Dryness is accelerating degradation of vulnerable shrublands in semiarid Mediterranean environments

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Abstract. Semiarid Mediterranean regions are highly susceptible to desertification processes. This study investigated the influence of increasing climate aridity in explaining the decline in vegetation cover in highly vulnerable gypsum semiarid shrublands of the Mediterranean region. For this purpose, we have used time series of the percentage of vegetation coverage obtained from remote sensing imagery (Landsat satellites). We found a dominant trend toward decreased vegetation cover, mainly in summer and in areas affected by the most severe water stress conditions (low precipitation, higher evapotranspiration rates and sun-exposed slopes). We show that past human management and current climate trends interact with local environmental conditions to determine the occurrence of vegetation degradation processes. The results suggest that degradation could be a consequence of the past overexploitation that has characterized this area (and many others in the Mediterranean region), but increased aridity, mainly related to global warming, may be triggering and/or accelerating the degradation processes. The observed pattern may be an early warning of processes potentially affecting more areas of the Mediterranean, according to the most up to date climate change models for the 21st century.

Keywords: desertification, drought, limiting factors, vegetation cover trends, remote sensing, land degradation, central Ebro valley

1. Introduction

Land degradation is a complex phenomenon that results from the combination of multiple potentially interacting factors (Le Houerou, 1984; Barrow, 1991). The term encompasses various processes that commonly converge in space, including loss of vegetation cover (Hostert et al., 2003; Kefi et al., 2007; Roder et al., 2008), changes in vegetation structure and composition (Schlessinger et al., 1990; Van Auken, 2000, 2009), loss of soil fertility (Schlesinger et al., 1996; Pei et al., 2008; Allington and Valone, 2010), and hydrological and/or aeolian erosion processes (Lal, 2001; Poesen et al., 2003; Ravi et al., 2010).

Although both human and physical factors commonly converge to trigger desertification processes in an area, some parts of the world are more prone to the effects of these processes (Middleton and Thomas, 1992). Arid and semiarid regions are at high risk of land degradation, given the marked abiotic constraints and the typically unstable equilibrium of these ecosystems, which are a consequence of human pressures, water limitations and climate change processes (Le Houerou, 1996; Dregne, 2002). In addition, organic decomposition is slow in these areas, which results in poor soils with low percentages of organic matter (Gallardo and Schlesinger, 1992). Therefore, human and/or climatic perturbations in these areas can trigger progressive loss of both vegetation cover and the productive capacity of the ecosystem. These processes can be irreversible, with restoration to initial conditions impossible because of the level of degradation of the soil (Le Houerou, 2002; Alados et al., 2011).

Among the semiarid regions of the world, those in the Mediterranean are highly prone to desertification (Puigdefábregas and Mendizábal, 1998) as a result of the historical high level of human pressure, and climate modification. The latter includes a substantial decrease in precipitation (Norrant and Douguédroit, 2005; López-Moreno et al., 2009) and a large temperature increase, both of which have markedly reduced the available water resources

(García-Ruiz et al., 2011). Thus, large areas are at high risk of desertification, which could be triggered or/and reinforced by the climate change processes affecting the region (Millán et al., 2005).

The central Ebro valley (northeast Spain) is the northernmost semiarid region of Europe. This area has a large water deficit as a consequence of the low level of precipitation and the high evapotranspiration rates (Cuadrat et al., 2007). It has a long history of high livestock pressure including centuries under a transhumant system that maintained large sheep flocks (Pinilla, 1995; Lasanta et al., 2011). Although livestock pressure has drastically decreased in the region in recent decades (Ameen et al., 2011), there are still large areas with degraded rangelands characterized by low vegetation cover (Pueyo and Alados, 2007a; Pueyo et al., 2009). Moreover, potential evapotranspiration (PET) and drought severity in the area have increased in recent decades (Vicente-Serrano and Cuadrat, 2007; Vicente-Serrano et al., 2010). In addition, the particular lithological characteristics of this region have exacerbated its vulnerability to degradation. The central Ebro basin has the largest surface gypsum outcrops in Spain (approximately 1.9 million ha (Navas, 1983)), which produces soil conditions that are very restrictive for the development of natural vegetation.

Gypsum soils commonly have very specialized flora adapted to the limitations imposed by the chemical and physical characteristics of these soils (Mota et al., 2004; Ferrandis et al., 2005). The chemical constraints of such soils include an unbalanced ionic concentration, with an excess of sulfates and Ca (Boukhris and Lossaint, 1975; Pueyo et al., 2007), and low concentrations of other nutrients including N, P and K, which limit the growth and survival of nonadapted plants (Guerrero-Campo et al., 1999). The physical restrictions are related to the occurrence of crusts that often form on the surface of soils with a high gypsum content, which restrict the germination and establishment of plants (Escudero et al., 1999). Moreover, gypsum soils have poor retention of moisture because of their low water potential (Boukhris and Lossaint, 1975). These lithological characteristics make gypsum ecosystems especially vulnerable to human perturbation and overexploitation (Pueyo et al., 2008), but probably also highly susceptible to the current aridification processes. The fragility and low resilience of gypsum soil plant communities (Pueyo and Alados, 2007b) may enhance the likelihood of irreversible degradation as a consequence of climatic change.

Iberian gypsophile plant communities are a European conservation priority (European Community, 1992), and consequently the central Ebro valley is included under the Habitats Directive of the European Commission in the network of Special Protection Areas (SPAs) and the Natura 2000 network, which are at the center of the European Union nature and biodiversity policy. The importance of this site highlights the urgency of assessing potential degradation processes in the region, and because of the limitations (climatic and edaphic) of the study area it is particularly urgent to assess whether land degradation processes are a response to current climate trends. The Identification of vegetation changes could provide an early indication of processes that may affect other semiarid areas of the Mediterranean region, given the projections of current climate change models for the 21st century (Weiss et al., 2007; Giorgi and Lionello, 2008; Evans, 2009; García-Ruiz et al., 2011).

Assessing potential degradation processes is difficult, as it is necessary to have a broad temporal perspective to assess the magnitude of the processes, and a spatial perspective covering large areas. Desertification processes have been monitored at various spatial and temporal scales using techniques including experimental plots and field samples (e.g. Lal, 1996; Paracchini et al., 1998; Martínez and Zinck, 2004). These approaches enable detailed knowledge of the processes that drive vegetation degradation, including soil–plant–atmosphere interactions (Paracchini et al., 1998), as well as the factors that constrain degradation, including biotic interactions among species (Maestre, 2004) and vegetation tolerance to water stress (Pueyo and Alados, 2007a). However, field samples and experimental plots are very local and spatially fragmented, do not provide a broad temporal perspective, and in particular do not clarify the spatial extent and magnitude of degradation processes.

Remote sensing provides useful information and a broader perspective on degradation processes. Sequential series of aerial photographs have been used to assess changes over large areas (Lindqvist and Tengberg, 1993; Alados et al., 2003; Bruelheide et al., 2003; Hirche et al., 2011). Nevertheless, in arid and semiarid areas, which are characterized by low percentage vegetation cover, it is difficult to assess vegetation changes over long periods from the subjective interpretation of aerial photographs, as the changes may be too small to be visually distinguished from the air. In contrast, satellite images provide spatially distributed quantitative and objective data for monitoring land degradation and desertification (e.g. Hill et al., 1995; Nicholson et al., 1998; Hostert et al., 2001; Evans and Geerken, 2004). Long time series (from the beginning of the 1980s to present) of satellite images are available because some of the earth observation programs (e.g. NOAA–AVHRR, Landsat) have been continuously recording information throughout that period, enabling a broad and updating perspective of current degradation.

Few studies have analyzed land degradation processes in Mediterranean rangelands using satellite images. Some studies have used NOAA–AVHRR images (Evans and Geerken, 2004; Geerken and Ilaiwi, 2004; Hill et al., 2008; Del Barrio et al., 2010), but most analyses have been based on Landsat data (Hill et al., 1998; Hostert et al., 2003; Symeonakis et al., 2007; Roder et al., 2008), as its spatial resolution (30 m compared with 1100 m for AVHRR images) allows the identification of processes with the degree of detail necessary, given the great heterogeneity of Mediterranean landscapes.

No studies in recent decades have used remote sensing images to analyze degradation processes in gypsum rangelands under highly semiarid conditions. The objectives of this study were to identify recent vegetation change in the gypsum shrublands of the central Ebro valley, and to determine the magnitude of vegetation change and its spatial variability. In addition, we investigated the role of various environmental factors potentially determining the observed changes, and focused on the possible influence of increasing aridity in explaining the general evolution and spatial patterns of land degradation in the region. Given the extreme vulnerability of vegetation communities in the region, our findings may be an early indication of forthcoming land degradation processes in dry Mediterranean environments.

2. Study area

The study area corresponds to the central Ebro valley, northeast Iberian Peninsula (Fig. 1). This area was selected on the basis of the presence of shrublands located on gypsum soils within the area covered by Landsat scene 199-31. The area comprises 75,791 ha with altitudes ranging from 200 to 400 m a.s.l., and its main characteristic is the presence of soils having high concentrations of gypsum. The gypsums of the area were formed during the Miocene, as chemical precipitates in the lake that occupied the center of the basin (Pellicer and Echeverría, 1989). The geomorphology of the area is characterized by hillocks and silted plain valleys (Peña et al., 2002) that are mainly cultivated with winter cereals (barley and wheat).

The area is subject to Mediterranean influences, and typically has a summer drought and semiarid conditions (mean annual temperature 15.0 °C; mean annual precipitation 318 mm). The climate is continental, which explains the extreme winter and summer temperatures and the very high temperature range.

The average precipitation is low (318 mm), and there is a negative water balance (precipitation minus evapotranspiration) as a consequence of the high PET, which reaches 1300 mm in some parts of the valley. Moreover, the high temporal variability in precipitation is a major limitation for vegetation growth, as severe droughts are frequent (Vicente-Serrano and Beguería, 2003; Vicente-Serrano and Cuadrat, 2007).

The gypseous soils in the area are poorly developed, have low nutrient and organic matter content, and are alkaline. The low nutrient content is a consequence of the low ionic transference capacity of the soil. The extreme mobility of gypsum in the profile and the high crystallization pressure cause a high degree of soil porosity, which results in high infiltration capacity, low soil water retention, and high rates of direct evaporation (Desir, 2001).

The vegetation communities are characterized by low vegetation cover, and are dominated by shrubs including *Rosmarinus officinalis* L., *Linum suffruticosum* L. and *Salvia officinalis* L., and the gypsophytes *Lepidium subulatum* L., *Gypsophila struthium* Loefl. subsp. *hispanica* (Wilk.) G. López and *Ononis tridentata* L. (Braun-Blanquet and Bolòs, 1957). Some community differences occur as a function of the degradation level; in the most degraded areas the vegetation density is lower and more dominated by chamaephytes including *Thymus vulgaris* L. and the gypsophyte *Helianthemum squamatum* (L.) Pers. (Braun-Blanquet and Bolòs, 1957; Guerrero-Campo, 1998).

To understand the abundance of shrublands with sparse coverage, three main factors need to be considered: i) the climate aridity; ii) the poorly developed and growth-limiting gypsum soils; and iii) the historical human land use over past centuries. The original open forests composed of

Juniperus thuriphera L., *Pinus halepensis* Mill. and *Quercus coccifera* L. were cleared several centuries ago to create agricultural lands and, particularly, rangelands to maintain large transhumant sheep flocks, which were the base of the economy of the region for many centuries (Ruiz and Ruiz, 1986). From the beginning of the 20th century the transhumant system declined and the flocks were reduced markedly (Pinilla, 1995). In the second half of the 20th century the development of agricultural mechanization led in several cases to unsuitable gypsum land being cultivated, but the poor productivity of these areas and the agricultural policies of the European Community resulted in their abandonment. The majority of the plain valleys are still cultivated, but the hillocks are covered by natural plant communities. Sheep grazing is the only form of exploitation of these plant communities, but the livestock pressure has declined abruptly in recent decades and is presently low in the study area (Pueyo, 2006; Ameen et al., 2011). There are also some small areas that are protected because of their ornithological and ecological values, and within which human activities are controlled (Pueyo et al., 2009).

3. Methodology

3.1. Data sources

The main data sources used in the study were two time series of Landsat images for the period 1984–2009, obtained from the sensors Landsat 5-TM and Landsat 7-ETM+; these correspond to Landsat scene 199-31. The database comprised 31 images; 18 from a summer time series and 13 from a spring time series (Table 1). The two time series were used to identify possible differences in degradation as a function of seasonal differences in vegetation activity, and to assess with more robustness any spatial and temporal degradation pattern. The data were processed using a protocol that included calibration and cross-calibration of the TM and ETM+

images, atmospheric correction using a radiative transfer model that included external atmospheric information, a nonLambertian topographic correction to avoid errors caused by differences in the illumination conditions, and a relative normalization between dates. The procedure allowed accurate measurements of physical surface reflectance units to be obtained. The correction applied to the images guaranteed the temporal homogeneity of the dataset, the absence of artificial noise caused by sensor degradation and atmospheric conditions, and spatial comparability between different areas, given the accurate topographic normalization applied. Details of the correction procedure applied to the images, and a complete description of the dataset and its validation can be found in Vicente-Serrano et al. (2008).

In addition to the Landsat data series we used complementary information sources including the land cover map for 1978, which was produced six years before the first Landsat TM-image was acquired (MAPA, 1978). The Spanish National Forestry Map for 2006 (MMA, 2006) and the geological map of Aragón (SITAR, 2007) were also used to accurately identify the shrublands on gypsum substrate. We also used a digital terrain model at a resolution of 30 m to obtain topographical variables. Digital maps of annual PET and precipitation were obtained from the Digital Climatic Atlas of Aragón at a spatial resolution of 1000 m (Cuadrat et al., 2007).

The meteorological records (daily precipitation, and maximum and minimum temperature) from the city of Zaragoza, which is located in the center of the study area, were also used to assess recent climate variability and trends. This station was selected because it provided data of high quality. The quality of the climate data was checked and temporal homogeneity was controlled (see details in Vicente-Serrano et al., 2010b and Kenawy et al., 2011).

We followed a detailed procedure to identify gypsum rangelands not subject to intensive anthropogenic transformation between 1984 and 2009, the period of the study. Within the area of the 199-31 Landsat scene we selected the gypsum areas using the Geologic Map of Aragón. To identify the categories of shrubs in these areas at the beginning and end of the study period we used the 1978 land use map and the Spanish National Forestry Map of 2006, respectively. After 2006 some shrublands were cleared for expansion of urban and commercial areas; we visually identified these areas in the images, manually digitized them, and removed them from the analysis.

3.2. Assessing vegetation cover from the Landsat series

Remote sensing images have been widely used to analyze vegetation dynamics and land degradation processes (e.g. Pickup et al., 1993; Hill and Schutt, 2000; Kepner et al., 2000; Collado et al., 2002; Hostert et al., 2003; Wessels et al., 2004). The utility of remote sensing for vegetation monitoring is based on the response of the vegetation cover to solar radiation in the visible and near-infrared regions of the electromagnetic spectrum. Radiation in the visible range is largely absorbed by vegetation during photosynthesis, while near-infrared radiation is mostly reflected because of the internal structure of the leaves (Knipling, 1970). Consequently, high vegetation activity and cover is characterized by low reflectivity of visible radiation and high reflectivity of near-infrared radiation.

The high spatial (30 m) and spectral (6 bands in the reflective spectrum) resolution of Landsat images make these data very suitable for analysis of changes in vegetation cover at a high level of spatial detail (Cohen and Goward, 2004). Long time series of images are available because the satellite (Landsat-5) was launched in 1984, providing more than 25 years of time series that enable highly detailed analysis of land surface processes. As Landsat images record spectral information for visible and near-infrared reflected radiation, quantitative evaluation of vegetation activity it is often undertaken using spectral vegetation indices (Bannari et al., 1995), the most common of which is the normalized difference vegetation index (NDVI). The NDVI exhibits a strong relationship with vegetation parameters including the green leaf area index (Baret and Guyot, 1991; Carlson and Ripley, 1997), green biomass (Tucker et al., 1983), and fractional vegetation cover (Duncan et al., 1993; Gillies et al., 1997). Nevertheless, there have been various studies reporting that these kinds of indices have limitations in their application to areas with a low percentage of vegetation cover and dominance of the soil background (Hostert et al., 2003; Röder et al., 2008); background soil properties, especially high reflectivity, affect the NDVI (Huete, 1988). This is particularly critical in the study region as gypsum soils are strongly reflective. For this reason we followed an alternative approach to quantifying the vegetation cover, using spectral mixture analysis (SMA). In areas of sparse vegetation SMA reduces the problems associated with the use of vegetation indices, and enables quantitative estimates of the percentage of photosynthetically active vegetation cover in each pixel of the image (Smith et al., 1990; Roberts et al., 1993). Various studies have shown that SMA outperforms the vegetation indices in areas with sparse vegetation cover (Elmore et al., 2000; Camacho-De Coca et al., 2004). Thus, in a recent study Sonnenschein et al. (2011) assessed various vegetation indices and the SMA for monitoring vegetation trends in semiarid Mediterranean rangelands, and showed that the SMA was superior. They stressed that at low levels of vegetation cover, as in the central Ebro valley, the differences between the NDVI and SMA increase, which would affect the assessment of gradual changes and vegetation trends in this type of environment.

The SMA assumes that the spectrum of each pixel is a linear combination of a small number of pure spectral signatures, which are denominated endmembers (Smith et al., 1990). The most common case is the presence of nonpure pixels; the SMA allows the proportion of each surface component in the pixel to be estimated. The linear SMA model is described according to the following equation (Settle and Drake, 1993):

$$\rho_{\lambda} = \sum_{j=1}^{n} F_j * \rho_{\lambda,j} + E_{\lambda} \quad \text{and} \quad \sum_{j=1}^{n} F_j = 0 ,$$

where ρ_λ is the reflectance of band λ, F_j is the percentage corresponding to the endmember j, $\rho_{\lambda,j}$ is the reflectance of the endmember j in band λ, n is the number of endmembers, and E_λ is the residual error in band λ.

The equation is solved by least-squares, assuming that the sum of the fractions of each endmember equals 1 for each pixel. The number of endmembers must be less than or equal to the number of existing spectral bands, to avoid autocorrelation between the bands (Small, 2004). The selection of endmembers is the most critical step in the process, and can be achieved using a spectral library or the spectral information contained in the image. In this study we assumed that there were two main components determining the reflectance of each pixel: the bare gypsum soil and the active vegetation cover. Other possible land coverages are very punctual in the study area. Therefore, we selected two endmembers, corresponding to bare soil and 100% photosynthetically active vegetation. For this purpose we used the image of 18 March 2009. Using visual identification and field work in April 2009 for validation, we identified two areas in the image corresponding to 100% active vegetation and bare soil, respectively. These areas were manually digitized in the image, and the spectral signatures of the endmembers were determined. These signatures were applied as the universal set of reference endmembers required for multi-date SMA (Camacho-De Coca et al., 2004; Sonnenschein et al., 2011), and were used to estimate the percentage vegetation cover for each pixel in the series of 31 images.

3.3. Analysis of vegetation trends

There are various techniques used to identify and analyze vegetation and landscape changes using satellite images (Singh, 1989; Mas, 1999; Lu et al., 2004; Deng et al., 2008). In this study a standard statistical test was used to determine trend significance. For this purpose a nonparametric coefficient (Mann-Kendall tau) was selected because it is more robust than parametric coefficients and does not assume normality of the data series (Lanzante, 1996). The values of tau measure the degree to which a trend is consistently increasing or decreasing. In our study positive values of tau indicated a trend of increasing vegetation cover, and negative values decreasing cover. Statistically significant trends were defined as those below the threshold $p < 0.05$ in each of the 496,384 spatial series of 30×30 m of grid size for the summer and spring seasons.

An important shortcoming of this method is that the nonparametric tau coefficient only showed the presence of significant trends in the series of vegetation cover, but not the magnitude of change. A small but sustained change can result in a higher coefficient than a bigger but abrupt change. To identify the areas that underwent the greatest changes in percent vegetation cover we used a regression analysis between the series of time (independent variable) and the temporal vegetation cover series (dependent variable). The results yielded one model for each spatial series of 30×30 m grid size, and took the form $y = m \times t + b$. The regression constant (b) and the coefficient (m) were calculated using a least-square fit, with the Landsat acquisition years (t) as the independent variable. The slope of each model (m) indicated the change in percentage vegetation cover (% change per year), with greater slope values coinciding with greater vegetation cover. Therefore, we first determined the areas showing a positive trend in vegetation cover, using a nonparametric correlation, and then analyzed the magnitude of change using linear regression.

3.4. Analyzing climate variability processes

To assess the evolution of the main climatic factors limiting vegetation development in the region we analyzed trends in various climate variables. Trends in annual precipitation, temperature and PET between 1970 and 1999 were obtained from the series for Zaragoza, and were analyzed using the Mann-Kendall tau statistic. PET was obtained using the Hargreaves equation (Hargreaves and Samani, 1985), which only requires data on the maximum and minimum temperatures, and extraterrestrial radiation. At monthly and annual timescales, the PET estimates derived from this equation are similar to those obtained using the Penmann-Montheith method (differences $< \pm 2$ mm day^{-1}; Droogers and Allen, 2002). Martínez-Cob (2002) and Martínez-Cob and Tejero-Yuste (2004) showed that the Hargreaves method provides very robust estimates of PET in semiarid areas of the central Ebro valley, and in the mountainous areas to the north of the study area (López-Moreno et al., 2009b). A climatic water balance (precipitation minus PET) was also calculated to enable assessment of trends in water availability.

We also calculated a synthetic drought index (the standardized precipitation evapotranspiration index, SPEI; Vicente-Serrano et al., 2010c). This index enables drought duration and magnitude to be objectively identified, and facilitates comparison of temporal changes in drought severity.

3.5. Factors determining spatial differences in land degradation trends

We analyzed the role of six factors in the observed changes of vegetation cover: i) terrain slope; ii) potential incoming solar radiation; iii) annual precipitation; iv) annual PET; v) average vegetation cover; and vi) rain use efficiency. We used two topographical variables: the terrain slope and the potential incoming solar radiation. The potential incoming solar radiation was computed quantitatively using the aspect derived from a terrain model (Pons and Ninyerola, 2008) using the MiraMon Geographical Information System (Pons, 2011). Higher levels of incoming solar radiation were recorded on southern slopes than on northern slopes. The spatial distribution of annual precipitation and PET was obtained for each pixel from the layers of the Digital Climatic Atlas of Aragón (Cuadrat et al., 2007). We also quantified the average vegetation cover per pixel during the period 1981–2009, to assess whether areas with less vegetation cover were more affected by degradation processes. We also estimated the rain use efficiency, which is a standard measure widely used to assess the health of ecosystems and their productive capacity (Le Houerou, 1984). We obtained an estimate of the mean Rain Use Efficiency (RUE) from the quotient between the average vegetation cover and the average precipitation. As different biomes tend to converge to a maximum rain use efficiency (Huxman et al., 2004), lower values of this variable are indicative of a decreased capacity for generating green biomass per unit of water, and are consequently indicative of more extreme degradation conditions (Le Houerou et al., 1988; Illius and O'Connor, 1999; Paruelo et al., 1999).

Trends in vegetation cover assessed using the Mann-Kendall tau statistic were classified as positive or negative (according to the sign of the statistic), and significant or nonsignificant (according to the p value of the test). Therefore, four groupings summarized the vegetation changes in summer and spring: i) negative and significant trend; ii) negative trend; iii) positive trend; and iv) positive and significant trend. For each of the Landsat pixels in each group we determined the values of each of the six factors noted above. We compared graphically the values of the factors in each of the four trend groups, and assessed the statistical differences among the groups using one-way analysis of variance (ANOVA); i.e. six ANOVAs for each of the two seasons (spring and summer). The Tamhane posthoc contrast, which does not require to assume homogeneity in the variance of the factors among the trend groups, was used to identify the statistically significant differences among the trend groups for each of the six factors. The contribution of the various factors in explaining the spatial differences in vegetation trends was estimated using predictive discriminant analysis (PDA). PDA is used to explain the value of a dependent categorical variable based on its relationship to one or more predictors (Huberty, 1994). Given a set of independent variables, PDA attempts to identify linear combinations of those variables (topographic, climatic and other) that best separate the groups of cases of the dependent variable. These combinations are termed discriminant functions (Hair et al., 1998). The procedure automatically chooses a first function that separates the groups as much as possible. It then chooses a second function that is not correlated with the first function and provides as much further separation as possible. This procedure continues, with further functions being adding until the maximum number of functions is reached, as determined by the number of predictors and categories in the dependent variable. The PDA enabled assessment of which predictor variables contributed to most of the inter-category differences of the dependent variable (the four trend groups).

4. Results

4.1. Patterns of vegetation cover in summer and spring

The percentage of vegetation cover was higher in spring than summer in the years analyzed (41% and 24.3%, respectively). Although lower levels of vegetation cover were generally recorded in summer, the temporal variability in this season (average coefficient of variation 0.25) was higher than in spring (0.2). Nevertheless, independently of season the areas with low levels of vegetation cover tended to have greater interannual variability in the percent vegetation cover

(Fig. 2). The relationship between the average vegetation cover and its coefficient of variation, quantified using the Mann-Kendall tau coefficient, was significant in both summer and spring, and clearly nonlinear. In areas with a cover of < 20%, the coefficient of variation tended to be very high in summer. In addition, there was a high degree of agreement in the spatial distribution of the percent vegetation cover in summer and spring (tau = 0.582). This indicates that independently of the seasonal differences in magnitude, areas with high levels of vegetation cover in one season also tended to have more vegetation in the other season (Fig. 2c). Areas that were more variable in spring also tended to be more variable in summer (Fig. 2d).

4.2. Temporal evolution of the vegetation cover

The temporal evolution showed a decrease in the average percentage vegetation cover in both summer and spring during the analysis period, but only in summer was the decrease statistically significant (tau = –0.29, $p < 0.05$; Fig. 3). The magnitude of change in the total cover in summer decreased by an average of 0.11% per year, which implies that between 1984 and 2008 the study area lost an average of 2.86% of the vegetation cover. In spring the trend was not statistically significant, but the linear regression analysis indicated an annual decrease of 0.16%. Table 2 shows the total surface area and the percentage with significant and nonsignificant positive and negative trends. In spring, the areas with no significant change dominated, occupying 96.8% of the total surface. Only in 2.2% of the surface area was there a significant decrease, while 1% showed a positive and significant increase in vegetation cover. Nevertheless, analysis of the signs indicated that negative changes dominated (61.3%) positive changes (38.7%). In summer, the dominant negative pattern was even more evident, with 72.1% of the surface area having negative tau coefficients between 1984 and 2008; it is notable that 14.7% of the study area showed a significant negative trend in vegetation cover. In contrast, only 27.9% of the surface area showed positive coefficients, and only 1.7% of the total area showed a positive and significant trend. As the two time series covered different periods (spring, 1989–2009; summer, 1984–2008) we also analyzed the trends in summer for a common period (1989–2008) to assess whether the differences were related to the period selected for analysis. This analysis indicated that negative trends dominated in the study area in summer between 1989 and 2008, with percentages similar to those observed for the 1984–2008 period.

Table 3 shows a cross-tabulation analysis of the spatial relationships between the trends observed in spring and summer. Areas with no significant trends dominated, but 11.3% of the surface area had negative but nonsignificant changes in spring, and negative but statistically significant changes in summer. The spatial association of the spring and summer trends was significant (χ^2 = 87.2, $p < 0.05$), which indicates that the results obtained in summer were not spatially independent of those found in spring. Nevertheless, the coefficient of contingency (cc), which measures the magnitude of the association, was not very high (cc = 0.29). When the period 1989–2008 was used to assess the trends in summer a similar association was observed (χ^2 = 95.2, cc = 0.295), which indicates that the results were not determined by selecting different periods for spring and summer.

The magnitude of the observed changes was assessed using linear regressions. Figure 4 shows the spatial distribution of the observed magnitude of change. In summer, the negative changes predominated; the main exceptions occurred in the hillocks to the southwest of the study area. There was no clear spatial pattern in the spatial distribution of the negative trends, although greater vegetation changes occurred in the southwest and in the shrublands to the north of the Ebro River. In spring, the spatial pattern was similar to that observed in summer, but the spatial heterogeneity of the changes was much greater. In general, the magnitude of the changes was small. In both summer and spring over most of the study area the changes in percentage vegetation cover were < ± 0.5% year^{-1}. Nevertheless, negative coefficients dominated in summer and spring. These results highlight the gradual character of the changes in these kinds of environments, as the maximum percentage change in the analyzed period ranged from 20–30% in very few areas, whereas the decrease in vegetation cover in most of the study area ranged from 0–5%.

The spatial association in the magnitudes of change observed in summer and spring was similar to those analyzed for trends (Fig. 5). The relationship between the spatial distribution of the magnitudes of change in summer and spring was statistically significant but not strong (tau = 0.27, $p < 0.05$). The coefficient for summer in the 1989–2008 period (tau = 0.26, $p < 0.05$) was similar.

4.3. Observed climate variability and trends

In the last four decades the central Ebro valley has shown a clear trend to drier conditions, mainly as a consequence of warming processes (Fig. 6). No significant trend has been found for annual precipitation, although interannual variability is very high, with some years having < 200 mm annual precipitation (e.g. 1995 and 1998). In contrast, the increase in temperature has been very evident since 1970, with an increase of approximately 1.5 °C having occurred in the last four decades. This has dramatically increased the PET rates in the region from approximately 1150 mm in the decade of 1970 to approximately 1250 mm in the decade of 2000. This has caused a decrease in water availability, as evidenced by the negative and statistically significant (tau = –0.22, $p = 0.048$) evolution of the climatic water balance.

In addition, the evolution of the SPEI clearly shows that, whereas in the decades of 1970 and 1980 the drought episodes were not excessively long or severe, in the decades of 1990 and 2000 there was a succession of severe drought episodes (Fig. 6) that exacerbated vegetation stress conditions in the region.

4.4. Factors affecting spatial patterns of vegetation trends

Figure 7 comprises six box-plots showing the values of the six factors analyzed to assess their impacts on the observed vegetation changes. The values are shown as a function of the identified vegetation trends in summer. Application of the Tamhane posthoc contrast test to each of the one-way ANOVAs indicated that most of the factors showed significant differences as a function of the vegetation trend groups (see Appendix). The terrain tended to have a smaller slope in areas with negative and significant trends. In general, the box-plots show that areas with more limiting water availability (i.e. higher levels of incoming solar radiation, less annual precipitation or greater PET) tended to show more negative trends, with statistically significant differences relative to areas with positive trends. For example, the average level of incoming solar radiation tended to be higher in areas with negative trends. Thus, solar radiation tended to increase the PET rates and to reduce the availability of water in the soil. In addition, the average annual precipitation was significantly lower and the PET significantly higher in areas with negative trends than in areas with positive trends. Although the values of the variables were not different between those areas with positive trends and (in some of the cases) those with positive and significant trends, the differences were significant between areas with negative trends and those with the strongest negative and significant trends. This highlights the possible role of these limiting factors in explaining the rates of decrease in the vegetation cover. Further, the parameters indicative of the health of the ecosystems (i.e. the average vegetation cover and the rain use efficiency) also showed significant differences in the vegetation trend categories. Thus, the decrease in cover mainly affected areas with a low percentage vegetation cover and low efficiency in the use of rainwater. The results for spring were quite similar to those in summer, and the differences in the values of the predictors among the vegetation trend groups were even more marked. Decreases in vegetation cover were also recorded in gently sloping areas, on south-facing slopes, and in dry areas. There were also large differences in vegetation cover and rain use efficiency between those areas having positive trends in the percentage of vegetation cover and those showing negative trends.

These results indicate that areas with less vegetation cover as a consequence of environmental constraints (mainly water availability, but also potentially because of poor soils, low nutrients and lower vegetation productive capacity, as shown by the rain use efficiency) were those that predominantly showed a greater decrease in vegetation cover. In contrast, areas with greater water availability (which commonly had more dense cover and made more efficient use of the available water) showed a predominant positive trend in vegetation cover.

The application of a predictive discriminant analysis enabled the relative importance of the various predictors to the vegetation cover trends to be identified. The first discriminant function accounted for 94.6% and 96.8% of the variance for summer and spring, respectively. For summer, the first function mainly represented factors that limited water availability: incoming solar radiation and the PET were positively correlated with function 1. In contrast, function 2 represented the vegetation cover and the RUE (Table 4). Function 3 was associated with an eigenvalue < 0.01, and was not further analyzed. The distribution of the centroids of the trend categories as a function of the two discriminant functions showed a clear separation of the groups, mainly with the function 1 axis (Fig. 8). This function discriminated between areas with negative trends (positive scores) and those with positive trends (negative scores). Positive values of function 1 (which mainly represented positive values of PET and solar radiation, and low values of RUE, vegetation cover and precipitation) are indicative of land degradation processes associated with a decrease in vegetation cover. Moreover, land conditions characterized by negative values of function 1 (low levels of solar radiation and PET, and higher levels of precipitation, vegetation cover and RUE) showed dominant positive trends in the vegetation cover. Function 2, the predictive capacity of which is much lower, separated groups with positive and significant trends from the remaining groups. The centroid of positive and significant trends in function 2 was negative in value. The correlations of RUE and vegetation cover with function 2 were also negative, which indicates that strongly positive trends were favored in areas with well-developed vegetation cover and a high capacity to generate biomass from available water.

A similar clear separation between the trend groups was obtained in spring, as function 1 separated positive (positive values of the function) and negative (negative values of the function) trends. Given the structure matrix of the discriminant analysis, the positive trend of vegetation cover in spring was favored by high values of vegetation cover, RUE and precipitation.

Precipitation was also important in explaining the difference in function 2 between trends that were positive and those that were positive and significant. Thus, whereas function 1 did not discriminate between the two groups of negative trends, function 2 (in which precipitation had the greatest role) separated both groups; those that received less precipitation and those that underwent the main decline in vegetation cover.

These results highlight the marked influence of the analyzed biotic and abiotic factors in explaining the spring and summer trends in the vegetation cover, and the extreme vulnerability of the most arid and water stressed areas to the effects of degradation processes in the region.

5. Discussion

Observed trends in the vegetation coverage

A gradual decrease in vegetation cover was the dominant trend in gypsum shrubland areas of the central Ebro valley, with an average loss of 2–3% over the entire study area for the analysed period. The decrease was independent of the seasonality of vegetation cycles, as the same pattern was found in spring and summer. The main result from this study was the identification of areas in which there has been a decrease in vegetation cover over the last 30 years. Although this area has very homogeneous landscape and vegetation characteristics, spatial differences in the trends in vegetation cover have been substantial and determined by a range of biotic and abiotic factors. Thus, the vegetation cover in some areas is progressively recovering, whereas in others there was a clear decrease during the study period. A novel aspect of the study was consideration of the natural seasonality of the vegetation cover. The use of time series in spring and summer enabled detection of major seasonal differences in vegetation change. The density of vegetation cover was higher in spring as a consequence of the presence of more therophytes than were present in summer; this was a result of the greater availability of water following winter, which aided seed germination. Vicente–Serrano (2007) showed that during early spring in the central Ebro valley the influence of climatic conditions on the vegetation cover is less important than in summer, because in most years the vegetation has sufficient water for physiological processes. In contrast, the climate conditions in summer are more limiting, and often cause plant mortality, loss of biomass, and consequently a reduction in vegetation cover. Therefore, it is not surprising that the greatest decrease in vegetation cover occurs in summer, when vegetation communities are near the limit of their physiological needs, and when changes in water availability rarely exceed these needs.

Observed climate change processes

Climate change explains many changes occurring in the vegetation cover in various ecosystems worldwide (e.g. Breshears et al., 2005; Ciais et al., 2005). Thus, the dominant negative trends in the vegetation cover observed in the central Ebro valley may be related to an increase in aridity in recent decades. Climate trends during the last four decades generally show that warming has occurred in the region. This is the dominant pattern observed in the Iberian Peninsula (Brunet et al., 2006) and other areas of the Mediterranean basin (Bethoux et al., 1998; Repapis and Philastras, 2004; Camuffo et al., 2010). Moreover, climate warming tends to have had a greater effect on the summer temperature extremes in the region, resulting in an increase in the severity of summer heat waves and the duration of warm spells (Kenawy et al., 2011b). This has occurred during a period of more acute water stress, which has increased the effects of the marked increase in evapotranspiration that has been observed in recent decades. Annual precipitation has not decreased in the last four decades, which indicates that the increase in aridity in the region, evidenced by the significant negative trends in the climatic water balance, has been driven by the trends in evapotranspiration. Nevertheless, the evolution of precipitation has not been seasonally uniform and the precipitation regime has changed considerably, which could explain some of the trends in vegetation cover observed in the region. López-Moreno et al. (2010) showed that there has been an increase in the duration of dry spells (consecutive days without precipitation) in the central Ebro valley, and that this has occurred in winter, spring and summer. In addition, a large decrease in precipitation has been recorded in late winter–early spring period in the region

(February and March) (González-Hidalgo et al., 2011), which indicates that there has been a change in the precipitation regime because the major rainfall period has shifted from winter to autumn (De Luis et al., 2010); a positive trend in precipitation in autumn has been recorded (González-Hidalgo et al., 2011). Therefore, although a decrease in annual precipitation has not been identified in the region, the observed seasonal changes and the occurrence of longer periods without rainfall may have increased water stress in the vegetation, and exacerbated the limits imposed by the increased evapotranspiration demand. Thus, the observed decrease in precipitation during late winter and early spring may explain the generalized negative trends observed in the spring vegetation cover in recent last decades, as most soil water accumulates in winter months, when vegetation activity and evapotranspiration rates are low (McAneney and Arrúe, 1993).

The increased climate aridity in the region has been accompanied by a marked increase in drought severity. Although drought is a common phenomenon in the Mediterranean region generally, and the central Ebro valley in particular, the magnitude and duration of drought episodes observed in the decades of 1990 and 2000 had not been observed since the decade of 1940 (Vicente-Serrano, 2006; Vicente-Serrano and Cuadrat, 2007), and the consecutive sequences of extreme drought episodes during the former decades have not occurred since the decade of 1900 (Vicente-Serrano, 2005). Drought has commonly been considered to be a major factor triggering desertification processes, and several studies have highlighted the importance of drought episodes in explaining the occurrence of serious degradation (e.g. Nicholson et al., 1998; Pickup, 1998). Drought itself cannot be considered a factor explaining degradation trends, given the common resilience of semiarid vegetation communities and the general recovery of the vegetation following an increase in precipitation (Hickler et al., 2005; Olsson et al., 2005). The gypsum shrublands of the Ebro basin are highly adapted to the natural variability in precipitation that characterizes the Mediterranean climate, and have various adaptation strategies that enable tolerance of drought (Braun-Blanquet and Bolós, 1957). Nevertheless, the vegetation cover of the central Ebro basin has been substantially modified by human activities over the centuries of inhabitation, which explains why it is prone to degradation processes. Given the vulnerability of the existing vegetation it is reasonable to assume that the general trend of vegetation cover loss and the greater decline observed in the most degraded lands may be related to the increase in aridity in recent decades. The unprecedented increase in temperature in recent decades may be reducing the resilient of the vegetation to severe drought events. This has been observed in other natural systems worldwide, where so-called global warming type-droughts are having a much greater impact than is expected from natural precipitation droughts (Breshears et al., 2005; Carnicer et al., 2011).

Human drivers of vegetation changes

Degradation trends have been observed in various semiarid areas of the Mediterranean region (Puigdefábregas and Mendizábal, 1998; del Barrio et al., 2010). Although persistent dry conditions may trigger or accelerate degradation processes, there is no doubt that human activities play a determining role (Puigdefábregas, 1995). The most important current degradation processes in the Mediterranean region are related to increased human pressure as a consequence of increasing populations and intensification of human activities, particularly in the countries of North Africa (Puigdefábregas and Mendizábal, 2005) but also in some overgrazed areas of southern Europe (Hostert et al., 2003) and the near east (Evans and Geerken, 2004). Overgrazing, cultivation of steep slopes and development of intensive crops all reduce the vegetation cover and the fertility of the soils, and can trigger and/or intensify degradation processes. Nevertheless, in the central Ebro valley the increase in degradation processes is not related to land use intensification and overgrazing, because livestock grazing has decreased dramatically in recent times (Ameen et al., 2011; Lasanta et al., 2011). At present, the big transhumant sheep flocks from the Pyrenees that traditionally grazed the gypsum rangelands of the central Ebro basin have practically disappeared (García-Ruiz and Lasanta, 1990). In addition, the livestock flocks maintained in the municipalities of the study area have also declined during the twentieth century. As a representative example, in the municipality of Zaragoza (the biggest of the study area), the number of sheeps in the decade of 1830 was in average 95597 (Pinilla, 1995). Most of them grazed in the alpine pastures of the Pyrenees during the summer but in winter they grazed the gypsum rangelands. In addition, the municipality also maintained in winter big transhumant flocks that moved from the Pyrenees. Andreu-Casadebaig and Proust (1979) and Hernanz Plaza (1986) indicated that most of the transhumant flocks from the Tena valley (located in the central Pyrenees) grazed in the municipality of Zaragoza. These authors showed that at the end of the XVIII century the number of sheeps in this valley was about 100000, and Fillat (1981) indicated that also some flocks of Ansó (other Pyrenean valley) rented pasture lands in Zaragoza.

At present, the livestock censuses of the region indicate that, between 1962 and 1999, the average number of sheeps in Zaragoza was 44573 and the last census of 2009 show a large decrease, since it records a total number of 22590 sheeps. This is in agreement with the statistics of the rest of municipalities of the study area, in which the number of extensive livestock systems has decreased 48% in the last decade.

Although overgrazing may still occur in some areas in the region (e.g. around shelters and watering points), Pueyo and Alados (2007b) showed that recent regional vegetation dynamics in the area are not related to current grazing use. Therefore, although the gypsum rangeland areas traditionally fed large transhumant flocks during the winter (Gomez Ibañez, 1977; Lasanta et al., 2011), as in other Mediterranean countries, nowadays the tendency in the region is and the abandonment of grazing areas (Barrantes et al., 2009). These areas are now only grazed by small stabled sheep flocks (around 850 heads) with light stocking rates (less than 1 head ha^{-1} on average). Animals receive most of their daily energetic requirements as supplementary feed (usually forages and cereals; Pueyo, 2006) and consequently, they do not take much biomass from shrublands (less than 50% of the available biomass, which is the recommended consumption for a sustainable use in semiarid Mediterranean shrublands; Boza et al., 1998), leading to actual grazing intensities below the carrying capacity in most of the territory. Livestock pressure has traditionally been the main factor limiting vegetation development and growth in these areas (Braun-Blanquet and Bolòs, 1957). Various studies have shown that moderate livestock pressure can have positive effects on the productivity and quality of range ecosystems, increasing productivity (McNaughton, 1985) and biodiversity (Montalvo et al., 1993), and favoring the dispersal of seeds (Pueyo et al., 2008). However, the cumulative effects of overgrazing are predominantly negative because of the reduction in vegetation cover and modification of the soil physical properties (Moret-Fernández et al., 2011), which affect seed germination and the storage of water. In the gypsum rangelands of the central Ebro valley the current livestock pressure is typically not high, which may be a positive factor favoring development of the vegetation cover. Nevertheless, centuries of high livestock use of these rangelands may have determinately affected both the density of the vegetation cover and the soil properties, limiting subsequent development of the cover following the decrease in livestock pressure. Most plant communities in the central Ebro valley are considerably modified from the natural communities, and in some cases the current degradation may be irreversible (Pueyo et al. 2009). Pueyo and Alados (2007b) showed that vegetation recovery in the gypsum rangelands of the center of the Ebro valley is impossible after a long period of livestock pressure and soil degraded conditions. If these conditions are aggregated to the observed trends in aridity, acceleration in the loss of vegetation cover appears to be the logical consequence.

Biotic and abiotic drivers

The limitations imposed by abiotic factors may also help explain the spatial patterns of land degradation processes in the study area. In particular, the microclimatic conditions determined by topography are central to understanding the observed differences in the trends in vegetation cover in the study area. Independently of the current adaptation of the vegetation to the limiting environmental constraints of gypsum soils (i.e. climate aridity, low soil fertility and water retention capacity), the current climate trends and past human uses have determined that in some places the vegetation is under environmental stress conditions that strongly favor degradation. The vegetation in the most arid sites (lower precipitation and higher rates of PET) and on southern slopes is affected by greater water stress, and it is these areas that are most affected by the loss of vegetation cover.

The terrain slope may also be a determinant of trends in vegetation cover in some areas. We observed a positive effect of slope, which was in contrast to that expected because the steeper slopes commonly have less developed soils and retain less water than flat areas. The observed positive influence of terrain slope on the evolution of vegetation in the study area is probably indirect and related to the limitations for grazing. Pueyo et al. (2007) showed that steep slopes in the gypsum rangelands of the central Ebro valley hold certain protection of the vegetation for livestock grazing. This could explain why some areas in the southwest of the study area, where steeper slopes are located, are showing the greatest increases in vegetation cover. This is because the soil and vegetation conditions are less degraded than the flat areas of the southeast, which were traditionally more heavily grazed and have been subject to most loss in vegetation cover. The other important drivers of vegetation degradation are intrinsic to the ecosystems, and closely related to the characteristics of the vegetation. In already degraded areas having low levels of current vegetation cover that is unable to efficiently use the available resources, vegetation recovery is less. This pattern has been observed in various semiarid regions of the world. For example, Pickup (1998) compared the response of degraded and nondegraded areas in Australia to climate variability, and showed that the most degraded areas had substantially reduced productive potential. Bonet (2004) showed that the recovery of vegetation was difficult in semiarid rangelands with low levels of vegetation cover, as these areas are not resilient to extreme perturbations including more frequent and severe droughts. Thus, it appears that areas with poor vegetation cover represent a highly unstable situation and are at greater risk of being affected by irreversible degradation processes. This is because of the positive feedback that occurs between the presence of vegetation and the soil conditions: vegetation cover protects the soil from erosion, surface sealing and compaction, which in turn promotes plant establishment and survival. When vegetation cover is reduced by climatic conditions or grazing, the soil is exposed and can become degraded, which in turn hampers plant establishment. In this scenario there may be a threshold of plant cover for recovery following high levels of plant mortality resulting for extreme climatic events or overgrazing; below the threshold ecosystems remain stably degraded (Rietkerk and van de Koppel, 1997; Rietkerk et al., 2004). This could explain why stably degraded states in semiarid rangelands are difficult to revert following the removal of grazing, even with the application of ecological restoration practices (van de Koppel et al. 1997; Pueyo et al., 2009).

Therefore, past human management and current climate trends interact with local environmental conditions to determine the occurrence of degradation processes. At present, the phenomenon is being observed in the most water-limited areas and areas already characterized by marked degradation and having very low levels of vegetation cover. However, if future climate trends exacerbate arid conditions, degradation processes could be triggered in areas that are currently not under limiting water conditions.

Future projections

In the Mediterranean region a decrease in ecosystem net primary production is expected in response to the projected climatic conditions for the future, particularly in the most arid areas.

Future climate projections for the Mediterranean region show a consistent trend toward warmer conditions (Gibelin and Déqué, 2003; Giorgi, 2006; Goubanova and Li, 2007; Alpert et al., 2008). The predicted magnitude of change for the 21st century varies between 1 and 6 °C, depending on the model and the greenhouse gas emissions scenario used. In addition to average temperature increase, a greater occurrence of extreme hot events in summer is expected (Beniston, 2004; Giorgi and Lionello, 2008).

Most studies project a general trend toward less precipitation in the next century (Ragab and Prudhomme, 2002; Gibelin and Déqué, 2003; Giorgi et al., 2004; Goubanova and Li, 2007; Giorgi and Lionello, 2008; Evans, 2009). Droughts of a severity expected every 100 years will occur every 10 years in the northern Mediterranean (Weiß et al., 2007). These changes may have very negative consequences for semiarid ecosystems, which are in a state of very unstable equilibrium and are highly vulnerable to change. Diffenbaugh et al. (2007) have shown that by the end of the 21st century the Mediterranean region may undergo a substantial increase in the northward extension of dry and arid lands, caused by a large increase in warming and a pronounced decrease in precipitation, especially during spring and summer. Thus, dynamic vegetation models predict a small general reduction in the net primary production of water-limited ecosystems of the Mediterranean region, in contrasts to the predictions for northern Europe, where higher temperatures, precipitation and atmospheric CO_2 levels may favor vegetation growth (Anav and Mariotti, 2011). This could cause increased stress in semiarid Mediterranean rangelands (particularly those affected by other limiting factors, such as soil properties in the gypsum rangelands), and result in the degradation patterns that are presently evident in the most limited areas having much wider distribution.

6. Conclusions

This study highlights the influence of climate aridification on the reduction of vegetation cover in highly vulnerable gypsum rangelands in the Ebro basin, the northernmost semiarid region in Europe. We have shown a dominant trend of decrease in vegetation cover, particularly in summer and in areas under the greatest water stress conditions (low precipitation, higher evapotranspiration rates and sun-exposed slopes). The pattern was observed during a period of marked decrease in water availability, with more frequent and severe droughts mainly as a consequence of the warming processes that have dramatically increased PET rates.

The study area is undergoing a progressive decrease in human pressure. Thus, in recent decades there has been a sharp decrease in grazing pressure and general land marginalization. This suggests that natural vegetation should recovery towards its state prior to perturbation. However, the substantial environmental limitations (mainly soil characteristics and aridity), the centuries of intensive land use that have contributed to the very degraded landscapes, and the recent climate evolution have determined the evolution of the vegetation in the gypsum rangelands of the Ebro basin in the last 30 years. Thus, the results reported here suggest that degradation could be indirect as a consequence of the overexploitation that has characterized this region, but increased aridity related largely to global warming conditions may be triggering and/or accelerating the degradation processes.

The methodological approach followed in this study, which considered seasonal differences and used a vast dataset of very high spatial resolution images covering a period of 26 years, has no precedent. Given that identification of degradation processes and rates at regional scales is very difficult, we stress the significance of the strong evidence provided by this study for the occurrence of degradation processes related to the trends in global warming that are influencing the Mediterranean region. The occurrence of the most negative trends in areas under water limiting conditions corroborates this pattern. For these reasons, the observed changes could be an early warning of processes that may affect wider areas of the Mediterranean, based on the current climate change models for the 21st century.

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Table 1. List of images used in the study.

Summer			**Spring**		
Date of adquisition	**Sensor**	**Hour of adquisition (GMT)**	**Date of adquisition**	**Sensor**	**Hour of adquisition (GMT)**
20/08/1984	TM	10.22	11/03/1989	TM	10.22
07/08/1985	TM	10.22	30/03/1990	TM	10.05
13/08/1987	TM	10.15	06/03/1993	TM	10.08
02/08/1989	TM	10.17	09/03/1994	TM	10.07
24/08/1991	TM	10.12	28/03/1995	TM	9.78
10/08/1992	TM	10.08	17/03/1997	TM	10.15
29/08/1993	TM	10.08	20/03/1998	TM	10.31
03/08/1995	TM	9.78	23/03/1999	TM	10.37
24/08/1997	TM	10.25	17/03/2000	ETM	10.6
14/08/1999	TM	10.35	10/03/2003	ETM	10.5
08/08/2000	ETM	10.57	07/03/2005	TM	10.5
26/07/2001	ETM	10.53	13/03/2007	TM	10.62
30/08/2002	ETM	10.52	18/03/2009	TM	10.48
27/08/2004	TM	10.43			
14/08/2005	TM	10.52			
01/08/2006	TM	10.6			
04/08/2007	TM	10.6			
06/08/2008	TM	10.48			

Table 2. Summary of the Mann-Kendall test for the entire study area for the series of summer and spring showing the surface area affected by positive and negative trends. The results are provided in surface area (ha) and percentage.

	Spring (1989-2009)		Summer (1984-2008)		Summer (1989-2008)	
Evolution	**surface (Ha)**	**%**	**surface (Ha)**	**%**	**surface (Ha)**	**%**
Negative (< 0.05)	968.3	2.2%	6554.9	14.7%	4743,6	10,6%
Negative (n.s.)	26411.4	59.1%	25662.5	57.4%	25211,2	56,4%
Positive (n.s.)	16836.2	37.7%	11709.9	26.2%	13783,2	30,9%
Positive (< 0.05)	458.6	1.0%	747.5	1.7%	936,5	2,10%
Total	44674.5	100.0%	44674.5	100.0%	44674,5	100.%

Table 3: Cross-tabulation analysis of the trends in vegetation cover found in spring and summer. The data are percentages of the entire study area.

		Summer (1984-2008)				
		Negative (< 0.05)	**Negative (n.s.)**	**Positive (n.s.)**	**Positive (< 0.05)**	**Total**
Spring (1989-2009)	**Negative (< 0.05)**	0.7%	1.2%	0.3%	0.00%	2.2%
	Negative (n.s.)	11.3%	36.6%	10.9%	0.40%	59.2%
	Positive (n.s.)	2.7%	19.3%	14.5%	1.20%	37.7%
	Positive (< 0.05)	0.0%	0.3%	0.6%	0.10%	1.0%
	Total	14.7%	57.7%	26.2%	1.70%	100.0%

Table 4. Structure matrix of the discriminant analysis. The table shows the correlation values of each predictor variable with the three discriminant functions. The variables represented in each of the first two functions are shown in bold.

	Summer				**Spring**		
Predictors	1	2	3	Predictors	1	2	3
Solar radiation	**0.66**	-0.08	0.55	Veg. coverage	**0.88**	-0.14	-0.09
PET	**0.52**	-0.43	-0.12	RUE	**0.70**	-0.43	-0.09
RUE	-0.33	**-0.55**	0.07	Solar radiation	**-0.63**	0.40	-0.06
Veg. coverage	-0.50	**-0.53**	0.19	Precip.	0.28	**0.72**	0.10
Slope	-0.37	0.43	-0.21	PET	-0.25	-0.43	-0.61
Precip.	-0.56	0.25	0.58	Slope	0.43	0.48	-0.49

FIGURE CAPTIONS.

Figure 1. Location of the study area and spatial distribution of the shrublands on gypsum soils analyzed in this study.

Figure 2. Relationship between vegetation cover and its interannual variability in summer (a) and spring (b), and between the average vegetation cover and the coefficient of variation in summer (c) and spring (d).

Figure 3. Evolution of the average percent vegetation cover in the study area from 1984 to 2009.

Figure 4. Spatial distribution of the changes in the vegetation cover in summer (1984–2008) and spring (1989–2009). The data are in percent per decade.

Figure 5. Relationship between the magnitude of change in vegetation cover in summer and spring (a), and between the changes observed in spring and summer in the 1989–2008 period (b).

Figure 6. Evolution of annual precipitation, mean temperature, potential evapotranspiration (PET) and the climatic water balance. The monthly evolution of the standardized precipitation index (SPEI) at a time scale of 12 months is also showed for the period 1970–2009.

Figure 7. Box-plots of the predictors as a function of the trend categories obtained using the Mann-Kendall test. A) summer, B) spring.

Figure 8. Centroids of the trend categories corresponding to functions 1 and 2 of the discriminant analysis for summer and spring.

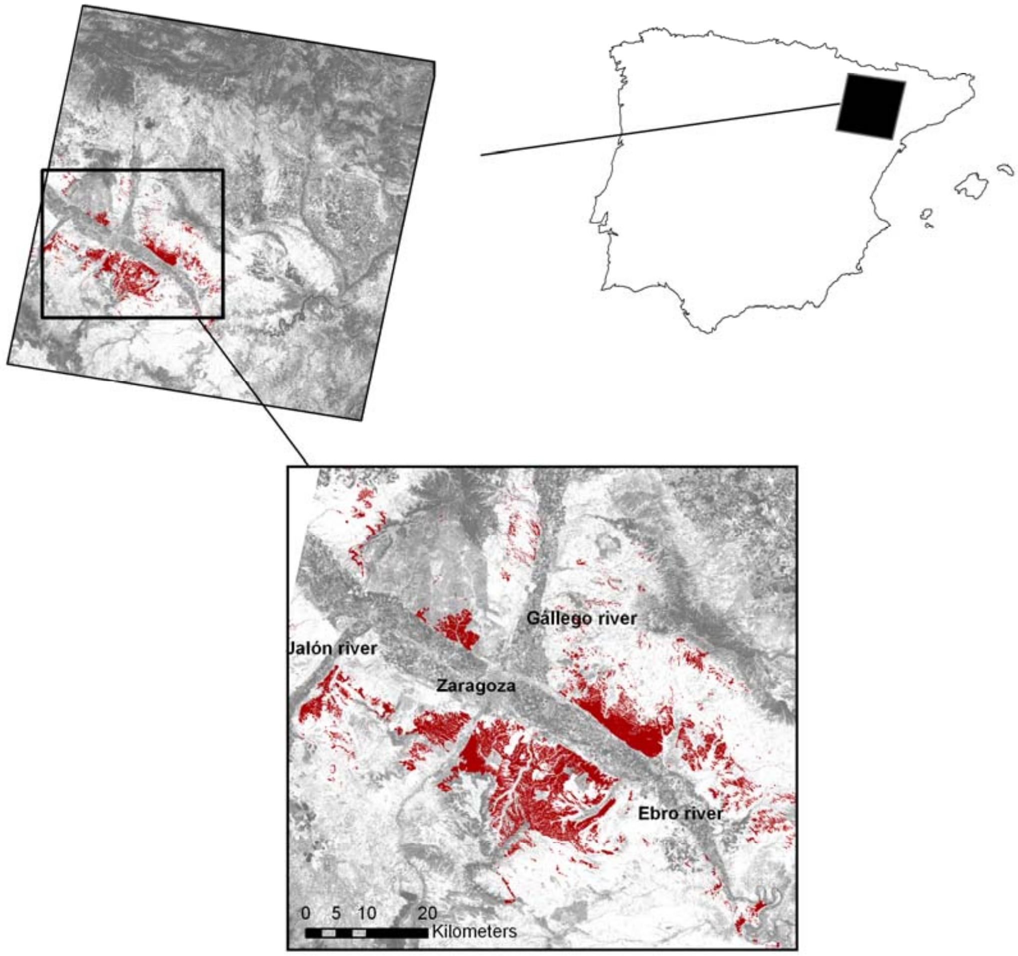

Gállego river
Jalón river
Zaragoza
Ebro river
0 5 10 20
Kilometers

Fig1

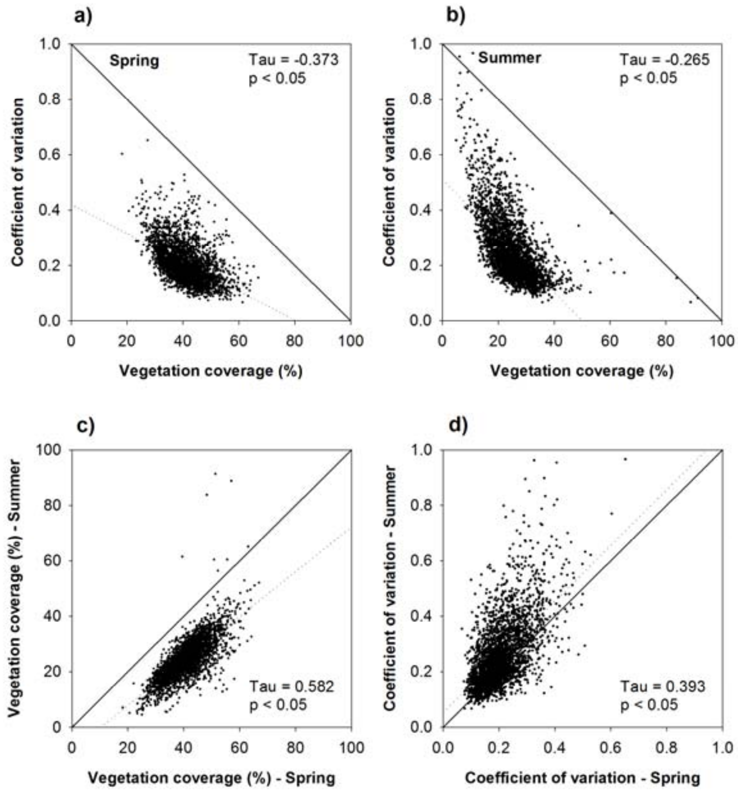

a)
Spring
Tau = -0.373
p < 0.05
Coefficient of variation
Vegetation coverage (%)
b)
Summer
Tau = -0.265
p < 0.05
Coefficient of variation
Vegetation coverage (%)
c)
Tau = 0.582
p < 0.05
Vegetation coverage (%) - Summer
Vegetation coverage (%) - Spring
d)
Tau = 0.393
p < 0.05
Coefficient of variation - Summer
Coefficient of variation - Spring

Fig2.

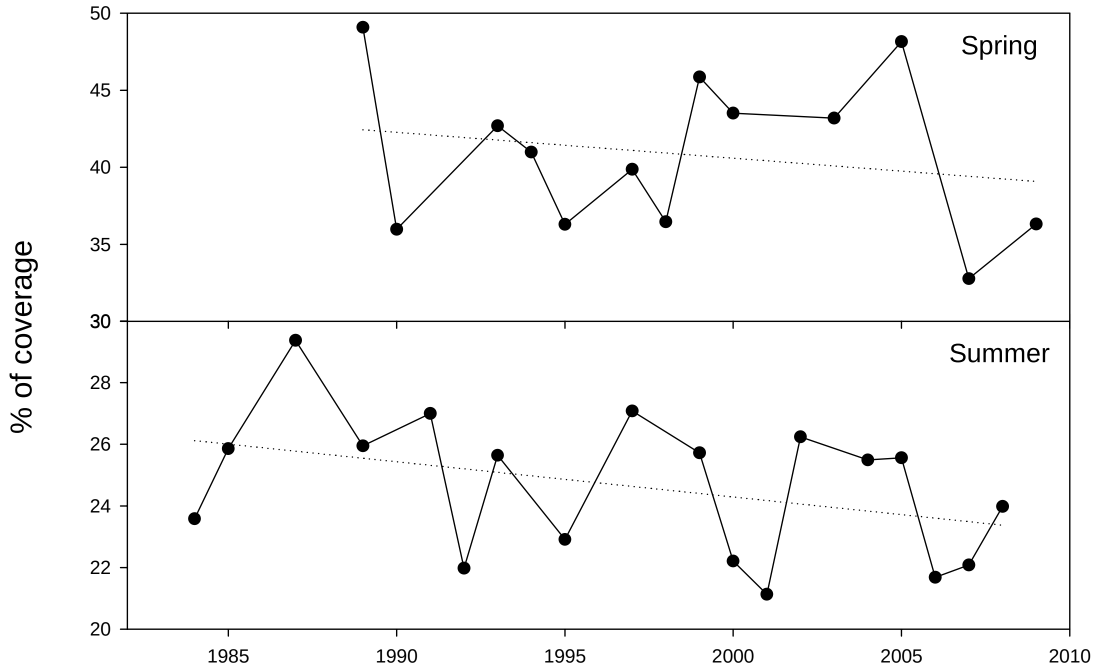

50
45
40
35
30
28
26
24
22
20
% of coverage
Spring
Summer
1985
1990
1995
2000
2005
2010

Fig 3.

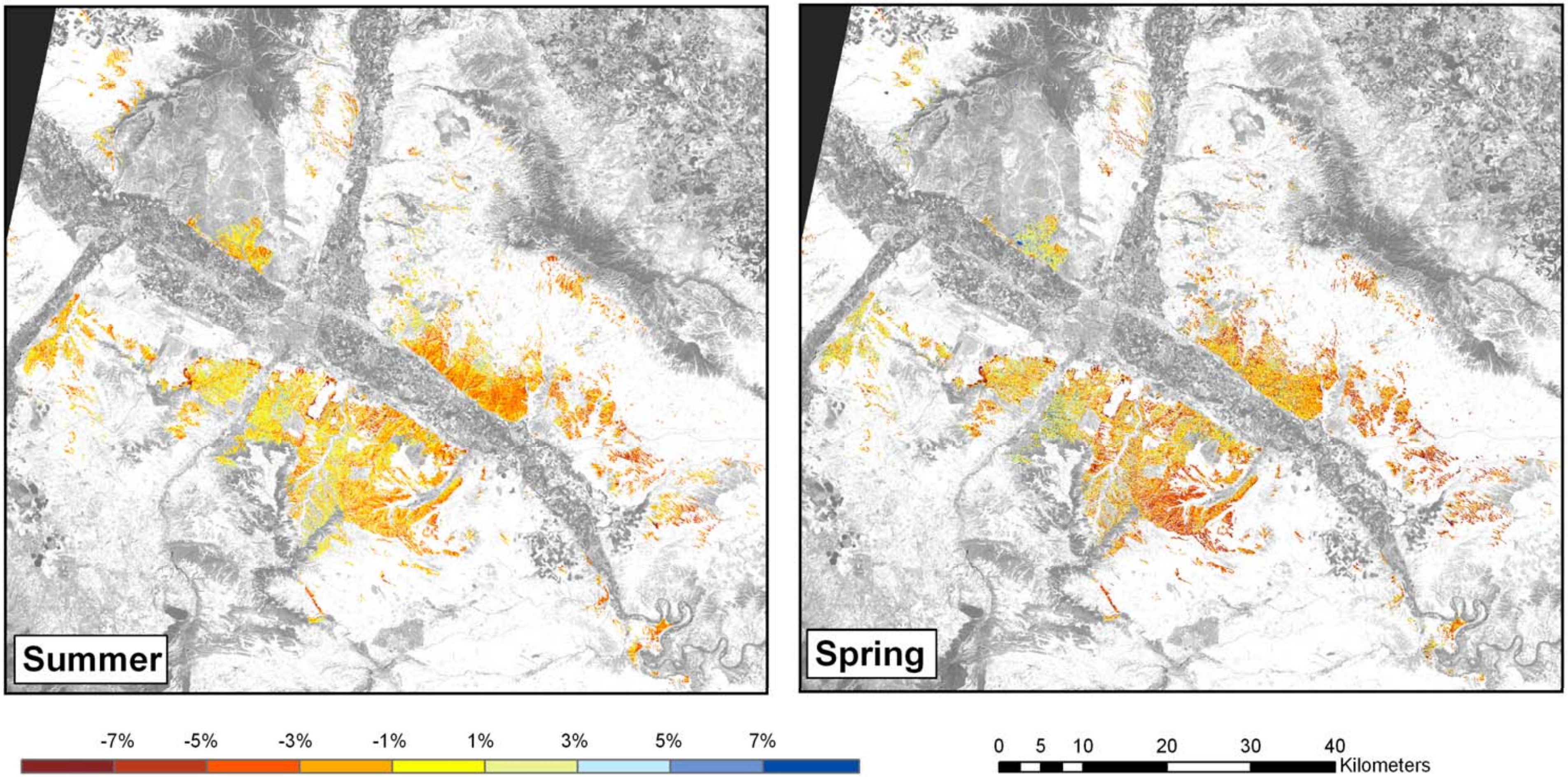


Fig 4.

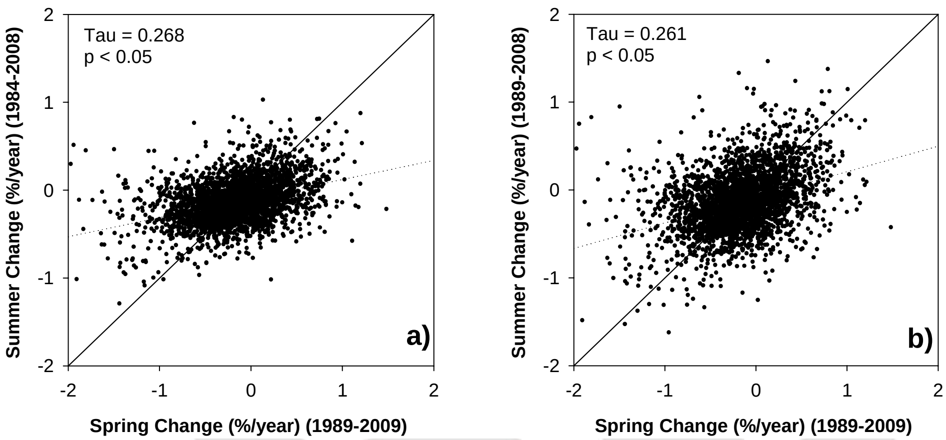

Tau = 0.268
p < 0.05
a)
Summer Change (%/year) (1984-2008)
Spring Change (%/year) (1989-2009)
Tau = 0.261
p < 0.05
b)
Summer Change (%/year) (1989-2008)
Spring Change (%/year) (1989-2009)

Fig 5.

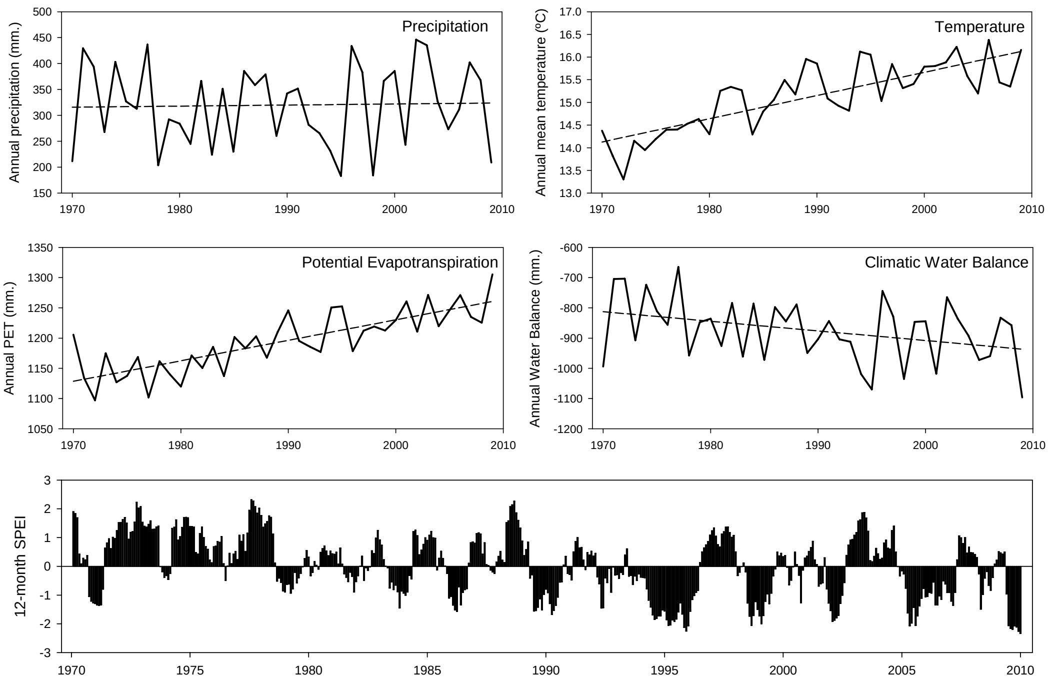
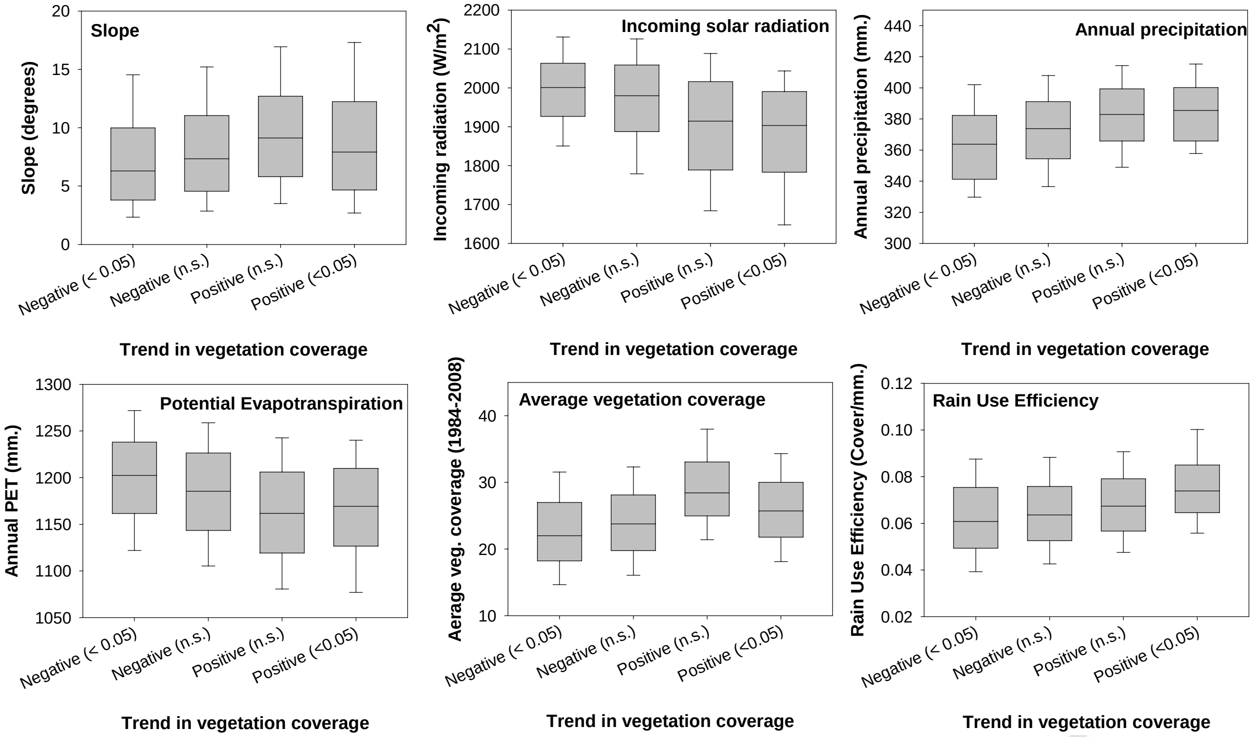

Precipitation
Temperature
Potential Evapotranspiration
Climatic Water Balance
Annual precipitation (mm.)
Annual mean temperature (ºC)
Annual PET (mm.)
Annual Water Balance (mm.)
12-month SPEI

Fig 6.

A)


Slope
Slope (degrees)
Incoming solar radiation
Incoming radiation (W/m2)
Annual precipitation
Annual precipitation (mm.)
Potential Evapotranspiration
Annual PET (mm.)
Average vegetation coverage
Aerage veg. coverage (1984-2008)
Rain Use Efficiency
Rain Use Efficiency (Cover/mm.)
Negative (< 0.05)
Negative (n.s.)
Positive (n.s.)
Positive (<0.05)
Trend in vegetation coverage

B)

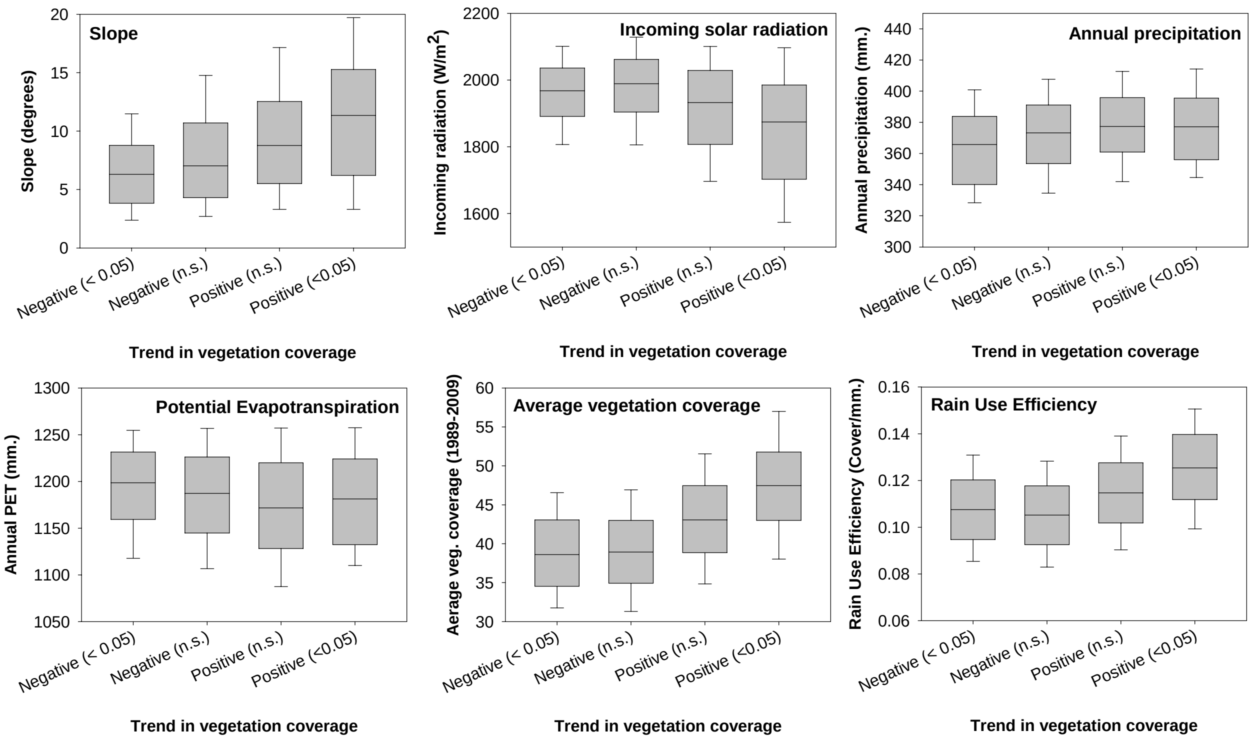

Slope
Slope (degrees)
Incoming solar radiation
Incoming radiation (W/m2)
Annual precipitation
Annual precipitation (mm.)
Potential Evapotranspiration
Annual PET (mm.)
Average vegetation coverage
Aerage veg. coverage (1989-2009)
Rain Use Efficiency
Rain Use Efficiency (Cover/mm.)
Negative (< 0.05)
Negative (n.s.)
Positive (n.s.)
Positive (<0.05)
Trend in vegetation coverage

Fig 7.

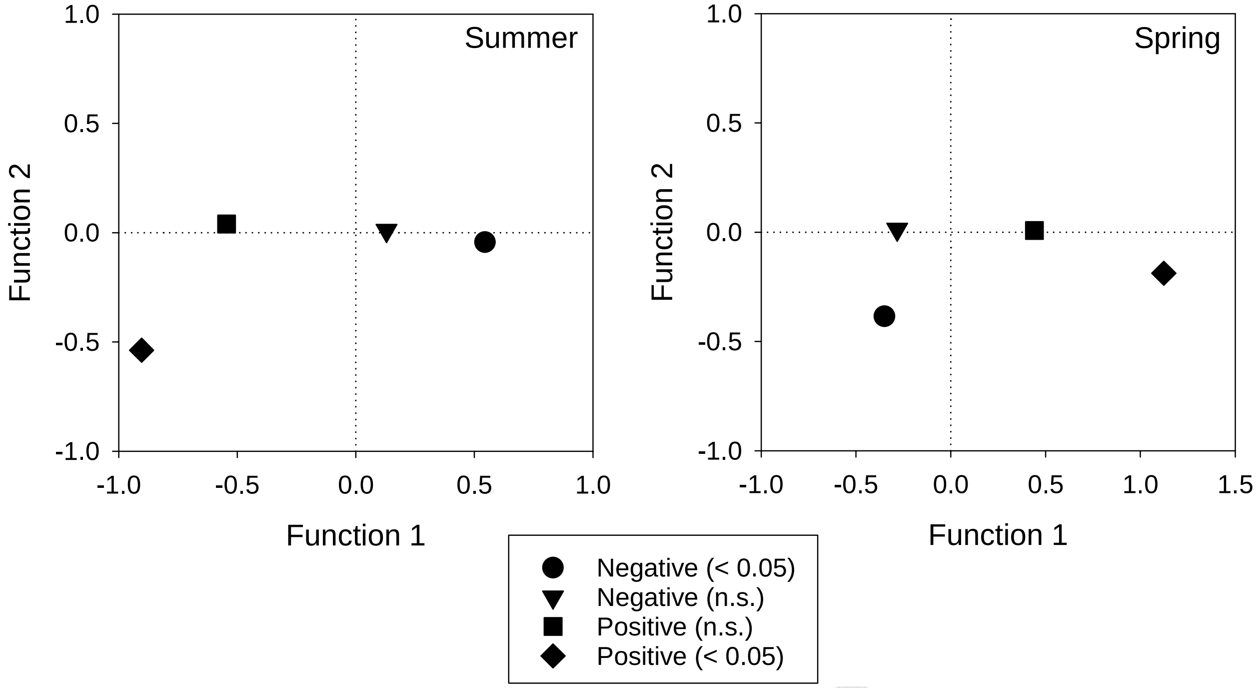

Summer
Spring
Function 2
Function 1
1.0
0.5
0.0
-0.5
-1.0
1.5
Negative (< 0.05)
Negative (n.s.)
Positive (n.s.)
Positive (< 0.05)

Fig 8.