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Trend analysis of Landsat-TM and -ETM+ imagery to monitor grazing impact in a rangeland ecosystem in Northern Greece

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ABSTRACT

Mediterranean rangelands are unique marginal ecosystems, which are characterized by a highly heterogeneous structure and are often interwoven with other ecosystems. Traditionally, rangelands provided resources for livestock grazing in transhumantic rotation schemes. In recent times, there has been a trend towards semi-intensive grazing systems, which is partly connected to the European system of agricultural and infrastructural subsidies, and which effectuates both intensification and extensification. This study employed trend analysis of a remote sensing data time series for a retrospective assessment of rangeland processes, and interpreted these in the light of land-use practices and previous management interventions.

We have selected a test area in Northern Greece that is representative of typical land-use transitions of the European Mediterranean. A time series of Landsat TM and ETM+ data covering the years 1984–2000 with one image per year was acquired, and for all images a geometric correction including digital elevation information and full radiative transfer modelling were carried out to attain surface reflectance data. For further analyses, proportional vegetation cover was selected as the target indicator, which was derived using Spectral Mixture Analysis. The resulting data set was used in a linear trend analysis to characterize spatio-temporal patterns of vegetation cover development. These could be interpreted based on knowledge of the local grazing regime and factors driving it, as well as using auxiliary spatial data sets. Results showed that temporal trends in the test area reflect the underlying pattern of potential livestock distribution at the per-pixel level, with a spatially differentiated pattern of both positive and negative trends in close proximity. On the other hand, no direct relation could be established between the development of vegetation cover and animal stocking rates at the community level. This suggests that this aggregation level is too coarse given the combination of highly heterogeneous landscapes with semi-intensive to intensive land tenure systems.

1. Background

1.1. Mediterranean rangelands

In the Mediterranean Basin, rangelands are highly heterogeneous, small-structured ecosystems, often interwoven with cultivated areas (Di Castri, 1981). Although these are marginal lands, they serve a variety of uses, including forage production. Mediterranean rangelands have a very long land-use history, often resulting in quasi-equilibrium states of semi-natural ecosystems (Di Pasquale et al., 2004). However, in the last decades, widespread land-use transformations have affected many regions of the European Mediterranean, leading to a departure from this equilibrium through both, intensification and extensification of land use. Resulting conflicts between ecological and economic priorities may lead to land degradation processes, which are aggravated by specific climatic and ecological properties encountered in the Mediterranean Basin (Hobbs et al., 1995; Noy-Meir, 1998; Stafford-Smith & Reynolds, 2002).

Livestock husbandry is a vital element of rural economies in Mediterranean rangelands (Le Houérou, 1981; Naveh, 1988). In the early 20th century, animal numbers decreased in the countries of the Northern Mediterranean, accompanied by land abandonment in marginal regions (Le Houérou, 1981). Yet, the political and socio-economic boundary conditions have drastically changed following the accession of Mediterranean countries to the European Community and later European Union (France, Italy: founding members 1951, Greece: 1981, Spain: 1986, Slovenia and Republic of Cyprus: 2004). Most of these countries have benefited ever since from different European funding schemes supporting rural infrastructure. In Greece, the Common Agricultural and Rural Policy (CAP) has revived livestock grazing through per-capita subsidies and led to re-increasing animal

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numbers in many areas (Dubost, 1998). Nonetheless, rising animal numbers did not coincide with a return to traditional transhumantic practices everywhere; rather, sedentary systems were frequently installed (Legg et al., 1998). The provision of additional feedstuffs, the ample availability of water, and animals returning to their sheds in the evenings are characteristic of such semi-intensive systems (Oba et al., 2000; Papanastasis, 1998).

Grazing of domestic animals is an important agent of land degradation in Mediterranean rangelands. It strongly affects plant physiology, for example through the reduction in photosynthetic leaf area, and environmental modifications for surviving plants (Briske and Noy-Meir, 1998). Furthermore, plant communities may be modified by favouring grazing resistant or tolerable species (e.g. *Quercus coccifera*) (Alados et al., 2004; Le Houérou, 1981). At the landscape level, reduced vegetation cover may increase soil erosion risk, while animal trails are potential starting features for linear erosion, and livestock trampling frequently leads to soil compaction and stratum loss, resulting in reduced water availability and soil fertility, as well as changes in soil texture (Thornes, 1990).

1.2. Remote sensing of grazed rangelands

Monitoring rangeland processes may be approached by using concepts from various fields of remote sensing and land-use (change) science. These need to be specifically combined to account for the specific properties of the small-structured, heterogeneous areas found in the European Mediterranean. On the one hand, different spatial and temporal observation scales may be chosen depending on the analysis task. On the other, dynamic aspects might be considered by assessing changes between classes or within classes. In the latter case, the sparse vegetation often found in such areas demands specific data interpretation strategies.

Remote sensing offers unique opportunities to monitor landscape processes, reflected in a large number of studies on land use/land cover change (Coppin et al., 2004; Lu et al., 2004). In these studies, a wide range of mostly diachronic techniques was suggested, such as change vector analysis (García-Haro et al., 2001; Johnson & Kasischke, 1998), multi-temporal PCA (García-Haro et al., 2001) or image differencing (Rogan & Yool, 2001), or a combination thereof (Petit & Lambin, 2001). In addition, multi-temporal analyses are complemented by the assessment of grazing-induced spatial patterns for singular dates (Pickup & Chewings, 1994; Röder et al., 2007; Sparrow et al., 1997; Washington-Allen et al., 2004). Such local-scale applications are often accompanied by approaches based on small-scale land-use change analyses that focus on seasonal dynamics (Archer, 2004; Lambin, 1996; Lambin & Ehrlich, 1997). In a recent study, Geerken and Ilaiwi (2004) analyzed desertification processes in Syria by integrating local knowledge with multi-temporal data sets from Landsat MSS/TM and NOAA-AVHRR.

While land use/land cover change analysis is well-suited to characterize major changes in land-use classes, it is not capable of quantifying gradual or 'slow' processes (Mulligan et al., 2004) within one class. In addition, quantifying grazing impact (compare Section 1.1) directly is not possible using remotely sensed imagery (Hill et al., 2004), such that suitable surrogates are required. In this respect, vegetation cover is a good indicator because it is directly affected by grazing animals, closely connected to above-ground biomass (Chiarucci et al., 1999; Tsiourlis, 1998), and may be related to processes of accelerated erosion, degradation, increase of flammable biomass etc. (Seligman & Perevolotsky, 1994; Thornes, 1990).

Quantifying vegetation cover from remote sensing data has traditionally been based on spectral vegetation indices (e.g. the Normalized Difference Vegetation Index, NDVI). Such indices suffer from a number of drawbacks, including saturation effects for dense vegetation canopies, and a negative influence of soil background especially for bright soils and sparse vegetation canopies (Baret et al., 1993; Rondeaux, 1995). Vegetation indices are confined to the pixel as the reference spatial entity, which is problematic for highly heterogeneous Mediterranean environments, where the signal is often a mixture of different surface materials (Fisher, 1997). Spectral Mixture Analysis (SMA, Smith et al., 1990b) allows to decompose the signal into proportions of reference materials (so-called 'endmembers') and is superior to traditional vegetation indices in sparsely vegetated areas (Elmore et al., 2000). SMA was successfully applied to characterize vegetation cover in semi-arid environments using both multi- and hyperspectral systems (Hostert et al., 2003a; McGwire et al., 2000; Okin et al., 2001; Roberts et al., 1993; Smith et al., 1994). Endmembers may be chosen based on actual reflectance measurements or using image endmembers, although these may introduce ambiguities as image endmembers are often themselves mixtures of different materials (Ustin et al., 1993). The approach may be extended by multiple endmember setups (García-Haro et al., 2005; Roberts et al., 1998) and multi-date approaches (Kuemmerle et al., 2006b; Shoshany & Svoray, 2002) to differentiate plant communities.

Despite the number of studies deriving vegetation cover using SMA, not many studies exist that build on time series of such data to characterize temporal dynamics in grazing-affected environments. To date only Hill et al. (2004) and Hostert et al. (2003b) have attempted a per-pixel, long-term analysis of temporal trends in grazing areas using dense, local-scale time series, and aimed at interpreting these in the context of the local grazing regime. In a traditional, extensive grazing area of central Crete they detected widespread land degradation matching the development of animal numbers and local transportation infrastructure.

As a consequence of these different aspects, a remote sensing based monitoring approach for grazing-affected rangelands needs to meet three basic prerequisites:

- a suitable indicator of grazing impact needs to be derived in an optimized manner
- temporal dynamics need to be adequately characterized by the procured dataset
- the remote sensing based results need to be interpreted under consideration of the local process framework.

2. Test site and objectives

2.1. Physical setting

The county of Lagadas belongs to the Region of Central Macedonia and is situated north of the Chalkidiki peninsula and east of the city of Thessaloniki, bordered by a frame between 22°56′56.85″E/41°00′07.82″N (ULX/ULY) and 23°45′55.26″E/40°28′56.72″N (LRX/LRY). The landscape is clearly structured into different elevation zones (Fig. 1).

While natural vegetation partially corresponds to these zones, there is a mosaic of land uses of various types and intensities according to which different pressure systems operate. There are some zones of intensified agriculture, but the region is dominated by its mosaic of rangelands mainly used for grazing, with embedded smaller agricultural plots.

Climate is typical Mediterranean, with annual precipitation between 410 mm and 685 mm and maximum rainfall in spring and autumn, although strong precipitation events may occur at other times.

Bedrock of the area is relatively homogeneous, with metamorphic rocks being abundant, accompanied by pyroclastic rocks and limestones in some areas. Shallow soils (leptosols and regosols) are most frequent, complemented by cambisols where local conditions sustain a thorough pedologic development. In larger depressions and intermediate plain areas, which are mainly used for agriculture, quaternary deposits and alluvial soils dominate.

The highest elevations of the County are dominated by submountainous beech forests and thermophilous oak forests, although the latter is frequently observed in a degraded state due to grazing and

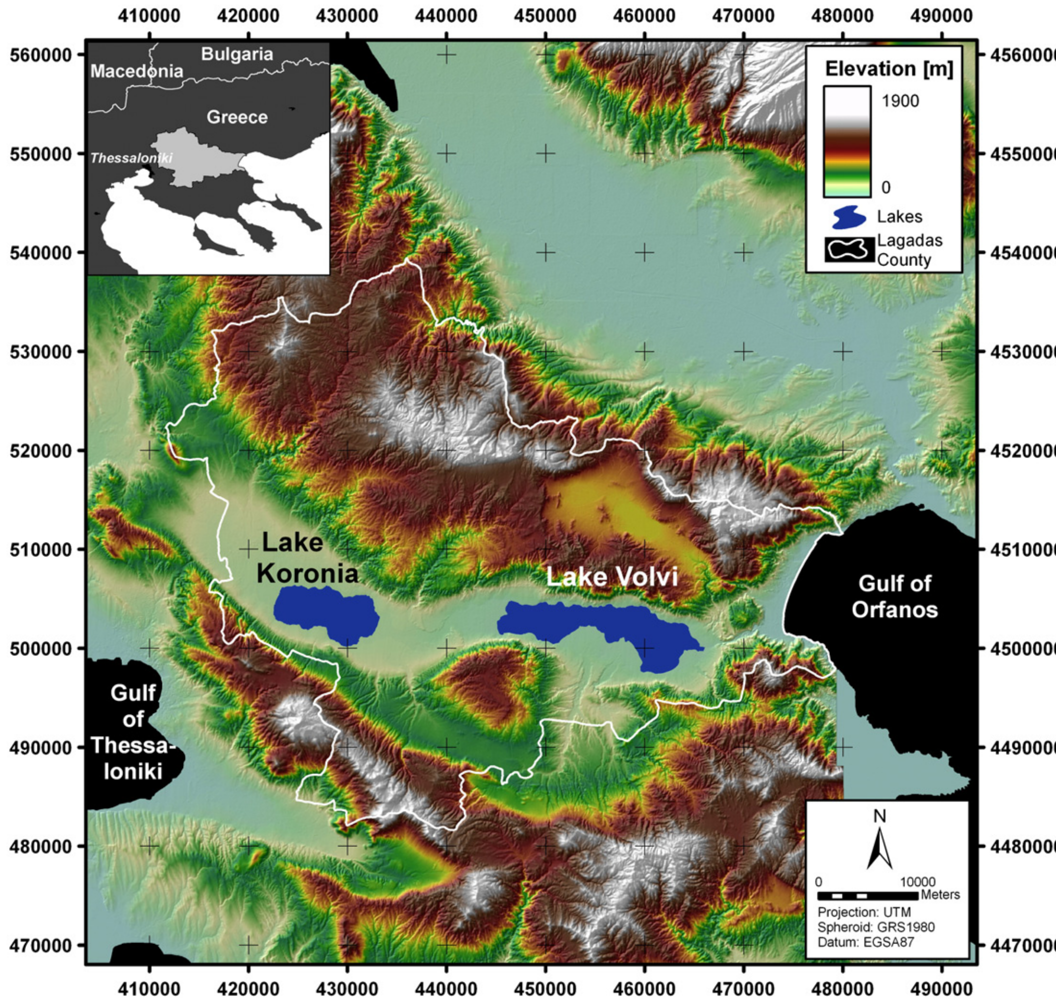

Fig. 1. Relief representation of the test area in Northern Greece with perimeter of Lagadas County.

forestry. Rangelands are dominated by grasslands and *Q. coccifera* shrublands. These shrubs and trees are highly adapted to rangeland environments due to their spiny small leaves, their resilience to fires and their fast resprouting and coppicing abilities (Di Pasquale et al., 2004). Herbaceous species are numerous in the shrubland areas and include Labiatae and Fabaceae. In the grasslands, annual species together with some perennial grasses occur (Konstantinidis & Tsiourlis, 2003). Because grasslands are a result of intense grazing activities, isolated thorny shrubs, such as *Q. coccifera*, *Crataegus monogyna*, *Juniperus oxycedrus* and *Prunus spinosa*, are common.

2.2. Socio-economic setting and objectives

In the county of Lagadas, rural economy is strongly dependant on agricultural production, fishery at the coast and in Lake Volvi, and livestock grazing in the rangeland areas. Besides, the economical and tourist centres city of Thessaloniki and the Chalkidiki peninsula are additional sources of income in the region.

Historically, the husbandry system can be characterized as a shepherded, extensive grazing system, while in the winter period animals are occasionally held in paddocks (Hadjigeorgiou et al., 1998; Yiakoulaki et al., 2002). The historic development of grazing largely follows the general fluctuation described before (Section 1.1) and it was only after Greece joined the EC in 1981 that animal numbers rose again. Especially the recent trend towards milk production aggravated the departure from the traditional transhumance system and replaced seasonal migration of herds by a more sedentary grazing system (Yiakoulaki et al., 2002). This lead to overgrazing for regions where livestock is concentrated. Contrarily, less accessible areas are often undergrazed, which may result in a thickening of shrublands and affects species composition, hydrological cycles, fire risk etc. (Noy-Meir, 1998).

During the summer months, grazing is often confined to the morning and late afternoon hours and animals are returned to their sheds during the hottest hours. In winter most time of the day is spent ruminating. Herds are usually bound to specific sheds, from which grazing activities originate, and transportation to more remote areas does not play a major role. This practice is supported by an increasing use of feedstuffs, which in turn contributed to the intensification of irrigated agriculture in the basin of Mygdonia (Yiakoulaki et al., 2002).

Given the present grazing scheme and the coexistence of intensification and extensification, this study aimed at identifying spatio-temporal patterns in grazing areas and investigating the possible influence of socio-economic driving factors. This was pursued by evaluating the validity of two major hypotheses:

a) The coexistence of intensification and extensification of grazing is reflected in the pattern of temporal trends of vegetation cover.
b) The development of animal numbers and stocking rates determines the development of vegetation cover.

3. Materials

A long time series of 15 Landsat-5 TM and Landsat-7 ETM+ data was acquired for the test area, covering the years 1984 to 2000 with one image per year where appropriate images were available (data missing: 1989 and 1991). These were selected to represent the period of maximum photosynthetic activity in the rangeland area around June. All scenes were supplied from Eurimage© as system-corrected products. A digital elevation model (DEM) at 30 m grid resolution was made available from Geoapikonisis Ltd., which had been derived from photogrammetric analysis of digital aerial photographs.

A detailed habitat map was derived from the visual interpretation of digital aerial photographs acquired at a scale of 1:20,000 in 1980, which was complemented and updated by numerous field surveys carried out in 2000 and 2001 by ecologists from the National Agricultural Research Foundation (NAGREF, Vasilika-Thessaloniki, Greece). Following the Natura 2000 habitat mapping key and EU directive 92/43, this vector data set provided information on dominating plant communities, including structural information and their average ground cover for 28 classes. In addition, agricultural, abandoned and barren areas were represented (Konstantinidis & Tsiourlis, 2003; Röder, 2005). Furthermore, vector data sets delineating administrational boundaries, settlements and water bodies were made available through local authorities. In addition to the spatial data layers, animal census information for sheep, goats and cattle was available from the National Statistical Survey of Greece for the communities of Lagadas County at decadal intervals from 1961 to 1991 and for 2002. In May and June 2002, vegetation cover was mapped for a total of 11 plots representing grasslands with a very low presence of shrubs, and sparse, medium dense, and dense shrublands. Two diagonal transects with a length of 35 m and intersecting at the centre coordinate were set up. Along each transect, 8 sampling locations of 1 m diameter were positioned to map ground cover of trees, shrubs, forbs and grass, which was integrated into a single cover estimate.

To support validation of the radiometric correction and definition of the endmember model, an extensive data base comprising more than 300 spectral reflectance measurements of typical Mediterranean soils, rocks and plant types was available from different campaigns. The spectra were measured using an ASD II Fieldspec Full Range spectroradiometer manufactured by Analytical Spectral Devices©. It measures reflectance between 350 and 2500 nm in 2151 bands and provides spectra interpolated to a spectral resolution of 1 nm; a Spectralon panel was used as spectral reference. Measurements were taken under natural illumination conditions on site for different vegetation types and species, which integrate the signal of whole specimen, including green components as well as stems and branches. These were complemented by laboratory measurements of fresh leave-stacks. Bedrock and soil samples were analyzed in the laboratory, measuring both natural and freshly cut rock surfaces, as well as soil samples in their original and in sieved and homogenized state. All of the samples were categorized and attributed to allow their subsequent stratification based on dominant vegetation types as well as lithological and soil conditions.

4. Methods

4.1. Satellite data pre-processing

The thematic analysis of multi-temporal data series requires differences between images to result exclusively from changes in surface properties, necessitating a precise geometric and radiometric correction of incorporated images (Song et al., 2001).

A topographic map 1:50,000 was used to identify ground control points (GCPs). Assuming that system-corrected Landsat data exhibit — apart from relief displacement — only scale and rotational distortions, a first order polynomial solution was employed to register one master image to the Greek UTM reference system (UTM projection, GRS 1980 spheroid, EGSA 87 datum) incorporating the DEM (Itten and Meyer, 1993). Subsequently, the remaining images were corrected based on a large number of GCPs identified using cross-correlation search windows (Hill & Mehl, 2003; Kuemmerle et al., 2006a).

Images were converted to surface reflectance using an approach that comprised sensor calibration and full radiative transfer modeling, including a correction of topography-induced illumination variations (Röder et al., 2008). To compensate for decreasing sensor sensitivity, data from vicarious calibration experiments (Teillet et al., 2001; Thome et al., 1997a,b) were used to calculate a time-dependent calibration function for Landsat-5 TM data (Teillet & Fedosejevs, 1995), while calibration of the ETM+ image was based on parameters published by the US Geological Survey (USGS, http://landsat7.usgs.gov/cpf/). Subsequently, a radiative transfer model based on the 5 S Code by Tanre et al. (1990) was parameterized for each scene. Atmospheric transmission factors were calculated using the Modtran-4 code (Berk et al., 1999) and the scattering behavior of aerosols at different viewing angles was characterized according to Aranuvachapun (1985). The relative constellation between sun, surface and sensor strongly affects radiance fluxes and the illumination budget. This was accounted for in the radiative transfer calculation by considering different radiance terms calculated from the digital elevation model for each acquisition date (Hill et al., 1995). The direct irradiance flux was corrected using a cosine correction. For the diffuse radiance term, isotropic and anisotropic components were separately corrected based on the anisotropy index (k) (Hay & McKay, 1985; Hay et al., 1986). The anisotropic component has its maximum in the circumsolar and circumzenithal region (Iqbal, 1983), and was cosine-corrected analogous to the direct irradiance term, while the isotropic component was corrected by considering the visible sky portion (H) for each pixel.

4.2. Spectral Mixture Analysis

Given the typical properties of sparsely vegetated Mediterranean rangelands, linear Spectral Mixture Analysis (Adams et al., 1986; Smith et al., 1990a,b) was chosen to infer quantitative estimates of proportional green vegetation cover on a per-pixel basis, which may be interpreted in the context of land degradation processes. Considering the spectral dimensionality of Landsat TM/ETM+ (Small, 2004), a pixel-adaptive, multiple 4 endmember model was set up by assessing the position of candidate endmembers. These had been selected from the spectral library in the feature space of the Landsat-7 ETM+ image that was used as a reference (Fig. 2). The model was composed of an integrated *Q. coccifera* spectrum and a zero shade endmember, which were forced to be included for all pixels, while laboratory measurements of a weathered gneiss rock and a developed cambisol soil were introduced to represent potential background

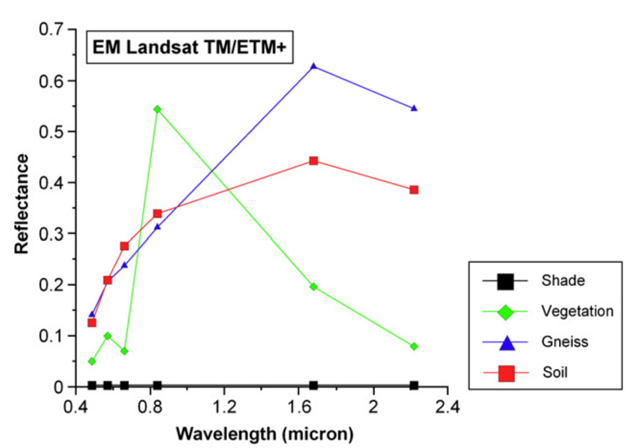

Fig. 2. Four endmember model employed to derive green vegetation cover for the Landsat-TM/-ETM+ time series.

materials. The latter two were selected per-pixel based on the lowest model RMS-error.

Although it is generally desirable to represent different types of vegetation (e.g. trees, shrubs, grasses and herbaceous) either within one model or through a stratified approach (Xiao & Moody, 2005), this was not possible due to the spectral dimensionality of Landsat TM/ETM+ (Small, 2004). Since photosynthetically active vegetation is distinct from other materials, it can be derived with high confidence while accepting larger error quantities introduced from a limited representation of potential background materials. Accordingly, abundance estimates for the different endmembers were derived according to

$$R_i = \sum_{j=1}^{n} F_j \cdot \mathrm{RE}_{ij} + \varepsilon_i \text{ and } \sum_{j=1}^{n} F_j = 1 \tag{1}$$

with

R_i= reflectance of the mixed spectrum in band i
RE_{ij}= reflectance of the endmember spectrum j in band i
F_j= fraction of endmember j
n= number of spectral endmembers
ε_i= residual error in band i.

The bias introduced by representing different types of vegetation with one endmember is partially compensated for through the shade endmember, which acts as a scaling factor in the subsequent normalization according to

$$f = \frac{1}{(1 - F_{\mathrm{Shade}})} \text{ and } \sum_{j=1}^{(n-1)} F_j \cdot f = 1 \tag{2}$$

with

f= shade normalization factor
F_{Shade}= fraction estimate for the shade endmember
F_j= fraction estimate for endmember j.

In the absence of prior knowledge of vegetation structures, this approach is a simplified alternative to the application of non-linear mixture models (Ray & Murray, 1996).

4.3. Linear trend analysis

Time series of data may be differentiated into deterministic (transient and cyclic) and noise components of stochastic nature (Schlittgen & Streitberg, 1999). These may be separated if data exhibit sufficient temporal coverage, such as hypertemporal NOAA-AVHRR data at daily to decadal intervals (Andres et al., 1994; Brunsell & Gillies, 2003). In such cases, seasonal vegetation cycles may be represented by phase, amplitude or length of growing season, while gradual long-term dynamics may be described by the linear trend as well as a phase shift indicating a potential change in plant communities. However, the rather coarse geometric resolution of such systems limits spatial patterns that can be resolved, which makes them less suitable for use in small-structured Mediterranean landscapes. In these cases, series of Landsat TM/ETM+ or Spot-XRS offer higher potential, but their temporal coverage and required processing efforts confines time series analyses of such data to the transient or linear component (Hostert et al., 2003a; Röder et al., 2008). We have hence represented rangeland dynamics by the available data set of green vegetation cover estimates. While spontaneous processes (e.g. fire) require a temporally stratified trend analysis (Röder et al., 2008), grazing effectuates 'slow' or gradual change of vegetation cover, justifying application of a consistent linear regression function. This was calculated for each pixel to minimize the sum of least squares between given and estimated values for each fix point (i.e. date available in the time series), yielding a function of the type:

$$y_t = g \cdot t + o \tag{3}$$

with

y_t= vegetation cover at date t
t= date of image acquisition (e.g. in days since launch of sensor)
g= regression coefficient (gain)
o= regressions constant (offset).

The offset of the resulting function characterizes the level of vegetation cover estimated for the date of sensor launch. Alternatively, it can be related to the starting date of the observation period if t_0 is set to the date of the first image acquisition. The gain describes the direction and magnitude of the development over the monitoring period. As these regression parameters are calculated on a per-pixel basis, the temporal development can be illustrated in a spatially differentiated way.

To assess the statistical robustness of the trend, the correlation coefficient (r), coefficient of determination (r^2), standard deviation (SD), two-sided t-test and root mean squared error (RMSE) were calculated, the latter being useful in areas with stable estimates as it is independent of the direction and magnitude of the regression coefficient.

Table 1
Factors considered in the topographic regionalization procedure and associated processes

Factor	Abbreviation	Comment
Slope angle [degrees]	SLO	First derivative of the DEM surface, associated to erosion potential
Profile curvature	PFC	Second derivative of the DEM surface, represents flow acceleration/ deceleration along hillslopes
Plan curvature	PLC	Second derivative of the DEM surface, represents flow convergency/ divergency across hillslopes
Catchment size [area] (Quinn et al., 1991)	ARE	Upslope area draining through each cell, associated to runoff potential
Wetness index (Beven & Kirkby, 1979)	ATB=ARE/SLO	Soil water accumulation in terms of the potential runoff and the water evacuation by gravitational forces
Slope length factor (Moore & Birch, 1986)	$\mathrm{LSF}=(\mathrm{ARE}/22.13)^{0.6} * (\sin(\mathrm{SLO}/0.0896)^{1.3}$	Related to potential sediment transport both in hillslopes and in stream channels
Distance to the nearest stream [m]	STRD	Based on a reclassification of ARE as a reference stream network
Solar exposure index	SUN	Computed from slope and aspect and referring in relative positive or negative values to a flat surface

4.4. Topographic regionalization

Water is a limiting factor in Mediterranean ecosystems. In turn, topography is the main control of water redistribution in the landscape (Wilson & Gallant, 2000). The test area has a distinct and complex relief that might explain, at least partially, the development of the vegetation cover in terms of potential water availability. Further to that, topography imposes also limitations to movement patterns of animal and shepherds and it can therefore have an indirect control on grazing intensity. For those reasons, a topographic regionalization of the test area was used to assess possible relationships between terrain complexity and vegetation cover development. It was carried out by applying Numerical Taxonomy techniques on a set of topographic variables. The full procedure is described in Del Barrio et al. (1996) and it is summarized below.

Table 1 presents the variables derived from the DEM by using appropriate algorithms.

Those layers were exported to a large data matrix with as many columns as layers being used (eight in this case) and as many rows as working cells are in a single layer. The classification was made in two steps, both of them using the Gower index as a dissimilarity measure. First, a Non Hierarchical clustering algorithm was applied to the cells to generate 34 preliminary groups. This step is done to avoid building an association matrix and to capture the main sources of variability in the data set (Belbin, 1987). Then, the group centroids were further classified using a Sequential Agglomerative Hierarchical Non Overlapping clustering. The median centroids were fused using UPGMA (Sneath & Sokal, 1973) and the resulting dendrogram was cut at an appropriate level after visual inspection. This generated 10 super-classes (groups of groups of cells) that, through the preliminary group composition, were subsequently expanded to the final 10 classes of cells. All these procedures were carried out using the PATN package (Belbin, 1989).

The resulting classes were interpreted in terms of both position in catenas and statistical distributions of the original topographic variables, to yield the final regionalization with the following legend: convex upperslopes, low elevation divides, hill tops, high elevation ridges and divides, steep mid and low slopes, upper steep stream channels, gentle areas with a high soil wetness, low transport capacity stream channels and high transport capacity stream channels. These names describe the topographic characteristics of the classes. The classes are statistically different among them, and their spatial autocorrelation facilitates a consistent pattern of patches in the final map.

5. Results

5.1. Satellite image pre-processing

The master image was geometrically corrected as described in Section 4.1, with a total of 48 visually identified GCPs. The total RMSE amounted to 0.544 pixels and the corrected image was found to match the reference topographic map. Then, all remaining images were registered following the same procedure, but with a larger number of GCPs (between 66 and 142) due the cross-correlation approach. RMSEs lay between 0.122 and 0.2274 pixels, while visual inspection confirmed a high geometric consistency of the time series.

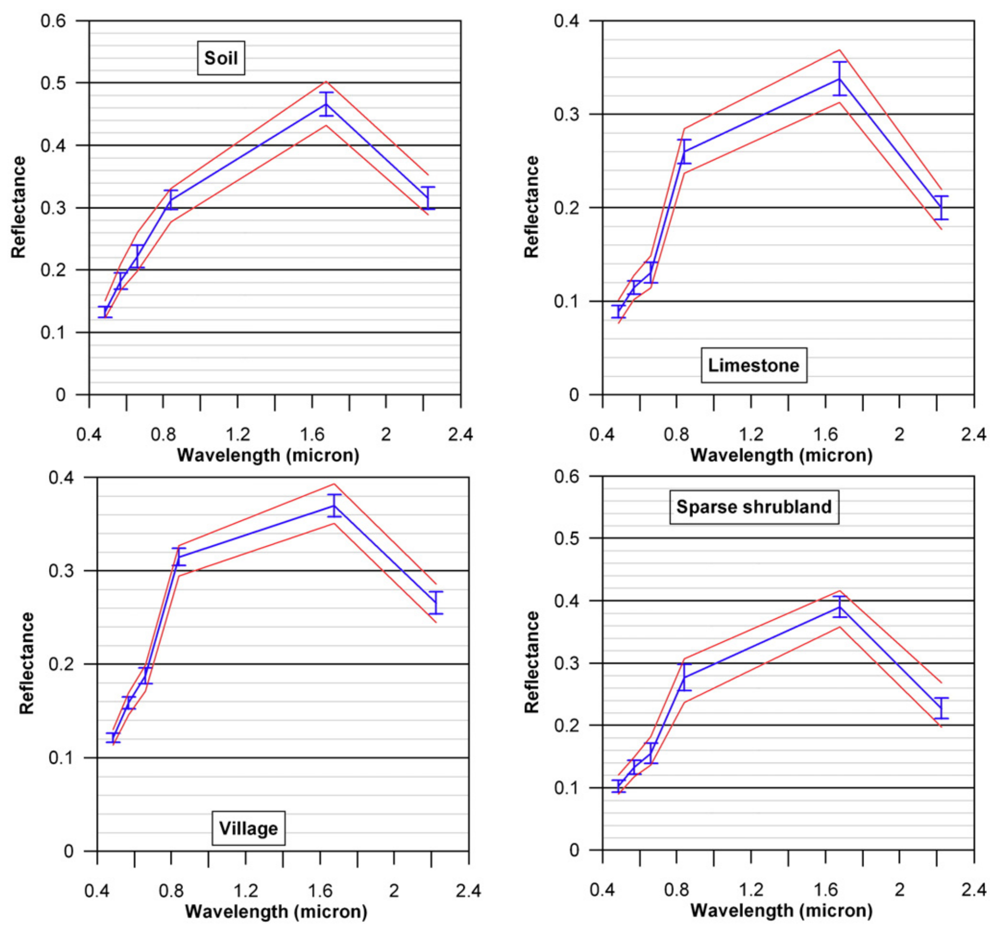

Fig. 3. Reflectance values for invariant targets after radiometric correction of the Landsat-TM/-ETM+ time series. Thick lines show average reflectance and standard deviations, dashed lines indicate overall minimum and maximum value for each feature (features include 15 to 80 pixels; scenes where the respective area was obscured by clouds were excluded).

Table 2
Areal statistics for the different degradation index classes [ha (%)]

	Low	Medium	High	Aggregated
Strong negative	609.03 (0.56)	5263.74 (4.85)	487.35 (0.45)	6360.12 (5.86)
Negative	1450.08 (1.34)	16,444.71 (15.16)	1560.51 (1.44)	19,455.3 (17.94)
Neutral	2211.21 (2.04)	34,739.19 (32.03)	9048.15 (8.34)	45,998.55 (42.41)
Increase	958.32 (0.88)	19,597.41 (18.07)	9182.7 (8.47)	29,738.43 (27.42)
Strong increase	212.22 (0.19)	4795.29 (4.42)	1888.29 (1.74)	6895.8 (6.35)
Sum	5440.86 (6.91)	80,840.34 (74.53)	22167 (20.44)	10,8448.2 (100)

Details on class ranges in the text.

Dark water signatures from the freshwater lakes within the test area and the Mediterranean Sea surrounding it were extracted from all scenes and processed using corresponding sensor calibration coefficients, but an identical set of atmospheric parameters. The resulting reflectance signatures were employed to assess aerosol and water vapour loading of the atmosphere, as well as the variability across the scenes. The image acquired in 1996 was identified as a suitable master image with intermediate reflectance levels and homogeneous conditions. For the image, the radiative transfer model was parameterized according to the methodology outlined in Section 4.1. The validity of the correction was confirmed by identifying spectrally ‘pure’ pixels (Boardman et al., 1995), using the lithology data set to assign them with the most probable bedrock or soil type, and comparing them with corresponding spectral reflectance measurements from the spectral library. Subsequently, 26 pseudo-invariant target areas were identified in all images. Sizes ranged from 5 to 51 ha for ‘pure’ regions, while also two stable rangeland areas with 200 and 500 ha, respectively, were included to represent large, homogeneous mixtures of different materials. Aiming at a close match of these areas, radiative transfer models were iteratively parameterized for all scenes and resulting signatures for the target areas were extracted. Results confirmed the quantitative consistency of the time series, and Fig. 3 shows average reflectance spectra, standard deviations as well as minimum and maximum reflectance values for exemplary target areas.

5.2. Spectral Mixture Analysis

Direct quantitative evaluation of vegetation cover estimates derived from SMA is complicated in highly heterogeneous and sparsely vegetated rangeland areas. One way of achieving an overall plausibility assessment is hence a combination of different validation approaches. First of all, the statistical validity of the spectral unmixing model was investigated by assessing histograms of the individual fraction estimates as well as the overall RMSE. For all images, histograms were generally found not to exceed the range of 0 to 1. In addition, residual bands exhibited no systematic patterns and overall RMSEs ranged below 2% reflectance for all images.

As it was closest in time to the years of the field campaigns, the image acquired June, 5th, 2000 was selected to validate the derived green vegetation cover estimates. Ideally, ground-based validation data should be mapped conceptually coherent with the target satellite data (i.e. green vegetation cover) and considering spatial autocorrelation in the sampling design and selection of sites (Atkinson, 1991; McGwire et al., 1993; Xiao et al., 2005). In our case, validation sites had been stratified using existing information on dominant plant communities to represent four distinctly different types of grassland/shrubland areas. As discussed in other studies focusing on

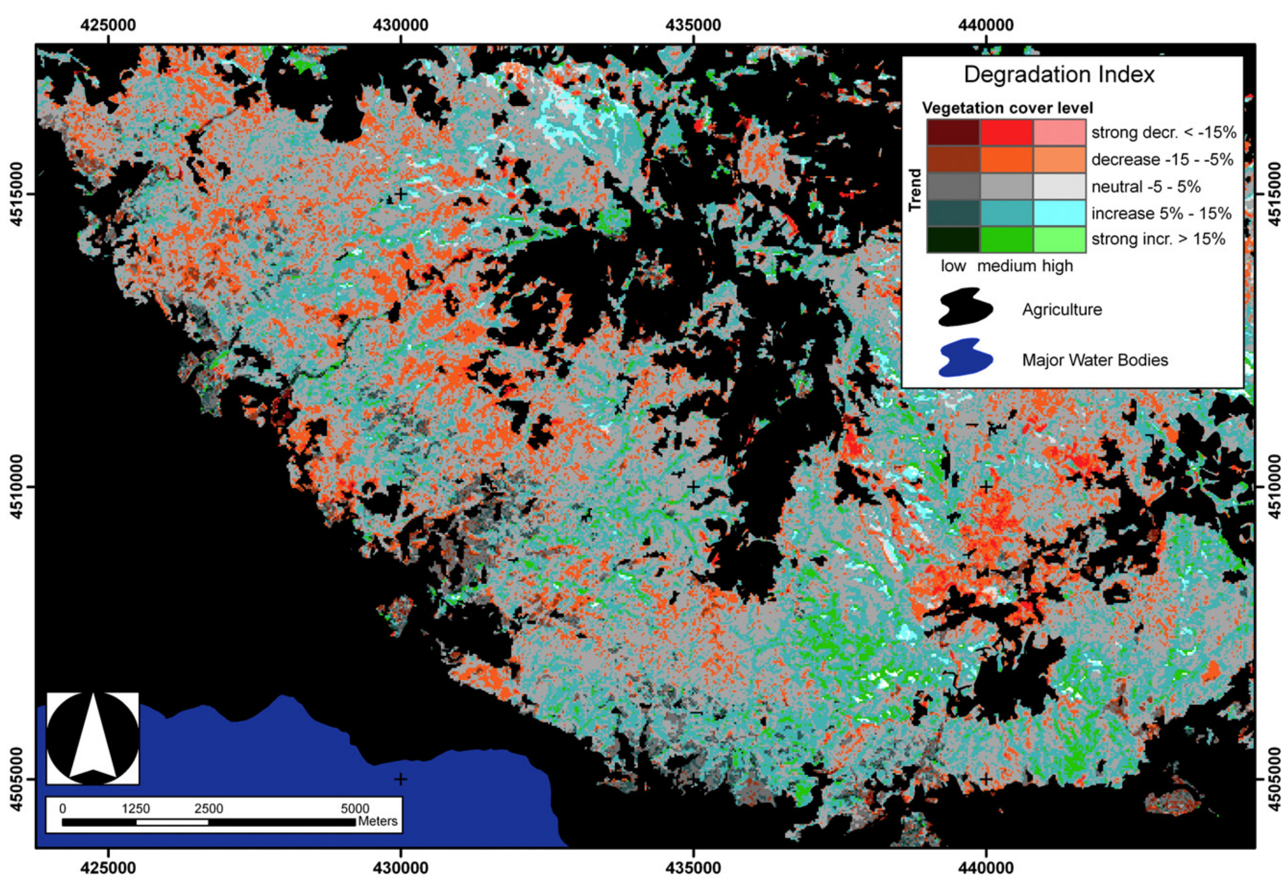

Fig. 4. Subset of the degradation index map integrating the gain coefficient and average value derived from linear trend analysis of the Landsat-TM/-ETM+ time series; agricultural areas were masked out based on the habitat vector information.

Mediterranean rangelands, a direct match between field-based estimates and satellite-based pixel- or buffer-estimates was impeded by the combination of the small-structured character of the rangelands as well as conceptual and scale-related constraints (Elmore et al., 2000; Kuemmerle et al., 2006b; Röder et al., 2008). Selecting the best-matching pixel in a search window (Elmore et al., 2000) resulted in a scatter plot oriented close to the 1:1 line, a correlation coefficient of 0.908 and a regression function given by $y = 0.81 * x + 0.091$. As SMA was shown to be very robust when green vegetation is assessed (e.g. Hostert et al., 2003a; Small & Lu, 2006) it could be assumed that applying the model to the full time series would result in a consistent time series of green vegetation cover.

5.3. Linear trend analysis

Considering the whole county area, different types of temporal profiles and corresponding statistical parameters were observed depending on the underlying type of process (Mulligan et al., 2004; Rindfuss et al., 2004b). Gradual, statistically significant trends with low to medium RMSEs were detected in areas where positive or negative development took place during the observation period. These were accompanied by temporal profiles showing no trend but again low to medium RMSEs in stable areas. On the other hand, 'disturbed' profiles appeared in forest clearcuts, agricultural areas and built-up land; given the focus of this study on grazing areas, these locations were masked out using the available vector information.

In the remaining investigation area, a distinct pattern was detected, where stability (i.e. no trend and low RMSE), increase and degradation of vegetation cover (positive/negative, significant trend with low to medium RMSE) appeared in close proximity. As the cover level at which processes take place is of great ecological importance, a degradation index was derived for the further interpretation similar to the approach adopted by Hostert et al. (2003b). It combined the direction and magnitude of the trend and the average level of vegetation cover during the observation period. Average cover values were assigned to three classes: low (0–35%), medium (>35–70%) and high (>70%) average cover. Similarly, overall gain rates were classified as follows: strong decrease (<−15%), decrease (−15–<−5%), increase (>5–15%) and strong increase (>15%), while the range between −5% and 5% was set to neutral to accommodate for possible errors in the processing and interpretation chain.

From the resulting spatial representation, areal statistics were derived for the grazed rangeland areas (Table 2).

Accordingly, less than half of the area shows a neutral behavior. Both negatively and positively developing areas yield higher percentages for the moderate compared to the strong gain classes. Within these aggregated classes, development on mean levels is dominating compared to low and high cover levels. This may, to a certain degree, also result from averaging over long periods. The spatial representation emphasizes the pattern described before, which is illustrated in Fig. 4 for a subset of the major grazing area of Lagadas County north of Lake Koronia.

Fig. 4 corroborates results from the gain representation described before and exhibits a pronounced spatial pattern. Spatially consistent patches of recovery and degradation emerge embedded in a matrix characterized as neutral. In general, negatively developing locations show a more compact texture, while for positively developing areas a reticulate pattern is prevalent.

This differentiated spatial pattern confirms the hypothesis that extensification and intensification of grazing activities in rangelands are reflected at the landscape level. Whether these patterns can be linked more explicitly to gazing properties will further be analyzed in Section 6.

5.4. Sensitivity

The trends calculated using time series analysis are potentially affected by different factors related to the processing and analysis steps, such as quality of the different steps in geometric and radiometric correction, as well as the related accuracy of green vegetation cover estimates derived from SMA. The latter also includes a potential mismatch between actual vegetation type and spectrum used, and phenological variations between the scenes. However, although accuracy estimates are provided for the different steps, these can only serve for a general estimation of the overall accuracy, but do not support a prediction of the accumulation of errors (Congalton & Green, 1999). In the context of the time series analysis, the impact of errors associated with a particular scene depends on the position of the respective scene in the time series. The effect is greater at the very beginning or end of the series, while errors in the middle of the series reduce significance of the derived trends without comprising the overall direction of the trend function (Schlittgen & Streitberg, 1999). In general, given an appropriate number of fix points (i.e. scenes) in a time series, sensitivity analyses indicated resulting trends to be very robust (Hostert et al., 2003b; Röder, 2005; Röder et al., 2008). In this context, the application of the same endmember model to all scenes adds to the stability of the trend analysis as it ensures a consistent relative level of cover estimates. Although vegetation is represented by one endmember, the potential over-/under-estimation is not considered a major source of error as it is partially balanced by the inclusion of a zero shade endmember and subsequent normalization (Adams et al., 1986, compare Section 4.3).

This study focused on the grazed rangeland areas, and agricultural areas were masked based on the available vector information (Section 3), which reflects the situation in 2000/2001. Where land-use conversions took place, this might affect the time series analysis, which was only targeted at changes within the natural and semi-natural rangelands. Given that the surface of arable land has increased during the observation period, this would only result in the exclusion of areas that had been used as grazing land at the beginning of the observation period. On the other hand, non-grazing areas would not be included, such that the potential error can be considered negligible.

The classes employed for analyzing the influence of terrain on observed trends (Section 6.2) were derived from a digital elevation model, and hence reflect the quality of the DEM. In addition, the analysis of topography features is scale-dependent, where coarser scales may camouflage patterns only visible at finer scales. On the other hand, the analysis strategy pursued here necessitates resolutions of the different input data sets to match, such that this limitation needed to be accepted. The 30 m grid resolution was shown to be an adequate analysis resolution for local-scale studies (Del Barrio et al., 1996). In particular, the regionalization procedure adopted here is based on relative differences between pixels rather than on their absolute topographic values. The resulting arrangement of topographic classes is therefore robust with respect to the local accuracy of the DEM.

6. Interpretation and discussion

Given that livestock grazing is the major land use in the rangeland areas of Lagadas County, the question on the influence of socio-economic factors will be addressed by linking available information on grazing animals and grazing system to spatial information derived from the time series analysis of satellite imagery as described in Section 5.3.

6.1. Stocking rates

The hypothesis of a general dependency of vegetation cover on livestock stocking rates was analyzed by linking average vegetation cover to animal numbers given per community and for three dates (where the 1984 image was assumed to adequately reflect the situation corresponding to the 1981 animal census). In order to represent effective grazing area, agricultural and built-up areas were excluded from the calculations. Goat and sheep units were equally

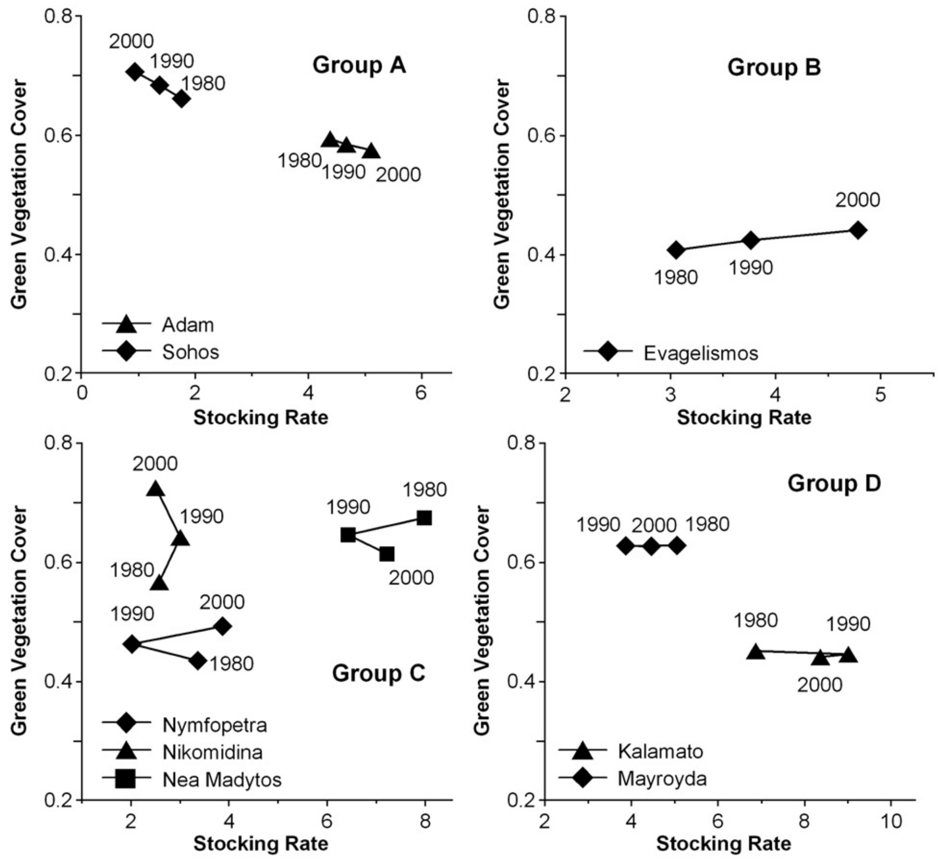

Fig. 5. Temporal trajectories relating vegetation cover development to stocking rate development (vegetation cover calculated for actual grazing areas based on the linear trend function; stocking rates relate to effective grazing area). The four plots represent major spatio-temporal pattern groups, with the chosen representative communities being indicated at the bottom.

treated and resulted in animal stocking rates [animal/ha] for 45 communities. The linear trend function was employed to derive vegetation cover estimates matching the animal census dates. This ensures them being free of a potential bias introduced by exceptional climate conditions prior to image acquisition for the respective dates; then, all pixel values were aggregated to the community level.

This procedure resulted in three value pairs of stocking rate and average vegetation cover for all communities in the county, which were illustrated in the form of temporal trajectories (Fig. 5). Given the basic assumption of a direct relation between the development of both factors, this representation should result in a direct negative relation, i.e. decreasing cover estimates with increasing animal stocking rates, and vice versa (Hostert et al., 2003b).

The resulting 45 trajectories did not conform to this pattern and can be grouped into four major categories, which are exemplarily shown in Fig. 5. Group A represents communities with a continuous negative relation between stocking rate and vegetation cover (Sohos, Adam). Group B shows an adverse relation, where increasing stocking rates coincide with increasing vegetation cover (Evagelismos), while the complementary effect is not observed. A third major group of trajectories (group C) is characterized by a significant change in the direction of the relation (Nymfopetra, Nea Madytos, Nikomidino), indicating a transition from type A to type B or vice versa. Finally, group D represents communities where vegetation cover seems to be unaffected by stocking rates (Mayroyda, Kalamato).

Different factors may contribute to this apparently non-significant relationship between stocking rates and vegetation cover development. Depending on whether stocking rates are above or below a level sustained by natural resources (Tsiouvaras et al., 1998), further changes may effectuate adverse consequences, i.e. increases or decreases in vegetation cover. In addition, areas that are characterized by a high vegetation cover level may behave resilient to slight increases in stocking rates; similarly, already degraded regions may not easily recover even if stocking rates decline. This is particularly severe where only little grazing areas exist and rotating systems cannot be installed. Another important factor is the noted intensification of production systems, with animals being kept in sheds and sustained by feedstuffs, while grazing is increasingly confined to specific times and locations. Together with the improved road network facilitating feedstuff transportation, this eventually leads to 'artificial' livestock systems where stocking rates may exceed natural resources (Hadjigeorgiou et al., 1998).

The hypothesis of a consistent relation between stocking rates and grazing impact indicated by vegetation cover can hence not be confirmed at the community level. Rather, the coexistence of physical and socio-economic factors driving the grazing system and its manifestations seems to request finer observation scales (Oba et al., 2000).

6.2. Effects of topography

The relation between socio-economic systems and landscape patterns (Section 5.3) was further interpreted at the pixel level. With the presumed strong influence of the specific local grazing regimes (Sections 2.2 and 6.1), the underlying hypothesis was that trends in vegetation cover are determined by the spatial distribution of grazing animals. This does not follow a stochastic distribution but is mainly driven by shepherd decisions. Given the present sedentary system, these are assumed to reflect to a large degree the accessibility of grazing areas.

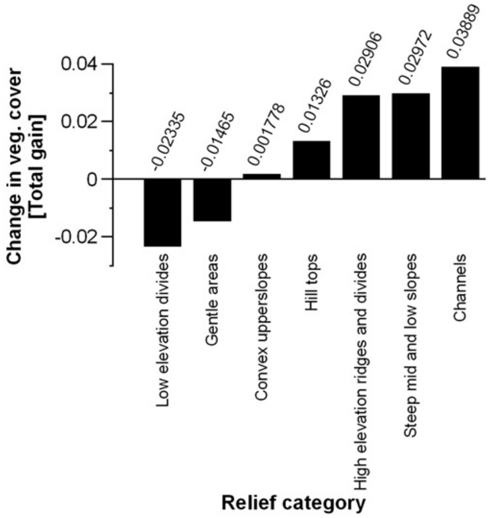

Fig. 6. Overall development of vegetation cover within major relief classes. Values were calculated by averaging gain factors from the linear trend analysis for the topographic regionalization classes; agricultural areas were masked out based on the habitat vector information, very small classes were aggregated to ensure sufficient number of samples per class (further information found in the text).

In this case, regions that are more easily accessible are assumed to show a stronger indication of degradation than those that are avoided by shepherds. The regionalization data set derived from the digital elevation model was employed to validate this. In order to ensure enough pixels within each class for further analysis, some small classes were aggregated, eventually yielding the following terrain zones: low elevation divides, gentle areas, convex upperslopes, high elevation ridges and divides, steep mid and low slopes, and channels. For each class, gain estimates from the trend analysis were averaged, as shown in Fig. 6.

With respect to the preliminary assumption, a likely coincidence between average cover development and terrain classes can be noted. Areas where shepherds move at ease show negative to neutral average trend gains ('low elevation divides', 'gentle areas,' 'convex upper-slopes' and 'high elevation ridges and divides'). Here, the 'low elevation divides' class encompasses the lower reaches of the Lagadas rangelands, which are easily accessed and hence frequently grazed. To the contrary, those classes which are more remote or where landscape features complicate movement, i.e. the 'hill tops', 'high elevation ridges and divides' and 'steep mid and low slopes' and 'channels', show positive overall trends. This corresponds to the pattern observed in Fig. 4 and supports the presumed relation. As the large number of pixels averaged for each class effectuates considerable variation, an ANOVA was calculated. For the resulting F-distribution ($F_{6,\ 673072}$) the null-hypothesis was rejected with α=0.05, indicating that average values for the terrain classes significantly differ and hence there is an influence of these classes on the gain values of the trend function. To further assess whether this difference applies only for some mutual class-pairs or for all classes, a post-hoc Scheffé test (Bahrenberg et al., 1992) was calculated, which confirmed the independence between all two-pair combinations of terrain classes.

It can hence be stated that through its influence on shepherd movement, topography exerts a significant influence on vegetation cover development. This underlines the dominance of socio-economic over physical factors in the case of the sedentary grazing regime in Lagadas County.

6.3. The regional context

Referring back to the overall questions addressed through different analysis steps, the pattern of rangeland use, in this case referring to the prevalent grazing regime, was shown to be reflected in the pattern of temporal trends of vegetation cover. On the other hand, a direct quantitative relation between animal stocking rates and the development of vegetation cover could not be confirmed due to limitations in possible aggregation levels and external factors.

Given the fine-textured character of Mediterranean landscapes, the driving factors of livestock grazing regimes as well as their consequences need to be comprehended in a local context, as this is often

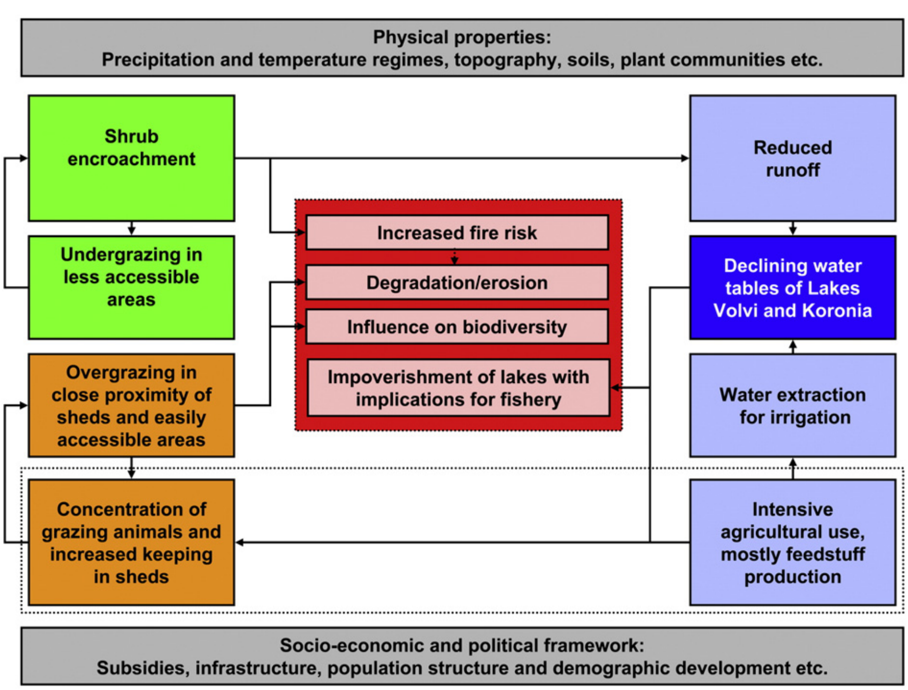

Fig. 7. Schematic representation of major determinants of the landscape processes responsible for the spatio-temporal trends derived in this study. White boxes and arrows indicate processes and potential feedback loops, the grey shaded box summarizes major ecological consequences of this process-feedback framework.

characterized by a complex configuration of causes, consequences and feedback loops (Hill et al., 2004; Mulligan et al., 2004; Rindfuss et al., 2004a; Stafford-Smith & Reynolds, 2002). Fig. 7 integrates results from this and other studies as well as local environmental knowledge (LEK, Reynolds et al., 2007) to sketch the present consensus on the overall process regime operating in Lagadas County.

In the case of Lagadas County, the socio-economic framework is mainly given by local demographic developments and the scheme of European subsidies (e.g. European Agricultural Guidance and Development Fund, European Rural Development Fund, European Social Cohesion Fund). This combination is largely responsible for the grazing scheme where the strong concentration of animals leads to both, an over-utilization close to sheds causing degradation and soil erosion, and undergrazing in more remote locations, causing shrub encroachment and increasing the risk of wildfires (Röder et al., 2007). With respect to biodiversity, field surveys of faunal diversity (Papoulia et al., 2003) presented highest total numbers as well as diversity of species in open, medium dense shrublands with a high number of grass–shrub interfaces, while both values declined towards dense shrublands and grasslands. Hence, both over- and undergrazing are expected to negatively affect biodiversity. A second important aspect is the influence of developments in vegetation cover on the hydrological regime, since with increasing shrub cover runoff of water to the lakes is declining. In particular, higher agricultural production is necessary to provide the feedstuffs required by the present grazing scheme. This has caused an expansion of agricultural areas and especially the increased water uptake for intensive irrigation has contributed to a significant reduction in the surface of the lakes (Stellmes et al., 2007). As a further aspect, modified hydrological properties and the inflow of pesticides and fertilizers are expected to affect fish stocks.

7. Conclusions and perspectives

For a test area located in Lagadas County (Northern Greece), this study analyzed temporal trends in proportional vegetation cover derived from a time series of Landsat TM/ETM+ data. It was investigated whether coexistence of intensification and extensification of grazing is reflected in the pattern of temporal trends, and whether a relation exists between the development of stocking rates and the development of vegetation cover. Integrating linear trends and terrain types, it was shown that negative to neutral trends prevail in low and plain areas, while positive trends dominate in steep areas. Given the highly concentrated grazing scheme, topography is a proxy for the accessibility of regions, as during the relatively little time actually spent outside of the sheds, shepherds prefer locations that are close to sheds and/or easily accessible. These findings are congruent with spatial trends derived using a grazing gradient approach (Röder et al., 2007).

On the other hand, a consistent relation between development of animal stocking rates and vegetation cover could not be established. This is due to specific characteristics of the grazing system, such as feedstuff provision, as well as to the community aggregation level corresponding to the available animal statistics, which fails to represent the variance encountered at the pixel level. As a typical example of a coupled human-environment system, this makes a strong case for the collection and integration of finer-scale socio-economic data, such as household surveys that link livestock to individual grazing plots (Lambin & Geist, 2006; Mertens et al., 2000; Rindfuss et al., 2004b). Such an analysis might well be expected to add to or modify the process framework depicted in Fig. 7.

On the methodological–technological level, it was shown that remote sensing data may serve as an essential component of landscape-level monitoring observatories if the temporal dynamics of suitable indicators are interpreted under consideration of the local process framework. In this context, important research opportunities arise from the comparison of linear trends shown in this study, with trends derived at coarser scales using hypertemporal data, as the small-structured rangelands of the European Mediterranean are an excellent test case to assess the influence of temporal and spatial scales on the patterns that may be detected.

With respect to management options of the extensive rangeland resources, this study underlines that the present, semi-intensive grazing system is partially unsustainable and sparks natural resource degradation and threats to ecosystems in many locations. The 2003 reform of the European CAP (Common Agricultural Policy, Regulation (EC) No 1782/2003, No 795/2004, No 796/2004 and subsequent documents) identifies sustainable resource utilization as a major objective, and intends to decouple the payment of subsidies and production numbers. Rather, incentives are foreseen for complying with 'good agricultural practice' and high environmental standards, and the implementation of this new system is currently underway. This study may hence serve as point of reference for a future extension of the data base to investigate whether positive effects arise from this major change in the socio-economic framework.

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