

Stability analysis for defining management strategies in abandoned mountain landscapes of the Mediterranean basin

Raffaele Pelorosso^{a,*}, Stefano Della Chiesa^{b,c}, Ulrike Tappeiner^{b,c}, Antonio Leone^a, Duccio Rocchini^d

^a Tuscia University, DAFNE Department - Environmental Engineering Group, via S. Camillo de Lellis snc 01100 Viterbo, Italy

^b Institute for Alpine Environment, EURAC, I-39100 Bozen/Bolzano, Italy

^c Institute of Ecology Leopold-Franzens-Universität Innsbruck Sternwartestr. 15 A-6020 Innsbruck, Austria

^d Biodiversity and Molecular Ecology Department, IASMA Research and Innovation Centre, Fondazione Edmund Mach, S. Michele all'Adige (TN), Italy

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ABSTRACT

The semi-natural landscapes of Mediterranean mountains underwent a remarkable land abandonment in the past decades. These large perturbation-dependent landscapes then evolved into new meta-stable states to balance human pressures and natural components with a general pattern homogenisation and several consequences on landscape services. These areas need effective management strategies to conserve a wide functionality allowing, at the same time, the sustainable development of population. Lack of resources and achievable restoration goals often hamper these objectives to be reached.

In this paper, a study of pattern change is proposed using five landscape metrics and a stability analysis of features derived from land cover maps in order to investigate their magnitude and rate of change in a mountain municipality of central Italy between two separate time periods (1954–1985 and 1985–1999). A Kappa statistic (Kappa Index of Agreement), a Markov chain model and a Kruskal–Wallis test were employed. The results showed that shape and size of woodlands, open areas and buildings patches were significantly changed during the second period (1985–1999), with a concurrent abrupt reduction in the rate of change for each land cover, confirming that a new meta-stable state of equilibrium between human land use and natural processes of secondary succession was being approached.

A discussion of management strategies for such equilibrium is therefore proposed to contribute to the development of effective conservation actions for the semi-natural landscapes of Mediterranean Basin. The presented approach aims to stimulate the inclusion of stability analysis into the planning and management of abandoned landscapes.

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1. Introduction

The semi-natural landscapes of Mediterranean mountains underwent a remarkable land abandonment in the past decades (Pelorosso, Leone, & Boccia, 2009; Sitzia, Semenzato, & Trentanovi, 2010; Tasser, Walde, Teutsch, Nogger, & Tappeiner, 2007) with a general mosaic simplification and several consequences on both ecosystem and human wellbeing. Indeed, such landscapes, as meta-stable perturbation-dependent systems, need an appropriate level of human pressure to maintain their structure and functionality (Naveh, 1987; Naveh, 2009). Otherwise, according with their resilience, they can turn into alternative equilibrium states (Gil-Romera et al., 2010; Suding, Gross, & Houseman, 2004), leading to alterations in several landscape services (sensus Termorshuizen & Opdam, 2009), as biodiversity levels (Geri, Amici, & Rocchini,

2010), fire spreading resistance (Lasanta, Arnàez, Errea, Ortigosa, & Ruiz-Flano, 2009), aesthetic pleasure, forage availability, regulation of water system, cultural heritage, vital countryside and economy (Lindborg et al., 2008; Sharma, Chettri, & Oli, 2010).

To ensure long-term benefits to people and ecosystems in these mountain areas as well as those downstream, a multi-scale, trans-disciplinary and holistic restoration and conservation approach to landscape functionality should be pursued (Naveh, 2005; Sharma et al., 2010). However, the recovery of all the landscape components and services is an uphill task (Moreira, Queiroz, & Aronson, 2006) and a shared trans-regional management strategy of such territories is not commonly accomplished. As a consequence, major attention has been focusing on those study/protected areas (e.g. Marignani, Rocchini, Torri, Chiarucci, & Maccherini, 2008), where environmental issues and public interest stimulated the development of restoration actions, taking into account a functionality concept strictly linked to land cover pattern and vegetation structure due to their relatively easy estimation by remote sensing (Koniak, Noy-Meir, & Perevolotsky, 2011; Verburg, van de Steeg, Veldkamp, & Willemen, 2009).

* Corresponding author. Tel.: +39 0761 357359; fax: +39 0761 357250.

E-mail addresses: pelorosso@unitus.it (R. Pelorosso), stefano.dellachiesa@eurac.edu (S. Della Chiesa), Ulrike.Tappeiner@eurac.edu (U. Tappeiner), leone@unitus.it (A. Leone), ducciorocchini@gmail.com (D. Rocchini).

The scrub encroachment and the forest recovery consequent to land abandonment, pointed out a general landscape homogenisation with loss of cultivated and grazed patches as well as the disappearance of isolated trees and hedges (Geri et al., 2010; Marignani et al., 2008; Rocchini, Perry, Salerno, Maccherini, & Chiarucci, 2006). Several restoration actions, recurring to scrub clearing and/or controlled grazing or burning, were usually proposed to retrieve the lost landscape functionality and biodiversity, recovering the vegetation pattern at suitable pre-abandonment reference states on the basis of in-depth knowledge of habitats, ecological processes and land use history (e.g. Bestelmeyer et al., 2009; Lasanta et al., 2009; Lindborg et al., 2008; Marignani et al., 2008). Furthermore, the ecosystem resilience concept and the definition of ecological thresholds among alternative states are raising of interest into the ecological restoration and environmental management plans (Briske, Bestelmeyer, Stringham, & Shaver, 2008; Hobbs, Higgs, & Harris, 2009; Miller, Belote, Bowker, & Garman, 2011; Swift & Hannon, 2010). Despite all the efforts made in the recent years, a clear definition of effective and achievable restoration goals which are ecologically sound, economically feasible and socially acceptable often lacks (Choi et al., 2008; Corsair, Bassman Ruch, Zheng, Hobbs, & Koonce, 2009; Hobbs, 2004) above all for the large abandoned landscapes of Mediterranean mountains. Indeed, the previous mentioned approaches, need substantial and conspicuous funds not only to correctly identify the reference landscape and the sites for restoration, but also for the long-term monitoring and the maintenance of a stable vegetation pattern, since the advancement of the spontaneous biological processes (successions). In such large abandoned landscapes, data and financial support for their functional conservation and restoration are, on the contrary, often scarce and sometimes insufficient. Thus, while these types of restoration projects could be destined to fail, significant ecosystem degradation could be still untreated. On the other side, important plans for the conservation of essential landscape function and services for human health (e.g. flood protection, fire spreading resistance) and the maintenance of cultural heritage (e.g. pastoral activity) could not be carried out due to incorrect funds assignment. A more cost effective approach to landscape management seems therefore to be necessary in order to overcome the actual impasse in environmental decision-making of such territories.

Restoration efforts do not usually aim at proving if the new state of a system is in a stable equilibrium (Suding et al., 2004). For abandoned lands, it would be interesting to understand if the landscape will move towards further pattern simplification with potentially negative consequences on landscape services, or towards a stable state in which sufficient levels of functionality are still realised. Indeed, abandoned landscapes can host novel ecosystems relatively stable that continue to provide similar or enhanced flows of services (such as habitat provision, CO₂ sink capacity, timber production and flood attenuation) (Hobbs et al., 2009). The analysis of landscape stability could therefore allow a better definition of restoration strategies and goals, and assess the application of a more conservative approach to reduce or prevent abiotic and biotic change instead to reverse it (Hobbs et al., 2009). An example of conservation strategies for abandoned landscapes was proposed by Pardini, Mosquera, and Rigueiro (2002) for a private farm of central Italy. The authors suggested a minimal farm management coupling a mechanical control of vegetation and a rotational grazing of pastures, woodlands and firebreaks to reduce the fuel availability and maintain a “beauty” landscape accessible by tourists. Such form of management could allow for a farm incomes diversification through the development of a naturalistic tourism even reducing the land abandonment process and fire spreading risks. Acknowledging that, after land abandonment, a stable or meta-stable state and the related flow of landscape services can be achieved,

conservation strategies requiring a minimal management by local farms and minor economic resources for their long-term maintenance as that proposed by Pardini et al. (2002) could be therefore employed for the preservation of the current landscape functionality and pattern; finally, the saved funds could be redirected to other initiatives for the balance between biophysical and cultural system components and the sustainable development of mountain populations.

Several methods, such as landscape metrics (Turner, Gardner, & O’Neil, 2001) and Markov chain models (Baker, 1989), are currently used to describe and to model the dynamics of landscape patterns and to correlate it with the processes driving the change. Although a wide range of variables (features) can be analysed to describe landscape equilibrium states (e.g. biomass, age and altitude of vegetation, seral stages, spatial pattern), their estimation over large areas and for multiple dates is often constrained by the limited available resources. Remote sensing images and derived land cover maps often represent the unique suitable data source and, therefore, their employment for a landscape pattern stability analysis over large scale is an attractive challenge.

However, multi-temporal analyses need to concern about different spatial resolution of the maps: classification errors and spatial inconsistencies among the compared maps can significantly distort the analysis of landscape pattern change and the appraisal of landscape dynamics. Several techniques have been proposed to improve the comparison between vectorial maps (Linke et al., 2009; Mas, 2005), while, dealing with raster data or post-classification change detection, resampling methods were usually employed to define a common Minimum Mapping Unit (MMU) between the maps (Pelorosso et al., 2009; Rocchini et al., 2006). However, resampling methods can introduce spatial errors and, to overcome them, Rocchini et al. (2006) proposed an alternative map interpretation method based on the visual classification of vector format grids with a pre-defined cell size. While time consuming, this method allows rigorous multitemporal analysis based on a pre-defined map resolution. However, such approach (i) has not been applied on larger areas and (ii) tested in terms of classification accuracy.

In this paper, a stability analysis of features derived from land cover maps integrated with landscape metrics analysis was pointed out to show the pattern dynamics and to prove the approaching to eventual stable states after land abandonment, where an equilibrium between human pressure and natural processes could be reached. In particular, three land cover maps (referred to years 1954, 1985 and 1999) were derived from aerial photos using the classification method proposed by Rocchini et al. (2006). *K* statistic, Markov chains and Kruskal–Wallis test were applied to assess the stationarity (constancy) in land cover typologies proportion, change rate, size and shape of patches. A mountain municipality of central Apennines (Italy), undergoing evident abandonment processes, was selected as a study case.

Since stability and equilibrium concepts are still debated in landscape planning, a brief resume of assumptions, methods and definitions, with particular reference to landscape pattern analysis, is reported in the following paragraph.

2. Background on landscape stability

The concepts of stability and equilibrium and their assessment methods are still largely discussed both in natural and semi-natural landscapes (Perry, 2002; Turner, Romme, Gardner, O’Neill, & Kratz, 1993; Wang & Finley, 2011). Landscapes are open systems controlled by both extrinsic and intrinsic factors with stochastic events (disturbances) and, since spatial heterogeneity influences ecological processes on multiple scales, the analysis of stability and equilibrium status requires the definition of a specific spatial

and temporal scale (Perry, 2002). Moreover, under the traditional ecological point of view, the stability of a state can be evaluated by three different stability properties as proposed by Grimm and Wissel (1997): *constancy* (staying essentially unchanged, e.g. in numbers, densities, relative proportions), *persistence* (persistence through time of an ecological system, e.g. non-extinction), and *resilience* (the return to a reference state after a temporary disturbance).

Several methods and tools have been developed in the last decades to assess the spatial and temporal landscape pattern dynamic. In landscape ecology, many studies focused on landscape metrics (e.g. Narumalani, Mishra, & Rothwell, 2004; Romero-Calcerrada & Perry, 2004; Rocchini et al., 2006) because they allow for an objective analysis of the landscape pattern change comparable with other study cases (Turner et al., 2001) and the recognition of the ecological processes that may have generated it (Rocchini et al., 2006). However, beside their meaningfulness in describing pattern dynamics, to our knowledge, previous studies have not explicitly tested issues related to the potential stability over time of such metrics. Instead, based on Andrey Markov's theory on stochastic processes and random variation (see Meyn & Tweedie, 1993), Markov chain models have been frequently used for modelling land patch dynamics and for testing the effect of disturbances (e.g. wildfire, human actions) on landscape evolution and its stationarity (i.e. the *constancy* sensus Grimm & Wissel, 1997) (Li, 1995; Yemshanov & Perera, 2002). In particular, for human-modified landscapes, Markov chain applications have been generally carried out to understand land cover dynamics assessing the stationarity of the transition rate over time (e.g. Petit, Scudder, & Lambin, 2001; Romero-Calcerrada & Perry, 2004; Weng, 2002). Since land use and cover change reflects dynamics and interplay of economic, social and biophysical factors over time, the stationarity in land cover data is unlikely (Weng, 2002) above all in heterogeneous areas (Pueyo & Bagueria, 2007; Romero-Calcerrada & Perry, 2004). However, even simple Markov models and projections are able to give useful indications for the environmental management when used to provide answers to "what if" scenarios hypotheses (Baker, 1989). Future projections can be employed as an early warning system for future land cover changes effects (Verburg, 2006), comparing the consequences of different management strategies or policies on the evolution of a landscape towards a steady state.

3. Study area

The study area (Fig. 1) was the municipality of Micigliano in Rieti Province, Italy (coordinates of the area centre X, Y: 339496, 4701808; projection Universal Transverse of Mercator (UTM) 33N European Datum 1950). The area is 3619 ha and is located between the Terminillo Mountain and the Velino River. The elevation ranges between 490 m and 2200 m above sea level, and the average slope is 28 degrees with a maximum of 68 degrees. The geology is mainly represented by extensive outcrops of carbonate platform lithotypes, arenaceous pelitic turbidites and hemipelagic marly-clayey deposits. Limestone is the main soil type (Martini, Moscatelli, & Scarascia Mugnozza, 2004).

Micigliano has 140 inhabitants and a typical agriculture based economy, strongly characterized by livestock (mainly sheep) and silviculture. A marked decrease in Usable Agricultural Surface (SAU) was accompanied by one of the highest rates of emigration (−76.8% between 1951 and 2001) and lowest population density (3.7 inhab/km²) in Lazio Region (ISTAT, 2001), revealing it as an area very susceptible to land abandonment.

The climate belongs to the temperate region with mesotemperate–supratemperate thermotype and humid–hyperhumid ombrotpe (Blasi, 1994). Mean annual temperature is between

6 °C and 13 °C and the mean annual precipitation ranges from 1200 mm to 1600 mm (average between the five closest meteorological stations). The landscape is dominated by pastures (97.6% of SAU). Orchards are practically nonexistent. Woods are the predominant coverage (ISTAT, 2000). To date, the vegetation found in the area comprises: open formation (e.g. high altitude grasslands dominated by *Sesleria tenuifolia*, *Carex kitaibeliana* and *Plantago atrata*) and small cultivated fields; shrublands, (*Juniperus nana*, *Vaccinium myrtillus* and *Arctostaphylos uva-ursi* above 1000 m, *Cytisus sp.*, *Rubus sp.*, under 1000 m); mixed broad-leaf woodlands between 500 and 1000 m, dominated by *Quercus pubescens*, *Cytisus sessilifolius* and *Brachypodium rupestre*; broad-leaf beech woodland above 1000 m, dominated by *Fagus sylvatica*, *Taxus baccata*, *Ilex aquifolium*; riparian woody vegetation mainly represented by *Salix eleagnos* and *S. purpurea*. Exotic conifers such as *Pinus Nigra* ssp. were planted in the 1950s.

4. Methods

To investigate the vegetation dynamics and the landscape pattern stability, a framework of three sequential phases was proposed as follows: first, the processing of aerial photos and the building of homogeneous land cover maps; second, the analysis of land cover change and landscape pattern dynamics by using some selected metrics; third, the stability analysis of the landscape pattern in terms of land cover typologies proportion, change rate, size and shape of the patches.

4.1. Data processing and map building

Panchromatic nadir aerial photos (grey scale) taken in 1954 and 1985 were scanned at 600 dpi. In order to achieve an accurate co-registration with recent aerial photographs from 1999, orthorectification was based on a Digital Elevation Model (DEM) with 10 m × 10 m cell size and 20 Ground Control Point (GCPs) for each photograph. The software used was ERDAS IMAGE 8.6 (Leica Geosystems). Each image's spatial resolution was 2 m. Positional accuracy was assured by means of 20 additional GCPs using the 1999 orthophotographs as reference; positional error (Ground X e Y RMSE) was always below 4 m. Images were projected into the Universe Transverse Mercator projection and European Datum 1950. The Minimum Mapping Unit was defined a priori overlaying a 10 m × 10 m grid on the study area (see Rocchini et al., 2006). Using ArcMap 9.2 (ESRI), each cell of the grid was subjected to visual photointerpretation with the support of ancillary information such as DEM, topographic maps, and others orthophotos of the same area (Pelorosso, 2008; Pelorosso, Della Chiesa, & Boccia, 2007). The land cover classes used were: woodlands, scrublands, isolated trees, hedges, open areas (bare rocks, pastures and fields), buildings, roads and coniferous plantations. If a cell contained more than one class, the value of the prevalent class, in terms of area, was used. This classification procedure was performed for each of the three periods (1954, 1985, 1999) by only one interpreter hence avoiding observer bias among different operators. A site survey was carried out in 2007 to facilitate the recognition of the most recent land cover map. To test the accuracy of the proposed classification method, a *K* statistic was applied on the 1999 map using as reference a random stratified sample. A minimum of 50 samples per category, increasing up to 100 samples for the larger category, were employed (Congalton, 1991). For each sample, the land cover patches were accurately delineated to obtain the maximum precision in the classification. The final file was converted into a raster having the same resolution and extension of land cover maps. The *K* statistic was then performed to estimate the accuracy of a careful digitization of polygons vs. the proposed classification of cells.

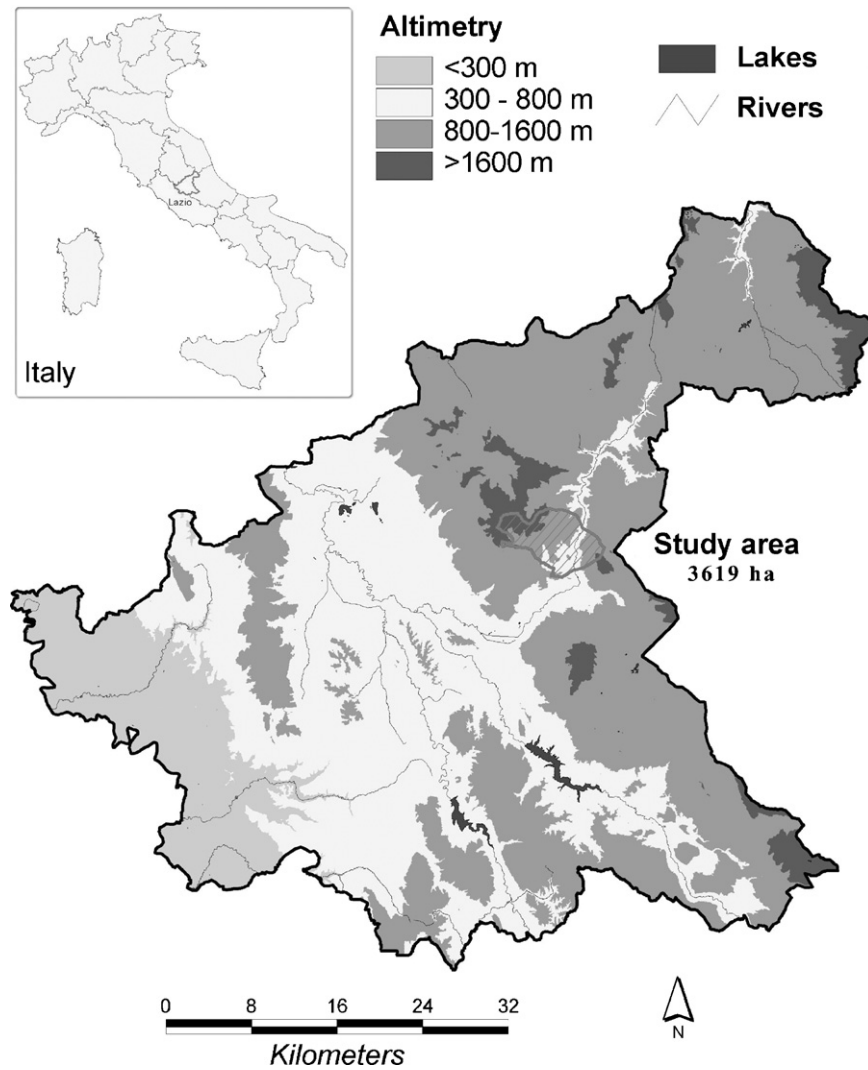


Fig. 1. Location of the study area.

4.2. Landscape analysis

Detection of land cover changes between pairs of the periods under analysis was performed through cross matrices realised with the *crostab* tool of the GIS software IDRISI 32 (Eastman, 2001). Fragstats 3.3 (McGarigal, Cushman, Neel, & Ene, 2002) was used for the landscape pattern metrics analysis.

Landscape structure was estimated at class level by means of some patch-, shape- and size-based metrics briefly described here. The number of patches per class (NP) is a patch-based metric, which is a simple measure of the extent of subdivision or fragmentation of each class into patches. Area Weighted Mean Shape Index (AWMSI) is a shape-based metric that measures the complexity of patch shape weighted by the patch size, as:

$$AWMSI = \sum_{p=1}^N \left(\frac{0.25P_p}{\sqrt{A_p}} \right) \left(\frac{A_p}{\sum_{p=1}^N A_p} \right) \quad (1)$$

where N = number of patches, P_p = perimeter of the patch p , A_p = area of the patch p .

Eq. (1) refers to the raster based calculation. We refer to Imre and Rocchini (2009) for a mathematical description of shape indices. AWMSI equals 1 when the patch is compact (i.e. a square for raster

files or a circle for vector files) and increases without limit as the patch shape becomes more irregular.

The following size-based metrics were used: Area Weighted Mean Patch Size (AWMPS) and Largest Patch Index (LPI). AWMPS is a measure of mean patches size with a lower weight given to the small patches in the landscape, as:

$$AWMPS = \sum_{p=1}^N A_p \left(\frac{A_p}{\sum_{p=1}^N A_p} \right) \quad (2)$$

where N = number of patches, A_p = area of the patch p .

LPI quantifies, at the class level, the percentage of the total landscape area comprised by the largest patch; LPI approaches 0 when the largest patch of the corresponding patch type is increasingly small, while LPI is equal to 100 when the entire landscape consists of a single patch.

Moreover, to investigate the isolation and fragmentation of a patch, a Mean Proximity Index (MPI) was used. This index estimates the isolation of patches for a certain class within a given search radius (in this study it was set as 100 m); a decrease of MPI is related to an increase of isolation and vice versa. Details for these metrics can be found on McGarigal et al. (2002) and Turner et al. (2001). For all these metrics an 8-cell rule was used to aggregate patches, i.e. diagonals were considered in the calculation. Finally, ranked patch

frequency-area distributions were plotted for each of the three time periods in order to highlight some aspects of the change in the dimension of the patches at the landscape scale.

4.3. Stability analysis

To evaluate the stability of change rate and to identify the probable future steady state, Kappa statistic and first order Markov chain models were performed.

The Kappa (K) statistic (also referred to as Kappa Index of Agreement, KIA) was used to analyse the stability of the different vegetation classes and their potential change (Rocchini et al., 2006; Romero-Calcerrada & Perry, 2004). This index allows to evaluate the agreement degree of a land cover category between two dates. The per-category K index was calculated through the following formula (Rosenfield & Fitzpatrick-Lins, 1986):

$$K = \frac{(P_{ij} - (P_i P_j))}{P_i - (P_i P_j)} \quad (3)$$

where P_{ij} = proportion of the entire image in which the i category agrees for both dates, P_i = proportion of the entire image in the i class for the reference image, P_j = proportion of the entire image in the i class for the non-reference image.

Its values range from -1 to $+1$. If the two input images are in perfect agreement (no change has occurred), K equals 1. If the two images are completely different, K equals -1 . If the change between the two dates occurred by chance, then Kappa equals 0. The *crosstab* tool of the GIS software IDRISI 32 (Eastman, 2001) was employed to analyse the K statistic.

First order Markov chains are based on a transition matrix (A), describing the probability of each cell to change from a state i to state j (for each class) in a discrete time step, and a vector (x_t) representing the area (abundance) of the considered class at time t . Landscape structure at time $t + 1$, described by the vector x_{t+1} , is obtained through the multiplication of A by x_t . A steady state, corresponding to stable class distribution, is reached after a sufficient number of iterations. Thus:

$$x_{t+1} = x_t * A \quad (4)$$

$$x_{t+n} = x_t * A^n \quad (5)$$

Yearly transition probability matrices were obtained by dividing each element in the matrix by the number of years separating the images; A_{ij} elements were then corrected such that each row summed to one (Romero-Calcerrada & Perry, 2004; Urban & Wallin, 2002). The Anderson-Goodman test (Pueyo & Bagueria, 2007) was used to verify the non-stationarity of transition probabilities, i.e. the inconstancy in land cover transition rate between the three dates, while percentage differences between the yearly transition probability of 1954–1985 and 1985–1999 were carried out to highlight coupled transition changes between each land cover; this matrix comparison was used to assess the conversion magnitude among land cover classes and to give further information on landscape stability. Finally, from each yearly transition probability matrix, future scenarios of land covers distribution until a converging steady state were developed. The difference in terms of Euclidean distances between such projections and the observed 1999 land cover distribution were therefore calculated.

The stability analysis regarding spatial pattern was carried out to investigate the potential significance of the difference in patches shape and size. Hence, besides plotting AWMSI and AWMPs per each class, a non-parametric Kruskal-Wallis test (see Zar, 1996) was applied within the R software (R Development Core Team, 2007) to test for differences in shape and size values. In particular, the null hypothesis (H_0) to be tested is that the mean ranks of

shape and size values are the same in each group (class); it is tested by a rank sum test.

A summarizing table of the landscape metrics and statistical analysis techniques employed was reported in Appendix A.

5. Results

5.1. General landscape changes

Fig. 2 shows the three maps produced after the orthophotos classification. They point out that evident changes occurred over the last 50 years in the study area. The 1999 land cover map reports overall Kappa of 0.95 using as reference the samples randomly individuated.

In 1954 “woodlands” was the most abundant and dominant class followed by open formations. Open areas were widely scattered in the landscape. In 1954–1985 time-span, a considerable decrease of open areas took place with an increase in the woodlands. This resulted in a defragmentation of woodlands in which all the small patches belonging to the open formation were dispersed; a clear increase in homogeneity in terms of spatial structure was recognisable in the landscape mosaic.

Fig. 3a displays the extension of each class expressed in percentage of the study area for the three periods. The total area of woodlands increased from 52.5% in 1954 to 63.1% in 1985 and to 63.5% in 1999, with a relative increase in area of 20.2% from 1954 to 1999.

Conifers, artificially planted after 1954, did not considerably vary their coverage between 1985 and 1999, with a territorial occupation of 2% in 1999. Scrublands covered 3.1% in 1954, 2.5% in 1985 and 2.6% in 1999 of the study area showing a relative decrease between 1954 and 1999 of -15.8% . A severe decline of open areas took place between 1954 and 1999, with a total cover loss of 29.9%. Isolated trees and hedges occupied a small part of the landscape under study. Nonetheless, they showed a strong reduction from 1954 and 1999 of 46.2% and 68.9%, respectively. While the urbanisation of the study area was reduced, buildings and roads were considerably expanded between 1954 and 1999, $+106.5\%$ and $+153.8\%$, respectively.

Table 1a and b show the transition matrices of land cover change. Each cell of the cross-matrices represents the hectares of the previous land cover class that has changed into another land cover class. The diagonal cells (bold characters) represent the unchanged area (persistence) for each class.

In the first period (1954–1985), open areas were characterized by the highest persistence (93% of unchanged area in 1954), followed by woodlands (80%), buildings (46%) and roads (32%). A low persistence was found for scrublands (19%), isolated trees (10%) and hedges (5%). Conversely, open formations were mainly changed into woodlands, conifers and scrublands, while semi natural vegetation like scrubland formations, hedges and isolated trees were mostly transformed into woodlands (Table 1a).

In the period 1985–1999 only small changes occurred in the landscape (Table 1b, Fig. 4) denoting a higher stability (also due to the aforementioned smaller time span) and similar conversion trajectories with the first investigated period (see Table 1a). Indeed, the persistence was even higher than 60% and almost 100% for woodland, open areas, roads and coniferous plantations. The spatial distribution of changed land covers during the two periods is shown in Fig. 4.

5.2. Landscape metrics analysis

The total number of patches in the landscape decreased by 29.3% from 1954 to 1985 (from 3432 to 2425 patches) and

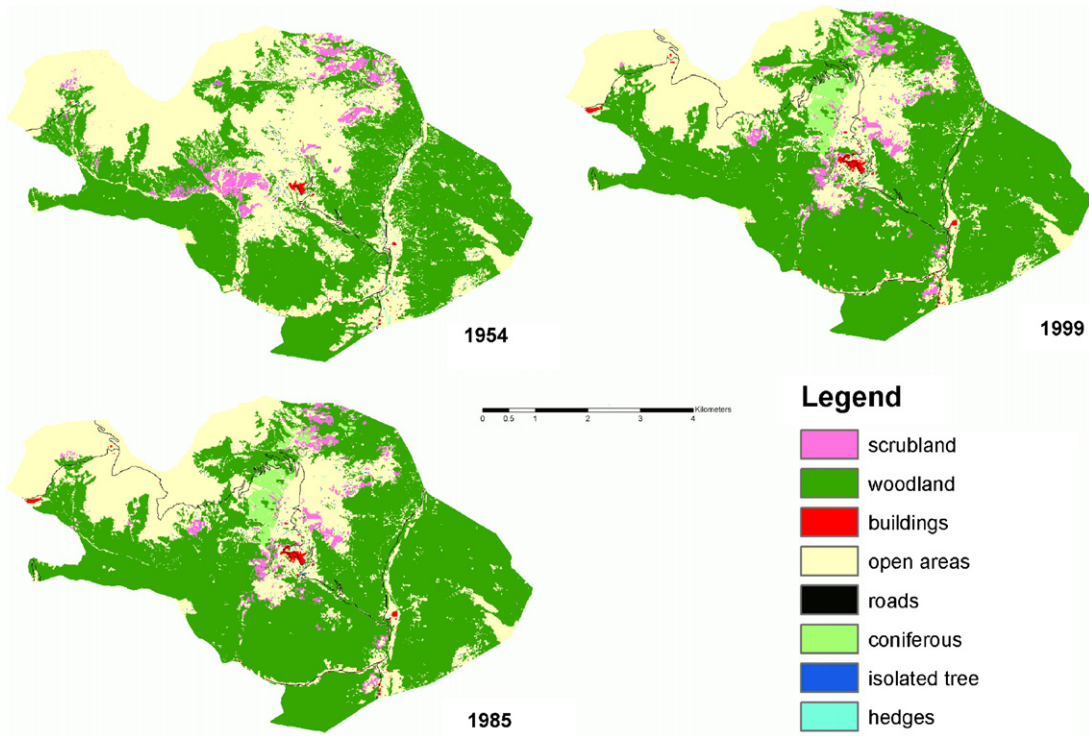


Fig. 2. Land cover maps derived from aerial photographs interpretation for the years 1954, 1985 and 1999.

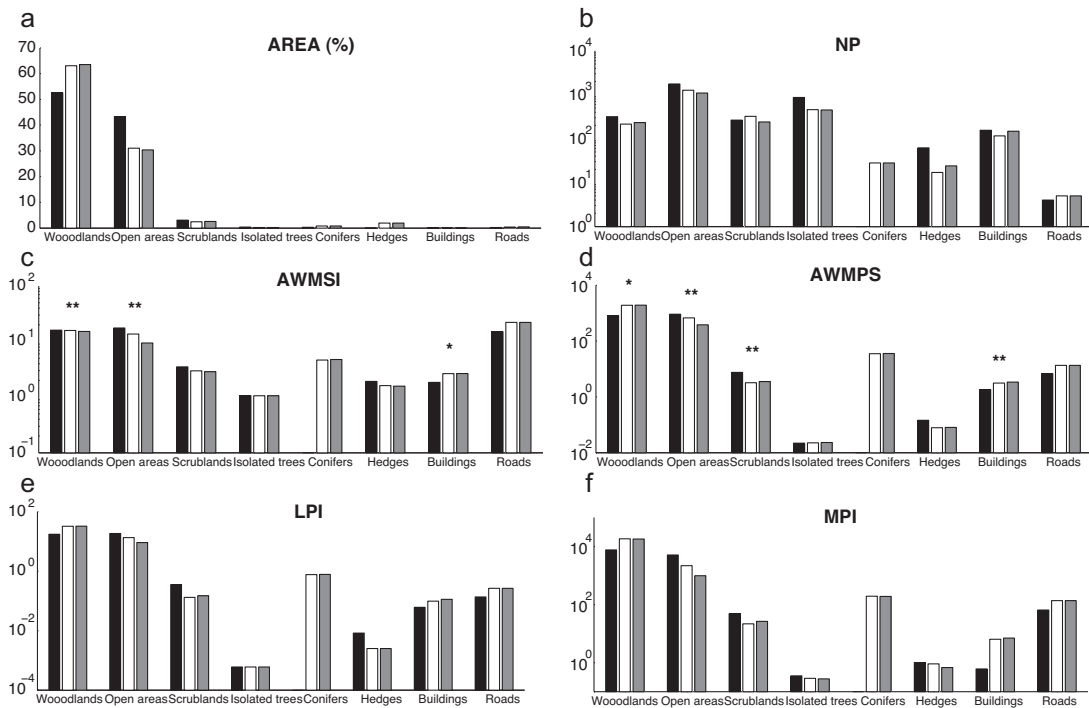


Fig. 3. Landscape metrics results. Black, white and grey columns represent, respectively, 1954, 1985 and 1999 years. Significant changes following the KW test in shape and area of patches are identified by markers, e.g. asterisks: * $p < 0.05$, ** $p < 0.01$. (a) Area occupied by each land cover class in the municipality of Micigliano (percentage). Notice the increase in woodlands and the consequent decrease in open formations. (b) Number of patches per class (NP). A patch is represented by an isolate or a contiguous group of cells belonging to the same class. (c) Area Weighted Mean Shape Index (AWMSI) per class. (d) Area Weighted Mean Patch Size (AWMPs) per class. (e) Largest Patch Index (LPI). The largest patch index at the class level quantifies the percentage of total landscape area comprised by the largest patch. Note the steep decrease in LPI for the open formations and the increase for the woodlands class. (f) Mean Proximity Index (MPI) per class. Notice the abrupt decrease in isolation for the woodlands and the increase in isolation for the open formations.

Table 1

Cross-matrices of the land cover classes (values in hectares). (a) Period 1954–1985. Columns represent land cover classes for the year 1985 while rows represent land cover classes for the year 1954. (b) Period 1985–1999. Columns represent land cover classes for the year 1999, records represent land cover classes for the year 1985.

(a) Time-span 1954–1985									
1954	1985								
	Woodlands	Open areas	Scrublands	Isolated trees	Conifers	Hedges	Buildings	Roads	Total 1954
Woodlands	1826.5	58.6	9.3	1.2	0.8	0.1	0.3	4.8	1901.6
Open areas	363.6	1044.8	61.7	5.5	68.9	0.8	6.9	15.3	1567.4
Scrublands	80.3	12.5	17.0	0.4	0.6	0.0	0.0	0.0	110.9
Isolated trees	8.0	4.3	0.8	0.8	0.7	0.0	0.1	0.1	14.8
Hedges	2.2	1.9	0.0	0.0	0.0	0.1	0.0	0.0	4.2
Buildings	0.9	0.8	0.0	0.0	0.0	0.0	6.6	0.2	8.5
Roads	1.1	0.8	0.0	0.0	0.0	0.0	0.3	9.7	11.9
Total 1985	2282.5	1123.8	88.8	7.9	71.0	0.9	14.1	30.2	3619.3

(b) Time-span 1985–1999									
1985	1999								
	Woodlands	Open areas	Scrublands	Isolated trees	Conifers	Hedges	Buildings	Roads	Total 1985
Woodlands	2262.5	15.4	3.8	0.4	0.0	0.2	0.2	0.0	2282.5
Open areas	28.9	1078.0	11.1	0.9	1.5	0.2	3.2	0.0	1123.8
Scrublands	5.4	4.9	78.4	0.1	0.0	0.0	0.1	0.0	88.8
Isolated trees	1.1	0.2	0.0	6.6	0.0	0.1	0.0	0.0	7.9
Conifers	0.9	0.4	0.1	0.0	69.7	0.0	0.0	0.0	71.0
Hedges	0.2	0.0	0.0	0.0	0.0	0.8	0.0	0.0	1.0
Buildings	0.0	0.0	0.0	0.0	0.0	0.0	14.1	0.0	14.1
Roads	0.0	0.0	0.0	0.0	0.0	0.0	0.0	30.2	30.2
Total 1999	2298.9	1098.9	93.4	8.0	71.2	1.3	17.5	30.2	3619.3

further declined by 8.4% from 1985 to 1999 (from 2425 to 2222 patches), resulting in an overall reduction of 35.3% for the entire period (1954–1999). In terms of the number of patches (NP) per class (Fig. 3b), from 1954 to 1999, all classes decreased in number, with the exception of roads (+25%). Woodlands showed a strong decrease in the first period (−32.2%) and a slight increase in the last period (+7.9%) with an overall decrease of 26.8%. Open areas have constantly diminished in the number of patches from 1954 to 1985 (−28.3%) and from 1985 to 1999 (−13.4%), with an overall loss of 37.9%, while scrublands were characterized by an increase in 1985 and then a decrease in 1999 (overall loss of 9.1%). The NP of isolated trees, hedges and buildings decreased in general between 1954 and 1999 by 48.2%, 61.3% and 4.5%, respectively, although the main reductions occurred between 1954 and 1985; the NP of hedges and buildings grew in the 1985–1999 time period (+41.2% and +27.6%,

respectively). The NP of conifers did not change between 1985 and 1999.

In terms of patch shape, open areas held the highest value of AWMSI in 1954, but there was an abrupt and significant ($p < 0.01$) decrease in the values of this index over time (Fig. 3c). The shape of woodlands, open areas, scrublands and hedges patches became simpler, going towards a more homogeneous shape. By contrast, buildings and roads increased in AWMSI with more diffused and complex shapes. High values in AWMSI for the class roads are due to their linear shapes. Conifers and isolated trees showed no significant change in shape over time. Over the three sampling dates, AWMPs of woodlands and open areas was higher than that of the other land cover classes (Fig. 3d). In particular, a significant increase of AWMPs in woodlands and a significant decrease in open areas were noted. The ranked patch frequency-area distributions over

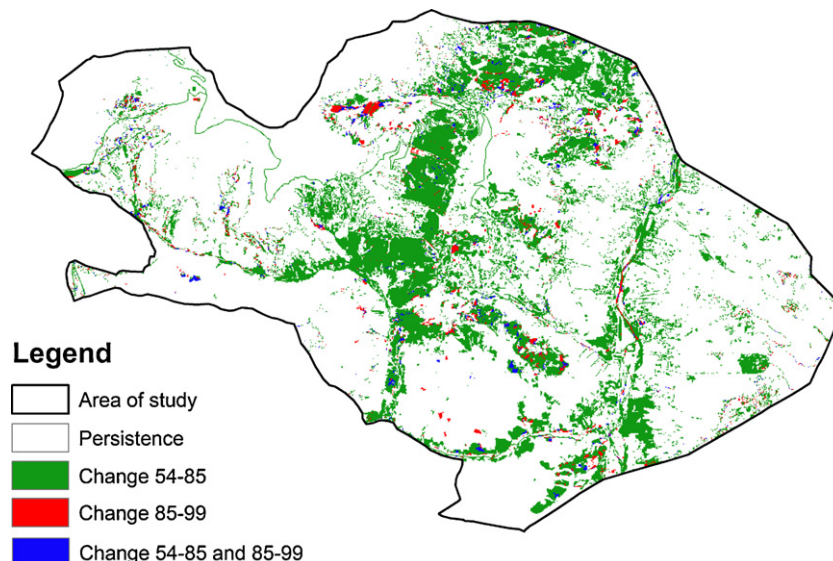


Fig. 4. Spatial distribution of landscape alterations.

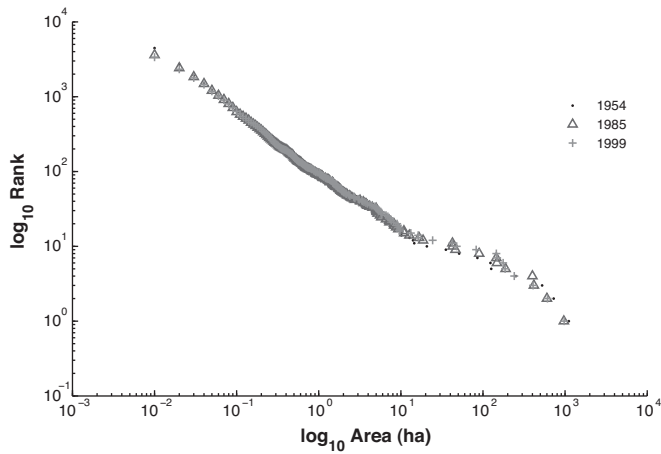


Fig. 5. Ranked patch frequency-area distributions for each year at landscape level; the X axis represents the logarithm of the area occupied by the patches and the Y axis represents the logarithm of the rank of the patches frequency-area.

the three sampling dates demonstrate very similar patterns, with a stable frequency of medium sized patches. However, a decrease in the smallest patches and an increase in frequency of the largest patches were observed (Fig. 5). Therefore, the Largest Patch Index (LPI, Fig. 3e) shows that open formations had the highest value in 1954, but suffered a remarkable decrease over the years. Woodlands LPI instead followed an opposite trend, mainly increasing between 1954 and 1985.

Woodlands and open formations were the only classes with a remarkable change in MPI over time (Fig. 3f). Woodlands had a considerable increase from 1954 to 1985 and a slight decrease from 1985 to 1999, suggesting an overall defragmentation over time; conversely, open formations displayed an abrupt reduction from 1954 to 1985, and a further decrease from 1985 to 1999, suggesting their increased isolation and therefore fragmentation over this period. The other classes had no change over time in MPI. The low values for buildings reveal their random distribution in the landscape. Conifers were planted in a few large groups and no spreading or shrinking has been observed. The roads class is characterized by a light increase in MPI due to the road network development that occurred in the period under consideration.

5.3. Landscape stability analysis

Between 1985 and 1999, the landscape was characterized by a higher stability in comparison with the period 1954–1985, with no further significant changes in the landscape mosaic, partially due to the shorter time span of study. Meanwhile, the rate of change (by means of the ratio between the cells where a change has occurred and the total extension in the period 1954–1985) was 19.7%, with a mean annual change rate of 0.64%, while the rate of change in the period 1985–1999 was 2.2% with a mean annual change rate of 0.16%.

To better explain the different degrees of change between the two time steps, the K statistic and a Markov chain model were used.

The Kappa statistic has allowed an estimate of the agreement degree of a land cover category between two dates; specifically in Fig. 6, grey columns represent the per-category K values for the period 1954–1999, and indicate that woodlands, roads, coniferous plantation and building were the most stable components of the landscape, followed by open formations. Meanwhile, scrublands, isolated trees and hedges were the most instable classes. The Kappa values for conifers were computed only for the period 1985–1999, since this land cover was absent in the 1954. The high K values from 1985 to 1999 for all land cover classes are related

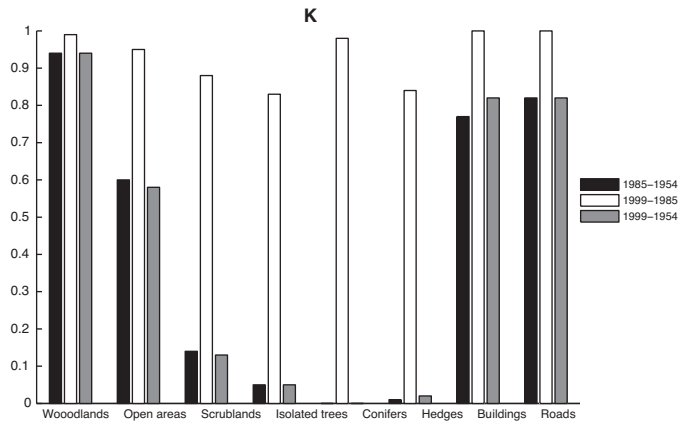


Fig. 6. Kappa (K) values for each class with the earlier image as reference. K represents the stability of each class in the corresponding time-span under analysis. Conifers are missing the K values for 1954–1985 and 1954–1999 time periods, since in 1954 this class was absent, making comparison impossible. Note the high values for the white columns, in the period 1985–1999, showing high stability of all the land cover classes for this period.

both to the high stability (most of the transformations occurred from 1954 to 1985), and to the shortness of this period in comparison with the first time span (14 years vs 31 years). To overcome this problem, a matrix of percentage differences between yearly Markovian transition probabilities for the two time periods was elaborated (Table 2). It clearly shows a decrease in change rate (negative values) over almost all transitions (small exceptions are in bold characters) and an increased stability in each land cover class along the diagonal (positive values). Fig. 7 represents future scenarios of a converging steady state (400 years). A rapid inspection of this graph indicates that the two distributions are distinct from one another, implying that the land cover change process in Micigliano is not stationary as confirmed by Anderson-Goodman test ($\chi^2 = 5292,78, p < 0.001$). According to the Euclidean distance, the expected steady state derived from the matrix of the first period (A_{54-85}) differs much more from the observed 1999 distribution than the projection of second matrix (A_{85-99}), respectively of 0.278 against 0.131, where 0 means a perfect agreement and 1 the highest discordance.

Regarding the stability analysis of the spatial pattern investigated by means of KW test, significant changes in shape and size of patches are respectively identified in Fig. 3c and d by markers. Concerning the patch shape, the KW test found the differences at the class level pointed out by AWMSI (Fig. 3c) to be significant only for woodlands, open areas and buildings. Although a clear difference in

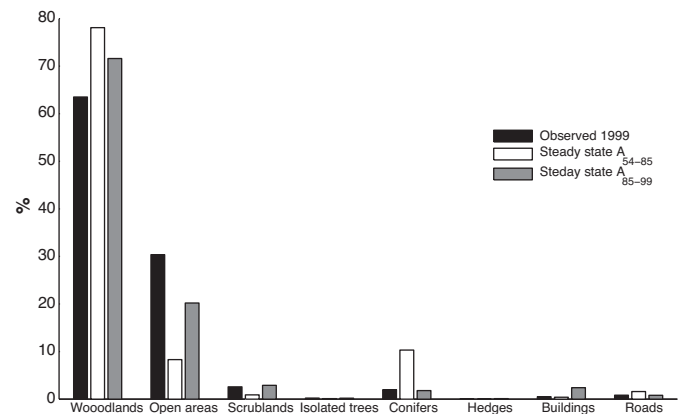


Fig. 7. Future scenarios of a converging steady state (400 years) derived by yearly transition probabilities of 1954–1985 (A_{54-85}) and 1985–1999 (A_{85-99}).

Table 2

Percentage variation of yearly transition probabilities between the two periods 1954–1985 and 1985–1999. Values represent % variation with respect to 1954–1985. Notice the positive values of the diagonal showing the increase in persistence during the last period.

1954/1985	1985/1999							
	Woodlands	Open areas	Scrublands	Isolated trees	Conifers	Hedges	Buildings	Roads
Woodlands	0.1	-51.5	-25.0	-50.0	-100.0	-	-	-100.0
Open areas	-75.5	0.8	-44.1	-45.5	-93.7	-50.0	50.0	-100.0
Scrublands	-81.5	9.1	1.9	-50.0	-94.4	-	-	-
Isolated trees	-42.3	-85.6	-100.0	1.9	-100.0	1057.1	-100.0	-100.0
Conifers	-	-	-	-	-	-	-	-
Hedges	-29.0	-100.0	-100.0	-100.0	-	2.1	-	-100.0
Buildings	-97.0	-93.7	-100.0	-	-	-	0.7	-100.0
Roads	-100.0	-100.0	-	-	-	-	-100.0	0.6

Table 3

Synthesis of stability analysis.

Class	Shape stationarity	Size stationarity	Transition stationarity
Woodland	NO	NO	NO
Open areas	NO	NO	NO
Scrubland	YES	NO	NO
Isolated trees	YES	YES	NO
Conifers	YES	YES	YES
Hedges	YES	YES	NO
Buildings	NO	NO	NO
Roads	YES	YES	YES

patch area distribution at the landscape level between three dates is not possible (Fig. 5), a variation in size at the class level was underlined by AWMPs (Fig. 3d) and LPI (Fig. 3e) values. Such differences were confirmed by the KW test, which found significant changes in area for woodlands, open areas, scrublands and buildings patches.

A synthesis of the stability analysis is finally reported in Table 3. Notice that the transition stationarity hypothesis is verified only for conifers and roads, a result of nonexistent or very few changes in the second period (Table 1b).

6. Discussions and conclusions

From the obtained results, it can be easily deduced that a strong simplification of landscape pattern for the Micigliano municipality occurred in the considered period of time. Despite the disappearance of most hedges and isolated trees, the main landscape homogenisation over time was realised through the increase of forest patches like in other Mediterranean areas (e.g. Geri et al., 2010; Marignani et al., 2008). With the exception of the large coniferous afforestation, the reduction of most open area patches was due to abandonment with a consequent secondary succession and spread of forests.

The proposed stability analysis allows for discussions about landscape pattern dynamics.

Woodlands and open areas, the two larger land covers in the study area, showed significant differences in shape and size of patches (KW test, Fig. 3c and d) during the analysed period, contributing to the general instability of landscape. A significant increase in compactness of woodlands and open areas patches, accompanied by a growth in woodlands patch size and a general reduction in the number of patches, led to a decrease in landscape diversity. Although a diminishing landscape pattern complexity seems to continue even in the second period (1985–1999), a reduction of land cover change rate over time was evident both at the landscape level, as underlined by the decrease in the mean annual change rate, and at the class level, as showed by the changes of K statistic (Fig. 6) and the annual transition probabilities pointed out by the Markovian matrices comparison (Table 2). These results suggest that the vegetation dynamics are far from being

stationary as confirmed by the Anderson-Goodman test and revealed in several papers on other mountain landscapes under abandonment (Pueyo & Bagueria, 2007; Romero-Calcerrada & Perry, 2004). Stationarity can be hypothesised only for conifers and roads which are not or very little changed in the second period (Table 1b and 3). For these two land covers, a steady state could be reached because their change rate seems to tend towards zero. Conifers, on the other hand, are an exogenous vegetation usually planted in scarcely productive soils or eroded slopes (Geri et al., 2010), hence they could find it more difficult to spread into surrounding areas than the native woodlands. However, further analysis should be carried out to verify the stationarity of these land covers in the recent dynamics.

The misregistration between cells of the three land cover maps was reduced since the low RMSE of the orthorectification and the pre-defined map resolution; moreover, a good accuracy in land cover classification was achieved. Landscape metrics and transition matrices show therefore a good reliability. In front of a high comparability of the maps, the presented method requires a high level of attention during the land cover classification, above all in the smaller patches of different land covers interspersed and/or juxtaposed. Its application on larger areas could therefore lead to classification errors due to weariness and concentration reduction of the operator. Different land cover map production methods more automated and less time consuming than the one here proposed, such as object-based segmentation (see Burnet & Blaschke, 2003; Gennaretti, Ripa, Gobattoni, Boccia, & Pelorosso, 2011) could be employed to study the landscape pattern dynamic on large scale always keeping in mind the map comparability maximization as central concept.

The stability analysis allowed to point out a reduction in change rate and the probable approaching to a new equilibrium (metastable) state of the landscape. In the first period (1954–1985), the landscape responded to human induced changes, such as abandonment of livestock, with a natural process of secondary succession defining a general pattern homogenisation. In recent years (1985–1999), the whole landscape pattern change rate was reduced. Such change decrease could represent dynamics of a landscape closer to a new metastable state, due to a balance between the actual human pressure (e.g. the recurring grazing/cropping activity and wood exploitation) and the spontaneous processes of succession, occurred over the abandoned or no more intensely human-exploited land patches. This equilibrium adjustment can be read even from the reduction of differences in land cover distribution between 1999 observations and the steady states derived from the two Markovian transition probability matrices (Fig. 7). Indeed, as confirmed by the Euclidean distance, the steady state derived by A_{85-99} better fits the observed 1999 land cover distribution than the steady state derived from A_{54-85} .

Neither particular environmental attention nor National or Regional policies seem to motivate and to support expensive

engagements in restoring a still indefinite traditional (reference) landscape and the linked landscape services. Moreover, it is costly and difficult to elaborate and maintain over the time a restoration action aiming to retrieve a past condition unrelated with the natural equilibrium between human activity (agriculture) and spontaneous processes of secondary succession. Indeed, designing a landscape pattern with restoration actions, which might seem adequate for some years, but that bears no relationship with the natural processes in the landscape system, is not likely to be effective (Marignani et al., 2008). In these conditions, a conservative approach, such that proposed by Pardini et al. (2002), could be considered a valid management strategy for the study area at least until emerging needs (e.g. increasing demand of “beauty” landscapes and traditional agro-sylvo-pastoral activity) or risks (e.g. land instability, fire and flood risk, disappearance of essential seed banks) for humans and ecosystems will not be drawn to public attention and clear ecological thresholds and economically and socially acceptable restoration goals will not identified.

In general, we argue that, since the limited resources and the difficult to maintain an optimal reference state for the large mountain landscapes, one of the major challenges could be to identify management opportunities maximizing ecological functions as well as the landscape services while minimizing restrictions on human land use and costs. In other words, an effective and reliable management strategy for abandoned landscapes should provide for a dynamic equilibrium between anthropic activity (land use), restoration/conservation costs (e.g. maintenance of the landscape pattern) and landscape capacity to provide goods and services. This equilibrium should be maintained over time through an adaptive management taking into account the spatial and temporal scale of natural processes as well as the shifting between

human and ecological perspective of the landscape. The reduction of wild fire risk and erosion processes should be the main objective to pursue on large scale for the maintenance of a wide system functionality and a stable and renewable flow of landscape services.

The development of a standardized analysis of landscape equilibrium has still to be realised due data scarcity and spatial- and time-variability of the processes driving the landscape evolution. In this work, we proposed a first framework to investigate the landscape stability in terms of land cover pattern. Such analysis can add a further judgment criterion in choosing the appropriate restoration and conservation strategies for abandoned lands and, therefore, its employment in plan management definition of the mountain landscapes of Mediterranean Basin should be considered.

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Appendix A. Summarizing table of the landscape metrics and statistical analysis techniques employed to describe and assess the landscape pattern dynamics and its stability. It aims to concisely explain how such metrics and stats address the research objectives.

Landscape metrics and statistical analysis techniques	Units	Range	Objective
Number of patches (NP)	None	$NP \geq 1$ It is 1 when the landscape contains only 1 patch of the corresponding patch type	It is a simple measure of the extent of subdivision or fragmentation of the patch type.
Area Weighted Mean Shape Index (AWMSI)	None	$AWMSI \geq 1$ It is 1 when the patch is maximally compact (i.e. a square for raster files or a circle for vector files) and increases without limit as patch shape becomes more irregular	Shape-based metric that measures the complexity of patch shape weighted by the patch size.
Area Weighted Mean Patch Size (AWMPS)	None	$AWMPS > 0$ It approaches 0 as the patch type becomes increasing rare in the landscape	Size-based metric that measures the mean patches size with a lower weight given to the small patches in the landscape.
Largest Patch Index (LPI)	Percent	$0 \leq LPI \leq 100$ It approaches 0 when the largest patch of the corresponding patch type is increasingly small, while LPI is equal to 100 when the entire landscape consists of a single patch	Size-based metric that quantifies, at the class level, the percentage of the total landscape area comprised by the largest patch.
Mean Proximity Index (MPI)	None	$MPI \geq 0$ It approaches 0 if a patch has no neighbors of the same patch type within the specified search radius	This index estimates the isolation of patches for a certain class within a given search radius (in this study it was set as 100 m).
Kappa Index of Agreement (KIA)	None	$-1 \leq KIA \leq +1$ If the two input images are in perfect agreement (no change has occurred), K equals 1. If the two images are completely different, K equals -1 . If the change between the two dates occurred by chance, then Kappa equals 0	This index allows to evaluate the agreement degree of a land cover category between two dates. It analyses the stability of the different classes.
Markovian matrices comparison	Percent	Negative values indicate a decrease of conversion rate. Positive values a increase of conversion rate. 0 means no change	This matrix comparison was used to assess the variation of the conversion rate among land cover classes between two periods.

Appendix A (Continued)

Landscape metrics and statistical analysis techniques	Units	Range	Objective
Markov model projections	Percent	0–100	This projection was used to compare the steady state land cover distributions derived from transition probability matrices of the two considered periods. Euclidean distance between distributions was used as measure of the approaching to a new metastable state.
Kruskal–Wallis test (patch shape)	None	≥0	This statistic assesses that the mean ranks of shape values are the same in each land cover class over the three considered dates.
Kruskal–Wallis test (patch size)	None	≥0	This statistic assesses that the mean ranks of size values are the same in each land cover class over the three considered dates.

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