

1 **Ecological and socio-economical thresholds of land and plant-community degradation in semi-**
2 **arid Mediterranean areas of southeastern Spain**

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10
11 **Abstract**

12 The climate of semi-arid Mediterranean areas is characterized by highly variable rainfall and
13 seasonal droughts, which favor highly specialized flora, high biodiversity, and the emergence of a
14 large number of endemic species. Such highly specialized complex ecosystems are expected to be
15 more difficult to return to a previous state following perturbation than are more mild ecosystems and
16 lead to dramatic transitions in vegetation. An understanding of the circumstances under which
17 irreversible transition shifts occur is vital in an age of climate change and socio-economic
18 globalization, which will have significant consequences for those highly specialized areas unless
19 management practices, conservation, and preventive measures are increased. Within ecosystems, the
20 interactions between multiple factors acting at multiple temporal and spatial scales, and the inertia of
21 ecological processes, make the consequences of current management practices persist for a long
22 time. With the task of understanding the relative importance of the underlying mechanisms that
23 influence the resilience of drylands in southeastern Spain, we reviewed what is known about plant-
24 soil interactions and the role of the self-organizing capacity of vegetation in the formation of
25 vegetation spatial patterns at multiple spatial scales. In addition, we evaluated the importance of
26 threshold indicators as early detectors of transition shifts in dryland ecosystems.

27
28 **Biophysical and socio-economical factors that influence landscape configurations**

29 Landscape configurations result from processes that act at multiple spatial and temporal scales.
30 Abiotic factors have a significant influence on the distribution of plant communities within a
31 landscape and act as the first species filter. Long-standing human management of the land,
32 particularly livestock grazing, acts as a secondary successional species filter (Swanson et al., 1988;
33 Cingolani et al., 2003; Sebastià et al., 2008). Humans have been modifying the landscape in
34 southeastern Spain since they began occupying it in remote period, constrained only by the physical

1 limitations imposed by the landscape. There are, however, uncertainties about the relative importance
2 of biophysical and socioeconomic factors and the dynamics of desertified and desertification-prone
3 areas (Herrmann and Hutchinson, 2005; Verón et al., 2006; Reynolds et al., 2007).

4 5 *Factors in the development of the Mediterranean landscape*

6 Like other Mediterranean coastal areas, the landscape of the southeastern region of the Iberian
7 Peninsula was affected by the widespread climate change that occurred at the end of the Tertiary
8 Period, which led to the region's characteristic summer drought, and tectonic activity of its
9 orographic systems, which increased erosion on hillslopes and sedimentation in drainage networks.
10 During the Holocene Epoch, arid conditions established in the basin; the area was particularly dry 6
11 ky and 4 ky BP, and humid periods occurred 3 ky and 250 y BP (Rognon, 1987). Carbon isotopes in
12 grain cereals from throughout the Iberian Peninsula indicated that the N-S aridity gradient tended to
13 become pronounced during the last 5 ky (Araus et al., 1997). A growing body of paleobotanical
14 evidence indicates that xeric plant species have been displacing mesic woodlands northward to
15 higher elevations (Parra, 1993). The activities of humans have influenced the current composition
16 and distribution of the vegetation in the Mediterranean region (Pons, 1981; Pons and Reille, 1988).
17 Humans have aided the expansion of sclerophyllous vegetation, e.g., evergreen oaks and pines in dry
18 soils at the expense of deciduous and semi-deciduous oaks. In that way, humans have created a
19 mosaic-like landscape that has a very complex spatial structure and high biodiversity (Naveh, 1991).
20 The changes that occurred over the long-term were at rates that were several orders of magnitude
21 lower than those that are occurring today. In the distant past, the influence of humans was absent or
22 very limited. In the last few hundred years, however, in southeastern Spain, the amount of land
23 degradation influenced by climatic and sociological factors acting synergistically has been significant
24 (Puigdefabregas and Mendizabal, 1998).

25 During the Little Ice Age (LIA) the Iberian Peninsula had a climate that was cooler and more humid
26 that it does today, but it was a period of high variability. The lowest maximum annual temperatures
27 occurred in the second half of the 17th C., which were about -0.5 °C below the average for the
28 Medieval Period (Creus, 1983). Since that time, temperatures have been slowly increasing as the
29 amount of precipitation has been decreasing (with the exception of a small increase at the beginning
30 of the 21st C. Since 1900 the rate of the decrease in annual rainfall has been about 3 mm/yr (Brandt
31 et al., 1991; Oñate Rubalcaba, 1993).

32 In Spain, socio-economic systems underwent three significant events, which led to changes in land
33 use over the last 500 yr. The first occurred in the 16th and 17th C. because of the establishment of
34 complete Christian rule throughout the Iberian Peninsula, which favored the southward expansion of

1 the dryland agriculture, and the colonization in America, which increased the demand for wool and
2 wood products to meet the needs of the settlements in America (Elliot, 1965) and the growing ship-
3 building industry (Dupre Ollivier, 1990). As a result, the area dedicated to grain crops and large scale
4 sheep industries expanded and the amount of forest was reduced. The population increase that began
5 in the 18th C. brought about the second phase (Dupre Ollivier 1990), which overcrowded the rural
6 area during the first half of the 20th C. The major technological, social, cultural, and economic
7 changes that occurred in rural areas in the 1960s led to the third significant period.
8 In the period encompassed by the 16th and 18th C., the cold and humid climate (at the peak of the
9 LIA) and extensive changes in land use that left large areas exposed to erosion had a significant
10 impact on the landscape (Hoffmann, 1988).

11

12 *Patterns of land degradation*

13 In the 19th and early 20th C., the growth of human populations in rural areas, which caused a need
14 for the expansion of agriculture, led to the degradation of the land. In the second half of the 20th C.,
15 the expansion of industry in urban areas reduced the rural population, and agriculture was
16 concentrated in the most productive areas; namely, within irrigation developments. Since the mid-
17 20th C., inland populations have decreased as populations increase in coastal areas. The
18 ‘litoralization’ of the economy is particularly acute in Spain, where the population density in the
19 coastal Mediterranean region is twice the national average (Grenon and Batisse, 1988). Marginal
20 and irrigated lands are the current hot spots of land degradation. In the semi-arid regions of
21 southeastern Spain, mining, extensive agriculture, and livestock production date to the Bronze Age
22 (Chapman, 1978). Those practices and the scarcity and irregularity of rainfall have created the
23 current landscape, which is dominated by xeric steppes on hillslopes with a few areas in the wetter
24 sites that have open shrublands and some pinewood forests at the higher altitudes. Those areas had
25 low productivity and, in the second half of the 20th C., anthropomorphic forces were exerted through
26 the use of hillslopes for extensive livestock grazing and the use of valley basins for rain-fed
27 agriculture, mainly cereal crops. Although humans have existed in the area since the Neolithic
28 Period, the low productivity has favored a reduced population density. With the aid of agricultural
29 technologies that were developed in the last 20 yr, the use of ground-water from aquifers and
30 desalination plants has enabled the rapid development of greenhouses and tourism in the coastal
31 plains of Almeria (Tout, 1990). The expansion of irrigation projects in the area relies mainly on
32 ground-water sources. The risks include overdrafting, degradation of aquifers by marine intrusion,
33 land degradation caused by salinization, topographic reshaping, and surface-mining of sand to meet
34 the demand from greenhouses (Puigdefabregas and Mendizabal, 2004). With the exception of the

1 latter human impact, the scarcity of water in the area has discouraged exploitation by humans, which
2 has led to a well preserved and well adapted xeric vegetation that has remained in a state similar to
3 that described in the scientific botanical explorations of Simón de Rojas 200 yr ago (Gil Albarracín,
4 2002). In some of the badlands in southern Spain, archaeological evidence indicates that conditions
5 existed 5 ky ago (Wise et al., 1982).

6 The abandonment of agricultural and mining activities in the 1960s did not lead to the recovery of
7 the shrubland vegetation. Resilience (the ability to return to a previous state after perturbation)
8 (Holling, 1973) in those semi-arid ecosystems is low because of the scarcity of resources and the
9 pronounced fluctuations in precipitation (Noy-Meir, 1990; Milton and Hoffman, 1994). Thus, in
10 Cabo de Gata-Níjar N.P., between 1957 and 1994, 45% of croplands in the area were abandoned and
11 was not transformed into shrubland (Alados et al., 2004b); only the amount of tallgrass steppe of
12 *Stipa tenacissima* (19%) and arid garrigues (24%) increased. The slow recovery of the vegetation
13 that is associated with xeric slopes was also reported by Cerdá (1998) in southeastern Spain. More
14 recently, an increase in the demand for biomass for biofuels has caused some reversal of this trend
15 and many set-a-side areas have become cultivated again. A large-scale survey of the state of the land
16 on the Iberian Peninsula (Barrio del et al., 2010) indicated an increase in land degradation in the
17 southeast, but it was associated with the decrease in annual rainfall, rather than to an intrinsic
18 worsening of the condition of the land, which is much less widespread. Mitigation programs have to
19 distinguish between active desertification and relict desertification. In the former, stressors must be
20 relieved, whereas in the latter, the rehabilitation or restoration of threatened areas, or even doing
21 nothing is sufficient.

22

23 **Vegetation spatial patterns at multiple scales**

24 A characteristic of complex biological systems is that they are governed by processes that act at
25 multiple spatial and temporal scales, and it is in the cross-scale interactions where irreversible
26 positive feedback processes can be triggered (Peters et al., 2004) and where higher levels constrain
27 the structure and function of lower levels (Brown and Allen, 1989; Holling, 1992). For example,
28 when the distance between ecosystem fragments exceeds a dispersal threshold, the disruption in
29 connectivity can lead to metapopulation extinction (Alados et al., 2009b). Species richness and
30 spatial patterns can depend on forces that operate across a wide range of temporal and spatial scales
31 (Brown and Lomolino, 1998). Grazing can enhance plant diversity at fine scales by reducing plant
32 competition, but it can inhibit diversity at coarse scales depending on the supply of propagules from
33 neighbouring areas Olff and Ritchie (1998). This is especially relevant in arid areas where species
34 that have a restricted spatial dispersal are abundant because the space suitable for germination is

1 scarce and it is advantageous to germinate close to the mother plant (Van Rheede van Outshoorn and
2 Van Rooyen, 1999). Consequently, species that have restricted spatial dispersal (reduction of
3 dispersal mechanisms or have anchorage mechanisms), which are predominant in arid regions (Ellner
4 and Shmida, 1981; Gutterman, 1993; Navarro et al., 2009), are very sensitive to habitat
5 fragmentation (Alados et al., 2010). In semi-arid areas in southeastern Spain, the isolation of remnant
6 fragments had a negative effect on plant species richness (Alados et al., 2009a) and led to an increase
7 in the differences in species composition among fragments (spatial variability in species distributions
8 was greater than that expected from a random spatial distribution). Indeed, species distributions
9 became more homogenized among large fragments because of diffusion processes.

10 The negative effects of habitat fragmentation on plant dispersal can be ameliorated by traditional
11 grazing practices because, in many areas, livestock are essential for the dispersal of many plant
12 species (Fisher et al., 1996). In dry environments, livestock are the primary dispersal agents of plants
13 and can increase seedling success through consecutive stages of establishment throughout seed
14 ingestion and deposition (Manzano et al., 2005; Robles et al., 2005; Ramos et al., 2006). In Spain,
15 free-ranging sheep traveled > 25 km per day during transhumance (Klein, 1981), thereby spreading
16 seeds over a large area. Although transhumance has almost disappeared in Spain, the daily
17 movements of free-ranging sheep are still important because they travel about 6 km per day.
18 Comparisons between fragmented and non-fragmented landscapes in the semi-arid regions of the
19 Middle Ebro Valley, where gypsophilic species exhibited dispersal patterns that are associated with
20 short dispersal distances (i.e., myxospermy (Escudero et al., 1999)), showed that grazing ameliorated
21 the detrimental effects of habitat fragmentation on species diversity (Pueyo et al., 2008b).

22 While landscape spatial patterns influence the spatial distribution of species, organisms can shape
23 their ecosystem by acting at the local scale, and influencing large-scale contagious processes such as
24 insect outbreaks and spreading fire (Holling, 1992). In semi-arid ecosystems, local interactions
25 between organisms and their environment, such as facilitation mechanisms (Bertness and Callaway,
26 1994; Pugnaire et al., 1996; HilleRisLambers et al., 2001), and limited dispersal ability (Miller et al.
27 2002), promotes the development of self-organized spatial patterns (Aarssen and Turkington, 1985;
28 Soro et al., 1999; Solé and Bascompte, 2006) that emerge at larger scales (Rietkerk (Rietkerk and
29 van de Koppel, 2008; van de Koppel et al., 2008). For example, woody species act as landscape
30 modulators by altering the structure of the soil and the redistribution of nutrients (Rietkerk et al.,
31 2002; Gilad et al., 2004; Meron et al., 2004), and provide refuge for grazing species (Rebollo et al.,
32 2002). Those species are important landscape modifiers that can generate a two-phase mosaic of
33 patches of woody phytomass and open space. In addition, the combined effects of spatial self-
34 structuring and epigenetic selection can result in niche complementarity. Communities assembled by

1 self-organizing processes such as niche complementarity, which includes niche differentiation and
2 facilitation interactions, develop an increase in the efficient use of resources (Venail et al., 2008) and
3 plant diversity has a positive effect on primary production (Loreau and Hector, 2001). In addition,
4 the greater likelihood of relatedness among neighbors favors community-level selection (Johnson
5 and Boerlijst, 2002) The destruction of the spatial structure of the community disrupts the self-
6 organized gene-complex of long-standing stable communities. Microcosm experiments demonstrated
7 that uniform populations rapidly diversified when distributed within spatially structured microcosms.
8 Diversity was lost when the spatial structure was broken by shaking the microcosms (test tubes)
9 (Rainey and Travisano, 1998).

11 **Detecting vegetation transitions from spatial patterns**

12 Most indicators of ecosystem-level stress are merely descriptors of past transitions between
13 ecosystem stages (Westoby et al., 1989); however, changes in ecosystem attributes should be
14 detected when management actions are cost-effective. In addition, in semi-arid regions, the transition
15 from vegetated to desert zones is not always gradual; rather, they can occur suddenly and irreversibly
16 (Scheffer et al., 2001; Rietkerk et al., 2004; Kéfi et al., 2007). Although, the random inter-annual
17 variations in precipitation that are typical of Mediterranean semi-arid environments (Noy-Meir,
18 1973) might result in an intermediate, statistically stable configuration between the two states
19 (D'Odorico et al., 2005). The ability to forecast system dynamics is limited; at least the emergent
20 patterns that reflect gradual or imminent transition shifts can be detected. The objective is not to
21 identify when the transition occurred, but to confirm whether the system is being pushed towards a
22 new trajectory, and to detect any threats to the integrity of the ecosystem. Models of complex
23 systems cannot predict where critical transitions will occur, but the vulnerabilities of the ecosystem
24 to critical thresholds can be identified (Guttal and Jayaprakash, 2009; Scheffer et al., 2009). Indeed,
25 complex systems need considerable time to manifest patterns in oscillations and, at the early stages
26 of colonization, species are randomly distributed (Solé and Bascompte, 2006; Cutler et al., 2008). As
27 the ecosystem develops, positive interactions between neighbours and population growth cause patch
28 nucleation and coalescence (Yarranton and Morrison, 1974; Cutler et al., 2008). On the other hand,
29 an increase in disturbance often leads to unpredictability (randomness) in species distributions
30 (Alados et al., 2003; Alados et al., 2004a). Spatial interactions in natural plant communities can
31 produce emergent patterns at higher spatial scales (Martens et al., 2000; Peterson, 2000; Rietkerk and
32 van de Koppel, 2008). Different kinds of models resulted in plant spatial pattern formation
33 (HilleRisLambers et al., 2001; Lejeune et al., 2002; Meron et al., 2004; Barbier et al., 2006). All of
34 those models are limited because they cannot incorporate all of the sources of the variability of the

1 system. In statistical models, however, signs that occur close to transition shifts can be used; e.g., a
2 model of changes in desert vegetation indicated that regular patterns change in a predictable way as
3 the critical transition threshold to bare soil is approached (Rietkerk *et al.*, 2004), which has increased
4 the use of the spatial distribution of vegetation to detect early warning signs of desertification (Kéfi
5 *et al.*, 2007; Scanlon *et al.*, 2007; Maestre and Escudero, 2009).

6 The ways to identify the early-warning signs of critical transitions range from analyses of the
7 autocorrelation and variance of the fluctuations (Scheffer *et al.*, 2009) to the long-term
8 autocorrelation that measures the memory of the process as one approaches to vegetation transitions
9 (Alados *et al.*, 1996; Alados *et al.*, 2003). To detect transition states in semi-arid ecosystems, recent
10 studies have examined the spatial variance or spatial skewness of the vegetation (Guttal and
11 Jayaprakash, 2009), or the change in the distribution of the sizes of patches of vegetation (Kéfi *et al.*,
12 2007; Scanlon *et al.*, 2007). Indeed, scaling relationships and fractals provide a powerful analytical
13 framework that incorporates the structural complexity of plant communities and can be used to
14 analyze “emergent patterns” of the ecosystem across a range of scales to predict catastrophic shifts.
15 The change in the fractal dimension of the spatial pattern can indicate a significant change in the
16 processes that generate plant spatial patterns (Sugihara and May, 1990; Li, 2000). For example, the
17 Information Fractal Dimension of a species distribution quantifies the relationship between species
18 diversity and area, and is equivalent to the exponent of the species-area relationship (Borda-de-Agua
19 *et al.*, 2002). The proportion of all possible species that are in a given area depends on the size of the
20 area sampled. Thus, the Shannon Diversity Index (H') (Shannon, 1948) changes with the size of the
21 area sampled, and the rate of change at several scales measures the Information Fractal Dimension
22 (IFD) (Farmer *et al.*, 1983). If the distribution of species is homogenous, the IFD is high, and
23 decreased as patchiness increased in semi-arid shrubland ecosystems (Alados *et al.*, 2003; Alados *et al.*,
24 2006). IFD and species diversity are inversely correlated (Alados *et al.*, 2006), which is
25 consistent with the general phenomenon that spatial diversity is greater in heterogeneous habitats
26 than it is in analogous homogeneous habitats, and with the community assembled by self-organizing
27 processes, which results in an increase in the efficient use of resources (Venail *et al.*, 2008). IFD
28 increases when the community assemblage changes to a more degraded state; e.g., when the dwarf
29 matorral of Cabo de Gata N.P. degraded to alpha steppe dominated by *Stipa tenacissima* (Alados *et al.*,
30 2003). At sites of natural vegetation in Cabo de Gata-Nijar National Park, Spain, the vegetation
31 was sampled along 69 500-m-long linear transects (Fig. 1). At those sites, the IFD of the dwarf
32 matorral ($x \pm se = 0.629 \pm 0.002$, $n = 37$) and the alpha steppe ($x \pm se = 0.725 \pm 0.007$, $n = 32$) differed
33 significantly ($F_{1,58} = 237.13$, $p < 0.001$) (Fig. 2).

1 When examining plant spatial distributions, it is important to distinguish between those that result
2 from external perturbations and those that are caused by dynamics intrinsic to the system (Cutler et
3 al., 2008); e.g., distinguishing between those patterns influenced by the topography of the terrain and
4 those patterns that originate at the level of the individual, but become apparent at larger scales, but at
5 scales that are lower than the environmental heterogeneity of the landscape. Those conditions pose a
6 problem in power spectrum calculations because the latter assume stationarity and cannot distinguish
7 between stationary and non-stationary signals (Ivanov et al., 2002). Separating spatial patterns
8 caused by pre-existing environmental heterogeneity from self-organizing processes allow us to detect
9 intrinsic changes in the control mechanisms. Detrended Fluctuation Analysis (DFA) removes the
10 trends (Peng et al., 1992) and avoids any spurious correlations caused by non-stationarities (Bunde et
11 al., 2002). DFA, which describes spatial autocorrelation and patchiness, can detect long-range spatial
12 autocorrelations in plant species' distributions (Alados *et al.*, 2003), which influence the degree of
13 spatial clustering of the vegetation cover, independent of the scale of measurement. Long-range
14 autocorrelations imply that positive feedback mechanisms dominate the system and provide valuable
15 information about the inherent memory of the process under examination.

16 In several Mediterranean rangeland ecosystems, evaluations of degradation gradients caused by
17 livestock grazing and associated increases in bare soil indicated that the long-range spatial
18 autocorrelations of the characteristic species of each community were highest in the most preserved
19 areas. As grazing pressure increased, most of plant species became more randomly distributed;
20 species adapted to perturbed ecosystems experienced an increase in spatial organization (Alados *et*
21 *al.*, 2004a). In addition, the slope of the negative correlation between grazing disturbance and long-
22 range spatial autocorrelation (high α values) appeared to be most pronounced among the most
23 perturbation-sensitive species; e.g., *Quercus coccifera*, *Phlomis purpurea*, *Cistus albidus*, and even
24 the grazing-tolerant *Periploca laevigata* exhibited a drastic reduction in spatial organization (Alados
25 *et al.*, 2003). Grazing-resistant species such as *S. tenacissima*, which have buried renewal buds that
26 livestock cannot reach, and species that have a capacity for persistence and colonizing unoccupied
27 microhabitats (e.g., annuals or pioneer plants) exhibited little change in spatial distribution along a
28 grazing pressure gradient (Alados *et al.*, 2003).

29 A reduction in the number and size of vegetation fragments and the degree of isolation had a
30 negative influence on the long-range spatial distributions of the characteristic plant species in Cabo
31 de Gata-Nijar N.P., Spain (Alados *et al.*, 2006). In particular, forest-gap species that have restricted
32 dispersal (*Phlomis purpurea*, *Cistus albidus*, *Teucrium pseudo-chamaepestis*, *Genista spartioides*,
33 *Brachypodium retusum*) and species that actively restrict or hamper dispersal (antitelechory) by
34 myxospermy (*Helianthemum almeriense* and *Linum strictum*) were very sensitive to the effects of

1 habitat fragmentation and exhibited a significant positive correlation between patch size and the
2 long-range spatial autocorrelations. In general, changes in the long-range spatial autocorrelations of
3 plant species in response to different degrees of land degradation indicated that spatial self-
4 organizing processes are important in the conservation of the ecosystem, and the most effective way
5 to protect against land degradation in semi-arid ecosystems is to reinforce self-organizing processes
6 as positive feedback interactions, which will allow self-structuring mechanisms to operate.

8 **Monitoring vegetation transitions using patterns in NDVI**

9 The use of remote sensing to monitor desertification processes has become common (Prince et al.,
10 1998; Asner and Heidebrecht, 2005; García et al., 2008; Weber et al., 2009; Hall et al., 2010) and the
11 availability of high-resolution satellite images permits an estimation of the relationship between
12 vegetation spatial patterns and the effects of environment. The satellite-derived vegetation index the
13 Normalized Difference Vegetation Index (NDVI) provides estimates of vegetation productivity
14 (Pettorelli et al., 2005) and can be correlated with the amount of green vegetation cover, vegetation
15 abundance, chlorophyll content, and leaf area index (Harrison et al., 2006). The NDVI, an expression
16 of the contrasting reflectance between the red and near-infrared wavelengths of a surface spectrum
17 (Rouse et al., 1974; Tucker, 1979), is given by the following formula: $NDVI = (RNIR - RRED) / (RNIR + RRED)$, where RNIR is near infrared reflectance (NIR) and RRED is red
18 reflectance. Values range between -1 and 1, with 0 representing the approximate value of no
19 vegetation. Thus, negative values usually indicate non-vegetated surfaces. A high NDVI indicates
20 high vegetation activity because of the strong relationship between the NDVI and the amount of
21 radiation absorbed in photosynthetic processes (Gallo et al., 1985). The NDVI can be used to
22 estimate species richness and vegetation spatial heterogeneity (Gould, 2000; Fairbanks and
23 McGwire, 2004; Gillespie, 2005; Levin et al., 2007); however, the ecological significance of
24 vegetation spatial patterns depends on the scale of the analyses, which can produce apparently
25 contradictory results. While some studies found a positive correlation between landscape
26 heterogeneity and species richness (Fairbanks and McGwire, 2004; Gillespie, 2005), others did not
27 find a clear relationship (Levin *et al.*, 2007). In general, studies conducted at higher scales observed a
28 positive correlation between species diversity and landscape heterogeneity because, accordingly to
29 habitat heterogeneity hypothesis (Simpson, 1949; MacArthur and Wilson, 1967), structurally
30 complex habitats might provide more niches and diverse ways of exploiting the environmental
31 resources and thus increase species diversity (Fahrig, 2003; Hobbs and Yates, 2003; Harper et al.,
32 2005). Depending on the spatial scale of the analysis, and the way organisms perceive the habitat,
33 species diversity can appear to decrease as habitat heterogeneity increases (Tews et al., 2004)

1 (ecosystem fragmentation). Thus, analyses at multiple scales are needed before spatial heterogeneity
2 can be advocated as a reliable means of monitoring ecosystem degradation.

4 **A case study of NDVI variation across a degradation gradient**

5 Coordination between ground- and remote sensing-based research and the implementation of sound,
6 cost-effective methods have promise as tools for monitoring desertification. To use the spatial
7 patterns in the NDVI index to monitor desertification, we need to know whether the average values
8 and spatial variability of NDVI differ depending on the ecosystem features. We compared the spatial
9 distribution of vegetation derived from ground transects and the NDVI values obtained from satellite
10 images of Cabo de Gata-Níjar Natural Park, in southeastern Spain, which was established in 1987.
11 Much (54%) of the 376-km² park is natural semi-arid vegetation on hillslopes within a matrix of
12 abandoned crop fields (Alados *et al.*, 2004b). The habitat in the northern part of the national park is
13 alpha steppe, which is dominated by *Stipa tenacissima* and, in the southern part, there is a dwarf
14 matorral of *Ulex baeticus*, *Phlomis* spp. and *Chamaerops humilis* on volcanic soils.

15 Values of the Normalized Difference Vegetation Index (NDVI) were derived from an image captured
16 on 2005-04-11 (11:10 GMT) by the Ikonos-2, a satellite-borne sensor (launched in 1999) that detects
17 four multi-spectral bands (blue, green, red, near infrared), simultaneously, at a spatial resolution of 4
18 m, and a panchromatic band (the entire range covered by the four multi-spectralbands) at a 1-m
19 resolution (Dial *et al.*, 2003). The analysis was based on an image captured in April (see Fig. 1)
20 because that is the month in which the vegetation exhibits its greatest phenological development at
21 the study area in Almeria, Spain. The image of the 74-km² area was orthorectified into a UTM
22 projection with a high precision (RMS <1 pixel). For the geometrical corrections, we used a Digital
23 Elevation Model (DEM), georeferenced aerial orthophotos, and Ground Control Points (GCPs). To
24 preserve the original image values, we used the nearest-neighbor re-sampling procedure.

25 The ground reflectance data retrieved from satellite images can be compared to ground-based
26 measurements, although haze removal and atmospheric-topographic corrections were required,
27 which reduce atmospheric effects and variation in solar illumination (sunny and shady hills). The
28 corrections were applied using algorithms implemented in the ATCOR software (Geosystem, 2008),
29 which was available as an extension to the remote sensing software ERDAS IMAGINE 9.2. To
30 remove any bias caused by the sensor's parameters (calibration files), the digital numeric values (raw
31 data) were converted into estimates of the radiance reaching the sensor. The radiance values were
32 converted into real surface reflectance values using standard estimates of atmospheric conditions
33 (e.g., visibility, aerosol type, water vapor content) in the study area. ATCOR consists of ATCOR2
34 and ATCOR3, and is based on MODTRAN-4 code. Typically, ATCOR-2 is used to correct for

1 atmospheric effects in the data from spaceborne optical sensors and assumes flat terrain and
 2 Lambertian reflectance. The algorithm works with a database of atmospheric correction functions
 3 that are stored in look-up tables (Richter, 2000b; a). ATCOR-3 accounts for rugged terrain effects by
 4 incorporating DEM data and their derivatives, such as slope and aspect, sky view factor, and shadow
 5 cast. The algorithm accounts a compiled database that contains the results of the radiative transfer
 6 calculations for a wide range of weather conditions and sun angles (Richter, 1997; Richter, 1998;
 7 2000a).

8 After the geometric and radiometric corrections, the NDVI was calculated at 4-m intervals (Fig 1d)
 9 along the 500-m-long transects that were extracted from the satellite image. The results were laid
 10 over the data collected from the ground-based transects in Cabo de Gata-Nijar N.P. For each transect
 11 on the ground, six random transects were taken from the satellite-based image (Fig 1c), which
 12 provided information from 36 and 48 transects in dwarf matorral and alpha steppe, respectively (Fig
 13 1b). At the alpha steppe site, transects were placed on either shallow (<10°) or steep (20°-30° slopes).
 14 The critical slope for incipient rill formation is 9 ° (Hudson, 1975; Berael et al., 1995). On the
 15 transects (n = 84), the average NDVI and the bare-soil ratio were significantly negatively correlated
 16 ($R^2 = 0.58$, slope = -0.62, $P < 0.001$). As in studies elsewhere (Fairbanks and McGwire, 2004;
 17 Harrison et al., 2006; Levin et al., 2007), in southeastern Spain, average NDVI and species richness
 18 were significantly positively correlated ($R^2 = 0.28$, slope = 0.002, $P < 0.001$).

19 The variation in NDVI within transects, as estimated by the coefficient of variation, increased
 20 significantly with the amount of bare soil ($R^2 = 0.42$, slope = 0.72, $P < 0.001$); i.e., the heterogeneity
 21 of the spatial distribution of NDVI increased with an increase in the bare-soil ratio. However, the
 22 spatial pattern of the variation differed among plant communities. To calculate the spatial pattern of
 23 the variation in the long-range spatial autocorrelation of NDVI values of each transect, we used
 24 Detrended Fluctuation Analyses (DFA). The NDVI data sequence $y(s) = \sum_i^s x(i)$ is integrated directly

25 from the NDVI values. The sequence was subdivided into non-overlapping sequential sets or 'boxes'
 26 of size b. A regression line was fit to each box of size b, ($y_b(s)$ was regressed on s) and the residual

27 variance per each box size was calculated from $F^2(b) = \frac{\sum (y_b(s) - \hat{y}_b(s))^2}{N}$; to provide the scale of the

28 relationship $F(b) \propto b^\alpha$. The α parameter reflects the long-range autocorrelations of the NDVI
 29 distribution sequences. Theoretically, α is unaffected by the magnitude of the fluctuations in the
 30 sequences, but is affected by the sequential ordering of the fluctuations. If $\alpha > 0.5$, the sequence
 31 depends on the history of the distribution and, when $\alpha = 0.5$, the sequence is randomly distributed.

1 As the amount of bare soil increased α -DFA of NDVI increases ($R^2 = 0.39$, slope = 1.04, $P < 0.001$)
2 in the dwarf matorral, but not in alpha steppe ($R^2 = 0.03$, slope = 1.04, $P > 0.05$) (Fig. 3). In dwarf
3 matorral, although a patchy vegetation distribution maximizes vegetation growth because vegetation
4 patches capture rainfall and nutrients in the patches and reduce inter-patch run-off (Ludwig et al.,
5 1999; Pueyo et al., 2008a), the contagious spread of erosion processes can dominate vegetation
6 dynamics when the system becomes over-connected and rapid change is triggered. Thus, the positive
7 feedback triggered by run-off and erosion tend to destabilize the system when the patchy distribution
8 of vegetation reaches a certain threshold beyond which the processes become irreversible (Rietkerk
9 et al., 2004). In the alpha steppe, the lateral root expansion of *S. tenacissima* favor to a stronger
10 spatial autocorrelation on bare soil, coincident with the spatial distribution of *S. tenacissima* ($R^2 =$
11 0.84, slope = 1.02, $P < 0.001$, for the regression between α -DFA of *S. tenacissima* vs. that of bare
12 soil from the ground-based transects). The effect was not as evident in the dominant species of the
13 dwarf matorral, e.g., *Ch. humilis* ($R^2 = 0.27$, slope = 0.57, $P < 0.001$) and *Ph. purpurea* ($R^2 = 0.21$,
14 slope = 0.39, $P < 0.001$). In the alpha steppe, effects of two opposing trends cancel out each other: (i)
15 the self-organization of bare soil because of the self-organization of the dominant species *S.*
16 *tenacissima* and (ii) the run-off process that increases with the bare-soil ratio.

17 To distinguish between those effects, we compared shallow and steep slopes in alpha steppe habitats,
18 where the effect of *S. tenacissima* on pattern formation is greater because it occupied 48% of the
19 area, whereas it occupied 24% on the dwarf matorral. Shallow slopes are very favorable to perennial
20 bunch grasses such as *S. tenacissima* because source and sink patterns of water and sediments
21 develop from bare ground areas into tussock (Sánchez and Puigdefábregas, 1994; Cerdá, 1997). The
22 effect is reduced on steep slopes where there is an increase in run-off and erosion when the slope is
23 $>9^\circ$ (Berael et al., 1995), which produces a contagious spread of erosion processes. Along a
24 degradation gradient, the long-range spatial autocorrelation of NDVI (α -DFA) was greater on high
25 (1.081 ± 0.024 , $n = 24$) than on low (0.865 ± 0.023 , $n = 24$) slope (one way ANOVA $F_{1, 46} = 41.85$, P
26 < 0.001) (Fig 4), which reflects the stronger effect of run-off processes on high slopes. NDVI self-
27 organization (α -DFA) declined with the amount of bare soil equally on high and low slopes, but the
28 effect was more pronounced on low slopes ($R^2 = 0.19$, slope = -0.81, $P < 0.05$) than on high slopes
29 ($R^2 = 0.07$, slope = -0.34, $P > 0.05$). In steep-sloped areas, the shallow roots of *S. tenacissima* are
30 exposed as the fine soil is lost during rain run-off, which reduces the competitiveness of this species
31 (Alados et al., 2006) and the negative relationship between α -DFA of NDVI and bare soil rate. In
32 contrast, in shallow-sloped areas in alpha steppes dominated by *S. tenacissima*, the species plays a
33 more important role in the organizing process.

1 The ecological strategy of the species occupying the habitat mediates the relationship between
2 species richness and NDVI spatial organization. In the dwarf matorral, the long-range spatial
3 autocorrelation of NDVI (α -DFA) declined as species richness increased ($R^2 = 0.14$, slope = -0.008 , P
4 < 0.05) (Fig. 5). Thus, when the spatial autocorrelation of NDVI is the result of run-off processes
5 that occur in slope areas, species richness declines as the contagious spread of erosion processes
6 increase. In contrast, as the spatial autocorrelation of NDVI in alpha steppe increases, the number of
7 species increases, particularly in shallow-sloped areas, where the contribution of *S. tenacissima* to
8 NDVI spatial organization is most apparent, which results in an increase in species diversity with α -
9 DFA of NDVI ($R^2 = 0.25$, slope = 0.004 , $P < 0.05$). The effect in high-slope areas is less evident (data
10 not shown). Thus, the NDVI spatial pattern is not always associated with a significant change in
11 species richness. While our study demonstrated a significant relationship between satellite-based and
12 ground-based data, some of the variance documented in the ground-based data remain unexplained
13 by the satellite-based methods (see also Hall et al. 2010). Other studies in semi-arid Mediterranean
14 ecosystems did not detect a relationship between spatial heterogeneity of NDVI and species richness
15 (Levin *et al.*, 2007).

16 Before landscape satellite images can be used to monitor ecosystems, we need to understand the
17 relationship between the spatial patterns in the satellite-derived vegetation index and ground-based
18 data and the mechanisms that trigger the self organizing positive feedback process. Once that is
19 understood, we can use the changes in the spatial autocorrelation of NDVI to monitor the thresholds
20 of change associated with irreversible desertification.

21

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Legends

Fig. 1. a) Limits of Cabo de Gata-Nijar Natural Park (SE Spain). Black dots are ground-transects areas. The box in the north represents the study area covered by the Ikonos-2 Image. b) Ikonos-2 image acquired on 2005-04-11. The 84 random image-transects are overlapped. c) Detail of the six random image-transect replicas in one of ground-transect areas (Montano Low Slope). d) Detail of the extraction points of NDVI values\ every 4 meters along one of the image-transect.

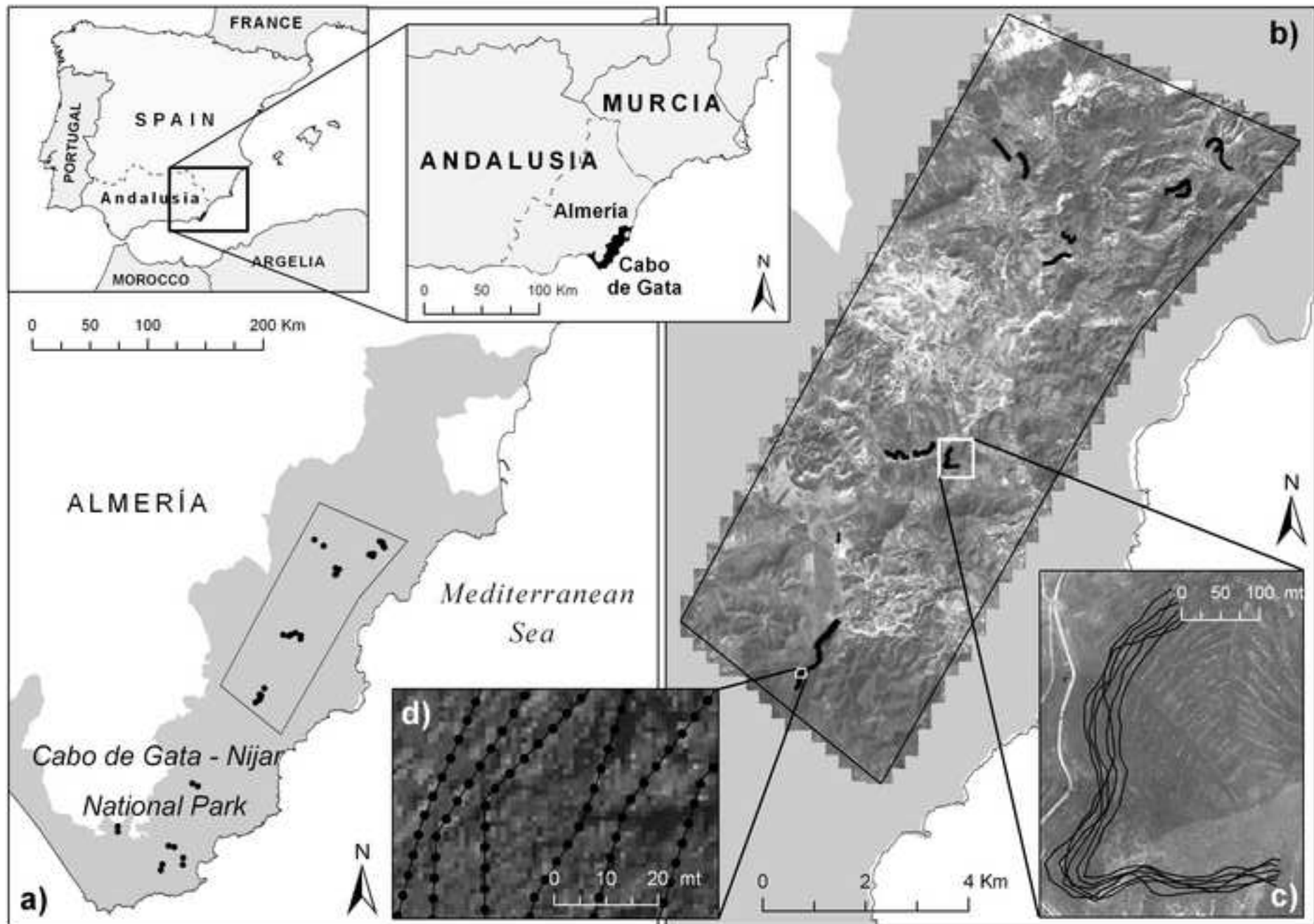
Fig. 2. Box-plot of the Information Fractal Dimension (IFD) obtained from 69 500-m transects collected at Cabo de Gata –Nijar National Park, Spain, showing significant differences between dwarf and alpha steppe communities.

Fig. 3. Scatter plot and linear regression fit between the long range spatial autocorrelation of NDVI (α -DFA) and bare soil rate (a) and species richness (b) obtained from 36 500-m transects extracted from Ikonos-2 image at the dwarf matorral of Cabo de Gata –Nijar National Park, Spain.

Fig. 4. Mean and standard error bar diagram of the long range spatial autocorrelation of NDVI (α -DFA) and bare soil rate obtained from 48 500-m transects extracted from Ikonos-2 image at the alpha steppe of Cabo de Gata –Nijar National Park, Spain, under high (20°-30 °) and gentle slope (< 10°) separately.

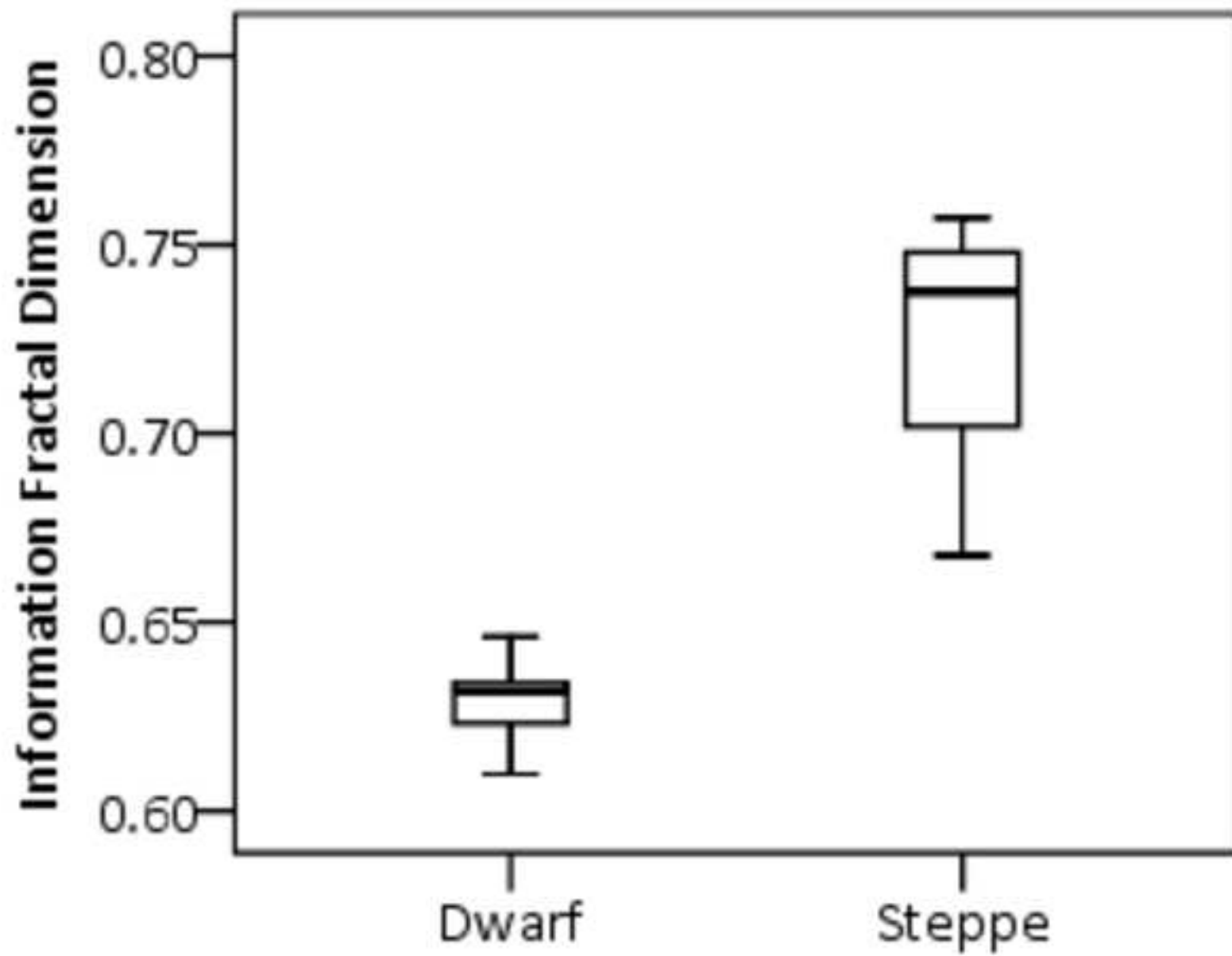
Fig. 5. Scatter plot and linear regression fit between the long range spatial autocorrelation of NDVI (α -DFA) and bare soil rate (a) and species richness (b) obtained from 24 500-m transects extracted from Ikonos-2 image at the gentle slope (< 10°) alpha steppe of Cabo de Gata –Nijar National Park, Spain.

Figure
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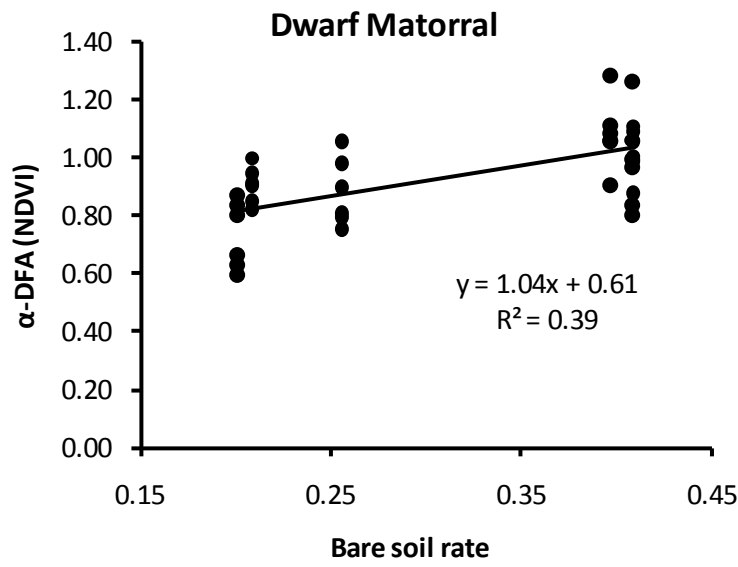


Figure

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a)



b)

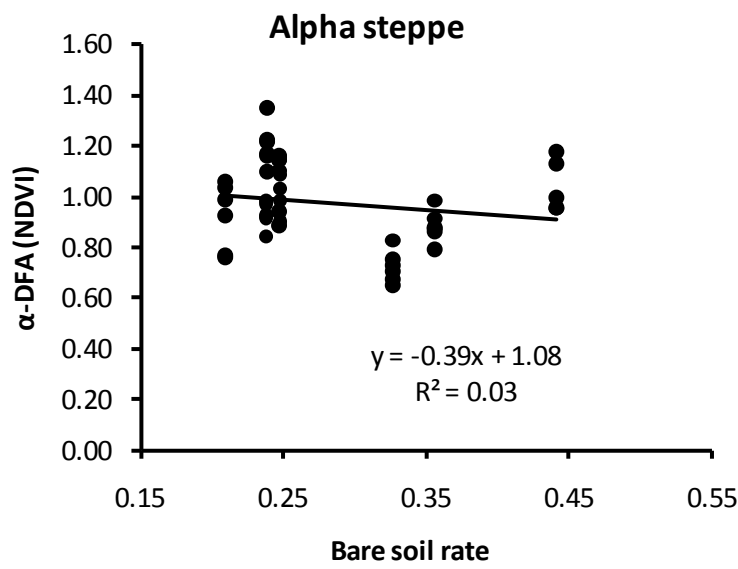


Fig. 3

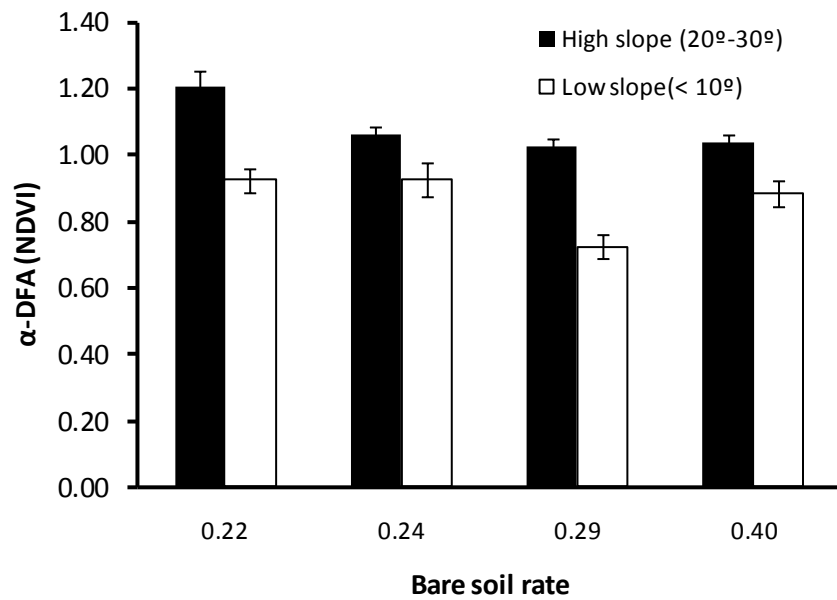


Fig. 4

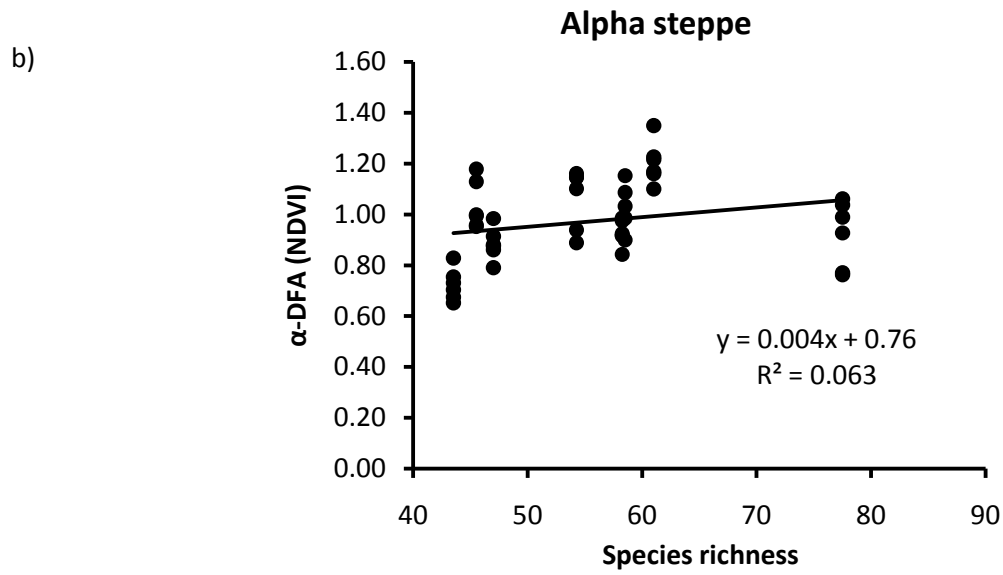
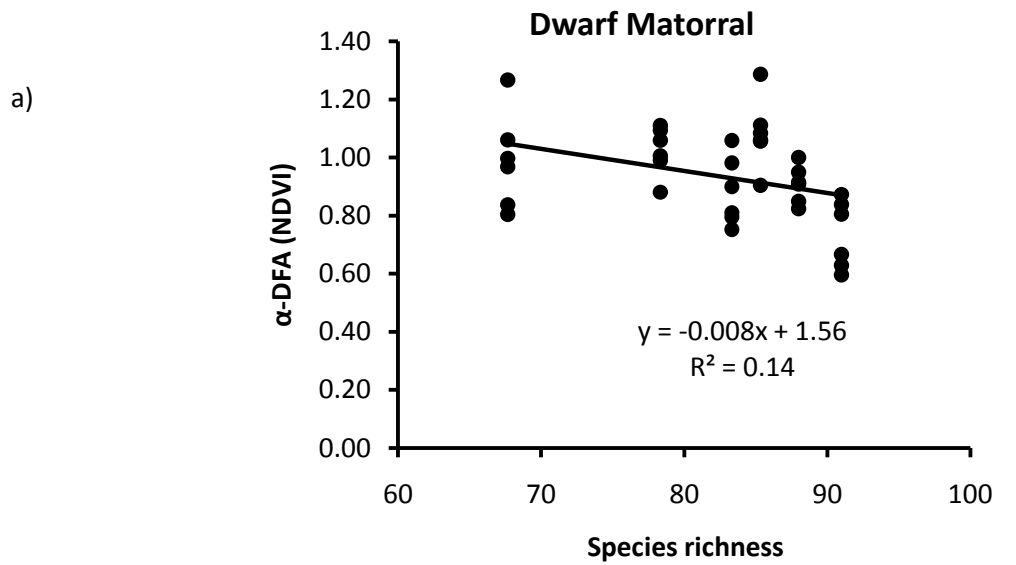


Fig. 5