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Interactions between land use/land cover change, forest fires and landscape structure in Sierra de Gredos (Central Spain)

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SUMMARY

In fire-prone areas, like the Mediterranean, land abandonment and forestation may interact with fire to alter landscape properties and eventually fire hazard and occurrence. However, the spatial interactions among the two processes (land-use/land cover change [LULC] and fire) are poorly known. Here, we analysed the relative effect of LULC change and fire on the landscape structure of an area of Central Spain frequently affected by fire. A series of Landsat MSS images from 1975 to 1990 was analysed to quantify annual changes in LULC, map fire perimeters and evaluate the changes in landscape properties. The temporal dynamics were analysed by annually computing the fraction occupied by each LULC type and landscape structural properties (number, size, shape and arrangement of patches) that might play a role in fire propagation. All of these were calculated separately for the unburned or the burned areas during the study period, as well as for the entire area. At the whole landscape level, or in the unburned area, LULC changes were small, yet the two more flammable LULC types tended to increase, and the landscape tended to become more homogeneous. In the burned area, the area covered by pine woodlands tended to decrease, and that covered by shrublands to increase. Burned areas turned into shrublands only five years after fire. Landscape indices indicative of reduced fragmentation were also found. Both LULC change and fire altered landscape patterns in the whole area to create a less fragmented and more contiguous landscape than in 1975. The changes induced in the whole landscape by fire, in spite of the overall low disturbance rate, were sufficient to closely determine the changes in landscape composition (LULC types) and patterns.

Keywords: forest fires, land-use change, Landsat MSS, landscape patterns, stationary state

INTRODUCTION

Land-use/land cover (LULC) change is one of the main drivers of global change (Vitousek *et al.* 1997), affects landscape structural patterns and, therefore, has the potential to alter disturbance regimes. In many areas of the world, fires are now occurring in landscapes that have been, or are being, subjected to pronounced LULC change. In developing countries, increasing human pressure is leading to deforestation, intensification of land-use and landscape fragmentation (Cochrane 2001; Siegert *et al.* 2001). In developed countries, such as the Mediterranean countries of the European Union, the reverse trend predominates: changes in LULC during the last decades have been characterized by depopulation of rural areas, land abandonment and forestation (Fernández Alés *et al.* 1992; García-Ruiz *et al.* 1996; Debussche *et al.* 1999; Peroni *et al.* 2000; Moreira *et al.* 2001). These LULC changes have favoured the expansion of shrublands and other flammable vegetation types, if only because afforestation with conifers was dominant. In parallel with these changes, fires have become more frequent and widespread (Moreno *et al.* 1998).

In countries affected by LULC change and fire, it is interesting to know how each of these processes affects landscape structure and, ultimately, whether such changes play a role in modifying fire occurrence and regime. There is evidence that changes in landscape structure have increased landscape fire hazard and thus may have played a role in increasing fire incidence in countries around the Mediterranean (Moreira *et al.* 2001; Mouillot *et al.* 2003) and elsewhere (Cochrane 2001; Johnson *et al.* 2001; Keeley & Fotheringham 2001).

The roles played by LULC change and fire in landscape spatial patterns depend, among other, on the rate and type of LULC change, properties of fires (frequency, size and intensity), their spatial autocorrelation (Moloney & Levin 1996; Turner *et al.* 1997) and the dynamics of the vegetation after fire. In the Mediterranean, natural vegetation change in the absence of fire is rather slow (Kadmon & Harari-Kremer 1999; Lavorel & Garnier 2002); thus, landscape changes as a result of abandonment may take more than 1–2 decades. That might not be the case when forestation is involved because after 1–2 decades forested areas will be very similar in vegetation structure, thus having the potential to quickly affect landscape patterns (Barbero *et al.* 1998; Bonet 2004).

Fires may induce rapid vegetation changes by transforming burned areas into the same state of recovery. It is possible that not all the area affected by a fire may regenerate equally, for reasons that include variations in the type of vegetation burned, fire severity, or past land-use and fire history. However, plot-type studies in the Mediterranean

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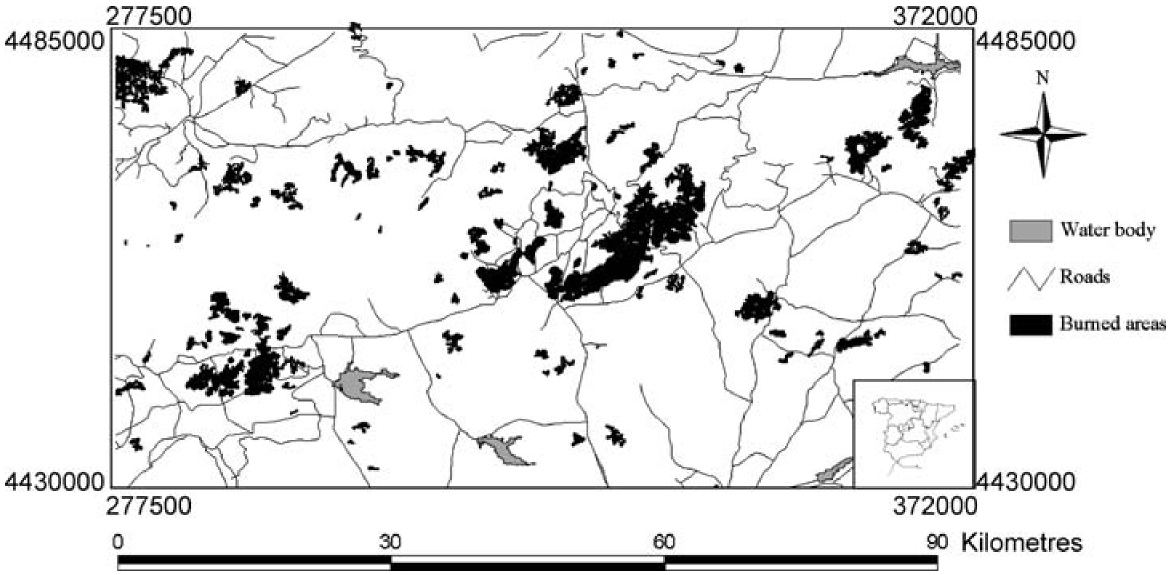

Figure 1 Location of the study site at Sierra de Gredos (Central Spain) and map of the 121 fires that occurred between 1975 and 1990.

indicate that the vegetation that regenerates after fire may be quite insensitive to these variations, and it quickly converges in composition and structure, irrespective of the type of vegetation burned (Faraco *et al.* 1993), fire severity (Pérez & Moreno 1998), land-use history (Pérez *et al.* 2003) or fire history (Pérez 1997). It is important to ascertain how various LULC types respond to fire at the level of burned areas and whole landscape. Knowledge of the effects of the parallel interplay of LULC change and fire occurrence on landscape spatial patterns is poor, particularly in view of the possible consequences for landscape fire-hazard and, ultimately, fire occurrence. Because of the transitory nature of many of these changes, studies over many years are needed, rather than mere comparisons between two end-point years of a given period, to gain perspective on the underlying temporal trends. Yet, such research is scarce, particularly in Mediterranean landscapes.

This work investigates the effects of LULC change and fire on landscape characteristics of a large area of Central Spain. During the second half of 20th century, this area was subject to a high incidence of forest fires, forestation and land abandonment. The objectives were to: map the fire perimeters of fires that occurred between 1975 and 1990 in the area; annually assess the changes that occurred in landscape composition and structure as a result of LULC and fire; evaluate whether landscape characteristics and trends were different among burned and unburned areas; and determine the effect of fire through time on the landscape characteristics of burned areas, particularly those that would more closely affect fire hazard properties, and in relation to the main LULC types. Overall, we were interested in finding the extent to which the combination of LULC and fire altered landscape properties, particularly those that ultimately may affect fire occurrence.

METHODS

Study site

The study area (518 331 ha) is located in Sierra de Gredos (province of Avila, Spain); UTM coordinates 4430–4485 and 277–372 in the zone 30 North (Fig. 1). The area is mountainous, with altitudes up to 2200 m, and a strong gradient from the south-east to the north-west. The majority of the territory is 400–600 m altitude, with $\leq 10\%$ inclination. Plutonic rocks, mostly granites and, to a lesser extent, metamorphic rocks at lower elevations, form the substrate. The soils are deep, siliceous, moderately acidic and quite fertile (Gallardo *et al.* 1980). The potential vegetation of the area corresponds mainly to four different types (Rivas-Martínez 1987): sclerophyllous oak forest (*Pyro bourgaeanae-Quercetum rotundifoliae*) from the lowest altitudes up to 600 m; humid deciduous oak forest (*Arbuto unedonis-Quercetum pyrenaicae*) at 600–800 m; sub-humid deciduous oak forest (*Luzulo forsteri-Quercetum pyrenaicae*) at 800–1600 m and woody-legume shrublands (*Cytiso purganti-Echinospartetum barnadessi*) above 1600 m. Despite the differences in temperature and precipitation indicated by these vegetation units, pastures, shrublands and managed pine woodlands are abundant in the landscape. At lower and middle altitudes, *Pinus pinaster* Aiton is the main species, whereas at higher elevations *Pinus sylvestris* L. is dominant (Blanco *et al.* 2005). Forestation started in the early 1960. Between the first (1976) and the second (1986) National Forest Inventories, the area under conifers increased by 65% in Avila province and croplands decreased by 23%, whereas the area occupied by pastures and shrublands grew by 26% during 1982–1989 (National Institute of Statistics 1982, 1989).

Image acquisition and pre-processing

Landsat Multispectral Scanner (MSS) images were used owing to their availability for the period from July 1972 to October 1992, which coincided with important socioeconomic changes and high fire occurrence in the area. MSS image data consists of four spectral bands, two from the visible region (green MSS 1 and red MSS 2), and two from the near-infrared region (MSS 3 and 4) with a spatial resolution of 79 m (1 pixel). The MSS images used were from the satellites Landsat 1–5. Each annual image was chosen during summer to autumn, according to the availability of images without clouds.

The images were geometrically and radiometrically corrected, and georeferenced to topographic maps at 1:50 000 scale with an error of less than 1 pixel (0.5–0.7 pixels; ENVI [Environment for Visualizing Images] 1999). All images were converted to apparent reflectances and an empirical atmospheric normalization was carried out to minimize the effects of the different atmospheric conditions among them (ENVI 1999). To avoid the radiometric effects of relief, a non-Lambertian model was applied to separate the signal owing to the ground cover from noise attributable to terrain variations (García-Haro *et al.* 2001). To accomplish this correction, a digital terrain model (DTM) with lines at 100 m (Vázquez & Moreno 2001) was resampled at the same spatial resolution as the MSS images (79 m).

Fire mapping

Fires with perimeters ≥ 10 ha that occurred during the period 1975–1990 were mapped using the Normalized Difference Vegetation Index (NDVI $=$ (MSS4 $-$ MSS2)/(MSS4 $+$ MSS2)), which has proven useful for discriminating areas with little or no vegetation cover from those with green and healthy vegetation. The fire maps were derived from three NDVI images corresponding to the pre-fire year and the two corresponding to the post-fire year, although in some years, the discontinuity of the temporal series forced us to compare the nearest (pre-fire and post-fire) images available. The difference between the post-fire NDVI and pre-fire NDVI images allowed us to identify areas that lost NDVI and that were likely to have been burned. We also considered the NDVI difference between the first and second post-fire years to quantify the regeneration of vegetation. A multitemporal unsupervised classification using the various images was carried out to identify the burned lands of each year.

To verify the fire-mapping procedure, maps of the perimeters of fires that occurred in the central portion of the study area were reconstructed using aerial photographs available for the period 1970–1990 (Vázquez & Moreno 1998) (Fig. 1). Annual statistics recorded by the Forest Service (ICONA) within 10×10 km^2 quadrats were also available for the period 1975–1990.

Annual LULC classifications

The LULC classifications used by the CORINE programme (EEA [European Environment Agency] 1990) were used to classify the 13 Landsat MSS images into: deciduous woodland, pine woodland, shrubland, pasture, *dehesa* (pasture with sparse trees), cropland, bare soil, urban area and water body. Annual unsupervised classifications were combined with a knowledge-base that used 'temporal context' dependences and topographic stratification of the main LULC types to relate spectral classes with real land cover types (Liu *et al.* 2002). We used the CORINE land-cover map for the year 1987 and a digital terrain model (DTM) with altitude classes separated by 200 m to construct the knowledge base. The accuracy of the annual classifications was assessed using the Kappa Index of Agreement (KIA) of each LULC spectral class with the CORINE LULC types of 1987, comparing the spatial coincidence of each CORINE LULC type with the annually-classified LULC classes.

The unsupervised classification of each annual MSS image was carried out using the K-means algorithm (ENVI 1999) on the five following bands: NDVI, first and second principal components applied to all MSS bands, and the soil brightness and greenness bands from the tasselled cap transformation. We extracted 20 spectral classes from each annual image to obtain their greatest spectral variability.

Knowledge of temporal neighbourhood (context) assisted the deduction of causal and spatial relationships. Furthermore, classification of single images was always subject to inevitable errors that were overcome through the analysis of sequential images. We compared the LULC map from CORINE (1987) with the nearest clustered MSS image (1986) by means of a GIS (geographic information system) (see Supplementary material at http://www.ncl.ac.uk/icef/EC_Supplement.htm, Fig. S1). A table with the area occupied by each LULC type (A_i) in each spectral class (I_i) was obtained and, through a dominance decision rule, any spectral class (I_i) was classified as any LULC type (A_i) when this was spatially dominant (LULC occupation greater than 60% in the class I_i). When the spatial dominance criterion was not met then decision rules based on altitudinal distributions were applied (see Supplementary material at http://www.ncl.ac.uk/icef/EC_Supplement.htm, Fig. S1). First, we crossed the real LULC types contained in each spectral class (I_i) with the DTM (altitude ranges) to ascertain the topographic position of the main LULC types included. When LULC types did not overlap in their dominant altitude ranges, the class (I_i) was identified by the LULC types located in such non-overlapping altitude ranges (see Supplementary material at http://www.ncl.ac.uk/icef/EC_Supplement.htm, Fig. S1). If two or more LULC types overlapped in their altitude distribution, then a Spatial Dominance Index (S $(i \cdot j)$) was applied to identify the relative dominance of each LULC type (A_i) with respect to another in such elevation ranges as:

$$S(i \cdot j) = (100 \times A_{i,l,h}/A_{j,l,h}) - 100 \qquad (1)$$

In this sense, if the $S(i \cdot j)$ of the LULC type (A_i) in spectral class (I_i) at the altitude range (h) was greater than $+50\%$ in relation to the other LULC type, then the spectral class (I_i) was identified as the LULC type (A_i) in that range. When the MSS image of 1986 was correctly classified according to the CORINE map (with a Kappa index of 0.84), it served as a reference image for the classification of the next nearest images ($t-1$ and $t+1$) following the same decision rules of spatial dominance, topographical distribution and topographical dominance.

The overall accuracy of the annual LULC classifications was high (overall Kappa $= 0.82 \pm 0.09$). The most dynamic LULC types were bare soil, cropland, pasture and pine woodland (Kappa index 0.6–0.8). In spite of the seasonal variability among image dates, no significant relationships were found between the annual KIA values of the main LULC types and day of year (seasonality), rainfall conditions or temporal distance from the reference image. Furthermore, the role of the reference image (CORINE map) over the annually classified LULC types was stable over time, indicating the lack of bias in the temporal context classification. The use of decision rules based on spatial dominance and environmental distribution of main LULC types reduces classification errors associated with radiometrical variations in the main LULC types induced by phenology and the effect of using a single reference map in the classification process, enhancing year-to-year changes in LULC.

Changes in LULC and landscape metrics

We undertook the temporal analysis of LULC changes by annually calculating the percentage occupation of the main LULC types for the whole area, or for the burned and unburned areas separately. Landscape structural changes for the whole area, the area that was burned and that which remained unburned were assessed by computing the landscape indices patch density (PD), mean core area of patches (MCA), contagion (CON) and Shannon's Diversity Index (SHDI) for each year during the 1975–1990 period using FRAGSTATS software (MacGarigal & Marks 1995). As absolute values were not comparable owing to the differing sizes and shapes of each area, results were expressed relative to the values of the initial year of study for each respective area.

At the level of each individual fire, the post-fire dynamics of landscape characteristics were computed for a set of 32 fires that occurred early on in the study period. For each fire, we calculated various landscape metrics (PD, MCA, SHDI, and Edge Density [ED]) for the year preceding the fire and those thereafter up to the eighth year after the fire. These metrics were summarized by means of a principal component analysis (PCA). The first axis explained near 64% of total variance and was interpreted as an 'index of spatial heterogeneity' with high positive coefficients for SHDI and ED. We used these scores to evaluate the relationship between the structural changes induced by fire and the recovery dynamic of the burned areas through time. Calculations were made for all the fires combined or separated by dominant land-cover type (pine woodland or shrubland that covered at the time of fire at least 60% of the area).

The effects of LULC changes and fire on the landscape dynamics of the whole area were assessed by partial regression models relating the area occupied by the main LULC types and the indices of landscape structure (PD, MCA, SHDI and CON), to the percentage area occupied by the various LULC types in the unburned and burned areas, as well as the annual burned area (immediate fire effects) and the area occupied by different early-successional ($\leq$ four years after fire) and mid-successional stages ($\geq$ five years after fire). Prior to any statistical analysis, the percentage area occupied was arcsine transformed. The main explanatory variables selected in each partial regression model were included in the final model using the stepwise method of variable selection.

RESULTS

During the study period (1975–1990), 121 fires were recorded, which swept across a total area of 29 144 ha ($\approx 8\%$ of the forest area). On average, there were nine fires per year and annual burning rate was 0.7% (see Supplementary material at http://www.ncl.ac.uk/icef/EC_Supplement.htm, Fig. S2*a*). Fire size ranged between 13 ha and 5008 ha, with the most frequent fires being between 50 and 200 ha (see Supplementary material at http://www.ncl.ac.uk/icef/EC_Supplement.htm, Fig. S2*b*). Our mapping procedure produced a mean of 81% (± 4.4 SD) for the annual burned area and 89.5% (± 4.9 SD) for the annual fire number as compared to the fire maps based on aerial photography. The accuracy with regards to the Forest Service fire statistics was 86% $\pm$ 2.3 and 75% $\pm$ 1.5 for annual burned area and number of fires, respectively. Fires were concentrated on the southern slopes of the Sierra, with an apparent high degree of contagion (Fig. 1).

The whole study area was dominated by shrublands and pastures embedded in a mosaic of dehesas and croplands, with pine-covered areas being relatively scarce (see Supplementary material at http://www.ncl.ac.uk/icef/EC_Supplement.htm, Fig. S3). There were marked differences among the areas that burned or did not burn. The areas that were swept by fire were mainly covered by shrublands and pine woodlands. In the unburned areas, pastures, dehesas and crops were more frequent (see Supplementary material at http://www.ncl.ac.uk/icef/EC _Supplement.htm, Fig. S3).

In spite of the apparent stability in area covered by the various LULC types between 1975 and 1990, there were some differences in the trends of change in the whole area, the area that remained unburned and the area burned over the time studied. In unburned areas, pine woodlands tended to increase their cover at the expense of shrublands and bare soil (Fig. 2*a*, *b*, *e*; Table 1), and pastures grew at the expense of croplands (Fig. 2*c*, *d*; Table 1). In contrast, in the burned areas, pine woodlands tended to disappear whereas shrublands tended to increase in close relationship with the area occupied by early and mid-successional stages,

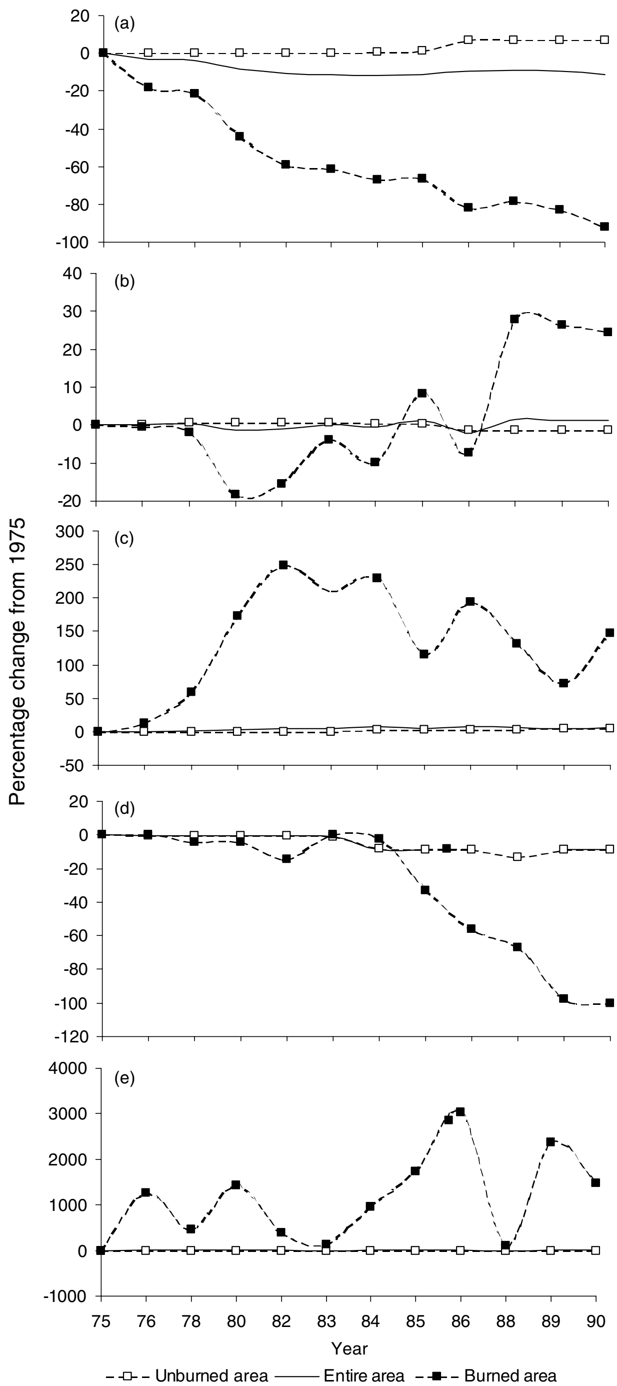

Figure 2 Percentage change in area during the period 1975–1990 (not all years were available) in Sierra de Gredos, Spain, with respect to the initial year of each LULC class, for either the whole area or the areas burned or not burned. (*a*) Pine woodlands, (*b*) shrublands (*c*) pastures (*d*) croplands and (*e*) bare soils.

respectively. (Fig. 2*a*, *b*; Table 1). Burned pastures increased along with the area occupied by early-successional stage and decreased as shrublands became spatially dominant (Fig. 2*b*, *c*; Table 1). The bare soil cover varied linearly with annual burned area (Fig. 2*e*; Table 1). Over the entire area, the occupation of the various LULC types was closely related to fire. Pine woodlands cover tended to decrease as a result of burning pine woodlands, whereas pastures increased at the expense of both burned pine woodlands and shrublands (Fig. 2*a–c*; Table 2). Croplands over the whole area followed the same trend of abandonment as unburned croplands, being replaced by pastures (Fig. 2*c*, *d*; Table 2). Bare soil cover showed a close relationship with the variable dynamics of annual burned area (immediate fire effects) (Fig. 2*e*; Table 2).

The dynamics of LULC types after fire indicated that recovery was rather quick, and after a short period (less than five years) in which vegetation compatible with herbaceous composition and structures was present, burned areas appeared dominated by shrublands, irrespective of whether they were burned shrublands or pine woodlands (Fig. 3).

Landscape structural changes

Between 1975 and 1990, net changes in the variables characterizing landscape patterns in the whole area and in the unburned area were small, but this was not the case in the burned area (Fig. 4*a–d*). In the whole and unburned areas there were nevertheless trends that could be explained by the interacting effects of LULC change and fire. In unburned areas, the landscape tended to become less fragmented (PD decreased and MCA increased), more contagious (CON increased) and compositionally more homogeneous (SHDI decreased) as result of cropland abandonment (thereby increasing pastures) and forestation with conifers (mainly in shrublands) (Fig. 4*a–d*; Table 1). In the burned areas, similar but more intense dynamics were observed. The burned landscape became less fragmented, more contagious and compositionally more homogeneous as the areas at mid-succession stages dominated (Fig. 4*a–d*; Table 1). Over the whole area, the concomitant effects of LULC change and fire explained the temporal dynamics of the main landscape features. The entire landscape became less fragmented (PD and MCA) mainly because of the increase in pastures (derived from croplands) and the increase in pine woodland (Fig. 4*a*, *b*; Table 2). Contagiousness (CON) increased through time as the areas in mid-succession stages became spatially dominant and the landscape became compositionally more homogeneous (SHDI decreased) from cropland abandonment and the mid-term fire effects (Fig. 4*c*, *d*; Table 2).

Landscape heterogeneity increased during the very first years after fire, either in areas burned as shrublands or pine woodlands (Fig. 5). This was because of the increasing fragmentation of pasture, shrubland and bare soil LULC types that developed in these areas (not shown). As time went by, heterogeneity tended to decrease, exhibiting values well below those before the fire, and the landscape of the burned areas became more homogeneous, owing to a contagious spatial dominance of shrublands. This lasted until the fifth year after fire, when a return to pre-fire patterns was observed

Table 1 Multivariate, step-wise linear regression models of LULC type and landscape indices (PD = patch density; MCA = mean core area; CON = contagion; SHDI = Shannon's diversity index) through time in unburned and burned areas as a function of the various LULC types and, in the burned areas, regeneration stages. Shown are the significant variables entering into the model, their coefficient, and the R^2 and significance of the model (* = $0.05 \geq p > 0.01$; ** = $0.01 > p \geq 0.001$; *** = $0.001 > p$).

LULC type	*Unburned area*	*Burned area*
Pine woodlands	Bare soil (−0.9) $R^2 = 0.98^{***}$	Early-succession (< 5 years old) (−0.9) $R^2 = 0.79^{***}$
Shrublands	Pine woodlands (−0.5) $R^2 = 0.98^{***}$	Mid-succession (> 5 years old) (+0.4) Pastures (−0.5) $R^2 = 0.81^{***}$
Pastures	Croplands (−0.6) $R^2 = 0.87^{***}$	Early-succession (< 5 years old) (+0.8) Shrublands (−0.9) $R^2 = 0.91^{***}$
Croplands	Pastures (−1.4) $R^2 = 0.87^{***}$	Mid-succession (> 5 years old) (−0.1) $R^2 = 0.62^{**}$
Bare soil	Pine woodlands (−1.1) $R^2 = 0.99^{***}$	Immediate-succession (0 years old) (+0.6) $R^2 = 0.78^{***}$
Landscape indices		
PD	Croplands (+6.8) Shrublands (+15.3) $R^2 = 0.99^{***}$	Mid-succession (> 5 years old) (−0.2) $R^2 = 0.77^{***}$
MCA	Croplands (+106.6) Shrublands (−238.8) $R^2 = 0.99^{***}$	Mid-succession (> 5 years old) (+2.4) Immediate-succession (0 years old) (+2.8) $R^2 = 0.76^{***}$
CON	Croplands (−28.9) Pine woodlands (+26.5) $R^2 = 0.99^{***}$	Mid-succession (> 5 years old) (+13.1) $R^2 = 0.36^{*}$
SHDI	Croplands (+0.6) $R^2 = 0.99^{***}$	Mid-succession (< 5 years old) (−0.5) Immediate-succession (0 years old) (+0.6) $R^2 = 0.80^{***}$

(Fig. 5). Nevertheless, there were some differences among fires, depending on the dominant type of vegetation burned. The impact of the fire pulse on the spatial patterns of pine woodlands was less intense than that on shrublands. But after this pulse, shrublands tended to return to the characteristics possessed before the fire in a more predictable way, even if by the end of the study period they remained less heterogeneous (Fig. 5). In contrast, the recovery process of areas covered by pine woodlands produced more heterogeneous landscapes than those before the fire (Fig. 5). This was more the result of greater variability among the areas burned as pine woodlands than of differences in structure of the vegetation, since the relationship between the index of heterogeneity and cover by shrubs was very similar for both types of land cover (Fig. 6*a–e*).

DISCUSSION

The landscape of the Sierra de Gredos experienced an overall net change in land cover types during the 16-year study period that, averaged over the entire area studied, was relatively small. Projecting the undisturbed area into the next century by means of a Markov-type simulation, and assuming stability in the processes observed during this period (LULC change and fire) and that past conditions remained stable, the overall net change in this landscape would be limited (not shown). Nevertheless, future projections must take into account possible changes in climate and fire-danger conditions; this was not done here. Calculations incorporating various greenhouse gas emission scenarios show that fire danger may increase several-fold during the course of this century (Moreno 2005), which could affect the area in an unknown way. The stability in land-cover types was also reflected in the various landscape indices calculated. This contrasts with the significant LULC changes identified in neighbouring areas over the study period (Romero Calcerrada & Perry 2004), or in the study area itself from the middle of the 20th century until prior to 1975 (O. Viedma & J.M. Moreno, unpublished data 2006). It thus appears that in this area the main LULC changes had occurred by the time images became available and that the pattern of change reflects growth and consolidation of the LULC changes prior to 1975. The low rate of burning (0.7% annual burned area) also contributed to the near-stability of the whole area. Quasi-steady state landscapes are likely where the average size of disturbances (fires in this case) is small relative to the total extent of the landscape (Shugart & West 1981). Landscapes characterized by small and frequent disturbances may be in some sort of equilibrium, whereas large infrequent disturbances usually lead to non-equilibrium conditions (Bormann & Likens 1979; Pickett & White 1985).

In summary, although the Sierra de Gredos as a whole was one of the areas in the country most severely affected by fire at this time, the dynamics of this fire-prone landscape appear to have been more stable than anticipated, and the overall net effect of LULC change and fire combined in the whole landscape was not great. Nevertheless, this cannot mask the fact that different parts of the landscape underwent very different dynamics, as attested by the evidence that fires tended to concentrate at particular locations, burned areas

Table 2 Multivariate, step-wise linear regression models of LULC types and landscape indices (PD = patch density; MCA = mean core area; CON = contagion; SHDI = Shannon's diversity index) in the whole area as a function of: LULC types in the unburned area; LULC types in the burned area; regeneration stages in the burned area; and LULC types and regeneration stages in either the unburned or burned areas. Shown are the significant variables entering into the model, their coefficient, and the R^2 and significance of the model ($^{*} = 0.05 \geq p > 0.01$; $^{**} = 0.01 > p \geq 0.001$; $^{***} = 0.001 > p$; ns = not significant).

LULC types in entire area	*Unburned LULC types*	*Burned LULC types*	*Regeneration stages*	*Burned–unburned LULC types and regeneration stages*
Pine woodlands	ns	Pine woodlands (+0.1) Pastures (−0.1) $R^2 = 0.86^{***}$	Early-succession (<5 years old) (−0.1) $R^2 = 0.70^{***}$	Early-succession (<5 years old) (−0.1) Burned pastures (−0.1) $R^2 = 0.80^{***}$
Shrublands	ns	Shrublands (+0.1) $R^2 = 0.73^{***}$	ns	Burned shrublands (+0.1) $R^2 = 0.73^{**}$
Pastures	Pastures (+1.1) $R^2 = 0.57^{**}$	Pine woodlands (−0.1) Shrublands (−0.1) $R^2 = 0.91^{***}$	Early-succession (<5 years old) (+0.1) $R^2 = 0.73^{***}$	Burned pine woodlands (−0.1) Burned shrublands (−0.1) $R^2 = 0.91^{***}$
Croplands	Croplands (+0.8) $R^2 = 0.93^{***}$	Croplands (+0.2) $R^2 = 0.42^{**}$	Mid-succession (> 5 years old) (−0.1) $R^2 = 0.81^{***}$	Unburned croplands (+0.8) $R^2 = 0.93^{***}$
Bare soil	ns	Bare soil (+0.1) $R^2 = 0.64^{***}$	Immediate-succession (0 years old) (+0.1) $R^2 = 0.69^{***}$	Immediate-succession (0 years old) (+0.1) $R^2 = 0.69^{***}$
Landscape indices in entire area				
PD	Pastures (−8.6) Pine woodlands (−5.1) $R^2 = 0.97^{***}$	Shrublands (+0.60) Pastures (+0.41) $R^2 = 0.80^{***}$	Mid-succession (> 5 years old) (−0.4) $R^2 = 0.77^{***}$	Unburned pastures (−8.6) Unburned pine woodlands (−5.1) $R^2 = 0.97^{***}$
MCAI	Pastures (+153.8) Pine woodlands (+96.0) $R^2 = 0.96^{***}$	Pine woodlands (−10.9) Pastures (−7.4) $R^2 = 0.82^{***}$	Mid-succession (> 5 years old) (+6.5) $R^2 = 0.78^{***}$	Unburned pastures (+153.8) Unburned pine woodlands (+96.0) $R^2 = 0.96^{***}$
CON	Croplands (−46.5) $R^2 = 0.89^{***}$	Shrublands (+1.84) Pine woodlands (−2.21) $R^2 = 0.88^{***}$	Mid-succession (> 5 years old) (+2.2) $R^2 = 0.91^{***}$	Mid-succession (> 5 years old) (+2.2) $R^2 = 0.91^{***}$
SHDI	Croplands (+0.98) $R^2 = 0.62^{**}$	Pine woodlands (+0.1) Bare soil (+0.1) $R^2 = 0.84^{***}$	Early-succession (<5 years old) (−0.1) Immediate-succession (0 years old) (+0.1) $R^2 = 0.75^{***}$	Mid-succession (> 5 years old) (−0.1) Immediate-succession (0 years old) (+0.1) Unburned croplands (+0.1) $R^2 = 0.95^{***}$

underwent rapid changes and not all LULC types were equally affected by fire. Therefore, small changes at the level of the whole landscape may dilute intense changes at local scales. Unravelling the factors that control the different levels of fire occurrence in the landscape is a subject to be pursued if we are to understand the overall effects of fire at the various scales.

The overall small changes at the level of the whole landscape implied that, at this level, the role of LULC and fire in altering fire hazard was probably small. Despite this, and even though difficult to evaluate, the changes observed were compatible with a trend of increasing fire hazard in the area. In general, the LULC changes in the area that did not burn during the study period, and in the area as a whole, were in the direction of decreasing landscape fragmentation, including the tendency of patch density to decrease and mean patch size and mean core area to increase. These changes might have contributed to a modified fire hazard, if only because stability through time in terms of land-cover type does not necessarily imply stability of flammability-related characteristics, as these may change with age as a result of, for example, the accumulation of litter, as happens in pine woodlands (Trabaud 2000), thereby increasing fire susceptibility through time (Vázquez & Moreno 2001). As for the burned areas, the shrublands that develop after fire can reach near-continuous cover in a few years (Faraco *et al.* 1993), as well as accumulate large quantities of highly flammable biomass (Pérez & Moreno 1998). Persistence in land cover may not imply persistence in the hazardousness of the landscape.

The small tendency detected for some of the more flammable land cover types to increase, together with their contagiousness, could have further increased fire hazard. Not all land cover types were equally affected by fire: pine woodlands in this area were burned in a much greater

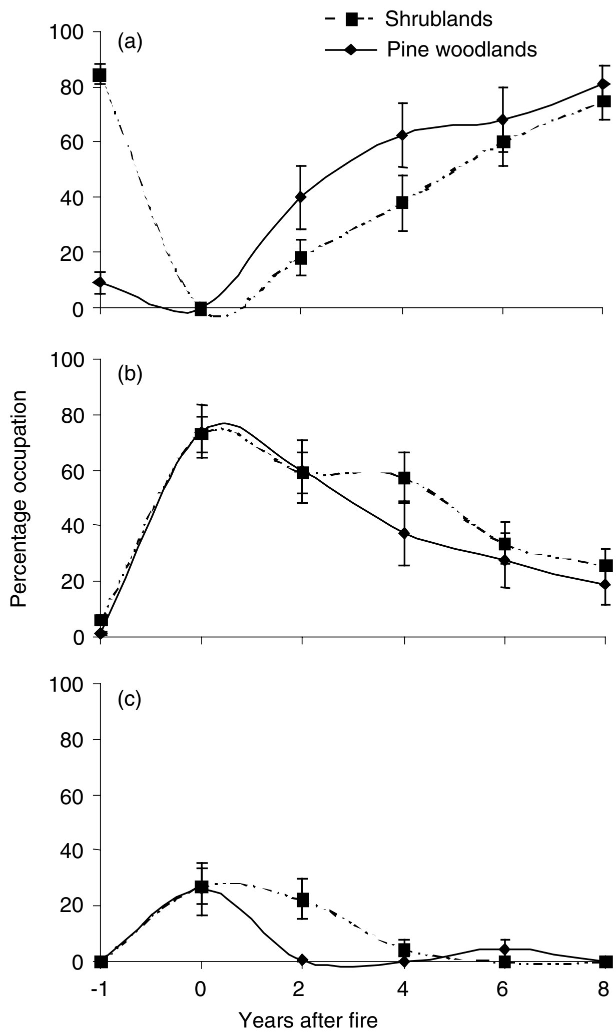

Figure 3 Variation in the percentage area occupied by the main LULC types for up to eight years after fire in areas burned as pine woodlands or shrublands: (*a*) shrublands, (*b*) pastures and (*c*) bare soil ($n = 32$).

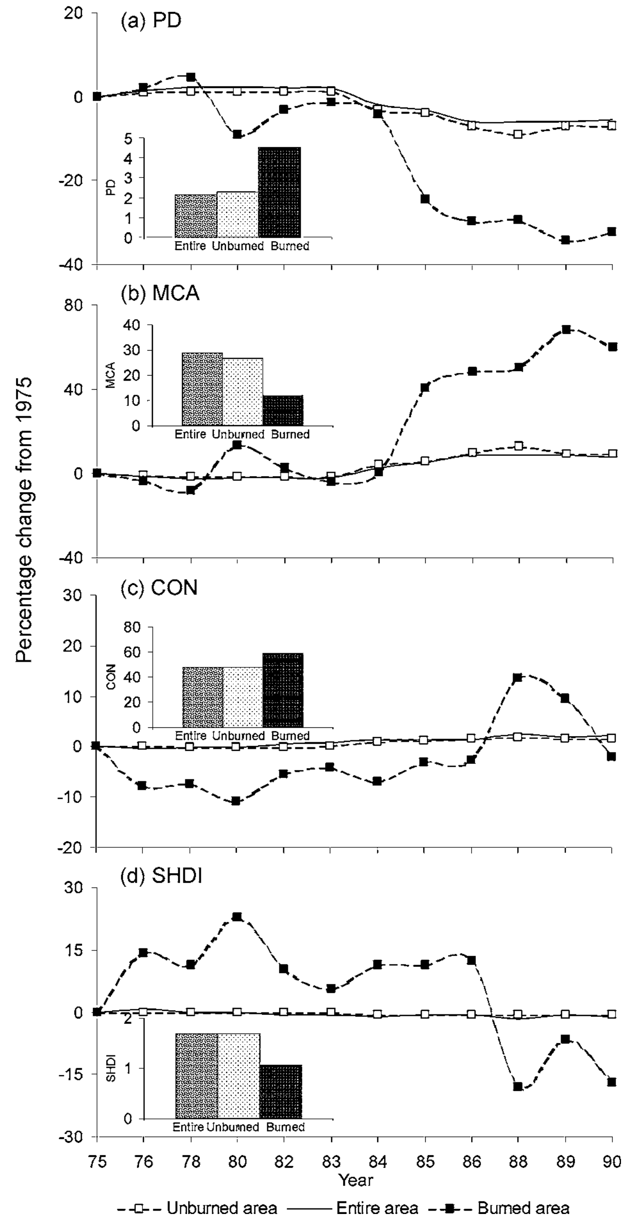

Figure 4 Percentage change with time (1975–1990) in (*a*) patch density (PD), (*b*) mean core area index (MCA); (*c*) contagion (CON); and (*d*) Shannon's diversity index (SHDI) for either the whole area or the areas burned or not burned in Sierra de Gredos, Spain. Absolute values of the landscape indices at the start of the period (1975) are shown as insets.

proportion (five times) to their cover in the landscape throughout the study period; shrublands were also affected by fire in greater proportion than their availability in the area, although much less intensely and constantly through time (J.M. Moreno, G. Zavala & O. Viedma, unpublished data 2006). Therefore, although small, changes in certain LULC types may have implied alterations of fire hazard and, ultimately, of fire occurrence that were not in direct proportion to the change in the area they covered.

Fires produced landscape structural characteristics that contributed towards their tendency to fire. Five years after a fire, burned areas exhibited land cover compatible with shrublands. In general, except during the years immediately following fire, burned areas tended to present reduced PD, increased MCA and reduced SHDI values. After 6–8 years, landscape characteristics of burned areas tended to return to previous conditions, albeit with some differences depending on the type of vegetation burned. While burned shrublands tended to quickly return to their pre-fire status, burned pine woodlands were more unstable and remained more heterogeneous some years after fire. These findings reflect field observations rather well. Monitoring over several years of a set of plots that made up a replicated chrono-sequence of areas that had burned as shrublands or as pine-woodlands showed that the variability in the ground-projected cover of woody species was much greater in pine-woodlands areas than in shrublands (Moreno *et al*. 1996).

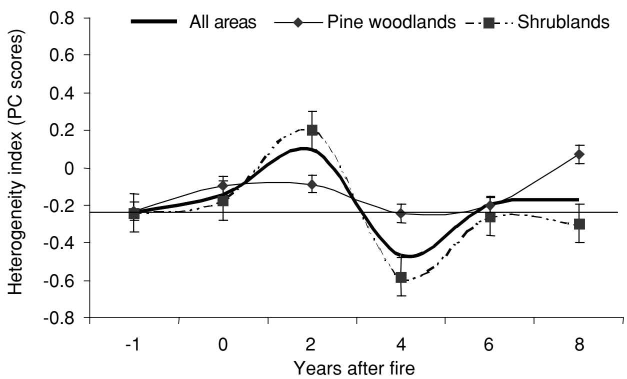

Figure 5 Variation in landscape heterogeneity (based on the scores of the first axis of a PCA on several landscape indices; see Methods) during the first eight years after fire for all burned areas, areas burned as pine woodlands ($n = 14$) and areas burned as shrublands ($n = 18$).

Overall, it appears that both processes (LULC and fire) operating in the landscape, either in the unburned or burned areas, tended to produce changes in landscape characteristics that contributed to reduce fragmentation in the landscape. Shrublands dominated the early regeneration phases after fire in the two types of LULC that burned most frequently (shrublands or pine-woodlands). It is possible that some of the shrublands arising from burned pine-woodlands might have been young pine woodlands that, in their early stages, could not be differentiated from shrublands. Field observations, however, indicate that this was not as frequent as might have been expected (Faraco *et al.* 1993; Pérez & Moreno 1998). How much these patterns contributed later to increased fire hazard could not be quantified, owing to the short period of time analysed. Yet, the tendency of all areas, irrespective of their land-use history (Pérez *et al.* 2003), management, or changes in fire severity (Pérez & Moreno 1998), to produce similar vegetative cover in short periods of time would tend to increase hazard. The trends observed by remote sensing coincided with field observations and may continue. At the present time, most areas are poorly managed because of the expense required, and vegetation dynamics are not directly controlled by humans. Changes in perceptions and values (Trabaud & Galtiè 1996), particularly when timber values disappear, and increased hazard of the post-fire vegetation may cause burned areas to burn again more frequently than initially (Vázquez & Moreno 2001; Salvador *et al.* 2005). In fact, the largest burn analysed here (nearly 6000 ha) burned almost totally again 15 years later (J.M. Moreno, personal observation 2002).

In summary, the main LULC changes increased the proportion of LULC types that are most frequently affected by fire and produced less fragmented landscapes. In addition, burned areas quickly turned into shrublands and, after a few years, changed structural patterns in the same direction as LULC change. The combination of reduced fragmentation

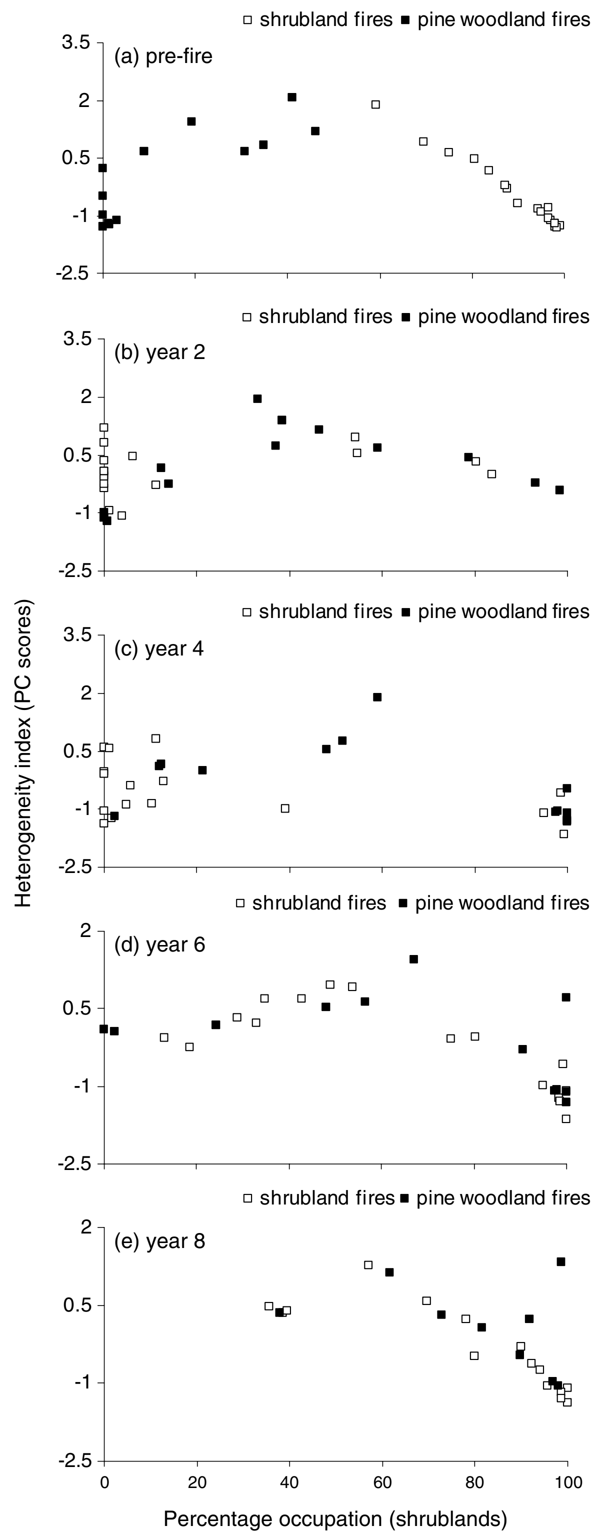

Figure 6 Variation in landscape heterogeneity (based on the scores of the first axis of a PCA on several landscape indices; see Methods) versus the percentage occupation of regenerated shrublands for areas burned as pine woodlands ($n = 14$) and areas burned as shrublands ($n = 18$). (*a*) Pre-fire, (*b*) two years after fire, (*c*) four years after fire, (*d*) six years after fire and (*e*) eight years after fire.

induced by land-use change and fire may further increase fire hazard in the area, making it more unstable through time.

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