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Analysis of Postfire Vegetation Dynamics of Mediterranean Shrub Species Based on Terrestrial and NDVI Data

Rocío Hernández-Clemente, R. M. Navarro Cerrillo, J. E. Hernández-Bermejo, S. Escuin Royo, N. A. Kasimis

Abstract

The present study offers an analysis of regen- eration patterns and diversity dynamics after a wildfire, which occurred in 1993 and affected about 7000 ha in southern Spain. The aim of the work was to analyze the rule in the succession of shrub species after fire, relating it to the changes registered in the Normalized Difference Vegetation Index (NDVI). Fractional vegetation cover was recorded from permanent plots in 2000 and 2005. NDVI data related to each time were obtained from Landsat images. Both data sets, from fieldwork and remote sensing, were analyzed through statistical and quantitative analyses and then correlated. Results have permitted the description of the change in plant cover and species composition on a global and plot scale. It can be affirmed that, from the seventh to the twelfth year after the fire, the floristic composition within the burned area remained unchanged at a global level. However, on a smaller scale (plot level), the major shrub species, *Ulex parviflorus*, *Rosmarinus offici- nalis*, and *Cistus clusii*, underwent significant changes. The regeneration dynamics established by these species condi- tioned plant species composition and, consequently, diversity indexes such as Shannon (H) and Simpson (D). The changes recorded in the NDVI values corresponding to the surveyed plots were highly correlated with those found in the regrowth of the main species. Areas dominated by *U. parviflorus* in a senile phase were related to a decrease in NDVI values and an increase in the number of species. This result describes the successional dynamics; the dry- ness of the main colonizer shrub species is allowing the regrowth and re-establishment of other species. Within the study area, NDVI shows sensitivity to postfire plant cover changes and indirectly expresses the diversity dynamics.

Keywords: Fire ecology, Diversity, Succession, Remote sensing, NDVI, Mediterranean communities

Irrespective of the variability of wildfire effects on vegetation, it is widely accepted (Hanes 1971; Trabaud and Lepart 1981) that Mediterranean ecosystems affected by wildfires show a direct regeneration pattern. Direct regen- eration begins from a more advanced level than the theoretical pioneer stages in a typical succession process (Clements 1916). This regeneration system is known as autosuccession. The regeneration of vegetation after fire depends principally on the resilience of the species in the community (Kuhnholtz-Lordat 1938; Naveh 1975; Pausas 2004). However, it is not possible to generalize this behavior for all Mediterranean plant communities. In fact, many species have not developed any kind of fire defense or postfire regeneration mechanism (Piussi 1992). Even fire-adapted species are conditioned by countless environ- mental factors, which limit their efficiency. As a consequence, Mediterranean ecosystem regeneration models are difficult to predict and classify. Such processes require a wide basis of scientific data during long obser- vation periods. Furthermore, for many centuries, these ecosystems have been continuously subjected to human disturbance: timber and charcoal production, fires, browsing, and, more recently, reforestations with allochthonous species of *Pinus* and, occasionally, of *Abies* or *Cupressus*. These human impacts have modified these communities, creating a fragmented landscape composed of different successional stages where the quickly resprouting shrub species complicate the plant canopy and its dynamism. This fact increases the difficulty of analyzing spatial postfire vegetation dynamics.

In recent decades, the understanding of vegetation dynamics has been improved by applying remote-sensing techniques. The temporal resolution of remote-sensing data sets enables the study of dynamic phenomena such as postfire ecosystem recovery over a multiple-year period. Previous research on the use of remotely sensed imagery in monitoring regeneration patterns in Mediterranean ecosystems has been presented by many authors (Viedma and others 1997; Díaz-Delgado and Pons 2001; Riaño and others 2002; Telesca and Lasaponara 2006). Some of these have reported that biomass-based vegetation indexes can be a useful detection tool for indicating changes in vegetation before and after fire (Marchetti and others 1995). The Normalized Difference Vegetation Index (NDVI) has been a widely used tool to assess recovery processes, as it is sensitive to changes in the fractional vegetation cover until a full cover is reached (Carlson and Ripley 1997).

Although ecologists and remote-sensing experts have performed numerous studies on postfire regeneration processes in the Mediterranean basin, few of these studies integrate both perspectives (Hall and others 1991). There is even less work at a community level (especially concerning shrubbery) associated with biodiversity changes and supported by enough fieldwork from multitemporal records for long-term studies. However, given current global climate change conditions, a more accurate knowledge of plant community recovery is needed. And, in short, the implementation of conservation and restoration actions in burned areas on a landscape scale and the development of more accurate models for interpreting remote-sensing data are required. Thus, the main objective sought in this analysis was to gain an understanding of postfire shrub regeneration dynamics and how the NDVI behaves with regard to those changes through (1) analyses of postfire changes in diversity and composition; (2) definition of the main species causing the changes, or indicator species; and (3) analysis of the NDVI changes brought about by plant community succession.

Study Area

The study area, Sierra de Huétor, is located in Beas de Granada, southern Spain (Fig. 1), with Universal Transverse Mercator (UTM) coordinates ranging from ULX: 457343 to LRX: 473918 and ULY: 4129353 to LRY: 4118806 in European Datum, zone 30 N (Fig. 1). The climate is transitional, varying between Mediterranean semiarid and Mediterranean subhumid, and the average annual rainfall is 600 mm. According to Rivas-Martinez and others (1997), the area belongs to the Malacitano-Almijarense sector, with a supra-Mediterranean climate. The annual mean temperature ranges from 7 to 17°C. Its highly homogeneous lithology is chiefly composed of basic soils, such as carbonate stone, limestone, and dolomites, covering almost 80% of the burned area. The rest is covered by acidic soils, such as mica-schists, gneisses, (phyllites), and slates. According to USDA Soil Taxonomy, the soils are classified as skeletal (lithosols and regosols) and are underdeveloped due to the original hardness of the high, rocky (limestone) slopes on which they have evolved.

A fire occurred between August 23 and August 26, 1993, and affected about 7000 ha. The fire severity level was ranked as extreme in 40% of the area, moderate in 47%, and low for the remaining 13% of the area (Hernández-Clemente and others 2007). Altitudes from 1000 to 1700 m are found within the burned area, although the topography is not very abrupt. After the 1993 wildfire, only a few reforestation work tasks have taken place in this area. The first one, at the end of 1996, consisted of aerial sowing, which gave poor results. The next attempt, direct machine planting in the field, was sporadic and not very significant.

Plant Community Types

The vegetation in the area has been greatly altered by human actions and, consequently, presents a state of advanced degradation. In the 1940s, an intense reforestation program planted *Pinus pinaster*, *Pinus halapensis*, *Pinus nigra*, and, to a lesser extent, *Pinus sylvestris*. According to Rivas-Martinez and others (1997), the potential vegetation found in the burned area can be defined by three main community types:

(a) Basophiles meso-Mediterranean Holm-oak woods, the dominant species being *Quercus ilex*, *Juniperus oxycedrus*, and *Crataegus monogyna*. In a degraded state, brush like *Ulex parviflorus*, *Rosmarinus officinalis*, *Cistus clusii*, *Lavandula lanata*, *Daphne gnidium*, and *Retama sphaerocarpa* are found. This is the most prominent formation in the burned landscape, sometimes even including species such as *Thymus*, *Fumana*, and *Helianthemum*.

(b) Basophiles supra-Mediterranean Holm oak woods with *Quercus ilex* as the dominant species, with the presence of *Quercus faginea*, *Prunus ramburii*, and *Berberis hispanica*, and sometimes turning into *Pinus*

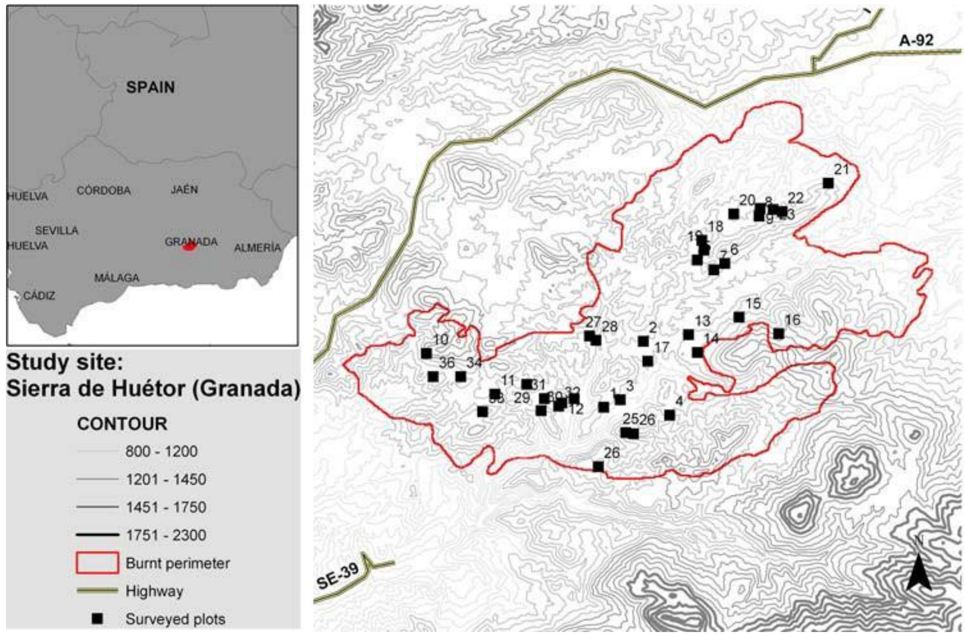


Fig. 1 Study site location, Granada, Spain; latitude/ longitude, 37815′38.94″N/ 3823′45.51″. A forest fire occurred in 1992. Sample plots are shown within the burned area

pinaster pine woods. Scrublands and thorn thickets at these elevations are composed of communities dominated by *Salvia lavandulaefolia*, *Lavandula lanata*, *Echinospartium boissieri*, *Ulex parviflorus*, *Euphorbia nicaensis*, *Thymus granatensis*, *T. mastichina*, *Coris monspeliensis*, and other chamaephytes and hemicryptophytes. In a degraded form these communities appear as open brush of *Cytisus reverchonii* scrubs, with *Berberis hispanica*, *Amelanchier ovalis*, *Salvia lavandulaefolia*, *Fumana ericoides*, *Santolina canescens*, *Thymus mastichina*, and *Daphne gnidium*, or even scrublands with *Echinospartium boissieri* as well as *Ulex parviflorus*.

(c) Siliceous supra-Mediterranean Holm oak and gall oak woods, occasionally transformed into black pine woods, are sometimes accompanied by *Quercus faginea* and, generally, *Adenocarpus decorticans*. Associated species are junipers (*Juniperus oxycedrus*) and other shrubby species such as *Doronicum plantagineum*, *Retama spaherocarpa*, *Cistus albidus*, and *Euphorbia pinea*. These formations sometimes degrade with *Cistus laurifolius* as the dominant species, together with *Thymus zygis* subsp. *gracilis*, *Fumana thymifolia*, *Halimium viscosum*, *Thymus mastichina*, and *Daphne gnidium*.

Plant Community Dynamics

Mediterranean shrubs play an important role after fire. Many of them have fire-resistant fruits and seeds due to their thick, insulated walls (*Cistus*), while other species store seeds in the soil (*Ulex*, *Rosmarinus*). There are also resprouting *taxa* like *Cytisus* spp. and *Daphne gnidium*. In this way, after fires, instead of initiating the succession process with herbaceous communities, shrubs similar to the prefire ones reappear, through autosuccession. This initial shrub land chokes the grassy plants and slowly allows for the resprouting of the macrofanerophytous species. The process continues, thus ensuring that this regeneration point is neither the beginning point of a pioneer regime nor the final point of a stable system.

Autosuccession does not represent a new reassuring paradigm to explain the response of Mediterranean vegetation to wildfires (Terradas 2001) since it does not always occur, nor does it always follow a foreseeable pattern. In some instances, the regeneration process is influenced by random phenomena, making it even more difficult to predict. While some environmental variables can be roughly modeled, others, like the previous historical factors (prior wildfires, soil seed banks, anthropogenic pressure, etc.), are countless. Mediterranean plant communities recovering from wildfire have been studied by many authors with the common intention of predicting their regeneration pro- cesses. Mateo Sanz and Mansanet (1982) studied the response model of Mediterranean communities in limy soils. These authors did not detect any significant variation in the number of species during the first 10 years after the fire. Morey and Trabaud (1988), in a study located in Mallorca (Balearic Islands), and Tárrega and Calabuig (1987), in the north of Spain, arrived at a similar conclu- sion: an increase in specific diversity during the first 3 years after fire. Later, this diversity decreases and becomes stable. Both observations were compatible and depended on the number of species in the soil seed bank. Pausas (1999) used prediction models, which simulate the regeneration of Mediterranean communities in Catalonia. Applying these models, the author simulated the behavior of populations of resprouting species of *Quercus* and *Erica*, mixed with other seedlings like *Cistus* and certain *Pinus* species. The prediction model established that recurrent fires would reduce the occurrence of *Quercus* species and increase those of *Cistus* and *Erica* and, even more so, those of *Pinus*. In a similar investigation, Lloret and others (2003) studied the response of species with a history of exposure to fire. As a result, obligate seedling species such as *Pinus halapensis*, *Rosmarinus officinalis*, *Cistus albidus*, and *C. saviifolius* had decreased in 20 years as a consequence of recurrent wildfires. The same occurred with the resprouting species *Erica multiflora*, while other resprouting species such as *Ampelodesmos mauritanica* and *Quercus coccifera* had increased. In the long term, Pausas (2004) insisted that the *Quercus* species will disappear with recurrent wildfires, with *Cistus* species and *Ulex parviflorus* becoming the dominant ones and a scant representation of *Quercus* and *Pinus*.

Leaving aside the community level, the postfire patterns of Mediterranean species are still unknown and have been widely discussed by many authors. Tárrega and others (1997) have studied the behavior of *Cistus laurifolius* subjected to experimental fires, confirming the presence of autosuccession processes. *C. laurifolius* grows quickly, within a few years taking the place of other herbaceous species, reducing diversity, and becoming the dominant species. Another remarkable shrub land, composed of *Ulex parviflorus* as the dominant species, was analyzed by Baeza (2001). He studied how dwarf gorse (*U. parviflorus*) responded to brush-cutting and burnings at two different stages of the community's development. He established three stages in the cycle of this species, which became apparent from the third or fourth year and onward: (1) 1–8 years after fire, with thin branches and green phyllodes with no defined vertical profile; (2) 8–20 years after fire, individuals of mature gorse contained up to 60% dead biomass, with live stems measuring 15–29 cm—the maximum cover is attained at this second stage; and (3) 20–25 years later, at the third stage, when the plant reaches the final senescent stage.

Materials and Methods

Image Preprocessing

This study used a Landsat Enhanced Thematic Mapper (ETM?) image taken on August 31, 2000, and a Thematic Mapper (TM) image from August 18, 2004, both covering the burned area (path 200, row 34). Although more advanced satellite sensors are available, time series of Landsat were chosen due to the suitability of acquisition along the time. With images available from 1972, Landsat presents the possibility of comparing postfire remote sensing data with images acquired before the fire. In addition, the spatial resolution of Landsat is 30 m, which represents an adequate spatial resolution to handle a burned area of more than 7000 ha. Image preprocessing involved the conversion of digital numbers into reflectance values and the georeferentiation of the image. Standard gain and offset coefficients for each satellite and period were applied in order to transform digital numbers into reflectance values. Finally, a relative atmospheric correction was required to normalize remotely sensed images for time-series analysis. The relative atmospheric correction was based on invariant targets following the methodology applied by Caselles and Lopez Garcia (1989). Results obtained from the image normalization yielded a mean correlation coefficient (r) of 0.98, with a standard error of 0.029. The 2000 image was georectified using ground control points and a high-resolution digital orthophoto quadraught (DOQ) of the study area obtained from the regional government of Andalusia. The 2005 image was georeferenced to the 2000 image. Corrected images provided a maximum root mean square (RMS) error of less than 1 pixel. Nontopographic correction was applied due to the samples selected having slopes of \10%. This criterion was established due to the high spectral distortion of the signature created by topographic correction.

The NDVI is an index calculated from reflectance measured in the visible and near-infrared channels. It is related to the fraction of photosynthetically active radiation. Change detection over time was calculated through the difference in NDVI between 2004 and 2000 as follows:

$$\mathrm{NDVI}[t_2 - t_1] = \mathrm{NDVI}_{ij}[t_2] - \mathrm{NDVI}_{ij}[t_1]$$

where ij is the pixel coordinate, t_2 = 2004, and t_1 = 2000, and where

$$\mathrm{NDVI} = (\mathrm{IR} - \mathrm{R})/(\mathrm{IR} + \mathrm{R}).$$

Digital values of each plot were extracted by calculating the average pixel reflectance of a 3 × 3 matrix from the satellite images. It has been widely accepted that the approach of using a pixel matrix reduces potential geolocation errors, which are difficult to avoid in time-series studies.

Ground Data

The field survey took place in August 2000 and 2005. The total area sampled in 2000 (36 plots, each with a 25-m radius) was 66,759 m^2, representing 0.09% of the total burned area. Using an Indian Remote Sensing Panchromatic

(IRS-PAN) image from 1998 and a 1999 orthophoto (1-m resolution), the test site selection was based on the maximum homogeneity, similar texture, and differences in vegetation types between plots. Survey points were then analyzed to define the main vegetation classes within the burned area by means of hierarchical cluster analysis. Based on these results, the most representative samples from each class were selected for a repeated inventory in 2005.

On the ground, plot locations were identified using a handheld global positioning system (GPS) with an average error of \5 m. Two 50-m-long perpendicular linear transects (one in the direction of the maximum slope) were defined for each plot and measured in both years (2000 and 2005) following the same methodology (linear interception method) (Bonham 1989). Along each transect, the morphologic variable measurements were the tree and brush canopy interception longitudes (l_k). The plants identified in the fieldwork were incorporated into the herbarium specimens with references COA00925 to COA00957 by the Herbarium of the Agricultural and Forest Sciences and Resources Department, University of Córdoba. This incorporation would guarantee the correct determination of specimens in future studies within the burned area.

Based on the data set surveyed in 2000, a complete cluster sampling analysis (Fig. 2) was performed. The identification of the main plant associations within the burned area allowed the selection of representative sample plots as training areas for species composition change analysis over time. The plots under analysis were only comprised by species with a shrubland profile. Ten plots were selected, relative to basophiles and acidophilus meso- and supra-Mediterranean vegetation types. All the plots were affected by fire at the extreme severity level. The severity levels were estimated according to Escuin and others (2006). The global fire severity level was ranked as extreme in 40% of the area, moderate in 47%, and low for the remaining 13% of the area (Hernández-Clemente and others 2007).

The following variables were calculated.

(a) Fractional vegetation cover (FVC). For each transect, the sum of intercepted fraction per species was calculated as $\mathrm{IF}_{t1} = \mathrm{R}l_k/d$ and $\mathrm{IF}_{t2} = \mathrm{R}l_k/d$, where l_k is equal to the distance intercepted for each species and d is the distance of each transect (50 m). By measuring two transects per sample, the FVC was obtained as $\mathrm{FVC} = (\mathrm{IF}_{t1}\ ?\ \mathrm{IF}_{t2})/2$.

(b) Richness (S) or total number of different species per sample. This variable measures the heterogeneity of the vegetation, providing information on the floristic composition of the plant community.

(c) Simpson's Reciprocal Index (D) measures the probability that two individuals randomly selected from a sample belong to the same species. Simpson's Reciprocal Index gives more weight to the most abundant species in a sample, decreasing the total species richness. The index equation is expressed as $D = 1/\mathrm{k}$, where $\mathrm{k} = 0$ represents infinite diversity and $\mathrm{k} = 1$, no diversity. k values are obtained from the equation $\mathrm{k} = \mathrm{R}p_i^{\,2}$, where $p_i = n_i/N$, where n_i is the total number of organisms of a particular species i, and N is the total number of organisms of all the species.

(d) Shannon-Weaber index (H), stated as $H = -\mathrm{R}p_i \log p_i$, where p_i is the same expression defined for the Simpson index. H is zero only if there is one species in the sample, and H is the maximum value ($H = \log_2 S$) when all S species are represented by a perfect even distribution of abundances. H measures the heterogeneity of the system in terms of entropy.

Vegetation Cover Change per Species

The analysis was carried out based on FVC per species from inventory years 2000 and 2005. The data were obtained by calculating the average of the FVC per species

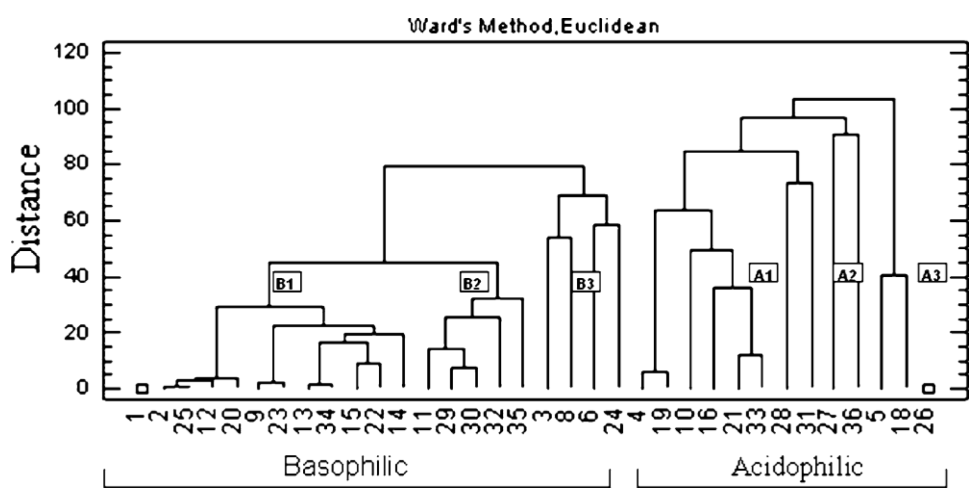


Fig. 2 Dendrogram based on fractional vegetation cover per species in all field plots surveyed in 2000. The *X* axis represents sample plot number identification; the *Y* axis, the distance between groups

measured in the 10 plots. With these data, two different approaches were taken: a statistical analysis to test the significance of the change and a quantification analysis to evaluate the FVC change registered per species. First, the effect of time on the diversity and floristic composition was tested with a statistical analysis of variance using the Wilcoxon test for related samples. Nonparametric statistics were chosen because the Kolmogorov–Smirnov and Shapiro–Wilks tests showed that the global change register per species differed significantly from a normal distribution (*p* [0.05). The data set was a 28 9 2 matrix, with the total average FVC per species of the *N* plots inventoried in 2000 and 2005. Next, FVC changes per species were quantified, enabling identification of the dominant plant species and revealing the regeneration dynamics of the community.

Vegetation Cover Change per Plot

This second analysis measured FVC per species per plot. The data were obtained by calculating the mean FVC per species and per plot. The goal of this analysis was to evaluate whether FVC changes were significant at the plot level over time (2000, 2005) and, if so, which were the main species promoting the change. In order to select an adequate statistical method, the Kolmogorov–Smirnov and Shapiro–Wilks tests were calculated, showing that the distributions of FVC per species in each plot fitted a normal distribution (*p* [0.05). To assess the composition of the change, a univariate general linear model (GLM) with repeated measurements was performed. This analysis was selected because it needs fewer subjects than a completely randomized analysis and it permits the elimination of any residual variation due to differences between subjects, since these are all the same. The model was designed based on the data computed in 2000 and 2005 (one repetition) of the FVC surveyed (28 inventoried species) for each plot (10 plots in total). The statistical analysis, calculated with SPSS 8.0 software, included a null hypothesis test to prove significant differences in species composition, a within-subject test, and a between-subject effect test, this being (*p* \ 0.05) for all of them. The final objective of those analyses was to identify if there were any significant species promoting the change. Those species would be called indicator species. Finally, diversity indexes were calculated to evaluate recovery changes.

NDVI Change Analysis per Plot

Changes in NDVI values were analyzed to study how this index behaved in relation to other ecological descriptors. The analyzed variables were calculated as the difference of the average in 2005 minus the average in 2000. The variables selected were as follows:

(a) Change ($t_2 - t_1$) in FVC of the main species responsible for the change
(b) Change ($t_2 - t_1$) in global FVC
(c) Changes ($t_2 - t_1$) in diversity registered throughout the regeneration processes, estimated based on differences in the Shannon and Simpson indexes and richness values per plot
(d) NDVI ($t_2 - t_1$) = $NDVI_{ij}$ (t_2) − $NDVI_{ij}$ (t_1), where ij is the pixel coordinate, $t_2 = 2004$, and $t_1 = 2000$.

Statistical relationships of all these variables were computed by a bivariate correlation matrix and described by Pearson, Kendall *s*-b, and Spearman correlation coefficients. All of them ranged from −1 (a perfect negative relationship) to ?1 (a perfect positive relationship). A value of 0 indicated no linear relationship. Finally, the best combinations were chosen to create different linear regression models between remote data and the diversity and FVC dataset.

Results

Sampling Area Analysis

The cluster analysis obtained from the FVC surveyed in 2000 permitted the description of the main communities in the area. The results of the cluster analysis are shown in Fig. 2. Two well-differentiated groups are distinguished at a Euclidean distance of 80. These groups were strongly influenced by the soil type (basophilic and acidophilic soils) and altitude location of the plots. Over a Euclidean distance of 20, six subgroups are shown in the dendrogram. These subgroups are representative of the different plant communities found in the area. Table 1 reports cluster analysis results. The clusters indicate the floristic classification described by Rivas-Martinez and others (1997). Different plant formations can be interpreted from the cluster analysis, according to the presence of dominant scrub or tree species, soil type, and altitude. The cluster analysis allowed the distinction of different combinations within the complex mosaic composed of *Ulex parviflorus*, *Rosmarinus officinalis*, and *Cistus clusii*. This codominance reappears in a mosaic shape in Sierra de Húetor. Based on those results, the most representative sample plots were selected in order to evaluate floristic changes occurring from 2000 to 2005.

FVC Change per Species

The change in floristic composition of the total FVC in burned area plots, analyzed by the Wilcoxon test, showed no significant variation (*p* [0.001). The results of the

Table 1 Composition of clusters derived from vegetation cover per species in 2000

Cluster	Plot nos.	Soil type	Plant community type	Main species	FVC (mean ± SE)	No. of species (mean ± SE)	Altitude (m; mean ± SE)
A			Gorse and rosemary thicket (G-R)				
B.1	2, 25, 12, 20	Basophilic	Strict G-R	*Ulex parviflorus*	0.24 ± 0.07	5.47 ± 2.01	1264.32 ± 40.21
				R. officinalis	0.21 ± 0.06		
				Cistus clusii	0.90 ± 0.03		
B.2	9, 23, 13, 34, 15, 22, 14		G-R with oak	*Ulex parviflorus*	0.26 ± 0.09	5.32 ± 2.11	1186.58 ± 284.00
				R. officinalis	0.19 ± 0.08		
				Cistus clusii	0.05 ± 0.01		
B.3	11, 29, 30, 32, 35, 3, 8, 6, 24		G-R ? a large number of accompanying species	*Ulex parviflorus*	0.27 ± 0.10	9.01 ± 1.24	1291.02 ± 79.47
				R. officinalis	0.16 ± 0.05		
				Cistus clusii	0.06 ± 0.04		
B			Pines, oaks, or rose thicket				
A.1	4, 19, 10, 16, 21, 33	Acidophilic	Pinewood	*Pinus pinaster*	0.45 ± 0.20	4.78 ± 2.00	1417.22 ± 99.10
A.2	28, 31, 27, 36		Holm oak	*Quercus ilex*	0.22 ± 0.14	6.10 ± 1.45	1310.12 ± 59.01
A.3	5, 18		Laurel-leaved rock rose	*Cistus laurifolius*	0.35 ± 0.06	7.01 ± 2.33	1413.23 ± 38.22
Residual	1, 26		Great diversity with no dominance		0.03 ± 0.06	11 ± 0.00	1305.34 ± 7.23

FVC fractional vegetation cover

Wilcoxon test ($p = 0.06$) were verified with the results of the Sign test ($p = 0.230$) and Student *t*-test ($p = 0.055$). These suggest that from the 7th to the 12th year after the fire, the general vegetation composition remained practically unchanged. At this point, it is interesting to highlight that the resulting values are close to the limits at which significance is accepted, so that, despite the nonvariance in the vegetation composition, this can be interpreted as being an indication that the latter is in a phase of progressive change. Nonetheless, the diversity indexes show a slow progress rate in the successional process (Table 2). The mean FVC of all species inventoried from 2000 and 2005 shows an average increase of 12%.

The results obtained from the quantitative change analysis are shown in Figs. 3, 4. Figure 3 depicts the mean FVC per species surveyed in 2000 and 2005. *Rosmarinus officinalis* and *Pinus halepensis* registered the highest FVC increase. *Ulex parviflorus* and *Rosmarinus officinalis* were the dominant species in both years, averaging, respectively, 24% and 11% in 2000 and 25% and 15% in 2005. Figure 4 depicts the mean FVC per species divided by the global FVC of all the species surveyed in 2000 and 2005. It is interesting to highlight that *U. parviflorus* depicts the highest decrease from 2000 to 2005. Other species registering a decrease were *Cistus clusii*, *Adenocarpus decorticans*, *Genista scorpius*, and *Cistus laurifolius.* A relative increase in the FVC from 2000 to 2005 was observed for *R. officinalis*, *P. halepensis*, and *P. pinaster*. More than 50% of the FVC mean measurement within the burned area is covered by one species, *U. parviflorus.*

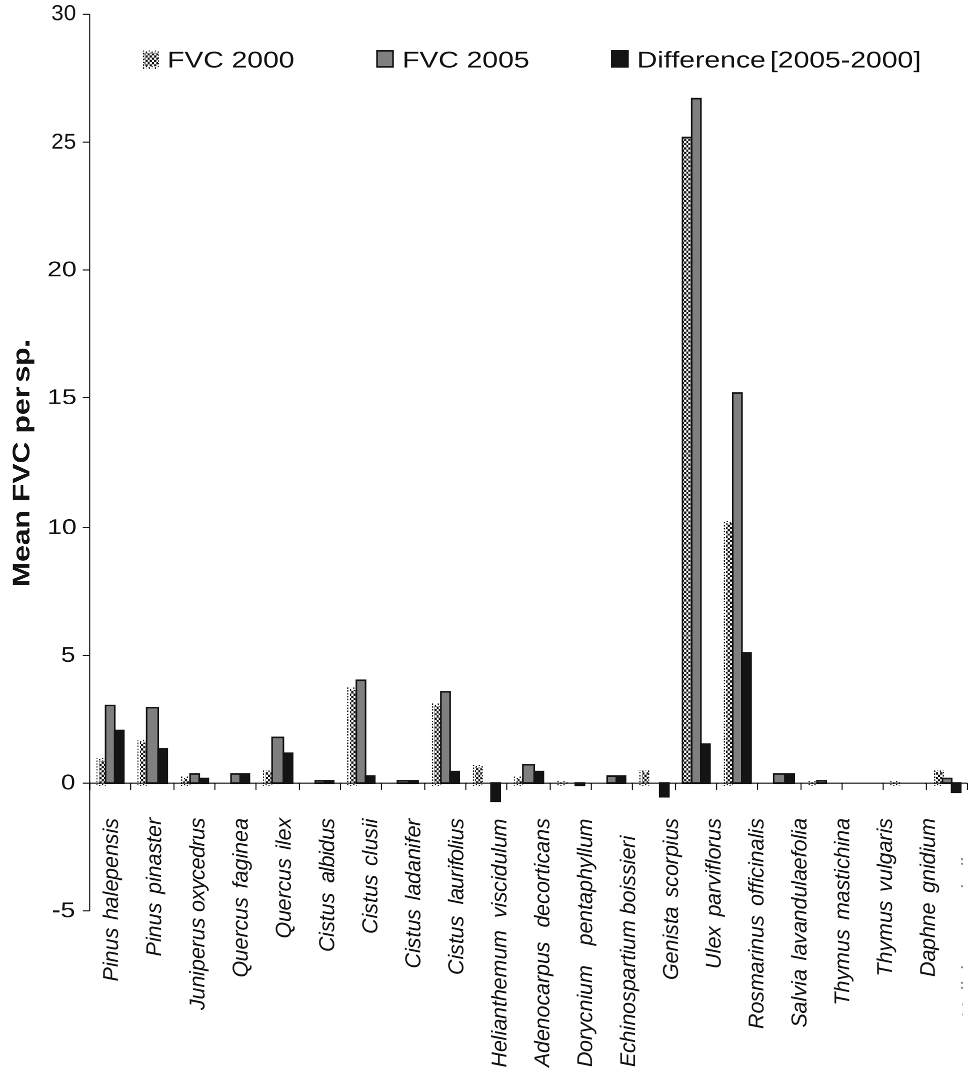
Table 2 Difference (2000–2005) in richness, Shannon and Simpson indexes, total fractional vegetation cover (FVC), and *Rosmarinus officinalis* and *Ulex parviflorus* cover (the dominant species) over the 10 plots analyzed (no. 2, 5, 9, 12, 13, 19, 22, 32 34, 36)

	2000			2005		
	Mean	SE	SD	Mean	SE	SD
Richness	5.00	0.55	1.76	5.50	0.42	1.35
Shannon index	1.34	0.15	0.50	1.60	0.10	0.30
Simpson index	2.07	0.21	0.68	2.60	0.19	0.60
Global FVC (%)	47.56	0.04	0.12	59.61	0.05	0.16
U. parviflorus (%)	24.83	0.05	0.14	26.68	0.05	0.16
R. officinalis (%)	10.83	0.20	0.09	15.25	0.03	0.10

Vegetation Cover Changes per Plot

According to the results obtained from the variance of repeated measurements analysis, the vegetation composition varied significantly in each plot from 2000 to 2005. The results of the GLM performed from repeated measurements describe different aspects of the change at plot level. The null hypothesis of no differences in species composition is rejected by Wilks' k test at *p* \ 0.05. A within-factor test showing *F*-statistics with *p* \ 0.001 related to the time factor reasserts the significance of this change. Finally, the results derived from a between-factors test identify only three species that contribute significantly

Fig. 3 Mean fractional vegetation cover (FVC) per species for the 10 plots surveyed in 2000 and 2005. The change is expressed as the difference (2005–2000) between the FVC measurement each year



to variance per plot ($p < 0.05$): *Ulex parviflorus*, *Rosmarinus officinalis*, and *Cistus clusi*. The latter were defined in this work as being succession indicator species. Finally, diversity index values obtained from each plot are depicted in Fig. 5. According to these results, it is not possible, in general, to define any specific trend in the diversity.

Spectral Response of the Change per Plot

Once the main indicator species promoting change were identified, the analysis of the change was directed toward the study of the relationships between the diversity indexes and the spectral change in NDVI within the sample plots. Table 3 presents the results of the 7×7 correlation matrix constructed with the differences in the following variables derived from 2000 and 2005 data: diversity indexes (H, D), richness (S), mean fractional vegetation cover (MFVC), *Ulex parviflorus* mean FVC, sum of dominant species mean FVC (*Ulex parviflorus*, *Rosmarinus officinalis*, and *Cistus clusi*), and NDVI per plot. All the variables were obtained as the difference in the average values related to 2000 and 2005 dataset per plot. Results derived from the correlation matrix offer the key to understanding postfire regenerations patterns and how these influence the spectral response.

To analyze the correlation matrix (Table 3), it was necessary to interpret the relations between the different variables separately. Starting with the diversity indexes analyzed, H and D show positive correlations with the change value in R. On the contrary, the variation in H, D, and R shows a negative correlation with dwarf gorse cover (*Ulex*, *Rosmarinus*, and *Cistus*) and with the MFVC. It should be pointed out that this correlation is more strongly related to *Ulex* cover than the correlation related to dwarf cover or the MFVC. So far, those results show an equal probability in the species participation in a tendency toward an increasing number of species, hopefully affected

Fig. 4 Relative mean fractional vegetation cover (FVC) in 2000 and 2005, calculated as the mean FVC per species divided by the global FVC of all the species surveyed in the 10 sample plots analyzed each year. The change is expressed as the difference (2005–2000) between the data sets

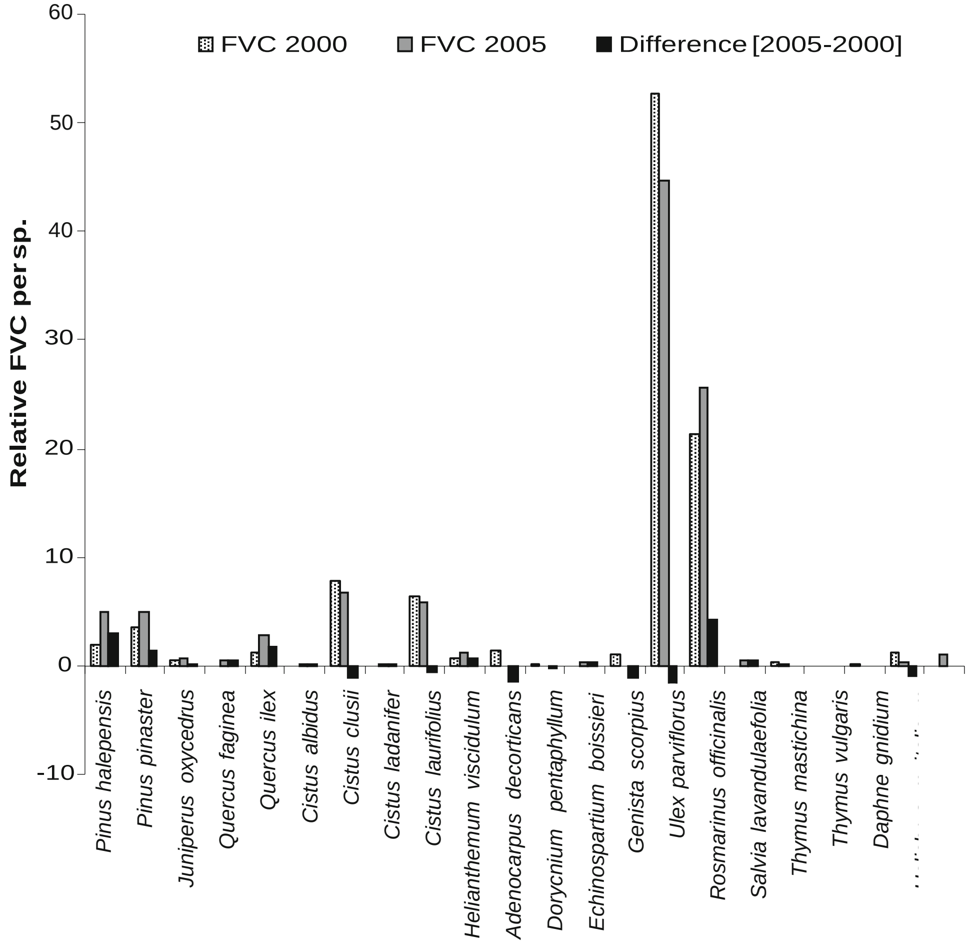
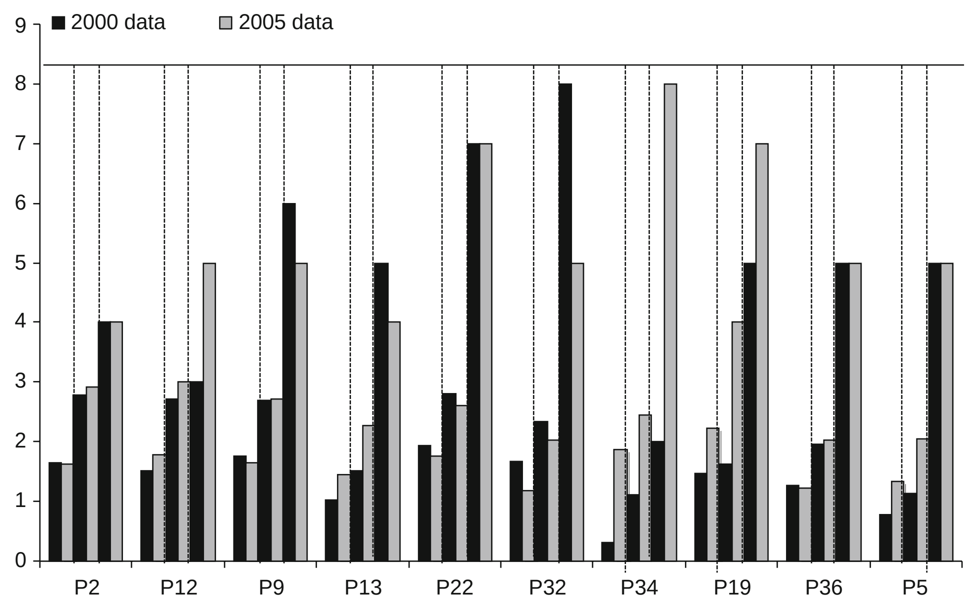


Fig. 5 Diversity index values corresponding to the sample plots measured in 2000 and 2005. Shannon, *H*; Simpson, *D*; richness, *S*



by the decrease in dwarf cover. The NDVI variations were directly correlated with the change in dwarf cover and MFVC. The highest correlations were found between NDVI and *Ulex parviflorus* FVC. Actually, these correlations were even higher and more significant than the ones between NDVI and the dominant species FVC or the

Table 3 Pearson matrix correlation for changes registered from 2000 to 2005 in diversity indexes, *Ulex parviflorus* mean fractional vegetation cover (FVC), sum of the mean FVCs of *U. parviflorus*, *R. officinalis*, and *C. clusii*, and the mean global FVC; *N* = 10

	Simpson (*D*)	Shannon (*H*)	Richness (*S*)	NDVI	FVC (*Ulex, Rosmarinus, Cistus*)	FVC (*Ulex*)	Global mean FVC
Simpson (*D*)	1						
Shannon (*H*)	0.957***	1					
Richness (*S*)	0.699*	0.858**	1				
NDVI	−0.748*	−0.842**	−0.887***	1			
Ulex, Rosmarinus, Cistus FVCs	−0.816**	−0.910***	−0.928***	0.848**	1		
Ulex FVC	−0.832**	−0.930***	−0.892***	0.936***	0.884**	1	
Global mean FVC	−0.578	−0.583	−0.58	0.713*	0.694	0.712	1

NDVI Normalized Difference Vegetation Index

* *p* \ 0.05, ** *p* \ 0.01, *** *p* \ 0.001 (two-tailed)

MFVC. On the contrary, the NDVI was inversely correlated with all the diversity indexes analyzed.

The linear regression analysis established among NDVI, FVC, and the diversity indexes is depicted in Fig. 6. According to Cook's test, there were no significant outliers in the data analyzed. The best linear fit was established between NDVI and *Ulex parviflorus*. So far, these equations cannot generally be extrapolated to other types of ecosystems. However, as shown by these results, monitoring the changes registered in NDVI can be helpful in evaluating the successional changes associated with the dryness of the shrub land.

Discussion

Cluster analysis allowed the distinction of different combinations within the complex mosaic composed of *Ulex parviflorus*, *Rosmarinus officinalis*, and *Cistus clusii*. This codominance reappears in a mosaic shape in Sierra de Húetor (Rivas-Martinez and others 1997). The analysis of a large number of ground sample measurements by clustering facilitated the classification of the main communities found in the burned area.

From the 7th to the 12th year after the fire the floristic composition in the burned area remained statistically unchanged on a global scale, although the global mean FVC increased. These results agree with those obtained by other authors (Mateo Sanz and Mansanet 1982), who observed a clear stage of brushwood invasion or stagnation after the fourth to fifth year. The high recovery values registered for *Rosmarinus officinalis*, *Pinus halepensis*, and *Ulex parviflorus* and the low ones for *Quercus* in the burned area agree with the results predicted by Pausas (1999) based on a regeneration simulation of Mediterranean formations. According to this model, *Ulex*, *Cistus*,

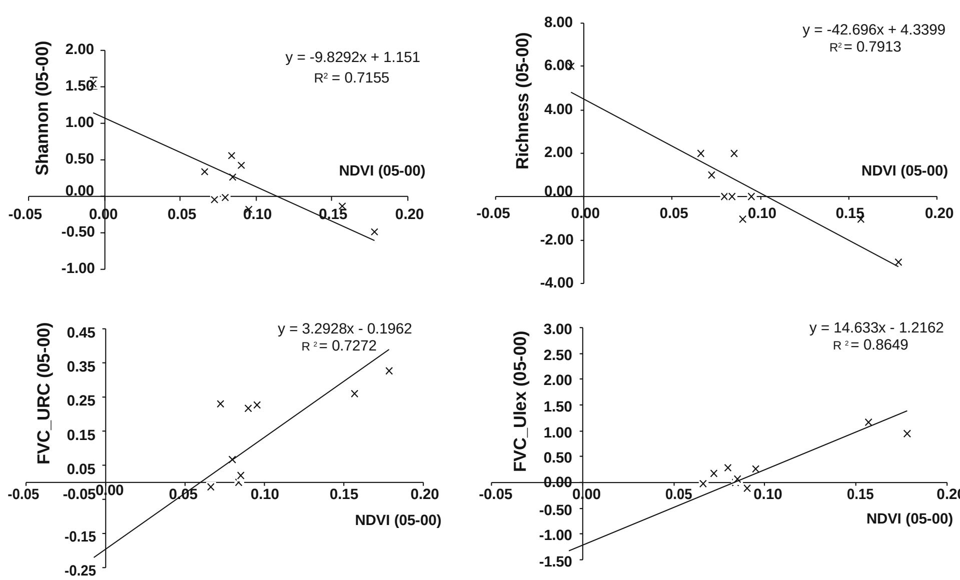


Fig. 6 Linear regression models derived from the correlation between Normalized Difference Vegetation Index (NDVI) and fractional vegetation cover (FVC) of *Ulex parviflorus* (U), FVC of *Ulex parviflorus*, *Rosmarinus officinalis*, and *Cistus clusii* (URC), Shannon index (*H*), and richness (*S*), all calculated as differences ($tB_{2B} - tB_{1B}$) per plot. T2, 2005; T1, 2000. *N* = 10

and *Erica* brushes and other ligneous species, such as *Pinus halepensis*, would have slown down the establishment of *Quercus* regeneration, causing a paraclimax that would stop the succession. Thus, the existence of recurrent fires in previous times could be contemplated. This recurrence would have diminished forest resilience and extended the persistence over time of the brush stage. More than 50% of the mean FVC measurement in the burned area is covered with one dominant species, *U. parviflorus.* Indeed, *U. parviflorus* depicts different trends comparing the quantitative results derived from the mean FVC and the relative FVC. The mean FVC of *U. parviflorus* per plot increased, but it registered a decrease quantified as the relative mean FVC (*U. parviflorus* FVC divided by the global FVC of all species measured in the plot). However, *R. officinalis* and *P. halepensis* registered the highest increase in both cases. Therefore although the dominant species was *U. parviflorus,* the relative increase in *R. officinalis* and *P. halepensis* was higher.

From the 7th to the 12th year after the fire, the floristic composition in the burned area recorded significant statistical changes at the plot level. However, according to the results derived from the diversity indexes estimated, it was not possible to define any specific floristic trend in general. The general linear model performed with repeated measurements allowed for the description of the main species promoting the change. These species were *Ulex parviflorus*, *Rosmarinus officinalis*, and *Cistus clusii.* The results obtained confirm a successional model that reaches a relatively mature stage caused by some species but, at the same time, promotes a slow developmental phase (Pausas 1999). In the 12th year, the profile of the surveyed dwarf gorse in Sierra de Huétor already showed more than 60% dead biomass, although there were green shoots at stem edges. According to Baeza (2001), this profile could correspond to an intermediary phase between phase 2 (8–10 years: mature scrubland with 60% dead biomass) and phase 3 (20–25 years: senescence).

The correlation of variables analyzed highlighted plant community dynamics in the burned area. Diversity indexes Shannon (H) and Simpson (D) and richness (S) exhibit positive correlations. Those results could imply a rising trend in the number of species per plot over time. However, the significant negative correlation between R and H or D and dwarf gorse indicated that these formations have exceeded the phase described by Baeza (2001). According to this author, after a momentary increase in diversity and richness, the brush reaches a period in which these values decrease. Indeed, new species inventoried in 2005 such as *Lithospermum fruticosum* and *Helichrysum italicum* were incorporated into these communities. This reintroduction could occur through the soil seed bank, or due to resprouting, or even because of dispersion from bordering zones.

Diversity indexes and richness are negatively correlated with the indicator species. In other words, the overall diversity dynamics and the indicator species regrowth depend on each other. Thus, it can be affirmed that these species dominate the succession in the community. The decrease in their relative participation allows a significantly higher increase in diversity and richness. This reinforces the above idea: that it is the increase in the number of species which triggers the succession progress. That is, in the current situation, the ecosystem goes farther than the autosuccession process, thanks to a decrease in, and the drying-out of, gorse, making the establishment of other species possible, increasing the diversity value, and reactivating the secondary succession. In contrast, none of the diversity estimators showed any relationship with the overall change in FVC. Therefore, the aging of *U. parviflorus* produces an increased richness in species diversity as shown below by measurements of the Shannon index from the study area.

Finally, the NDVI also displayed a significant negative correlation with all the diversity indexes. The latter result cannot be interpreted without the previous ecological relationships having been established. The correlation among the diversity indexes, the NDVI, and the FVC associated with dwarf gorse shows an indirect relationship. In other words, changes in the dominant species, *Ulex parviflorus*, condition the diversity dynamics. The dryest *Ulex* areas were associated with a decrease in NDVI values and an increase in the number of species. Based on these relationships, it may be possible to design stochastic models for monitoring the succession process established in the area. Thus, it would be possible to predict the regeneration progress of plants depending on the decline of *U. parviflorus*.

Conclusions

Twelve years after the fire, the dominant shrub species in the burned area are *Ulex parviflorus*, *Rosmarinus officinalis*, and *Cistus clusii.* More than 50% of the burned area is covered with one species, *U. parviflorus.* Although the appearance of this species could have initially encouraged the autosuccession process in Sierra de Huétor, the dominance of this species could be slowing down the succession progress. The floristic composition remained unchanged on a global scale, while significant changes were recorded at the plot level. The diversity indexes have not varied on a global scale either but showed erratic variations at the plot level. Nevertheless, the global cover of *U. parviflorus* has begun to slow down slightly, and it is allowing the introduction or recovery of other species. The long-term role of dominant shrub species in autosuccession processes is fundamental for determining the evolution of Mediterranean plant communities.

Changes in the NDVI reveal high correlations with FVC changes in the dominant postfire shrub species. FVC changes in *U. parviflorus* condition the recovery of other species and, therefore, diversity dynamics. Elimination of accumulated *Ulex* dry biomass will decrease wildfire threat and accelerate the restoration of the biodiversity within the area. The NDVI can help to distinguish different successional stages, as it shows a sensitivity to postfire FVC changes and diversity dynamics. For accurate monitoring, analysis of shrub regeneration trends after fire using the NDVI should be repeated at other test sites.

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