

# Modelling the Ecological Vulnerability to Forest Fires in Mediterranean Ecosystems Using Geographic Information Technologies

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**Abstract** Forest fires represent a major driver of change at the ecosystem and landscape levels in the Mediterranean region. Environmental features and vegetation are key factors to estimate the ecological vulnerability to fire; defined as the degree to which an ecosystem is susceptible to, and unable to cope with, adverse effects of fire (provided a fire occurs). Given the predicted climatic changes for the region, it is urgent to validate spatially explicit tools for assessing this vulnerability in order to support the design of new fire prevention and restoration strategies. This work presents an innovative GIS-based modelling approach to evaluate the ecological vulnerability to fire of an ecosystem, considering its main components (soil and vegetation) and different time scales. The evaluation was structured in three stages: short-term (focussed on soil

degradation risk), medium-term (focussed on changes in vegetation), and coupling of the short- and medium-term vulnerabilities. The model was implemented in two regions: Aragón (inland North-eastern Spain) and Valencia (eastern Spain). Maps of the ecological vulnerability to fire were produced at a regional scale. We partially validated the model in a study site combining two complementary approaches that focused on testing the adequacy of model's predictions in three ecosystems, all very common in fire-prone landscapes of eastern Spain: two shrublands and a pine forest. Both approaches were based on the comparison of model's predictions with values of NDVI (Normalized Difference Vegetation Index), which is considered a good proxy for green biomass. Both methods showed that the model's performance is satisfactory when applied to the three selected vegetation types.

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## Introduction

During the second half of the 20th century, profound socioeconomic changes affecting most northern Mediterranean countries implied many farmlands being abandoned (Moreira and others 2011), which have generally reverted to semi-natural vegetation, often leading to fire-prone communities (Baeza and others 2002). Furthermore, large-scale afforestation plans fundamentally based on flammable species such as *Pinus* spp. (Dimitrakopoulos 2001), were extensively implemented in many of these countries (Vallejo and Alloza 1998). The resulting accumulation of flammable fuels over large parts of these landscapes has

been one of the major causes of the drastic alteration of fire regimes in recent decades (Vallejo and Alloza 1998; Pausas and Keeley 2009). In Spain, the number of fires and areas burned increased in the mid 1970s, and this trend has remained for fire occurrence (Pausas 2004; MMA 2006).

Fire is a major ecological factor in Mediterranean ecosystems and major changes in such regimes can have serious consequences at both ecosystem and landscape levels (Zedler and others 1983; Díaz-Delgado and others 2002). Indeed, wildland fires have become one of the major causes of land degradation (soil loss, decreasing soil fertility and vegetation cover) in Mediterranean landscapes (Vallejo and others 2009).

Most climate change predictions imply increased air temperatures and less summer rainfall for the Mediterranean basin (Hertig and Jacobeit 2008; Schröter and others 2005). The resulting unfavourable shift in moisture availability could favour large fire occurrence and severity (Pausas 2004).

In this context, fire risk spatial evaluation (defining the risk as the product of hazard and vulnerability; <http://glossary.eea.europa.eu/>) is critical to design and implement fire prevention and post-fire restoration strategies at ecosystem and landscape levels as it allows monitoring when and where a fire is more likely to occur, or its more negative effects (Chuvieco and others 2010).

Although vulnerability aspects (relating to the potential damage of a given hazard) are frequently considered in assessment systems for most natural hazards (Bachmann and Allgower 2001), they are not generally included in the operational fire danger indices used in European Mediterranean countries, which mostly rely on meteorological indices (Chuvieco and others 2010). Initially, a conceptual fire risk assessment model was developed within the European Spread Project, and afterwards within the FIREMAP project (Chuvieco and others 2010), and includes two aspects: fire hazard (ignition or propagation potential) and vulnerability (potential damage). This paper focuses on the ecological vulnerability assessment (not including socio-economic vulnerability) included in that fire risk model's vulnerability component (Alloza and others 2006).

Vulnerability has many different definitions. Based on the definition proposed by IPCC (2007), it is understood here as the extent to which a system is susceptible to, and unable to cope with, adverse effects of any driver of change or hazard (fire, in this case). Ecological vulnerability can be used at several hierarchical levels, but is applied here as synonym for ecosystem vulnerability, defined as the inability of an ecosystem (biological community and habitat combined) to tolerate stressors over time and space (Williams and Kapustka 2000; <http://glossary.eea.europa.eu/>). Ideal ecosystem vulnerability assessments should involve the abiotic

environment, the system's different organisation levels and several temporal and spatial scales (Ippolito and others 2010). A comprehensive vulnerability analysis is, thus, unrealistic and, in practice, "reduced" assessments are conducted (Turner and others 2003).

Effects of fire on soils and post-fire vegetation dynamics have been studied in Mediterranean basin ecosystems (Arianoutsou-Faraggitaki 1984; Trabaud 1994; Giovannini and Lucchesi 1997; Kazanis and Arianoutsou 2004; Duguay and others 2007b; Duguay and Vallejo 2008). However, integrated, spatial approaches considering the whole ecosystem's short- to long-term response to fire are scarce, mainly due to the difficulty of establishing appropriate long-term field-based experimental designs for documenting such responses.

Analysing different ecological processes requires various process-specific resilience assessments (Lavorel 1999) with several temporal scales. Post-fire soil degradation caused by water erosion is expected to occur from fire extinction to some months afterwards when vegetation cover recovery is still poor (Pausas and Vallejo 1999). However, re-establishing pre-fire plant communities spans several years, even decades, depending on pre-existing vegetation, fire regime and environmental conditions (Baeza and others 2007; Keeley 2009).

In the integrated fire management strategy context, involving minimizing fire risk and promoting resilience to fire and biodiversity in Mediterranean landscapes, it is essential to predict how ecosystems may evolve in the short to long term after fire. Methodologies combining remote sensing and GIS tools have been recently proposed to assess particular post-fire processes (e.g., erosion) at the landscape level and to, thus, support post-fire management (Ruiz-Gallardo and others 2004; Bisson and others 2008). Arianoutsou and others (2011) presented a GIS-based multi-criteria evaluation approach for a comprehensive ecological knowledge-based assessment of the post-fire recovery ability of *Pinus halepensis* forests.

This paper presents a conceptual scheme for assessing an index of ecological vulnerability to forest fires in order to integrate major ecosystem properties (through the responses of two major components: soil and vegetation) that determine its potential for modulating response to fire over time and space. The theoretical, modelling approach, combining the use of remote sensing and GIS, considers short-term post-fire potential degradation of soil and vegetation's short/mid-term recovery capacity. It was implemented for Mediterranean ecosystems of the Iberian Peninsula within the FIREMAP project framework (Alloza and others 2006; Chuvieco and others 2010). It also presents the results of a partial validation approach proposed for our ecological vulnerability assessment model.

## Methods

### Conceptual Scheme for Ecological Vulnerability Assessment

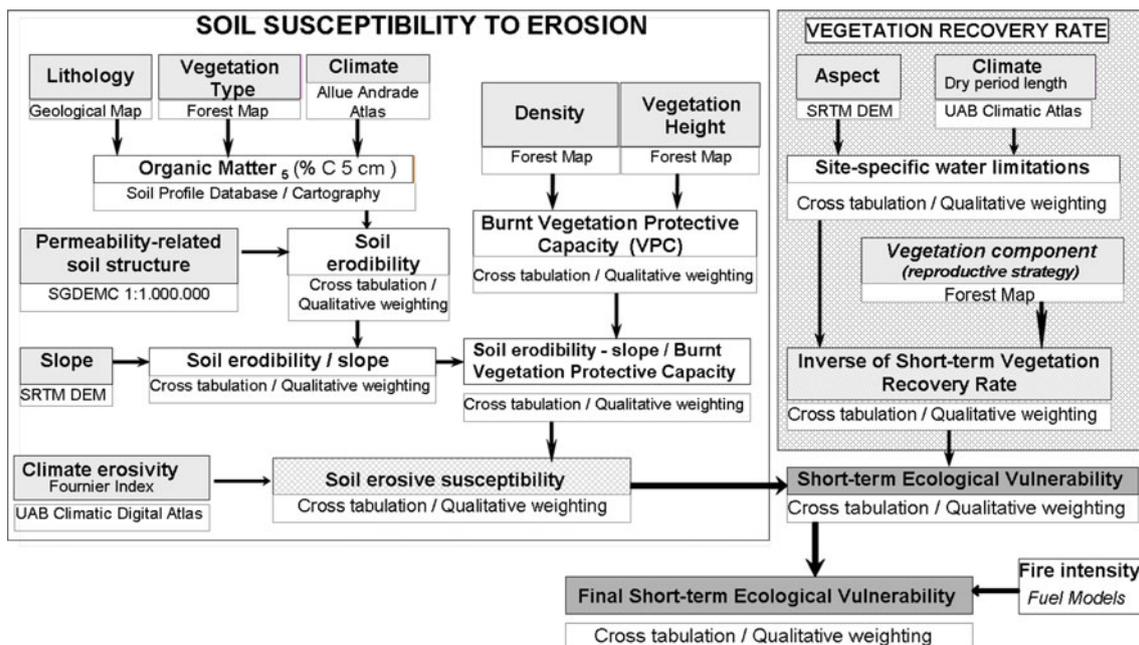
Our assessment approach was to be as integrative as possible by considering the abiotic environment, the biological community and various temporal scales.

Early post-fire stages are critical for assessing ecosystem vulnerability to fire. Some fire-caused impacts (e.g., on soil nutrient balance) may result in relevant direct or indirect effects on various system parts over longer temporal scales. Post-fire plant establishment patterns, for instance, can be influenced by fine-scale patterns of ash depth and soil properties (Moreno and Oechel 1992; Rice 1993).

In Mediterranean ecosystems, characterised by low productivity (generally due to water and nutrients limitation), fire may change soil properties that are difficult to predict and to observe in the short term (subtle, spatially heterogeneous, not clearly identified), but can significantly alter the availability of essential nutrients in early post-fire succession stages, thus crucially affecting ecological processes and interactions (e.g., germination, seedling establishment, inter-plant species competition). The latter strongly determine post-fire vegetation over longer time scales.

Moreover, the long, spatially heterogeneous and often unknown history of disturbances (e.g., fire and land use) characterising Mediterranean ecosystems renders the predictions of the effects of potential fire very difficult. Indeed, the inference of post-fire mid-term vegetation from short-term studies is not always evident, even under a low frequency fire regime, particularly when vegetation is dominated by species with irregular germination (Rodrigo and others 2004). Pine forests may change considerably from a single fire (Rodrigo and others 2004; Baeza and others 2007).

Hence, the assessment of ecological vulnerability to forest fires was made on two temporal scales (short and mid terms after fire) with three stages. The first was a short-term vulnerability analysis (under 1 year after fire), by estimating the post-fire soil degradation risk (Fig. 1), and the second a mid-term vulnerability analysis (20/25 years after fire), by assessing *vegetation's capacity of persisting with no changes* in species composition or structure (considering species' relative abundance and vertical structure). This parameter was considered the most explanatory indicator (among the easiest to assess) of the whole ecosystem's (in)ability to cope with fire over that temporal scale. We assumed that any ecosystem whose vegetation is highly capable of mid-term post-fire persistence, as previously defined (with no short-term critical damage) is globally



**Fig. 1** Conceptual scheme for the short-term vulnerability analysis (main data sources are indicated) *Geological Map*: Mapa Geológico de España (1:50.000). Instituto Geológico y Minero de España, Madrid. *Forest Map*: Mapa forestal de España (1:50 000), 1990–1996. Dirección General de Conservación de la Naturaleza, Ministerio de Agricultura, Pesca y Alimentación, Madrid. Allue Andrade JL (1990) Atlas fitoclimático de España. Taxonomías. Ministerio de

Agricultura, Pesca y Alimentación, Instituto Nacional de Investigaciones Agrarias, Madrid, pp 221 *SGDEM*: Soil Geographical Database of Euro-Mediterranean Countries, <http://eu soils.jrc.ec.europa.eu/> *UAB Climatic Digital Atlas*: <http://opengis.uab.es/wms/iberia/> *SRTM DEM*: Shuttle Radar Topography Mission Digital Elevation Model <http://www2.jpl.nasa.gov/srtm/index.html>

considered able to tolerate fire over time and space; i.e., low vulnerability. Other structural or functional characteristics that are more difficult to assess (e.g., aboveground plant biomass) have already recovered from a pre-disturbance situation, or will do so (if a longer time span is needed), unless a new disturbance occurs.

The third stage estimated a synthetic index of ecosystem vulnerability to forest fires by integrating the short- and mid-term vulnerability assessments (Fig. 1).

The mid-term assessment time span was sufficiently long to allow most forest-type ecosystems to recover from a similar pre-fire situation (or clearly tending towards it), but short enough to avoid succession-related processes in most vegetation types. Our model was particularly devised to be applied in low productivity ecosystems with shrub and tree species, especially pine forests as they are among the most extended communities most affected by fire in the Mediterranean region (Arianoutsou and others 2011). Post-fire vegetation dynamics may be very slow in such systems (Baeza and others 2007) due to limiting abiotic interacting features (Vallejo and others 2000) and low seedling recruitment of resprouters (Vallejo and Alloza 1998). At one of our study sites (Ayora, E Spain), most ecosystems (forests, mixed shrublands with pines) affected by a large crown fire in July 1979 had not recovered their pre-disturbance situation 23 years (Baeza and others 2007) and 31 years (Duguay and others in preparation) after the fire.

Throughout the assessment process, the variables (most represented by a raster layer in ArcGIS) that were successively integrated into the model (Fig. 1) were qualitatively rated with *Low* to *Very High* categories (except soil permeability-related physical characteristics, and vegetation height). They were combined in the GIS into new integrated qualitative spatial variables following the model structure (Fig. 1). All the cross tabulation processes followed expert knowledge-based decision rules. In most cases, the integration of two qualitative categories (from two different variables) was rated by considering the worst case scenario vs. the vulnerability estimation; i.e. overvaluing the category leading to higher vulnerability.

As vulnerability assessments are carried out for potential fire events, fire regime and the post-fire weather conditions are unknown. However, these factors strongly determine post-fire vegetation dynamics. In this study, when attributing short- and mid-term vulnerabilities to vegetation types (App. B and C, Electronic supplementary material), we considered a low historical fire frequency (no more than 2 fires in the last 20 years, time interval between fires  $\geq 10$  years) where the time since the last fire was beyond 10 years and the worst case fire scenario under Mediterranean conditions: a summer fire with very low fuel moisture and similar climatic conditions to those of

summer period historical averages. High intensity severe fires (Keeley 2009) are, thus, expected.

Our model addressed a regional-scale mapping of vulnerability at a 100 \* 100 m spatial resolution (linked to the Digital Elevation Model (DEM) resolution), suitable for supporting regional-level planning. This work scale conditioned the input variables selection and rendered some generalisations and simplification processes necessary.

### Short-Term Vulnerability Analysis

An ecosystem's short-term post-fire ecological vulnerability, i.e., its response capacity for post-fire periods  $< 1$  year, depends on abiotic and biotic factors which determine the post-fire soil erosion risk. For any ecosystem, we considered (Fig. 1):

- abiotic factors relating to post-fire *soil susceptibility to erosion*
- vegetation characteristics and environmental conditions crucially influencing the *short-term vegetation recovery rate*. We evaluated vegetation's capacity for rapidly protecting bare soil from post-fire erosion processes in terms of the speed of the post-fire vegetation re-establishment.

Both soil erosive susceptibility (1) and the vegetation recovery rate (2) were integrated into the short-term ecological vulnerability (3).

### Soil Erosive Susceptibility

Despite the numerous modifications and criticisms to which the Universal Soil Loss Equation (USLE) was subjected, the empirical soil loss model by Wischmeier and Smith (1978) is still the major reference to predict post-fire soil erosion risks (Giovannini 1999).

In order to facilitate its application in this model, and given the digital cartography availability limitations, we used a qualitative approach for its use. We assumed the same factors in USLE: climate (rainfall erosivity index), soil erodibility, topography (slope) and land cover; then each was classified into three erosion susceptibility categories (low, medium and high) and integrated by reduction matrices. Similar analyses have been done in European projects for regional-scale cartographic applications, such as the CORINE soil erosion risk assessment for southern European Community regions (<http://www.eea.europa.eu/publications/CORO-soil>) or the Pan-European soil erosion risk assessment (PESERA) ([http://eussoils.jrc.ec.europa.eu/ESDB\\_Archive](http://eussoils.jrc.ec.europa.eu/ESDB_Archive)).

The criteria and information sources used for all four factors are:

- *Climate factor*: The Fournier Index (FI), the quotient between the square of the highest average monthly precipitation and average annual precipitation (for a defined period), was used as a climate erosive capacity indicator (De Luis and others 1997). To implement the model in the selected Spanish Mediterranean regions, we set three intervals based on the overall range of FI across the Valencia region (De Luis and others 1997) and expert opinion:  $FI < 60$ , low erosive susceptibility;  $60 \leq FI \leq 120$ , medium erosive susceptibility;  $FI > 120$ , high erosive susceptibility. The Climatic Digital Atlas of the Iberian Peninsula (<http://opengis.uab.es/wms/iberia/>) provides precipitation values, unless more accurate data are available (Pérez Cueva 1994).
- *Soil erodibility*: This parameter, influenced by numerous factors, is complex to estimate. Factor K in USLE model is a measure of the susceptibility of soil particles to detachment and transport by rainfall and runoff. It measures the influence of several soil properties (texture, organic matter content, structure and soil permeability) on soil susceptibility to erosion. Fire and degrading phenomena (e.g., rain) occurring after it mainly alter superficial soil layers (Debanó and others 1998), and particularly affect soil organic matter content and soil permeability. Our model's soil erodibility assessment was, therefore, based on two superficial layer characteristics: (a) organic matter (OM) content in the first 5 cm of top soil (OM<sub>5</sub>, hereafter); (b) soil permeability-related physical characteristics; e.g., top soil structure and soil crusting risk.

#### OM Content in the First 5 cm of Top Soil

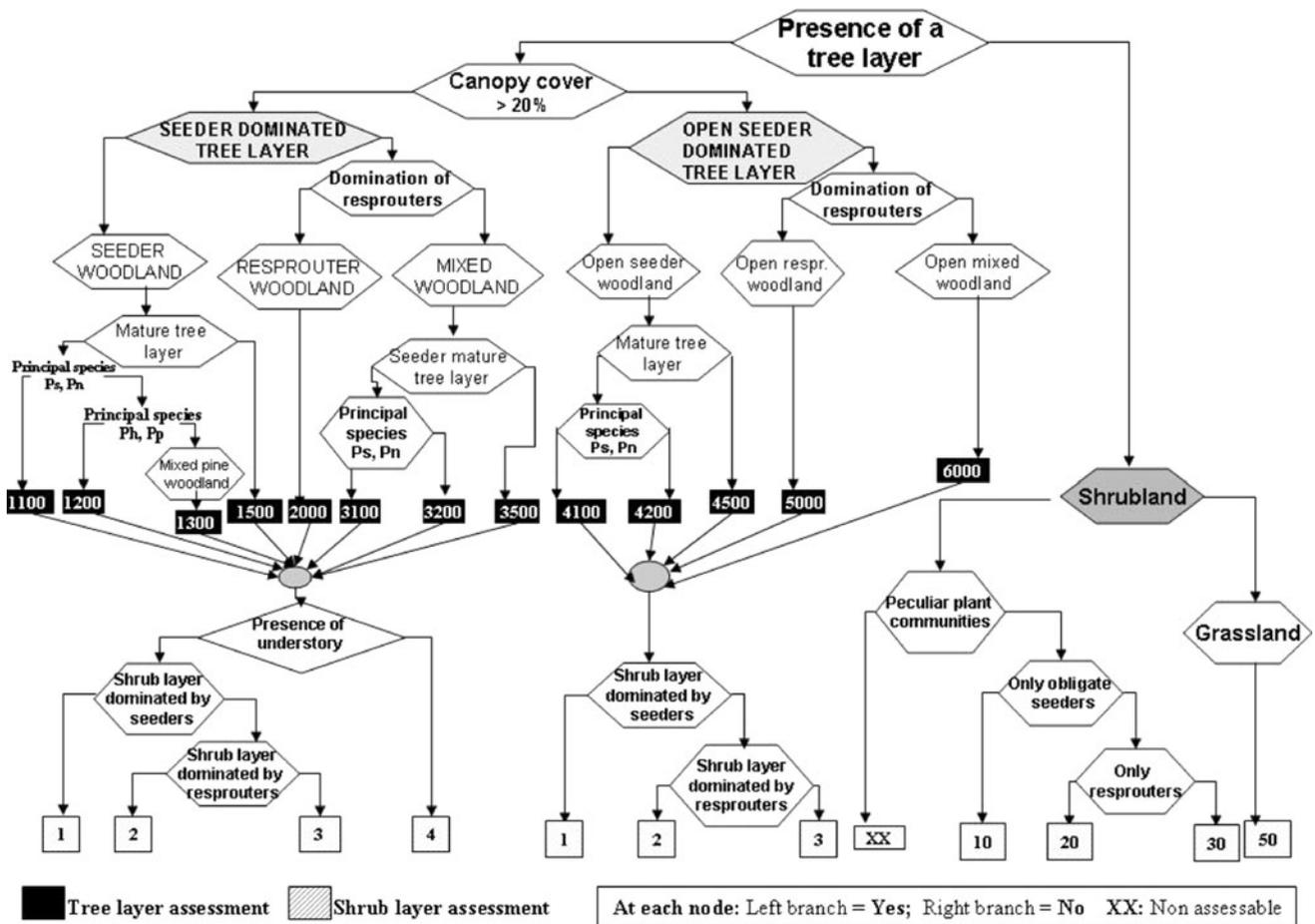
Within the CarboInvent project framework (Multi-source inventory methods for quantifying carbon stocks and stock changes in European forests), based on an existing soil profile database covering Spain (Rovira and others 2007), an average soil organic carbon content (1 m maximum soil depth) was attributed to any possible combination of three variables for which spatial information is available nationally (Fig. 1): climate type (Allue Andrade 1990), bedrock (type, mineral composition, hardness) and vegetation type (vertical structure, dominating regeneration strategy) (Fig. 2).

We derived OM<sub>5</sub> (in % weight/weight) at any location and, to simplify its use in the model, we grouped it into five qualitative classes, based on the OM<sub>5</sub> ranges found throughout the territory and experts' knowledge: <1%, Very low; 1–2%, Low; 2–4%, Medium; 4–8%, High; >8%, Very high.

#### Soil Permeability-Related Physical Characteristics

We used the soil cartography (scale 1:1.000.000) of the *Soil Geographical Database of Euro-Mediterranean Countries* (Montanarella 2001), which forms part of the European Soil Information System (<http://eusoils.jrc.ec.europa.eu/esbn/EUSIS.html>). The database defines a list of Soil Typological Units (STU) described by variables specifying the nature and properties of soils (texture, water regime, stoniness, etc.). On this coarse scale of representation, STUs were not delineated, but were grouped into Soil Mapping Units (SMUs) to form soil associations and to illustrate the functioning of pedological systems within landscapes. Most SMUs are larger than 25 km<sup>2</sup>. The database also includes a series of expert-based pedotransfer rules (PTRs) (Van Ranst and others 1995) to infer some additional soil properties from those available for practical purposes (preparing soil erosion risk maps, soil organic carbon estimates, etc.). Of the several hundred PTRs, we chose Top soil structure (STR\_TOP) and Physi-chemical factor of soil crusting & erodibility (PHYS\_CHIM), after verifying that the information required to estimate them was not redundant with other variables considered in the model. STR\_TOP is divided into four categories (Good, Normal, Poor and Humic or peaty topsoil), although only Good and Normal appear in Spain. PHYS\_CHIM is divided into five categories (Very favourable, Favourable, Medium, Unfavourable and Very unfavourable), of which Favourable, Medium and Unfavourable are present in Spain. Both variables were integrated (App. A-1, Electronic supplementary material) to qualitatively estimate a permeability-related soil structure variable, which was then combined with OM<sub>5</sub> to obtain a qualitative soil erodibility estimation (App. A-2, Electronic supplementary material).

- *Slope factor*: This variable was spatially quantified after the DEM. Traditionally, erosion models associate larger slopes with higher overland flow speeds, thus larger erodibilities. For slope-related assessments of susceptibility to erosion, we distinguished, the following slope intervals: <15 %, Low; 15–45 % Medium; >45 % High.
- *Vegetation development factor*: Although vegetation protective capacity is reduced by fire, charred or partially burnt plant remains may diminish precipitation kinetic energy, increase infiltration rates and limit the overland flow velocity (Pérez-Cabello and others 2002; De Luis and others 2003). We assumed a positive relationship between quantity of plant remains and biomass present in the pre-fire plant community. Hence, we considered dense vegetation (generally including a mature tree layer, but not always) as the most capable of withstanding the negative effects of



**Fig. 2** Scheme for the classification of plant communities into vegetation types. (Ps: *Pinus sylvestris*; Pn: *Pinus nigra*; Ph: *Pinus halepensis*; Pp: *Pinus pinaster*) The codes attributed to the vegetation types correspond to those given in Appendices B and C (Electronic supplementary material). For shrublands: 10: seeder-dominated; 20:

resprouter-dominated; 3: mixed *Vegetation types with a tree layer*: The first 3 digits of the code appear in the black boxes; the 4th digit, informing about the shrub layer (1: seeder-dominated; 2: resprouter-dominated; 3: mixed), appears in the last row

potential fire on infiltration processes and soil erodibility.

Based on the Spanish Forest Map information (MAPA 1990–1996), we estimated a *burnt vegetation protective capacity* factor (VPC) for any pre-fire plant community in a given area by integrating two parameters: vertical structure or *Height* (i.e., height of the highest vegetation layer) and *Density* (App. A-4, Electronic supplementary material). They derived from the “*Overload*” and “*Evolutionary level*” fields (Ruiz de la Torre 1990), respectively, the latter classifying vegetation into nine evolutionary (or maturity) levels. Level 0 corresponds to ecosystems whose vegetation cover is <5% and Levels 8 and 9 correspond to dense mature forests, dominated by mostly hardwood species.

(1) -Integration of factors for estimating soil erosive susceptibility

We first combined slope with soil erodibility (App. A-5, Electronic supplementary material). The resulting *Soil*

*erodibility-slope* factor was combined with the VPC factor (App. A-6, Electronic supplementary material). The obtained parameter was finally integrated with climate erosivity to estimate *Soil erosive susceptibility* (Table 1).

(2) Short-term vegetation recovery rate

The post-fire period to reach minimum vegetation cover that effectively protects soil from a high erosion risk is the factor differentiating ecosystems, under similar climatic and soil conditions, relating to the magnitude of post-fire soil erosion processes. Here we consider this minimum protective cover to be 30–40% (Francis and Thomes 1990), below which, degradation processes likely enter a positive feedback and become irreversible (Vallejo and others 2000).

In order to predict the speed by which minimum cover is reached, we contemplated the plant community’s intrinsic short-term recovery capacity and the environmental conditions influencing it. To estimate the former variable, we used the plant functional types approach (Gitay and Noble

**Table 1** Soil erosive susceptibility

	Climate erosivity			
		High	Medium	Low
Soil erodibility-Slope/ VPC factor	High	High	High	Medium
	Medium	High	Medium	Medium
	Low	Medium	Medium	Low

1997). For fire, we considered a set of functional traits to determine species' response to fire, particularly, the so-called regeneration traits; e.g., post-fire regeneration mode, seed storage mode or dispersal mode (Pausas and others 2004). After examining the information available on regeneration traits for a wide range of species in Mediterranean-type ecosystems (Paula and others 2009), we chose the post-fire regeneration strategy of species as the predictive attribute to estimate the intrinsic recovery capacity. Despite the simplification of using this regeneration trait as the only predictive attribute, previous analyses of post-fire vegetation response on a regional scale based on this vital attribute provided good results (Alloza 2003; Arianoutsou and others 2011).

#### a) Vegetation characteristics

We assumed that (1) the plant species able to resprout after fire will establish faster after a fire than obligate seeder species; i.e., species relying on only the re-cruitment of seedlings for post-fire recovery (Pausas and Vallejo 1999); (2) the resprouting capacity is maintained, regardless of soil water availability (although resprouting rates might be affected by dry post-fire conditions; Pausas and others 1999). Conversely, seed germination and seedling establishment will be strongly dependent on soil moisture and, hence, on post-fire rainfall patterns (Moreno and Oechel 1992).

We classified the major plant communities in the Iberian Peninsula into vegetation types (Fig. 2) according to their physiognomy (forest/shrubland), tree canopy cover (if present), tree layer maturity and the post-fire reproductive strategy (resprouters/seeder) dominating each vegetation layer. The spatial information about plant communities was provided by the digital Spanish Forest Map and by more updated regional cartographies, as in Aragón (Escudero and Franchés 2004).

Each vegetation type was identified by a numeric code and then attributed a *short-term vulnerability qualitative value* (App. B, Electronic supplementary material) among the five values (Very Low, Low, Medium, High, Very High), estimated as the inverse of its short-term post-fire recovery rate. The latter was estimated according to the post-fire reproductive strategy dominating the vegetation layer conferring its physiognomy to the whole plant

community (tree layer in forests; shrub layer in shrublands). Based on the literature and our field observations, we distinguished between seeder tree species with good or low post-fire response capacity. For the latter, we considered species like *P. nigra* and *P. sylvestris*, which produce few seedlings after fire, or *P. pinea*, with a high seedling mortality in the first post-fire years (Rodrigo and others 2004), and all the species showing no particular fire-survival strategy (e.g., serotinous cones, early flowering) (Tapias and others 2001).

#### b) Environmental conditions

Climatic and terrain parameters influence fire behaviour and post-fire vegetation responses (Pausas and Vallejo 1999). Precipitation and aspect strongly determine ground water balance and, thus, post-fire vegetation dynamics (Díaz-Delgado and others 2002; Röder and others 2008). Aspect determines site-level short wave radiation budgets, which control the evapotranspiration of vegetation. Locations receiving considerable irradiance are less favourable for plant development since more water is lost through transpiration effects (Röder and others 2008).

We also mention the dependence of obligate seeders on post-fire precipitations when evaluating vulnerability according to reproductive strategy. However, as the real amount of rain to fall after a potential fire is unknown, we relied on water deficit indicators based on historical records for evaluation purposes. We selected the dry period length, estimated by the Bagnouls-Gaussens Index (Bagnouls and Gaussens 1953), employed as a proxy for ecosystem vulnerability to climatic stimuli, and we crossed it with aspect to estimate the water limitations factor (Table 2). This was then combined with intrinsic recovery capacity (short-term vulnerability value) to obtain the inverse of the *Short-term vegetation recovery rate* (the higher this value, the lower the vulnerability; Table 3), which we next combined with *Soil erosive susceptibility* to estimate *Short-term ecological vulnerability* (Table 4).

#### Final Short-Term Ecological Vulnerability

As a standard criterion, the worst possible burning conditions (summer fires) were presumed to implement the model in its stages. However, it is known that fire behaviour parameters

**Table 2** Limitation to plant regeneration due to water availability (water limitations)

Aspect	Dry period length (months)		
	≥3	2	<2
South and Plains	High	High	Medium
Southwest, Southeast	High	Medium	Low
North, Northwest, Northeast	Medium	Medium	Low

**Table 3** Integration of short-term vegetation vulnerability (App. B, Electronic supplementary material) with site-specific water limitations to obtain the inverse of the Short-term vegetation recovery rate

Short-term vegetation vulnerability	Water limitations		
	High	Medium	Low
Very high	Very high	Very high	Very high
High	Very high	High	High
Medium	High	Medium	Medium
Low	Medium	Low	Low

**Table 4** Short-term ecological vulnerability

	Inverse of the short-term vegetation recovery rate				
	Very high	High	Medium	Low	
Soil erosive susceptibility	High	Very high	Very high	High	Medium
	Medium	High	High	Medium	Medium
	Low	Medium	Medium	Low	Low

(spread rate, fireline intensity, etc.), thus fire effects, may be very heterogeneous spatially, and strongly determined by fuel characteristics. In this final integration phase, the evaluation of short-term vulnerability is modulated according to potential fire behaviour and, more specifically, to expected fireline intensity. This parameter, which describes the energy release per unit length of flame front, is described as the best fire behaviour descriptor to establish correlations with above-ground fire effects (Andrews and Rothermel 1982). Moreover, positive relationships have been identified between fire intensity and magnitude of post-fire erosive activity (Debanco and others 1998; De Luis and others 2003; Scott and others 2009).

Following the photographic identification key of the Spanish Forest Administration (ICONA 1990), vegetation types were reclassified into 13 standard fire behaviour fuel models (Anderson 1982). We used the flame lengths documented for these fuel models (Anderson 1982) to attribute them predicted fire intensities using an algorithm derived from Byram’s Intensity Equation (Ryan 1981) by incorporating flame length.

For various plant communities in E Spain, we compared the intensities estimated with the data obtained from experimental fires (Baeza and others 2002) or fires simulated with the FARSITE model (Finney 1998), which we parameterised for the Mediterranean conditions in Ayora using Anderson’s 13 fuel models (Duguay and others 2007a). We finally aggregated those models in relation to the expected fire intensities: 1, 8, 9 = low intensity; 2, 5, 6, 7, 10, 11 = medium intensity; 3, 4, 12, 13 = high

**Table 5** Final short-term ecological vulnerability

Fire intensity	Short-term ecological vulnerability			
	Very high	High	Medium	Low
High	Very high	Very high	High	Medium
Medium	Very high	High	Medium	Low
Low	Very high	High	Medium	Low

**Table 6** Synthetic Ecological Vulnerability to Forest Fires

	Medium-term ecological vulnerability				
	Very high	High	Medium	Low	
Final short-term ecological vulnerability	Very high	Very high	Very high	High	High
	High	Very high	High	High	Medium
	Medium	High	High	Medium	Medium
	Low	High	Medium	Medium	Low

intensity. Table 5 offers the *Final short-term ecological vulnerability* assessment.

Mid-Term Ecological Vulnerability Analysis

Mid-term ecological vulnerability was estimated by vegetation’s ability to persist 20/25 post-fire years with no substantial changes in species composition or structure.

By taking a low historical fire frequency, we estimated the mid-term vulnerability qualitative value for each vegetation type with an expert knowledge-based process (App. C, Electronic supplementary material). The key parameters for experts’ estimations were the post-fire regeneration strategy dominating each vegetation layer and the age (mature or not) of tree species. We distinguished between seeder tree species with a good or poor post-fire response capacity following the same rules applied for short-term vulnerability (App. B, Electronic supplementary material). The *Synthetic index of ecological vulnerability* was finally obtained by integrating the *Final short-* and *Mid-term* ecological vulnerabilities (Table 6).

Cartographic Outputs

We generated our final cartographic outputs (i.e., short- and mid-term and synthetic vulnerability maps) at a 100 × 100 m spatial resolution. Most original thematic maps were available as a vectorial format on a 1:50,000 scale (climatic type, bedrock, vegetation) and we rasterised them in ArcGIS for overlaying purposes.

## Validation Approach

Validation is often a complex issue in ecological models (Rykiel 1996). With the model herein presented, a classical approach for its validation involves comparing the model-predicted values for the vulnerability synthetic index with field observations of mid-term post-fire ecosystem recovery in burned sites with known pre-fire vegetation. This analysis is done over several years for numerous cross-country summer fires to cover the largest range of environmental conditions and vegetation type combinations. However, the immense human and economical efforts required for this large-scale, long-lasting sampling design make such validation approaches unrealistic. Moreover for many vegetation types in Spain, field-based studies documenting mid-term post-fire dynamics are lacking, which would be valuable to complement any partial validation results.

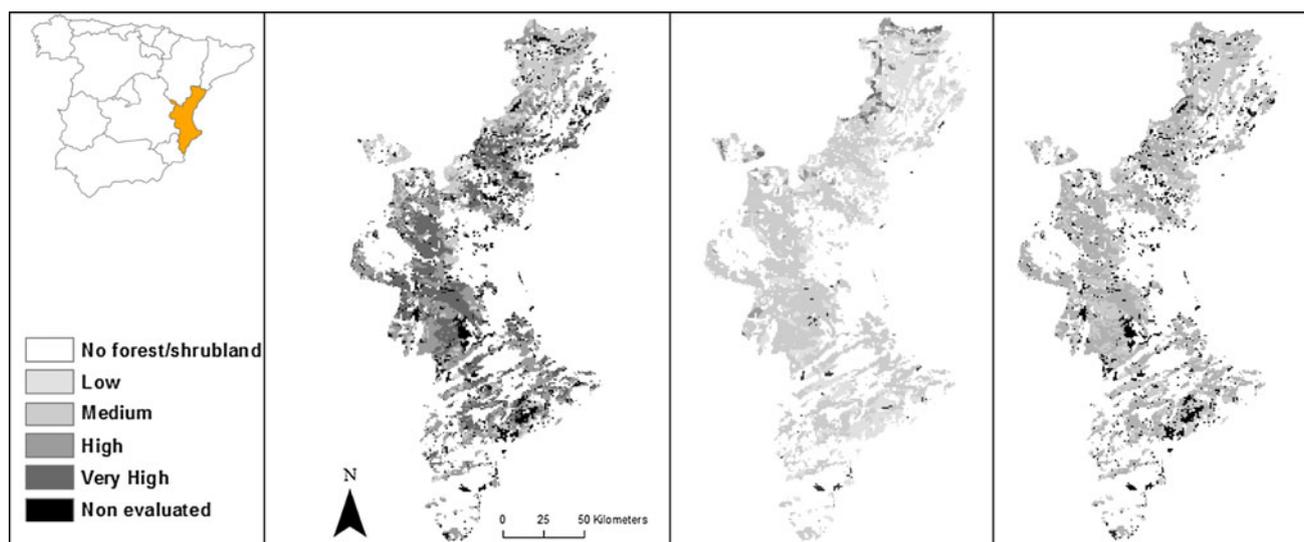
Given those limitations, we will hereafter refer to the implemented partial validation exercise as an evaluation. It focused on testing our model's adequacy to support regional-level fire management planning in Mediterranean fire-prone landscapes. It was carried out with three vegetation types (open, dense pine forest and two shrub type formations), all of which were among the most frequent and representative communities found in fire-prone landscapes of E Spain. Nonetheless, testing the adequacy of the model's predictions for these ecosystems does not allow generalising its applicability to other ecosystems (Goodall 1972).

This evaluation protocol combined two complementary approaches applied at the Ayora study site (Valencia, E Spain), defined as 39° 20' 15.13"N/1° 10' 33"W (ULX/ULY) and 38° 49' 53.19"N/0° 32' 3.53"W (LRX/LRY).

In July 1979, the area was affected by a very large fire (31,700 ha), with important socio-economic and environmental repercussions. The site was extensively documented in previous projects by an extensive digital database (Duguy and others 2007a; Röder and others 2008).

The first evaluation approach tested the vulnerability analysis results based on the most updated Spanish Forest Map at the time, published in 1993 for Ayora, and on 1989 field data (Fig. 3). The synthetic vulnerability predicted for a hypothetical fire of 1989 corresponds to the situation reached by around 2014. We compared these predictions in a set of sites with variation in the Normalised Difference Vegetation Index (NDVI) between the pre-fire situation (1991) and 2014. The NDVI values derived from the regressions of a previous study to compare the suitability of four vegetation indices to monitor 9 years of succession in several shrub communities in Ayora after a fire in 1991 (Duguy and others in preparation). It burned 5,000 ha, which were previously affected by the 1979 fire (Baeza and others 2007). The indices were NDVI,  $Q_{NDVI}$  (Díaz-Delgado and others 2002), NRI (Riaño and others 2002) and SAVI (Huete 1988). At most sites, for a given regression model, the best fit (the largest R-Pearson value) was observed for NDVI. The high spatial homogeneity of substrates found in that landscape, and the rather large post-fire vegetation covers reached quite quickly for most plant communities, probably explain NDVI's suitability for monitoring post-fire dynamics in these ecosystems.

Several studies suggest that methods such as spectral mixture analysis (SMA) could be more performant than vegetation indices to monitor post-fire vegetation dynamics (tree cover, in particular) in arid or semi-arid landscapes (Yang and others 2012). A comparison made recently in



**Fig. 3** Maps of Short-term, Medium-term and Synthetic Ecological Vulnerability to fire (from left to right) in the Valencia Region (indicated within Spain in the upper left localization map), based on the Spanish Forest Map (MAPA, 1990–1996)

Crete (Greece) between several Landsat-based vegetation trend analyses, based on four different vegetation measures (SMA, NDVI, SAVI and Tasseled Cap Greenness), revealed, however, that all the methods resulted in highly comparable trends (Sonnenschein and others 2011). Indeed, in spite of being a proxy for green biomass only and having its limitations, NDVI remains the most widely used vegetation index for deriving vegetation cover estimates (Sonnenschein and others 2011).

Our sites (selected on the Spanish Forest Map and checked in the field) were covered by either seeder-, or resprouter-dominated shrublands (S- and R-sites, respectively). *Ulex parviflorus* and *Rosmarinus officinalis* dominated in the former, with *Quercus coccifera* in the latter. Fire affected no site between the fires of 1979 and 1991. Therefore, their 1989 and 1991 pre-fire vegetations were similar.

More vulnerable plant communities likely undergo major structural alterations, thus their green biomass changes. In the studied shrublands, very high vulnerability to fire would correspond to a regressive post-fire dynamics with reduced green biomass and, therefore, a lower NDVI. Therefore if our model is valid, stable (or even increasing) NDVIs would be observed between the pre-fire situation and 2014 at the sites predicted as the least vulnerable, whereas the greatest NDVI decreases would occur at the sites where vegetation is predicted to be very negatively affected by fire (i.e., when most pixels at the site are predicted High or Very High vulnerability).

Major changes in species composition may occur with high fire frequencies (Zedler and others 1983) which may not always be monitored with NDVI, and we do not expect these shifts in the studied plant communities under the prevailing fire regime (low fire frequency).

In ArcGIS, we extracted the areas corresponding to the S- and R-sites from the synthetic ecological vulnerability map produced by the model, and we calculated the proportion (i.e., percentage of pixels) of the different vulnerability classes at each site.

In the second evaluation approach, we ran the vulnerability model using an available 1979 Ayora vegetation map (pre-fire situation) generated in a previous remote sensing-based study (Röder and others 2008). The predicted synthetic vulnerabilities corresponded to the situation at around 2004. A set of geometrically and radiometrically corrected Landsat images (Röder and others 2008) enabled us to monitor the post-fire NDVI evolution until 2000, 21 years after the fire, in open or dense areas covered by pine forests (*P. halepensis* and *P. pinaster*) before 1979. We were able to compare the model's predictions with the NDVI variation values obtained from the images.

## Results and Discussion

### Cartographic Outputs

We implemented the model in two Mediterranean Spanish regions: Aragón (inland NE Spain) and Valencia (E Spain) covering wide-ranging vegetation types and climatic conditions, which are extensively documented with spatially-explicit information. We present only the maps of Valencia (Fig. 3).

The short-term vulnerability assessment provides a good indicator of the urgency to carry out post-fire restoration treatments in recently burnt areas. Our maps show that these treatments should be implemented in at least 65 and 74% of the areas evaluated in Aragón and Valencia, respectively (e.g., High or Very High short-term vulnerability areas). The mid-term assessment reveals the risk of permanent degradation, or even the disappearance, of the affected plant communities. This risk is significant in 20 and 5% of the evaluated Aragón and Valencia territories, respectively (High or Very High mid-term vulnerability areas). For both regions, the synthetic map showed that around half the evaluated lands (51 and 52%, respectively), there are ecosystems with High or Very High ecological vulnerability to fire. Large parts of both regions are very, or extremely, susceptible to the adverse effects of a potential fire. It is likely that many of these landscapes are not able to cope with the more severe fire regimes predicted for the Mediterranean regions in the future scenarios of climate and socio-economic changes (IPCC 2007). These results highlight the urgent need to further investigate and implement management and restoration alternatives to reduce fire risks and the vulnerability of ecosystems and landscapes to fire with change and uncertainty (Duguay and others 2012).

### Evaluation

The first evaluation approach always showed an increase in the mean NDVI value between 1991 (pre-fire situation) and 2014 at the R-sites, whereas NDVI did not reach its pre-fire value yet (more than post-fire 20 years) at no S-site (Table 7). These results indicate that the latter communities exhibited much slower post-fire recovery and were vulnerable to the 1991 fire.

They could also reflect a permanent change of the S-communities to an earlier successional stage due to recurrence of fires (1979 and 1991). This hypothesis, however, seems rather unlikely as most seeder shrubs in these communities are fire-recruited species (*Ulex parviflorus*, *Cistus* spp.), generally reaching sexual maturity in post-fire year 3 (Carreira and others 1992; Roy and Sonié 1992). Only high fire recurrences with very short time

**Table 7** Regression-based estimations of mean NDVI variation and model predictions of synthetic ecological vulnerability (estimated with percentage of pixels occupied in each test site)

	Estimated mean NDVI (2014)	NDVI variation (1991–2014)	Medium synthetic vulnerability (%)	High synthetic vulnerability, %Hv (%)
<b>R-sites</b>				
R1	0.49 (0.77)	8.9	78.3	21.7
R2	0.51 (0.75)	10.6	100	0
R3	0.43 (0.55)	2.2	20	80
<b>S-sites</b>				
S1	0.42 (0.86)	−2.1	53.3	46.7
S2	0.36 (0.57)	−2.2	68.8	31.2
S3	0.41 (0.76)	−7.6	0	100
S4	0.38 (0.77)	−7	0	100

R resprouters, S seeders; Values in parentheses indicate the  $R^2$  of the corresponding regression of NDVI with time after fire

intervals (<3 years) between fires would significantly deplete their persistent soil seed banks (Roy and Sonié 1992). The model did not predict any Very High vulnerability pixel (i.e., good likelihood of a shift towards a different community) at the S-sites or the R-sites.

Rainfall during the first autumn after a fire is particularly crucial for the germination of most seeders (Moreno and Oechel 1992). Records from the nearby Salto de Millares weather station (<http://www.ceam.es/ceamet/>) show that autumn and spring rainfalls after the 1991 summer fire were well below historical averages in the area: 76 and 31%, respectively. These dry conditions likely delayed post-fire regeneration in the S-communities.

Moreover before the 1991 fire, all sites were covered by dense mature shrublands, classified as fuel model 4 (Anderson 1982); i.e., highly flammable vegetation where high (therefore severe) fire intensities are expected. Fire severity has been correlated with post-fire seedling recruitment, but never with post-fire resprouting (Keeley 2009). Some studies suggest there might be a negative relation between fire intensity and post-fire germination for *Ulex parviflorus*, *Cistus albidus* and *Rosmarinus officinalis*; i.e., intense fires can destroy shallow-buried seeds (De Luis and others 2005). Cistaceae and Leguminosae seeds, can generally withstand severe fires thanks to their hard coats (Arianoutsou 1998). However, *R. officinalis* (a dominating species at all the S-sites) may have been more affected by intense fire.

Finally, fire severities are often directly related to larger post-fire soil losses and, thus, to soil degradation processes, which may significantly affect vegetation recovery in the short to long term (De Luis and others 2003; Scott and others 2009). This may also apply to our S-sites too as they

lie mostly over old croplands with marly soils, which are highly sensitive to erosion (Vallejo and Alloza 1998; Vallejo and others 2000).

Considering the proportion of sampled sites occupied by the highest synthetic vulnerability (%Hv hereafter), i.e., “High” vulnerability pixels since “Very High” vulnerability was never predicted, quite a consistent inverse relationship was found between that variable and mean NDVI variation (between the pre-fire situation and 2014). The slighter increases (or greater decreases) in NDVI are observed at the sites with a larger %Hv (Table 7). This relationship improves when separately considering each community type (R- or S-sites), and is totally consistent for the three R-sites, as confirmed by the correlation analyses. When considering all the communities, significant correlations ( $r = -0.85$ ,  $P < 0.05$ ) were observed between %Hv and NDVI variation. When considering only R- or S-sites, we found even more significant correlations between these two variables ( $r = -1$ ,  $P < 0.05$  and  $r = -0.98$ ,  $P < 0.05$ , respectively).

It is noteworthy that even the largest %Hv (80%) was associated with an increase (2%) in NDVI among the R-shrublands (Table 7), suggesting a window totally occupied by High vulnerability pixels showing a stable (or slightly smaller) NDVI for 2014 vs. 1991 (i.e., vegetation would have reached, or almost, the pre-fire green biomass). Moreover, sites such as R<sub>1</sub> and S<sub>2</sub>, with a similar %Hv (22 and 31%, respectively), but different dominant post-fire strategies, displayed very different NDVI post-fire evolutions (8.9 and −2.2%, respectively).

The post-fire vegetation dynamics appears to substantially differ between the studied S- and R-communities as the latter are highly fire-resilient with a much faster vegetation recovery rate.

Our results suggest that the various qualitative synthetic vulnerability categories used in the model require a different interpretation in terms of post-fire vegetation dynamics when applied to those two ecosystems types (more generally to different ecosystems). Thus, predicted High vulnerability in S-shrublands would correspond to reduced green biomass (7–8%, as indicated by sites S<sub>3</sub> and S<sub>4</sub>) in the mid term after a fire, whereas among R-shrublands, it reflects a slower post-fire dynamics than in a Medium- or Low-vulnerability area, but is still characterised by total pre-fire green biomass recovery.

Indeed, our model overvalues the synthetic vulnerability of R-communities and probably attributes them excessively high vulnerabilities. In ecosystems characterised by highly resilient plant communities (most R-dominated Mediterranean shrublands), site-level abiotic limitations are often overcome by vegetation’s good capacity of quickly recolonising the open space created by fire with its undamaged below-ground organs. Modifying the integration table

(Table 6) to attribute lower synthetic vulnerabilities to certain combinations of short- and mid-term vulnerabilities does not seem suitable as the model loses its capacity of identifying the most critical situations (from an abiotic- or a vegetation-related point of view) in its final output. Indeed, under more adverse situations (e.g., more fire recurrences affecting shrubs' resprouting capacity), limiting abiotic conditions at the site level would probably slow down the post-fire dynamics in some R-communities.

Indeed, the slowest post-fire recovery among the R-sites was observed in R<sub>3</sub> (Table 7), characterised by a steeper slope: 44.2% of its area reached a slope >15% (and >20% for 21.4% of its area), whereas this slope was only 6% and 0% of the area in R<sub>1</sub> and R<sub>2</sub>, respectively. That feature of R<sub>3</sub> is probably associated with shallower soil (i.e., a smaller soil water holding capacity), which might negatively affect vegetation recovery (Di Castri 1973; Thorne 1990), particularly under the post 1991 fire dry conditions. A steeper slope would favour greater post-fire soil losses.

The second evaluation approach resulted in the predicted synthetic vulnerability map for 2004 (25 years after the 1979 fire). We selected all the areas covered by dense or open pine forests before the 1979 fire which were affected by it. In both cases, the model predicted only the "Medium" and "High" synthetic vulnerabilities for 2004.

Across all the dense forest areas predicted as "Medium" vulnerability, NDVI decreased (26%) between the 1979 and 2000 images. When considering those predicted as "High" vulnerability, this decrease enlarged (31.2%). Regarding open forests, NDVI also lowered, but not so significantly, for both the predicted "Medium" and "High" synthetic vulnerability areas (22.5% and 27.0%, respectively).

Model predictions were again consistent for a given vegetation type (pine forest) when comparing the NDVI post-fire evolutions with the predicted synthetic vulnerabilities. The greater reductions in green biomass (i.e., the least successful post-fire recoveries) always corresponded to areas with higher predicted vulnerability to fire.

The NDVI decreases (associated with either "Medium" or "High" synthetic vulnerability) were considerably larger than those previously noted for the S-communities (Table 7). Forest-type ecosystems are generally characterised by larger aboveground biomasses (green biomasses in particular) and by larger biomass losses due to intense fires. The post-fire time spans required to recover the original biomass are expected to be longer.

The NDVI decreases observed for dense forests indicate that their canopies were strongly affected by the 1979 fire (i.e., many pines were killed) and had not recovered their original state even after 25 years. These ecosystems, dominated by two seeder trees, were much more vulnerable to fire than the two previously studied shrubland types.

The 1979 fire was a severe crown fire causing profound long-term changes in vegetation throughout the area (Baeza and others 2007). Hundreds of hectares of dense mature pine forests reverted to shrublands, sometimes with open tree layers (Tapias and others 2001; Duguay and others 2007a). Small isolated pine stands not affected by fire remained. As previously mentioned, high fire severities often cause soil degradation processes, particularly when fire affects substrates that are very sensitive to erosion. The pine forests in Ayora were located on former cultivated lands, very often on Mediterranean marly soils (Vallejo and Alloza 1998). Post-fire soil degradation processes (mainly causing soil nutrient losses) might have affected vegetation recovery there. As resprouting species were very scarce (uprooted for centuries for agricultural use), post-fire woody regeneration was primarily produced from obligate seeders, resulting in the spread of highly flammable *Ulex parviflorus*-dominated shrublands (Baeza and others 2002, 2007).

After very large fires, site-level post-fire succession may prove most unpredictable as it is greatly influenced by the large spatial heterogeneity, generally created by fire, and particularly by the heterogeneous pattern of surviving organisms and propagules (i.e., residuals *sensu* Turner and others 1998). The 1979 fire killed trees over vast areas. Consequently when new trees were established, their seed cone formation may have been vastly delayed by lack of pollen (Tapias and others 2001).

Post-fire adverse weather conditions may have also contributed to such low recovery. The autumn of 1979 was extremely dry. The precipitation recorded from September to November at the Salto de Millares weather station was 83.7 mm, 61% less than the average value for the same months in the 1979–2000 period. Pausas and others (1999) observed that the post-fire recovery of *P. halepensis* mature woodlands in that region may have been negatively affected by very dry post-fire conditions.

## Conclusions

The modelling ArcGIS-based approach presented herein represents an innovative methodological framework to support the assessment of ecological vulnerability to fire in Mediterranean ecosystems. This integrated analysis intends to assess the response of main ecosystem components (soil, vegetation) in fire terms by considering the various time scales which, in each case, fire-caused impacts may affect them more critically, and could alter the whole ecosystem's generic capacity to cope with fire over time and space.

Implementation of this spatially-explicit model in two Spanish regions helps identify the most erodible areas, possible in greater need of short-term post-fire mitigation

actions, and those where permanent mid-term post-fire changes in vegetation composition and structure are expected. The maps of the synthetic index of ecological vulnerability to fire indicate the most problematic situations in relation to the risk of permanent degradation of ecosystems and, therefore, their decreasing resilience to fire.

Our model's cartographic outputs may prove powerful supportive tools to adapt current fire management strategies in Mediterranean fire-prone landscapes, and contribute to develop fire prevention and restoration alternatives to cope with projected future fire regimes driven by climate and socio-economic changes. Combined with fire hazard conditions models, vulnerability models like that presented herein could implement more integrated fire risk assessment systems to also consider the potential damage by fires to ecosystems and landscapes.

The evaluation approaches results reveal that the model's predictions are consistent for the studied ecosystems, which extend across fire-prone landscapes in E Spain. Major synthetic vulnerability situations (low, medium, high) can be identified and the predictive performance of the model appears satisfactory in relation to the objective of validating a tool to support regional-level fire management (prevention and restoration) planning.

Further model evaluations with other vegetation types should be implemented. In particular, the interpretation of the NDVI-based approaches, like those presented herein, can be improved with further studies (including field work) to establish a suitable correspondence between the mid-term NDVI post-fire variation (trend and magnitude of change) and the various synthetic vulnerability categories predicted by the model for a wide range of vegetation types and environmental conditions. Other remote sensing-based methodologies (combining several vegetation measures) should be explored.

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