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Landscape change and the dynamics of open formations in a natural reserve

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Abstract

Remote sensing, when used in conjunction with landscape pattern metrics, is a powerful method for the study of ecological dynamics at the landscape scale by means of multi-temporal analysis. In this paper, we examine temporal change in open formations in the natural reserve of Poggio all'Olmo (central Italy). This area has undergone rural depopulation and the cessation of traditional methods of agriculture, resulting in the subsequent re-establishment and spread of other vegetation formations. Aerial photographs taken in 1954, 1977 and 1998 were orthorectified and classified based on the physiognomic characteristics of the vegetation. An objective definition of the minimum mapping unit (MMU) was guaranteed by using vector format grids for this classification. We applied landscape pattern metrics based on landscape composition, the shape and size of patches and patch isolation. Our results demonstrate the key roles of shrubland, woodland and coniferous plantations in the ongoing fragmentation of open formations in the landscape. Multi-temporal landscape analyses, and, in particular, a restricted suite of landscape metrics, proved useful for detecting and quantitatively characterizing dynamic ecological processes. We conclude with some recommendations on the management alternatives feasible for the protection of the remaining grassland formations in the natural reserve of Poggio all'Olmo.

Keywords: Aerial photos; Biodiversity management; GIS; Landscape metrics; Minimum mapping unit; Multi-temporal analysis

1. Introduction

In many ecosystems, biodiversity is rapidly being lost, and Reid's (1992) estimate of a 13% loss over the period 1990–2015 is beginning to appear conservative (Nagendra and Gadgil, 1999). At the same time, new integrated approaches for ecosystem monitoring and management have proven increasingly successful in regional planning and natural resource management (Innes and Koch, 1998). Landscape analyses conducted over large spatial extents are important for biodiversity conservation (Forman and Godron, 1986;

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Forman, 1995). The sheer complexity of many ecosystems suggests that characterizing ecosystems by means of the more conventional species-level approach may be either expensive or even impossible when historical data are not readily available (Innes and Koch, 1998; Roy and Tomar, 2000). Remote sensing and Geographical Information Systems (GIS), when integrated with the tools of landscape ecology, can be used to investigate the changing spatial patterns of biodiversity (Innes and Koch, 1998; Roy and Tomar, 2000).

Of the features of the ecosystem most easily identified via remote sensing, the *physiognomy* and *spatial pattern* of the vegetation are among the most important, since they characterize the landscape both structurally and functionally. A description of the shape, size and spatial arrangement of patches of vegetation in the landscape can be used to link the observed pattern with the ecological processes that may have generated it. For example, the fragmentation of ecological units (a spatial characteristic) has often been reported to be related to ecosystem degradation (Ludeke et al., 1990; Mertens and Lambin, 1997).

Alongside spatial scale, temporal scale is equally important when assessing the change of a landscape over time. Multi-temporal analysis based on remotely sensed data (e.g. grey-scaled aerial photographs) has played an important role in landscape ecology (Butaye et al., 1999; Defries and Townshend, 1999; Viedma and Meliá, 1999). In most studies, the classification of airborne data is performed by digitizing polygons based on photo-interpretation, with the minimum mapping unit (MMU) defined by means of the smallest polygon identified. This method leads to problems in comparing maps whose MMU is defined so subjectively. Landscape metrics show variations across a range of scales (Turner et al., 1989; Jelinski and Wu, 1996; Wu et al., 1997, 2000, 2002), and so the translation of information across scales is problematic if the MMU is not clearly defined. Moreover, all of the resampling methods commonly used to compare maps with different input MMUs introduce some spatial error (Turner et al., 1989; Gustafson, 1998), since they do not take account of the 'true' resolution of the maps; for example, a map with a MMU of 100 m, but resampled at 20 m MMU, will not reveal patterns recognized at <100 m. In this paper, we used vector format grids with a pre-defined cell size, alongside landscape pattern metrics. This method of interpretation, although very time consuming, ensures that the analysis is undertaken using a precise and consistent MMU, and thus allows for the comparison of different processes at the same resolution.

The aim of this paper is to quantify landscape change in an abandoned rural landscape and give some recommendations on the management alternatives feasible for the protection of the grassland formations.

2. Study area

The natural reserve of Poggio all'Olmo, on the slopes of Mt. Amiata (latitude 42°52′00″, longitude 11°28′21″, datum WGS84), occupies an area of about 440 ha, with calcarenite, clay and sandstone being the main soil types. The elevation ranges from 664 to 1016 m, and the climate belongs to humid type B2, at the limit with B1, in terms of Thornthwaite's (1948) classification. Total annual rainfall averages 1045 mm and mean annual temperature averages 12.5 °C (climatic station: Castel del Piano, 639 m a.s.l.). Recently, the area has undergone a rapid re-colonization by woody vegetation in open formations (pastures and abandoned fields). Land-use has changed considerably in the area surrounding the reserve, particularly since the Second World War, due to rural depopulation and a cessation/decrease in traditional agricultural practices (e.g. terracing, grazing, maintenance of grazed grasslands by burning, cultivation of chestnut groves, woodcutting, cultivation of fields, maintenance of hedgerows at field boundaries). The natural reserve was established in 1998 to conserve remnant semi-natural grassland habitats.

As described by Angiolini et al. (1999), the vegetation found in the reserve today comprises: open formations, e.g. semi-natural grasslands (*Bromus erectus* grasslands and *Arrhenatherum elatius* grasslands) and small cultivated fields; *Prunus spinosa* and *Cytisus scoparius* shrublands; woodlands, e.g. *Quercus cerris* woodlands, mixed broad-leaf woodlands (dominated by *Quercus pubescens*, *Ostrya carpinifolia* and *Q. cerris*), riparian woody vegetation (dominated by *Populus nigra*, *Ulmus minor*, *Acer campestre*, *Tilia platyphyllos* and *O. carpinifolia* or *Salix* sp. pl.), maple thickets and chestnut woodlands. Cultivated fields once covered about one-third of the area (Comune di Cinigiano, unpublished data) but are now almost completely absent. Exotic conifers (*Abies alba* Miller, *Cupressus arizonica* Green, *Pinus nigra* Arnold, *Pinus sylvestris* L., *Pseudotsuga menziesii* (Mirbel) Franco), which do not reproduce spontaneously, were planted in the 1950s–1970s, with two large plantations at Poggio la Torretta and Poggio Matoraio (Maccherini et al., 2001). Current agro-forestry practices are largely extensive, consisting of sheep grazing, wood-cutting (coppice with standards), cultivation of chestnut groves and small arable fields.

3. Methods

3.1. Derivation of land-use maps

Aerial photographs (grey-scale) taken in 1954, 1977 and 1998 (scale: 1:33,000) were scanned at high resolution (600 dpi). Orthorectification, based on a Digital Terrain Model (DTM) derived from a 1:10000 topographic map (pixel size: 10 m) of the study area and on 30 ground control points (GCPs), was performed using ERDAS IMAGINE 8.4 (www.erdas.com). Each image's spatial resolution was approximately 2 m. Positional accuracy was tested by means of 20 additional GCPs; error never exceeded 4 m. Images were projected onto the National (Italian) Coordinate System (Gauss Boaga Projection, datum Roma 40).

The minimum mapping unit was defined a priori by the superimposition of a vector format grid with a cell size of $10\,m \times 10\,m$, giving a total of 44,000 cells; this analysis was conducted using the ArcView 3.2 GIS (ESRI). Each cell was subjected to photo-interpretation by means of pixel radiance and the physiognomic characteristics of the vegetation it contained. Vegetation classes recognized using these methods were: woodlands, shrublands, open formations (pastures and fields), buildings, isolated or grouped trees, linear formations (hedges) and coniferous plantations. If a cell contained two or more classes, the value of the prevalent class (in terms of area) was used. This procedure was performed on the same $10\,m \times 10\,m$ grid for each of the three periods (1954, 1977, 1998); thus, each cell represents a spatial unit whose identity may, potentially, change over time.

3.2. Landscape analysis

For the landscape analysis, vector format grids were converted into a raster format at a cell resolution of $10\,m \times 10\,m$, and a suite of non-redundant landscape pattern metrics was calculated for each of the grids. Landscape *composition* was quantified by means of the area covered by each class. Transition matrices, based on cells occupied by classes over the three time periods, were generated to quantify the overall gains and losses of area (converted from cells) for the different landscape classes. The Kappa (K) statistic (Chust et al., 1999; Romero-Calcerrada and Perry, 2004) was used to quantify how stable the different vegetation classes were: this index is based on a comparison between pairs of maps and ranges from 0 (no pixels change) to 1 (all pixels change). We used a series of landscape pattern metrics to quantify change in the landscape. Such metrics have been widely applied in similar studies, and allow objective description of the temporal pattern of landscape change and comparison of this with other (similar) landscapes (Turner et al., 2001). Landscape *structure* was assessed by means of patch-based metrics, such as the total and per class number of patches, shape-based metrics, such as the Area Weighted Mean Shape Index (AWMSI) and size-based metrics, such as the maximum, minimum, mean and standard deviation of patch size (MaxPS, MinPS, MPS, PSSD, respectively). Shape metrics approach zero when patches have perfectly square (raster format) or circular (vector format) shapes. For the size metrics, frequency–size distributions were also analysed; these allow an analysis of changes in the patch structure through time (e.g. what proportion of the landscape is covered by patches below a certain size). The slope of the patch size versus frequency plot quantifies the ratio of the number of large to small patches in the landscape; as the slope increases, large patches become rarer with respect to small ones. Isolation was measured by means of the Mean Proximity Index (MPI). This index is directly related to the area covered by patches and inversely related to the nearest neighbour distance, and represents the "non-isolation" of patches. The MPI equals zero if all patches of the corresponding class have no neighbour of the same type within a given search radius (McGarigal and Marks, 1995); in these analyses the search radius was set to 100 m. A thorough explanation of these metrics is given by McGarigal and Marks (1995) and Turner et al. (2001).

4. Results

4.1. General temporal trends

The three maps show that over the last 50 years important changes have occurred in the landscape (Fig. 1). In 1954, open formations were the most abundant and dominant class, and they constituted a matrix within which patches of the other vegetation classes were dispersed. By 1977, a considerable decrease in the abundance of open formations, with commensurate increases in the other classes, had taken place. This resulted in increased fragmentation; a clearly dominant matrix was no longer recognizable and the landscape mosaic showed a higher heterogeneity, in terms of spatial structure. By 1998, open formations had decreased

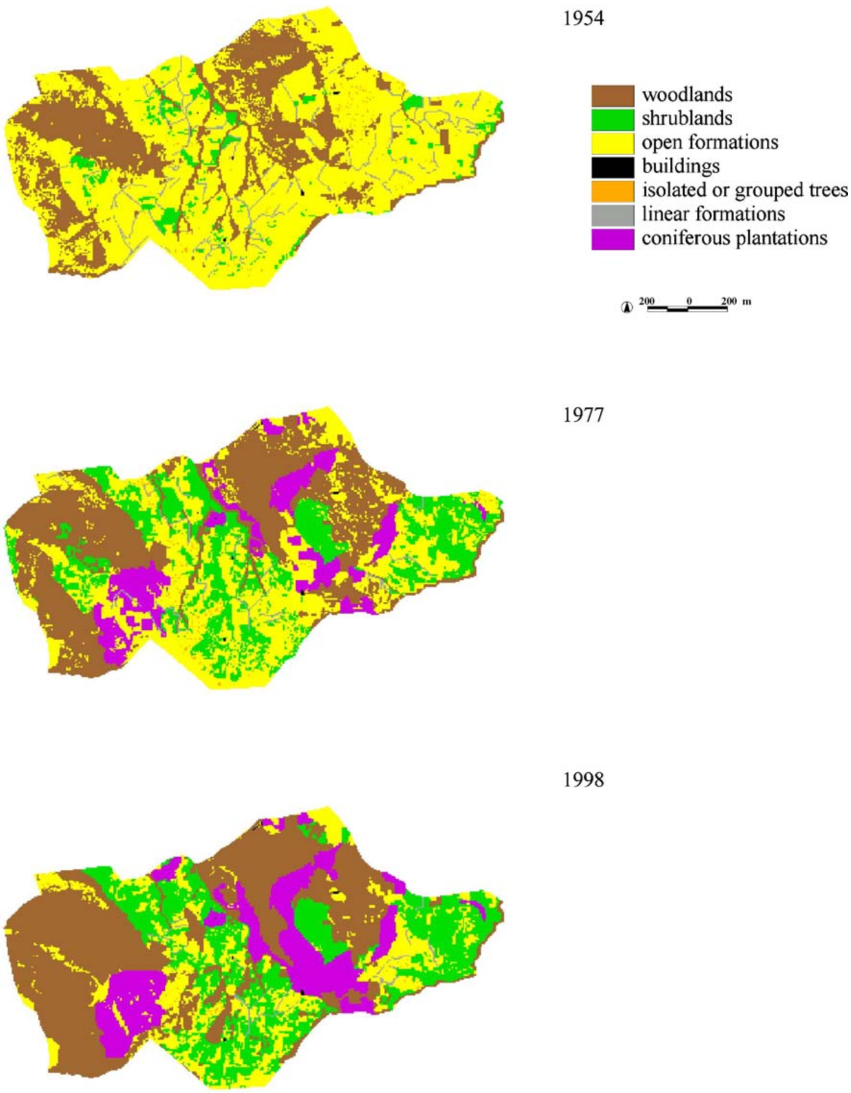

Fig. 1. Change in vegetation cover over the period 1954–1998 derived from interpretation of aerial photographs.

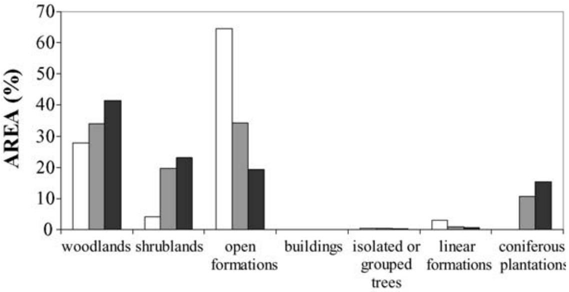

Fig. 2. Area (as a percentage of total reserve area) occupied by each class. White columns: 1954; grey columns: 1977; black columns: 1998.

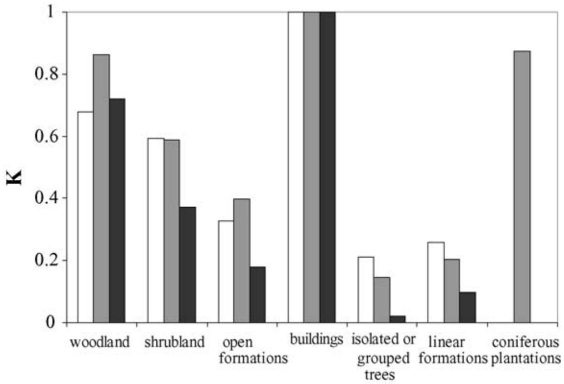

Fig. 3. Kappa values for each class, representing the level of permanence. White columns: 1954–1977 period; grey columns: 1977–1998 period; black columns: 1954–1998 period. Note that the only non-zero value for the 'coniferous plantations' class is for the interval 1977–1998; this is because this vegetation type was absent from the landscape in 1954 making comparison meaningless.

still further and woody vegetation (woodlands and coniferous plantations) had increased.

4.2. Landscape composition

The total area of woodland increased from 28% in 1954 to 34% in 1977 and to 41% in 1998 (Fig. 2). The relative increase in woodland abundance, from 1954 to 1998, was 48%. Coniferous plantations were not present in the landscape in 1954, but constituted 15% of the total area in 1998. Shrublands represented the vegetation type with the highest relative increase, being five times more abundant in 1998 than in 1954, and covering about a quarter of the total area. Conversely, open formations underwent a consistent decrease; cover of this class declined from 65% in 1954 to 19% in 1998, with a loss of more than 70% of the area covered in 1954. By 1998, this class was largely replaced by shrublands, woodlands and coniferous plantations. Isolated trees and linear formations (hedges), although always only present at low abundances, also decreased markedly; both of these vegetation classes suffered about a 70% decline in cover over the period 1954–1998.

Based on transitions occurring in the period from 1954 to 1977, woodlands and shrublands were found to be highly stable (Table 1). Conversely, open formations were frequently transformed into woodlands, shrublands or coniferous plantations. Isolated trees and linear formations showed low stability, losing 78.50 and 73.85% of their areal cover, respectively. A similar pattern was observed in the period from 1977 to 1998 (Table 2), with some differences resulting from the presence of coniferous plantations, which increasingly replaced the open formations, isolated trees and linear formations. The Kappa statistic (Fig. 3) indicated that woodlands, shrublands, coniferous plantations and buildings were the most stable components of the land-

Table 1
Transition matrix for 1954–1977 period (values in hectares)

1954	1977						
	Woodlands	Shrublands	Open formations	Buildings	Isolated or grouped trees	Linear formations	Coniferous plantations
Woodlands	91.5	13.3	11.2	0	0.2	0.2	6.7
Shrublands	1.4	11.8	4	0	0	0.2	1.1
Open formations	55.7	57.7	132.2	0	1.5	0.6	36.9
Buildings	0	0	0	0.2	0	0	0
Isolated or grouped trees	0.9	0.2	0.4	0	0.4	0	0.1
Linear formations	0.7	3.8	2.8	0	0.1	3.4	1.8

Table 2
Transition matrix for 1977–1998 period (values in hectares)

1977	1998						
	Woodlands	Shrublands	Open formations	Buildings	Isolated or grouped trees	Linear formations	Coniferous plantations
Woodlands	134.7	2.8	2.1	0	0	0.1	10.3
Shrublands	19	56.1	9.2	0	0.1	0.9	1.4
Open formations	24.1	40.6	70.5	0	0.2	1.1	13.9
Buildings	0	0	0	0.3	0	0	0
Isolated or grouped trees	0.2	0.7	0.7	0	0.3	0	0.3
Linear formations	0.8	1.1	0.1	0	0	0.1	0.6
Coniferous plantations	3.8	0.2	1.4	0	0	0	41.2

scape, while open formations, isolated or grouped trees and linear formations were the least stable. This is consistent with the patterns of change in the landscape since 1954 described above.

4.3. Landscape structure

The total number of patches in the landscape increased by about 20% from 1954 to 1977 (609–734) and then declined by about 36.6% from 1977 to 1998 (734–465), resulting in an overall decrease of 23.6% for the entire period (1954–1998). In terms of the number of patches per class (Fig. 4), there was a progressive decrease from 1954 to 1998 in all classes, with the exception of shrublands and open formations, which showed an increase from 1954 to 1977 and a subsequent decrease from 1977 to 1998. The number of shrub patches in 1998 was half that of the original number, while the number of patches of open formations was similar to that seen in 1954.

In terms of patch shape, the open formations showed the highest value of AWMSI in 1954, but there was an abrupt decrease in the values of this index over time (Fig. 5). The shape of woodland patches became increasingly simple. By contrast, shrublands showed a trend towards having more diffused and complex shapes. No discernible trends were observed in the shape of patches of buildings, isolated trees, linear formations and coniferous plantations.

Over the three “sampling dates”, PSSD values were higher than MPS values, for all classes (Fig. 6), suggesting a right-skewed distribution with small patches being numerically dominant for all classes (see also Fig. 8). This may be related to the classification method, using a fixed grid cell size at a very fine resolution. All classes showed an increase in MPS over time, with the exclusion of open formations.

In 1954, the minimum patch size for all classes was 0.01 ha (the MMU), with the exception of coniferous plantations (0.08 ha). By contrast, the MinPS values in 1998 reveal the aggregated and artificial nature of

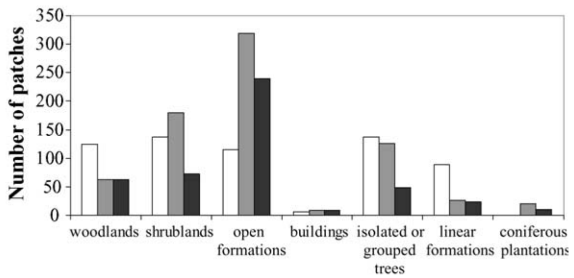

Fig. 4. Number of patches per class. White columns: 1954; grey columns: 1977; black columns: 1998.

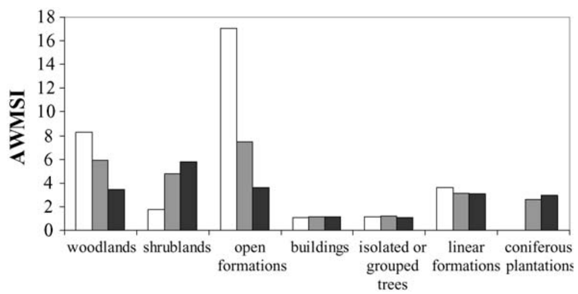

Fig. 5. Area Weighted Mean Shape Index (AWMSI) per class. White columns: 1954; grey columns: 1977; black columns: 1998. Notice the high value of open formations in 1954 and the abrupt decrease in following dates, showing a move towards a simpler shape over time.

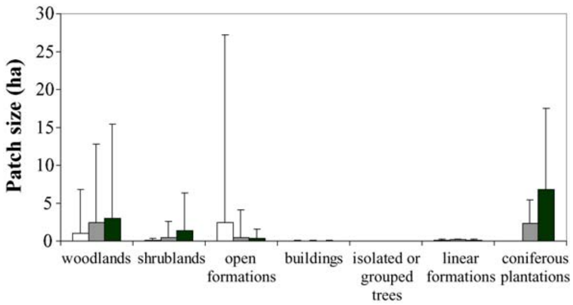

Fig. 6. Patch size: mean patch size (MPS), columns and patch size standard deviation (PSSD), error bars, per class. White columns: 1954; grey columns: 1977; black columns: 1998.

the coniferous plantations. As for the MPS (Fig. 7), open formations had the highest value in 1954, but they suffered a remarkable decrease over subsequent years. Woody vegetation (woodlands, shrublands, coniferous plantations) showed opposite trends to the open formations. The frequency–size distribution of patch areas (Fig. 8) showed large differences between the maps of 1954, 1977 and 1998, with a steeper slope in 1954 corresponding to the abundance of small-medium patches, while a shallower slope was found in 1977 and 1998.

The value of the MPI for open formations in 1954 exceeded that of all other patch types (Fig. 9). Open formations showed an abrupt decrease from 1954 to 1977, and a further decrease from 1977 to 1998, suggesting their increased fragmentation over this period. Shrublands and coniferous plantations showed a progressive increase in the MPI, suggesting a decrease in their isolation. Woodlands showed a different trend: a lack of change from 1954 to 1977 and then a decrease

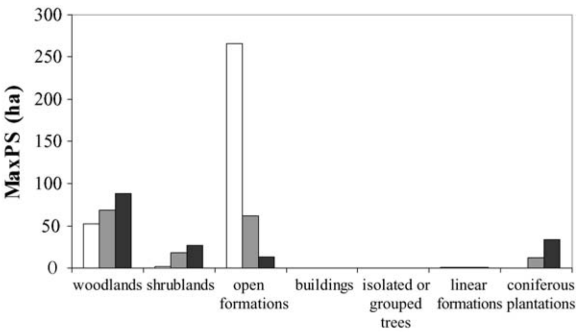

Fig. 7. Maximum patch size per class (MaxPS). White columns: 1954; grey columns: 1977; black columns: 1998.

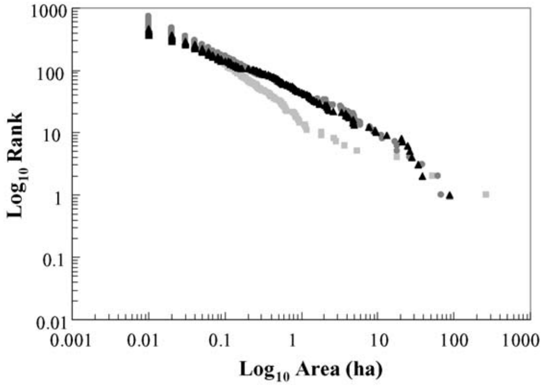

Fig. 8. Patch size frequency–area distributions for each year; light grey squares: 1954; dark grey circles: 1977; black triangles: 1998.

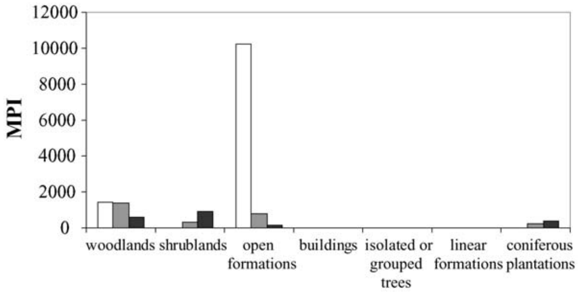

Fig. 9. Mean Proximity Index (MPI) per class. White columns: 1954; grey columns: 1977; black columns: 1998.

from 1977 to 1998. Buildings, isolated trees and linear formations had the lowest values of MPI, resulting from their scattered occurrence in the landscape.

5. Discussion

The cessation/decrease of traditional agricultural practices in the natural reserve of Poggio all'Olmo has resulted in a decline in the abundance of open formations, linear formations (hedges) and isolated or grouped trees, and the spread of woody vegetation (shrublands, woodlands, coniferous plantations). The open formations have become increasingly fragmented since 1954, when a single patch occupied about 60% of the total area; by 1977 and then 1998 these formations had become fragmented into several small patches with a commensurate increase in patch isolation. The spread in shrubland areas largely accounts for this increase in landscape fragmentation. This vegetation type has expanded by forming a number of patches characterized by diffuse edges. Over time these patches have coalesced and "closed" the previously open formations, linear formations (hedges) and isolated or grouped trees. This phenomenon is basically related to the establishment of several nuclei shrubs, which initially colonize the open areas in a stochastic manner, and become closer to one another over time, before being replaced by woodland patches. This invasion of the open formations by isolated woody shrubs, and the subsequent coalescence of these woody invasive species into larger patches explains the trend in the number and size of patches of open formation vegetation—an increase during the early invasion phase followed by a decrease as individual patches coalesce. Other authors (e.g. Loehle et al., 1996) have speculated that, once the woody vegetation cover exceeds a critical threshold, this colonization process will 'self-accelerate'. Woody formations (woodlands and coniferous plantations) increased from less than 25% to about 50% of the reserve area (1954–1998), whereas the area of open formations has fallen by 70%, when compared to the area occupied in 1954. These values emphasize the magnitude of changes that have led to, and will continue to result in, the replacement and loss of the open formations.

The expansion of woody vegetation over open vegetation types, following the cessation of traditional management, has been observed in several Mediterranean areas (e.g. see Blondel and Aronson, 1999; Carmel and Kadmon, 1999; Romero-Calcerrada and Perry, 2004), including elsewhere in central Italy (Vos and Stortelder, 1992; Peroni et al., 2000; Torta, 2004). The general successional trend in many Mediterranean landscapes is from pasture to shrublands (abandoned pastures) and woody vegetation; the latter is considered to be the endpoint of secondary succession in these ecosystems. This type of old-field succession may have important implications for a wide range of ecological processes such as, for example, nutrient cycling (Blondel and Aronson, 1999; Wainwright and Thornes, 2004). Carmel and Kadmon (1999) studied the long-term effect of grazing on the vegetation dynamics in a Mediterranean ecosystem in the Northern Galilee Mountains, Israel, from 1964 to 1998. They point out that a decrease in grazing intensity resulted in an increased cover of woodland coenoses, from 2% of total area in 1964 to 41% in 1998, accompanied by a commensurate decrease in the cover of herbaceous vegetation, from 56% in 1964 to 24% in 1998. Their results show that the regeneration of woodland vegetation can occur quite rapidly (certainly over a period of 30 years).

In Italy, similar trends have been observed in various parts of the Apennine mountains, such as the Solano Valley (Vos and Stortelder, 1992), the Aterno Valley (Peroni et al., 2000), the Aniene Middle Valley (Blasi et al., 2001), the Northern Apennines (Torta, 2004) and the Pisan Hills (Bertacchi and Onnis, 2004). Blasi and Di Pietro (1998) and Di Pietro and Filibeck (2000) report that 30–40 years of abandonment were sufficient for the regeneration of woodland coenoses in hilly areas characterized by terracing in Central and Southern Italy. Other authors, however, have found that it can take much longer for woodland colonization of abandoned unterraced agricultural fields in Mediterranean areas to occur (e.g. Escarrè et al., 1983). As Debussche and Lepart (1992) conclude, it is difficult to make broad generalisations about the invasion of old field sites by woody vegetation in the Meditteranean. Nevertheless, Vos and Stortelder (1992) consider that the "remnants" of traditional Tuscan mountain landscape will have changed radically in another 50 years and may totally vanish. These concerns are echoed by Bertacchi and Onnis (2004), who note that modification of the 'typical' landscape in Tuscany is prevalent in hilly areas close to urban centres and on foothills at the edge of plains. In the present study, a 50-year time period has been sufficient for the regeneration of woodland coenoses. The observed decrease in the number of woodland patches corresponds with simpler patch shapes developing as individual patches begin to coalesce. Likewise, previously separate patches of woodlands and coniferous plantations have merged, resulting in the formation of fewer and larger patches, again with simpler shapes. For coniferous plantations, this is simply a consequence of the way in which such plantations are managed. As for woodlands, this pattern of forest dynamics has been already recognized at other Italian sites that have undergone abandonment and a cessation of traditional forestry and agricultural practices (Vos and Stortelder, 1992; Peroni et al., 2000; Blasi et al., 2001). As Vos and Stortelder (1992) and Bertacchi and Onnis (2004) discussed, the fine-grained patterning seen in some traditional agricultural landscapes is gradually being lost and is progressively replaced by a more coarsely grained pattern, characterized by the dominance of woodland assemblages over the other vegetation types. In this study, a significant decrease in the total number of patches was observed (1977–1998), after an initial increase in the period 1954–1977. From 1977, the change in landscape structure, and the subsequent fragmentation of open formations, has resulted in the isolation of patches belonging to this class. These patches have become embedded within the woody classes (shrublands, woodlands, coniferous plantations). The reduced patchiness observed in 1998 fits the conceptual model of 'patch fusion' for the dominant vegetation type (woodlands, see Vos and Stortelder, 1992), which has replaced the other vegetation types following a change in land-use management.

6. Implications for restoration management

The changes in the open formations that occurred in the natural reserve of Poggio all'Olmo are typical of a general trend occurring across Europe; in particular, semi-natural grassland communities have, due to changes in land-use, declined in area and quality during the second half of the 20th century across all of Europe (van Dijk, 1991; Bakker and Berendse, 1999). These plant communities are of great interest, due to their landscape aesthetic value, the high number of rare and/or endangered species they harbour, and their high species richness (see Willems, 2001). Abandoned pastures and lowered grazing pressure are the main factors triggering natural succession of semi-natural grasslands into increasingly dense shrublands, characterized by lower species richness and fewer rare species (Barbaro et al., 2001).

Pastures are the vegetation type containing the highest number of species of phytogeographic interest and are the most threatened habitat within the natural reserve (Maccherini et al., 2001); semi-natural grasslands on calcareous substrata are also classified as habitats of 'community importance' in Annexure I of the Habitats Directive 92/43/ECC (Romão, 1996).

The restoration of semi-natural grasslands depends on the degree of degradation (Muller et al., 1998; Bakker and Berendse, 1999). An important factor limiting the success of this type of management is that often only a few grassland species remain present in either the (soil) seed bank and/or the seed rain (see Bissels et al., 2004). This slows restoration, especially for formerly arable land, if target species are not sown and the establishment rate of competitive, early successional species, is not artificially reduced (Hutchings and Booth, 1996; Kleijn, 2003).

The restoration of grassland vegetation after conifer forestry also seems to be a slow and difficult process (Csontos et al., 1996, 1996/1997; Maccherini and De Dominicis, 2003), as the grasslands are overtopped by trees (Dzwonko and Loster, 1998). Encouraging results, however, have been obtained in the restoration of grassland overgrown by shrubs, by the clearing of shrubs and mowing or grazing (Bobbink and Willems, 1993; Zobel et al., 1996; Willems and Bik, 1998; Barbaro et al., 2001). An additional problem for the management of protected areas is the presence of exotic species (D'Antonio and Meyerson, 2002): they can out-compete native species, infect them with diseases, out-compete them, or otherwise alter ecosystem functioning, making making the restoration of the original ecosystem a difficult and expensive task (Vitousek et al., 1997).

Based on these considerations, restoration of semi-natural grasslands for the natural reserve of Poggio all'Olmo is recommended only for the areas encroached by shrubs, on calcarenite by appropriate management (cutting of shrubs, burning and grazing); the same actions are also recommended for maintenance of the remaining semi-natural grasslands in the rest of the natural reserve. Abandoned ex-arable lands on clay and sandstone are actually covered by dense shrublands and are very hard to restore. Given their reduced value as habitat of rare and phytogeographycally interesting species they can optimally be left to their natural succession processes. On the contrary, maintenance of hedgerows, riparian woody vegetation, chestnut groves, maple thickets and the gradual substitution of exotic conifers with native woody species should be considered a priority for conservation of a high ecosystem heterogeneity and, consequently, a high species diversity.

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