Agroforest Syst (2012) 85:331–349
DOI 10.1007/s10457-011-9423-2

Landscape vulnerability to wildfires at the forest-agriculture interface: half-century patterns in Spain assessed through the SISPARES monitoring framework

M. Ortega · S. Saura · S. González-Avila · V. Gómez-Sanz · R. Elena-Rosselló

Received: 10 February 2011 / Accepted: 3 August 2011 / Published online: 26 August 2011

Abstract Large-scale socioeconomic changes in recent decades have driven shifts in the structure of Spanish rural landscapes, particularly in those located at the forest-agriculture interface (FAI), as well as in their wildfire regime. Using data from more than 200 16 km^2 landscape plots in Spain surveyed between 1956 and 2008 through the SISPARES monitoring framework, we assessed the FAI vulnerability to wildfires and identified the main landscape structural factors related to an increased number of wildfire events. We found that the most vulnerable landscapes were those with high road density, high diversity of land uses and, most importantly, with fine-grained forest-agriculture mixtures. Ignition frequency was lower in those landscapes where crops and woodlands coexisted but distributed in large and well-separated patches, and much lower where both land uses were combined within an integrated production and management system ("dehesas"). We discuss the geographical distribution patterns and temporal trends of the different FAI types during recent decades. We conclude that such approach is useful to forecast the mutual interactions between land use pattern changes and wildfire regime in the Mediterranean agroforestry mosaics. This would also provide an ecological base for developing a complementary, cost-effective and durable passive strategy of wildfire management targeted to modify the inherent FAI susceptibility to ignition events.

Keywords Agroforestry mosaics · Landscape dynamics · Spatial pattern vulnerability · Structural fire risk · Wildfire occurrence

M. Ortega
Unit of Forest Fires, Department of Silviculture and Forest Management, Centre of Forest Research CIFOR, Instituto National de Investigación y Tecnología Agraria y Alimentaria, INIA, Carretera de La Coruña, km 7.5, 28040 Madrid, Spain

M. Ortega · S. González-Avila · V. Gómez-Sanz · R. Elena-Rosselló (✉)
ECOGESFOR-UPM, Research Group of Ecology and Sustainable Forest Management, Escuela de Ingeniería Forestal y del Medio Natural, Universidad Politécnica de Madrid, Ciudad Universitaria s/n, 28040 Madrid, Spain
e-mail: ramon.elena.rossello@upm.es

S. Saura
Department of Forest Management and Economics, Universidad Politécnica de Madrid, Ciudad Universitaria s/n, 28040 Madrid, Spain

Introduction

Wildfires play an essential role in shaping the landscapes worldwide (Bond et al. 2004). A complex combination of wildfire disturbances of varying frequency, intensity and extension might be a main direct contribution to landscape heterogeneity and to interspersed mosaics together with other changes in human land uses (Pickett and White 1985; Krumel et al. 1987; Turner 1989; Forman 1995). This is particularly true in the Mediterranean, where landscapes have been modified by fires during millennia (di Castri and Mooney 1973; Naveh 1975; Trabaud 1980; Trabaud et al. 1993). On the other hand, landscape structure strongly modulates fire occurrence, spread and behavior (Turner et al. 1994; Scheller and Mladenoff 2004; Malamud et al. 2005). In summary, wildfire regime and landscape pattern have been shown to have strong mutual interactions (Romme 1982; Finney 2001).

The exact timing, location and extent of a particular wildfire are impossible to forecast. However, at coarser temporal and spatial scales the patterns of fire occurrence can be successfully predicted, mainly based on climatic and topographic variables and on models that relate fire ignition and behavior to different characteristics of the fuel (vegetation) types over which the fire may spread (Scott and Burgan 2005). Since climatic variables cannot be controlled through management, the efforts in wildfire prevention mainly rely in modifying forest structure and composition (Agee and Skinner 2005). In turn, forest composition and structure are themselves influenced by quite the same variables that drive fire regime, such as climate, landform, disturbance history, different human impacts and socioeconomic pressures (Romme 1982; Flannigan and Harrington 1988; Bessie and Johnson 1995; Finney 2001; Malamud et al. 2005). Therefore any shift in these factors may have large impacts not only on fire patterns and landscape structure themselves, but also in the highly complex interactions between the two. This may produce nonlinear and synergistic responses that go beyond what could be expectable from the change in each of these variables alone.

Among the factors affecting fire regime, the role of landscape configuration is one of the most poorly understood and commonly underestimated. Just recently, several studies have analyzed its importance in different regions (Schmidt et al. 2008; Catry et al. 2009; Gralewicz 2010; Moreno et al. 2011; Padilla and Vega-García 2011; Hely et al. 2010). The research efforts in this direction are however overwhelmingly small compared to those devoted to other wildfire drivers. Some possible causes are (1) the assumed relatively high stability of landscape patterns, (2) the difficulty of analyzing, monitoring and managing landscape patterns over the large spatial and temporal scales at which these changes and interactions operate, and (3) the lack of a conceptual and methodological background to model those landscape mosaics and their interactions with ecological processes.

Nowadays, the development of landscape ecology offers new ways for the analysis of landscape patterns and their interactions with disturbance events. Indeed, the relationship between landscape patterns and processes is one of the main objectives of landscape ecology studies (Forman and Godron 1986). Therefore, this seems an appropriate approach to unravel the factors behind the changes in wildfire occurrence and spread.

Mediterranean landscapes are far from being static ingredients of the complex combination of fire regime drivers. Southern Europe has experienced large scale socioeconomic changes since the mid XXth century that have driven profound shifts in the traditional rural landscape structure (Mazzoleni et al. 2005). Migration to cities depopulated rural areas (Ayuda et al. 2010). Abandonment of marginal productivity crops, decline in grazing and increase of stabled livestock caused the transition to more uniform rural landscapes. The decrease of firewood harvesting, coupled with an expansion of forest plantations to replace some of the more labor demanding agricultural crops, have led to the coalescence of forested areas and to larger woodland patch sizes. The recent changes are in contrast with the secular society-environment coevolution of Mediterranean landscapes, where forestry and agriculture have competed and coexisted. Their coexistence in Spain took place either within the same agrosilvopastoral integrated system (landscapes of “dehesas”) or as an intermixture of forest and agricultural patches with spatially well separated management systems (Rey Benayas et al. 2007). These forest-agriculture interfaces are frictional zones among different social mentalities and interests. The conflicts among foresters, shepherds and farmers have been stronger in the Mediterranean because the arid climate increases the competition for the scarce water resources.

Increased landscape vulnerability to wildfires may be expected when the unstable balance between structure and functional processes was broken down during the past century. This might be the case in those areas where the use of fire as an agropastoral tool is now coupled with a lack of silvicultural treatments in the contiguous woodlands that has increased forest biomass and fuel continuity. A higher occurrence and frequency of ignition sources in these border forest lands may be a key factor behind an increased wildfire risk and vulnerability at wider spatial scales. As a consequence of those facts, the wildfire frequency in Spain has hugely increased in the last five decades, mostly intentionally lighted by farmers in the forest-agriculture interface (EGIF 2009).

Despite its importance in the Mediterranean, few research projects have been devoted to the forest agriculture interface in comparison with the urban forest interface or the pure forest landscapes (Badia and Pallares 2006, among others). Analyzing the interactions between the forest-agriculture interface and the wildfire regime requires long term data on the agroforestry mosaics, which has been an important methodological bottleneck hampering progress in this direction. At the moment, just studies at a limited spatial or temporal scale have been carried out (Romero-Calcerrada et al. 2008; Vilar et al. 2010; Chuvieco et al. 2010). Only through the implementation of long term monitoring systems that problem can be solved. Several national monitoring initiatives have been recently developed in Europe, which deliver nation-wide samples on selected landscapes, habitats, and regions, such as the UK countryside survey, the Swedish environmental mapping project NILS, and other continental-wide initiatives (European Project EBONE, www.ebone.wur.nl/UK). In Spain, the SISPARES monitoring system has been designed to evaluate the trends in the structure of rural landscapes (Elena-Rosselló et al. 2005). Such system was conceived with an appropriate size of the sampling sites, a homogeneous spatial scale, and a time window long enough to record changes from 1956 to present. Besides, more than 60% of the SISPARES samples are located in the forest agriculture interface. This makes SISPARES particularly adequate for assessing the trends in the landscape biophysical internal features that make the landscape susceptible to wildfire. In addition, after 40 years of systematic wildfire data acquisition in Spain, we are now in a good position for a deeper study on those relationships and hypothetical causes of fire occurrence.

In this study we aimed to assess the spatial and temporal patterns of landscape vulnerability to wildfires at the forest agriculture interface across Spain in the last five decades. We intended to provide (1) a better understanding of the drivers of fire ignition and damage in Mediterranean agroforestry interfaces as related to the abundance, pattern and mixture of the different land uses, and (2) conceptual bases to land planners for designing landscape patterns which minimize wildfire vulnerability. These new planned patterns should decrease the inherent high susceptibility of the forest agriculture interface to fire ignition events. They should be key features for developing a cost-effective and long-term, durable approach for wildfire management focussing on a passive protection strategy.

Materials and methods

Study area characterization and stratification

Our study area covered all the Spanish lands within the Iberian Peninsula and the Balearic Islands (about 497,000 km^2 in total). Geographically, Spain is a Southern European country located at the western part of the Mediterranean basin. Its position among the Atlantic Ocean and the Mediterranean sea, its high mean elevation (2nd in Europe, after Switzerland) and the abundant mountain systems are responsible for varied climatic types: Oceanic, Mediterranean, Continental and Alpine. Lithologically, there is a wide diversity of bedrocks from most geological periods. All these biophysical factors have made possible a diversified vegetation of sclerophillous, evergreen conifer and deciduous hardwood forests. Historically, agriculture, forestry and husbandry have coexisted in most of the regions and have modelled its rural landscapes after millennia of human use of wildfire as a powerful management tool.

Landscape spatial structure data have been recorded in a spatially and temporally consistent manner over all the study area in four different years (1956, 1984, 1998 and 2008) in 206 selected sample plots within the SISPARES monitoring framework (Elena-Rosselló et al. 2005, www.sispares.com). The sampling intensity, taking into account the plot size and the total extension of the study area, is about 1/146. Landscape data were derived from the interpretation of aerial photographs combined with field surveys in the two most recent dates. Each sample has an extension of 16 km^2, as in other landscape monitoring studies (Honnay et al. 2003), and a minimum mapped patch size of 1 ha. However, when the delineated patches (of at least 1 ha) comprise multiple portions of different land uses (each smaller than 1 ha) that are intermixed at such fine scale, these patches are classified as a fine-grained mixture of those land use types.

The distribution of these sample plots is fully representative of the biogeoclimatic structure of Spain, because they were selected using a stratified sampling procedure as related to the CLATERES classes (Elena-Rosselló et al. 1997), with each SISPARES sample representing one of these CLATERES classes. CLATERES was designed for getting estimates at a national scale of a wide variety of ecological features and used a classification approach adapted from the country survey land classification system (Bunce et al. 1996). The CLATERES classes consist in ecological zones with similar biotic and abiotic factors such as topography, vegetation, and climate. The CLATERES classes were used to aggregate the SISPARES plots in five climatic strata (humid, sub-humid, sub-arid, arid and hyper-arid) in order to evaluate the differences in wildfire occurrence across different climatic conditions.

Characterizing landscape structure at the forest-agriculture interface

We considered as the forest-agriculture interface those landscapes in which agricultural and forest lands occupied each at least 10% but no more than 90% of total area. Two sets of variables were selected to characterize the spatial pattern composition and configuration of the landscape plots at the forest agriculture interface and to analyze the relationships between those variables and wildfire occurrence in the rural areas across Spain. The first set of variables referred to the area covered by the following eight major land use types: forests, forest plantations, riparian woodlands, shrubs, pastures, agro-forestry areas called "dehesas", crops, heterogeneous agricultural areas (lands mainly occupied by agriculture, with significant areas of natural vegetation), rocks, water bodies, and artificial surfaces (urban and industrial areas). These land use types correspond to level 3 of land cover CORINE classification (EEA 1995). The second set was made up with eleven landscape structure variables described in Table 1.

Wildfire data and statistical analysis of the relationships between ignition frequency and landscape structure

For characterizing the spatiotemporal patterns of wildfire occurrence we used the national fire database (EGIF), the most complete collection of wildfire data in Spain, which is compiled by the *Area de Defensa Contra Incendios Forestales* of the Spanish Ministry of Environment. EGIF records daily wildfire data since 1974, including their causes, which are summarized yearly at a 10 km × 10 km UTM square resolution. Wildfire ignitions intentionally caused by agricultural and forestry practices are included in EGIF. Such a database allows for a combined analysis of fire occurrence and landscape structure (as provided by the SISPARES framework described earlier) from 1974 to present. In particular, we analyzed the frequency of wildfire ignitions since 1974 in a total of 5101 UTM 10 km × 10 km squares. A total of 468,471 fire ignitions were recorded in the EGIF database within this period. Based on the available data, landscape vulnerability is better measured through data of the wildfire ignition rather than of burnt area (Gralewicz 2010), because the former is independent from the level of post-fire suppression and therefore better related to the structural landscape characteristics that determine the occurrence of wildfire events. Consequently the underlying intrinsic conditions leading to ignition are emphasized (Stocks 2002).

In order to spatially link the SISPARES and wildfire databases, we overlaid the 10 km × 10 km UTM grid of wildfire occurrence with the CLATERES classes. This process was facilitated because the CLATERES classes were designed with 2 × 2 km resolution and a UTM projection as for the wildfire data. For each CLATERES class (each corresponding to a continuous region with homogeneous biogeoclimatic conditions and represented by one SISPARES plot) we calculated 38 descriptors of wildfire occurrence from the UTM 10 km × 10 km squares intersecting that class: the number of ignitions in each of the 35 different years in that period and in the periods from 1974 to 1984, from 1985 to 1998 and from 1999 to 2008 (with the last year of each these three periods

Table 1 Metrics used to describe landscape composition (PR and SDI) and configuration (the rest of the metrics) in the 16 km^2 SISPARES plots across Spanish rural landscapes

Metrics	Description
Patch richness (PR)	Number of different land cover types present
Shannon diversity index (SDI)	Shannon index calculated from the proportions of total area covered by each land cover type
Patch density (PD)	Number of patches per unit area
Edge density (ED)	Total length of edges between patches per unit area
Mean patch fractal dimension (MPFD)	Twice the logarithm of patch perimeter divided by the logarithm of patch area (average for all patches)
Mean patch size (MPS)	Average size of patches in the landscape
Mean patch edge (MPE)	Average edge length of patches in the landscape
Largest patch index (LPI)	Proportion of total area covered by the largest patch in the landscape
Mean shape index (MSI)	Sum of the ratios between the perimeter and the square root of area for every patch in the landscape, divided by the number of patches
Forest-agriculture boundaries (FAB)	Total length of agricultural-forest edges per unit area
Fine grained forest-agriculture mixtures (FGFAM)	Percentage of landscape total area covered by fine-grained forest-agriculture mixtures. These mixtures are defined as conglomerates of adjacent units that are each smaller than 1 ha and correspond to both forest and agricultural lands. These are mapped as a single mosaic (heterogeneous) patch in the SISPARES monitoring framework
Road density (RD)	Total road length per unit area

FAB and FGFAM have been purposefully defined for the objectives of this study. Further details on the rest of the metrics can be found in McGarigal and Marks (1995)

matching to one of the successive SISPARES surveys). We hypothesized that the landscape composition and configuration recorded by SISPARES at each survey would eventually be associated with wildfire occurrence in the subsequent years. Consequently, landscape structure in 1956 was related with fire events in the period 1974–1984, 1984 with the period 1985–1998, and 1998 with the period 1999–2008. Large differences between wildfire occurrences in these three periods did not allow for a joint statistical analysis for the whole 1974–2008 period. Therefore, each period was normalized (mean 0, variance 1) resulting in three values of NYFF (normalized yearly fire frequency) at the 206 SISPARES sample squares.

An analysis of the factors determining landscape vulnerability to wildfire occurrence was performed through a two step statistical model aiming to assess (and remove) the climatic influence, and to assess the landscape structure influence across climate strata. The first step consisted in a general linear model of repeated measures (three periods) with the annual total rainfall (WorldClim, version 1.3., Hijmans et al. 2005) as the independent variable and the NYFF as the dependent variable. In the second step we used as the dependent variable the residuals obtained in the previous step in order to assess through a general linear model to what extent FAI landscape structure variables could explain wildfire occurrence beyond what already captured by climate. Since landscape structure variables can be highly correlated (e.g. Riitters et al. 1995), the model in this second step only included those landscape structure variables that presented a Spearman correlation between them lower than 0.6; therefore not all the variables listed in Table 1 were finally entered in the model. The model also included those quadratic terms of the variables in Table 1 that presented a significant relationship with the residuals from the previous step.

Classifying the rural landscapes based on their vulnerability to wildfire occurrence at the forest-agricultural interface

We classified the rural landscapes in six types depending on the amounts of forest and agricultural lands and on the types of mixtures between them. This classification included two non-FAI landscapes, Types I (pure agricultural) and VI (pure forest). Type V was differentiated because it corresponds to a FAI predominantly managed under an integrated agroforestry system where most patches are sharing forest and agricultural uses (dehesas). The other three types corresponded to mosaics of spatially separated forest and agricultural lands that were differentiated based on the landscape structure variable that was most significantly associated to wildfire occurrence (step 2 in the model described in the previous section). The differentiation of these three types was performed through a regression tree analysis (CART) of the wildfire occurrence in the 206 SISPARES samples using the corresponding landscape structure variable, FGFAM.

The ability of the resulting FAI landscape typology to effectively discriminate the FAI landscape wildfire vulnerability was tested through ANOVA of the NYFF per partial period and for the total study period. This allowed us to estimate how much variation of the wildfire frequency was explained by the FAI landscape types. For each period, the 206 SISPARES plots were classified in the FAI landscape types (Fig. 3) and characterized by the NYFF of their own CLATERES class.

Finally, the temporal patterns were assessed by analyzing the changes in FAI landscape type distribution through time. We also considered data on rural population and agrarian employment rates during the studied period as provided by the INE (Spanish National Institute for Statistics 2008) in order to support our discussion on the underlying causes for these temporal patterns.

Results

Spatial and temporal patterns of wildfire occurrence and causality

Total wildfire occurrence showed a mean value of 13,413 events per year from 1974 to 2008, but with three partial periods with remarkably different records as shown in Fig. 1. The first period from 1974 to 1984 was characterized by the lowest frequency of fire events (average baseline of 5,807 events) and a slow increase in wildfire occurrence. In the second period, from 1985 to 1998, the increase rate accelerated resulting in a maximum in the number of fires towards the end of the XXth century (average baseline of 15,758 events). The third period, starting from 1999, had the highest frequency (average baseline of 18,497 events). Nevertheless, the number of fires was initially stabilized and then decreased significantly in the last years.

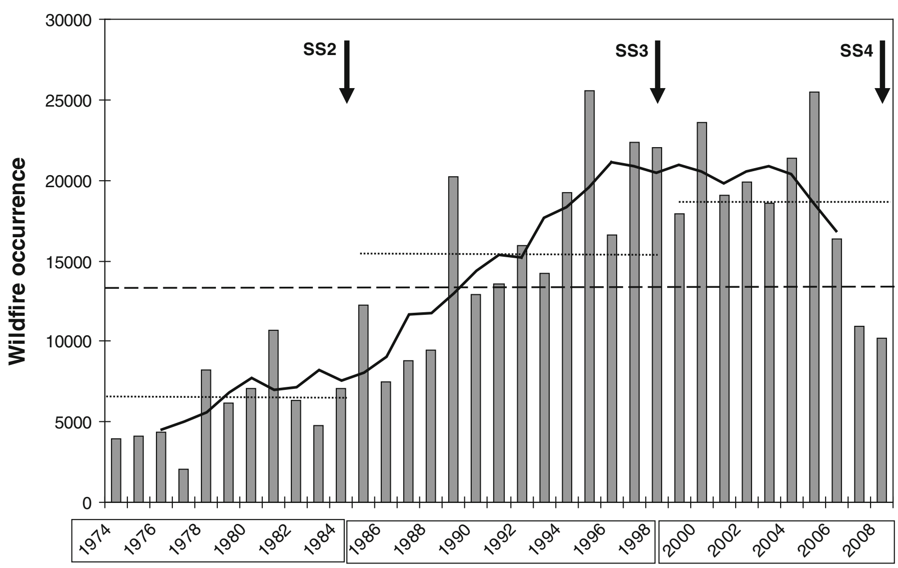

Fig. 1 Diagram of the yearly evolution of wildfire occurrence along the study period (1974–2008). *Solid* trend line corresponds to the 5 year mobile mean. *Narrow dotted* lines indicate average values for the three partial periods (5,807, 15,758 and 18,497 events, respectively). *Dotted* line is baseline for the whole study period (13,413 events). SISPARES surveys (SS# *arrows*) appear closely related to critical changes in the evolution trend: 1984 is closed to a inflection point in the growing trend, and 1998 is linked to the highest fire occurrence records

In all the three partial periods indicated in Fig. 1, the highest number of fire ignitions with respect to the mean number in each climatic stratum was found in the north and northwest of Iberian Peninsula (Fig. 2), that is, in the humid extreme of the Spanish climatic gradient. Some mountain systems like the Pyrenees and low lands over the south and the centre of Spain were in general less prone to wildfires (Fig. 2). This distribution showed an even more polarized and sharply defined ignition pattern through time, with a reinforcement of the high wildfire occurrence in the north-western regions, followed to a lesser degree by some areas around mountainous systems located towards the centre of Spain (Central and Iberian ranges), as shown in Fig. 2. However, the geographic pattern has been very stable in the whole studied time span. See Fig. 6.

The analysis of variance of NYFF in the three partial periods using the climatic strata of CLATER-ES as the classification factor indicated that the wildfire occurrence was significantly higher in the humid extreme of Spanish gradient. Significant differences between the first period and the next two periods were found in all strata ($F = 221.85$; $P < 0.001$).

As far as the direct causes are concerned, according to the EGIF database, in the period 1996–2008, 78.47% of total ignitions recorded in Spain were caused by humans: 47.45% deliberately lighted, and 31.02% by accidental or neglected activities. 4.78% were caused by natural factors, and 14.81% remained with unknown causes (MARM 1996–2005, 2006, 2007, 2008). But these trends are quite different in the humid and sub-humid regions, including Galicia,

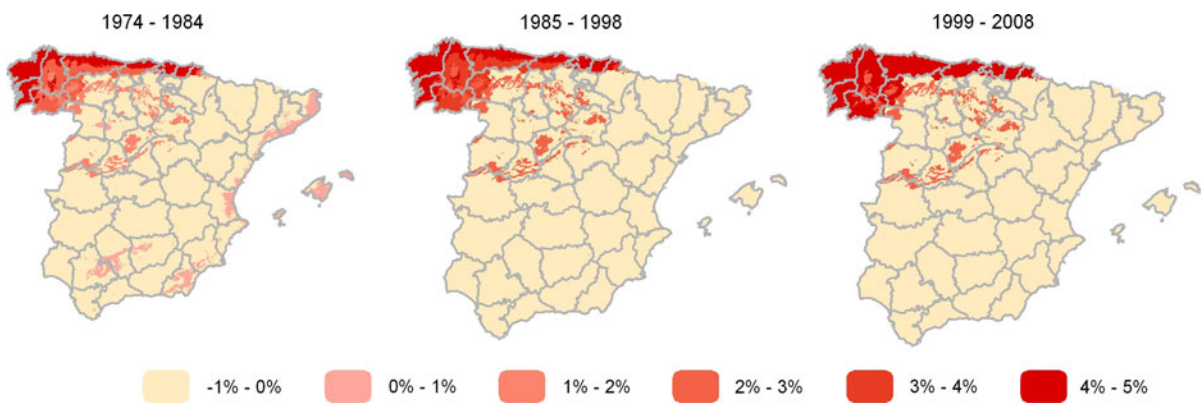

Fig. 2 Maps of the relative ignition rates of the geoclimatic strata in the three studied periods: 1974–1984, 1985–1998 and 1999–2008. The relative ignition rate of a stratum in a given period was calculated as the deviation from the average of fire frequency in the whole Spain during that period, expressed as a percentage of the baseline (Negative values indicate lower fire frequency than the average, and positive ones mean higher frequency). Absolute fire frequency in any stratum is calculated as the mean value of the yearly ignition frequencies of their 10 × 10 km squares. The three maps allowed comparing the geographical distribution pattern of fire occurrence, once the effect of the period factor was removed

LANDSCAPE VULNERABILITY TAXONOMY KEY				LANDSCAPE TYPES	
NO FAI	FOREST LUCT < 10%			Type I	PURE AGRICULTURAL
FOREST AGRICULTURE INTERFACE FAI	FOREST LUCT ≥ 10% & AGRIC LUCT + FGFAM >10 %	DEHESA < FOREST LUCT & DEHESA < (AGRIC LUCT + FGFAM)	FGFAM ≥ 10 %	Type II	FINE GRAINED INTERMIXED
			FGFAM < 10 %	Type III	MEDIUM GRAINED INTERMIXED
			FGFAM = 0 %	Type IV	COARSE GRAINED INTERMIXED
		DEHESA ≥ FOREST LUCT & DEHESA ≥ (AGRIC LUCT + FGFAM)		Type V	AGRO FOREST INTEGRATED
NO FAI	FOREST LUCT ≥ 90%			Type VI	PURE FORESTED

Fig. 3 Classification of forest agriculture interface of Spain based on the landscape structural features. *FAI* Forest agriculture interface, *AGRIC* agricultural coarse grain fields, *LUCT* landuse land cover type

Asturias, Cantabria and the Basque Country where ignitions recorded in the same period were 62.05% deliberately lighted, and 15.54% by accidental or neglected activities. Only 1.09% was caused by natural factors, and 19.52% remained with unknown causes.

Landscape structure variables related to wildfire occurrence

The effect of climate represented by annual total rainfall on wildfire occurrence was significant in the three periods explaining close to 50% of variance (Table 2). Residuals of this analysis were explained by means of landscape structure variables in each period. Fine grained forest-agriculture mixtures (FGFAM) and road density (RD) had significant effects in the three partial periods. Shannon diversity index (SDI) and mean patch size (MPS) had only significant effects in the second period (1985–1998).

FGFAM was the variable most significantly associated to wildfire regime. The quadratic relationship between wildfire occurrence and FGFAM (see Table 2) indicated that the landscapes with a moderate amount of fine-grained mixtures of forest and agricultural land uses (between 9.3 and 26.0%) were those with a higher number of fire ignitions, as shown in Fig. 4.

Testing the potential of the FAI types for assessing landscape vulnerability to wildfire

NYFF in the whole study period 1974–2008 was significantly higher in those landscapes classified as fine and medium-grained mixtures (Types II and III), and particularly in Type II. After applying the Games-Howell test, three groups of FAI landscape types were positively tested as different in terms of wildfire vulnerability: group 1: Type II, group 2: Types III and VI, Group 3: Types I, IV and V. these groups have implicitly tested the discriminant potential the FAI intermixed landscape of Types II, III and IV, showing significant differences in vulnerabilities (Fig. 5). In the overall 1974–2008 period, the landscape vulnerability to ignition events of the fine-grained forest-agriculture mixtures was about four times higher than the average vulnerability of all Spanish landscapes (Fig. 6). In the three partial periods, Type II was proved to be significantly the most vulnerable FAI landscape structural type. Among the other types, only Type III was positively tested as more vulnerable than Type IV. In general terms, the partial analysis has shown the same results than the overall one: the pure agriculture (Type I), the Integrated FAI (Type V) and the coarse grained intermixed FAI (Type IV) are significantly less vulnerable than national averages. On the other hand, medium grained intermixed FAI (Type III) and pure

Table 2 General linear analyses of (1) normalized wildfire frequency (NYFF) explained by annual total rainfall (ATR) in SISPARES plots using repeated measures; (2) residues of (1) explained by landscape structure variables of SISPARES plots of three dates: 1956, 1984, and 1998, matching with the wildfire occurrence period 1974–1984, 1985–1998 and 1999–2008, respectively

Dependent variable	Independent variable	*df*	1974–1984			1985–1998			1999–2008		
			R^2	*F*	*P*	R^2	*F*	*P*	R^2	*F*	*P*
(1) NYFF	ATR	204	0.45	169.7	0.001	0.50	206.2	0.001	0.49	193.0	0.001
(2) Residues of NYFF		197	0.14	3.9	0.000	0.23	7.3	0.000	0.21	6.5	0.000
	FGFAM			4.9	0.03		20.0	0.000		15.8	0.000
	FGFAM2			4.3	0.04		17.3	0.000		8.4	0.004
	SDI			n.s.	n.s.		10.1	0.002		n.s.	n.s.
	RD			8.1	0.005		7.6	0.006		7.6	0.006
	MPS			n.s.	n.s.		5.01	0.03		n.s.	n.s.
	LPI			n.s.	n.s.		n.s.	n.s.		n.s.	n.s.
	MPFD			n.s.	n.s.		n.s.	n.s.		n.s.	n.s.
	FAB			n.s.	n.s.		n.s.	n.s.		n.s.	n.s.

Acronyms of variables are the same as those described in Table 1

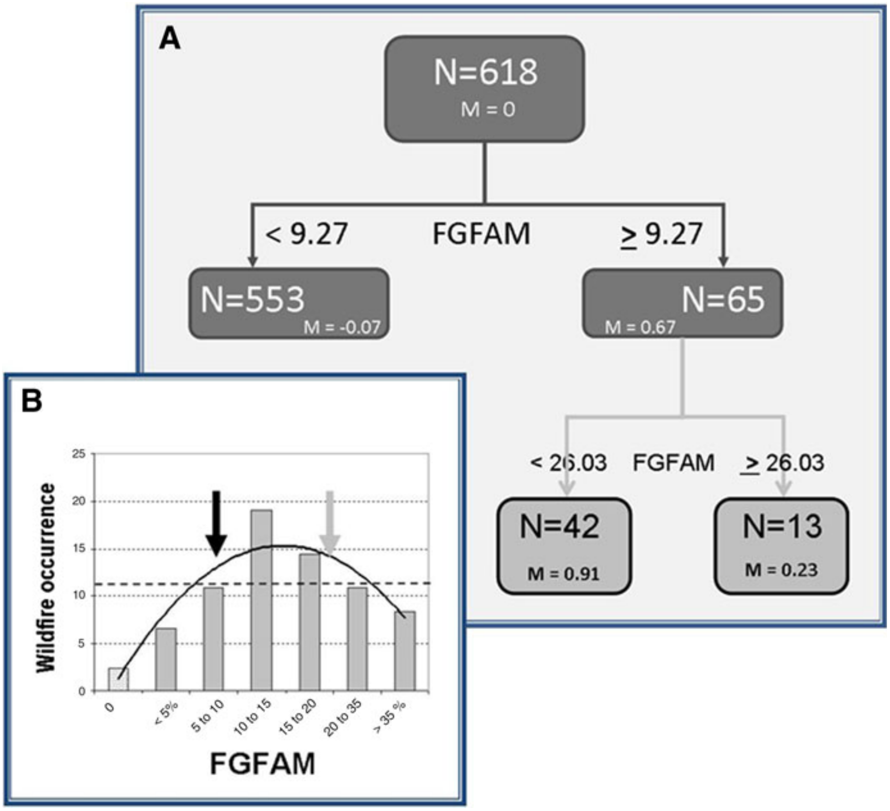

Fig. 4 **a** Dendrogram of tree regression analysis for the whole 1956–2008 period: 618 (206 SISPARES squares × 3 periods) residues from general linear analysis of wildfire occurrence per 10 × 10 km UTM cell (number of ignitions/year), once detracted rainfall factor, have been analysed using fine grained forest-agriculture mixtures index (FGFAM) as independent variable (risk estimation = 0.45; SE = 0.07). FGFAM was used since this landscape structural variable most significant correlated to wildlife occurrence, as shown in Table 2. Number of SISPARES plots and residual wildfire occurrence average in each *box* is indicated by N and M, respectively. The most significant division is established for FGFAM = 9.27%: taxonomic threshold of 10% for discriminating FAI Type II from FAI Type III. **b** Diagram of the distribution of wildfire occurrence among FGFAM. The quadratic model fitted shows a maximum wildfire occurrence at 15% of fine grained forest agriculture mixture. *Arrows* indicate the two break points identified for FGFAM in the CART analysis of **a**

forest (Type VI) showed the average vulnerability (Fig. 5).

Spatial and temporal patterns in the forest-agricultural interface types

While the pure and coarse-grained mixture landscapes were distributed throughout all or most of Spain, the fine-grained mixture landscapes (Type II) and the integrated agroforestry systems ("*dehesas*") (Type V) were respectively concentrated towards the NE (Galicia) and the SE (Extremadura), as shown in Fig. 6. These two latter landscape types were those that decreased most in the half-century evaluated period (Fig. 7). They did not only reduce their presence but their geographic scope as well, being increasingly concentrated in the two cores of their distribution (Fig. 6). A decrease was also found for the number of plots in the coarse-grained landscape category (Type IV) (Fig. 7). The landscapes that increased more in the 53 year period were those corresponding to Types VI (pure forest) and III (medium-grained mixtures), as shown in Fig. 7. Overall, the landscapes located at the forest agriculture interface (Types II, III, IV and V) decreased from 153 to 136 plots in the 1956–2008 period, although they always represented more than 60% of the total number of SISPARES plots (Fig. 7).

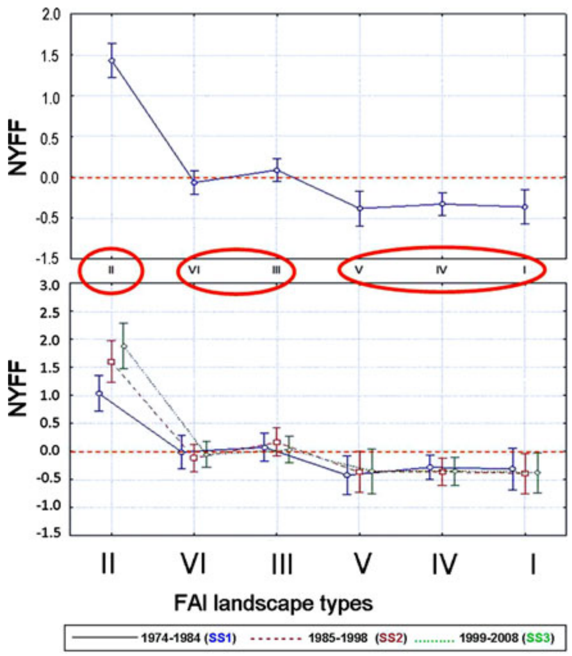

Fig. 5 Diagrams showing the results of analysis of variance of normalized yearly fire frequency (NYFF) in the proposed FAI landscape types. *Upper* diagram shows the general ANOVA of NYFF in 618 SISPARES sample squares (206 squares × 3 study periods) classified in the FAI landscape typology. *Lower* diagrams show the results of the three partial ANOVAs of NYFF carried out separately with the 206 SISPARES sample classified in 1956, 1984 and 1998 using the FAI landscape typology (SS = SISPARES Survey). According to Games-Howell test, the general ANOVA establishes three statistically different groups of FAI types marked with *circles*: group 1: Type II, group 2: Types III and VI, group 3: Types I, IV and V ($P < 0.05$)

62% of the SISPARES plots did not change their landscape structure type during the five decades. The most dynamic structural types over the entire period were those of the three intermixed structures in the forest agriculture interface (Types II, III and IV), while the most stable categories were the pure landscapes (Types I and VI), as summarized in Fig. 7. The most dynamic partial period in the agricultural-forest interface was the one between 1956 and 1984 (see Fig. 7), with the major flows occurring between the different scales of mixture of the two land uses: from Types III–VI, from Types II–III, from Types IV–III and reciprocally. Most of the losses of agricultural-forest mixture areas were due to their conversion to purely forested landscapes (Fig. 7).

Discussion

Landscape structure features most correlated to wildfire occurrence: diversity of land uses, fine grained mosaics and road density

First of all, our results have confirmed that climate regime is the overall key ecological factor for determining the landscape vulnerability to wildfire in Spain. Both CLATERES climate stratification and the annual total precipitation are significantly correlated to fire frequency. Accordingly, landscapes in the more humid climates have higher biomass production and consequently are much more vulnerable than those under arid conditions. (Vázquez et al. 2006).

However, our results indicate that, even when the climate effect on fire vulnerability has been accounted for, a high wildfire occurrence is associated to more diverse landscapes, a higher road density and the presence of fine-grained agroforestry mixtures. Our results for the impact of roads on fire ignition frequency concur with other studies that have studied in detail the role of transportation networks as a source of fire ignitions (Romero-Calcerrada et al. 2008; Gralewicz 2010; between others). Roads are an incision and a pathway into the forest landscape that favours the presence of wildfire sources such as those related to urban picnickers and cross-country motor bikers.

The impact of landscape diversity on ignition frequency is also well known, and is a clear consequence of the coexistence of different land uses that most probably produces conflicts among the different land management objectives in a particular landscape. In our analysis, the values of this diversity index are largely determined by the relative amounts of forest and agricultural areas, which are the two land uses dominating the landscapes here studied. A diversity of decision makers eventually increases socio-economic conflicts and results in the use of fire as a powerful tool to transform and shape the landscapes towards those stages that are considered more beneficial for some land owners (Romme 1982).

Indeed, a very special and prominent feature of the Mediterranean landscapes has been the strong competition among foresters, shepherds, and farmers for lands where water is a scarce limiting resource. Although the vegetative period is rather long in the

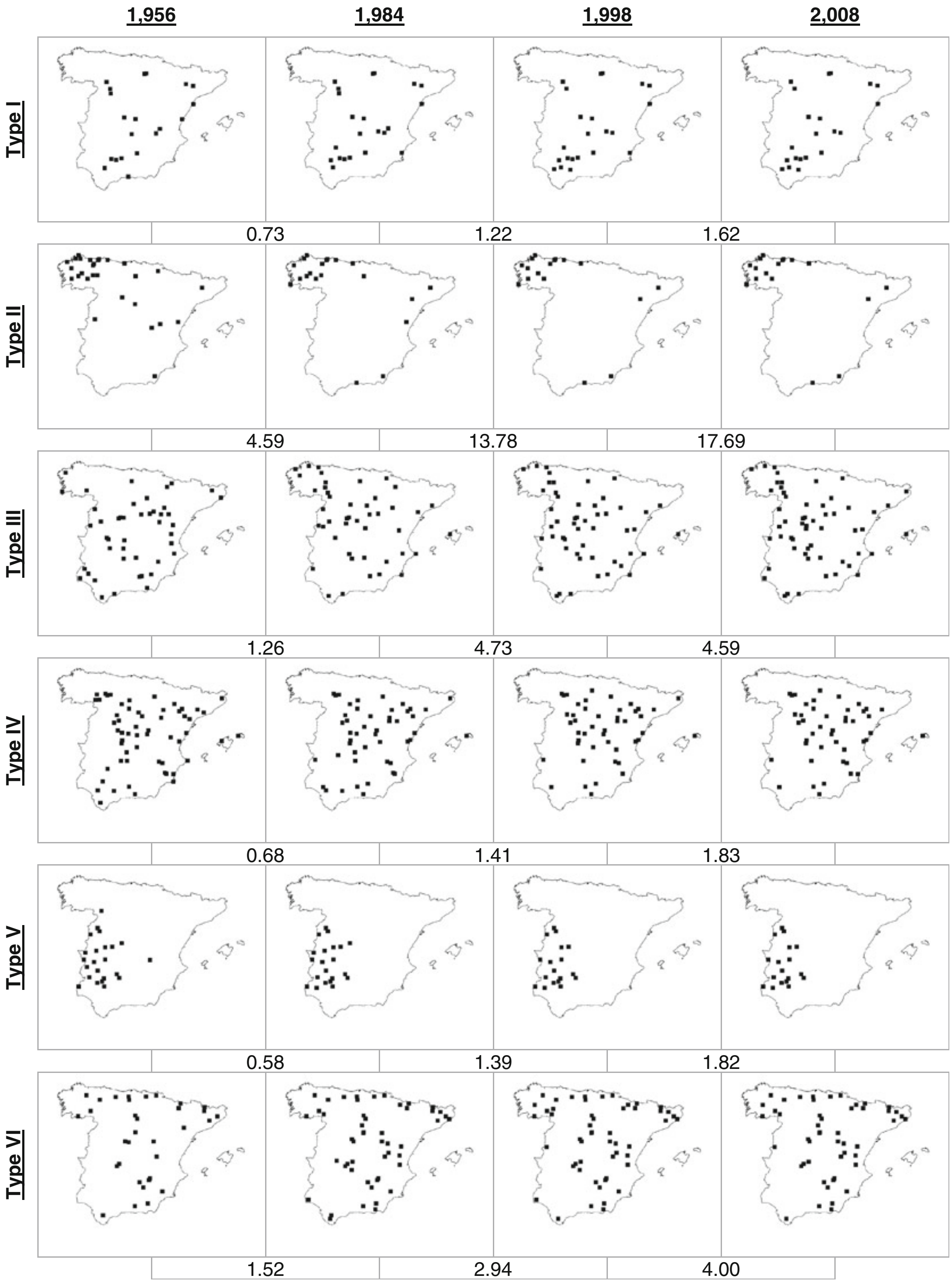

Fig. 6 Spatial distribution of the SISPARES landscape plots corresponding to each of the six vulnerability typologies for the four different dates in which these plots were surveyed. Observed mean annual values of wildfire occurrence per 10 × 10 km for each typology are indicated

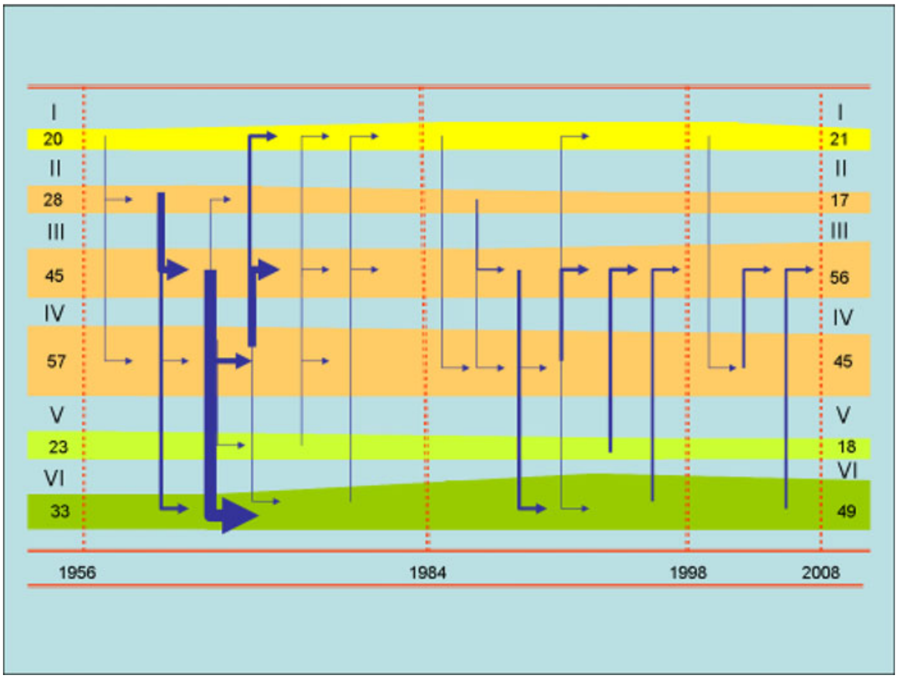

Fig. 7 Flow chart of the dynamic evolution and changes among structural landscape types. The *arrows* indicate changes from one type to another within a given period, with *thicker arrows* corresponding to a larger number of SISPARES landscape plots changing between those types. For example, the *first arrow* by the *left* indicates that between 1956 and 1984, one plot changed from Types I–II, and another one changed their type from II–IV. The initial (1956) and final (2008) number of plots within each landscape type are indicated at the *left* and *right* of the chart respectively

Mediterranean, there is a long drought season of about two months in summer that largely reduces the biomass production. Only by irrigation the summer season can be productive in intensive agricultural systems. Otherwise, extensive non-irrigated crops and pastures require much larger field areas with rotations of several years. Consequently, farmers have been managing against woody vegetation as a water competitor to their crops. The lack of hedge-rows among fields is a landscape feature resulting from that competition (Jongman and Bunce 2008). Since the middle ages, the shepherds, well organized in a common institution called Mesta, had lobbied in Spain for ensuring pastures all year around for their flocks. Because this Mediterranean summer drought affected most of the lowlands, it was necessary to move their flocks up to the mountain highlands, looking for the summer pastures, and grazing through forested areas (or through their interface with other land uses) along their way. This transhumance was very important in terms of landscape impacts because it occurred at a national scale (affecting almost the whole Spain) and because it involved frequent burning to avoid the competition of woody plants for the scarce resources (Pineda 2001). Our results suggest that these effects and relationships, although not as prominent today as in other historical times, still play an important role to determine wildfire occurrence patterns, at least in some Mediterranean areas.

Fine-grained versus coarse-grained forest-agricultural mixtures: a key feature of Mediterranean landscape structure, a great difference for wildfire occurrence

The impacts of forest-agriculture diversity on wildfire occurrence were much more prominent when these land uses were interspersed at fine spatial scales (below 1 ha). Our results showed that fine-grained forest-agricultural mixtures were associated to ignition events much beyond what could be expectable by simply considering the total amount of these two land uses in the landscape. Fine-grained mixtures seem to enhance the competition for space and resources at the more local scales in which management and traditional landowners use and perceive the landscape (Chas-Amill et al. 2010). Therefore, either intentionally or not, these mosaics make the overall landscape more susceptible to ignition events associated to uncontrolled slash burning, woody vegetation control, etc. It seems clear from our results that not only landscape composition (as captured for instance by the diversity indices discussed earlier) but also spatial configuration are determinant of fire regime in the Mediterranean rural landscapes.

Indeed, we have shown that the amount of fine-grained forest-agricultural mixtures (FGFAM) is the main structural landscape factor behind an increased landscape vulnerability to wildfires. Interestingly, such mixture type does not need to be dominant in the landscape for triggering a high wildfire occurrence. The most vulnerable landscapes were those in which those fine-grained mixtures represented between 9 and 26% of total area. Within the complex and synergistic set of factors that drive fire regime, a relatively modest amount of these fine-grained agricultural mosaics seem to be a sufficient and potentially explosive ingredient boosting fire occurrence. That amount of fine-scale mixtures guarantees the required number of fire ignitions that, coupled with other adequate conditions (e.g. meteorological) and with the rest of the landscape presenting other more fuel-abundant and continuous forest types, determines the largest vulnerability to wildfires of all the landscape types at the forest-agriculture interface.

"Dehesas" as an integrated and non-fire prone agroforestry system

On the other extreme of the vulnerability gradient in the Spanish rural landscapes, agroforestry integrated landscapes (Type V) showed the less intense wildfire regime, at similar levels than pure agricultural (Type I) or coarse grained mixture (Type IV) landscapes. These landscapes are defined by the predominant presence of agrosilvopastoral systems called "dehesas", similar to the Portuguese "montados" (Pinto-Correia 2000). Geographically, "dehesas" and "montados" are mainly located in the south western regions of the Iberian Peninsula: Extremadura, western Andalucía, and south western Castilla-León in Spain and Alentejo in Portugal. Extensive granite, quartzite and acid slates peneplains under arid Mediterranean climate conditions have developed poor and rocky soils where low dense sclerophillous forest are the climax vegetation, mainly cork oak and holm oak.

Historically, the low biomass productivity as well as the landownership system, made these regions less populated than Iberian average rates. Very large properties, over 500 hectares, dominate the landscape where local population is concentrated in a few rural towns. The farm average size has produced a very coarse grained pattern, even coarser than the one in the pure forested landscapes.

The reasons for the very low vulnerability of those landscapes dominated by "dehesas" are twofold. First, from a biophysical point of view, the "dehesa" has a reduced forest biomass productivity, with low fuel density and continuity that make difficult the wildfire ignition and spread. Typical stem density for the sclerophillous climax tree species (holm and cork oaks) are about 20–40 trees/hectare. Simultaneously, cattle, sheep and pigs keep herbs and scrubs under control. Second, from the socioeconomic and management perspective, the "dehesa" farms are based on very large properties where land uses coexist within an integrated agroforestry production system. Only one person makes the decisions in case of conflicts among agriculture, forestry and husbandry uses (Campos et al. 2007). Fire is under control of the "dehesa" managers, and is very rarely used unless for charcoal production. On the other hand, the urban population has restricted access to the private states, so their impact on wildfire occurrence is marginal in these agroforesty lands (Houérou 1993).

Temporal trends in wildfire occurrence and landscape structure: socioeconomic factors as driving forces

The investigation of the wildfire causes in Spain concludes that human activities (deliberate or involuntary) are responsible of the 70% of ignition events. Consequently, we have to consider the socioeconomic factors as the most important drivers of the short term changes in the fire frequency and regime that have been here reported during this period.

During the 1970s the rural population was abandoning their traditional activities, such as firewood harvesting, crops in the marginal arable lands and extensive sheep and goat grazing practices. The Spanish population employed in the primary sector with agrarian activities has dramatically decreased from 47.6% in 1950 to 6.86 in 2005 (INE 2008). The abandonment was generalized across Spain, but started earlier in the northern regions and then in the central and southern ones. During almost the whole 20th century, the decrease of agrarian jobs induced the migration to urban areas. But in 1980 a reverse migration flow started, keeping the rural population in rates around the 20%. All these processes were part of a vast social development across Spain that has transformed a rural to an urban society, both mentally and geographically (Hoggart 1997).

The transformation of the rural population is the sociological process that triggered the outbreak of wildfires in Spain (Vázquez and Moreno 1998). It produced an increasing fuel continuity and accumulation of biomass that previously was largely extracted for firewood and other rural uses, as it still happens nowadays in Northern Africa (Alexandrian et al. 1998). The few remaining farmers and shepherds kept controlling the woody biomass, but they had to handle larger quantities than in earlier times with a scarcer labour force in their depopulated areas, which induced them to increasingly rely on fire as a management tool in their agricultural lands and surrounding landscapes. At the same time, a new social factor and ignition source raised as urban tourists started to spend their weekends and holidays in the countryside, barbequing with little control and poor knowledge of the associated risks and responses in the forest environment.

Starting in the mid 1980s another socio-economic factor was reported for producing 2.4% of wildfire occurrence until end of the century (EGIF 2009). The legal regulation for nature protection and biodiversity conservation of about 28% of the total area of Spain took place in less than 5 years. Local population, farmers, shepherds, landowners, and hunters reacted against what they interpreted as confiscation of their rights and future potential revenues. Some wildfires were ignited as a complain reaction.

All these changes were coincident with a steady increase in wildfire occurrence in the forest-agriculture interface. When matching the evolution of the wildfire occurrence (Fig. 1) and the population rates of the rural areas of Spain, 1985 appears as a critical year: wildfires increased its occurrence, the rural migration flows reversed and the diversification of the rural employment started (Hoggart 1997).

In 2000 the increasing wildfire occurrence trend reached a summit and stabilized till 2006 when the trend reversed into a significantly lower ignition frequency. Among other factors, this evolution might be interpreted in political terms: in 2004, after a catastrophic period, a very strict regulation was passed (RD 11/2005). At the same time, educational programs have been implemented for changing the population attitude (FAO 2005).

All these large-scale socioeconomic changes and related policy-making have changed not only the vulnerability of rural landscapes to wildfire, but their own spatial structures as well. The area covered by forests and other woodlands has constantly and largely increased since the 1950s decade not only due to the abandonment of marginal agricultural lands but also as directly resulting from vast reforestation programs. The large-scale Spanish reforestation plan towards the mid of the XXth century (1935–1978) was followed since the 1990s by the application of series of policies established by the European Union (Common Agricultural Policy). These policies were determined to promote reforestation in abandoned agricultural lands and were very successful in Spain, with about 685,000 ha of agricultural lands converted to forests in the period from 1994 to 2006 (Sociedad Española de Ciencias Forestales 2009).

Increments of forest lands and densification process of dehesas and shrub land use types has been noted also by SISPARES monitoring system. These combined processes have resulted in important changes in the forest-agriculture interface. Generally speaking, the area covered by crops has decreased, as well as the fragmentation of their landscapes (Ortega et al. 2008). Therefore, those types at the extremes of the vulnerability gradient, such as the integrated agroforestry "dehesas" or the intermixed fine grained landscapes, have decreased their geographical distribution being increasingly concentrated at their traditional cores respectively in Extremadura and Galicia. In the case of the "dehesas", the abandonment of grazing practices, development of irrigation programs including reservoirs, conversion into game hunting states, urbanization by the main towns, including golf courses, are described as the most frequent change driving processes (Pérez-Soba et al. 2007). However, since 1984 "dehesa" landscapes still remained dominant in 18 SISPARES samples mainly where cattle, pigs, coal and cork productions make the system profitable (García del Barrio et al. 2004).

Could climate change be key responsible of recent trends in FAI vulnerability? Our conceptual approach to the complex web of FAI driving forces and components

Our results and discussion allow us to establish a conceptual summary aiming to clarify the complex web of causes and interactions existing around the forest agriculture interfaces. The proposed conceptual framework shown in Fig. 8 is largely based on landscape ecology principles (Forman and Godron 1986):

1. FAIs are cultural landscapes; they result from the action of climate, landform, vegetation and human factors.
2. These four factors are fully determinant of the FAI landscape structure and its functions, such as the vulnerability to wildfire.
3. Climate and landform are the predominant natural factors, and their impacts are effective at much longer term than the vegetation and human factors.
4. On their turn, vegetation and human population are directly dependent on climate and landform.
5. Human population, often by using fire, has impact at the shortest term in both the FAI structure and wildfire vulnerability.
6. FAI structure and its functional vulnerability have shown mutual interactions.

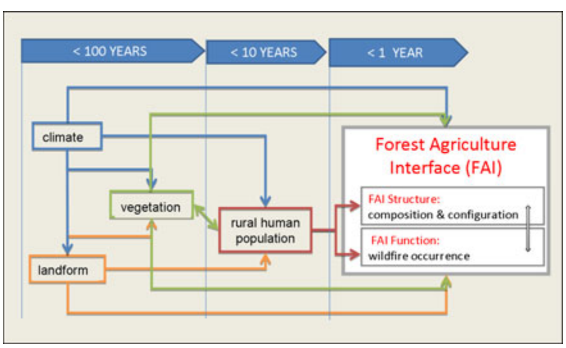

Fig. 8 Flow chart diagram summarizing the conceptual approach to the FAI driving factors, based on our results and other general landscape ecological assumptions (Forman and Godron 1986). The four main factors have different time and spatial scales of expression, as well as levels of relative independence. *Boxes* are driving factors and *arrows* show the direction of the predominant relationship of dependence between factor and FAI components

Such a conceptual approach is helpful to explore whether climate change can be responsible for the detected changes in FAI configuration or wildfire vulnerability. As far as the FAI structural features it concerns, most of the recorded changes have been produced by new human land uses or by the abandonment of inefficient activities rather than by climatic causes, possibly because is too soon for climate to fully express their impact on neither vegetation nor landform. However, in an indirect way, climate change could affect the human action mainly by promoting new meteorological conditions for agricultural extensive crops or forest plantations.

From the functional landscape perspective, only 4.87% of the total number of wildfires in Spain has reportedly been caused by natural hazards, during the 1996–2008 period. Consequently, it is very difficult to establish a direct causal link between the increase of FAI fire frequency and the climate change at national scale. However, there are regional variations from 1.09% in the humid and subhumid areas to 8.70% in the more arid areas. These natural fires are ignited by lightning, mainly located in the eastern Mediterranean regions where the thunderstorms in summertime are frequent. Because of that, some research has been carried out on the impact of meteorological conditions in wildfire occurrence in the regions of Valencia and Catalonia (Pausas 2004; Piñol et al. 1998). These two papers have positively tested the correlation among higher temperatures, lower relative humidity and lower summer rainfall rates with wildfire occurrence. Such findings are consistent with our data since lightings are much more risky for dry thunderstorms.

A deeper study on the implication of the climate change on the vulnerability would require a longer time period and the analysis of information on fire spread and extension, features much more related to vegetation fuel characteristics after changes in the climatic condition.

Monitoring the forest-agriculture interface: importance and requirements for delivering solid information for decision making

The spatial–temporal dimension of the wildfire regime-landscape pattern relationships implies the need for information that is usually not easily available for large study areas like the entire Spain. This is a significant bottleneck for carrying out any landscape ecology research of this kind at a national scale (Turner 1989). In particular, an important requirement on such information is the consistency of the data gathered from all the studied land (spatial consistency) at different periods (temporal consistency). The spatial consistency is being solved thanks to the growing development of remote sensed land information and related data processing technologies (Martínez et al. 2010). However, the temporal consistency is much harder to achieve when assessing trends comprising several decades. This is an important limitation, since information from the past is crucial to support present management decisions and to project changes into the future. In general, the vaster the study area is, the lower past data availability, which hampers the adequate disentangling and understanding of the factors driving wildfire regime and their changes at sufficiently wide spatial scales. Indeed, in order to monitor changing dynamics of wildfire, reduce human risk and damage from wildfire, and maintain region specific natural wildfire regimes, we must first understand historic spatial–temporal patterns of wildfire (Gralewicz 2010). A comprehensive approach including landscape vulnerability assessment, monitoring, planning and finally management has to be developed as a key tool towards this end, particularly in those forest-agricultural mixture mosaics that have been here demonstrated to be the most vulnerable to wildfire occurrence.

At European level, the Council on prevention of forest fires within the European Union (2010) invites the use of monitoring systems. The connections with potential trends under the influence of wildfire and social risk factors also need step up efforts within the EU research programs. In this framework, the usefulness of land use monitoring systems at a national level and the need for their harmonization at European level to maintain biodiversity and mitigate environmental change are recognized (Metzger et al. 2010). However, the connections with wildfire hazard still need to be more specifically addressed.

To achieve the harmonization of monitoring systems, a consistent classification of land into relatively homogeneous environmental strata is needed to provide a robust framework for comparison and analysis of environmental data and sampling ecological resources over large geographic regions (Bunce et al. 1996). Many countries have adapted quantitative environmental stratifications to support environmental management and planning, e.g. Australia, Austria, New Zealand, Senegal, Sweden, Norway, and Spain where the CLATERES classification described earlier has been developed (Elena-Rosselló et al. 1997). The Great Britain Countryside Survey provides one of the best documented examples of a national monitoring scheme designed to assess stock of habitats and vegetation. Other countries with similar monitoring schemes include the Spanish SISPARES systems described above (Elena-Rosselló et al. 2005), Austria, and Sweden. Currently, these stratifications and monitoring systems are being assessed to be integrated at a single system at European level by means of the European Collaborative Project EBONE (Ortega et al. 2011). This background supports the usefulness of CLATERES classification and SISPARES monitoring systems as appropriate databases to analyze the relationship between landscape structure and wildfire occurrence.

Conclusions and final remarks

Our study analyzed the mutual interaction between landscape and wildfires in the forest agriculture interface of Spain and showed that rural landscapes played an important role as spatial structured systems of ecosystems with proved impact in the ignition and eventually in the propagation of wildfires. This is particularly important in Spain and other Mediterranean areas, where most of the cultural landscapes are dominated by mixtures of woodlands and crops that coexist in a variety of composition and configuration patterns. The widespread forest-agriculture interface that can be currently observed throughout Spanish landscapes is the result of millennia of human disturbance, with wildfires being one of the most important drivers and mostly induced by humans to shape landscape conditions. Therefore, under the predominant climate constraints, sociological and cultural factors as well as the susceptibility to wildfire are among the most important drivers for past and recent spatiotemporal pattern.

We have found in Spain a quite varied menu of forest agriculture blends, developed with very different socioeconomic backgrounds somehow dependent from the biophysical land capability and suitability. In fact, we have a vast variety of cultural landscapes, some of which can be considered as real fossils because they do not balance with the current human society's demands. The trends in these patterns and land use mixtures have changed in recent decades and are expected to undergo major changes in the short and mid term, following underlying socioeconomic drivers and triggering significant shifts in the wildfire regime.

Of particular relevance as holders of landscape vulnerability to wildfires were the fine-grained forest-agricultural mixtures, with an ignition frequency four times higher than the pure forested landscapes, and six times higher than the average of the rural landscapes. Other landscape types such as "dehesas" were much less prone to wildfire occurrence and therefore should be advocated as low-vulnerability agroforestry system rather than a heterogeneous and interspersed pattern composed by spatially separated patches with different land uses. The forest-agriculture interface types that we here defined and statistically tested in Spain are of broader implications since they are similarly replicated throughout most of the Mediterranean basin. However, some differences between Spain and these other Mediterranean areas do exist and enhance the interest of our focus in the Spanish agroforestry landscapes. First, Spain shows a very high diversity of environmental conditions and landscape settings because of its large extent and its characteristic biophysical transition from temperate oceanic and alpine until true Mediterranean climate. In addition, Spain has recently experienced dramatic socio economic changes that have impacted notably the rural landscapes where the forest-agricultural interface occurs, unlike other countries in North Africa where population largely remains to rely on grazing and firewood harvesting as it occurred in the Iberian Peninsula some decades ago. Finally, the studied Spanish landscapes present an historical background of human disturbances and landscape modifications that is not so prominent in other Mediterranean countries like California, Chile, South Africa or Australia.

In any case, we believe that the characterization and classification of agroforestry landscapes, here proposed and tested, has broad implications regarding the landscape vulnerability to wildfires and hence has as well relevant consequences applicable to landscape planning and management. From the perspective of long-term wildfire prevention, results from our research should be considered as overall guidelines for land use planning in the most vulnerable rural landscapes. This would provide an important complementary and durable approach for wildfire management based on a passive protection strategy targeted to modify the inherent high susceptibility of the forest agriculture interface to fire ignition events.

Acknowledgements The present work has been partially financed by the European Union Project EBONE (7FWP, ENV.2007.4.1.1.2.) and the Spanish project DECOFOR (AGR2009-07140). We thank the Spanish Ministry of Environment (Dirección General del Medio Natural y Política Forestal) for providing wildfire ignition data (EGIF). Many thanks also to the two anonymous reviewers who improved earlier drafts of the manuscript.

References

Agee JK, Skinner CN (2005) Basic principles of forest fuel reduction treatments. For Ecol Manag 211:83–96

Alexandrian D, Esnault F, Calabri G (1998) Forest fires in the Mediterranean area. FAO Corporate Document Repository, http://www.fao.org/docrep/x1880s/x1880s07.htm

Ayuda MI, Collantes F, Pinilla V (2010) From locational fundamentals to increasing returns: The spatial concentration of population in Spain, 1787–2000. J Geograph Syst 12(1):25–50

Badia A, Pallares M (2006) Spatial distribution of ignitions in Mediterranean periurban and rural areas: the case of Catalonia. Int J Wildland Fire 15:187–196

Bessie WC, Johnson EA (1995) The relative importance of fuels and weather on fire behavior in subalpine forests. Ecology 76(3):747–762

Bond WJ, Woodward FI, Midgley GF (2004) The global distribution of ecosystems in a world without fire. New Pytologist 165:525–538

Bunce RGH, Barr CJ, Clarke RT, Howard DC, Lane AMJ (1996) The ITE meriewood land classification of Great Britain. J Biogeogr 23:625–634

Campos P, Caparrós A, Cerdá E, Huntsinger L, Standiford R (2007) Modeling multifunctional agroforestry systems with environmental values: Dehesa in Spain and woodland ranches in california handbook of operations research. Nat Resour Int Ser Oper Res Manag Sci 99(1):33–52

Catry FX, Rego FC, Bação FL, Moreira F (2009) Modelling and mapping wildfire ignition risk in Portugal. Int J Wildland Fire 18(8):921–931

Chas-Amill ML, Touza J, Prestemon JP (2010) Spatial distribution of human-caused forest fires in Galicia (NW Spain). In: WIT transactions on ecology and the environment, vol 137. WIT Press, Southampton, www.witpress.com. ISSN 1743-3541 (online) doi:10.2495/FIVAI00221

Chuvieco E, Aguado I, Yebra M, Nieto H, Salas J, Martín MP, Vilar L, Martínez FJ, Martín S, Ibarra P, De la Riva J, Baeza J, Rodríguez F, Molina J, Herrera MA, Zamora R (2010) Development of a framework for fire risk assessment using remote sensing and geographic information system technologies. Ecol Model 221:46–58

Council of the European Union (2010) Council conclusions on prevention of forest fires within the European Union. In: 3010th general affairs council meeting. Luxembourg

Di Castri F, Mooney HA (eds) (1973) Mediterranean type ecosystems. Origin and structure. ecological studies 7. Springer, Berlin

EEA (1995) CORINE land cover. Part II. Nomenclature. Europeam Environmental Agency

EGIF (2009) Los incendios forestales en España. Technical Report of "Area de Defensa Contra Incendios Forestales" of the Spanish Ministry of Environment

Elena-Rosselló R, Tella G, Castejón M (1997) Clasificación biogeoclimática de España peninsular y balear. Ministerio de Agricultura pesca y Alimentación, Madrid, Spain

Elena-Rosselló R, Bolaños F, Gómez V, González S, Ortega M, García del Barrio JM (2005) The SISPARES (Spanish Rural Landscape Monitoring System) experience. Proceedings of European IALE conference. Landscape ecology in the Mediterranean: inside and outside approaches. Faro, Portugal

FAO (2005) Community based fire management in Spain. Forest protection working papers, working paper FFM/4/E. Forest Resources Development Service, Forest Resources Division, FAO, Rome Italy Forestry Department. Based on the work of Mr Ricardo Velez Ministry of Environment Madrid, Spain

Finney MA (2001) Design of regular landscape fuel treatment patterns for modifying fire growth and behavior. Forest Science 47:219–229

Flannigan MD, Harrington JB (1988) A study of the relation of meteorological variables to monthly provincial area burned by wildfire in Canada (1953–80). J Appl Meteorol 27:441–452

Forman RTT (1995) Land mosaics: the ecology of landscapes and regions. Cambridge University Press, Cambridge, UK

Forman RTT, Godron M (1986) Landscape ecology. Wiley, New York

García del Barrio JMG, Bolaños F, Ortega M, Elena-Rosselló R (2004) Dynamics of land use and land cover change in Dehesa Landscapes of the REDPARES network between 1956 and 1998. In: Schnabel S, Ferreira A (eds) Advances in geoecology: sustainability of agrosilvopastoral systems—Dehesa, Montados, vol 37. Catena GmBh, Reiskirchen, Germany, pp 47–54

Gralewicz, NJ (2010) Spatial and temporal patterns of wildfire occurrence and susceptibility in Canada. Electronic thesis and dissertations. Department of Geography, University of Victoria, Canadá

Hely C, Fortin MJ, Anderson KR, Bergeron Y (2010) Landscape composition influences local pattern of fire size in the eastern Canadian boreal forest: role of weather and landscape mosaic on fire size distribution in mixedwood boreal forest using the prescribed fire analysis system. Int J Wildland Fire 19(8):1099–1109

Hijmans RJ, Cameron SE, Parra JL, Jones PG, Jarvis A (2005) Very high resolution interpolated climate surfaces for global land areas. Int J Climatol 25:1965–1978

Hoggart K (1997) Rural migration and counter urbanization in the European periphery: The case of Andalucía. Sociologia Ruralis. Wiley Online Library

Honnay O, Piessens K, Van landuyt W, Hermy M, Gulinck H (2003) Satellite based land use and landscape complexity indices as predictors for regional plant species diversity. Landsc Urban Plan 63:241–250

Houérou HN (1993) Land degradation in Mediterranean Europe: can agroforestry be a part of the solution? A prospective review. Agrofor Syst 21(1):43–61

INE (2008) Anuario Estadístico de España. Instituto Nacional de Estadística

Jongman RHG, Bunce RGH (2008) Farmland features in the European Union: a description and pilot inventory of their distribution. Report 31–08-2008. Alterra, Wageningen, UR

Krumel JR, Gardner RH, Sugihara G, O'Neill RV (1987) Landscape patterns in a disturbed environment. Oikos 48:321–324

Malamud BD, Millington JDA, Perry GLW (2005) Characterizing wildfire regimes in the United States. PNAS 102(13): 4694–4699

MARM (1996–2005, 2006, 2007, 2008) Los incendios Forestales de España. Technical Reports of Dirección General de Medio Natural y Política Territorial del Ministerio de Medio Ambiente y Medio Rural y Marino. http://www.marm.es/es/biodiversidad/temas/defensa-contra-incendios-forestales/estadisticas-de-incendios-forestales/default.aspx

Martínez S, Ramil P, Chuvieco E (2010) Monitoring loss of biodiversity in cultural landscapes. New Methodol Satel Landsc Urban Plan 94:127–140

Mazzoleni S, di Pasquale G, Mulligan M, di Martino P, Rego F (eds) (2005) Recent dynamics of the Mediterranean vegetation and landscape. Wiley, London

McGarigal K, Marks BJ (1995) FRAGSTATS: spatial pattern analysis program for quantifying landscape structure. USDA For Serv Gen Tech Rep PNW-351

Metzger MJ, Bunce RGH, van Eupen M, Mirtl M (2010) An assessment of long term ecosystem research activities across European socio-ecological gradients. J Environ Manag 91:1357–1365

Moreno J, Viedma O, Zavala G, Luna B (2011) Landscape variables influencing forest fires in Central Spain. Int J Wildland Fire (in press)

Naveh Z (1975) The evolutionary significance of fire in the Mediterranean region. Vegetatio 29(3):199–208

Ortega M, Bunce RGH, García del Barrio JM, Elena-Rosselló R (2008) The relative dependence of Spanish landscape pattern on environmental and geographical variables over time. Investigación Agraria: Sistemas y Recursos Forestales 17(2):114–129

Ortega M, Metzger M.J, Bunce RGH, Wrbka T, Allard A, Jongman RHG, Elena-Rosselló R (2011) The potential for integration of environmental data from regional stratifications into an European monitoring framework. J Environ Plan Manag (in press)

Padilla M, Vega-García C (2011) On the comparative importance of fire danger rating indices and their integration with spatial and temporal variables for predicting daily human-caused fire occurrences in Spain. Int J Wildland Fire 20(1):46–58

Pausas JG (2004) Changes in fire and climate in the eastern Iberian Peninsula (Mediterranean basin). Climatic Change 63:337–350

Pérez-Soba M, San Miguel Ayanz A, Elena-Rosselló R (2007) Complexity in the simplicity: the Spanish dehesas: the secret of an ancient cultural landscape with high nature value still functioning in the 21st century. In: Pedroli B, van Doorn A, de Blust G, Paracchini ML, Wascher D, Bunce F (eds) Europe's living landscapes. Essays exploring our identity in the countryside. KNNV Publishing, The Netherlands, in cooperation with Landscape Europe, pp 369–387

Pickett STA, White PS (1985) Patch dynamics: a synthesis. In: Pickett STA, White PS (eds) The ecology of natural disturbance and patch dynamics. Academic Press, New York, pp 371–384

Pineda FD (2001) Intensification, rural abandonment and nature conservation in Spain. In: Bunce RGH, Pérez-Soba M, Elbersen BS, Prados MJ, Andersen E., Bell M. Smeets PJAM (eds) Example of European agro-environment schemes and livestocks systems and their influence on Spanish cultural landscapes. Proceedings of a European workshop, Soto del Real, 13–15 July 2000, Wageningen, Alterra-Rapport 309, pp 23–46

Piñol J, Terradas J, Lloret F (1998) Climate warming, wildfire hazard, and wildfire occurrence in coastal eastern Spain. Clim Change 38:345–357

Pinto-Correia T (2000) Future development in portuguese rural areas: how to manage agricultural support for landscape conservation? Landsc Urban Plan 50:95–106

Rey Benayas JM, Martins A, Nicolau JM, Schulz JJ (2007) Abandonment of agricultural land: an overview of drivers and consequences. CAB reviews: perspectives in agriculture, veterinary science, nutrition and natural resources 2, No. 057

Riitters KH, O'Neill RV, Hunsaker CT, Wickham JD, Yakee DH, Timmins SP, Jones KB, Jackson BL (1995) A factor analysis of landscape pattern and structure metrics. Landscape Ecol 10(1):23–39

Romero-Calcerrada R, Novillo CJ, Millington JDA, Gomez-Jimenez I (2008) GIS analysis of spatial patterns of human-caused wildfire ignition risk in the SW of Madrid (Central Spain). Landscape Ecol 23:341–354

Romme WH (1982) Fire and landscape diversity in subalpine forests of Yellowstone National Park. Ecol Monogr 52: 199–221

Scheller RM, Mladenoff DJ (2004) A forest growth and biomass module for a landscape simulation model, LANDIS: design, validation, and application. Ecol Model 180:211–229

Schmidt D, Taylor AH, Skinner CN (2008) The influence of fuels treatment and landscape arrangement on simulated fire behavior, Southern Cascade range, California. For Ecol Manag 255:3170–3184

Scott JH, Burgan RE (2005) Standard fire behavior fuel models: a comprehensive set for use with Rothermel's surface fire spread model. Gen Tech Rep RMRS-GTR-153. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO

Sociedad Española de Ciencias Forestales (2009) Situación de los bosques y del Sector forestal en España. V Congreso Forestal de España, Ávila

Stocks BJ (2002) Large forest fires in Canada, 1959–1997. Journal of Geophysical Research- Atmospheres 108:511–512

Trabaud L (1980) Impact biologique et écologique des feux de végétation surl' organisation, la structure et l' evolution de la végétation des garrigues du Bas-Languedoc. Thése d'Etat Univ Sci Tech Languedoc, Montpellier

Trabaud LV, Cristensen NL, Gill AM (1993) Historical biogeography of fire in temperate and Mediterranean ecosystems. In: Crutzen PJ, Goldammer JG (eds) Fire in the environment: The ecological atmospheric and climatic importance of vegetation fires. Wiley, Chichester, UK, pp 277–295

Turner MG (1989) Landscape ecology: the effect of pattern on process. Annu Rev Ecol Syst 20:171–197

Turner MG, Hargrove WH, Gardner RH, Romme WH (1994) Effects of fire on landscape heterogeneity in Yellowstone National Park, Wyoming. J Veg Sci 5:731–742

Vázquez A, Moreno JM (1998) Patterns of lightning and people-caused fires in Peninsular Spain. Int J Wildland Fire 8(2):103–115

Vázquez A, García del Barrio JM, Ortega M, Sánchez-Palomares O (2006) Recent fire regime in peninsular Spain in relation to forest potential productivity and population density. Int J Wildland Fire 15:397–405

Vilar L, Woolford DG, Martell DL, Martín P (2010) A model for predicting human-caused wildfire occurrence in the region of Madrid, Spain. Int J Wildland Fire 19(3):325–337