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Land use change in the high mountain belts of the central Apennines led to marked changes of the grassland mosaic

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Abstract

Questions: High mountain pastures are hotspots of biodiversity, but grazing cessation and climate change are causing tall-grass encroachment and expansion of scrublands and forests. As part of biodiversity conservation efforts, grassland variation needs to be investigated at different spatial scales. We aimed to assess the landscape mosaic variation that occurred between 1988 and 2015 in the higher Mediterranean mountains. We investigated the recovery or land degradation processes related to land use change, the effects of site condition, the impacts on grassland mosaic heterogeneity, and the threats to biodiversity.

Location: Sibillini Mountains (central Italy), over 1650 m a.s.l.

Methods: We used two-step object-based supervised classification on Landsat 5 and 8 satellite images to analyze changes in landscape patterns and vegetation cover on formerly low intensity pastures, by assessing the Normalized Difference Vegetation Index variation between 1988 and 2015. Twenty percent of the polygons obtained from segmentation were visually interpreted and assigned to five land cover classes. We generated a land use transition matrix and used Fourier Transforms to detect trends in variation of landscape mosaics and fragmentation.

Results: We observed prominent dynamics of the grassland mosaic leading to the homogenization of its structure through decreasing patch heterogeneity, especially on south-facing slopes. Grasslands shifted from open communities to dense pastures, with reduction of scree and spread of tall grasses. The former trend could be understood as a recovery process reverting screes to conditions in equilibrium with local landform and climatic features, while the invasion of tall grasses is a land-degradation process that might lead to local species extinction and loss of habitat connectivity.

Conclusions: Pronounced changes in the large-scale landscape characteristics, mainly due to land use changes, of which scientists and managers of protected areas are not fully aware, are underway in the top mountain sectors of the study area.

KEYWORDS

climate change, dominant species invasion, grazing cessation, habitat degradation, habitat regeneration, landscape dynamics, landscape fragmentation, landscape homogenization, remote sensing

Nomenclature: Conti et al. (2005)

1| Introduction

High mountain pastures are hotspots of plant diversity, typically related to the mosaic of different habitats and the presence of dispersal barriers between mountains that facilitate speciation processes, but that make biodiversity vulnerable to loss (Körner, 1995), especially in mountainous Mediterranean regions, where the abandonment of farming is moreover contributing to threaten the plant diversity (Catorci, Ottaviani, & Cesaretti, 2011). In fact, farming cessation led to dominant tall grass encroachment (Grime, 2001), modifications of grasslands plant community composition (Peco, Carmona, De Pablos, & Azcarate, 2012), and/or expansion of scrubland and forests (Bracchetti, Carotenuto, & Catorci, 2012). In particular, the flora of Mediterranean high mountains is affected by a decrease of small-statured cryophilic species and an increase of caespitose hemicryptophytes and dwarf shrubs (Evangelista et al., 2016). These changes could be attributed to climate change, but also to grazing cessation (Catorci et al., 2011; Peco et al., 2012), or could be a by-product of both climate change and grazing cessation (Le Houérou, 1996). Whatever the cause, grassland variations have become a major concern for biodiversity conservation of Mediterranean mountains (Kaligarič, Culiberg, & Kramberger, 2006). However, to date, most studies on grassland systems have focused on small-scale plots rather than examining dynamics at the landscape scale, which instead would be highly valuable, as the results could be used to develop general models that would serve in restoration and conservation plans and projects and to improve environmental policies (Irwin & Geoghegan, 2001; Jobin, Latendresse, Grenier, Maisonneuve, & Sebbane, 2010). Landscape mapping is an important tool for studies on this scale, because it provides information through the quantification of vegetation at a given time point or over a continuous period (Attorre et al., 2014; Malatesta et al., 2013). In fact, it has been adopted for purposes such as the conservation of habitats (Lindenmayer & Cunningham, 1996) and biodiversity (Leathwick, Overton, & Mcleod, 2003), as well as environmental monitoring (Attorre et al., 2014). From this point of view, Remote Sensing (RS) has proven to be a practical and economical tool for studying vegetation cover and mapping vegetation from local to global scales (Xie, Sha, & Yu, 2008), and has shown strong potential for quantifying important biotic characteristics of grasslands (Kallenbach, 2015; Primi et al., 2016). In particular, the Normalized Difference Vegetation Index (NDVI), obtained from a spectral transformation of the red and near infrared bands, has been used to estimate numerous vegetation properties, since these bands are closely related to the absorption of photosynthetically active radiation by green vegetation (Feoli, Gallizia-Vuerich, Ganis, & Woldu, 2009; Peñuelas & Filella, 1998; Purevdorj, Tateishi, Ishiyama, & Honda, 1998; Rasmussen, 1998; Rouse, Haas, Schell, & Deering, 1973). Vegetation properties, such as aboveground phytomass, are often derived by correlating space-derived NDVI values with ground-measured values of the considered factor (Finley, Munoz, Gehl, & Kravchenko, 2010; Mangiarotti, Mazzega, Hiernaux, & Mougin, 2012).

In mountain landscapes, altitude and landform are major factors in determining variations in species composition and community structure (Burrascano et al., 2013). These factors partly determine the range of aboveground phytomass typical of each plant community (Scocco, Piermarteri, Malfatti, Tardella, & Catorci, 2016). Moreover, it has been demonstrated that, when livestock grazing ceases, the sward tends to increase in height and thickens, and subsequently the canopy closes and the aboveground phytomass increases (Louault, Soussana, & Perrodin, 2002). Consequently, modifications of grassland mosaic can be effectively approached by analyzing the pattern of local variations in aboveground phytomass (as denoted by modifications in NDVI values), and, on a large scale, by assessing changes of landscape metrics such as fragmentation (Gardner, O'Neill, & Turner, 1993). Analysis of changes in landscape fragmentation proved useful for detecting landscape pattern, structure and complexity, and for describing changes in landscapes biodiversity (Dunn, Sharpe, Guntensbergen, Stearns, & Yang, 1991).

Taking into account all the foregoing considerations, in this study we used two-step object-based supervised classification to analyse the changes in landscape patterns and vegetation cover, and Fourier Transforms to analyse landscape fragmentation on the formerly extensive pastures of the high-mountain, subalpine and alpine belts over 1650 m a.s.l. (the upper timberline runs at 1800-1900 m a.s.l.; Catorci, Scapin, Tardella, & Vitanzi, 2012) of the Sibillini Mountains (central Italy). This area, of great importance for biodiversity conservation in Italy, has been marked by traditional shepherding activities for centuries. It includes screes considered habitats of community interest for biodiversity conservation, *sensu* 92/43/EEC Directive [8120 - Calcareous and calcshist scree of the mountain to alpine levels (*Thlaspietea rotundifolii*)], that are also recognized as potential habitats for priority bird species, such as *Alectoris graeca* (Rippa, Maselli, Soppelsa, & Fulgione, 2011), as well as xerophilous grassland communities, which are also deemed habitats of community interest [6170 – Alpine and subalpine calcareous grasslands; 6210* - Semi-natural dry grasslands and scrubland facies on calcareous substrates (*Festuco-Brometalia*) *important orchid sites]. However, in recent decades there has been a marked progressive decrease in grazing intensity (starting from the 1970s and further declining up to the 1990s), and at present several areas are totally abandoned. This is due not only to socio-economic factors, but also to policy choices by the managers of the "Monti Sibillini" National Park, who have decided to prohibit sheep grazing in some sectors, in order to foster the reintroduction of Apennine chamois (*Rupicapra ornata*) and to avoid transmission of disease between sheep and wild herbivores.

Our aim was to assess the landscape variation that occurred between 1988 and 2015, in terms of changes of the grasslands mosaics of the upper montane, subalpine and alpine belts in the central Apennine ridge, where harsh environmental conditions strongly limit forest and shrub expansion, and stakeholders may not perceive easily the variations occurring in the system. The main questions were: i) are processes of recovery or land degradation underway in the system due to land use changes? ii) do site condition influence these processes? iii) do these processes affect the grassland mosaic heterogeneity? iv) especially in habitats of conservation interest, does variation in the system potentially threaten biodiversity?

2 | Methods

2.1 | Study area

The study area (Fig. 1) of about 9200 ha lies within the “Monti Sibillini” National Park (Italy, central Apennines: central coordinates 42°49’26” N 13°16’32” E) and is characterized by limestone bedrock. It encompasses the pastoral landscapes over 1650 m a.s.l. up to the highest peaks (M. Vettore, 2476 m a.s.l.). The bioclimatic characteristics are shown in Appendix S1. The plant landscape is composed of a very rich mosaic of herbaceous plant communities. At lower altitudes (below 1750 m a.s.l.), some small patches dominated by *Fagus sylvatica* or *Rhamnus alpina* are scattered amid pastures. The flora of this area (about 700 species and subspecies) includes several rare or endemic species (about 180 and 150 taxa, respectively) and arctic-alpine species (about 100 taxa) sheltered on the highest peaks after the end of glaciation (Ballelli et al., 2010).

For centuries, the entire study area was intensely grazed during the summer. Flocks were led to high pastures in early July and remained until the end of September. However, in recent decades the number of flocks and their size have constantly decreased. At present, the area above 1600 m a.s.l. has been mostly abandoned.

2.2 | Data collection and processing

2.2.1 | General approach

Since it has been proven that the inter-annual variation in rainfall amount has a strong influence on phytomass production (Chelli et al., 2016; Scocco et al., 2016), we preliminarily verified that the summer rainfall amounts of 1988 and 2015 were equivalent (1189 mm in 1988 *vs* 1286 mm in 2015; data were taken from the Marche Region meteorological data set – ASSAM, 2018). Given the slight differences in rainfall between 1988 and 2015, it can be said that inter-annual precipitation variability had a negligible effect on the observed land cover changes.

To assess patterns of landscape changes, we used Landsat images. These images were classified to obtain Land Cover (LC) maps, using a two-step approach consisting in automatic segmentation and supervised classification of the resulting polygons. Training areas were obtained from visual interpretation of historical aerial photographs and Very High Resolution (VHR) satellite images. The classification of the two sets of images was executed independently. The Digital Elevation Model and NDVI were used as ancillary data for classification and change analysis, while ground truth data (obtained from field surveys) were compared to the map generated in 2015 to identify the vegetation types and the plant communities corresponding to our LC classes. We mapped the changes in LC to assess the reduction of plant communities of conservation interest. We also performed a Fourier Transforms analysis on Landsat images to analyse the changes in landscape fragmentation and verify possible ongoing landscape homogenization processes. A flow chart of all the data processing and analysis is provided in Fig. 2. All the spatial data were processed using Quantum GIS (QGIS) 2.12 (QGIS Development Team, 2015) and GRASS GIS 7.0.3 (Neteler, Bowman, Landa, & Metz, 2012).

2.2.2 | Data sets and pre-processing

Landform features, elevation data, aspect angle (azimuth degrees) and slope angle (vertical degrees) were elaborated basing on the Global Digital Elevation Model (GDEM) with 30 m pixel resolution obtained from the NASA Advanced Spaceborne Thermal Emission and Reflection Radiometer (ASTER) mission website (http://asterweb.jpl.nasa.gov/gdem.asp). We derived aspect, slope and raster maps by a multi-scale approach, fitting a bivariate quadratic polynomial to a window size of three pixels using least squares. To do this, we used *r.param.scale* function in GRASS GIS.

Concerning RS data, for 1988 we used a Landsat 5 Thematic Mapper (TM) image acquired on August 7, 1988. Landsat 5 TM provides orthorectified multispectral images with 30 m pixel resolution and 7 spectral bands. For 2015, we used a Landsat 8 Operational Land Imager (OLI) image acquired on July 1, 2015. Like the earlier Landsat 5 TM, Landsat 8 OLI also provides orthorectified multispectral images with 30 m pixel resolution, but the number of spectral bands is increased to 9. (Landsat 5 TM and Landsat 8 OLI images courtesy of the U.S. Geological Survey - https://landsat.usgs.gov/). Several studies (e.g. Teillet et al., 2001; Brown, Pinzón, Didan, Morisette, & Tucker, 2006; Yuan & Bauer, 2007; Li, Jiang, & Feng, 2013) demonstrated how different Landsat sensors can be successfully used for long term analyses based on NDVI. Both dates were selected to obtain the minimum cloud cover within the ideal period for vegetation growth (phytomass peak). All the images were pre-processed to obtain top-of-atmosphere reflectance values and then topographically corrected before being used for all the analyses described below. We applied radiometric correction to transform the calibrated digital number of Landsat imagery products to top-of-atmosphere reflectance, using uncorrected at-sensor values (Chander, Markham, & Helder, 2009). Radiometric collection was applied using the *i.landsat.toar* function in GRASS GIS. Topographic correction was based on an illumination model obtained from ASTER GDEM. This illumination model represents the cosine of the incident angle *i*, i.e. the angle between the normal to the ground and the sun rays. We used the c-factor method, which takes into account the overcorrection of low illuminated slopes by the original C method, to obtain corrected reflectance values (Riaño, Chuvieco, Salas, & Aguado, 2003). Topographic correction was performed using the function *i.topo.corr* in GRASS GIS.

NDVI was computed from the RS data obtained for the analyzed periods. NDVI is obtained from a spectral transformation of the Red (RED) and Near Infrared (NIR) bands: its values range between -1 and 1 (Broge & Leblanc, 2001). We chose the NDVI for this purpose since, despite many possible disturbing factors, NDVI remains a valuable tool for quantitative vegetation monitoring when the photosynthetic capacity of the Earth's surface has to be studied in a large spatial scale and/or to define local trends of variations in land cover type (Lunetta, Knight, Ediriwickrema, Lyon, & Worthy, 2006; Primi et al., 2016).

2.2.3 | Image classification and accuracy assessment

Given the unavailability of ground truth points for the year 1988, we used an approach based on supervised object-based classification of LC units, carried out in two steps. First, we mapped landscape patterns through an image segmentation procedure. Image segmentation is the process of grouping similar pixels into unique segments, also referred to as objects, basing on the pixel values of a single-band or multi-spectral image (Pitas, 2000). Each object found during the segmentation process is delimited by a polygon, and is a collection of contiguous pixels recognized as a homogeneous portion of the image. We applied a region-based algorithm called "region-growing" for segmentation, since region-based algorithms have been found to be the most reliable, given their resistance to noise or texture (Carleer, Debeir, & Wolff, 2005). In this algorithm, the procedure starts at each point in the image with one-pixel objects, and in numerous subsequent steps, smaller image objects are merged into bigger ones, through a pair-wise clustering process. The merger is based on three criteria: color, smoothness, and compactness. The segmentation was performed using the function *i.segment* in GRASS GIS. We excluded the Aerosol (Band 1), Panchromatic (Band 8) and Cirrus (Band 9) from the set of bands analyzed for segmentation of the Landsat 8 image (2015) to avoid generating noise.

The second step to create our Land Cover maps consisted of a supervised classification of the polygons obtained from segmentation. The supervised classification was based on Landsat bands, NDVI, landform, slope angle and aspect data. In order to obtain training areas for the supervised classification, 100 (approximately 20%) of the polygons resulting from segmentation were visually interpreted and assigned to five different LC classes (Table 1). To facilitate the visual interpretation, we overlaid the aforementioned polygons to fine resolution imagery from different sources (Fig. 3). For 1988, we used historical aerial photographs of the study area provided by the Italian Military Geographical Institute (IGM - https://www.igmi.org/en/descrizione-prodotti/aerial-photography). For 2015, the polygons were overlaid on VHR satellite images in Google Earth.

We used a machine-learning algorithm implemented through the *v.class.mlpy* function in GRASS GIS to classify the remaining polygons, basing on the statistics extracted from the training dataset. The datasets used for statistics extraction included: satellite bands (Blue, Green, Red, Near Infrared, Shortwave Infrared 1 and 2), NDVI, landforms, slope angle and aspect. The *v.class.mlpy* function is a tool for supervised vector classification built on top of the Machine Learning Python (MLPY) library, a collection of machine learning algorithms written in the Python programming language (Albanese et al., 2012). The function has been specifically developed to be used in GRASS GIS for supervised vector classification, but the MLPY library is open source and free to use (http://mlpy.sourceforge.net/). In our case, a Maximum Likelihood classifier based on the Diagonal Linear Discriminant Analysis (Dudoit, Fridlyand, & Speed, 2002; Pique-Regi, Ortega, & Asgharzadeh, 2005) was used for classification. This approach uses a linear discriminant rule to assign polygons to the different classes, based on the assumption that the class densities must have the same diagonal covariance matrices (Dudoit et al., 2002). This simplification increases the reliability of the covariance estimation, but is vulnerable to noise from variables that are irrelevant to the classification. Therefore, the variables used for the classification were carefully selected after searching for the subset that best improved the class separation, following the procedure described in Pique-Regi et al. (2005).

The accuracy of the two maps produced for 1988 and 2015 was assessed by comparing the final classified map with a test set of 50 pre-classified polygons (approximately 10% of the total polygons generated by the segmentation process). The test set was generated independently from the training set. The accuracy assessment consisted in superimposing the test set and the classified map to compare the classification of each map pixel with its corresponding test pixel, and computing the number of correct and incorrect assignments (Congalton, 1991). This operation was carried out in GRASS GIS using the function *r.kappa* (Neteler et al., 2012). The percentage of correct observations and the “kappa” statistic were calculated with this method for the maps obtained for 1988 and 2015 (Xie et al., 2008). The observed classification accuracy was equal to 100% for both years (kappa = 1). To detect the correspondence of the LC classes with the different types of plant communities present in the study area, we also performed field surveys in about 100 polygons. This also allowed us to verify the reliability of our classification in the field. The correspondence of LC classes (mapped for 2015) with vegetation types is shown in Appendix S2.

2.2.4 | Land Cover change analysis

To detect the pattern of changes in landscape patches, and thus to understand the main trends of vegetation modification between 1988 and 2015, we generated a land use transition matrix (Pontius, Shusas, & McEachern, 2004). This matrix indicates the number of pixels shifted from a certain category to another, while the values on the matrix diagonal indicate the persistence (Pontius et al., 2004). Using the QGIS *AccurAssess* plug-in (Mas et al., 2014), we elaborated the transition matrix between 1988 and 2015, by a pixel-wise comparison of the classified images from 1988 and 2015. The result data as a .csv file was entered into a spreadsheet processing software for further analyses.

Surface areas and percent cover of the LC classes in 1988 and 2015 with respect to aspect and slope angle were calculated using zonal statistics: the classified maps were superimposed to aspect and angle raster maps, and the distribution of each LC class in the different aspect and angle classes (described below) was calculated. We used *r.report* and *r.statistics* functions in GRASS GIS to obtain zonal statistics. We categorized aspect values in two classes: south-facing areas (values ranging from east south-east to north north-west clockwise) and north-facing areas (values ranging from north north-west to east south-east clockwise). We categorized slope angle values in the following classes: less than 15°; ranging from 16° to 30°; ranging from 31° to 45°; higher than 45°.

2.2.5 | Landscape fragmentation analysis

In order to analyze the changes in fragmentation at the landscape scale, we applied an approach based on Fourier Transforms (Fourier, 1822), which have proven to be a very effective method for analyzing landscape fragmentation, since they are based on a continuous function that requires neither a-priori field information, nor a specific model based on the data being used (Rocchini et al., 2013). Considering a raster image as a continuous function defined in the spatial domain, based on the Fourier theorem (Fourier, 1822), this function can be transformed into a continuum of sinusoidal functions of varying frequencies. These functions associate with the original spatial domain an alternative coordinate space, also termed frequency domain. When applying Fourier Transforms on a raster image, a plot of the frequency domain is generated. Lower frequencies are generally plotted at the center of the Fourier spectrum, while higher frequencies are plotted in the more external areas. Homogeneous landscapes are generally characterized by a constant sinusoidal wave, showing high values in the low frequency part of the domain, since higher frequencies are restricted to a small range of (dominant) values (Rocchini et al., 2013). Hence, low-frequency homogeneous images are expected to show high values at the center of the spectrum and very low values in the more external areas. An increase in fragmentation should provoke an increase in higher frequencies, resulting in a more complex (heterogeneous) Fourier spectrum with higher values in the high-frequency part of the frequency domain (Rocchini et al., 2013). Probability density functions can be generated from Fourier spectrum plots to improve readability. In the density function curve, a skewed distribution of Fourier frequency values, characterized by the dominance of few values, indicates high homogeneity over the landscape, while lower skewness results from an increase in equitability of values due to heterogeneity (Rocchini et al., 2013). An extensive mathematical description of Fourier Transforms is provided by De Bie (2012), while an example of application is described in Rocchini et al. (2013).

To obtain Fourier Transforms of the 1988 and 2015 satellite images, we first executed a Principal Component Analysis (PCA) on the Landsat 5 (for 1988) and Landsat 8 (for 2015) bands. We used PCA to obtain an image representing the first Principal Component (PC) of each multi-spectral image, resuming the original variability in a single band. We excluded Aerosol (Band 1), Panchromatic (Band 8) and Cirrus (Band 9) from the set of input bands for PCA of the Landsat 8 image (2015) to avoid generating noise. To perform the PCA, we used the function *i.pca* in GRASS GIS. Then, we applied the Fourier Transforms algorithm (Rocchini et al., 2013) as described above on the first Principal Component (PC1) of each image, to construct the real and imaginary Fourier components in frequency space. In these components, represented as raster images, the low frequency components are in the center and the high frequency components are toward the edges. The Fourier components were produced with the function *i.fft* in GRASS GIS. A kernel density function was applied to the real component of the Fourier output to observe the degree of dominance in each of the two images analyzed (Bowman & Azzalini, 1997, 2015). To apply the kernel density function and obtain probability density curves, we used the *sm* package in R software (R Core Team, 2016).

3 | Results

3.1 | Land Cover changes

In 1988 the landscape was composed mainly of closed-turf grasslands (CGr), which covered 4739 ha (51.0% of the total area), followed by open-turf grasslands (OGr), with about 2601 ha (28.0%), while the remaining territory was covered by scree and rocky areas (ScRk, 1060 ha, 11.4%), dense grasslands which are dominated by *Brachypodium genuense* (DGr, 601 ha - 6.5%) and wooded areas (Wd, 272 ha, 2.9%). In 2015, the landscape remained dominated by CGr with 4311 ha (46.5%), followed by OGr (2293.6 ha) and DGr (1766.5 ha), while the remaining area was characterized by ScRk (562.6 ha), and Wd (341.5 ha) (Appendix S3).

In the long run, we observed a decrease of ScRk (-47%), OGr (-12%), and CGr (-9%), and an increase of DGr (+194%). These changes led to a deep modification of the landscape structure, since habitat with open turf (ScRk and OGr) decreased from 3662 to 2856 ha (-22%); while habitat with thick turf (CGr and DGr) increased from 5340 to 6078 ha (+14%). The extension of wooded landscape increased only slightly (+69.5 hectares).

The transition matrix (Table 2) shows for the ScRk class that 39% of the surface did not change, while 52% evolved into OGr. For OGr, it shows that 47% did not change, while 42% evolved into CGr. Instead, 60% of CGr remained unchanged and 26% evolved into DGr. For DGr, the transition matrix shows that 71% remained unchanged, while 21% changed into CGr. For Wd, 69% was stable while 31% changed into DGr (16%) and CGr (15%).

3.2 | Relation between Land Cover changes and landforms

As far as aspect is concerned, Table 3 shows that on north-facing slopes, grasslands with closed turf (CGr and DGr) essentially did not change (3022 *vs.* 3139 ha), while DGr (dense turf) strongly increased from 338 to 1116 ha and CGr (closed-turf) decreased from 2684 to 2023 ha. Instead, on south-facing slopes, the sum of CGr and DGr increased from 2319 to 2939 ha with an increase of both CGr (2055 *vs.* 2288 ha) and DGr (264 *vs.* 651 ha). This is reflected in the reduction of both ScRk- scree and rocky areas (619 *vs.* 264 ha) and OGr - open-turf grasslands (1694 *vs.* 1368 ha).

As regards slope, all the considered classes substantially showed the same trend, with an increase of the sum of CGr and DGr, even if the increase of DGr was sharper in less steep slopes. On the other hand, on the steepest slopes, the coverage of open-turf grasslands increased, while that of scree/rocky areas decreased. Wooded lands increased in all the slope classes, except in the lowest one (Table 4).

3.3 | Landscape fragmentation

Statistical analyses applied on Fourier Transforms show that in 1988, the study area was characterized by greater overall landscape heterogeneity; in fact, the probability density curve was flat and the equitability of Fourier frequency values was higher. Instead, in 2015, the curve showed a skewed distribution, with the dominance of a small range of values indicated by the peak, related to a higher homogeneity of the landscape (Fig. 4).

4 | Discussion

First of all, since the amount of rainfall for the two years was comparable, it is safe to say that the inter-annual climatic variation did not play a major role in determining the results of the study case. We observed a decrease of areas covered by communities with open turf (ScRk and OGr) and an increase of pastures with thick and closed sward (CGr and DGr). It is well known that extensive abandonment of the pastoral system or reduction in livestock pressure leads to changes of community structure by favoring the development of taller and denser vegetation (Vassilev, Pedashenko, Nikolov, Apostolova, & Dengler, 2011), mainly through encroachment of tall grasses (Peco et al., 2012). In fact, we observed a marked increase of DGr, which is primarily *Brachypodium genuense*-dominated grasslands. *Brachypodium genuense* is a rhizomatous, competitive-stress tolerant, coarse, tall grass, whose competitive ability is related to the plasticity of its leaves (Tardella, Bricca, Piermarteri, Postiglione, & Catorci, 2017) and its high tiller density (Pottier & Evette, 2010), as well as to its ability to adopt a clonal integration strategy (de Kroon & Bobbink, 1997). It was demonstrated that the invasive spread of *Brachypodium* sp. pl. leads to modifications of community composition and to a decrease in species richness, especially of the small-statured ones (Catorci et al., 2011). Moreover, the spread of *Brachypodium* has been linked to a reduction of forage quality for both domestic (Vitasović Kosić, Tardella, Grbeša, Škvorc, & Catorci, 2014) and wild (Corazza, Tardella, Ferrari, & Catorci, 2016) herbivores. Analogously, the increase of turf density following grazing cessation was considered harmful for several bird species, including *Alectoris graeca* (Rippa et al., 2011).

More in detail, we observed a quite clear dynamic pattern of land cover changes. Basically, scree (included in ScRk) developed into open-turf grasslands (OGr), with the result that there was a strong decrease in the surface area covered by scree. The observed trend could be attributed to the decrease of sheep disturbance, since it has been proven that trampling and grazing cause soil erosion in arid and steep systems (Thornes, 2007) and that grazing cessation leads to vegetation recovery (Cipriotti & Aguiar, 2012). Therefore, the observed changes might be understood as a process in which the scree distribution reverts to ancient, non-anthropogenic conditions, in equilibrium with the local landform and climatic features. However, it can also be considered problematic, in that scree are considered habitats of community interest for biodiversity conservation, *sensu* 92/43/EEC Directive [8120 - Calcareous and calcshist scree of the mountain to alpine levels (*Thlaspietea rotundifolii*)] and host several plants of high naturalistic value (e.g. *Adonis distorta*, *Campanula alpestris*, *Isatis apennina*, *Hornungia alpina* susbp. *alpina*, *Ranunculus alpestris*, *R. brevifolius*, and *Papaver alpinum* subsp. *ernersti-mayeri*).

OGr had a double shift. On the one hand, 551 ha of former scree developed into new open grassland patches, but on the other, 1085 ha of former open grasslands changed into CGr class. The net outcome of this process was a decrease of the total surface of open grasslands. In addition, only 46.7% of open grasslands did not change during the considered time interval. This could threaten small dry xerothermic calcicolous specialist species, since regeneration of these species within closed vegetation cover is limited (Wallis DeVries, Poschlod, & Willems, 2002) and seed recruitment of subalpine xerophilous species is extremely limited (Zeiter, Stampfli, & Newbery, 2006), thus largely hampering the colonization of new patches and increasing the risk of local species extinction (Willis, Hall, Rubio de Casas, Wang, & Donohue, 2014).

Some of the CGr patches developed into DGr, while in other cases, new CGr were formed because open, dry grasslands (OGr) changed into CGr. The net outcome of these changes was a slight reduction of these pastures (4739 vs. 4311 ha). Since this class includes some peculiar high-altitude vegetation types (*Nardus stricta*- and *Plantago atrata*-dominated grasslands) that host endemic or rare plant species, it will be of major importance to understand the impact of these shifts on the specific composition, because there are several indications that stability of grasslands reflects on their composition, especially in the case of rare species (Pikälä, 2003). Moreover, as formerly stated, the spread of *Brachypodium genuense*-dominated communities (DGr) leads to a marked change in species composition (Catorci et al., 2011).

As regards landforms, we observed that aspect had a certain importance in diversifying the pattern of patch changes, in that the net outcome of modification of south-facing slopes was an evident decrease of habitats with open turf, whilst on north-facing slopes their extension did not change as much. This finding seems to support the suggestion that grazing cessation has a stronger impact than climate change, since the increase of water shortage (as forecasted by climate change models in Mediterranean regions - Giorgi & Lionello, 2008) should lead to an increase of open habitats (Sternberg & Shoshany, 2001), especially on south-facing slopes. Actually, previous studies on subalpine grasslands suggested that subalpine grasslands are resilient to climate changes (Jung et al., 2014), and more sensitive to land use changes (Vittoz, Selldorf, Eggenberg, & Maire, 2009). In any case, disentangling the effects of climate change from those of disturbance remains a significant challenge (Palumbo, Marchetti, & Tognetti, 2014).

North-facing, gentle slopes were characterized by the greater increase of *Brachypodium genuense*-dominated communities (DGr). In fact, it was demonstrated that this plant community thrives in more productive conditions (deeper soil, north-facing slopes, and concave surfaces) (Tardella et al., 2017).

We also found that encroachment by shrubs and trees (Wd) was higher on north-facing slopes as well, probably because of the deeper and more humid soil. However, on south-facing slopes, wide encroachment of individuals of *Juniperus communis* subsp. *nana* has been observed, but with sprouting only in small patches (Allegrezza et al., 2016). The recovery of woody vegetation was less effective than expected (Bracchetti et al., 2012). Two factors may contribute to this situation: the mortality of beech seedlings due to *Brachypodium genuense* competition for resources (Catorci et al., 2012), or the local extinction of subalpine shrubs, like *Pinus mugo*, which has meant that *P. nigra* (an anthropogenic species) is the only tree species expanding at high altitudes, in a scattered and transitory process that started 35–40 years ago (Piermattei, Lingua, Urbinati, & Garbarino, 2016). Actually, observing the data set carefully, it emerged that about 30% of wooded areas regressed to grassland communities, probably because of the effect of avalanches and landslides, while only 5% of dense grasslands evolved into new wooded areas. This again highlights the strong patch turnover that characterises the system.

With respect to the landscape mosaic structure, in confirmation of the observations of other authors (Campagnaro, Frate, Carranza, & Sitzia, 2017), we observed a decrease of the overall landscape heterogeneity as a consequence of the cessation of traditional land use, which underlines a general process of landscape homogenization at different spatial scales. The landscape homogenization may compound the threat to plant diversity caused by climate change, because it limits the possibility of plants to disperse through landscapes in search of local conditions to which they are adapted (Pearson & Dawson, 2005). In addition, Duparc et al. (2013) found that the mosaic of subalpine grassland communities is heterogeneous in the timing of flowering phenology, creating small-scale variability in forage quality and quantity that could be of basic importance for the conservation of wild herbivores, birds and insects.

5 | Conclusion

Notwithstanding the possibility that the exact amount and percentage of patches variation might be influenced by some inaccuracies related to the method used, the observed general trend may be considered reliable mainly because of the marked concordance in patch variations of different vegetation types. The main output of this research is the demonstration that marked changes in grassland mosaic patterns are underway in the top mountain sectors of the study area. Until now, such variations in the large-scale landscape characteristics have not been readily perceived, and scientists and those who seek to protect nature have not had cause for alarm. Instead, the observed variation is a harbinger of a significant threat for plant diversity in the higher belts of Mediterranean mountains, due to the reduction of the extent of LC classes related to plant communities constituting habitats of interest for biodiversity conservation, and hosting several plants of high naturalistic value.

We detected a pronounced dynamics throughout the grassland mosaic, involving high patch turnover and leading to homogenization of the landscape structure, especially on south-facing slopes. Two main processes emerged from our study, the first related to the strong reduction of scree and the second to the spread of the *Brachypodium genuense*-dominated community. The first case could be understood as a recovery process to conditions in equilibrium with the local landform and climatic features. The second case, instead, is a process that likely prevents the recovery of more natural conditions and fosters a loss of biodiversity, thus contributing to land degradation. Actually, the question of whether the abandonment of mountain grazing activities enhances or threatens biodiversity conservation remains open, since the various impacts of abandonment are evaluated in different ways, through the choice of which metrics or taxa are assessed, or which aspects of conservation receive attention (Queiroz, Beilin, Folke, & Lindborg, 2014).

In conclusion, actions devoted to managing these changes and recovering a higher degree of landscape heterogeneity are urgently needed. In particular, a modern system of shepherding in the summit sectors of the study area should be promoted. In particular, it would be advisable to organize intermittent periods of intense grazing pressure in specific areas of south-facing slopes, in order to promote grassland recovery. In addition, those responsible for nature conservation should identify areas along the main environmental gradients to be managed with grazing at varied and alternating levels of intensity. In fact, orienting corridor linkages along environmental gradients may assist species in tracking climatic suitability in the future (Pearson & Dawson, 2005), maximizing their potential to persist in face of rapid global climate change (Jewitt, Goodman, Erasmus, O'Connor, & Witkowski, 2017).

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Supporting Information

Appendix S1. Bioclimatic characteristics of the study area.

Appendix S2. Correspondence of land cover classes, mapped for 2015, with vegetation types, obtained from field surveys in about 100 polygons.

Appendix S3. Distribution of land cover types in the study area in 1988 and 2015.

Table 1

Land Cover (LC) classes used in the present study, with related abbreviations and approximate NDVI values.

LC class	Abbreviation	NDVI range
Scree/rocky areas	ScRk	0.10-0.25
Open-turf grasslands	OGr	0.26-0.40
Closed-turf grasslands	CGr	0.41-0.60
Dense grasslands	DGr	0.61-0.70
Wooded areas	Wd	> 0.70

Table 2

Percent values of transition between different land use classes from 1988 to 2015. The percent values of unchanged land use are highlighted with light grey background.

		2015				
	Land use class	(ScRk) Scree/rocky areas	(OGr) Open-turf grasslands	(CGr) Closed-turf grasslands	(DGr) Dense grasslands	(Wd) Wooded areas
1988	(ScRk) Scree/rocky areas	39.0	52.0	7.9	0.6	0.4
	(OGr) Xeric grasslands	3.0	46.7	41.7	7.6	1.1
	(CGr) Closed-turf grasslands	1.5	10.7	60.1	25.6	2.0
	(DGr) Dense grasslands	0.1	2.8	21.2	71.2	4.6
	(Wd) Wooded areas	0.0	0.7	15.5	15.1	68.6

Table 3

Cover of land use classes in two slope aspect categories in 1988 and 2015. Values are in hectares.

	Slope aspect class			
	North		South	
Land use class	1988	2015	1988	2015
(ScRk) Scree/rocky areas	441.5	298.5	619.1	264.1
(OGr) Open-turf grasslands	907.4	925.9	1694.3	1367.7
(CGr) Closed-turf grasslands	2684.4	2023.2	2054.8	2288.3
(DGr) Dense grasslands	338.0	1115.8	263.8	650.6
(Wd) Wooded areas	204.0	253.5	68.4	88.1

Table 4

Cover of land use classes in four slope angle categories in 1988 and 2015. Values are in hectares.

	Slope angle class							
	≤ 15°		16°-30°		31°-45°		> 45°	
Land use class	1988	2015	1988	2015	1988	2015	1988	2015
(ScRk) Scree/rocky areas	57.5	43.2	453.6	237.4	472.0	226.4	77.6	55.6
(OGr) Open-turf grasslands	292.0	102.0	1159.6	962.3	1014.3	1048.1	135.8	181.3
(CGr) Closed-turf grasslands	764.6	610.3	2095.7	1863.5	1563.5	1564.6	315.4	273.0
(DGr) Dense grasslands	127.5	478.8	297.1	904.9	157.7	352.2	19.5	30.6
(Wd) Wooded areas	68.9	68.4	153.6	195.3	43.5	69.0	6.3	8.8

Fig. 1. Location of the study area (indicated with a small square in the map in the upper right corner) and surface area of grasslands considered in the study (dark gray area), inside the "Monti Sibillini" National Park.

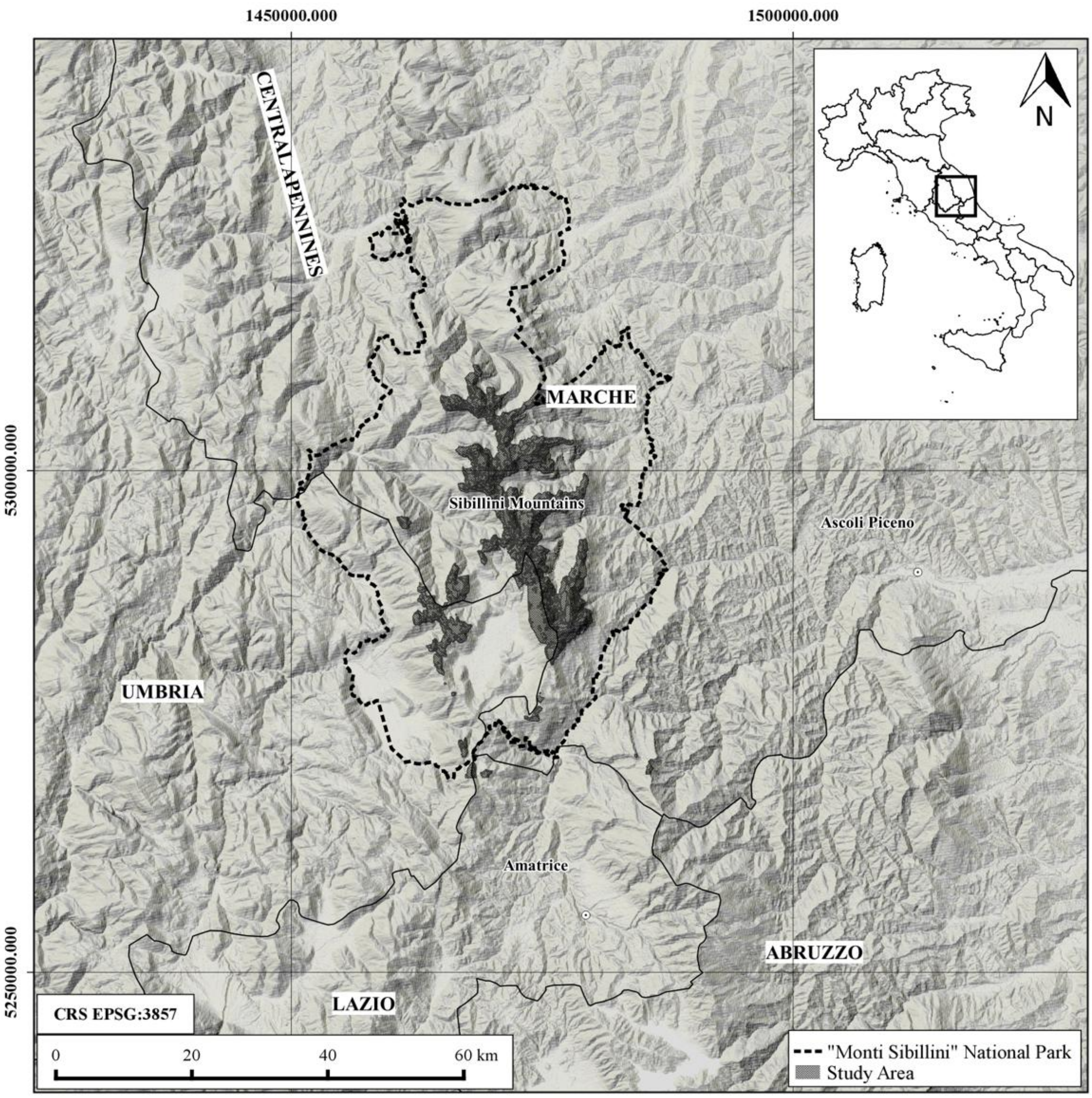

Fig. 2. Flow chart describing the data processing and analysis performed in this study

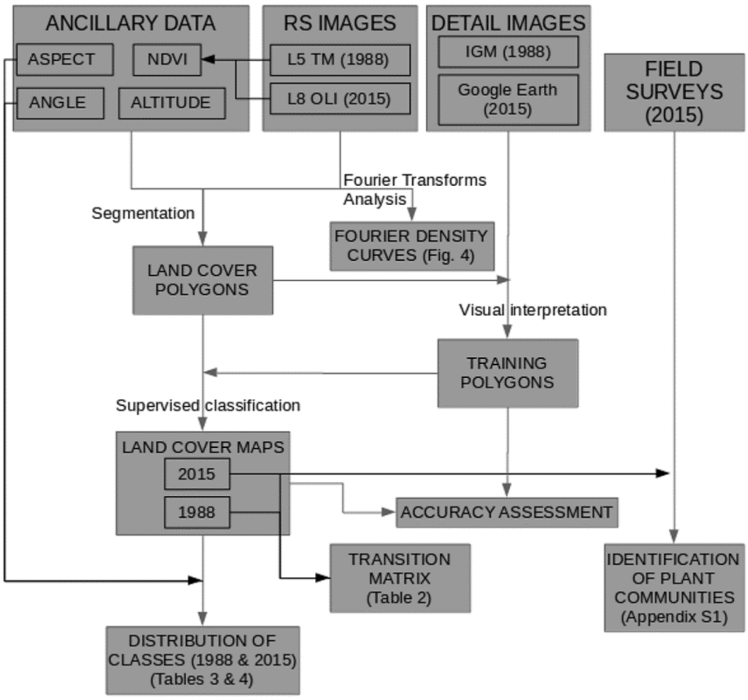

Fig. 3. Example of the fine resolution images used for visual interpretation of training polygons: historical aerial photographs from 1988 provided by IGM (left), and VHR satellite images from Google Earth (right). This example shows how vegetation cover in rocky areas/screes increased from 1988 to 2015.

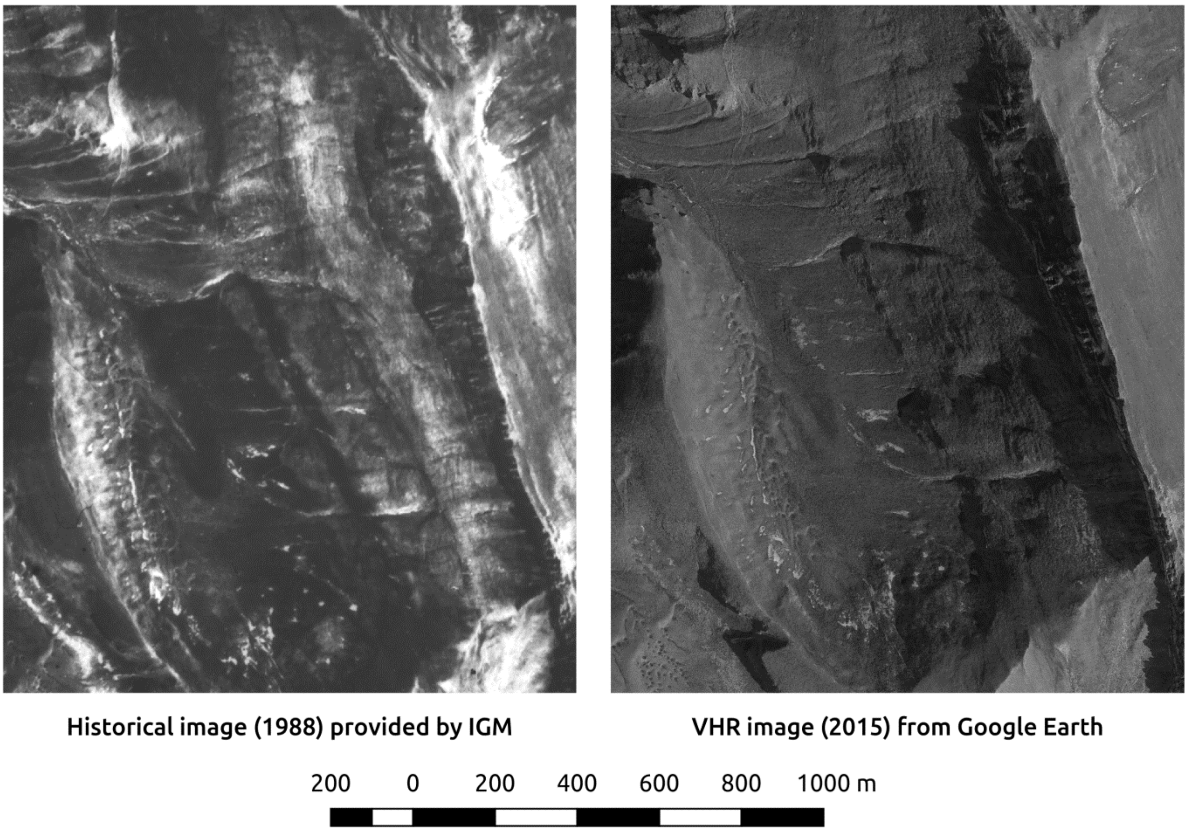

Fig. 4. Curves of probability density function of vegetation cover distribution in 1988 and 2015.