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A method for regional-scale assessment of vegetation recovery time after high-severity wildfires: Case study of Spain

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Abstract

This study aims to develop a method to estimate the recovery time of plant communities after high-severity wildfires. The designed methodology is based on map algebra and a geographical information system, which enabled calculation of the approximate time required to restore vegetation to conditions similar to pre-fire regarding plant height and canopy cover. The methodology considered, first, the vegetation in the territory, characterized by the structure of the dominant plant community (tree, shrub, or grassland) and its regeneration strategy (resprouter or seeder); and, second, two of the main factors determining recovery time – water availability and soil loss. We also considered the influence of observed rainfall trends over the past 50 years on these latter two factors. The methodology was applied to Spain to test its performance. The results suggest a period of 2 and approximately 100 years for grassland communities and tree communities with low germination, respectively. There are significant differences in plant communities between the two biogeographic regions (Euro-Siberian and Mediterranean) as well as within each community, directly linked to variability in terrain and climatic conditions.

Keywords

GIS, plant communities, recovery time, wildfire

I Introduction

Forest fires have traditionally been linked to the Mediterranean climate due to the coexistence, in some months of the year, of high

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temperatures and low rainfall (Camia and Amatulli, 2009). The indigenous vegetation has lived with fire for millennia, and thus it is not an extraneous factor to the Mediterranean environment or, more specifically, to peninsular Spain (Pausas and Vallejo, 1999; Pyne, 2009; Wagtendonk, 2009). However, recent changes in socio-economic models and climatic patterns have significantly affected the historical fire regime in southern Europe (González et al., 2010; San-Miguel-Ayanz et al., 2012a), with potential damage far greater than traditionally experienced (Bodí et al., 2012; Bowman and Boggs, 2006; Meyn et al., 2007; Pausas and Vallejo, 1999). In Spain, the total area burned (with an annual average of over 125,000 hectares from 2000 to 2008, but almost 250,000 hectares per year from 1980 to 1989; Schmuck et al., 2009) has decreased in recent years, while the number of fires has increased (18,150 compared to 15,300 in the corresponding periods; San-Miguel-Ayanz et al., 2012a; Schmuck et al., 2009). Furthermore, the chances of suffering an especially dramatic fire season, as in several countries in the last decade as a result of extreme heatwaves (Rebetez et al., 2006) (Spain, 2000 and 2005; Portugal, 2003 and 2005; Greece, 2007; Australia, 2009; Russia, 2010), appear to have increased (Allen et al., 2010; Camia and Amatulli, 2009; San-Miguel-Ayanz et al., 2012b; van Mantgem et al., 2009) and are likely to occur more frequently in the coming decades (Seidl et al., 2011). The main trends of degradation induced by fire in the medium and long term may include permanent changes in the floristic composition of the plant community, reduction of vegetation cover, biomass loss, and alteration of landscape patterns. Forest fires can also induce long-term changes in floristic and physiognomic parameters of vegetation through their impact on the physical and chemical properties and nutrient availability of soil (MMA, 2006; Vallejo et al., 2009). After the burning of vegetation, the contribution of ash to the soil temporarily increases the availability of some nutrients (P, Mg, K, Ca, Na). This initial fertilization depends on the severity of the fire and the amount of biomass (fuel) prior to the fire. However, other nutrients such as nitrogen may volatilize or be washed away as a result of wind or water erosion post-fire (Neary et al., 2009; Shakesby and Doerr, 2006). In addition, the loss of vegetation cover after fire increases surface erosion because the bare soil is exposed to raindrop impact and surface runoff, especially in the first months after burning (Giovannini et al., 2001; Inbar et al., 1998).

Therefore, it is necessary both to improve our early warning systems and to encourage the assessment of potential environmental damage (Chuvieco et al., 2010, 2012), as such natural and semi-natural ecosystems provide many important functions (or 'services') of economic, cultural and aesthetic value to human societies (Costanza et al., 1997). In this sense, assessment of vegetation response after fire can support governments' forestry policies, forest services' activities, and fire-risk modeling. This point is acute because the lack of spatial data on this subject has hindered natural resources management agencies from identifying priority areas for adaptation measures (Brooks et al., 2006; Hannah et al., 2002). This hindrance is more common in Mediterranean ecosystems where fire is the main natural disturbance and exerts a decisive influence on the structure and dynamics of plant and animal communities (Arianoutsou et al., 2011; Bajocco et al., 2011; Cerdà and Doerr, 2010; di Castri and Mooney, 1973; Gill et al., 1981; Naveh, 1975; Trabaud and Lepart, 1980).

This study focuses on the development of a method to assess the time required for vegetation to reach a state approximating pre-fire with similar levels of vegetation cover, restoring its physiognomic properties on regional scales, assuming that the dominant community species remain the same after fire (Broncano et al., 2005). Here vegetation means more specifically the dominant plant community. Other methodologies for assessing vegetation response to forest fires in Mediterranean-type ecosystems have already been designed. Bisson et al. (2008) presented an index of plant community resilience to fire. Arianoutsou et al. (2011) evaluated the post-fire resilience of *Pinus halepensis* in Cape Sounion National Park, Greece, using GIS and multi-criteria analysis. De la Riva et al. (2008), Alloza et al. (2006), and Duguy et al. (2012) produced a qualitative index of ecological vulnerability to forest fire in Mediterranean environments. In any case, these methods provide qualitative results; however, while they may be useful in some areas for territorial management, they are inadequate for other kinds of analyses such as quantitative assessment of fire-induced economic losses due to interruption of environmental services (e.g. timber, hunting, and mushroom gathering). For these, it is essential to know the period during which that service was lost (Román et al., 2013).

To overcome this limitation, our methodology follows a different approach, estimating the post-fire recovery time of vegetation by integrating some of the major factors and processes influencing vegetation development after fire: the pre-fire structure of the dominant plant community (grassland, shrubland, or trees), the post-fire regeneration strategy of the dominant plant community (resprouter or seeder) (Baeza and Roy, 2008), water availability for vegetation development (from rainfall), and soil loss as a consequence of loss of canopy cover. The first two (vegetation structure and regeneration strategy) are intrinsic characteristics of the plant species, and are used to define the post-fire response capacity of plants (Alloza et al., 2006; de la Riva et al., 2008). The dichotomy of resprouters versus seeders is an important factor when analyzing the consequences of fire for vegetation (Pausas et al., 2008). The latter two (water availability and soil loss) are parameters that mainly depend on the characteristics and temporal evolution of the climatic conditions (Certini, 2005), influencing plants by modifying the amount of available nutrients and water or soil chemical composition (Shakesby and Doerr, 2006). Climatic conditions and soil loss are considered key parameters when modeling relationships between wildfire and vegetation (Daly et al., 2000; Lenihan et al., 2008). To include the influence of possible changes in climatic conditions, the temporal evolution of seasonal rainfall is considered. Our method focuses on obtaining a quantitative result that is easily transformable into a qualitative one. However, it is not intended to provide a definitive recovery time, because our main goal is to develop a methodological approach for its assessment. Vegetation recovery time may vary to a greater or lesser degree depending on local characteristics, such as the type and characteristics of vegetation, climatic conditions, or terrain (Baeza et al., 2007; Keeley, 2009). Consequently, we intend to supply an indicative result, though a more accurate one than provided by qualitative analysis. A method for validation of the results, based on monitoring the post-fire evolution of NDVI in fire-affected plant communities, is also proposed and discussed.

II Materials and methods

The study area covered the whole of peninsular Spain, thus excluding the Balearic and Canary Islands as well as the autonomous cities of Ceuta and Melilla. The study region was further restricted to forested areas, meaning that urban, agricultural, and inland water zones were also excluded from the assessment. No data are reported for these areas or shown on the maps.

The methodology for estimating the post-fire vegetation recovery time (RT) is based on calculating the regeneration time of plant communities. An initial RT (recovery time under optimum conditions, RTOC) is assigned according to the dominant plant communities' structure (grassland, shrubland, or trees) and regeneration strategy (resprouter or seeder).

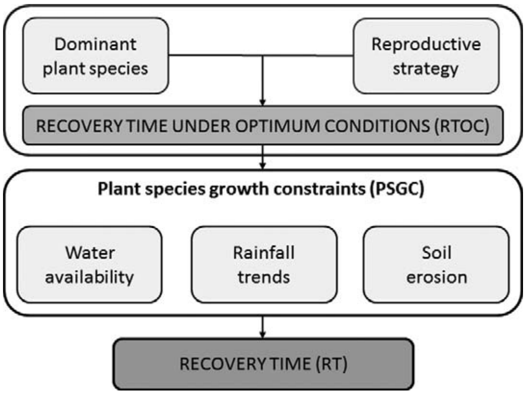

Figure 1. Methodology for RT calculation.

The increase in time is then calculated by introducing the influence of plant species growth constraints (PSGC): water availability from annual rainfall, soil erosion due to loss of protective vegetation cover, and seasonal rainfall trends, which influence both water availability and soil loss mainly after the fire. The influence of water availability and soil erosion is introduced as a weight factor of RTOC. In turn, seasonal rainfall trends, specifically winter and summer trends, are introduced by weighting water availability and soil loss. RTOC is assigned based on experts' criteria supported by a literature review (detailed later), in a scenario of optimal conditions for vegetation development. This means that we consider that the recovery process takes place with no constraining factors for vegetation development, such as water and/or nutrient availability, chemical alteration of the soil, or fire recurrence. Figure 1 shows the process followed for calculation of the recovery time.

The following subsections describe in detail each stage of this methodology, beginning with the assignment of the RTOC, and then the PSGC. Finally, we present the method for calculating the vegetation RT and a validation procedure to test the performance of the method. The methodology was implemented in a GIS environment using map algebra and spatial analysis tools to calculate and map the recovery time. The spatial resolution of the input parameters was 1 km × 1 km, except for the rainfall trend maps which were 15 km × 15 km.

1 Recovery time under optimum conditions (RTOC)

Some plant species are better adapted to fire than others and either better resist the impacts of fire or recuperate more quickly, depending on the regeneration strategies and horizontal and vertical continuity (Baeza and Roy, 2008). Initially, we made the RTOC assessment from lists of dominant plant species in the Forest Map of Spain (MAGRAMA, 1997), giving an individual characterization, in terms of their structure and regeneration strategy, to more than 500 species. As stated above, for characterization we assumed that the vegetation recovery process occurs under optimal conditions. We based the plant characterization on our experience (de la Riva et al., 2008; Duguy et al., 2012) and several studies of post-fire vegetation and response (e.g. Baeza and Roy, 2008; Barbéro et al., 1998; Buhk et al., 2007; Martínez Ruiz, 2005; Pausas et al., 2004; Tárrega and Luis-Calabuig, 1989; Trabaud, 1990, 1998, 2002; Vera de la Fuente, 1994). We did not find all the information required for the characterization of all species in Spain; as a result, several species are classified according to the authors' criteria alone. Accordingly, the initial time assigned could vary significantly depending on the local characteristics of each site and the influence of some parameters, as is the case for local topography (slope or aspect), climatic conditions, steppe vegetation or open scrub, and climatic aggressiveness from heavy rainfall and steep slopes (Baeza et al., 2007; Keeley, 2009). Table 1 shows the resulting combinations of structure and regeneration strategy, the RTOC assigned to each, as well as representative plant species for each vegetation structure and regeneration strategy category. Figure 2 shows RTOC spatial distribution in Spain.

Table 1. Approximate RTOC depending on vegetation structure and regeneration strategy and representative examples of plant species.

	Time (years)	Representative species	Source
Grassland	2	–	Pereira et al. (2013)
Resprouter shrubland	6	*Buxus sempervivens*, *Quercus coccifera*	Trabaud (1990); Martínez Ruiz (2005)
Seeder shrubland	10	*Juniperus thurifera*, *Ulex parviflorus*	Baeza and Roy (2008); Martínez Ruiz (2005)
Resprouter tree	20	*Quercus robur*, *Quercus ilex*, *Quercus faginea*	Tárrega and Luis-Calabuig (1989); Martínez Ruiz (2005)
High seeding tree	15	*Pinus halepensis*, *Pinus pinaster*	Barbéro et al. (1998); Martínez Ruiz (2005)
Low seeding tree	45	*Pinus sylvestris*, *Pinus nigra*	Barbéro et al. (1998); Martínez Ruiz (2005)

2 Plant species growth constraints (PSGC)

This section describes the process followed to obtain the values of PSGC due to both water availability and soil loss.

a Water availability. We derived the increase in RTOC depending on water availability in the area (F_w) from the precipitation data reported in the Vegetation Series map of Spain (Rivas and Gandullo, 1987). This map was initially developed to delineate areas of recognized vegetation units (also referred to as series) to determine the great diversity of forest ecosystems in Spain. However, each of the different series was also assigned a typical rainfall category (arid, semiarid, dry subhumid, humid, and hyper-humid) based on annual local precipitation, which allows the assessment of water availability by grouping these rainfall categories, subsequently recoding them to a numeric value of the increase ratio (F_w) of the RTOC. This map is particularly suitable for achieving the objectives of this research, because orographic parameters and bioclimatic characteristics were considered in the process of mapping the vegetation series. Table 2 and Figure 3 show the correspondence between typical rainfall intervals and the ratio of increase (assigned following the criteria of the present study) and rainfall distribution.

b Post-fire soil erosion. Soil erosion is another major negative outcome of forest fires, particularly in the Mediterranean region (San-Miguel-Ayanz et al., 2012b). Within Europe, the risk of water-driven soil erosion is particularly high in the Mediterranean region where autumn rainstorms often follow summer wildfires (Pausas and Vallejo, 1999). The susceptibility of a burnt area to soil erosion depends on the intensity of the fire and the degree to which the vegetation cover is removed (San-Miguel-Ayanz et al., 2012b). The evaluation of the RT increment as a function of soil loss (F_e) was carried out using a spatial analysis of the distribution of soil erosion in post-fire conditions. To this end, the Pan-European Soil Erosion Risk Assessment model (PESERA; Kirkby et al., 2004) was used. PESERA is a spatially distributed model at 1×1 km resolution for quantification of water soil erosion. A model of erosion at regional level is necessary to serve as a starting point for modifications to the RTOC. PESERA, which is more detailed than models such as USLE (Wischmeier and Smith, 1960), includes information on several soil parameters, such as soil erodibility, readily available soil water capacity, and propensity

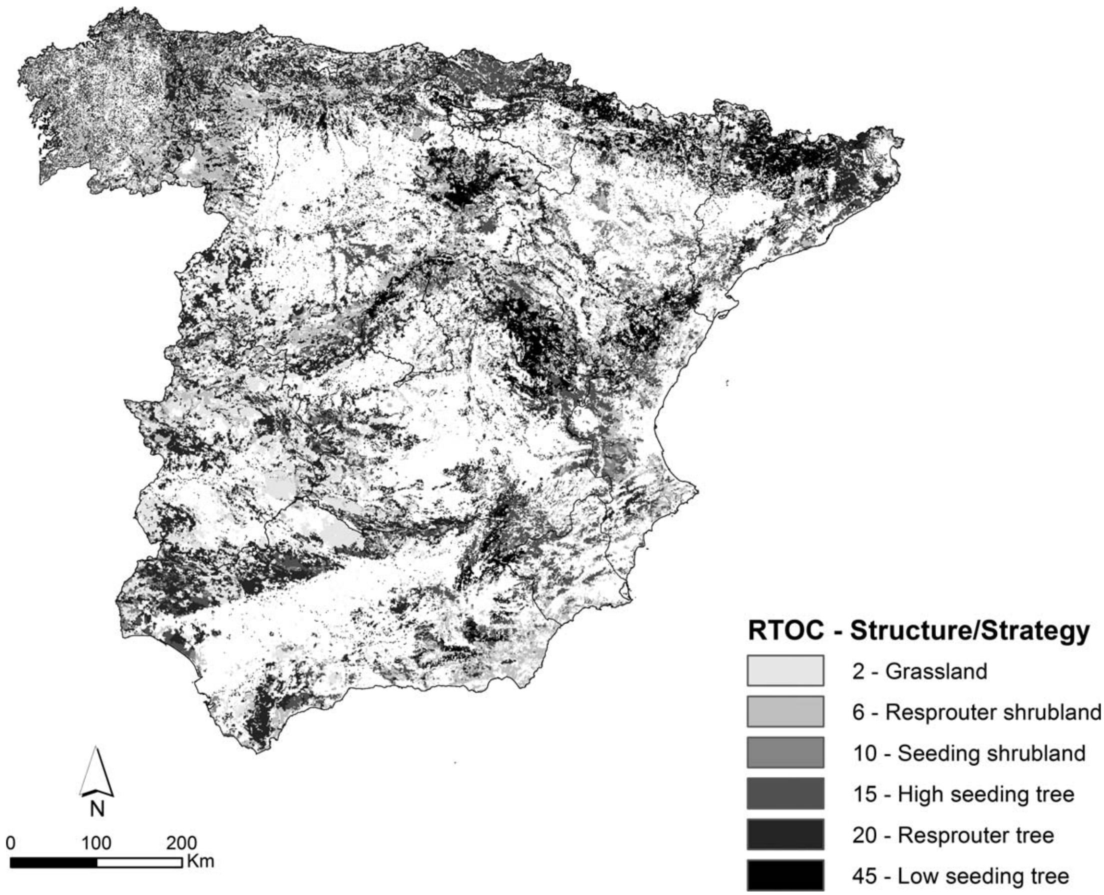

Figure 2. Spatial distribution of RTOC.

Table 2. Water availability, post-fire erosion rates, and corresponding RT increase ratios.

Rainfall category	Precipitation (mm)	F_w	Post-fire erosion rate (E_f) (Mg ha^{-1} year^{-1})	F_e
Hyper-humid	>1600	0.000	<0.04	0.000
Humid	1000–1600	0.075	0.04–0.13	0.075
Subhumid	600–1000	0.150	0.13–0.36	0.150
Dry	350–600	0.600	0.36–0.86	0.225
Arid-Semiarid	<350	1.200	>0.86	0.325

for crust in order to define soil water storage capacity. The PESERA model was developed to provide spatial information on erosion risks at European level using a simple conservative erosion model, which is broken down into components that depend on climate, vegetation, soil factors, and topography. The physical model is based on a one-dimensional soil–vegetation–atmosphere transfer type scheme for surface hydrology, coupled where appropriate to a dynamic model for generic vegetation growth and/or remotely sensed land-use data (Kirkby

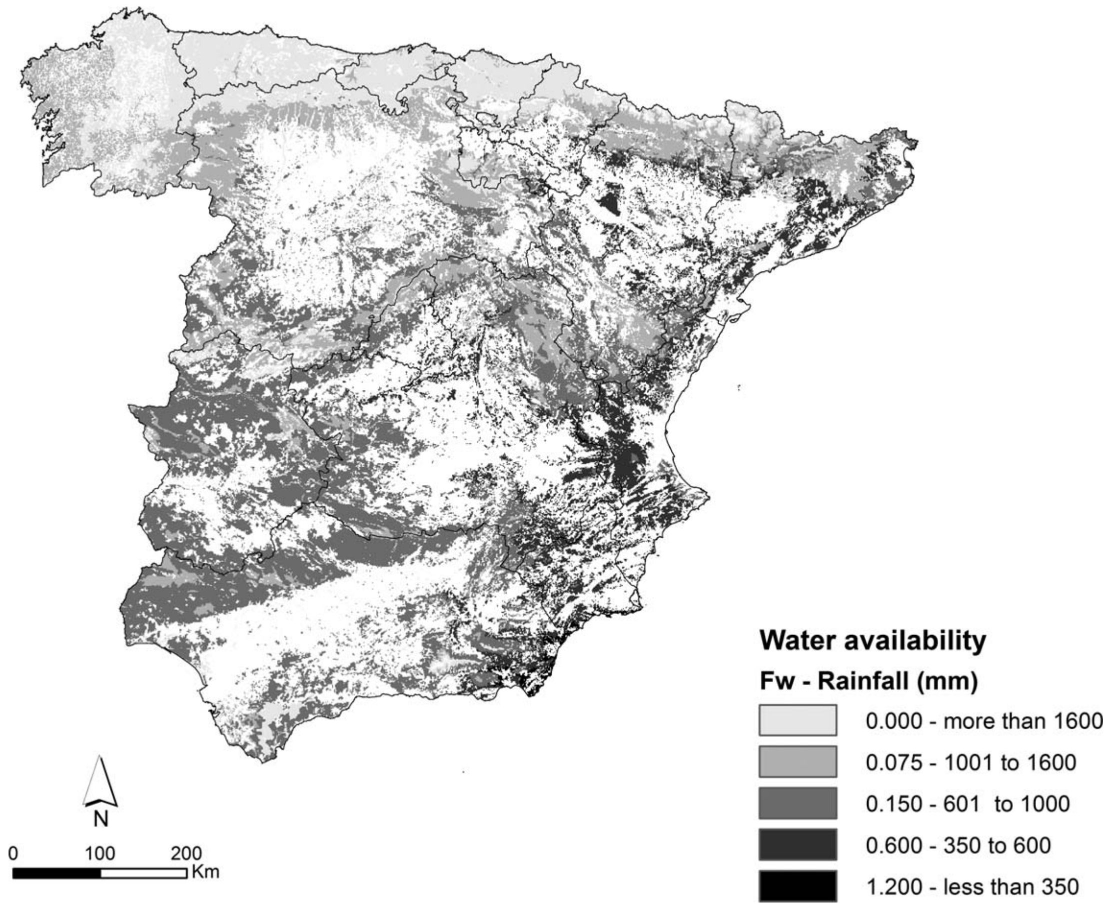

Figure 3. Spatial distribution of annual rainfall and water availability increase ratio.

et al., 2004). This model can be used as a tool at regional level, comparable to others such as the USLE (Wischmeier and Smith, 1960). Model results are validated at a basin scale and compared with data obtained using different methods of erosion measurement. More specifically, PESERA validation is based on comparison with erosion plot (40 m^2), small catchment (0.01–1 km^2), and reservoir (1–100 km^2) data (Cerdan, 2003; Tsara et al., 2005; Van Rompaey et al., 2003). These data have been used primarily to modify the pedo-transfer functions, particularly for soil erodibility.

In the current study, the Spanish subset of the European-scale PESERA map was used, although modifications have been made relating to the erosion processes that follow severe wildfires. An extensive literature review indicated great variability in the effect on erosion of plant cover loss resulting from fire. The erosion rate (Mg ha^{-1} year^{-1}) increments range from an increase of 18.6 (Soto and Diaz-Fierros, 1998; Soto et al., 1994) to 5200 (Shakesby, 2011; Shakesby et al., 1994, 2002) times the initial erosion rate. Given the very considerable heterogeneity of these values, due to differences both in ecological conditions where the experiments were carried out and in the design and techniques used (erosion traps, rainfall simulations, erosion plots, etc.), we selected the ERMiT model

Table 3. Increase factor of soil erosion by bioclimatic region, vegetation structure, and slope.

Structure	Slope (%)	Mediterranean region	Euro-Siberian region
Forest	<15	1.80	1.95
	15–45	1.70	1.75
	>45	1.70	1.75
Shrubland	<15	1.80	1.75
	15–45	1.80	1.75
	>45	1.80	1.80
Grassland	<15	1.80	1.70
	15–45	1.75	1.75
	>45	1.70	1.75

(Robichaud et al., 2006) to modify the PESERA pre-fire erosion rates. The ERMiT model integrates information on climate indicators, soil (texture), topography (slope and slope length), plus the type of vegetation affected and the severity level of the fire, thus allowing simulations to assess fire-caused increases in erosion rates. The model uses a probabilistic approach that incorporates temporal and spatial variability in weather, soil properties, and burn severity for forests, rangeland, and chaparral hill slopes. ERMiT allows calculation of the percentage increase in the pre-fire erosion rate (PESERA) in several vegetation communities, which are characterized in terms of climate, soil, and topography indicators, given a specific fire severity (high severity in our case). The ERMiT model simulations were carried out in several locations considered representative of each Spanish bioclimatic region (derived from Rivas and Gandullo, 1987) and where climatic data were available (four locations in the Euro-Siberian region – A Coruña, Oviedo, Santander, and Bilbao – and five in the Mediterranean – Madrid, Barcelona, Valencia, Seville, and Zaragoza). This enabled us to develop different scenarios covering several combinations of vegetation structure, slope, and fire severity. We calculated the increase factor for each location, dividing the erosion rate obtained for a high-severity fire by the pre-fire erosion rate. These quotients were calculated for each combination of vegetation structure, slope, and bioclimatic region and expressed as an average value. We carried out this process using the results provided by ERMiT for the first two years after a fire. Experimental data and measurements demonstrated that soil losses are significantly higher after a forest fire, being quickly reduced after 2–4 years (Cerdà and Doerr, 2005). We summarized the average increase factors obtained for the two years after burning into a single value to calculate the amount of erosion increment as consequence of wildfires. Table 3 summarizes the average increase factors.

We then modified the soil erosion rates reported in the PESERA model, including the factor of erosion increase obtained from the simulation with ERMIT, using the following equation:

$$E_f = \sum E_{pre} F_{resy}$$

where E_f is the corrected erosion rate (Mg ha^{-1} year^{-1}), E_{Pre} is the original erosion rate reported in the PESERA model, r is the bioclimatic region, e is the vegetation structure, s is the slope interval, and F_{re} is the soil erosion increase factor in bioclimatic region r and year y.

Once the erosion rates were corrected to take account of the effect of losing the protection of the vegetation cover as a result of fire, we reclassified the values into five intervals (by quantiles) to assign the RTOC increase ratio.

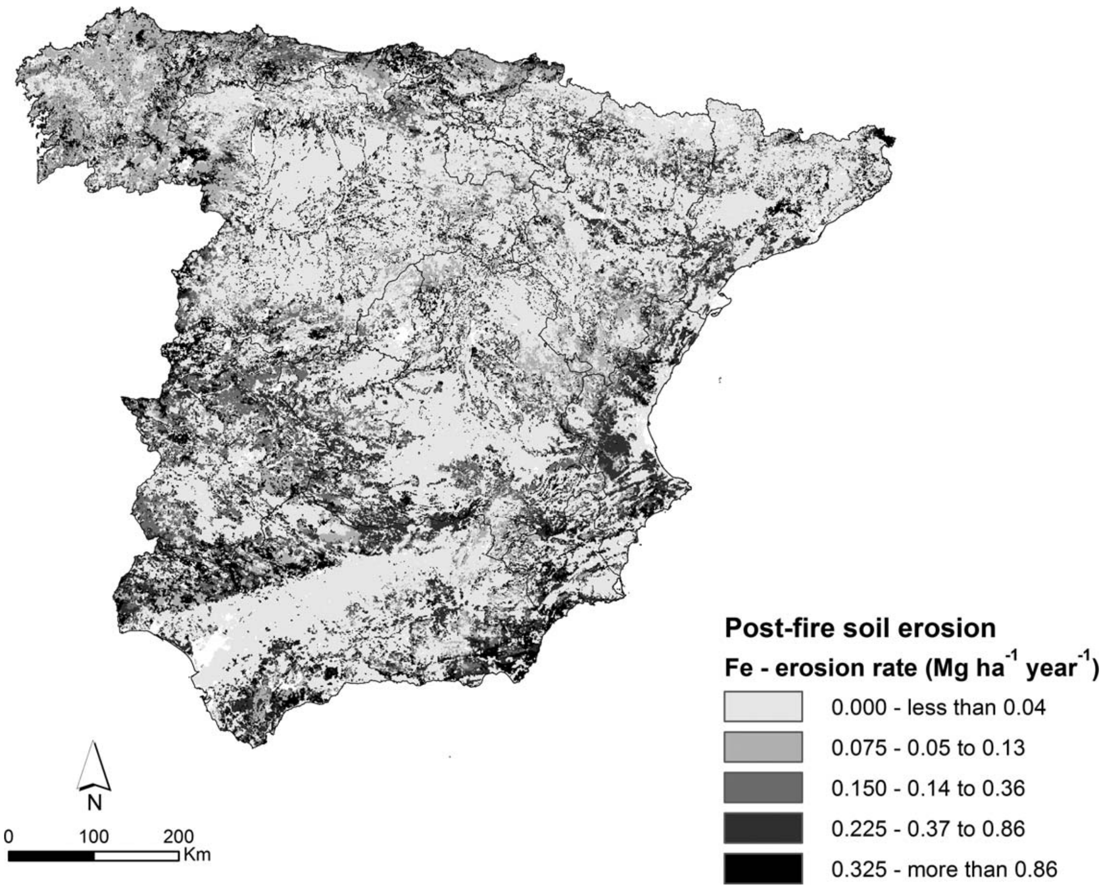

Figure 4. Spatial distribution of post-fire erosion rates and increase ratio.

Table 2 and Figure 4 show the soil erosion increment factor and its spatial distribution.

c Rainfall trends. Climate trends are a key factor in vulnerability assessment (González et al., 2010). Most climate-change predictions imply increased air temperatures and less summer rainfall for the Mediterranean basin (Hertig and Jacobeit, 2008; Schröter et al., 2005). Adverse climatic conditions (i.e. drier conditions) in many of the areas affected by fires may have caused lower rates of post-fire vegetation recovery (San-Miguel-Ayanz et al., 2012a). Hence, the observed changes in temperature and precipitation provide indicators of the potential change of the biome of an ecosystem (González et al., 2010). In this context, using observed climate data accounts for the impact of climate change that has already occurred (González et al., 2010).

Rainfall trends were included in the RT calculation as a weighting factor of the PSGC. In this sense and in general terms, we consider that a decrease in precipitation (negative rainfall trends) should imply a decrease in water availability; thus, the influence of a lack of water increases (San-Miguel-Ayanz et al., 2012a). A similar behavior is expected in the case of soil erosion, though in the opposite direction: here an increase in precipitation (positive rainfall trends) should increase its effect on the RT (Pausas and Vallejo, 1999), if water erosion is

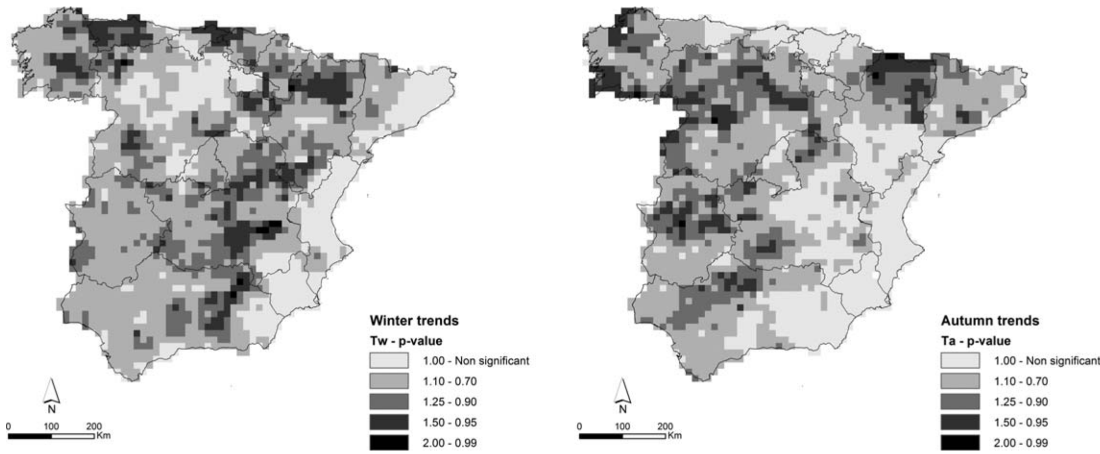

Figure 5. Spatial distribution of seasonal rainfall trends and their increase ratio. Left: negative winter trends. Right: positive autumn trends.

considered the main erosion mechanism. To include this behavior in the recovery time model, we used the reported rainfall trends in de Luis et al. (2010). In that study, the spatial variability of seasonal precipitation regimes in the Iberian Peninsula were calculated for a temporal period of observations of 50 years from 1946 to 2005, using the Mann–Kendall test (Mann, 1945). The spatial variability of the seasonal trends is characterized according to the sign and significance level of the observed trends. As the rainfall trends were calculated only at seasonal level, we used winter trends to weight water availability, considering this to be the most effective season for plants to capture water, due to low potential evapotranspiration. We used autumn trends for soil erosion weighting, as this is the most critical season due to the dryness of the soil following summer (Pausas and Vallejo, 1999), the decreased vegetation cover from the loss of leaves in deciduous communities, and torrential rains (de Luis et al., 2010). It should be noted that when considering seasonal trends instead of annual trends we are including the analysis of intra-annual variability of the precipitations. Rainfall trend weights ranged from 1 in locations where there was no significant trend (p-value < 0.70) to 2 in locations with significant with p-value > 0.99. Figure 5 shows the spatial distribution of winter (T_w) and autumn (T_a) trends, respectively, the significance levels, and their corresponding PSGC weights.

3 Recovery time (RT)

The RT was calculated as the sum of RTOC and the time increase from the PSGC:

$$RT = RTOC + T_{Fw}T_w + T_{Fe}T_a$$

where T_{Fw} is the time increase from water availability, T_w is the winter rainfall trend weight, T_{Fe} is the time increase from soil loss, and T_a is the autumn rainfall trend weight.

III Model validation

Validation is often a complex issue in ecological models (Rykiel, 1996). Here we suggest a validation procedure based on the previous work by Pérez-Cabello (2002) and Pérez-Cabello and Ibarra (2004). These works proposed the multi-temporal monitoring of changes in NDVI (Normalized Difference Vegetation Index) in burned plots as a tool for assessing the reconstruction process of various forest communities. In the present work we applied the same method to determine an approximate time span for the validation of the RT values. The NDVI has been the most frequently used tool for monitoring, analyzing, and mapping temporal and spatial post-fire variations (Díaz-Delgado et al., 2002, 2003; Riano et al., 2002; Viedma et al., 1997). NDVI is also used as a validation instrument in analyses similar to that in this paper (Bisson et al., 2008).

The NDVI is related to changes in the amount of green biomass, pigment contents and concentrations, and leaf water stress (Gong et al., 2003); that is why it emphasizes the regeneration process of burnt areas more clearly than the respective spectral signatures (Riano et al., 2002; Viedma et al., 1997). However, NDVI responds more to changes in leaf area than to changes in the overall biomass (Henry and Hope, 1998), reaching saturation levels at high LAI (leaf area index) values (Wang et al., 2005). Therefore, tracking post-fire vegetation recovery using NDVI should be limited to the most recent development stages – 10–20 years after fire – as it registers information related to the vegetation cover (Tanase et al., 2011).

We based the validation methodology on monitoring the temporal evolution of the recovery process in burned plots by measuring the NDVI values for several plant communities affected by severe wildfires during several years after burning. We calculated NDVI values from Landsat TM images corrected geometrically and radiometrically to ensure the consistency of the results. Cloud spots were deleted from each image to avoid undesired radiometric effects. The calculated NDVI values were compared with the pre-fire conditions (also characterized in terms of NDVI) to determine an approximate time for plant recovery. The method has been applied to seven plant communities (*Pinus sylvestris*, *Pinus nigra*, *Pinus halepensis*, *Quercus ilex*, *Quercus faginea*, *Quercus coccifera*, and *Buxus sempervirens*) affected by high severity (dNBR > 660; normalized burn ratio; Cocke et al., 2005) wildfires in 1985 and 1986 in the Huesca Pyrenees region (see supplementary material at http://ppg.sagepub.com/content/by/supplemental-data). These plant communities are considered as representative examples of vegetation structure and regeneration strategy categories (see Table 1). Moreover, the analysis region is particularly suitable for validation since it is located in a transition area from Mediterranean to Euro-Siberian regions and, therefore, plant communities in this area are a representative example for our purposes. Ten examples of affected plant communities, four in 1985 and six in 1986, compose the validation sample. The NDVI data for 1985 were obtained from Pérez-Cabello (2002) who calculated the NDVI for burned communities of *Pinus sylvestris*, *Quercus ilex*, *Quercus faginea*, and *Buxus sempervirens* since 1984 (pre-fire) until 1997. NDVI values for 1986 fire-affected communities were calculated from 15 Landsat TM images during the period 1984–2007 in *Pinus sylvestris*, *Pinus nigra*, *Quercus ilex*, *Quercus faginea*, *Quercus coccifera*, and *Buxus sempervirens* communities. We determined the recovery time span by comparing post-fire and pre-fire NDVI values, considering the affected community as recovered from the fire disturbance when the post-fire NDVI is higher than the pre-fire one. However, in those cases where the affected communities did not reach pre-fire NDVI values during the analysis period, a logarithmic profile evolution curve was projected from the observed NDVI data to establish a recovery time span. To complete the validation procedure, we compared predicted RT values with the outputs from the NDVI monitoring.

IV Results

1 Recovery time

The main result obtained from applying the proposed methodology in mainland Spain is the RT map (Figure 6). A statistical summary of the results is also given in Table 4, which we

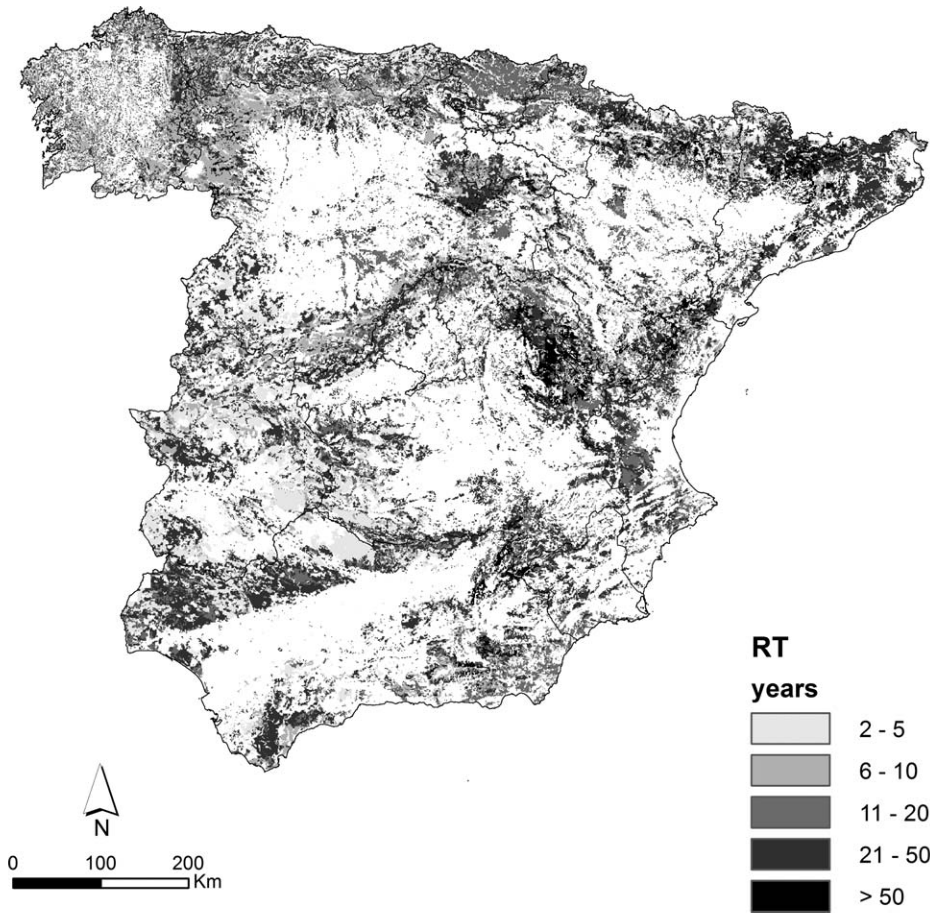

Figure 6. Spatial distribution of the post-fire vegetation recovery time.

constructed by using a zonal statistics algorithm, with RT values, and the categories from Figure 2 as zonal layer.

Results suggest a RT range from 2 to approximately 100 years for grassland communities and tree communities with low germination (mainly *Pinus nigra* and *Pinus sylvestris*), respectively. However, significant differences exist in the geographical distributions of times, mainly between Euro-Siberian and Mediterranean biogeographical regions. The higher RTs were obtained for low seeding tree communities, located mainly on the Mediterranean coast, ranging from 45 to 100 years. We also found both high seeding and resprouter tree communities, with average RTs around 21 and 25 years, respectively. Despite having similar average RT values, great difference exists between the maximum RTs, with values near 40 years in the case of high seeding trees and 50 years for resprouter trees in the Mediterranean region. Shrubland communities showed a RT span near 8 and 13 years in resprouter and seeder communities. Finally, grassland areas presented the lowest RT, at an average of 2.5 years.

Regarding the PSGC influence, although they contributed significantly to the recovery time, this contribution had an average of 22%, but exceeding 60% of RT in some areas. The highest values of PSGC contribution were found in the Mediterranean region in areas with

Table 4. Statistical summary of the post-fire recovery time for each vegetation category.

Plant community	Min time	Max time	Avg time	Stdev
Grassland	2	5.3	2.6	0.40
Resprouter shrubland	6	15.8	8.1	1.95
Seeder shrubland	10	26.4	13.5	2.80
Resprouter tree	20	51.1	25.5	4.37
High seeding tree	15	39.6	21.5	4.64
Low seeding tree	45	100.7	52.9	7.62

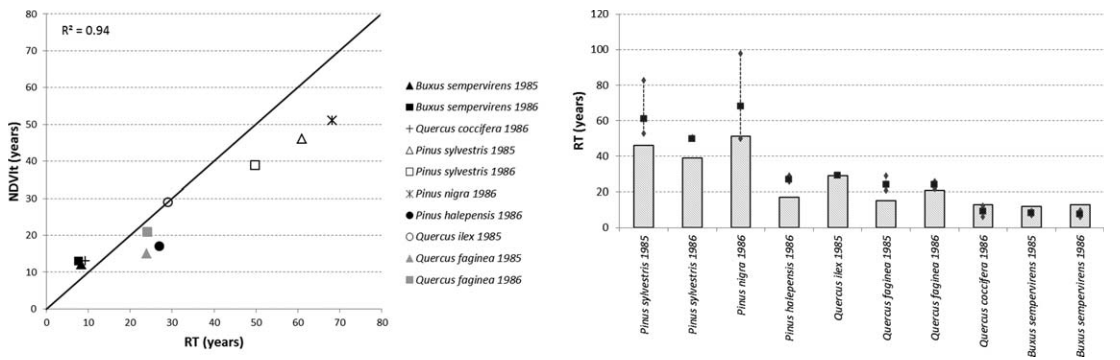

Figure 7. Validation results in burned plots. Left: scatterplot RT-NDVIt. Right: RT range (dotted line), average RT (black square), and NDVI recovery time values (filled bars).

low water availability influenced by significant negative winter rainfall trends. Thus the RTOC, which reflects the structure and regeneration of the dominant plants, has the higher contribution to the RT (c. 78% of the final RT).

2 Model validation

According to the validation results (Figure 7 and Table 5), the observed NDVI recovery span is reasonably similar ($R^2 = 0.94$) to the RT prediction, although shorter. The overall observed behavior is an overestimation of the RT in tree communities, whereas in shrubland communities RT seems to be underestimated. The best performance is achieved in resprouter shrubland (*Quercus coccifera* and *Buxus sempervirens*), and resprouter tree (*Quercus ilex* and *Quercus faginea*) communities, with differences between RTs and NDVI under five years. Seeding tree communities, both high and low seeding (*Pinus halepensis*, *Pinus sylvestris*, and *Pinus nigra*), showed the poorest agreement with differences of six years in high seeding tree communities and more than 15 years in low seeding trees (*Pinus nigra*).

V Discussion

Some plant species are better adapted to fire than others, depending on the regeneration strategies and horizontal and vertical continuity (Baeza and Roy, 2008). Particularly, plant communities in the Euro-Siberian region present lower RT values due both to the presence of resprouter communities, considered as highly resilient (Rodrigo et al., 2005), and to the higher water availability due to Atlantic climate conditions. On the other hand, in the Mediterranean region, predominantly on the Mediterranean

Table 5. Statistical summary of validation results. The results are characterized in terms of year of burning (Year), number of pixels of RT (N), maximum value of RT (RT max), minimum value of RT (RT min), average value of RT (RT avg), recovery threshold according to the NDVI evolution ($NDVI_t$), pre-fire NDVI value (NDVI 84), and accuracy of the projected NDVI evolution curve (R^2 Regr).

Species	Year	N	RT min	RT max	RT avg	$NDVI_t$	NDVI 84	R^2 Regr
Pinus sylvestris	1985	4	53	83	61	46	0.69	0.79
Pinus sylvestris	1986	5	49	51	50	39	0.69	0.82
Pinus nigra	1986	6	50	98	68	51	0.67	0.72
Pinus halepensis	1986	4	26	29	27	21	0.62	–
Quercus ilex	1985	1	29	29	29	29	0.60	0.70
Quercus faginea	1985	5	21	29	24	15	0.61	0.71
Quercus faginea	1986	4	22	26	24	21	0.68	–
Quercus coccifera	1986	6	7	13	9	13	0.68	–
Buxus sempervirens	1985	2	7	10	8	12	0.57	–
Buxus sempervirens	1986	12	6	10	8	13	0.59	–

coast, higher RTs were reported. This occurs as a consequence of, among other factors (ecological, edaphic, topographic, land use, etc.), low water availability due to low rainfall, as well as to the frequent torrential rainfall events in autumn (Baeza et al., 2007; Bisson et al., 2008) thus increasing soil loss. Rainfall during the first autumn after a fire is particularly crucial for the germination of most seeders (Moreno and Oechel, 1992) since dry conditions likely delay post-fire regeneration in seeding communities (Rodrigo et al., 2004). However, the site-level soil water availability is the result of the interaction of precipitation inputs with various factors such as soil depth, type and degradation, and topography. Besides, the temporal distribution of rainfall and factors such as the history of disturbances also influences the recovery time. In any case, it seems that the type and characteristics of the vegetation are the most important parameters influencing the post-fire regeneration process (Alloza et al., 2006; de la Riva et al., 2008). Accordingly, the post-fire vegetation dynamics seem to differ substantially between the studied seeding and resprouting communities as the latter are highly fire-resilient with a much faster vegetation recovery rate (Broncano et al., 2005; Duguy et al., 2012; Pérez-Cabello and Ibarra, 2004). However, differences in RT are not restricted to average values. Resprouter communities, mostly resprouter tree communites, show less variability in the RT values than the seeding ones according to the standard deviation values reported in Table 4 (4.37 in resprouter tree communities, 4.64 in high seeding trees, and 7.62 in low seeding trees). This might occur because in ecosystems characterized by highly resilient plant communities (resprouter Mediterranean species), site-level abiotic limitations are often overcome by the resprouter's ability to quickly recolonize the open space created by fire with its undamaged below-ground organs (Duguy et al., 2012).

On the other hand, the results from the validation procedure have confirmed that the overall performance of the proposed methodology ($R^2 = 0.94$) is sufficient to consider the method a useful tool for supporting regional forest management and planning. According to the validation results, the best performance is observed in resprouter shrubland (*Quercus coccifera* and *Buxus sempervirens*), and resprouter tree (*Quercus ilex* and *Quercus faginea*) communities, whereas seeding tree communities (*Pinus halepensis*, *Pinus sylvestris*, and *Pinus nigra*) showed the poorest agreement. In tree communities, RT values appear to be overestimated, compared to the NDVI recovery time span. This result is likely related to the fact that a similar spectral response of the plant does not always implicate a complete physiognomic recovery, mainly because the NDVI is more related to vegetation cover (Tanase et al., 2011). In addition, it should be noted that in some cases (*Pinus halepensis*, for instance) the NDVI evolution could be strongly influenced by the presence of other associated plant communities, such as shrublands or grasslands (Pérez-Cabello, 2002; Pérez-Cabello and Ibarra, 2004).

Despite the promising overall results, the validation sample should be increased to cover the high variety of plant communities in mainland Spain. Consequently, the results reported for the validation should be considered as a pilot validation that mainly aims to exemplify the procedure. Developing a validation sample sufficiently wide to be used for full validation would be very time-consuming, involving the characterization of numerous burned plots, and gathering and correcting a great amount of remote sensing images. Nevertheless, RT values obtained for the different communities analyzed are reasonably similar to the expected periods in accordance with the existing literature. In the case of resprouter communities, Rodrigo et al. (2005) indicate that they reach similar pre-fire cover about 30 years after the fire, a time span close to the average 25.5 years obtained following the method proposed in this work (RT values are showed in Table 4). Broncano et al. (2005) and Pérez-Cabello and Ibarra (2004) indicated similar recovery time intervals. In high seeding tree communities (*Pinus halepensis*), Broncano et al. (2005), Kazanis and Arianoutsou (2004), Ruano et al. (2012), and Trabaud (1998) suggest recovery times starting from 15 years, reasonably similar to the 21.5 years obtained in this study. Finally, the high RT values calculated in the case of low seeding trees (*Pinus nigra* and *Pinus sylvestris*) are consistent with Rodrigo et al. (2004) who indicated that there is little chance of recovery of the original pre-fire situation which supports the existence of a very long RT. However, it should be noted that RT values are not equally reliable since the validation sample does not cover the whole RT range. Accordingly, the most reliable recovery times are the values below 21 years, since this is the time period for which NDVI recovery is directly measured in the calibration data. The next most reliable period extends to 51 years, the time period for which NDVI recovery has been indirectly measured. Beyond 51 years is the least reliable period which should be carefully considered.

We can conclude that the recovery time calculated from the RT method is reasonably similar to the recovery threshold obtained from the NDVI evolution, particularly in resprouter communities. However, there is a certain degree of uncertainty insofar as we are providing a regional-scale assessment. This uncertainty is mainly linked to the quality of the input data and to the capacity of the NDVI for monitoring vegetation recovery. The first source of uncertainty, quality of the input data, is inherent to the geographical information, since it is not possible to find a data source that perfectly represents real conditions, especially when working at regional scales. For instance, errors in the spatial distribution of plant communities in the Spanish Forestry Map are influencing the whole RT calculation process because all the assumptions are made on its basis. On the other hand, the uncertainty related to the NDVI as a tool for monitoring post-fire vegetation recovery comes from the fact that the NDVI takes into account the vegetation cover rather than the physiognomy of the plants, which can lead to misinterpretation of the observed recovery time span. However, the comprehension of its limitations allows the use of the NDVI as a reference for validation purposes. Furthermore, note that the proposed method for RT assessment also has some limitations and drawbacks, although it may be considered as a relative improvement compared to similar methods designed to evaluate post-fire dynamic processes like Duguy et al. (2012) or Bisson et al. (2008), since our methodology offers a framework for its implementation at regional scale as well as quantitative results. The main drawback arises from the fact that, although we conducted an extensive literature review to support our choices, some subjectivity remains in the values of RTOC or PSGC increase ratios, which is particularly important when applying the method to a different region, as it is very likely that either plant species and/or environmental conditions differ significantly from those described herein. For example, it is difficult to establish a direct comparison of our work and the one developed by Duguy et al. (2012) or Bisson et al. (2008) mainly because both outputs and analysis scale are different.

VI Conclusions

Our results indicate a high heterogeneity in RT values, both between the examined plant communities and between the various regions of peninsular Spain. This is not surprising, given that peninsular Spain has a wide range of physical and environmental conditions. This is mainly due to the coexistence of two contrasting biogeographic regions (Euro-Siberian and Mediterranean), which also show a high internal variability in conditions, directly linked to variability in terrain and the resulting different climatic conditions. This fact increases the complexity of the analysis of any environmental parameter or process, especially at regional scales.

The PSGC contributed significantly to the reconstruction time; however, type and characteristics (structure and regeneration strategy) of the dominant plant community seem to be the most important parameters influencing the post-fire regeneration process.

On the other hand, although the validation results are restricted to 10 fire events that occurred in 1985 and 1986, and extending the validation to the whole of peninsular Spain will be a very time-consuming process, involving the gathering and correction of a great amount of remote sensing images, in general we consider that the RT values obtained are reasonably well adjusted to the expected evolution of plant communities after fire disturbance.

We believe that the proposed method is sufficiently strong to be valuable in several fields, such as land management, forest fires, assessment of socio-economic vulnerability, and environmental services. This applicability is mainly due to the simplicity of the method, which requires few variables. Additionally, the methodology is integrated and developed within a GIS that allows one not only to map the results but also to perform different kinds of spatial analyses and mapping. Additionally, we are working on extending the validation procedure in the framework of another project focusing on the development of predictive models of ecological vulnerability to fire.

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