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Landscape metrics and topographical determinants of large-scale forest dynamics in a Mediterranean landscape

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ABSTRACT

The analysis of land-use and land-cover change has long been a key topic in Landscape Ecology. In particular, forest fragmentation is known to affect species composition and diversity, thus threatening the integrity of forest habitats. This paper examines the forest cover dynamics in a Mediterranean area (the Province of Siena, central Italy), by comparing historical (1933) and recent forest maps (2000). The historical map has been geo-referenced, digitised in a GIS environment, and classified in three forest classes: broad-leaved, conifer and mixed forests. The same classification was used for the recent forest map. Image processing techniques and landscape pattern metrics were applied to quantify the changes in forest cover patterns, while appropriate statistical descriptors were adopted to investigate the relationship between land-cover changes and topographical factors. A general afforestation process was detected in the investigated area over the period 1933–2000, resulting in a high landscape transformation overall considering zones with higher elevation and slope. The forest landscape structure changed in terms of decreased fragmentation and patchiness. The general trend observed in this area was in line with previous results achieved in similar ecological situations, thus reinforcing the need to link landscape change patterns with appropriate management decisions.

1. Introduction

The analysis of land-cover change is a key topic in Ecology (Braimoh, 2006). The estimation of land-cover status and change can provide crucial ecological information for science-oriented resource management and policy making for a range of human activities (Cihlar, 2000). Land-cover change, such as forest fragmentation, affects forest-species composition and diversity, thus threatening the maintenance of stable and rich forest habitats (Brown et al., 2000). In particular, forest fragmentation represents one of the crucial factors among the human-induced processes that cause local, regional and global decline of biodiversity (Wilcox and Murphy, 1985). Generally, the major difference between forests and other vegetation types, such as grasslands and heathlands, is that forests represent the final successional stage in most terrestrial ecosystems (Packham et al., 1992; Chapin et al., 2002). This implies that when a non-forest land-use type is abandoned, vegetation dynamics will inevitably occur leading to a spontaneous forest regeneration (see e.g. Romero-Calcerrada and Perry, 2004), often passing from a mixture of fragments of different ages and sizes (Honnay et al., 2005) to a whole homogenization of the forest landscape (see e.g. Rocchini et al., 2006). Early-successional forest patches differ from ancient forest patches in both vegetation structure and composition, and many studies evidenced the importance of forest history and forest continuity for biodiversity conservation (Rackham, 1980; Peterken, 1981; Honnay et al., 2005).

The Mediterranean area is one of the most significantly altered hotspots on Earth (Myers et al., 2000), since it has been intensively affected by human activity for millennia (Covas and Blondel, 1998; Lavorel et al., 1998; Blondel and Aronson, 1999; Vallejo et al., 2005). As a result, only 4.7% of its primary vegetation remained unaltered (Falcucci et al., 2007). In fact, agricultural lands, evergreen woodlands and maquis habitats that dominate the Mediterranean basin are the result of anthropogenic disturbances over centuries or even millennia (Blondel and Aronson, 1995; Blondel, 2006). In many developed countries, a particular land-cover change pattern has taken place during the last decades: plains are being increasingly utilized for human activities, while mountain areas are being abandoned and are undergoing natural reforestation processes (Ales et al., 1992; García-Ruiz et al., 1996; Debussche et al., 1999; MacDonald et al., 2000; Santos, 2000; Lambin et al., 2003). These generalized patterns can have significant impacts on biodiversity distribution and conservation (Ales et al., 1992; Covas and Blondel, 1998).

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When developing conservation planning strategies, it is essential to link land-cover spatial patterns over long time periods to understand the processes underlying their change, in order to: (i) better understand the ecological mechanisms of these temporal changes, (ii) generate predictions about future rates of change and vulnerability of ecosystems, and (iii) design appropriate spatially explicit conservation policies (Lambin, 1994; Nagendra et al., 2004). A characterisation of the shape, size and spatial arrangement of patches of forests within a landscape can be used to link the detected spatial patterns to the driving forces generating them, such as natural ecological processes or human management practices (De Blois et al., 2002; Corry and Nassauer, 2005). Spatial patterns, such as the fragmentation of ecological units, have often been reported to be related to ecosystem degradation, i.e. an environmental process (Ludeke et al., 1990; Mertens and Lambin, 1997). In this perspective, multi-temporal analysis of air-borne imagery, with the support of GIS and remote sensing has a great potential for assessing and monitoring landscape diversity (e.g., Stoms and Estes, 1993; Defries and Townshend, 1999; Viedma and Meliá, 1999; Braimoh, 2006; Rocchini et al., 2006), and for investigating changes in the spatial patterns of vegetation and landscape structure (Innes and Koch, 1998; Roy and Tomar, 2000). Other “historical” resources, such as topographic and cadastral maps, can also be extremely useful for this purpose (Cousins, 2001; Benitez and Fisher, 2004).

The aims of this paper were: (i) to analyse the forest dynamics over 70 years in a relatively large Mediterranean area by means of a comparative examination of historical and present-day land-cover maps and (ii) to examine the relation between topography and forest dynamics.

2. Materials and methods

2.1. Study area

The study area is the whole Siena Province, about 3820 km^2 in size, located in Tuscany, Italy (centroid coordinates: longitude 11°26′54″E, latitude 43°10′12″N, datum WGS84). This province is characterised mainly by hilly topography, but high mountains (up to 1667 m) are also present. The mean annual temperature, over a period of 10 years (data from the Regional Agency for Research and Innovation in Agriculture) ranges from 11.8 to 14.3 °C, with mean annual rainfall ranging from 554 to 1080 mm.

Corine Land-Cover data (Bossard et al., 2000) shows that this area is currently predominantly occupied by agricultural land (33% of herbaceous cultivations, 11% of mixed cultivation patterns and 6% of woody plantations), while natural and semi-natural forests cover about 39%. Forests are mostly dominated by deciduous (mainly *Quercus cerris*, *Q. pubescens*) and evergreen (*Q. ilex*) oak formations, while chestnut (*Castanea sativa*) and beech (*Fagus sylvatica*) dominate forests at higher elevations.

2.2. Map derivation

A historical forest map of the entire territory of the Siena Province, produced in 1933 by the Milizia Forestale was located in the local department of the State Forest Department (Corpo Forestale dello Stato). This map contained detailed information about the status of forest in Siena Province like forest typologies, silvicultural practice, and dominant species. The 1933 forest map has been produced by field surveys and manual digitisation, using as a topographic base each of the forty-two 1:25,000 topographic maps that cover the entire province of Siena (Fig. 1a). The 1933 forest map was scanned and digitised within a geographic information system (GIS, see Fig. 2). Three general forest classes were extracted for the 1933 map: broad-leaved, conifer and mixed forests. This map was then overlaid to the 2000 Corine Land-Cover map (see Bossard et al., 2000) for which the same classes were obtained (Fig. 1b).

In order to smooth the error expected from the scanning, rectification and digitisation of the 1933 map, we (i) registered the 1933 maps to the recent topographic maps (scale 1:25,000, year: 1992) by further (ii) calculating the residuals in the x and y axes achieved from the co-registration process and (iii) using a pixel size higher than the spatial error for further analysis. In particular, we made use of 100 control points randomly scattered over the study area. Then we detected the nearest landscape feature which remained unchanged over time in the two maps (e.g. crossroads, houses, etc.). Finally, the deviation between the 1933 and the 2000

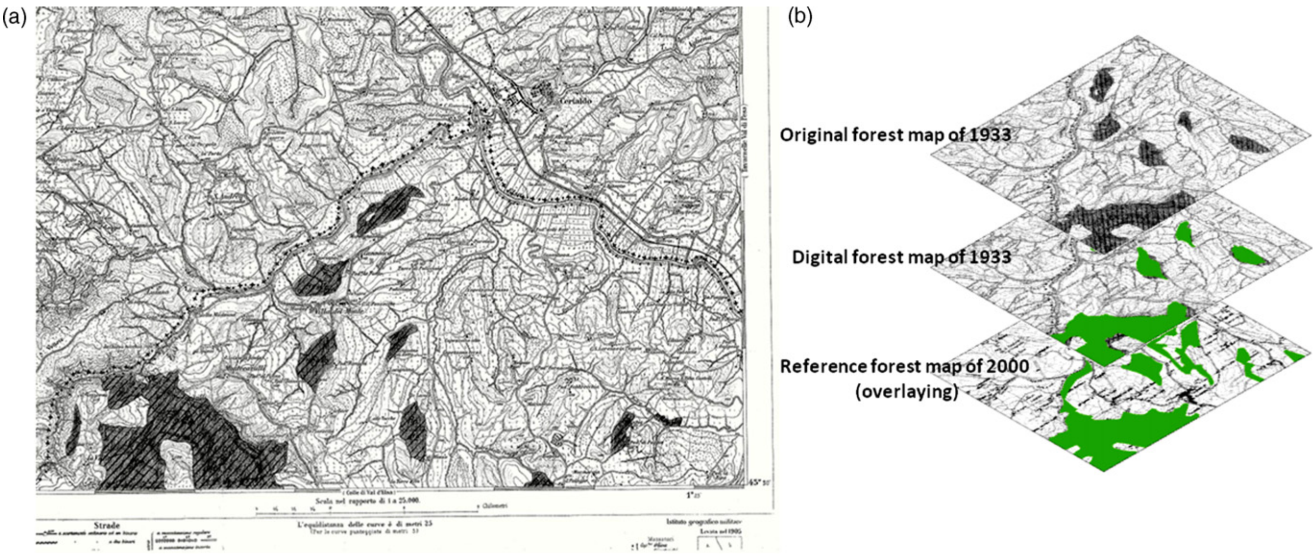

Fig. 1. (a) A detail of the forest map of 1933, (b) the process involving the acquisition (upward) and digitisation of the 1933 forest map, and overlay (downward) with the 2000 reference map. Notice that we showed analog data in grey and digital data in green. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of the article.)

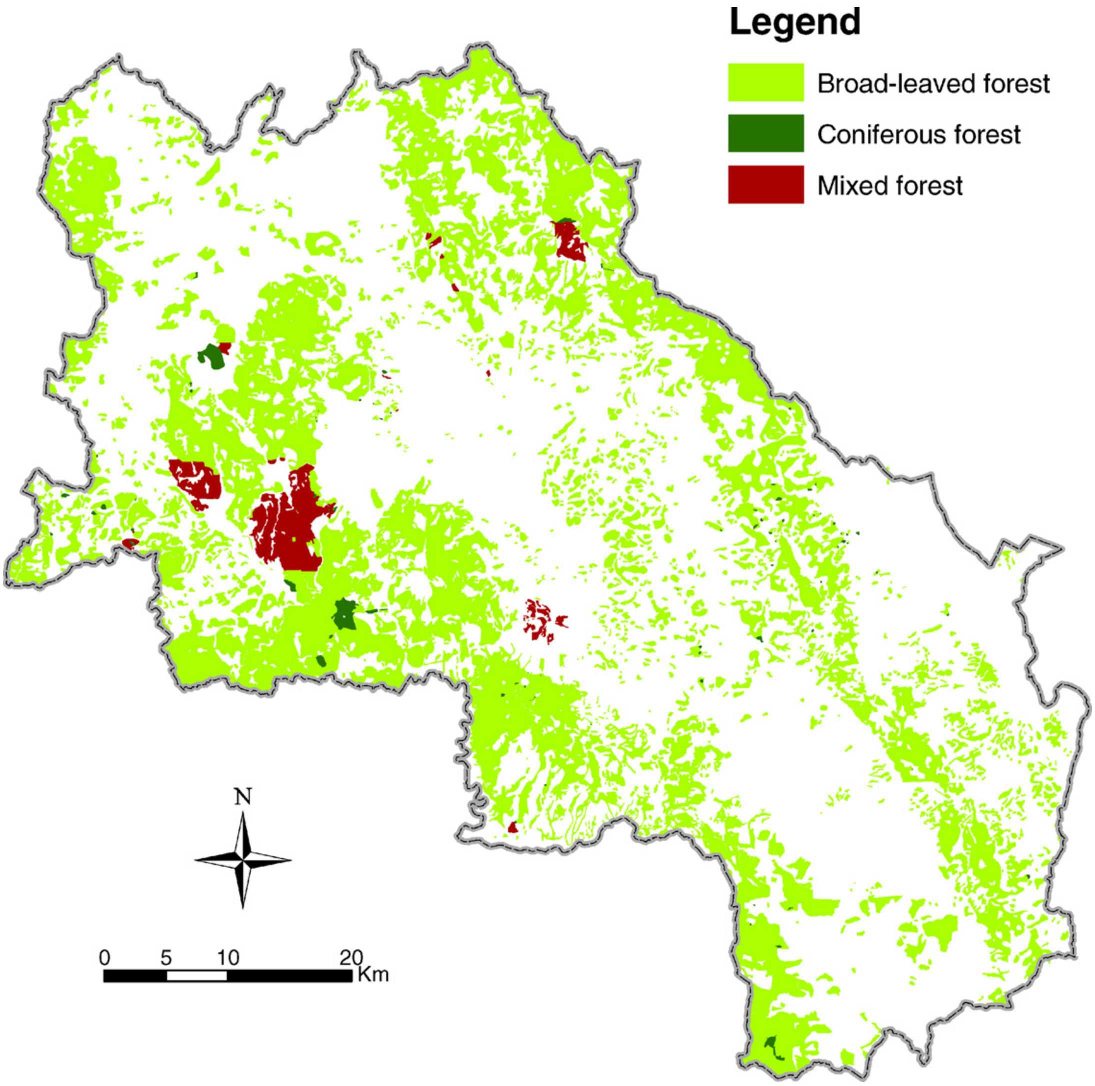

Fig. 2. Historical forest map derived from a digitisation of a map produced by the Milizia Forestale in 1933.

maps was calculated by the Root Mean Squared Error (RMSE) as:

$$RMSE = \sqrt{\frac{1}{n}\sum_{i=1}^{n} u^2 + v^2} \qquad (1)$$

where n = number of control points (in this case $n = 100$), u = residual in the x axis; v = residual in the y axis.

The RMSE corresponded to 75.07 m. Hence, we resampled both 1933 and 2000 forest maps using a grid with a spatial resolution of 100 m, i.e. a pixel dimension larger than the spatial inaccuracies calculated by RMSE. Such grids were derived from the original vector datasets by a majority filter.

2.3. Calculation of landscape metrics

A series of non-redundant landscape metrics was applied to the historical and Corine maps (using the grids with a resolution of 100 m, see Section 2.2). Landscape metrics have long been used in similar studies and they allow the objective description of the temporal patterns of landscape change (Turner et al., 2001).

Forest landscape *composition* was quantified by means of the area covered by each class (CA, Class Area). Moreover, the historical and Corine maps (100 m grids) were compared by overlaying and deriving transition matrices (also referred to as confusion matrices, see Baessler and Klotz, 2006; Rocchini et al., 2006) in order to quantify changes in landscape forest composition. This process typically implies to (i) overlay two different maps, (ii) derive for each pixel the correspondent change from one class to another one, or its conservation over time, and further (iii) summarise transitions of classes into a matrix.

Landscape *structure* was assessed by means of patch-based metrics such as the total and per class number of patches, shape-based metrics, size-based metrics and edge-based metrics (see e.g. Haines-Young and Chopping, 1996). Patches were derived using the 100 m grids and aggregating neighbour pixels (horizontally, vertically and diagonally) belonging to the same class.

The number of patches (NumP) is a useful measure to evaluate the weight of landscape configuration in a large number of ecological processes: as an example, it may determine the number of sub-populations in a spatially dispersed metapopulation for species exclusively associated with that habitat type (McGarigal and Marks, 1995). Patch shape has been shown to influence inter-patch processes such as small mammal migration (Buechner, 1989), woody plant colonization (Hardt and Forman, 1989) and may influence animal foraging strategies, Shape is a difficult parameter to be quantified concisely by a single metric (McGarigal and Marks, 1995). In order to analyse patch shape we used the area weighted mean shape index (AWMSI) that measures the average patch shape, weighted on patch shape size. Specifically, larger patches are weighted more heavily than smaller patches in calculating the average patch shape for the considered class or landscape (McGarigal and Marks, 1995). Area weighted mean patch size and patch size standard deviation (PSSD) were used as size-based metrics (McGarigal and Marks, 1995).

Edge metrics were used in order to analyse habitat loss and forest fragmentation (Bender et al., 1998). Statistics representing the amount of edge or degree of edge effect, like total edge (TE) and mean patch edge (MPE), were computed.

2.4. Calculation of topographical variables

A Digital Elevation Model (DEM) of the Province of Siena, with a spatial resolution of 75 m, was acquired in order to derive elevation, slope, aspect and incident solar radiation variables and to evaluate the differences in topography between the areas that experienced afforestation or deforestation processes. According to the resampling procedure of forest maps described in Section 2.2, we resampled the DEM to a 100 m spatial resolution in order to match the spatial resolution of the forest grids used for the landscape change analysis. In order to resample the DEM, a nearest neighbour algorithm was chosen, because it does not alter the original pixel brightness (Duggin and Robinove, 1990; Rocchini, 2004).

The estimate of direct incident radiation (hereafter referred to as radiation) was performed by a multiple regression algorithm based on a trigonometric function of slope, aspect and latitude as described by McCune and Keon (2002). This parameter provides an approximation of the total amount of radiation received by a point on the ground given its latitude, slope and aspects.

For each forest dynamic process (afforestation, deforestation and conservation) we extracted the values of the topographical variables (elevation, slope and direct incident radiation) and compared them in terms of differences in topography. In particular, we firstly generated three Boolean maps, one for each forest dynamic process (afforestation, deforestation and conservation). Each map was formed by pixels with Boolean values (0/1), showing the event of the considered process (afforestation, deforestation and conservation). The values of topographical variables (elevation, slope and direct incident radiation) were extracted per each dynamic process. Hence, afforestation, deforestation and conservation were compared in terms of their differences in elevation, slope and radiation

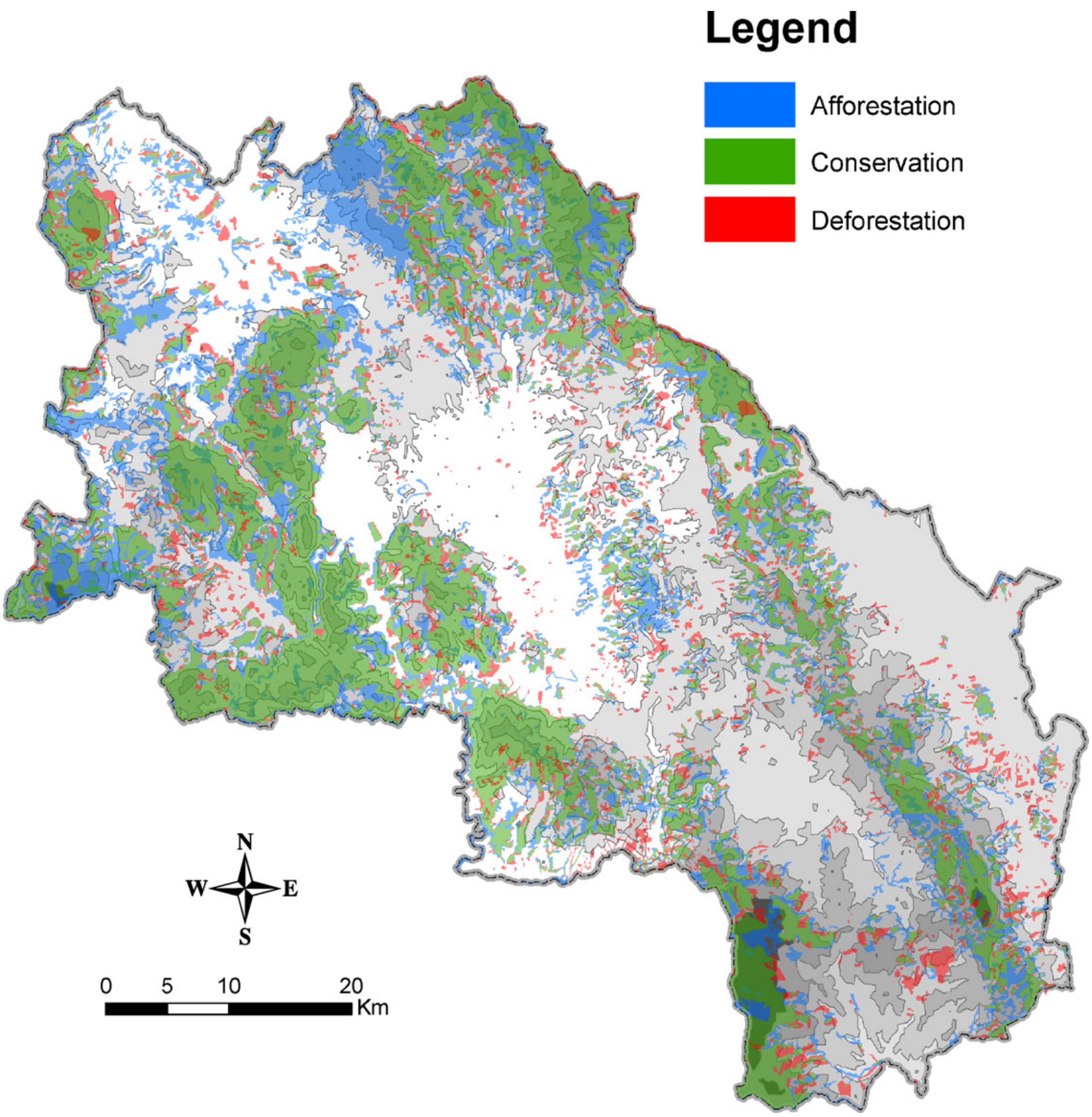

Fig. 3. Cross-classification image derived from the overlay procedure between the 1933 historical map and the 2000 Corine Land Cover. In this image each pixel shows the combination of categories in the two images that are being compared. Notice that the Digital Elevation Model was overlaid to the cross-classification image (For interpretation of the references to color in this figure legend, the reader is referred to the web version of the article).

Table 1
Contingency matrix (value in km^2) showing the transitions between different forest types and unforested land from 1933 to 2000.

	2000 forest				Total
	No forest	Broad-leaved	Coniferous	Mixed	
1933 forest					
No forest	2053.8	502.7	19.6	48.9	2625.9
Broad-leaved	263.8	775.5	28.5	67.1	1134.9
Coniferous	1.5	2.6	3.7	4.5	12.3
Mixed	5.5	30.4	5.5	7.1	48.5
Total	2324.6	1311.2	57.3	127.6	

by using univariate statistics and applying the Kruskal–Wallis non-parametric test.

All the spatial analyses were performed using Arcview 3.x and extension Grid Tools v. 1.7 by Jenness Enterprise (Jenness, 2006) and Idrisi (Eastman, 2003). The statistical package R (R Development Core Team, 2007) was used for statistical analyses.

3. Results

3.1. Landscape metrics

The overlay of the two maps showed that the larger forest patches were maintained during the 1933–2000 period (Fig. 3). Deforestation concerned a high number of relatively small-sized patches, while afforestation happened in both small and large size patches (Fig. 3). In addition, conservation and afforestation were both largely associated with mountains or hill ridges. The plain areas were mostly without forests in both periods, and only showed a certain pattern of deforestation of small-sized forest patches (Fig. 3).

The overall trend corresponded to an absolute increase in forest cover of about 300 km^2 (about 25% of the forest surface present in 1933); this was mostly due to the increase in broad-leaved forests (Table 1). In relative terms, the highest changes were those of coniferous forests, with an increase of about 365% (from 12.3 to 57.3 km^2, see Table 1), while broad-leaved forests increased about 15.5%. The Class Area index (CA, Fig. 4a) showed the quantitative changes in forest cover of the different classes as obvious from the map (Fig. 3).

The forest surface showed a pronounced fragmentation in 1933, with a high number of patches (Fig. 4b), and this fragmentation dramatically decreased in 2000, with the exception of mixed forests (Fig. 4b). The general trend of increase in forest area and of the reduction in the number of patches was reflected by the increase in mean patch size, observed for both broad-leaved and coniferous forests (Fig. 4c): from 94 to 271 ha and from 14 to 85 ha for broad-leaved and coniferous forests, respectively. Mixed forests, on the other hand, showed an opposite trend. The patch size standard deviation index (Fig. 4d) showed an increase in the size variability for broad-leaved and coniferous forests and a corresponding decrease for mixed forests.

The shape of the forests also changed in a clear way, with an increase of AWMSI (area weighted mean shape index) for the broad-leaved and conifer forests (Fig. 4e), while mixed forests showed a slightly different pattern. In general, an increase in the geometric complexity of forest patch shapes was observed. The investigation of the edges length showed a substantially similar total edge (TE) between 1933 and 2000 (Fig. 4f), with a relatively high increase for mixed forests, while mean patch edge (MPE) showed a marked increase for broad-leaved and conifer forests and a slight increase for mixed forests (Fig. 4g).

3.2. Topographical analysis

The sites with afforestation, deforestation or conservation processes significantly differed in elevation, slope and solar radiation (Table 2), as shown by the Kruskal–Wallis test. The sites that experienced afforestation processes were on average more elevated and steeper but with a lower radiation with respect to the sites that experienced deforestation processes. The sites where forest was conserved were even more elevated and steeper than the sites experiencing afforestation (Table 2).

4. Discussion

In this paper, we made use of historical maps derived from manual digitisation of polygons without knowledge about the uncertainty related to polygon boundaries or thematic attribution. These are expected to contain a certain amount of bias in the spatial configuration of forest patches and consequently problems in analysing the spatial variation of forest patterns. This bias can derive from (i) a subjective definition of the minimum mapping unit (MMU, see e.g. Burnett and Blaschke, 2003; Rocchini et al., 2006), (ii) the impossibility of checking the uncertainty of thematic attribution (Bailey et al., 2007; Rocchini and Ricotta, 2007), (iii) poor information available to check for reproducibility of the mapping process (Cihlar, 2000). In addition, given the dependence of almost all the landscape metrics on the spatial resolution associated with the investigated patches (Jelinski and Wu, 1996), the same spatial resolution and mapping procedure should be used for comparing maps from different areas or different time of the same area. In fact, the study of the landscape structure and dynamics cannot ignore the scale of observation (Turner et al., 1989; O'Neill et al., 1991, 1996; Wu et al., 2000). As pointed out by Turner et al. (1989), measures made at different resolutions cannot be directly compared.

Table 2
Univariate statistics considering topographic variables (altitude, slope and solar radiation) per each forest dynamic (afforestation, deforestation and forest conservation). Mean and standard deviation are shown, as well as the results of the Kruskal–Wallis tests. Notice that (i) univariate statistics concerning slope were based on circular statistics according to Fisher (1993), (ii) direct radiation is reported on a logarithmic scale for comparability with McCune and Keon (2002) who developed the regression function for its calculation, but it can be returned to an arithmetic scale by an exponential function.

	Altitude (m)	Slope (°)	Direct radiation (ln(MJ/cm^2/year))
Afforestation	385 ± 164	7.58 ± 4.41	0.892 ± 0.060
Deforestation	360 ± 148	6.13 ± 3.54	0.902 ± 0.042
Conservation	435 ± 185	8.71 ± 4.66	0.885 ± 0.052
Results of the Kruskal–Wallis test H ($\alpha = 0.05$)	134.2	172.5	22.6

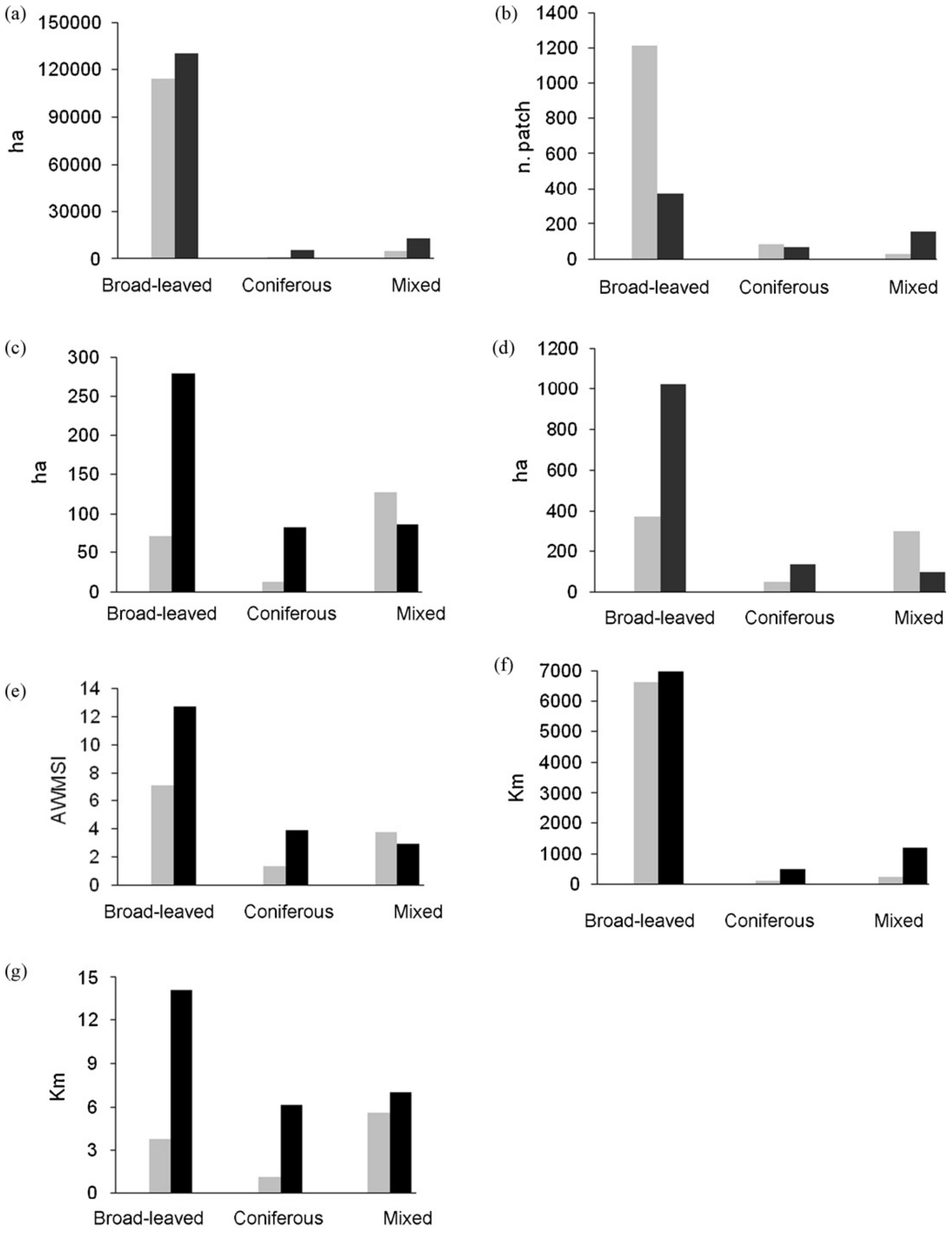

Fig. 4. Results of landscape metrics ((a): CA; (b): NumP; (c): MPS; (d): PSSD; (e): AWMSI; (f): TE; (g): MPE). In grey the values for the 1933 historical map, in black those for the 2000 Corine map.

Nevertheless, the use of maps obtained by manual digitisation represents the only possible way of gathering information about the spatial configuration of ancient forests. It should be noted that, in general, classification is in its very nature a human construct, with the analyst being a key node of the entire classification process. Any attempt to standardise the classification process is, of course, scientifically sound, but the “human factor” is always part of the process (Burnett and Blaschke, 2003).

In this paper we accounted for the uncertainty in the accuracy of manually derived maps by resampling procedures. Of course, we are aware that, when there is no possibility to directly check for the accuracy of historical information (e.g. by remotely sensed imagery) the above-described bias may impact further analysis.

Even with the above discussed limitations, some clear trends emerged for the investigated Mediterranean area (the Province of Siena), i.e. the increase of forests, mostly depending on: (i) the abandonment of pasture and cultivation activities on hilly or mountain sites (see also Rocchini et al., 2006) and (ii) the afforestation of scarcely productive soils, such as ultramafic substrates (Chiarucci and De Dominicis, 1995) or eroded slopes (Gautam et al., 2004). This phenomenon is mostly due to the social-economic changes that occurred in Italy after about 1950, and that resulted in deep land-use changes (Vos and Stortelder, 1992).

Although the general trend of afforestation resulted in an increase in forest surface, many small forest patches interspersed in the agricultural or urban matrix (forest remnants) in the plain areas disappeared during the analysed period. On the other hand, the hilly and mountain areas experienced a parallel disappearance of the open areas patches, once used as small agricultural lands or pastures. These open areas patches were abandoned and experienced a consequent spread of forests, that merged and included the smaller forest patches previously existing in the surroundings. This type of landscape dynamics has already been observed for other areas of central Italy as a result of the mountain abandonment (e.g. Vos and Stortelder, 1992; Rocchini et al., 2006). This led to a reduction of the number of forest patches and an increase in the average forest patch size. The trend of forest defragmentation also resulted in a higher variability of the perimeter and shape of forest fragments, as already observed in other studies (Forman, 1995; Moreira et al., 2001).

The three considered forest classes showed an increase in their total surface cover, with the coniferous forests showing the highest relative increase (more than 400%). This result reflected a process affecting the whole Italian country during the post-war period, when many lands were devoted to the plantation of native or exotic conifer species. The main reasons for the re-afforestation planning during the last century were mainly related to soil protection, slopes reinforcement and the reduction of water runoff in the mountainous and hilly watersheds (AA, 2000). It is worth remembering that, even if exogenous environmental factors, such as climate, soil type, stochastic disturbances, and geomorphology (Cox and Moore, 2004) may impact land use, human management represents a crucial driver of land-cover patterns (Antrop, 2005). As an example, the high rate of forest deforestation in the plains (or at low altitudes) found in this paper is likely to be due to an increase in human pressure on these areas. On the contrary, forest conservation has been found in sites with unsuitable conditions, i.e. those with higher elevation and steeper slopes. This can be explained by the high cost expected for extending modern agricultural practices or the difficulty of setting new urban settlements in these areas (Meeus, 1993; Rühl et al., 2005). A lower solar radiation was found within forested areas compared to deforested ones. This phenomenon can be explained by considering the vocation for agriculture activities within areas with higher solar radiation, while areas with cold exposures commonly do not fit cultivation scopes and are more easily reforested (Field et al., 1995; Caviglia et al., 2004; Fuhrer et al., 2006). This trend can be related with the shift from a traditional self-sufficiency agricultural system, characterised by a high complexity in its organisational schemes, to a new agricultural scheme, with the support of modern technologies. Such practices intensify on most fertile and better-connected areas, leaving aside less profitable ones (García-Ruiz and Lasanta, 1990).

5. Conclusion

The analysed Mediterranean landscape (the Siena Province), as most of the hilly Tuscan landscape, showed important changes in the last decades with a marked increase in forest surface. This was certainly due to socio-economic changes such as the abandonment of countryside and mountains, and the subsequent colonization of previously agricultural and pasture surface by forest vegetation, but also to the plantation of conifers. Similar patterns were observed in mountain and hilly habitats, leading to an increase in their naturalness, while the diffusion of intensive monocultures and the enlargement of urban settlements determined a parallel reduction of the forest cover in the plain areas (Luoto, 2000; Zechmeister and Moser, 2001).

On the one hand, the historical landscape dynamic produced an increase in naturalness, in particular in habitats at higher elevations. On the other hand, the decrease in traditional land-use practices dramatically changed landscape structure, including the replacement of manpower by technology (machines, biocides, chemical fertilizers), with a loss of non-monetary values (Pinto-Correia and Vos, 2004).

As in many other landscapes, two types of processes were active: (i) the abandonment and fragmentation of semi-natural remnants and (ii) the intensification of industrial agricultural practices, with the destruction of forest remnants and the homogenization of the landscape (Vos and Stortelder, 1992). We demonstrated how the abandonment of typical human practices and the intensification of modern agricultural practices may lead to the conversion of a complex landscape matrix into a homogeneous system, by reducing the its landscape diversity.

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