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Multi-scale analysis of alpine landscapes with different intensities of abandonment reveals similar spatial pattern changes: Implications for habitat conservation

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ABSTRACT

The abandonment of traditional anthropogenic activities is an important driver shaping landscape patterns. Therefore, multi-scale pattern analysis over time is needed to identify appropriate scales for biodiversity conservation and monitoring of abandoned landscapes. We compared spatial and temporal changes in a pair of alpine watersheds in Italy (Cajada and Tovanella), which are similar in size, geo-climatic conditions, and land-use histories; but have had divergent anthropogenic abandonment processes since the 1950s. We hypothesize that this divergence has led to corresponding dissimilarities in multi-scale patterns of landscape change. To examine this hypothesis, we analyzed land cover maps from three years (1954, 1980/83, 2006) and described the changes using transition matrices. For each year and watershed, landscape heterogeneity and a set of class-level metrics (i.e. percentage of the landscape, area-weighted mean patch size, patch density, area-weighted mean shape index, edge density, and aggregation index) were also measured at different scales using random sampling techniques, and the results were summarized by using scalograms. Woodland expansion occurred mainly at the expenses of grasslands, meadows, and shrublands. These changes were greater during the first time-period (1954-80/83) than in the more recent period (1980/83-2006), with a mean annual value that decreased from +5.18 to +1.33 ha/year and from +4.08 to +1.96 ha/year in the abandoned and managed watersheds, respectively. Landscape heterogeneity decreased over time with a similar pattern in both watersheds, which indicates a general process of homogenization. Management regime affected the spatial-scale response of class-level metrics; these metrics showed a variety of multi-scalar responses, which were not always consistent over time and under different management regimes. When considering the response of the indices across spatial-scales for both watersheds, certain historical curves showed a scale break, representing a significant change in the shape and slope of the curve (i.e. scale divergence). The presence of scale breaks in the scalograms can potentially reveal important thresholds for biodiversity. For example, grassland and meadow patch density at small spatial scales (<200 m radius), which was found to be important for protected butterfly species, had a greater reduction over time in the managed watershed when compared to the abandoned watershed. In conclusion, the findings of this study indicate that there is good potential for understanding changes in landscape patterns under different management abandonment regimes by combining spatial and temporal analysis of class-level metrics.

1. Introduction

Anthropogenic activities tend to modify the heterogeneous scales and patterns of natural landscapes (Turner et al., 2013), and therefore a corresponding change in scale and pattern is expected when anthropogenic pressure decreases. Consideration of multiple spatial scales is fundamental to understand spatial complexity (Wu et al., 2011) and to define landscapes through the use of landscape indicators (Lustig et al., 2015). Indeed, the identification of appropriate spatial scales of analysis is a critical step in environmental and biodiversity monitoring (Mairota et al., 2015). The effect of changes in grain size (i.e. spatial resolution of data; Turner et al., 1989a) have been widely examined (Frazier, 2016), however there

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is still the need to further analyze the effect of extent (i.e. size of the study area; Turner et al., 1989a) on landscape indicators (Lustig et al., 2015; Siímovaí and Gdulovaí, 2012), and to develop an improved understanding of the implications of these changes.

Application of the findings of multi-scale assessments into practical management actions is recognized as an outstanding challenge (Nash et al., 2014). For example, given that landscapes tend to show distinct patterns at different spatial scales, the findings from a single-scale analysis may often be overly reductive (Zurlini and Girardin, 2008). In addition, not all metrics used to quantify landscape pattern respond consistently at different spatial scales. Metrics can be classified depending on whether or not their response is consistent (Wu, 2004), and not all metrics have shown consistent responses among different studies (Siímovaí and Gdulovaí, 2012). Research on the response of landscape metrics to changes in spatial scale, in particular when coupled with temporal changes under different anthropogenic pressure intensity, can help shed light on impacts to biodiversity (e.g., Frate et al., 2015; Riiters et al., 1997). In this context, a comparison of pattern metrics between different areas will be most valid when the spatial extent is the same and the proportion of land-use categories are approximately equal (Baldwin et al., 2004; Remmel and Fortin, 2013; Turner et al., 2001). Furthermore, coupling the analysis of changes in spatial scale to different time periods may provide further indications on the variable responses of landscape metrics, and what this variability implies about ecological impacts.

Traditional agricultural and forestry practices have altered natural heterogeneous landscapes in many rural mountain areas, resulting in a complex mosaic of sparse open areas and woodland patches. Conversely, many landscapes around the world are now changing again as a consequence of the abandonment of these practices (e.g., Haddaway et al., 2014; Mukul and Herbohn, 2016; Navarro and Pereira, 2012). Management abandonment in mountain landscapes leads to natural succession processes, which typically results in shrub and woodland encroachment (e.g., Chemini and Rizzoli, 2003; Dullinger et al., 2003; MacDonald et al., 2000).

In a review of studies on the impacts of rural abandonment, Sitzia et al. (2010) found general trends towards an increase in size and number of woodland patches and a decrease of open semi-natural habitats linked to anthropogenic activities (e.g., meadows and pastures). The ecological consequences of these changes may be either positive or negative, depending on the geographic and economic context, and the spatial-scale of analysis. Tree encroachment usually results in a simplification and homogenization of these landscapes (Bracchetti et al., 2012), with a decrease in landscape diversity and a reduction in complex mosaics (Frate and Carranza, 2013; Frate et al., 2014; Geri et al., 2010 Frate et al., 2014; Geri et al., 2010). Another potential consequence is the reduction in ecological connectivity across open semi-natural habitats, such as meadows and pasturelands (Sitzia and Trentanovi, 2011).

Analyses of landscape pattern change related to land abandonment typically consider a single scale and assume a dichotomous representation of the landscape by focusing on forest habitats (Otero et al., 2015). However, a broader focus (i.e. studying changes of different land covers) can provide a better understanding of the implications on biodiversity conservation. Analyzing landscape pattern change at various spatial scales also enables a better differentiation of landscapes with different management regimes (Garcia-Feced et al., 2010), and can also help expand our understanding of the complex patterns resulting from land abandonment (Frate et al., 2014). Making comparisons between different landscapes with similar spatial dimensions and geographic conditions, but with different management, would be beneficial in multi-scale analysis (e.g., Bracchetti et al., 2012; Martinez del Castillo et al., 2015; Pan et al., 1999). However much of the previous research on this topic has either focused only on a single study site, has compared sites with very different landscapes (e.g. in terms of area/characteristics), or has not considered landscape metrics (e.g., Beilin et al., 2014; Hall et al., 2012; Tasser et al., 2007).

Changes in landscape composition and configuration due to management abandonment can affect biodiversity both positively and negatively (Queiroz et al., 2014). Navarro and Pereira (2012) highlighted the likely positive effects derived from forest expansion due to farmland abandonment. For example, they suggest that reduced anthropogenic pressure and forest restoration could favor approximately 60 bird, 24 mammal and 26 invertebrate species in Europe. Furthermore, forest expansion increases the area of land suitable for forest species, as some are shade-tolerant plants (Carranza et al., 2012), as well as for some vertebrate species (Bracchetti et al., 2012). Nevertheless, species well suited to semi-natural open habitats may be negatively impacted by the expansion of forests. For example, in the Central Massif region of France, the abundance of open-habitat adapted birds which are of conservation concern decreased in the vicinity of forest edges (Fonderflick et al., 2013). Little attention has been given to landscape features' configuration in relation to their possible suitability as habitat for wildlife within the context of forest expansion (Bowen et al., 2007). It is understood however that shape irregularity metrics correlate with wildlife and vegetation diversity (Carranza et al., 2012; Saura et al., 2008). Yet, analyzing how landscapes change over different spatial extents can better inform scientists about the possible effects of anthropogenic activities on plants and wildlife (Holland et al., 2004; Morelli et al., 2013; Schindler et al., 2013).

The objective of this study is to examine how landscape metrics vary at different scales. Specifically, we aim to analyze spatio-temporal changes in complex landscapes with high biodiversity, which have had different amounts of recent anthropogenic pressure. We examined two forested watersheds with similar sizes and environmental conditions, but with differing management intensity: (1) low-intensity management (gradual abandonment), and (2) no management (abrupt abandonment due to forestry and pasture cessation). We hypothesize that (i) the loss of open habitat types (e.g. grasslands and meadows) due to woodland encroachment will be greater in the abandoned watershed; (ii) management abandonment will smooth the disturbance re-scaling and re-shaping effects occurring in the historic managed watershed, and we should expect a shift from a landscape pattern with pronounced scale-breaks in contrast to a landscape with linear scaling relations (Frate et al., 2014); (iii) after a strong initial divergence between patterns and scales, the differences in landscape metrics between the two areas will tend to disappear; and (iv) landscape metrics for grasslands and meadows, and shrublands will show similar responses, but differing from that observed for woodlands within the same watershed. To our knowledge, this is the first study investigating the responses of landscape indicators to changes in spatial scale coupled with temporal and management regime changes. These changes in landscape pattern at different scales can potentially have effects for habitat and species; therefore we discuss the results of this study in the context of their implications for biodiversity conservation.

2. Materials and methods

2.1. Study area

This study is based in the Tovanella and Cajada watersheds (1040 ha each), located within the south-eastern Alps in the Veneto Region of Italy, in the Alpine biogeographical region (Fig. 1). The two watersheds are located less than 6 km from each other. The climate is temperate-continental, typical of the south-eastern

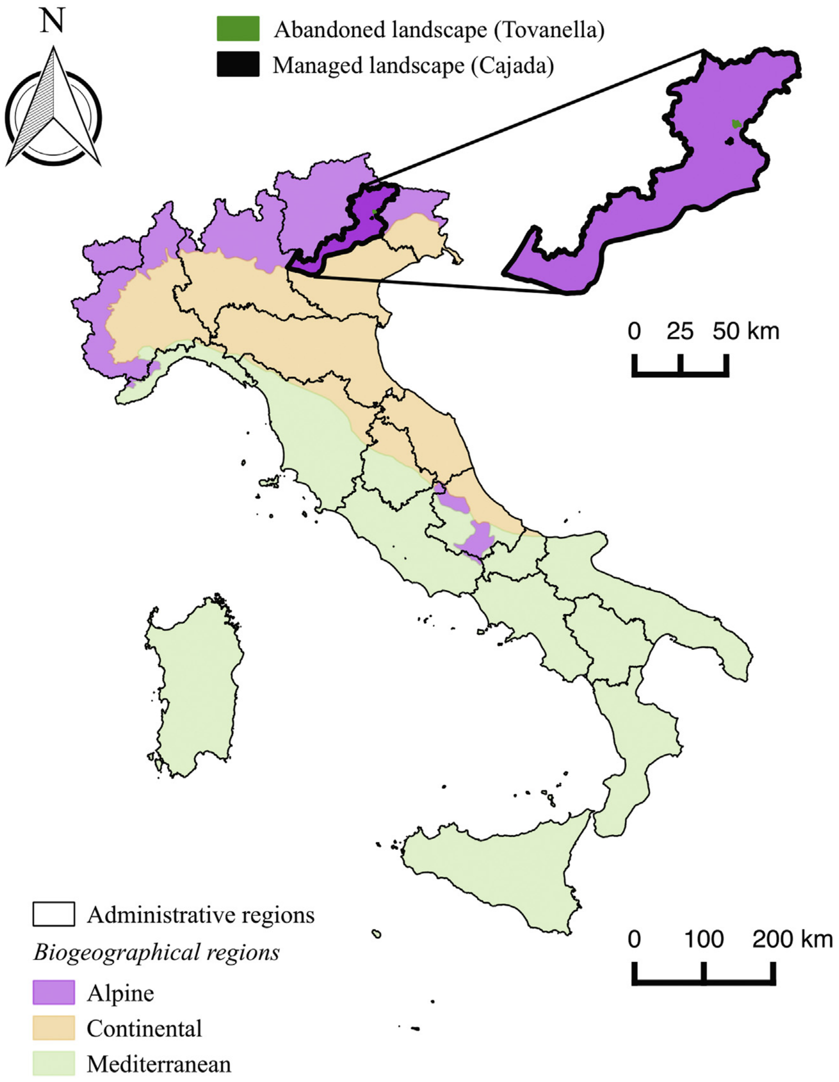

Fig. 1. Study area location within the Alpine biogeographical region in Veneto (north-eastern Italy).

alpine region, with relatively high mean annual precipitation (1300–1500 mm year^{-1}) concentrated in May-June and October-November, and a mean annual temperature of 7.2 °C with harsh winters. The main rock substrate is dolomitic limestone formed during the secondary and tertiary. Both watersheds have an altitudinal range from approximately 550 to 2500 m a.s.l.

While the historical management regime of both watersheds is characterized by traditional forestry and pastoral activities, their management trajectories diverged after the 1950s. The forests in both watersheds were heavily logged by the Republic of Venice between the 15th and 17th centuries, which relied on timber from this region for ships construction. More recently, timber extraction carried out between 1943 and 1953 resulted in a very low growing stock (<200 m3 ha-1) in these forests (Susmel, 1958). In addition to forestry, the pastures and meadows of both watersheds were important for the pastoral activities of local communities. For example, in Tovanella in the second half of the 14th century, around one hundred cattle and thousands of sheep and goats grazed the pastures and meadows during the summer season (Viola et al., 2008).

After the 1950s, the management approach of the two watersheds diverged significantly. In Tovanella, forestry and pasture activities were abruptly abandoned, and when the area became an ‘Oriented Biogenetic Nature Reserve’ in 1971, all anthropogenic activities were legally banned (Viola et al., 2008). By contrast, in Cajada forestry has continued at a low intensity (i.e. near-to-nature silviculture, applying group shelterwood system) and pastoral activities have continued at gradually decreasing rate, up until the present time period (Cassol, 1996).

Both watersheds are in the Natura 2000 network, Tovanella falls under the ‘Site of Community Importance (SCI): Val Tovanella – Bosconero – IT3230031’ and the ‘Special Protection Area (SPA): Dolomiti del Cadore e Comelico – IT3230089’, while Cajada falls under the SPA and SCI ‘Dolomiti Feltrine e Bellunesi – IT3230083’. The establishment of this protection underlines the importance of both areas as habitats for biodiversity conservation (Table 1) and species of Community interest (Ente Parco Nazionale Dolomiti Bellunesi, 2009; Lasen et al., 2008). Both watersheds are dominated by woodlands, consisting mainly of beech and fir, which were classified as *Asperulo-Fagetum* beech forests (code: 9130), following the Habitats Directive (Directive 92/43/EEC) classification. Calcareous rocky slopes with chasmophytic vegetation (code: 8210) are widespread in the higher altitudes, while brush-land areas of *Pinus mugo* and *Rhododendron hirsutum* (code: 4070) are common in Tovanella. Grassland habitats are less prevalent, and the most common categories are alpine and subalpine calcareous grasslands (code: 6170). Several flora and fauna species of European interest are present, including those related to forests (e.g.,

Table 1
Habitats of Annex I of the Habitats Directive (Directive 92/43/EEC) and related land-cover classes in the two watersheds (Tovanella and Cajada).

Cover class	Habitats	Watershed
Alpine grasslands	6210 Semi-natural dry grasslands and scrubland facies on calcareous substrates (*Festuco-Brometalia*) (* important orchid sites)	Cajada
Alpine grasslands, or Grasslands and meadows.	6170 Alpine and subalpine calcareous grassland	Cajada and Tovanella
Grasslands and meadows	6430 Hydrophilous tall herb fringe communities of plains and of the montane to alpine levels	Cajada and Tovanella
	6520 Mountain hay meadows	Cajada
	7230 Alkaline fens	Cajada
Shrublands or Woodlands	4060 Alpine and Boreal heaths	Tovanella
Shrublands	4070 * Bushes with *Pinus mugo* and *Rhododendron hirsutum* (*Mugo-Rhododendretum hirsuti*)	Cajada and Tovanella
Rocks and screes	8120 Calcareous and calcshist screes of the montane to alpine levels (*Thlaspietea rotundifolii*)	Cajada and Tovanella
	8210 Calcareous rocky slopes with chasmophytic vegetation	Cajada and Tovanella
Woodlands	9130 *Asperulo-Fagetum* beech forests	Cajada and Tovanella
	9140 Medio-European subalpine beech woods with *Acer* and *Rumex arifolius*	Cajada and Tovanella
	9150 Medio-European limestone beech forests of the *Cephalanthero-Fagion*	Tovanella
	9180 * *Tilio-Acerion* forests of slopes, screes and ravines	Tovanella
	91K0 Illyrian *Fagus sylvatica* forests (*Aremonio-Fagion*)	Cajada
	9420 Alpine *Larix decidua* and/or *Pinus cembra* forests	Tovanella
	9530 * (Sub-)Mediterranean pine forests with endemic black pine	Cajada and Tovanella

*Priority habitats under the Habitats Directive (Directive 92/43/EEC).

Glaucidium passerinum L., *Cypripedium calceolus* L.), open grasslands (e.g., *Parnassius mnemosyne* L., *P. apollo* L.), ecotones and mosaics (e.g., *Tetrao tetrix* L., *Lanius collurio* L.), and rocky slopes and screes (e.g., *Campanula morettiana* Rchb., *Physoplexis comosa* (L.) Schur.) (Argenti and Lasen, 2008; Ente Parco Nazionale Dolomiti Bellunesi, 2009; Hardersen and Dal Cortivo, 2008; Mezzavilla et al., 2008).

2.2. Land cover maps

To characterize the changes in the two watersheds, we used aerial photographs for the years 1954 (flight from the Italian Military Geographical Institute-GAI), 1980 (flight from Aerofoto Consult), 1983 (flight from Rossi srl), and 2006 (flight from Regione Veneto). While the photos of the latter year were already orthorectified and georeferenced (TIFF and ECW images), those of the former years were acquired in paper format, and then georeferenced and digitalized in a Geographic Information System.

The aerial photos from 1980 (for Tovanella) and 1983 (for Cajada) were scanned as TIFF images with a resolution of 800 dpi, and the 1954 photos were scanned at 1200 dpi. The resolution for these photos were selected based on the clarity of the output, as settings at higher resolutions resulted in grainy images All photos were orthorectified using the ErMapper 7.0 software with a 25 m Digital Terrain Model (DTM). For the images from 1980 and 1983, a minimum of 10 ground control points were used for orthorectification, with a resulting average root-mean-square error (RMSE) of 1.15 and 0.81 m for 1980 and 1983, respectively. As the calibration certificate was missing for 1954 images, the spline method was applied by using an average of 20 points for each photo from previously georefenced images.

To produce land-cover maps, a manual classification process was carried out. A classification grid with a mapping unit of 250 m^2 (15.8 × 15.8 m) at a fixed scale of 1:5000 was used. This resolution enabled consideration of a 1 mm minimal possible mapping accuracy, which corresponds to 5 m at a scale 1:5000 (Sitzia and Trentanovi, 2011). Six cover classes were used: forests, grasslands and meadows, shrublands, bare rock, buildings, and alpine grasslands (i.e., grasslands above the forest line). We selected these cover classes as they represent the most important habitats for the plant and wildlife species of interest for this study.

2.3. Data analysis

2.3.1. Landscape scale dynamics

To analyse the main land-cover changes between the periods under investigation (1954–1980/83 and 1980/83-2006), we built specific transition matrices using the 'combine tool' of the GIS software 'ArcGIS 10.1' (ESRI, 2011). The transition matrices describe the temporal dynamics of the analysed watersheds over each time period (1954–1980/83 and 1980/83-2006). Each cell of the matrix represents the hectares belonging to one land cover class in a given year that has changed into another land cover class. The diagonal cells represent the unchanged (or persistence) area. For each period we calculated the mean annual change (ha/year).

2.3.2. Spatial pattern analysis at multiple scales

To quantify the spatial pattern changes across scales, a set of landscape indices were selected and computed using the software 'FRAGSTATS 4.2' (McGarigal et al., 2012). These metrics were calculated at the landscape level (taking into account all cover types together) and at the class level considering three cover types (woodlands, grasslands and meadows, and shrublands) that host habitats (Table 1) and species of Community interest. Landscape and class pattern indices were selected due to their ability to relate the observed landscape pattern to the underlying ecological processes. Furthermore, these metrics were previously reported as ecologically meaningful and have proven useful for describing and comparing the spatial structure of abandoned land (Algeet-Abarquero et al., 2015; Frate et al., 2014; Otero et al., 2015; Schindler et al., 2013).

The Shannon diversity index (SHDI) was used to define landscape heterogeneity (Díaz-Varela et al., 2009b) and to detect possible homogenization processes as a consequence of land management abandonment. Class level indices, as a percentage of the landscape (PLAND), area-weighted mean patch size (AREA_AM), and patch density (PD) were analyzed to assess changes in the extent, patch size, and spatial distribution of each cover type. As the active management of forests and pastures should allow the maintenance of open habitats, management abandonment should promote a progressive reduction in the number and size of these habitats (e.g. Rocchini et al., 2006). Area-weighted mean shape index (SHAPE_AM), edge density (ED), and aggregation index (AI) were computed to observe changes in shape and connectivity of each cover type. Indeed, forest expansion usually leads to the oversimplification of patch structure (i.e. more regular) and to a decrease of open habitat connectivity (Sitzia et al., 2010). The description of the pattern metrics used in the study along with their respective variation range is provided in McGarigal et al. (2012).

Since landscape patterns and processes are scale-dependent (e.g., Turner et al., 1989b; Wu, 2004; Wu et al., 2002), and landscape indices vary with landscape extent (Frate et al., 2014; Gardner et al., 1987; Wu et al., 2002), we quantified landscape pattern over time at multiple scales. There are several techniques available for landscape analysis at multiple scales, including nested quadrat design (e.g. Turner et al., 1989b), diagonal expansion of the study area (e.g., Frate et al., 2014; Wu, 2004; Wu et al., 2002), step-wise expansion of the original area (e.g. Baldwin et al., 2004), grid-based sampling design (Sitzia et al., 2014) and moving window analysis (e.g. Díaz-Varela et al., 2009a). Here we adopted a sampling strategy (Carranza et al., 2014; Ramezani et al., 2013; Stehman, 2012), which provides a good method of describing the relationship between land cover and spatial pattern changes (Carranza et al., 2014; Díaz-Varela et al., 2009b), which is crucial for the correct interpretation of on-going landscape processes (Frate et al., 2014; Hargis et al., 1998). Moreover, a sample-based approach allows for the production of statistically valid estimates of class and landscape metrics at different scales (Hassett et al., 2012). In particular we quantified landscape pattern change using random sampling techniques on the multi-temporal maps. One hundred points were randomly distributed across the land cover map, and circular windows with different radii at increasing dimensions were used for each point. Selected radii were 100, 200, 300, 500, 700 and 1000 m. In this way, each window defined a series of sub-landscapes on which the selected pattern metrics were computed. Scalograms for all indices were built by plotting index value against spatial scales, and a simple spline regression model (with the relative bootstrapped 95% confidence interval) was fitted to evaluate the response of the indices (the shape and the slope of the regression curve) and their changes over time (e.g. small-scale vs. large-scale changes).

3. Results

3.1. Landscape change

The temporal maps of the two watersheds (Fig. 2) indicate that significant changes have occurred in the whole area over the last 50 years. In 1954, woodlands were the dominant land cover for both watersheds, followed by shrublands, and grasslands and meadows. During the first time span in Cajada, a steady decrease in grasslands and meadows (from 8% to 4%), and shrublands (from 11% to 7%) was observed, along with an increase in woodlands (from 63% to 76%). The corresponding mean annual change was −1.37, −1.36 and +4.08 ha/year, respectively. During the second time-span, grasslands and meadows, and shrublands showed a slight decrease (from 4% to 3% and from 7% to 6%) whereas woodlands increased from 76% to 81%. However, the mean annual change was lower (−0.27, −0.49, and +1.96 ha/year, respectively) than during the previous period. In Tovanella watershed, from 1954 to 1980 the area covered by grasslands and meadows decreased from 4% to 1%, shrublands decreased from 28% to 21%, and woodlands increased from 49% to 62%. This corresponds to a mean annual change of −1.09 ha/year for grasslands and meadows, −2.89 ha/year for shrublands, and +5.18 ha/year for woodlands. In the period 1980–2003, grasslands and meadows almost disappeared, shrublands decreased from 21% to 19%, and woodlands expanded from 62% to 65%. The mean annual change was lower compared to the period 1954–1980, corresponding to −0.16 ha/year for grasslands and meadows, −0.24 ha/year for shrublands, and +1.33 ha/year for woodlands.

The transition matrices (Table 2 and Table 3) and the relative maps of change (Fig. 3) show that during the first time period both watersheds had the same percentage of stable (25%) and dynamic areas (75%). In the last time-period, these values differed slightly between watersheds, with an unchanged area of 11% for Cajada and 10% for Tovanella. In the period 1954–1980/1983 woodlands and bare rock showed high values of persistence for both watersheds. Despite these similarities, differences between the two watersheds were observed for other land-cover categories. In Cajada, shrubland was the category most affected by change (mainly conversion into woodlands), whereas in Tovanella, grasslands and meadows showed the greatest change (95% reduction from the original cover). In the second time-period, smaller changes were observed in both watersheds. In Cajada, the largest change was recorded for shrublands, and for grasslands and meadows; while in Tovanella grasslands and meadows, and alpine grasslands were less persistent.

3.2. Multi-scale analysis of landscape change

The temporal analysis revealed different patterns emerging at specific scales in the two watersheds. The scalograms of Shannon Diversity Index (SDI) revealed differences in heterogeneity across scales and over time (Fig. 4). Overall, Tovanella had higher heterogeneity values than Cajada. The SDI was almost always higher in 1954, and significantly decreased in the recent years of analysis. However, it is interesting to note that in Tovanella at very local scales there were no differences in the SDI. When looking at the response of the metrics across scales for both areas, the historical curves showed a scale break representing a significant change of the shape and slope of the curve ("scale divergence" Wu et al., 2000). After this point the curves were relatively consistent, and further increasing the scale did not cause substantial variations in the metric value. Conversely, the 2006 curves did not show such a break, but they grew slightly in a linear fashion.

The scalograms describing the class pattern metrics (Figs. 5–7) showed specific responses to changing scale that varied among cover classes, year, and between the study areas. According to the land-cover classes, two main trends can be distinguished for percentage of the landscape (PLAND) across scales: (i) a scale break for grasslands and meadows, and shrublands and (ii) a steady-linear response curve for woodlands. The percentage of grasslands and meadows in 1954 was very similar between Tovanella and Cajada at the local scale, however at larger scales PLAND was higher in Cajada. In 1954, PLAND was characterized by a curve with a sharp scale break (close to 400 m radius) after which the curve was relatively constant, while in the more recent years of analysis PLAND tended to be stable across scales. This means that in 1954 grasslands and meadows formed a significant element of the small-scale patterns, while more recently they were close to disappearing at the local scale as well. Shrublands had a similar response compared to grasslands and meadows, except for in Tovanella where they were still an important factor in landscape heterogeneity. At local scales, shrublands were extensive in all compared years. In 1954, the percentage of shrublands decreased until reaching a break point (at 400 m radius) where the values tended to become constant. In 1980 and 2006, this percentage decreased faster across scales and the break point occurred at larger scales. This indicates that shrublands have become less prominent, while conversely woodlands did not show any sort of scale break, but were dominant at all scales.

Concerning patch density (PD), all classes showed a similar scalar response characterized by a strong linear-decay pattern at very small scales that tended to stabilize at small scales (200–400 m radius). Thus, the small-scale patchiness was replaced by large-

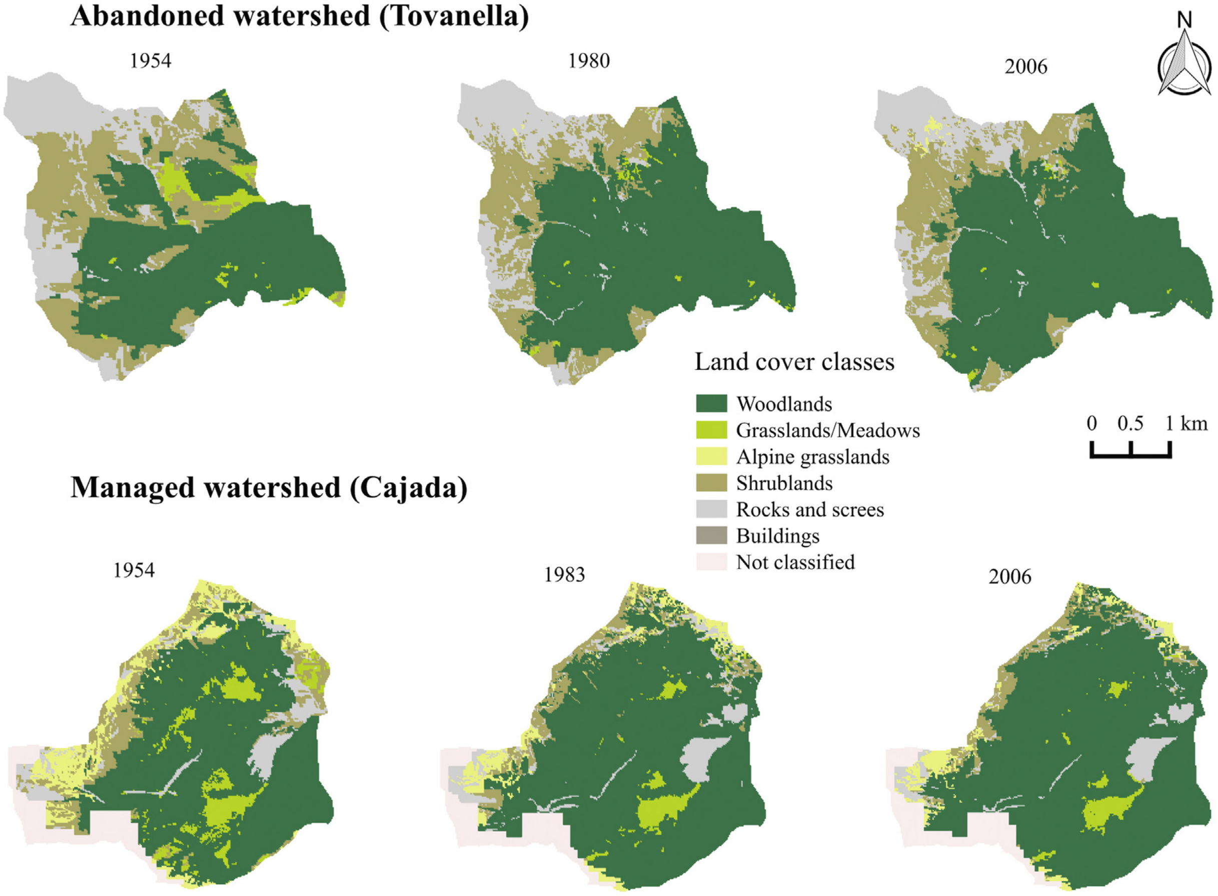

Fig. 2. Land-cover maps of the abandoned (above) and managed (below) watersheds for the different analyzed years (1954, 1980/83, 2006).

Table 2
Transition matrix of the abandoned watershed (Tovanella) for the two time-periods (1954–1980 and 1980–2006). Areas in bold did not change land cover class.

(a) Time-period 1954–1980

		1980						
		Woodlands	Grasslands and meadows	Shrublands	Rocks and screes	Buildings	Alpine grasslands	Total 1954
1954	Woodlands	**487.35**	2.4	11.5	4.90	0.00	0.00	506.15 (49)
	Grasslands and meadows	31.18	**2.03**	3.40	0.90	0.00	0.00	37.50 (4)
	Shrublands	108.65	3.95	**153.15**	25.00	0.00	0.1	290.85 (28)
	Rocks and screes	13.58	0.90	47.63	**143.45**	0.00	0.68	206.23 (20)
	Buildings	0.00	0.00	0.00	0.00	**0.00**	0.00	0.00 (0)
	Alpine grasslands	0.00	0.00	0.00	0.00	0.00	**0.00**	0.00 (0)
	Total 1980	640.75 (62)	9.28 (1)	215.68 (21)	174.25 (17)	0.00 (0)	0.78 (0)	1040.73 (100)

(b) Time-period 1980–2006

		2006						
		Woodlands	Grasslands and meadows	Shrublands	Rocks and screes	Buildings	Alpine grasslands	Total 1980
1980	Woodlands	**632.3**	0.43	6.35	1.68	0.00	0.00	640.75 (62)
	Grasslands and meadows	5.13	**2.33**	1.68	0.15	0.00	0.00	9.28 (1)
	Shrublands	32.43	0.73	**166.70**	15.63	0.00	0.2	215.68 (21)
	Rocks and screes	5.38	1.60	27.35	**135.30**	0.00	4.63	174.25 (17)
	Buildings	0.00	0.00	0.00	0.00	**0.00**	0.05	0.05 (0)
	Alpine grasslands	0.00	0.00	0.13	0.60	0.00	**0.05**	0.78 (0)
	Total 2006	675.23 (65)	5.08 (0)	202.20 (19)	153.35 (15)	0.00 (0)	4.93 (0)	1040.78 (100)

scale cohesion between patches belonging to a specific class. Over the three years there were no differences in patch density across scales, except for Cajada where both grasslands and meadows, and shrublands, had lower patch density values in more recent years.

The area-weighted mean patch area (AREA_MN) showed different scaling relations for grasslands and meadows, and shrublands in the two watersheds. For Tovanella in the 1954 scalogram, grasslands and meadows exhibited a linear-increase trend, whereas the 1980 and 2006 scalograms showed flat curves with values close to zero. This pattern was very similar to that of Cajada shrublands. Conversely, both Tovanella shrublands and Cajada grasslands and meadows had a linear-increase response curve with the 1954

Table 3
Transition matrix of the managed watershed (Cajada) for the two time-periods (1954–1980 and 1983–2006).

(a) Time-period 1954–1983

		1983						
		Woodlands	Grasslands and meadows	Shrublands	Rocks and screes	Buildings	Alpine grasslands	Total 1954
1954	Woodlands	**556.73**	5.88	5.40	13.75	0.00	4.65	586.40 (63)
	Grasslands and meadows	43.45	**27.25**	2.75	0.75	0.05	0.03	74.28 (8)
	Shrublands	62.33	1.10	**27.20**	6.68	0.00	7.68	104.98 (11)
	Rocks and screes	14.98	0.20	5.78	**47.88**	0.00	2.25	71.08 (8)
	Buildings	0.00	0.00	0.00	0.00	**0.00**	0.00	0.00 (0)
	Alpine grasslands	27.30	0.00	24.50	5.88	0.00	**36.88**	94.55 (10)
	Total 1983	704.78 (76)	34.43 (4)	65.63 (7)	74.93 (8)	0.05 (0)	51.48 (6)	931.28 (100)

(b) Time-period 1983–2006

		2006						
		Woodlands	Grasslands and meadows	Shrublands	Rocks and screes	Buildings	Alpine grasslands	Total 1983
1983	Woodlands	**693.08**	2.78	5.18	1.20	0.00	2.55	704.78 (76)
	Grasslands and meadows	9.13	**24.93**	0.35	0.03	0.00	0.00	34.43 (4)
	Shrublands	25.78	0.40	**34.63**	1.10	0.00	3.73	65.63 (7)
	Rocks and screes	9.18	0.20	5.30	**53.43**	0.00	6.83	74.93 (8)
	Buildings	0.00	0.00	0.00	0.00	**0.05**	0.00	0.05 (0)
	Alpine grasslands	12.68	0.00	8.90	2.98	0.00	**26.93**	51.4810)
	Total 2006	749.83 (81)	28.30 (3)	54.35 (6)	58.73 (6)	0.05 (0)	40.03 (4)	931.28 (100)

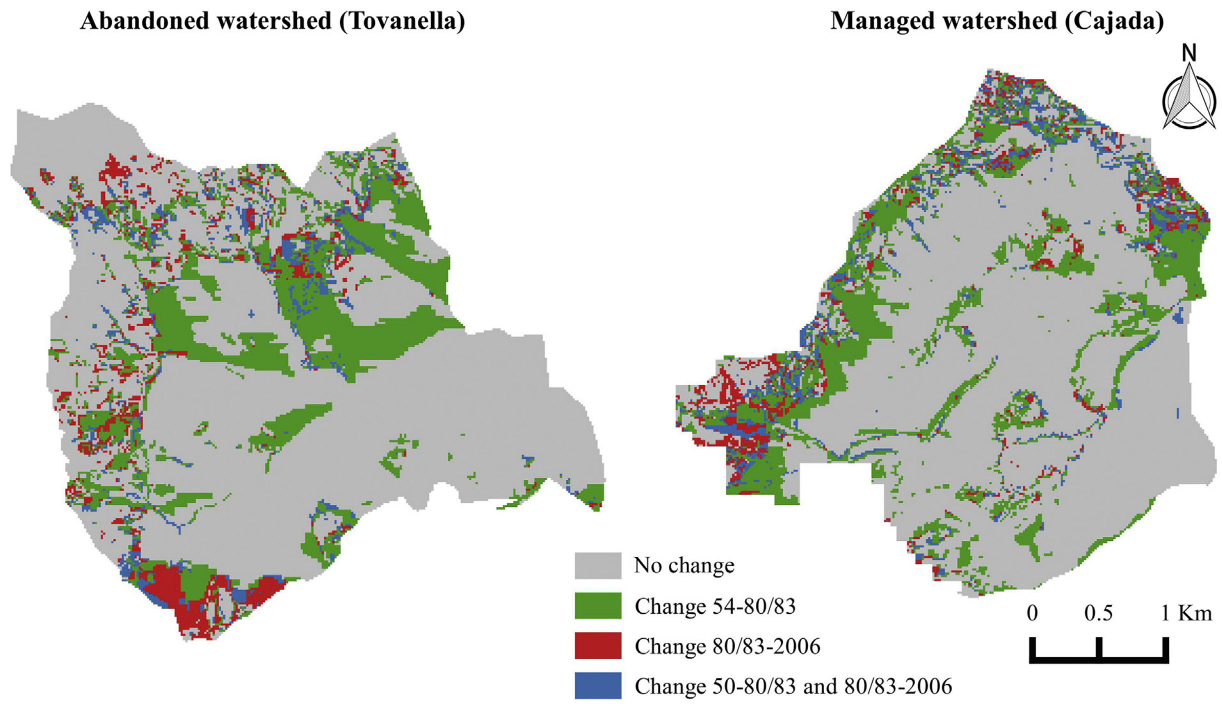

Fig. 3. Maps indicating the areas where land cover changes occurred during the first (1954–1980/83 in green), second (1980/83- 2006 in red) and both (1954–1980/83 and 1980/83 –2006 in blue) time-periods. Abandoned watershed is outlined in the left and managed one in the right. Areas where no changes occurred are also reported (grey). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article).

Cajada grasslands and meadows scalogram presenting a scale break over the 700–800 m radius scale. Overall, 1954 curve had higher AREA_MN values. Woodlands exhibited a power-law increasing trend for both areas, with higher values in more recent years.

The aggregation index (AI) showed different scale response according to the land cover classes under consideration. Grasslands and meadows AI had an erratic response, as the metric seemed not to follow any predictable trend. Conversely, shrublands exhibited a flat scalogram with AI values that were always lower in the two recent dates compared to that of the 1954. Woodlands had a similar scale response but without significant difference over time.

Edge density (ED) showed a similar response for grasslands and meadows, and shrublands presenting scale-breaks at different extents. However, scale-breaks for grasslands and meadows tended to disappear in 1980/83 and 2006 compared to 1954. Furthermore, differences in ED were more evident in Cajada between the first year and the other two years of analysis. Whereas, differences between years for shrublands were more evident in Tovanella than in Cajada. Unexpectedly, the scale-break in Tovanella tended to become more evident in recent years while the opposite occurred in Cajada. A change in woodland edge density was evident at all scales, and higher in 1954 in Cajada, while in Tovanella occurred only between 200 and 600 m radii.

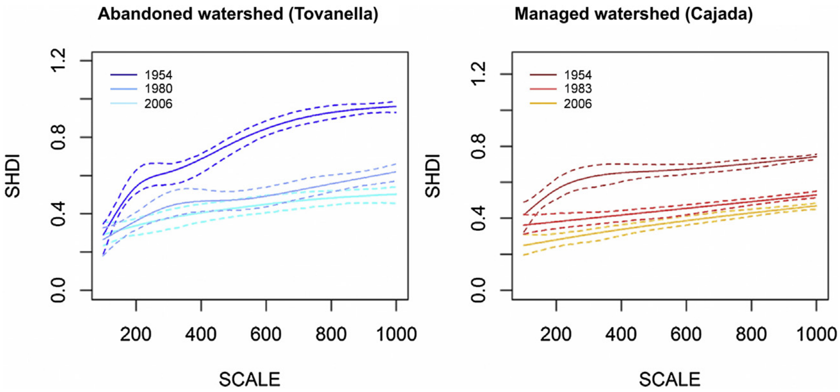

Fig. 4. Shannon Diversity Index (SDI) at different scales for the three years (1954, 1980/83, 2006) in abandoned (left) and managed (right) watersheds.

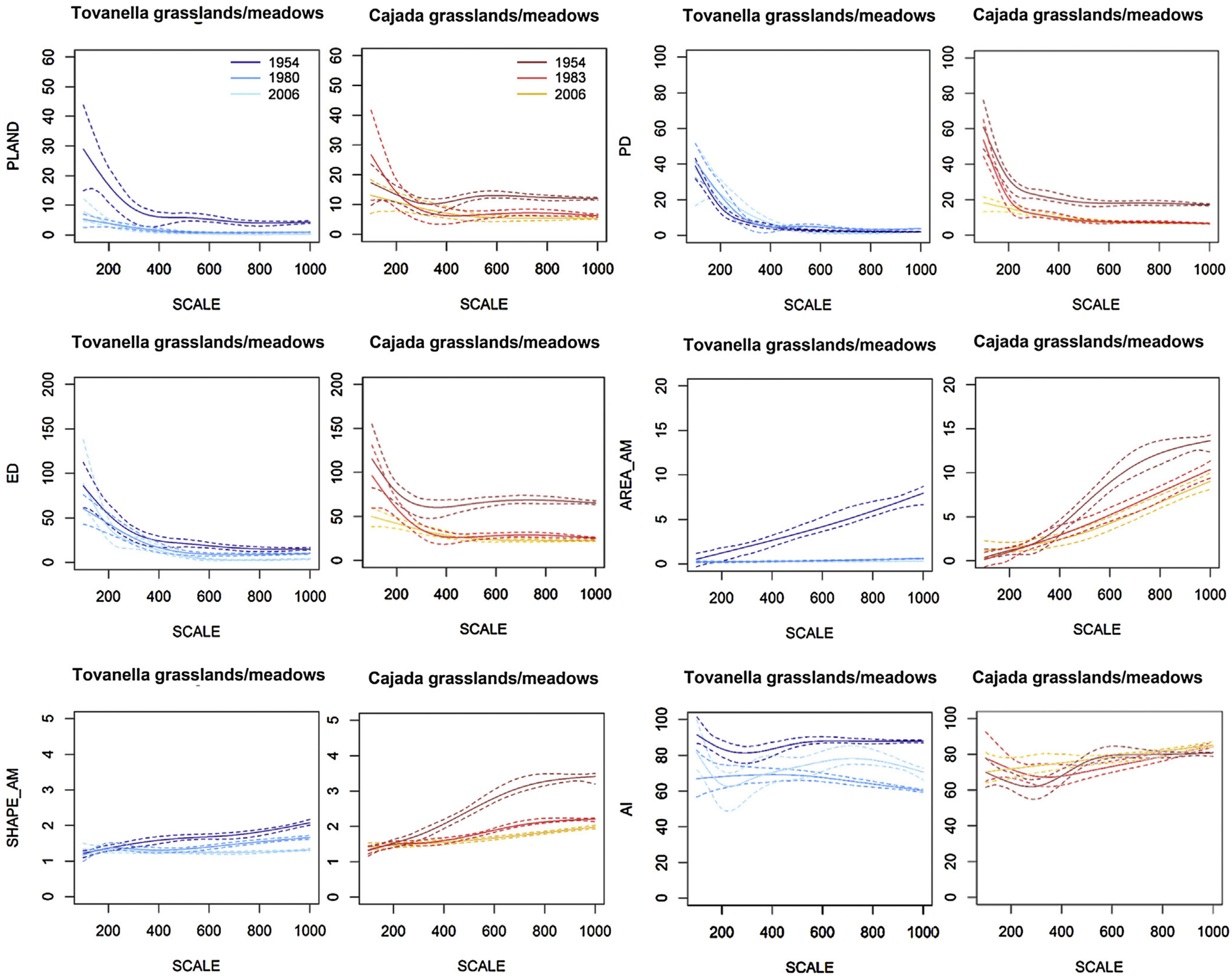

Fig. 5. Multi-scale response of landscape metrics (PLAND, PD, ED, AREA_AM, SHAPE_AM, AI) of grasslands and meadows for the three years in abandoned (left) and managed (right) watersheds.

Area-weighted mean shape index (SHAPE_AM) scalogram for grasslands and meadows, shrublands and woodlands had similar response in Cajada: an increasing trend with spatial extent and decreasing over time except at small scales. Also in Tovanella SHAPE_AM increased with increasing extent. For grasslands and meadows it was higher in 1954, except at small scales, while for shrublands it was lower.

4. Discussion

4.1. Landscape-scale dynamics

In contrast to our hypothesis, similar results were observed in terms of habitat loss regardless of the differences in the management regime between the two watersheds. Woodland area initially

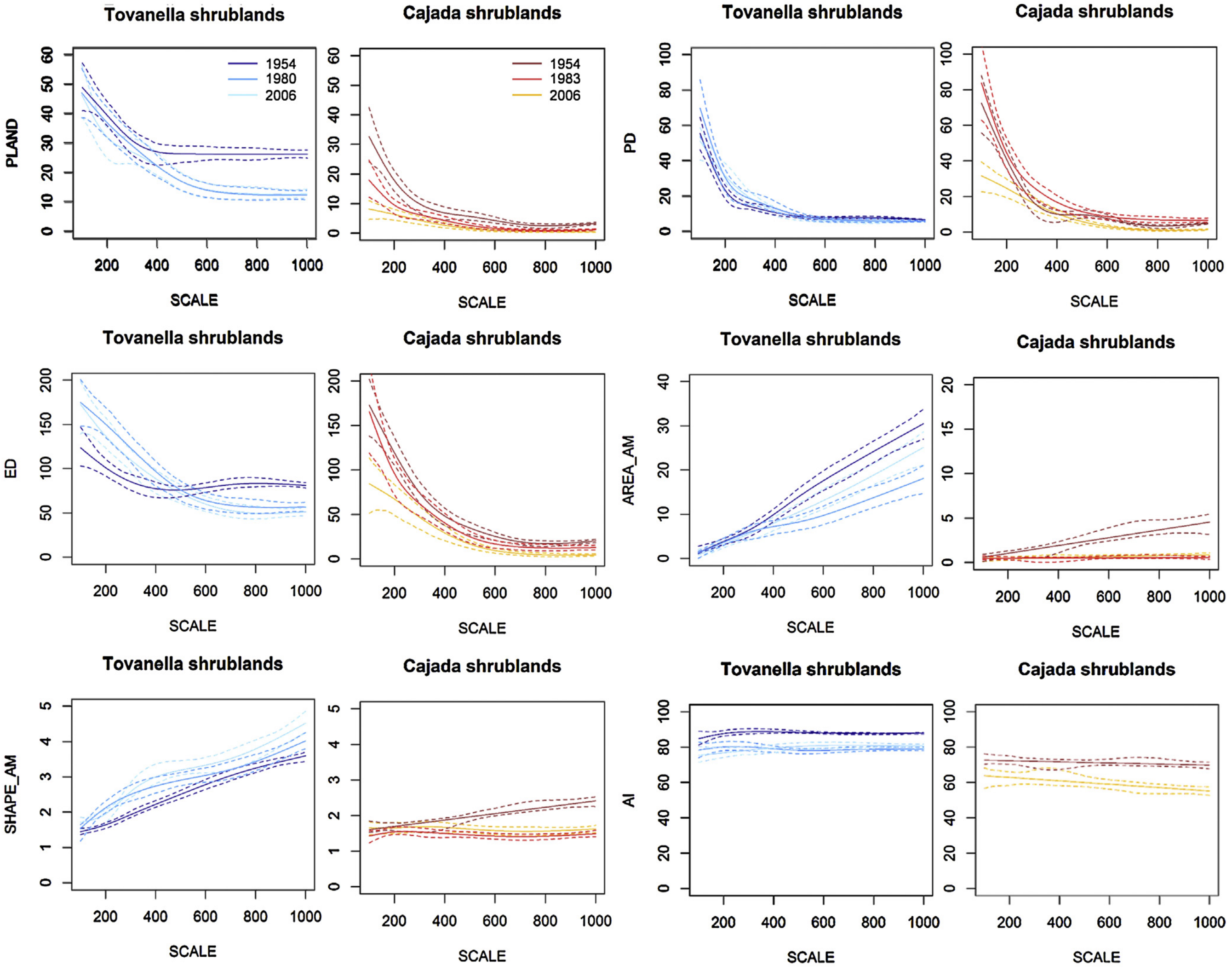

Fig. 6. Multi-scale response of landscape metrics (PLAND, PD, ED, AREA_AM, SHAPE_AM, AI) of shrublands for the three years in abandoned (left) and managed (right) watersheds.

expanded rapidly in the period of 1950-80/83, while in the period 1980/83-2006 the mean annual expansion decreased substantially in both watersheds irrespective of the initial cover. Both watersheds also showed similar trends for grasslands and meadows, and shrublands; which showed a strong initial loss followed by a reduction in the mean annual change over the second period. The pattern is common in the Alps, where the reduction of these land covers has been widely reported as an effect of management abandonment and consequent woodland encroachment (e.g., Orlandi et al., 2016; Sitzia et al., 2010). Climate change has also likely impacted vegetation changes at high altitudes, resulting in an increase in forest cover (Dainese and Sitzia, 2013; Evangelista et al., 2016; Jackson et al., 2016). This trend in cover reduction is likely to have negatively affected species that prefer grasslands and meadows, and shrubland habitats. Furthermore, the various grassland and heath habitats protected under the Habitats Directive found in the two watersheds (i.e. Alpine and Boreal heaths – code: 4060, Alpine and subalpine calcareous grassland – code 6170, Semi-natural dry grasslands and scrubland facies on calcareous substrates (*Festuco-Brometalia*) (* important orchid sites) – code 6210, Hydrophilous tall herb fringe communities of plains and of the montane to alpine levels – code 6430, Mountain hay meadows – code 6520, Alkaline fens – code 7230), are all at least in part dependent on anthropogenic activities (Halada et al., 2011).

It is probable in both watersheds that species linked to these habitats will have moved towards other areas and/or their population will have reduced over time (Pernollet et al., 2015). Indeed, such a trend (i.e. reduction of suitable habitat due to woodland succession) can lead to local extinction of these species (Balmer and Erhardt, 2000; Schlossberg and King, 2009). By contrast, woodland species have probably benefited from these changes (Sirami et al., 2007), as the differences in specific habitat features between the two watersheds have shown an influence on several species (Nascimbene et al., 2013; Sitzia et al., 2015). The analysis of land cover change and related habitat loss over time, as conducted in this study, enables the spatial identification of areas that underwent changes in recent years, which should be preferred areas for restoration actions (Öckinger et al., 2006). However, attention should be given to time lags in specialist species local extinction and woodland specialist colonization (Bagaria et al., 2015).

4.2. *Multi-scale response of landscape change*

Landscape heterogeneity decreased in both watersheds over time, but with slightly different changes across scales. A decrease in landscape heterogeneity was recorded in other parts of the Alps (Kulakowski et al., 2011). In Tovanella (hereafter called abandoned) watershed the decrease in heterogeneity occurred between 1954

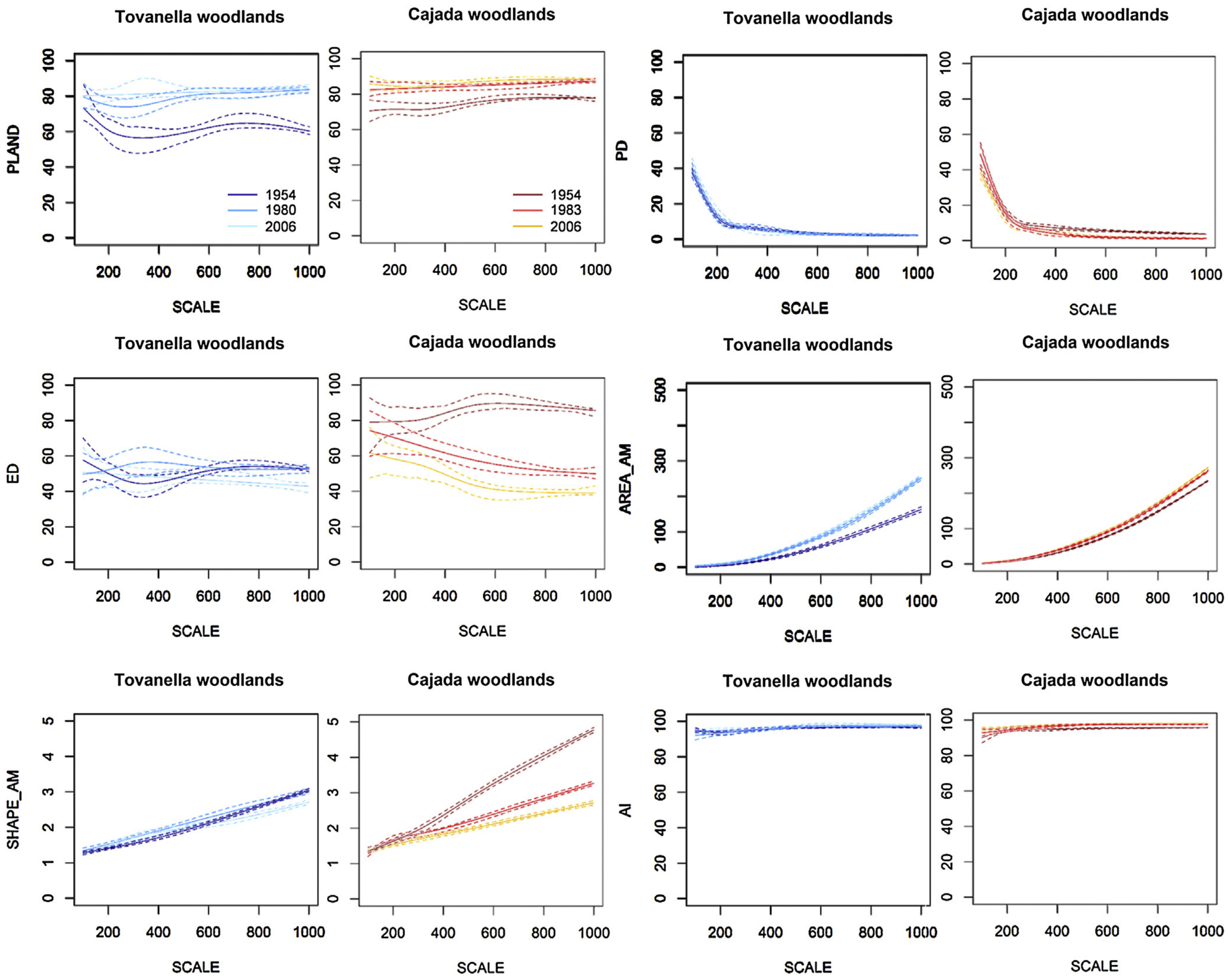

Fig. 7. Multi-scale response of landscape metrics (PLAND, PD, ED, AREA_AM, SHAPE_AM, AI) of forest for the three years in abandoned (left) and managed (right) watersheds.

and 1980, and tended to stabilize between 1980 and 2006. In Cajada (hereafter called managed) watershed the decrease was evident between all the three years. Furthermore, in the abandoned watershed there was a marked gradual incorporation of different patches with increasing scale that was less pronounced in the managed watershed for 1954; namely, the curve tended to flatten at lower scales in the managed watershed. This indicates a higher initial heterogeneity in the abandoned areas than in the managed watershed. However, in both areas the observed pattern indicated a similar presence of mainly continuous woodland cover over time (1980/83, 2006) and scales. Indeed, this demonstrates an important reduction of landscape heterogeneity in both the abandoned and managed watersheds. This was particularly evident in the abandoned watershed at larger extents, and in general the reduction of landscape heterogeneity was greater at larger extents in both watersheds.

Wildlife species have various scales of habitat selection (Ducci et al., 2015; Mayor et al., 2009; Sitzia et al., 2014). Species that are "multi-habitat" specialist (i.e. preferring high heterogeneity at broader scales; Russo, 2007), are likely to have been impacted from the reduction in heterogeneity over time in both watersheds given the reduction in landscape heterogeneity over time. For example, bird species are frequently affected by landscape changes at intermediate extents (Mairota et al., 2015), for which we observed a greater reduction in landscape diversity. Heterogeneity was reduced at finer scales in the managed watershed, suggesting that taxa needing high heterogeneity at finer scales were more vulnerable in this location. For example, in a study in Germany, Steckel et al. (2014) found that wasp richness was positively influenced by fine, rather than medium, scale landscape heterogeneity.

Our results suggest that low-intensity management, both with respect to forestry and agro-pastoral activities, was not sufficient for the maintenance of a heterogeneous landscape. It is likely that the low-intensity management applied in the managed watershed was masking a substantial abandonment of agro-pastoral activities, and that forest management alone did not fully prevent tree encroachment. This is particularly relevant for protected areas (e.g. Natura 2000 sites) in which grassland and shrubland habitats are of conservation interest. For example, both long continuity grassland management and current activities are fundamental for high species density (Cousins et al., 2007; Eriksson et al., 2002) which is a common indicator of the conservation status of semi-natural habitats. Furthermore, compositional and configurational heterogeneity support taxonomically diverse butterfly communities and vulnerable species linked to grassland habitats, respectively (Perović et al., 2015). Therefore, in addition to near-to-nature forestry, extensive grazing activities (Cocca et al., 2012) and mowing should be promoted to maintain a degree of landscape heterogeneity, as heterogeneity is likely to have a positive effect on the conservation of habitats and species.

The analysis of relationships between single land covers and the response of landscape metrics over extent and time gives additional insights into the changes occurring in the two watersheds. Understanding the response of class-level metrics at multiple scales is fundamental to characterize and monitor landscape heterogeneity (Wu, 2004). Our results highlight the huge variability in the response of class-level metrics to changes in scale (Kelly et al., 2011; Wu, 2004) and time under different anthropogenic pressure regimes. For example, woodlands had stable, linear-decay, and power-law increasing responses to scale with or without scale-breaks. All metrics examined were sensitive to changes in extent, however certain metrics showed less predictable trends (e.g. aggregation index) than others (e.g. area-weighted mean patch size). Previous studies have described different types of responses to change in extent (e.g., Baldwin et al., 2004; Wu, 2004); however certain metrics can be considered to respond following specific scaling relationships (Siímovaí and Gdulovaí, 2012).

As for landscape diversity, response of class metrics among scales and time tended to be similar between the abandoned and managed watersheds. Therefore, both watersheds shared similar scaling and shaping dynamics not showing substantial influence over time. Furthermore, while most of the metrics seemed to generally have a consistent response regardless of the considered class (grasslands and meadows, shrubland, woodland), the percentage of landscape and edge density had a different response when we considered woodland, or grasslands and meadows, or shrubland classes. These differences can be linked to the different changes in cover of the three classes and to their different total cover within the whole watershed.

Class-level edge density is believed to have an unpredictable response with increasing extent (type III metrics, Wu, 2004), and this seemed to be confirmed in our study. Patch density is also considered to be unpredictable, and in our study it tended to have a decreasing power function, characterized by a strong linear-decay pattern at very small scales in nearly all cases. However, patch density tended to also level at low scales for recent grasslands and meadows, and shrublands in the low-intensity managed watershed. Furthermore, other metrics responded differently that what has been reported in previous studies; for example, Wu (2004) found a staircase-like response with changing extent for area-weighted mean shape index, in contrast to the findings of our study. Argan˜araz and Entraigas (2014) found that changes in grain size and extent varied for different landscapes and among class types. Furthermore, our results suggest that metrics response depends on the landscape class under investigation, and that the response is also likely to change with changing anthropogenic pressure. Hence, the response of metrics to changes in extent will depend on the underlying processes occurring in the landscape. Observing the response of metrics among different time periods helps in comparing the intensity of change over extent after changes in management.

This interaction between landscape management and the response of class-level metrics of landscape patterns at changing scales holds an important informative function for the conservation of habitats and species. The capacity of landscape metrics to predict species occurrence is affected by the spatial scale of analysis; for example, Schindler et al. (2013) observed that woody plants, orthopterans, and small terrestrial birds are better predicted at smaller extents than reptiles. In our study, differences in the response of metrics over time were more evident at smaller spatial extents. Therefore, species that respond to changes at small scales are likely to have been impacted by the management regime in the two watersheds. Indeed, species are likely to respond to changes in landscape pattern in relation to their home range and dispersal capability. Species with small home ranges and short dispersal distance will be more influenced by changes at small scales, while species with larger home ranges and longer dispersal distance will be more affected by differences at larger scales. For example, *P. mnemosyne*, a butterfly of conservation concern found in the study region, is dependent on grassland habitats. The dispersal distance of this species is considered to be relatively small (253 m ±12.59), indicating that patch density at small distances is important for their migration (Välimäki and Itämies, 2003). In our case study, grasslands and meadows patch density at small scales (<200 m radius) slightly decreased over time in the abandoned watershed, whereas in the low-intensity managed watershed it underwent a drastic decrease over time. This phenomenon, together with a decrease at smaller scales of the percentage of the landscape and a more disperse pattern of this habitat type, indicates a strongest reduction of suitable conditions for this species in the low-intensity managed watershed during the last time period examined. Similarly, other vulnerable species may have experienced changes occurring at small/medium spatial scales. For example, *L. achine* has a relatively low dispersal distance (<500 m) and females favour the edges of woodland openings for laying eggs (Bergman, 1999; Bergman and Landin, 2002); therefore it is likely that changes in edge density at small scales (i.e. stronger in our managed watershed) had an influence on the communities of this species. Furthermore, species such as *Tetrastes bonasia* L. may have responded to different features at small spatial scales (Sitzia et al., 2014). Nevertheless, species with longer dispersal distance may have not responded to these small-scale changes, but rather at those occurred at larger scales; for example *Alectoris graeca saxatilis* Meisner, which has an average dispersal distance of 4–15 km (Bernard-Laurent, 1991; Cattadori et al., 2003).

5. Conclusion

This study compared landscape pattern changes occurring over time and space in one watershed where management was abandoned and in one were management continued over time, but with low-intensity. In both watersheds, woodland cover increased with similar trends (at the expenses of grasslands and meadows, and shrublands). A loss in landscape heterogeneity occurred regardless of the management regimes in place in the two watersheds, primarily between 1954 and 1980/83. The landscape metrics showed a variety of responses depending on scale, time, habitat type, and anthropogenic pressure. Indeed, these complex interactions, as shown by landscape metrics, highlight the importance of taking into account multiple perspectives for characterizing different landscapes (Lustig et al., 2015).

Our study indicates that management regime can affect the spatial scale response of landscape and class-level metrics. A reduction in scale breaks for grasslands and meadows, and shrublands over time highlighted the relevant spatial changes. Understanding the changes in response of specific landscape metrics over scale, time, habitat type, and management regime are important, as they have implications for biodiversity conservation, especially for species that may be sensitive to habitat modification (Sitzia and Trentanovi, 2011). Monitoring landscape metrics have the potential to help the assessment of the conservation status of habitats under the Habitats Directive (Perrino et al., 2013; Vaz et al., 2015) and this should be further investigated in other landscape settings taking into account relevant habitat characteristics. Our study highlighted that the landscape response of grasslands and meadows, and shrublands was similarly affected by abandonment but also by low-intensity management. These results suggest that the local extinction of many species linked to grasslands and meadows, and shrubland habitats, may have occurred in both watersheds; probably earlier in the abandoned than in the low-intensity managed watershed. Indeed, future studies should investigate time lags following changes in landscape metric response, in order to adopt conservation measures for habitats of high biodiversity value in a timely and adequate manner (Bagaria et al., 2015).

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