Landscape Ecology **19:** 543–559, 2004.

Research article

Variations in landscape patterns and vegetation cover between 1957 and 1994 in a semiarid Mediterranean ecosystem

Alados[1], C.L.[1,*], Y. Pueyo[1], O. Barrantes[2], J. Escós[3], L. Giner[1] and A.B. Robles[4]
*[1]Pyrenean Institute of Ecology Avda. Montañana 1005, Apd. 202, 50080 Zaragoza, Spain; [2]Dpt. Agricultura y Economía Agraria, Universidad de Zaragoza, Miguel Servet 177, 50013, Zaragoza, Spain; [3]Infraestructura and Ecología. Avda. Cardenal Herrera Oria, 65 3ª pl, 28034, Madrid, Spain; [4]Estación Experimental del Zaidín. Profesor Albareda 1, Apd. 419, 18008, Granada, Spain; *Author for correspondence (e-mail: alados@ipe.csic.es)*

Received 11 March 2003; accepted in revised form 23 February 2004

Key words: Fragmentation, Land-use, Mediterranean, Semi-arid ecosystem, Vegetation cover transition, Cabo de Gata-Nijar Natural Park, Spain

Abstract

The aim of this study was to analyze the main processes that determine changes in landscape patterns and vegetation cover from 1957-1994 to develop a model for land cover dynamics. Land cover and landscape patterns were assessed and compared using aerial photographs taken in 1957, 1985, and 1994. Over this period, tall grass steppe and arid garrigues increased by 6% and 4%, respectively, while crop fields decreased by 15% and tall arid brush remained the same. Over the same period, tall grass steppes and arid garrigues became less fragmented. Changes in land use were triggered by socioeconomic forces, which were constrained by the underlying structure of the physical landscape. The best preserved vegetation (tall arid brushes) was concentrated at higher elevations, with a pronounced slope, not oriented towards the sea, and in volcanic substrate. Communities tended to be better preserved further away from towns and at lower house densities. Tall grass steppe was present on more gradual sea-oriented slope and in calcareous substrate, and increased at higher elevations, although not far from the town but away from high anthropogenic influence. Previous studies have revealed that traditional land uses of this landscape, particularly grazing, favoured the transition from tall arid brush to tall grass steppe. In this study, we analyzed to what extent the underlying structure of the physical landscape imposes limitations to the vulnerability to human activity of the main vegetation types. According to the data on the probability of vegetation transition over the 37-year period, the shift from tall arid brush to tall grass steppe appeared to be favoured by gradual slopes. Tall arid brush recovered from either arid garrigues or tall grass steppes at steeper slopes. Thus, steep terrain had a favourable effect on the formation of brushwood and more gradual terrain favoured tall grass steppe. The prevalent trends were confirmed by a projection of a transition matrix over 100 years.

Introduction

Changes in landscape use and cover take place over long periods of time as a result of changes in population (Turner et al. 1990, 1993). Traditional land use activity such as farming has modelled the Mediterranean landscape for centuries (Stevenson and Harrison 1992; Barberá et al. 1997; Valero-Garcés et al. 2000). Humans colonized the Almería coast (SE Spain) during the Neolithic period and settled in river valleys. No major evidence of climate change has been reported since that period (Chapman 1978). During Roman times, mountain slopes provided timber and the lowland steppes consisted of abundant *Stipa tenacissima* L. grass and a few areas of climax evergreen vegetation (Chapman 1978). Traditional land use included cereal crops in the valley and husbandry on the slopes. Since the middle of the 20th century, the landscape has undergone important changes with drastic consequences for the ecosystem. These include farming intensification (irrigation, pesticides, fertilizers, machinery, etc), increased migration to urban centres and more summer houses on coastland (housing projects, infrastructure, transportation, etc.). Similar processes are occurring in other Mediterranean areas (Ispikoudis et al. 1993; Lyrintzis 1996; Papanastasis and Kazaklis 1998), resulting in significant landscape degradation (Arianoutsou-Faraggitaki 1985; Aidoud et al. 1998). Many changes have occurred so quickly that the natural environment cannot adapt and is significantly impacted. As a result, habitat fragmentation has increased dramatically over the last century (Saunders et al. 1991; Groom and Schumaker 1993).

Habitat fragmentation (habitat loss and isolation) is a serious threat to biological diversity (Soulé 1986; Kareiva et al. 1993; Edwards et al. 1994). Populations become more isolated since dispersal success and patch colonization are disrupted, resulting in lower population persistence and more regional extinctions for the entire metapopulation across the landscape (Lande 1987; With and King 1999a).

Proper management and conservation of landscapes should consider the relationship between landscape patterns (composition and spatial configuration) and how they are generated. Factors that affect landscape can be divided into abiotic (topography, climate, soil), biotic (organism interaction, ecosystem succession) and human (Levin and Paine 1974). On the spatio-temporal scale of ecological studies, human related factors are the driving force of landscape evolution (Krummel et al. 1987; Iverson 1988; Huebner et al. 1999; Turner et al. 1996; Wickham et al. 2000). Broad-scale variability in abiotic factors will constrain the interactions among biotic factors (Turner et al. 2001). More information is needed on how the stable physical attributes of landscape elements constrain land use and determine changes in land cover (Pan et al. 1999; Swenson and Franklin 2000).

The aim of this study was to analyze the processes that have generated changes in landscape patterns and vegetation cover from 1957-1994 in Cabo de Gata-Níjar Natural Park (Almería province). We wanted to know to what extent physical characteristic of the landscape determined vegetation cover and influenced land degradation. The primary focus was on land cover change and fragmentation, and the driving processes that generated such changes. We also tested whether vegetation transitions were significantly influenced by anthropogenic factors and we described the main constraints on transitions in vegetation/ land cover.

Previous studies report that grazing (the main activity in the slopes of Cabo de Gata-Níjar Natural Park) has favoured the colonization of the tussock-graminoid *Stipa tenacissima* L. (Alados et al. 2003), in agreement with other work in semi-arid Mediterranean areas where grazing favoured perennial grasses (Hadar et al. 1999). The dominant plant cover community (tall arid brush) leads the vegetation through a progressive series of stages towards the end-successional steppe of *S. tenacissima* (Tomaselli 1981). This species is clearly the most successful species in semi-arid south eastern Spain because of its reproductive strategy based on combined sexual and asexual reproduction. In the latter, the underground stem develops extensive branched rhizomes that fragment by withering, providing a good spatial spread (Hessen 1999). The buried renewal buds cannot be reached by livestock and have a high capacity to retain soil and runoff, especially in fine material soils (Cammeraat et al. 2002). However, when the slope gradient increases, perennial bunch-grasses also disappear due to the loss of fine material. This leaves the root exposed, reducing the survival of species with shallow roots, such as perennial grass. In contrast, woody species with deeper roots can survive in pronounced slopes with greater stoniness and rugosity (Hodgson et al. 1994; Guerrero-Campo et al. 1999). Slope gradient is then an important factor determining vegetation succession in semiarid areas. Indeed, the thinner structure of calcareous soils appear to be more suited to the thin and dense roots system of dominant *S. tenacissima*, while the greater stoniness and rugosity of volcanic soils delays surface runoff, providing a greater infiltration rate (Cerdá 2001) and being more suitable for the thicker roots of shrubs.

Thus, the dynamics of transition from tall arid brush to tall grass steppe must be related to the topographical characteristics, soil properties, geographical location and the favourable living conditions of *S. tenacissima*.

We hypothesize that:

(i) The climax vegetation of Cabo de Gata-Níjar N. P., tall arid brush, would be more abundant in areas with pronounced slopes where *S. tenacissima* is less competitive. Conversely, moderated slopes would favour *S. tenacissima* configuration.

(ii) The vulnerability of the vegetation to the degradation depends on both the dynamic of the erosion-sedimentation processes and the sensitivity of the vegetation to human impact. We expect the transition from tall arid brush to tall grass steppe would be retarded in rocky soils, such as volcanic soils, whereas gradual slopes would promote the transition to tall grass steppe. Indeed, human impact would accelerate the transition from tall arid brush to tall grass steppe.

(iii) Spatial structure of the vegetation in the landscape is the result of both of pre-existing environmental heterogeneity and spatial self-organization caused by local interactions and local dispersion. Changes in vegetation spatial patterns among years reflect changes in the spatial self-organization, whereas topographic variables remain invariable. We expect those changes would be related with the degree of preservation of each vegetation cover class.

Methods

Study Area

Cabo de Gata-Níjar Natural Park (SE Spain) is a protected area (37 570 ha) that has been preserved since 1987. It ranges in altitude from sea level to 493 m at El Fraile peak. The climate is Mediterranean semi-arid. The average annual rainfall between 1973 and 1996 at 43 m elevation was 193.9 mm, and the mean annual temperature was 19.4 °C (Passera 1999).

The potential vegetation is a dense matorral of *Pistacia lentiscus* L., *Chamaerops humilis* L. and *Rhamnus lycioides* L. (*Chamaeropo humilis-Rhamneto lycioidis* sigmetum) (Peinado et al. 1992). *Periploca laevigata* is also a common species. Recently, the landscape has been altered by human activities such as cropping, grazing, burning and deforestation and is composed of various types of shrubland.

The area was mainly exploited under traditional agro-pastoral systems, with cereal cultivation on the floodplains and livestock (sheep and goats) grazing on the slopes. Esparto-harvesting *(Stipa tenacissma)* has been frequent since Roman times until recently replaced by plastic fibre (Mota et al. 1996). After the 1960s, the human population increased rapidly due to an increase in intensive crops and a booming tourist industry (mainly on the coast). Inland farms were abandoned since they were less profitable, partly due to the low precipitation regime in the area (Castro and Guirado 1993; Guirado and Mendoza 2000; Rodríguez and Sánchez 2000).

We developed a GIS database to study the structure, spatial patterns and evolution of the landscape at Cabo de Gata-Níjar Natural Park. Fine-resolution data sets were generated from manual digitization and automated scanning of aerial photographs. A topographic map from 1985 (scale 1:25.000) was obtained from the Geographic Institute. Contours were digitized every 20 m as well as roads, settlements and streams. Vegetation and land/cover use in 1994, 1985 and 1957 were calculated from aerial photographs (1994, scale 1:15 000; 1985, scale 1:20 000 and 1957, scale 1:33 000). A geology map was obtained from the Geological and Mining Institute (scale 1: 50 000). All the thematic maps were prepared and stored in GIS using ARC/INFO and ERDAS software on UNIX workstations and PCs. All the layers were stored in vector format, where information is defined by points, lines or polygons.

Vegetation/land use was classified into nine groups (Figure 1) according to CORINE (1991): (1) Tall arid brushes (*Phlomido-Ulicetum canescentis* ; (2) Tall arid brushes (*Anthyllidetalia terniflorae*); (3) Tall grass steppes of *Stipa tenacissima (Lygeo-Stipetalia tenacissima)*; (4) Arid garrigues (*Artemisio – Salsoletum genistoidis, Siderito-Teucrietum chaeridemi*); (5) Halonitrophylous shrub of *Tamarix boveana* (*Tamaricetalia africanae*); (6) South-eastern gypsum scrubs (*Thymo-Teucrion verticillati*); (7) Ruderal communities or extensive crop fields; (8) Cliffs, rocks and sand dunes; (9) Towns, mines and salines. The phytosociological nomenclature of vegetation communities follows Peinado et al. (1992) and Mota et al. (1997).

A reconnaissance survey was also carried out in the study area to identify and relate the vegetation types to the photos. Stratified random sampling was used to distribute sampling points (n = 189) in different vegetation strata. The overall accuracy was 73.54%. Producer's and user's accuracy (Congalton 1991) were higher for tall arid brush (97.14 and 87.18 respectively) and tall grass steppe (75.86, 84.61%), than arid garrigues (60.0, 48.98%) and ruderal communities/extensive crop fields (66.07, 75.51%). The accuracy of arid garrigues was lower because of the time lag between aerial photographs (1994) and the year of ground observation (2002). The transition of an abandoned crop field to arid garrigue may take about seven years. Indeed, some abandoned fields were cultivated again due to demand for land (the population

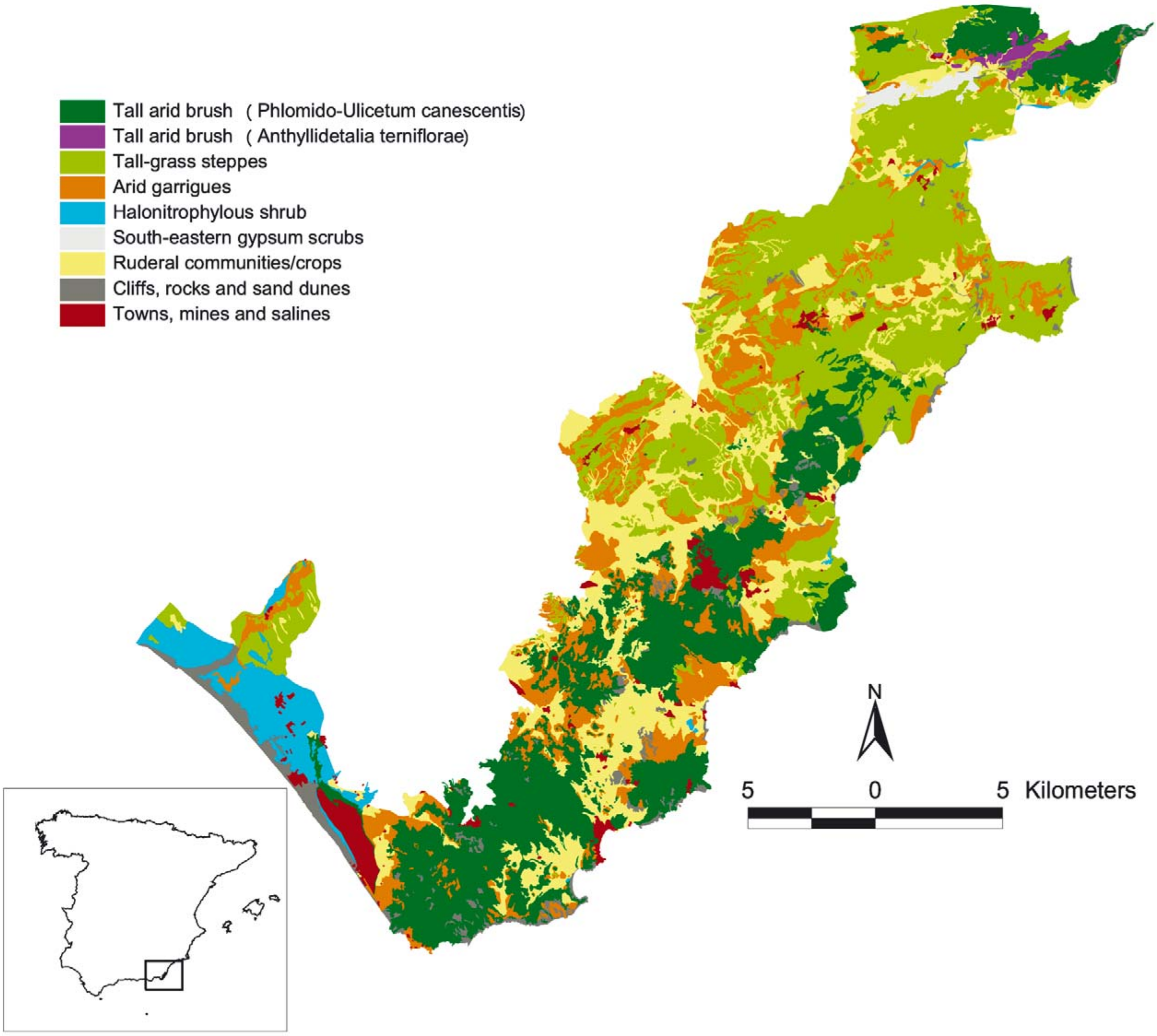


Figure 1. Vegetation cover map of Cabo de Gata-Nijar N. P., SE Spain, in 1994.

in the area between 1900 and 2000 ranged from 144000 to 407000; http://ine.es). Tall grass steppe and tall arid brush were rarely confused even though they were the most difficult to distinguish in the aerial photos (1 out of 35 shrubs and 3 out of 47 steppes were misclassified). In addition, crop fields were very well differentiated from arid garrigues in the aerial photo. Given this fact, we assumed that the photo-interpretation of arid garrigues was accurate, being the difference observed due to the time lag between photos (1994) and field identification (2002).

Derived variables

Topographic variables including elevation, slope angle and slope aspect were derived from the digital elevation model (DEM) using ARCVIEW. Slope aspect was calculated in reference to shoreline to reflect the dominant influence of sea winds, which determine, for example, the presence of *Periploca laevigata* (one of the most frequent shrubs of that vegetation community). Sea oriented slopes (i.e., slopes oriented 135° to the SE), were obtained by subtracting 135° from north orientation ($\Phi = 0°$), where, $\cos(\Phi\text{-}135) = 1$ when sea oriented, and –1 for NW slopes.

We also considered house density (number of houses per km^2), and town distance (distance to the nearest town in km).

Data analysis

We identified the areas that changed and the direction of change using the information about overlying layers of vegetation in different years. We also determined the changes in class in these areas and predicted the direction and magnitude of future changes.

Spatial patchiness was measured and quantified in terms of composition (patch types and abundance) and configurations (patch area and patch aggregation).

Because the term spatial heterogeneity has been used in diverse ways (Kolasa and Rollo 1991), it is important to differentiate when it corresponds to spatial variability (measured using non-spatial statistics) and when it corresponds to spatial patterns (using spatial explicit metrics). We used the latter approach, defining spatial heterogeneity in terms of spatial dependence or spatial autocorrelation (Adler et al. 2001). Strong spatial dependence implies spatial heterogeneity, while spatial independence implies a random pattern. Spatial self-structuring can arises through local interactions between individuals and their neighbours and local dispersion and produces non-random spatial patterns at scales much larger than the scale of the interaction among individuals (Peterson 2002). The destruction of the landscape spatial structure will disrupt self-organization of long standing stable communities (Johnson and Boerlijst 2002). Changes in patch spatial patterns of vegetation among years reflect the change in spatial heterogeneity independent of the underlying topographic variables, which remain invariables along the 40 years of this study. Theses changes are hypothesized to be related with the degree of preservation of each vegetation cover class.

The spatial distribution of patches was analyzed by detrended fluctuation analyses (DFA), which measures the degree of autocorrelation of a spatial sequence (Alados et al. 2003). DFA was used to estimate the degree of spatial aggregation of each land/use cover independent of the scale of measurement. DFA describes habitat connectivity by providing information about fragmentation and spatial autocorrelation as well as patchiness. DFA is described by Peng et al. (1992) and Alados and Weber (1999) and has often been applied in the literature (Peng et al. 1992, 1994; Hausdorff et al. 1995, 1997). It is systematically better than other algorithms that measure fractal correlations (Taqqu et al. 1995) and yields more powerful prognostic information than traditional measures of variability (Mäkikallio et al. 1999; Huikuri et al. 2000).

A binary sequence ($z(i)$) of cover presence was denoted by 1, and absence was denoted by –1 at 25 m intervals (i) taken along continuous transects of 1025 points along the Cabo de Gata-Níjar land cover map. Integration of the sequence $y(t)$, obtained by adding the previous sequence such that $y(t)=\sum_{i=1}^{i=N} z(i)$, provides a graphical representation that can be used to calculate the degree of autocorrelation in the landscape spatial sequence. The entire sequence of data of length N is divided into non-overlapping sub-sequences, or 'boxes', of length b. In each box, we fit a least square line to the data. Let $\hat{y}_b(t)$ be the regression estimate of $y_b(t)$ for each box size b. The fluctuation of $y_b(t)$ about $\hat{y}_b(t)$ was calculated for each data point, and then the average fluctuation for that box size b was determined as

$$F^2(b) = \frac{\sum \left(y_b(t) - \hat{y}_b(t)\right)_2}{N}$$

If this procedure is repeated for various box sizes, b, the residual variances should fit the relation,

$$F(b) \cong b^{\alpha}$$

where $\alpha = 0.5$ indicates no correlations in the sequence (random distribution) and $\alpha \neq 0.5$ indicates long-range power-law correlations. In general α is equivalent to the Hurst exponent H (Hausdorff et al. 1997), and inversely related to the fractal dimension by the relation $H = 2 - D$, for one-dimensional series. We considered boxes of sizes scaled to the nearest integer in the power-law relation 2^2, $2^{2.5}$, 2^3, $2^{3.5}$, ... $2^{9.5}$. Sizes were equally spaced on a log scale to avoid biasing the linear regression in favour of larger boxes. The α values were compared among years using a one-way analysis of variance (ANOVA).

A log likelihood ratio (G-test) was performed to compare land cover changes and vegetation transition rates. The variation in mean patch size among years and cover classes was determined using an ANOVA.

The regression analysis was performed using a random selection of 10% pixels (61872 pixels) to minimize problems with autocorrelation. In order to separate the variation due to physical constraints from the variation due to anthropogenic influence, we performed a step-backward logistic regression analyses between the topographic variables (elevation in m, slope angle, and slope aspect) and geology (volcanic *vs.* non volcanic) as the predictor and vegetation cover class as the dependent variable (presence or absence as 1 or 0). Studentized residuals with absolute value larger than 1.96 were computed and evaluated for each vegetation cover (Cardille et al. 2001). Visual representation of residuals was inspected to determine if spatial autocorrelation existed. The residuals were then regressed against the anthropogenic variables (house density and town distance).

Data on changes in land use over time were obtained by comparing chronologically sequenced coverage. The probability of vegetation transition was calculated by dividing the number of pixels from class *i* to class *j* across the landscape by the total number of class *i* pixels (i.e., the conditional probability of being j given the condition of being *i* in the previous transition period).

Land cover changes between 1985 and 1994 (i.e., 1 if transition took place and 0 otherwise) for the three most abundant cover classes (tall arid brush, tall grass steppe and arid garrigues/extensive crops) were regressed by step-backward logistic regression. Topographic (elevation, slope angle and slope aspect), geological (volcanic *vs.* no volcanic) and anthropogenic (house density and town distance) variables, were used as predictors. Comparisons between the 1994 cover class and those predicted by the model were performed by Tau $_p$ statistic (Ma and Redmond 1995; Cardille et al. 2001).

To determine the probability of degrading or recovering from a previous stage, vegetation cover classes were ordered in successional regression order, from tall arid brush to tall grass steppe and to arid-garrigues/extensive crops. Due to the fast transition between extensive crops and arid garrigues, both classes were joined for the transition matrix analysis. The probability of degrading to a less evolved cover class from 1985 to 1994 was assigned a score of 1 if: i) it belonged to a tall grass steppe cover class in 1994 but was tall arid brush in 1985, or, ii) it belonged to arid-garrigues/cropland cover class in 1994 but was tall arid brush or tall grass steppe in 1985. Otherwise, the probability of degrading was zero. The probability of recovery was also assigned 1 if: i) it belonged to tall arid brush cover class in 1994 but was tall grass steppe or arid-garrigues/cropland in 1985, or ii) it belonged to tall grass steppe in 1994 but was arid-garrigues/cropland in 1985. Otherwise, the probability of recovery was zero.

Results

Land cover change and fragmentation

Tall arid brush of *Phlomido-Ulicetum canescentis* was the best preserved vegetation, occupying 9332.41 ha in 1994. The *Anthyllidetalia terniflorae* brushland community was more scarce, representing only 0.52% of the cover land in the park (201.31 ha). For that reason, this cover is not included in further analyses concerning tall arid brush community. The tall arid brush community decreased between 1957 and 1985, but recovered in 1994 (log likelihood ratio G-test = 16.58, $P < 0.001$; Figure 2). At the other extreme, the tall grass steppes, corresponding with the end successional regression stage of tall arid brush community, occupied 9093.14 ha in 1957, increasing to 11834.12 ha in 1994 (G-test = 252.14, $P < 0.001$). Arid garrigues correspond to early succession stages after abandoning of traditional cereal cropland. Its cover has changed drastically over the last 50 years (G-test = 505.89, $P < 0.001$). Extensive crop fields that occupied 12023.97 ha in 1957 decreased by 45% in 1994, occupying only 6927.77 ha (G-test = 2424.97, $P < 0.001$). The change was parallel to the increase of arid garrigues (Figure 2). Town cover increased as a result of increasing tourism in the area (G-test = 170.53, $P < 0.001$). Other communities were Halonitrophylous shrub, South-eastern gypsum scrubs and the already mentioned *Anthyllidetalia terniflorae*.

Patch size varied significantly among the different land cover ($F_{8,\ 5245} = 5.00$, $P < 0.001$). With the exception of the Gypsum scrubs (mean ± se is 299.74 ± 13.36), tall-arid brushes of *Phlomido-Ulicetum canescentis* were significantly larger than arid garrigues (Tukey test pairwise comparison $P < 0.001$), although no significant differences were observed with tall grass steppes or crop lands.

Patch size did not vary significantly ($F_{2,\ 460} = 0.26$, NS) in the study period (1957, 1985, and 1994) for tall arid brushes of *Phlomido-Ulicetum canescentis*, neither for tall grass steppes ($F_{2,\ 1148} = 0.91$, NS). Only arid garrigues increased significantly with time ($F_{2,\ 1390} = 20.58$, $P < 0.001$), ranging from 7.27 ±

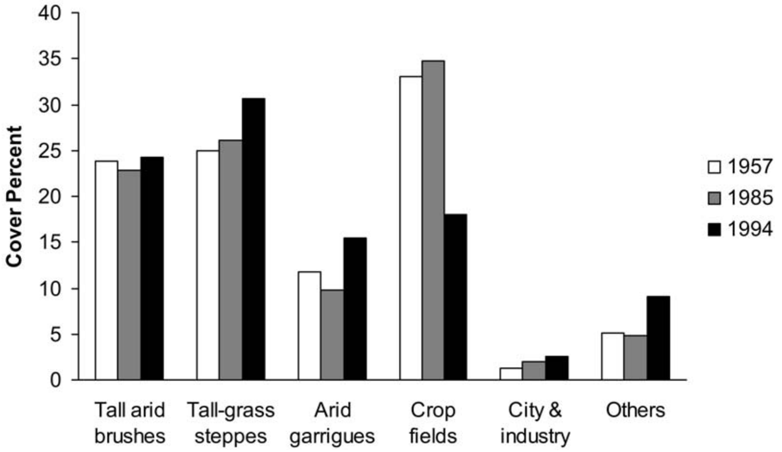


Figure 2. Percent area of different land cover types during the period 1957-1994.

Table 1. Means ± se alpha values calculate by DFA analysis from 26 transects along C. Gata vegetation cover map. Means with different letters are significant at the 0.5 level, DHS Tukey contrast.

	Year 1957	Year 1985	Year 1994	$F_{2,\ 75}$	P
Tall arid brush	1.085 ± 0.01[a]	1.131 ± 0.01 [b]	1.093 ± 0.01 [a b]	3.60	0.03
Tall-grass steppe	0.943 ± 0.01 [a]	0.945 ± 0.02 [a]	1.031 ± 0.02 [b]	7.44	0.001
Arid-garrigues	0.902 ± 0.01 [a]	0.955 ± 0.01 [b]	0.994 ± 0.01 [b]	11.46	0.0001
Crop fields	1.040 ± 0.01 [a]	1.075 ± 0.01 [b]	1.052 ± 0.01 [a b]	3.09	0.05

0.69 ha to 20.8 ± 3.98 ha. The average patch size of the crop field was also similar for the study period ($F_{2,\ 745} = 0.03$, NS).

Alpha values obtained using DFA represent the degree of aggregation (inversely related with fractal dimension) of each cover class, which is larger with increasing values of alpha. The degree of aggregation increased significantly with time for tall grass steppe and arid garrigues (low fractal dimension), in comparison with tall arid brushes, that increased in 1985 but declined again in 1994 (Table 1).

Topographic and anthropogenic factors determining land cover

An attempt was made to separate the driving forces of landscape evolution from the physical boundary conditions and to understand the processes that generated the observed patterns. To do this we carried out a logistic regression using the topographic variables (elevation, slope angle and slope aspect) and geology (volcanic vs. no volcanic) as predictors with the separate dependent variables 1994-tall arid brush, 1994-tall grass steppe and 1994-arid garrigues covers. Tall arid brush was significant and positively related with elevation and slope (Table 2). Indeed, volcanic substrate favoured tall arid brush cover. Tall grass steppe was also positively related with elevation, but slope had a negative influence on the establishment of tall grass steppe and arid garrigues. In addition, sea oriented slopes favoured the presence of tall grass steppe but selected against tall arid brush and arid-garrigues. Volcanic substrate was negatively related with tall grass steppe cover.

Significant studentized residuals (i.e., larger than 1.96) represented the probability of belonging to a specific cover that is not explained by topographic and geologic variables. We investigated the relationship of the significant residuals (Figure 3) to the anthropogenic variables (house density, number of houses per km^2 and town distance, distance to the nearest town in km). As summarized in Table 3, tall arid brush increased significantly as house density decreased and distance to the town increased. Conversely, tall-arid steppe and arid garrigues increased significantly, as town distance declined and house density decreased.

Table 2. Logistic regression analysis for landscape cover and topographic variables as predictor in three different communities of Cabo de Gata N.P in 1994.

Predictors	Tall arid brush Log-like. χ^2 = 21623.74*** d = 4		Tall-grass steppe Log-like. χ^2 = 3518.144*** df = 4		Arid-garrigues Log-like. χ^2 = 1798.18*** df = 4	
	β	Wald	β	Wald	β	Wald
Constant	− 4.254	13826.06***	− 0.906	2310.99***	− 1.317	3190.54***
Elevation	0.003	633.06***	0.003	868.24***	0.0004	10.61**
Slope angle	0.108	6311.85***	− 0.002	3.36	− 0.054	1106.86***
Aspect (sea oriented)	− 0.041	5.99*	0.177	193.82***	− 0.295	329.87***
Volcanic	1.740	4536.15***	− 0.990	2166.45***	0.234	79.98***

* $P \leq 0.05$, ** $P \leq 0.01$, *** $P \leq 0.001$.

Main factors influencing changes in land cover

Comparisons of transition probabilities for the three main succession stages (tall arid brush, tall grass steppe and arid garrigues/extensive crop) revealed a significant increase in the probability of permanency of tall arid brush (log likelihood ratio *G*-test = 73.67, $P < 0.001$) and tall grass steppe (*G*-test = 60.88, $P < 0.001$) in 1985-1994 (Table 4). On the contrary, the probability of permanence of arid garrigues/extensive crops decreased (*G*-test = 853.04 $P < 0.001$). The probability of switching towards a more degraded community (arid garrigues/extensive crop) was larger between 1957-85 than 1985-994 (*G*-test = 313.24, $P < 0.001$ when developing from tall arid brush, *G*-test = 304.35, $P < 0.001$ from tall grass steppe). The probability of recovery from any of the more degraded stages of the vegetation increased from 1985-94 for arid garrigues to tall arid brush (*G*-test = 136.98, $P < 0.001$), and to tall grass steppe (*G*-test = 288.84, $P < 0.001$).

The vegetation transition graph is presented in Figure 4. We constructed the vegetation stage-class matrices for the Cabo de Gata-Níjar N.P. landscape using the data from a transition matrix between 1985 and 1994, which reflects the actual driving forces. A cover class grouping *Anthyllidetalia termiflorae* shrubs, halonitrophylous shrubs, gypsum scrubs, cliffs, rocks and dunes, and towns and industry were included into the projection matrix as ‘Others’ for analysis. Projection of the 1985-94 transition matrix through eight consecutive transition periods of nine years each revealed an increase in tall grass steppe and a decline in arid garrigues/croplands, until they stabilize around 2020. After an initial increase, tall arid brush was maintained (Figure 4).

To evaluate land cover transition model between 1985 and 1994 we calculated the coefficient of the logistic regression analysis for each transition cover (Table 5). The predicted observation for 1994 classified 76% more pixels correctly than would be expected by random assignment ($\text{Tau}_p = 0.76$). There was good agreement between the observed and predicted cover maps (Figure 5).

To understand the main causes of transition since 1985, we calculated the probability of transition to a more degraded cover class. The results of the logistic regression analysis between the probability of degrading and the topographic variables and geological substrate are presented in Table 6 and Table 7. There was a negative relationship with elevation and slope angle (Table 6; steeper slopes and higher elevations had lower degradation probability). Sea oriented slopes were also negatively related with the probability of transition to a more degraded community. Studentized residual maps are presented in Figure 6. Analysis of residuals revealed that the probability of degrading to a lower cover class was higher at increased house density. Indeed, the transition to a more degraded cover class was higher closer to the town (Table 7). The probability of recovery from tall grass steppe to tall arid brush, or from arid-garrigues/cropland to either tall-arid steppe or tall arid brush between 1985 and 1994 was positively related with slope (Table 6), in concordance with land abandonment at high slopes. Volcanic substrate also favoured vegetation recovery. Anthropogenic variables significantly influenced the capability of vegetation recovery (Table 7). As expected, recovery increased further from the town and at lower house densities.

Discussion

The most important social and economic changes that have taken place at the Cabo de Gata-Níjar N. P. be-

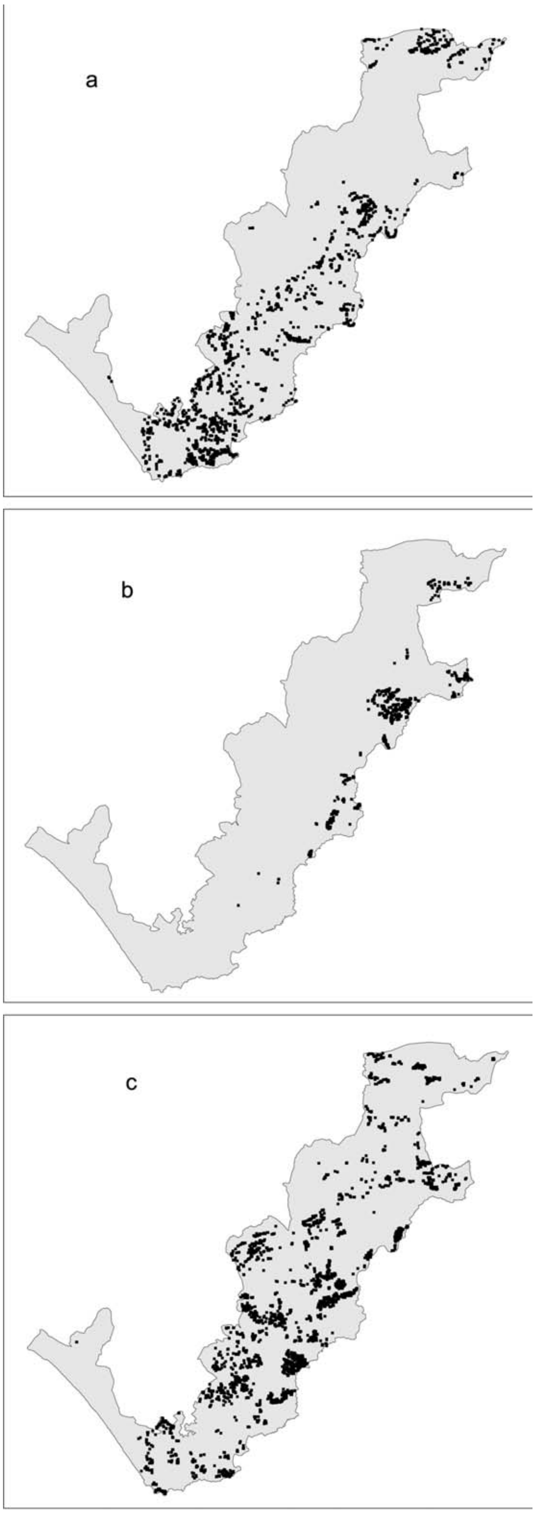


Figure 3. Significant studentized residuals of logistic topographic model for: (a) tall arid brush; (b) tall grass steppe; and c) arid garrigues covers.

tween 1957 and 1994 were that around 45% of croplands have been abandoned since 1957 and that the area was proclaimed a Natural Park in 1987. As a result, the evolution of the landscape can be divided into two distinct periods. First, land use was more or less stable between 1957-1985, without significant area shifts but with an emergent fragmentation process of natural vegetation. Second, from 1985 to 1994, the vegetation dynamics changed towards recovery of vegetation cover as a consequence of cropland abandonment and land use regulations for Natural Parks (PORN 1996). Changes in land cover/use between 1957 and 1994 included a 19% increase of tall grass steppe and 24% of arid garrigues as a result of abandonment. During the 37 years included in this study, tall arid brush has remained (with only a small net increase, especially after the area was declared a Natural Park in 1987). The percentage of towns and buildings increased significantly, but were quite negligible with regard to the whole study area (only 2.5%).

Average patch size increased between 1957 and 1994 for arid garrigues and tall grass steppe. As a result, the number of patches decreased because small patches begin to coalesce into larger ones (Gustafson and Parker 1992). No clear trend was observed for tall arid brush. Although classic fragmentation indices provide useful results to characterize landscape patterns, they lack spatial information (the distribution of patches in the landscape). Hargis et al. (1998) examined the behaviour of landscape metrics in nine series of generated landscapes varying in fragmentation intensity. They considered edge density, contagion, mean nearest neighbour distance, mean proximity index, perimeter-area fractal dimension and mass fractal dimension but none of the measures could differentiate between dispersed and aggregated landscapes. The authors concluded that the current landscape metrics do not quantify spatial arrangement. Indeed, Schumaker (1996) observed a weak correlation between landscape indices and habitat connectivity. Habitat connectivity has been identified as one of the most important factors preserving dispersion among populations (Taylor et al. 1993; Pearson et al. 1996; With et al. 1997). Landscape connectivity depends on the degree of habitat loss as well as the degree of aggregation (fractal dimension). Clumped landscapes (low fractal dimension) remain connected across a larger range of habitat loss than in a landscape with high fractal dimension (With et al. 1997; Gustafson 1998; With and King 1999b).

Table 3. Linear regression analysis for the plant cover residuals and the anthropogenic variables (house density per km^2, and town distance in m) for tall arid shrub, tall-grass steppe and arid garrigues in 1994.

Predictors	Tall arid brush F=287.89*** df = 2, 59974		Tall-grass steppe F=479.95*** df = 2, 59974		Arid-garrigues F = 57.55*** df =2, 59974	
	β	t	β	t	β	t
Constant	− 0.051 ± 0.003	− 17.45***	0.068 ± 0.004	18.00***	0.018 ± 0.003	5.93***
Town distance	0.018 ± 0.001	21.86***	− 0.018 ± 0.001	− 16.30***	− 0.0046 ± 0.001	− 5.25***
House density	− 0.00034 ± 0.00	− 4.44***	− 0.0029 ± 0.000	− 29.45***	− 0.00082 ± 0.000	− 10.33***

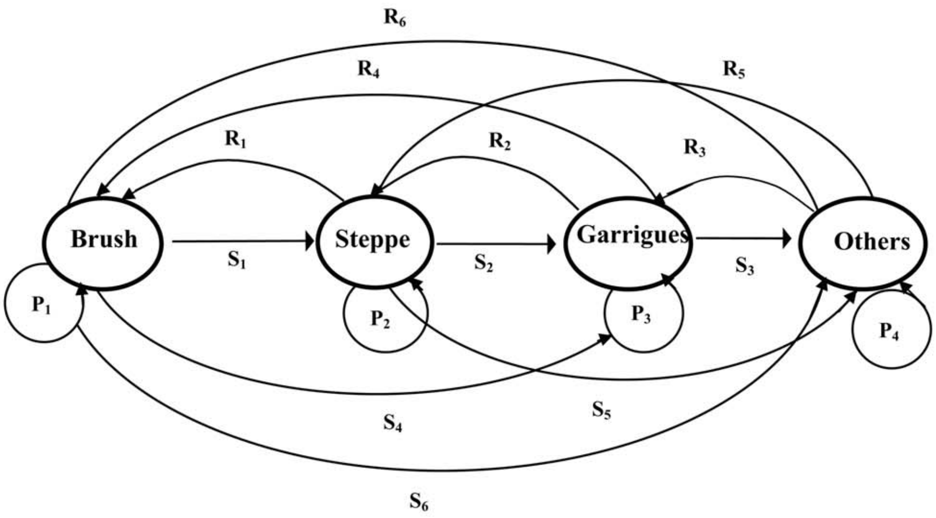
Table 4. Transition probabilities between 1957 and 1985 and between 1985 and 1994 along the vegetation succession from tall arid brush, tall grass steppe and arid garrigues/crop lands of Cabo de Gata N.P.

	Tall arid brush	Tall grass steppe	Arid garrigues/crops
1957-1985			
Tall arid brush	0.729	0.031	0.182
Tall grass steppe	0.031	0.770	0.188
Arid garrigues/crops	0.087	0.117	0.761
1985-1994			
Tall arid brush	0.822	0.042	0.100
Tall grass steppe	0.023	0.852	0.110
Arid garrigues/crops	0.120	0.176	0.661

According to the spatial aggregation measured by DFA, the alpha exponent for tall grass steppe and arid garrigues increased evenly in the past 37 years. The results agree with previous reports of recovery for these formations. The distribution of tall arid brush, on the contrary, was only more clumped in 1985, compared with 1957, but that was reversed in 1994.

On the other hand, comparisons among land cover classes revealed that, in terms of patch size and spatial aggregation (alpha values), tall arid brush (the better preserved vegetation cover of the area) was less fragmented than tall grass steppe and arid garrigues, which represent the degradation stages of the Mediterranean-type ecosystem (DiCastri et al. 1981; Le Houérou 2001). Other authors have also observed a correlation between fractal dimension and the degree of human transformation of the landscape (Krummel et al. 1987; O'Neill 1989). In a parallel study in this area, we have also observed that fragmentation of natural vegetation is higher as a result of recent tourist development compared with traditional land uses (Pueyo et al. in prep). These results agree with previous studies (Alados et al. 2003, in press) and with the hypothesis that plant spatial patterns are the result of spatial self-organization processes (Rietkerk et al. 2002) under the constraints of other processes such as landscape heterogeneity (Pan et al. 1999).

Analysis of the processes that have generated the given patterns revealed that, as expected, tall arid brush (the climax community of this area), occurred at high elevation and steep slopes not oriented towards the sea shore. Volcanic soil substrate (more nutrient rich than calcareous soil substrate) also favoured tall arid brush formation. Tall grass steppe was found on more gradual slopes, at high elevation, and sea oriented, while arid garrigues were found at low elevation and not sea oriented. Our results confirmed the importance of the topographical variables when explaining patterns of plant communities' distribution (Swanson et al. 1988; Davis and Goetz 1990). The reason is that vegetation growth is affected by environmental variables, such as nutrient availability, soil moisture or sun light, which are affected by topographical variables (Bolstad et al. 1998). The brushwood community is more suited to very rock slopes that retard surface runoff, giving smaller interrill runoff discharges and greater infiltration rates (Cerdá 2001), providing many crevices and fissures where the thicker roots of shrubs can penetrate the soil (Hodgson et al. 1994; Guerrero-Campo et al. 1999). Rain simulation experiments in south of Spain reported that when rock fragments were removed, infiltration rate diminished 50% and the erosion rate increased 39 times in comparison with not removed



1985-94 Transition matrix

$$\begin{pmatrix} P_1 & R_1 & R_4 & R_6 \\ S_1 & P_2 & R_2 & R_5 \\ S_4 & S_2 & P_3 & R_3 \\ S_6 & S_5 & S_3 & P_4 \end{pmatrix} = \begin{pmatrix} 0.822 & 0.023 & 0.120 & 0.177 \\ 0.042 & 0.852 & 0.176 & 0.093 \\ 0.100 & 0.110 & 0.661 & 0.136 \\ 0.035 & 0.015 & 0.042 & 0.594 \end{pmatrix}$$

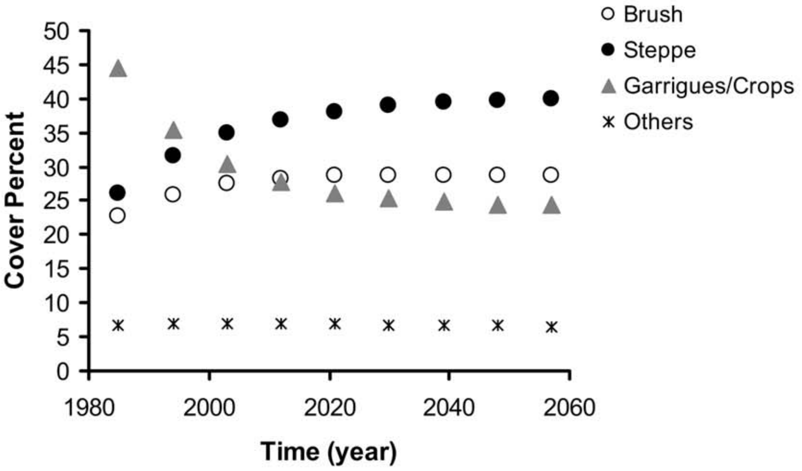


Figure 4. (a) Vegetation succession graph where *P* is the probability of remaining in the same vegetation-class (P_1 as a tall arid brush, P_2 as tall grass steppe, P_3 as arid garrigues/extensive crops, P_4 as other, which include te remainder cover classes). *R* is the probability of vegetation recover (R_1 is from tall grass steppe to tall arid brush, R_2 is from arid garrigues/extensive crops to tall grass steppe, and R_3 is from others to arid garrigues/extensive crops to tall arid brush, R_4 is from arid garrigues/extensive crops to tall arid brush, R_5 is from others to tall grass steppe, and R_6 is from others to tall arid brush). *S* is the probability of succession to the next vegetation stage (S_1 is from tall arid brush to tall grass steppe, S_2 is from tall grass steppe to arid garrigues/extensive crops, S_3 is from arid garrigues/extensive crops to others, S_4 is from tall arid brush to arid garrigues/extensive crops, S_5 is from tall grass steppe to others, and S_6 is from tall arid brush to others). (b) Vegetation-cover projection.

Table 5. β exponent from the logistic regression analysis for transition probabilities among vegetation cover between 1985 and 1994, and topographic and anthropogenic variables. Only significant variables included.

β exponent	Predictors	Tall arid brush	Tall grass steppe	Arid garrigues/crops
Tall arid brush	Constant	− 0.577	− 1.192	0.219
	Elevation	0.002	–	− 0.003
	Slope	0.054	− 0.037	− 0.078
	Aspect	0.072	–	− 0.283
	Volcanic	0.632	− 1.159	− 0.424
	T. distance	0.045	− 0.152	− 0.073
	H. density	− 0.016	–	0.007
Tall grass steppe	Constant	− 6.296	1.709	− 1.698
	Elevation	− 0.001	0.002	− 0.001
	Slope	0.093	− 0.005	− 0.028
	Aspect	− 0.369	0.088	− 0.114
	Volcanic	0.643	− 0.369	0.553
	T. distance	0.401	− 0.086	− 0.034
	H. density	–	− 0.013	0.010
Arid garrigues/crops	Constant	− 4.061	− 1.802	1.982
	Elevation	0.001	0.002	− 0.002
	Slope	0.135	0.015	− 0.114
	Aspect	0.102	0.159	− 0.212
	Volcanic	1.043	0.065	− 0.299
	T. distance	− 0.061	–	− 0.032
	H. density	− 0.014	− 0.028	0.002

rock experiment (Cerdá 2001). Slopes that are more gradual maintain sediments around the tussocks *S. tenacissima* (Cerdá 1997) favouring the spatial propagation of its rhizomes and the establishment of tall grass steppe. According with Berael et al. (1995) 10% was the critical slope angle for incipient rills formation on very stony slopes, and 4% was the critical slope angle below which sedimentation occurred.

Anthropogenic influence was analyzed after removing the variance due to topographical and geological factors, revealing that closer distance to the town favoured the most degraded formations (tall grass steppe and arid garrigues). Tall arid brush was more developed further from the town and at lower house density per square km. Thus, vulnerability to human activities was different between the two main vegetation types in the study area. Indeed, high house density did not favour any of the natural vegetation formations. When grazing disturbance is high, close to the villages and settlements, the decreased and scattered distribution of brushwood community favoured the invasion of the colonizing grazing-tolerant species, such as steppe grasses, well represented by *S. tenacissima*.

The capacity of the system to self-organize and adapt is an intrinsic property that depends on species composition (Tilman et al. 1996, 2001; Naeem and Li 1997) and successional dynamics (Burrows 1990; Connell and Slatyer 1977; Pickett et al. 1987). In semi-arid regions, such as Cabo de Gata, disturbance is responsible for particular vegetation changes in the natural vegetation, where arid garrigues (seral-thyme-brushwood) and tall grass steppe of *S. tenacissima* finally develop (Tomaselli 1981; Peinado et al. 1992; Alados et al. 2003). In addition, soil erosion and runoff distribution is directly related with the spatial distribution of the vegetation (Cerdá 1997; Sole Benet et al. 1997; Martinez Mena et al. 2000, 2001). After land abandonment soil erosion is very high during the first 3 years after abandon, but when vegetation cover of *Artemisia herba-alba* community develops (after 10 years of abandonment), water infiltration increases, reaching the highest values with the *S. tenacissima* community (Cerda 1997; Martinez Mena et al. 2000, 2001).

To assess the effect of perturbation on an ecosystem it is important to know the transition probabilities among different vegetation cover. The probability of permanency as tall arid brush and as tall grass steppe increased approximately 10% between 1957-85 and 1985-94, while the probability of remaining as arid garrigues/croplands declined in a similar quantity, being the abandon of rural areas one of the main driving forces of this human-dominated

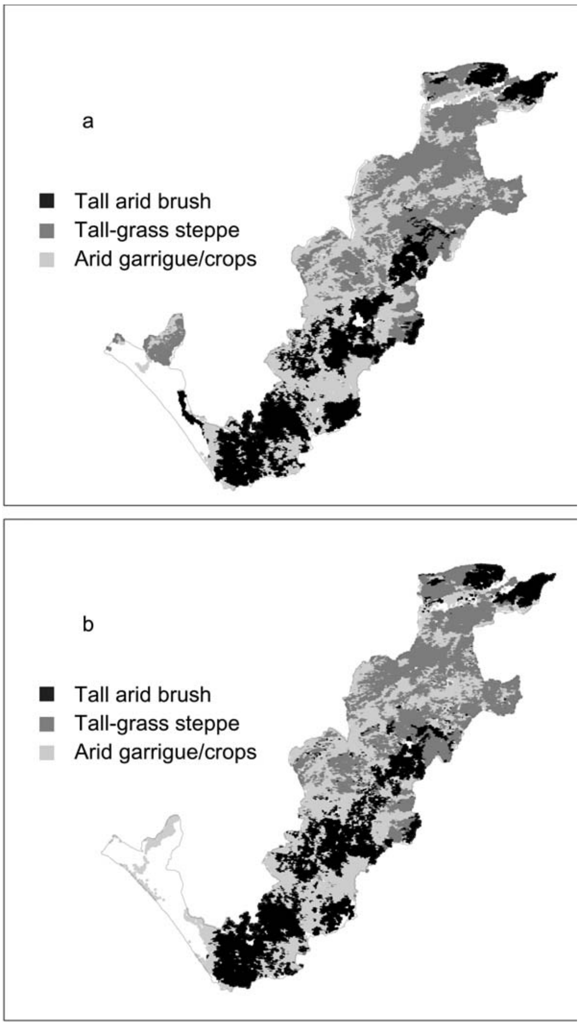


Figure 5. (a) Observed vegetation cover (tall arid brush, tall grass steppe and arid garrigues/extensive crops) in 1994. (b) Predicted vegetation cover (tall arid brush, tall grass steppe and arid garrigues/extensive crops) from the 1985-1994 transition model.

Figure 6. Significant studentized residuals of logistic topographic model for: (a) Recovering probability map; (b) Degrading probability map.

landscape (Castro and Guirado 1993; Guirado and Mendoza 2000; Rodríguez and Sánchez 2000).

The probability of degrading from tall arid brush or tall grass steppe to arid garrigues/croplands declined by approximately 8% from 1957-85 to 1985-94. Conversely, the recover probability from arid garrigues/croplands to tall arid brush or to tall grass steppe increased ca. 3% and 7% respectively.

Comparisons were carried out between the probabilities of degrading/recovering with the topographic and anthropogenic variables to elucidate the main causes determining the transition probability of degrading and recovering between 1985 and 1994 (after the establishment of the land use regulations in Cabo de Gata-Níjar Natural Park in 1987). We observed that the higher elevations and steeper slopes within our study area were less likely to be degraded and were more likely to recover. Sea oriented slopes receive the morning dew as an important water resource that mitigate water stress in this xeric area, being less prone to degradation than no sea oriented slopes. Conversely, slower vegetation recovers and larger hydraulic stress associated with low aggregation stability in more xeric slopes were also observed after 10 years experiments in south-eastern Spain (Cerdá 1998). As expected, high house density promoted degradation, while distance to the town

Table 6. Logistic regression analysis for probability of degrading/recovering and topographic variables as predictor of Cabo de Gata N. P between 1985 and 1994.

Predictors	Degrading Log-like. $\chi^2 = 550.84^{***}$ df = 4		Recovering Log-like. $\chi^2 = 3214.30^{***}$ df = 2	
	β	Wald	β	Wald
Constant	− 1.091	537.60***	− 2.387	9366.00***
Elevation	− 0.002	116.60***	–	–
Slope angle	− 0.036	267.03***	0.042	743.91***
Aspect (sea oriented)	− 0.168	43.38***	–	–
Volcanic	0.193	22.94***	1.137	1741.74***

* $P \leq 0.05$, ** $P \leq 0.01$, *** $P \leq 0.001$.

Table 7. Linear regression analysis for probability of degrading/recovering residuals between 1985 and 1994, and the anthropogenic variables (house density per km^2, and town distance in km).

	Degrading from brush F=38.51*** df =2, 27653		Recovering F=66.79*** df =2, 39882	
	β	t	β	t
Constant	0.009 ± 0.004	2.23*	− 0.00009 ± 0.004	− 0.24
Town distance	− 0.004 ± 0.001	− 3.55***	0.003 ± 0.001	2.32*
House density	0.001 ± 0.0001	7.24***	− 0.001 ± 0.0001	− 10.56***

* $P \leq 0.05$, ** $P \leq 0.01$, *** $P \leq 0.001$.

favoured recovery, supporting the hypothesis that colonization of tall arid brush by *S. tenacissima* is favoured by human impact. This can be related to previous studies on grazing (the main activity on mountain slopes), which has been found to induce a directional effect on vegetation of semiarid land (Fuhlendorf et al. 2001), leading to a successional regression from tall arid brush to tall grass steppe (Alados et al. 2003).

In summary, the vegetation dynamics of semiarid marginal areas from south-eastern Spain can allow the recovery from arid garrigues formation after land abandonment (towards tall grass steppes), in spite of the extensive husbandry in the area. It is important to note that the grazing pressure in the area (0.68 goats/ha) was under the estimated carrying capacity (0.72 goats/ha; Robles et al. 1997). Nevertheless, results from previous studies confirm that effective stocking rates above 0.5 individuals ha^{-1} year^{-1} would increase the risk of vegetation regression from tall arid brush to tall grass steppe (Alados et al. 2003).

According to Turner et al. (1993), the ability of an ecosystem to recover from disturbance depends on the ratio of the disturbance interval to the time required to return to the mature stage, and on the ratio of the size of the disturbance to the size of the landscape. When the disturbance interval is shorter than the recovery time and a large portion of the landscape is affected, the system may shift into a different trajectory. Projection of the transition matrix over 100 years confirmed the prevalent trend observed in the 40 years considered in this study. Briefly, it consisted in the reduction of arid garrigues/croplands until they stabilized around 20% cover in 2020, the increase of tall grass steppe to 35% cover in the same period, and the permanency of brushwood to the amount observed in the present period.

Hence, it can be concluded that land cover changes in Cabo de Gata-Níjar underwent two important processes. First, the reduction of croplands and arid garrigues as a result of land abandonment, and second, an evolution towards tall grass steppes as a consequence of vegetation successional dynamics (from garrigue to steppe) and vegetation successional regression (from brushwood to steppe). The regeneration of tall arid brush from cropland abandonment appears quite improbable, although vegetation cover is preserved due to the favourable effect of stepped terrain in the brushwood formation. Finally, the increase of tourist infrastructures will increase the actual fragmentation of the Cabo de Gata-Níjar tall arid brush.

Acknowledgements

The work was supported by the EU under its INCO-DC Program, contract number ERBIC18-CT98-0392: DRASME (Desertification risk Assessment in Silvopastoral Mediterranean Ecosystems) and by Spanish CICYT program, project number REN2002-04668/GLO (Efectos de la fragmentacióny uso del suelo en la conservación y dinámica de las zonas esteparias Mediterráneas). The support from both programs is gratefully acknowledged.

References

Adler P.B., Raff D.A. and Lauenroth W.K. 2001. The effect of grazing on the spatial heterogeneity of vegetation. Oecologia 128: 465–479.

Aidoud A., Aidoud-Lounis F. and Slimani H. 1998. Effects of grazing on soil and desertification: a review from the southern Mediterranean rim.. In: Papanastasis V.P. and Peter D. (eds), Ecological basis of livestock grazing in Mediterranean ecosystems. European Commission EUR 18308, Brussels, Belgium, pp. 133-148.

Alados C.L., Pueyo Y., Giner M.L., Navarro T., Escos J., Barroso F., Cabezudo B. and Emlen J.M. 2003. Quantitative characterization of the regressive ecological succession by fractal analysis of plant spatial patterns. Ecological Modelling 163: 1–17.

Alados C.L., ElAich A., Papanastasis V.P., Ozbek H., Navarro T., Freitas H., Vrahnakis M., Larrosi D. and Cabezudo B. 2004. Change in plant spatial patterns and diversity along the successional gradient of Mediterranean grazing ecosystems. Ecological Modelling, (in press).

Alados C.L. and Weber D.N. 1999. Lead effects on the predictability of reproductive behavior in fathead minnows (*Pimephales promelas*): a mathematical model. Environmental Toxicology and Chemistry 18: 2392–2399.

Arianoutsou-Faraggitaki M. 1985. Desertification by overgrazing in Greece: the case of Lesvos Island. Journal of Arid Environment 9: 237–242.

Barberá G.G., López-Bermúdez F. and Romero-Días A. 1997. Cambios de uso del suelo y desertificación en el Mediterráneo: el caso del Sureste Ibérico.. In: García-Ruiz J.M. and López-García P. (eds), Acción humana y desertificación en ambientes mediterráneos, Instituto Pirenaico de Ecología: Zaragoza, Spain, pp. 9-39.

Berael C., Poesen J., Van Wesemael B., Fantechi R., Peter D., Balabanis P. and Rubio J.L. 1995. Rill and gully erosion in relation to rock fragment cover and surface slope along a hill slope in south eastern Spain.. In: Desertification in a European context: physical and socio economic aspects. Proceedings of the European School of Climatology and Natural Hazard. El Campello. Pueblo Acantilado, 6-13 October 1993, Alicante, Spain, pp. 371-375.

Bolstad P.V., Swank W. and Vose J. 1998. Predicting Southern Appalachian overstory vegetation with digital terrain data. Landscape Ecology 13: 271–283.

Burrows C.J. 1990. Processes of vegetation change. Unwin Hyman: London, UK.

Cammeraat L.H., Willott S.J., Compton S.G. and Incoll L.D. 2002. The effects of ants' nests on the physical, chemical and hydrological properties of a rangeland soil in semi-arid Spain. Geoderma 105: 1–20.

Cardille J.A., Ventura S.J. and Turner M.G. 2001. Environmental and social factors influencing wildfires in the Upper Midwest, Unites States. Ecological Applications 11: 111–127.

Castro H. and Guirado J. 1993. El Plan de Ordenación de los Recursos y el Plan de Desarrollo Integral del Parque Natural marítimo-terrestre del Cabo de Gata-Níjar.. In: Activitats Economiques I Espais Protegits. Monografies, 23. Diputación de Barcelona. Servei de Parcs Naturales: Barcelona, Spain.

Cerdá A. 1997. The effect of patchy distribution of *Stipa tenacissima* L. on runoff and erosion. Journal of Arid Environments 36: 37–51.

Cerdá A. 1998. The influence of aspect and vegetation on seasonal changes in erosion under rainfall simulation on a clay soil in Spain. Canadian Journal of Soil Science 78(2): 321–330.

Cerdá A. 2001. Effects of rock fragment cover on soil infiltration, interrill runoff and erosion. European Journal of Soil Science 52: 59–68.

Chapman R.W. 1978. The evidence for prehistoric water control in south-east Spain. Journal of Arid Environment 1: 261–274.

Congalton R.G. 1991. A review of assessing the accuracy of classifications of remotely sensed data. Remote Sensing and Environment 37: 35–46.

Connell J.H. and Slatyer R.O. 1977. Mechanism of succession in natural communities and their role in community stability and organisation. The American Naturalist 111: 1119–1144.

CORINE 1991. Corine biotopes manual. Commission of the European Communities: Brussels, Belgium.

Davis F.W. and Goetz S. 1990. Modelling vegetation pattern using digital terrain data. Landscape Ecology 4: 69–80.

DiCastri G., Goodall D.W. and Specht R.L., 1981. Mediterranean type shrublands. Ecosystems of the World 11. Elsevier, Amsterdam, The Netherlands, 644 pp.

Edwards P.J., May R.M. and Webb N.R. 1994. Large-scale ecology and conservation biology. Blackwell Scientific Publications: Oxford, UK.

Fuhlendorf S.D., Briske D.D. and Smeins F.E. 2001. Herbaceous vegetation change in variable rangeland environments: The relative contribution of grazing and climatic variability. Applied Vegetation Science 4: 177–188.

Guerrero-Campo J., Alberto F., Hodgson J., Garcia Ruiz J.M. and Montserrat Marti G. 1999. Plant community patterns in a gypsum area of NE Spain. I. Interactions with topographic factors and soil erosion. Journal of Arid Environments 41: 401–410.

Guirado J. and Mendoza R. 2000. La regeneración del medio forestal almeriense.. In: Grupo Ecologista Mediterrá neo (eds), Desertificación de Almería, Artes Gráficas: Almería, Spain, pp: 101-110.

Gustafson E.J. and Parker G.R. 1992. Relationships between landcover proportion and indices of landscape spatial pattern. Landscape Ecology 7: 101–110.

Gustafson E.J. 1998. Quantifying landscape spatial pattern: what is the state of the art? Ecosystems 1: 143–156.

Groom M.J. and Schumaker N. 1993. Evaluating landscape change: Patterns of worldwide deforestation and local fragmentation.. In: Kareiva P.M., Kingsolver J.G. and Huey R.B. (eds), Biotic Interactions and Global Change, Sinauer Associates: Sunderland, Massachusetts, USA, pp 24-44.

Hadar L., Noy-Meir I. and Perevolotsky A. 1999. The effect of shrub clearing and grazing on the composition of a Mediterranean plant community: functional groups versus species. Journal of Vegetation Science 10: 673–682.

Hargis C.D., Bissonette J.A. and David J.L. 1998. The behavior of landscape metrics commonly used in the study of habitat fragmentation. Landscape Ecology 13: 167–186.

Hausdorff J.M, Peng C.K., Ladin Z., Wei J.Y. and Goldberger A.L. 1995. Is walking a random walk? Evidence for long-range correlations in stride interval of human gait. Journal of Applied Physiology 78: 349–358.

Hausdorff J.M., Mitchell S.L., Firtion R., Peng C.K., Cudkowicz M.E., Wei J.Y. and Goldberger A.L. 1997. Altered fractal dynamics of gait: reduced stride-interval correlations with aging and Huntington's disease. Journal of Applied Physiology 82: 262–269.

Hessen I. 1999. Life strategies in a semi-arid grassland community mechanisms of dispersal and reproduction within Lapiedro-martinezii-stipetum tenacissimae (southeastern Spain). Feddes Repertorium, 110: 265–285.

Hodgson J. Montserrat G., Alberto F., García Ruiz J.M. Guerrero J. and Colasanti R. 1994. A comparison of the functional characteristics of plants from sedimenting and eroded areas with particular reference to the gypsum hills of the Ebro Depression.. In: Arnáez J., García Ruiz J.M. and Gómez Villar A. (eds), Geomorfología en España, Vol. 2, Sociedad Española de Geomorfología: Logroño, Spain, pp. 239-251.

Huebner C.D., Vankat J.L. and Renwick W.H. 1999. Change in the vegetation mosaic of central Arizona USA between 1940 y 1989. Plant Ecology 144: 83–91.

Huikuri H.V., Mäkikallio T.H., Peng C.- K., Goldberger A.L., Hintze U. and Møller M., 2000. Fractal correlation properties of R-R interval dynamics and mortality in patients with depressed left ventricular function after an acute myocardial infarction. Circulation 101: 47–53.

Ispikoudis I., Lyrintzis G. and Kyriakakis S. 1993. Impact of human activities on Mediterranean Landscapes in Western Crete. Landscape and Urban Planning, 24: 259–271.

Iverson L.R. 1988. Land-use changes in Illinois, USA: The influence of landscape attributes on current and historic land use. Landscape Ecology 2: 45–61.

Johnson C.R. and Boerlijst M.C. 2002. Selection at the level of community: the importance of spatial structure. Trends in Ecology and Evolution 17: 83–90.

Kareiva P., Kingsolver J.G. and Huey R.B. 1993. Biotic interactions and global change. Sinauer Associates: Sunderland, Massachusetts, USA.

Kolasa J., and Rollo C.D. 1991. Introduction: the heterogeneity of heterogeneity, a glossary.. In: Kolasa J. and Pickett S.T.A. (eds), Ecological heterogeneity, Springer, Berlin Heidelberg New York, pp. 1-21.

Krummel J.R., Gardner R.H., Sugihara G., O'Neill R.V. and Coleman P.R. 1987. Landscape patterns in a disturbed environment. Oikos 48: 321–324.

Lande R. 1987. Extinction thresholds in demographic models of territorial populations. The American Naturalist 130: 624–635.

Le Houérou H.N. 2001. Biogeography of the arid steppeland north of the Sahara. Journal of Arid Environment 48: 103–128.

Levin S.A. and Paine R.T. 1974. Disturbance, patch formation and community structure. Proceedings of the National Academy of Sciences 71: 2744–2747.

Lyrintzis G.A. 1996. Human impact trend in Crete: the case of Psilorites Mountain. Environ. Conservation 23: 140–148.

Ma Z. and Redmond R.L. 1995. Tau coefficient for accuracy assessment of classification of remote sensing data. Photogrammetric Engineering Remote Sensing 61: 435–439.

Mäkikallio T.H., Høiber S., Køber L., Torp-Pedersen C., Peng C.-K., Goldberger A.L., Huikuri H.V. and the TRACE Investigators. 1999. Fractal analysis of heart rate dynaimis as a predictor of mortality in patients with depressed left ventricular function after acute myocardial infarction. American Journal of Cardiology 83: 836–839.

Martinez Mena M., Rogel J.A., Albaladejo J., Castillo V.M. 2000. Influence of vegetal cover on sediment particle size distribution in natural rainfall conditions in a semiarid environment. Catena 38(3): 175–190.

Martinez Mena M., Castillo V., Albaladejo J. 2001. Hydrological and erosional response to natural rainfall in a semi-arid area of south-east Spain. Hydrological Processes 15(4): 557–571.

Mota J.F., Peñas J., Castro H. and Cabello J. 1996. Agricultural development *vs*. biodiversity conservation: the Mediterranean semiarid vegetation in El Ejido (Almería, southeastern Spain). Biodiversity and Conservation 5: 1597–1617.

Mota J.F., Cabello J., Cueto M., Gómez F., Giménez E. and Peñas J. 1997. Datos sobre la vegetación del Sureste de Almería. Universidad de Almería: Almería, Spain.

Naeem S. and Li S. 1997. Biodiversity enhances ecosystem reliability. Nature 390: 507–509.

O'Neill R.V. 1989. Perspectives in hierarchy and scale.. In: Roughgarden J., May R.M. and Levin S.A. (eds), Perspectives in Ecological Theory, Princeton University Press: Princeton, New Jersey, USA, pp. 140-156.

Pan D., Domon G., de Blois S. and Bouchard A. 1999. Temporal (1958-1993) and spatial patterns of land use changes in Haut-Saint-Laurent (Quebec, Canada) and their relation to landscape physical attributes. Landscape Ecology 14: 35–52.

Papanastasis V.P. and Kazaklis A. 1998. Land use changes and conflicts in the Mediterranean-type ecosystems of western Crete.. In: Rundel P.W., Montenegro G. and Jaksic F. (eds), Landscape degradation and biodiversity in Mediterranean-Type Ecosystems. Ecological Studies 136. Springer-Verlag, Berlin, Germany, pp. 141-154.

Passera C.B. 1999. Propuestas Metodológicas para la gestión de ambientes forrajeros naturales de zonas áridas y semiáridas. Ph.D. Universidad de Granada. Granada. Spain. 243 pp.

Pearson S.M., Turner M.G., Gardner R.H. and O'Neill R.V. 1996. An organism-based perspective of habitat fragmentation.. In: Szaro R.C. and Johnston D.W. (eds), Biodiversity in managed landscapes: theory and practice, Oxford University Press: Oxford, UK, pp. 77-95.

Peinado M. Alcaraz F. and Martínez J.M. 1992. Vegetation of Southeastern Spain. Flora and Vegetation. Flora et Vegetatio Mundi. Vol. X. J. Cramer. Berlin, Germany.

Peng C.K., Buldyrev S.V., Goldberger A.L., Havlin S., Sciortino F., Simons M. and Stanley H.E., 1992. Long-range correlations in nucleotide sequences. Nature: 356: 168–170.

Peng C.K., Buldyrev S.V., Havlin S., Simons M., Stanley H.E. and Goldberger A.L. 1994. Mosaic organization of DNA nucleotides. Physical Review 49: 1685–1689.

Peterson G.D. 2002. Contagious disturbance, ecological memory, and the emergence of landscape pattern. Ecosystems 5: 329–338.

Pickett S.T.A., Collins S. L. and Armesto J.J. 1987. A hierarchical consideration of causes and mechanisms of succession. Vegetatio 69: 109–114.

PORN 1996. Plan de ordenación de los recursos naturals y plan rector de uso y gestión del Parque Natural de Cabo de Gata-Níjar. Junta de Andalucía, Consejería de Medio Ambiente: Sevilla, Spain.

Rietkerk M., Boerlijst M.C., Langevelde F., HilleRisLambers R., Van de Koppel J., Kumar L., Prins H.H.T. and Van de Ross A.M. 2002. Self-organization of vegetation in arid ecosystems. American Naturalist 160: 524–530.

Robles A.B., Boza J. and Gonzalez Rebollar J.L. 1997. Análisis de la recuperación de la cubierta vegetal en un agroecosistema pastoral del sueste semiárido Ibérico. Actas Etnobotánica 92: 333–339.

Rodríguez J.E. and Sánchez A. 2000. Despoblamiento y desertificación.. In: Grupo Ecologista Mediterrá neo (eds), Desertificación de Almería, Artes Gráficas: Almería, Spain, pp. 35-39.

Saunders D.A., Hobbs R.J. and Margules C.R. 1991. Biological consequences of ecosystem fragmentation: A review. Conservation Biology 5: 18–32.

Schumaker N.H. 1996. Using landscape indices to predict habitat connectivity. Ecology, 77: 1210–1225.

Sole Benet A., Calvo A., Cerdá A., Lazaro R., Pini R., Barbero J. 1997. Influences of micro-relief patterns and plant cover on runoff related processes in badlands from Tabernas (SE Spain). Catena 31(1-2): 23–38.

Soulé M. 1986. Conservation biology. Sinauer Associates: Sunderland, Massachusetts, USA.

Stevenson A. and Harrison R. 1992. Ancient forests in Spain: a model for land-use and dry forest management in south-east Spain from 4000 BC to 1900 AD. Procceding of Prehistoric Society 58: 227–247.

Swanson F.J., Kratz T.K., Caine N. and Woodmansee R.G. 1988. Landform effects on ecosystem patterns and processes. Bioscience 38: 92–98.

Swenson J.J. and Franklin J. 2000. The effects of future urban development on habitat fragmentation in the Santa Monica Mountains. Landscape Ecology 15: 713–730.

Taqqu M.S., Teverovsky V. and Willinger W. 1995. Estimators for long-range dependence: an empirical study. Fractals 3: 785–798.

Taylor P.D., Fahrig L., Henein K. and Merrian G. 1993. Connectivity is a vital element of landscape structure. Oikos 68: 571–573.

Tilman D., Wedin D. and Knops J. 1996. Productivity and sustainability influenced by biodiversity in grassland ecosystems. Nature 379: 718–720.

Tilman D., Reich P.B., Knops J., Wedin D., Mielke T. and Lehmen C. 2001. Diversity and productivity in a long-term grassland experiment. Science 294: 843–845.

Tomaselli R. 1981. Main physiognomic types and geographic distribution of shrub systems related to Mediterranean climates.. In: Di Castri G., Goodall D.W. and Specht R. (eds), Mediterranean Types shrublands. Ecosystems of the World 11, Elsevier: The Netherlands, pp 95-105.

Turner B.L. II, Clark W.C., Kates R.W., Richards J.F., Mattews J.T. and Meyer W.B. 1990. The earth as transformed by human action: global and regional changes in the biosphere over the past 300 years. Cambridge University Press: New York, New York, USA.

Turner M.G., Romme W.H., Gardner R.H., O'Neill R.V. and Kratz T.K. 1993. A revised concept of landscape equilibrium: disturbance and stability on scaled landscapes. Landscape Ecology 8: 213–227.

Turner M.G., Wear D.N. and Flamm R.O. 1996. Land ownership and land-cover change in the Southern Appalachian Highlands and the Olympic Peninsula. Ecological Applications 6: 1150–1172.

Turner M.G., Gardner R.H. and O'Neill R.V. 2001. Landscape ecology in theory and practice. Springer-Verlag: New York, New York, USA.

Valero-Garcés B., Navas A., Machin J., Stevenson T. and Davis B. 2000. Responses of a saline lake ecosystem in a semiarid region to irrigation and climate variability. The history of Salada Chiprana, Central Ebro Basin, Spain. Ambio 29: 344–350.

Wickham J.D., O'Neill R.V. and Jones K.B. 2000. Forest fragmentation as an economic indicator. Landscape Ecology 15: 171–179.

With K.A., Gardner R.H. and Turner M.G. 1997. Landscape connectivity and population distribution in heterogeneous environments. Oikos 78: 151–169.

With K.A. and King A.W. 1999a. Extinction thresholds for species in fractal landscapes. Conservation Biology 13: 314–326.

With K.A. and King A.W. 1999b. Dispersal success on fractal landscapes: a consequence of lacunarity threshold. Landscape Ecology 14: 73–82.