportions of WUI classes in Catalonia based on (A) total area and (B) structure count.

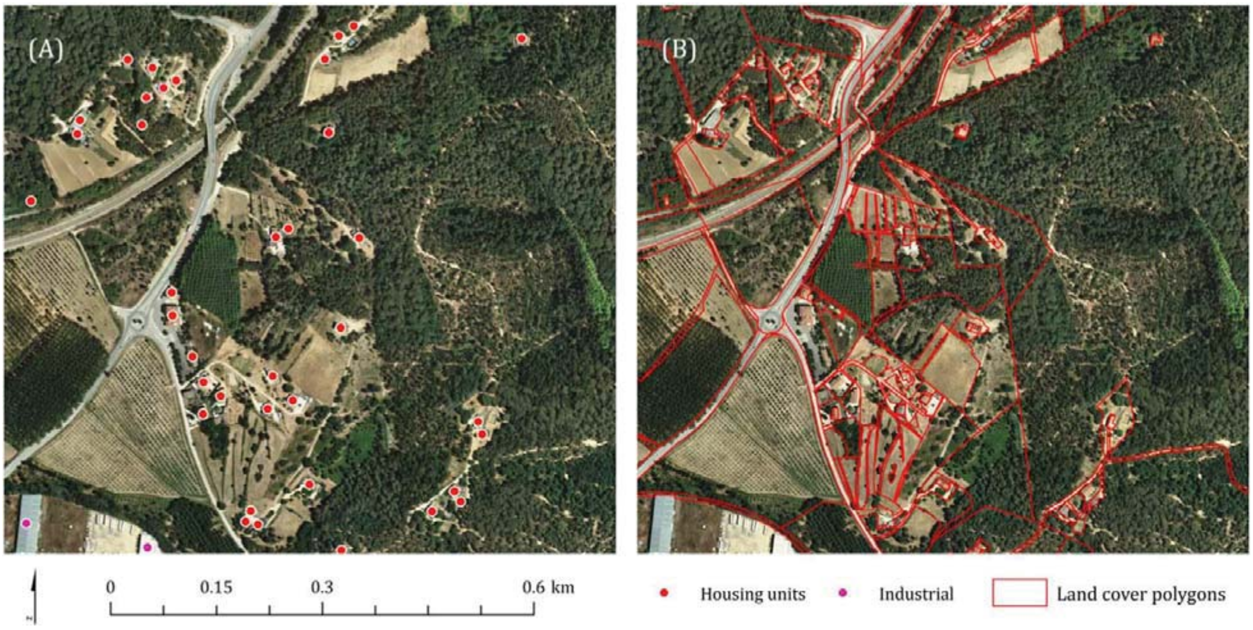

Fig. 3. Closed-up view sample of the vegetation and structure location data used to generate de WUI map for Catalonia in the municipality of Massanes. (A) Individual housing structure centroids were used to calculate structure density [6]. (B) Land cover polygons were used to identify the forest land and tree covered-forested areas [7].

2. Experimental design, materials, and methods

2.1. Residential housing and vegetation data

We mapped residential houses from the 1:25,000 scale Spanish national topographic platform (BTN25) map [6] (Fig. 3A). The BTN25 is the reference cartography used by municipalities and other official governmental agencies for multiple landscape and urban planning purposes and to accurately identify locations of individual structures. We did not distinguish between different type of residential houses (e.g., rural, housing block or chalet) and we excluded industrial structures, commercial buildings, and agricultural warehouses. We used the 2016 land parcel identification system (LPIS) vegetation map of Catalonia [7] to map areas where hazardous fuels can carry fire, torch stands, and threaten structures (Fig. 3B). We identified forest land polygons using the definition in Article 5 of the National Forest Law 43/2003, of 21 November, which includes natural pastures, shrubby pastures, open woodlands and tree covered-forested land.

2.2. Geospatial analysis

We used housing density, forest cover, and ember exposure grids in the WUI classification. Reference values for structure density and forest cover in the classification were obtained from similar studies that classified the WUI in other fire-prone areas [5,8]. Risk exposure from ember showers was assumed to occur up to 2 km from forested lands based on the large-fire spotting distances observed in Catalonia [9]. All data were compiled at a 150 m resolution projected at ETRS89 UTM 31N coordinate system.

To construct the housing density layer, we first extracted individual residential house polygons from the BTN25 map (n=801,336) to generate a point file with structure location centroids. In the absence of census block information (a data source commonly used in similar US studies), we considered a 450 m regular grid (20.25 ha) as reference to calculate structure density at 150 m resolution. Pixels containing development were then reclassified as very low ($<$6.18 houses km^{-2}), low ($\geq$6.18–$<$49.42 houses km^{-2}) and medium-high ($\geq$49.42 houses km^{-2}) density.

We generated the vegetation cover and ember exposure grids using the LPIS forest land polygons. To generate the vegetation cover grid we converted forest land polygons into a 150-m resolution raster grid and reclassified the pixels into vegetated ($\geq$ 50% cover) and non-vegetated ($<$ 50% cover). Concurrently, to generate the ember exposure grid, we merged contiguous forested area polygons into larger blocks (the LPIS database subdivides polygons according to land ownership), and used resulting polygons >5 km^2 to identify non-vegetated interface areas within a 2 km buffer that may be exposed to ember showers during catastrophic events.

Finally, we combined the three previous grids and assigned each 150-m raster grid cell as one of the 6 following classes:

- (1) Very low and low housing density: Forest land cover $<50\%$, housing density ≤ 49.42 houses km^{-2}, and >2 km from a forested land area ≥ 5 km^2 in size.
- (2) Medium and high housing density: Forest land cover $<50\%$, housing density ≥ 49.42 houses km^{-2}, and >2 km from a forested land area ≥ 5 km^2 in size.
- (3) Vegetated: Forest land cover $\geq 50\%$ and no housing.
- (4) Dispersed rural: Housing density <6.18 houses km^{-2} and forest land cover $\geq$ 50%.
- (5) Intermix WUI: Housing density ≥ 6.18 houses km^{-2} and forest land cover $\geq$ 50%.
- (6) Interface WUI: Housing density ≥ 6.18 houses km^{-2}, forest land cover $<50\%$ and houses located $<$ 2 km from a forested area $\geq$ 5 km^2 in size.

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Transparency document. Supporting information

Supplementary data associated with this article can be found in the online version at http://dx.doi.org/10.1016/j.dib.2017.12.066.

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GENERAL DISCUSSION

Despite escalating investments in fire suppression resources during the last decades, recent catastrophic episodes of Portugal and Greece are strong evidence that the current exclusion policy is failing to protect humans, property and natural values from wildfires (Castellnou and Miralles 2009; Otero and Nielsen 2017). Moreover, often tragic entrapment episodes during these events continue to pose a real threat not only to civilians escaping from flames but also to well-trained and equipped firefighting crews (Cardil *et al.* 2017; Molina-Terrén *et al.* 2019). Fire-prone areas in the Mediterranean regions need a radical new strategy to cope with contemporary fires, and this Ph.D. Thesis advances a framework to help develop a wildfire risk management comprehensive strategy for southern European countries.

In southern European areas, fire-regimes are not fuel-limited since the fragmented cultural landscape is rapidly evolving to dense regenerating forests where managed agricultural lands reduce to valley bottoms (Seijo and Gray 2012; Cervera *et al.* 2016). As a result, extreme events spreading over this hazardous fuel continuum resist suppression efforts and burn large areas (Tedim *et al.* 2018). At the same time, the suburban sprawl that first developed on peri-urban open plains is now creating highly exposed human communities while expanding on the wildland-urban interface (Pellizzaro *et al.* 2012). To cope with this evolving scenario, we require a new approach beyond the full suppression and ignition prevention policy (Bovio *et al.* 2017). In this Ph.D. Thesis, I purpose a long-term strategy aimed at creating fire resilient landscapes, restoring cultural fire regime, facilitating safe efficient fire response, and creating fire-adapted communities (USDA Forest Service 2014; Alcasena *et al.* 2019a).

The massive fires spread over long distances and fire occurrence models derived from historical ignition locations are bad predictors for burned areas (Miller and Ager 2013). On the other hand, fire spread models can be used to simulate fire growth and replicate historical burn patterns on Mediterranean landscapes (Oliveira *et al.* 2016; Salis *et al.* 2016a). Nonetheless, humans are responsible for the vast majority of fires and assuming random distributions would markedly overestimate the ignition probability on poor-access remote areas with a limited lightning activity (Costafreda-Aumedes *et al.* 2017). Also, firefighting resources are very efficient in controlling most ignitions by initial attack, and thus, a fire ignition by itself does not necessarily imply a risk (Rodrigues *et al.* 2019). This complexity was captured in this Thesis through the use of fire occurrence models to generate the ignition location input data required for fire simulation modeling, and assuming that all ignitions occurring under extreme weather conditions (>97th percentile) escape first-response fire-contention efforts. The burn probability modeling results captured the spatial variability and changing shapes and extent of fire perimeters across the landscape, and provided a valuable wildfire likelihood estimate. Definitely, the burn probability is a crucial wildfire risk causative factor, essential for quantitative risk assessment (Finney 2005; Scott *et al.* 2013).

In Mediterranean areas, the WUI problem has been usually addressed as a local issue, where wildfire risk has been almost exclusively attributed to the individual homeowners or residential neighborhoods. Accordingly, prevention plans cope with administrative divisions and with urban planning and focus on preventing ignitions and reducing wildfire hazard (potential for loss) by implementing fuel treatments on structure vicinities within the home ignition zone (Badia *et al.* 2011; Herrero-Corral *et al.* 2012). Alternatively, housing-unit clustering patterns and surrounding forest fuel types have been often used to characterize the wildfire risk in the wildland-urban interface (Lampin-Maillet *et al.* 2010; Sirca *et al.* 2017; Fernández-Álvarez *et al.* 2019). In this Thesis, I demonstrate that previous works provide a partial solution to problem-fires since those cross-boundary events would continue to expose a large number of structures in the WUI areas. Far from being a local issue, wildfire risk management activities on neighboring communities and forested areas hold direct implications. Understanding the scale of risk to communities and the identification of individual housing structures expecting highest losses would help inform risk perception and identifying responsible landowners in implementing mitigation strategies (Ager *et al.* 2016). Ultimately, the wildfire risk management scale and the treatment priorities aimed at protecting values at risk are strongly determined by large fires. To cope with this fact, I first propose replacing the administrative division by the community fireshed extent in management plans aimed at protecting human communities (Bahro *et al.* 2007). Indeed, community firesheds extend far beyond the administrative division boundaries in fire-prone areas (Ager *et al.* 2018). This approach would allow concentrating the landscape fuel treatments in specific "high-priority" portions of the landscape at effective intensities and exclude the vast forested areas where managing fuels would not have any effect in mitigating wildfire exposure to communities (Alcasena *et al.* 2019b). And second, I encourage using wildfire metrics that integrate wildfire likelihood and intensity to target and prioritize HIZ fuel management on housing unit clusters within large communities (Dillon *et al.* 2015; Alcasena *et al.* 2019a). Again, modeling large fire spread is essential to understand wildfire exposure to the WUI (Haas *et al.* 2013; Salis *et al.* 2013). Understanding which are the housing units or structure clusters where the fire intensity and burn probability is the highest is fundamental to prioritize fuel treatments. Exceptionally, using susceptibility relations and structure level economic values where available would allow assessing expected economic losses (Alcasena *et al.* 2017).

From among all the existing management options, fuels reduction is the main long-term management option to mitigate uncharacteristic and destructive events in fire-prone Mediterranean areas. Preventing all human ignitions is impossible, firefighting efforts are ineffective during extreme events, thousands of structures on the WUI are totally embedded in hazardous fuels, and the aging population in rural areas plus lack of fire-risk-awareness in crowded touristic sites makes impossible protecting the whole population during catastrophic events. In this Thesis, I explore the existing collocation opportunities for prioritizing and scheduling treatments in order to share the cost for treatments on strategic locations. Public forest managers, private forest owners, and human communities in the WUI, all can benefit from the treatment collocation opportunities. All these findings highlight the collaborative effort needs between fire-connected neighboring communities and the individual housing units or homeowners within them (Palaiologou *et al.* 2018). Currently, most forest management subsidies present separate and competing subsidies that distinguish between land ownership (private, public owned by municipalities, and public managed by the regional government), land designation or the different degrees of protection, and the treatment objective (e.g., wildfire hazard reduction, pasture improvement, game species habitat improvement, and endangered species habitat restoration). Moreover, most treatments are designed and implemented according to expert criteria, and I used quantitative metrics in the strategic treatment design, which facilitated comparisons and spatial prioritization within large areas. All the results are provided at operational scales as required by the landscape and wildfire managers implementing the fuel treatments.

Ignoring the wildfire management implications of the large fire exchange patterns, as well as the existing interactions between human ignition source areas and risk receptors, would lead to poor prescriptions in preventing catastrophic events. On the other hand, the fire policies developed from the deep understanding of major risk causative factors and feasible management options would provide a consistent solution to deal with cross-boundary uncharacteristic fires. While increasing budgets are allocated to mitigate wildfire risk on vast landscapes, a minimal effort is made to unveil the real opportunities for facilitating implementing appropriate management options on the areas where these efforts would results effective. In this Thesis, I present a comprehensive strategy to prioritize widely accepted risk management options aimed at achieving four core goals (Alcasena *et al.* 2019a). Methods from this approach integrate wildfire occurrence, accurate structure location data on the WUI, and wildfire exposure metrics from 10,000 wildfire season modeling results for capturing variability in the factors (e.g., changing fire-weather scenarios) driving burn patterns and potential conditioning effects across the landscape. Currently, most decision-making for wildfire management at the regional scale is conducted considering the incidence of historic events, which represent real outcomes from a minimal set of potentially occurring fire-conditioning scenarios (< 20-30 wildfire season or years). My results were presented in a set of priority maps which revealed very strong spatial patterns and allowed identifying best areas for fuel treatments, ignition prevention, natural fire re-introduction, fire suppression, and community action. Methods from this Thesis can be replicated and adapted elsewhere considering the local conditions in terms of fire ignition distributions, fuel loads and fire-weather scenarios. Ultimately, this framework was developed to serve as a baseline and help develop a long term comprehensive strategy for fire-prone southern European landscapes.

Future research lines may consider combining firefighting efforts in fire spread modeling, assessing cost-efficiency for management options, a management-oriented characterization for the WUI, integrating social vulnerability in the risk assessment framework, assessing cultural and lightning fires feedbacks in reducing large fire spread, and a long-term sustainability plan for management prescriptions. Coupling the existing fire spread models with the models that predict fire control by initial attack, as well as escaped large fire spread contention, would provide more realistic and accurate burn probabilities (O'Connor *et al.* 2016; Rodrigues *et al.* 2019). Latest WUI studies used urban development patterns and vegetation gradients to cluster community archetypes and then evaluate risk mitigation options according to large wildfire exposure profiles (Evers *et al.* 2019). Integrating social vulnerability and community preparedness would help better understand human entrapment and home destruction during catastrophic events (Paveglio *et al.* 2016). Conversely, lighting re-introduction in remote areas could be allowed and even prescribed where unplanned ignitions pose a minimal risk to property and could positively contribute to fire-adapted ecosystem conservation (Barnett *et al.* 2016). Although reversing current landscape dynamics to the original fire-resistant mosaic-type cultural landscape seem impossible, promoting a sustainable forest management within a bio-based circular economy would make rural communities less subsidy-dependent, reduce fuel load and continuity, and make cultural landscapes less vulnerable to fires (Verkerk *et al.* 2018).

FINAL CONCLUSIONS

Continuing with a fire exclusion policy will only enhance the mega fire escalade in Mediterranean areas. We need to understand which are the triggering factors behind the massive fire episodes in order to develop a science-based wildfire management comprehensive and proactive strategy. Accordingly, there is a clear need to integrate the essential risk causative factors (fire occurrence, large fire spread and potential effects, both negative and positive) into a framework capable of prioritizing the most suitable management options locally and regionally. The implementation of management prescriptions would ultimately require a cross-scale collaborative solution, where different socioeconomic agents including citizens, homeowners and human communities in the WUI, forest landowners, public forest managers, and firefighting institutions assume their involvement in mitigating risk. In this context, current treatment collocation opportunities in multi-functional forest systems may play a vital role to overcome budgetary restrictions and make fuel reduction programs sustainable over time. Ultimately, the knowledge of existing relations between fires and humans on rewilding Mediterranean cultural systems will determine the long-term success of the wildfire management policies.

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