Patterns and drivers of forest landscape change in the Apennines range, Italy

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Malandra Francesco[1], Vitali Alessandro[1], Urbinati Carlo[1], Weisberg Peter J.[2], Garbarino Matteo[3]

Malandra F. · Vitali A. · Urbinati C.

[1]Marche Polytechnic University, Department of Agricultural, Food and Environmental Sciences, Via Brecce Bianche 10, 60131 Ancona, Italy

f.malandra@pm.univpm.it

alessandro.vitali@univpm.it

c.urbinati@univpm.it

Weisberg P.J.

[2]Department of Natural Resources and Environmental Sciences, University of Nevada, Reno, Mail Stop 186, Reno, NV 89557, USA

pweisberg@cabnr.unr.edu

Garbarino M.

[3]University of Turin, Department of Agricultural, Forest and Food Sciences, Largo Braccini 2, 10095 Grugliasco, Italy

matteo.garbarino@unito.it

Corresponding author: Vitali Alessandro

e-mail: alessandro.vitali@univpm.it

Phone number: +390712204274

ORCID ID: 0000-0002-1713-2152

ABSTRACT

Human activities and natural processes over millennia have shaped the forest landscapes of European mountain ranges. In the Apennines, the second largest range in Italy, the post-World War II abandonment of traditional activities has led to forest expansion. Previous analyses of land-use change related to forest landscape were performed for relatively small localities and used different sampling protocols. Consequently, a replicate landscape approach and a systematic sampling design were crucial for quantifying changes at regional scale. We investigated land-cover change and landscape configurational shifts comparing different slope exposures and altitudinal zones and discussed the main drivers affecting post-agricultural forest dynamics. We selected two paired study landscapes (North-East vs. South-West) of 16 km^2 for each of 10 sites located along the entire range. We applied object-based classification to aerial photography from 1954 and 2012, resulting in 40 land-cover maps. We assessed: i) overall landscape changes by computing land-cover transitions; ii) landscape patterns through key metrics; iii) reforestation dynamics through multivariate statistics and binomial generalized linear models (GLMs). Apennine landscape mosaics experienced structural simplification at lower elevation due to tree establishment in abandoned pastures, but a diffuse fragmentation of historical grasslands at higher elevation due to development of woody vegetation patches beyond the forest-grassland ecotone. Forest expansion occurred more rapidly at lower elevations, on steeper slopes, and closer to existing forests and cultivated areas. A replicate landscape approach proved useful for quantifying changes to forest cover and landscape structure along complex gradients of topography and land-use history, following a diffuse agro-pastoral abandonment.

KEYWORDS: land use change; mountain forest landscape; landscape mosaic; reforestation; Apennines; forest regeneration.

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INTRODUCTION

Land-use change in mountain landscapes

Land-use change (LUC) is one of the main drivers affecting mountain ecosystems globally (Bugmann et al. 2007). LUC phenomena are occurring at unprecedented rates and magnitudes, and interact with ecosystem processes, biogeochemical cycles, biodiversity and climate (Turner et al. 1994). LUC regimes are defined by the type, intensity, extent, duration of land-use, as well as by the spatial and temporal scales of analysis (Turner et al. 1994). Historical land-use is widely considered a fundamental constraining factor driving current landscape configuration (Gimmi et al. 2008; Garbarino et al. 2013) and constraining future landscape response to environmental change (Foster et al. 1998).

Human pressure and consequent landscape modifications

In Europe, mountain areas have been deeply transformed by human presence (Debussche et al. 1999; Geri et al. 2010a) so that ecosystems and biota coevolved under anthropic pressure, generating the so-called "cultural landscapes" (Naveh 1995). However, during the last century, European mountain landscapes have experienced a progressively decreasing intensity of human impacts (Debussche et al. 1999) due to the decline of small-scale agriculture, pastoralism and forest utilization, especially in areas of marginal productivity for agriculture (Chauchard et al. 2007). The increasing abandonment of mountain and rural areas, often triggered by the decline of livestock grazing (MacDonald et al. 2000; Fernández et al. 2004), induced a natural expansion of forest cover arising from secondary succession or gap filling in pre-existing woodlands (Améztegui et al. 2010). Natural reforestation is a heterogeneous and site-dependent process (Garbarino et al. 2013) that is driven by topographic, climatic and socio-economic factors (Debussche et al. 1999). The future landscape structure depends on how such processes interact over time. These are common dynamic processes observed across Europe, from the Spanish (De Aranzabal et al. 2008), and French Pyrenees (Roura-Pascual et al. 2005), to the Greek mountains (Petanidou et al. 2008), as well as in the Alps (Tasser et al. 2005) and Carpathians (Weisberg et al. 2013). Biodiversity loss and structural simplification are commonly reported outcomes of land abandonment in Mediterranean mountain ecosystems such as the Apennines (Falcucci et al. 2007; Petanidou et al. 2008). Worldwide, the effects of farmland abandonment on biodiversity is still debated. Some researchers consider it a threat and others an opportunity for habitat regeneration. In various regions of the world, both negative and positive effects are reported (Plieninger et al. 2014; Queiroz et al. 2014).

Farmland abandonment and forest expansion in the Apennines (Italy)

The Apennines are the second largest mountain range of Italy, extending along the peninsula for over 1200 km. Strongly heterogeneous natural features have interacted with human pressure to shape the forest landscape mosaic, which is very rich in plant biodiversity. During the late Holocene (after ca. 6000 years BP), the Apennines forest landscape was dominated by broadleaf forests, intensively coppiced and extensively converted to cropland or rangeland until the 1950's (Vacchiano et al. 2017). Coniferous forests are naturally present at only a few sites, but between the 1930's and 1980's, approximately 1 million hectares of pine and spruce forest were planted to reduce the severe slope erosion induced by former over-exploitation of steep mountain slopes (Vacchiano et al. 2017). Moreover, the outlawing of sharecropping and tenant farming in the 1950's caused a diffuse abandonment of resource use in marginal areas and a severe depopulation in mountain municipalities (Falcucci et al. 2007; Bakudila et al. 2015). This in turn led to widespread forest expansion into abandoned grasslands and croplands (Cimini et al. 2013) and an overall decrease of landscape heterogeneity (Peroni et al. 2000).

Previous analyses of LUC in the Apennines have been implemented for relatively small localities and have used varying sampling protocols, such that they are often not directly comparable (Malandra et al. 2018). To better understand the influence of LUC on landscape structure at the regional scale, we conducted a land-cover change analysis of the entire Apennine range with a homogeneous sampling design and a rigorous method of image analysis, using 20 replicate mountain landscapes. Our goals were: i) to identify the most important land-cover transitions over the 60-year period, at the two prevailing slope exposures (North-East vs South-West) and at lower and higher elevations (>/< 1300 m a.s.l.); ii) to measure the mosaic shifts that occurred at each landscape over time along elevational gradients; and iii) to detect the main drivers (natural or human-induced) affecting forest cover change. We hypothesized that natural reforestation would occur mainly in mountain areas where the decrease of agro-pastoral activities is associated with favorable site conditions. Assuming the same abandonment rate, we expected that at lower elevation sites, where mean annual temperatures are higher and the growing season longer, reforestation should be more relevant. Moreover, we expected forest expansion to be significantly greater on warmer SW slopes, subjected to more intensive past land use and providing more suitable conditions for natural reforestation after abandonment (Vitali et al. 2017).

MATERIAL AND METHODS

Study areas

Our ten study areas are all located within the Apennines along 4.30° of latitude (about 660 km), extending from North -East (NE) to South-West (SW) between 38-45° N and 8-17° E. They encompass a region 1200 km in length and 40-200 km of width, from the Ligurian sea to the Calabrian tip. The study areas encompass several mountain peaks higher than 2000 m a.s.l., from Mt. Cimone (North) to Mt. Pollino (South) together with comparable land covers along the altitudinal gradient and suitable for a change-detection analysis. The highest elevation is Corno Grande (2914 m a.s.l.) of the Gran Sasso massif in the central Apennines (Figure 1). Most of the study areas are included in the European Union Natura 2000 network of protected sites: about 78.5% of the analyzed areas is in the European Union Natura 2000 network of protected sites. Mean annual temperatures range from 6.2 to 10.0 °C and annual precipitation ranges from 730 to 877 mm. NE slopes (Adriatic side) are in general more continental than SW slopes (Tyrrhenian side), whereas precipitation is greatest for NE slopes. At each study area, we analyzed two paired study landscapes (NE and SW aspect), each extending for 16 km^2. The 20 study landscapes cover a total surface of approximately 32000 ha within an elevation range of 347-2500 m a.s.l., including all vegetation zones, from hilly (< 600-m a.s.l.) to alpine.

The forest cover is largely dominated by broadleaf forests, belonging to the Mediterranean and temperate forest biomes. Lower elevations and steep rocky slopes host xeric oak forests dominated by *Quercus pubescens* and *Quercus ilex*. Deciduous forests of *Quercus cerris, Ostrya carpinifolia, Acer* spp., and *Castanea sativa* dominate the sub-montane zone. *Fagus sylvatica*, locally mixed with *Abies alba*, largely dominates the montane zone. Especially in the central and southern sectors of the Apennines, *Pinus nigra* forests were planted during the mid-20th century to reduce slope-erosion (Piermattei et al. 2016). Limited natural forests of *Pinus mugo* and *Pinus heldreichii* (Vitali et al. 2017) occur at higher elevations.

Image analysis

We collected, processed and analyzed two types of aerial imagery: i) 1954-1955 flight aerial photos (b/w, 1 m cell size) from IGMI (Italian Geographic Military Institute) GAI (Italian Aerial Group); ii) 2010-2014 orthophotos from AGEA (National Agency for Funding in Agriculture) (RGB, 0.5 m cell size). For Mt. Pollino only, we processed 1948 IGMI b/w photos and 2003 AGEA orthophotos. Here we refer to 1954 for older aerial photos (1948, 1954, 1955) and to 2012 for newer ones (2003, 2010-2014). Several IGMI 1948 images were scanned at 1200 DPI, mosaicked and resampled at 1 m resolution. Mt. Pollino is a representative southern location, where peaks > 2000 m a.s.l. are very rare. Historical GAI aerial photos were orthorectified using the AGEA orthophotos and a 20-m resolution DTM (Ispra – Italian Institute for Environmental Protection and Research) as reference data. We used PCI Geomatica 2012 software for geometric correction of historical images (mean RMSE overall = 23 m ± 2 SD; mean RMSE for Mt. Pollino = 82 m ± 2 SD). To facilitate the comparison between historical and recent aerial photographs, we resampled the higher resolution (0.5 m) AGEA images to 1 m as for the IGMI images. We applied a semi-automatic object-based classification by combining the automatic segmentation through eCognition software (scale factor 100, color factor 0.5) with on-screen photointerpretation of segmented polygons (Garbarino et al. 2013). For the 40 land-cover maps (20 landscapes x 2 time periods) each polygon was classified into 9 land-cover classes: bf (broadleaf forest), cf (conifer forest); sh (shrubland), dg (dense grassland dense), sg (sparse grassland), or (orchard, vineyards, other tree groves), cr (cropland, herbaceous crops in general), un (unvegetated, bare soil and water bodies), ur (urban, buildings and infrastructures). The 40 land-cover maps (see examples in Figure S2, S3) were post-processed in ArcGIS 10.4 software so as to enforce consistency among the two datasets (Figure S1). This two-step process aimed for a minimum mapping unit (MMU) of 100 m^2. At first, the polygons with surface area < 100 m^2 were merged with neighboring larger ones by using the ArcGIS tool "Eliminate". After a rasterization of vector data (1 m resolution) the raster maps were smoothed by using a moving-window (3 x 3) majority filter (Jensen et al., 2001). Overall classification accuracy (Figure S1 – table insertion) ranged from 70% (Morone SW 1954) to 96% (Gorzano NE 2012) with a K coefficient between 62% (Cimone SW 1954) and 92%

(Gorzano NE 2012). For validation data, we randomized 100 points on each map and classified them visually using the same land-cover categories adopted in the automatic segmentation.

Data analysis

For the change detection analysis, land-cover raster data were divided into two altitudinal zones above (H) and below (L) 1300 m a.s.l. of elevation, obtaining 4 sub-landscapes for each study site. We adopted a 1300 m a.s.l. threshold after a preliminary analysis of forest cover elevation, in order to separate and analyze forest cover into two altitudinal belts equally represented in each landscape. The land-cover change analysis provided 20 transition matrices combined to detect overall transitions and differences between NE-SW exposures and L-H elevation zones. We performed the overall transition analysis using the 20 transition matrices but we excluded the two Pollino study landscapes from the NE-SW and H-L land-cover change analysis due to fundamental differences in physiography and quality of the photogrammetric materials, leaving 18 transition matrices. We converted the overall transition matrix into a transition diagram showing gain, loss, net change and persistence for each land-cover category (Cousins 2001).

To analyze 1954-2012 landscape patterns of the 20 study landscapes, we computed suitable landscape and class metrics from each raster image using the Fragstats 4 statistical package (McGarigal and Marks 1994). We selected 5 metrics (patch density PD, patch area mean AREA_mn, mean shape index SHAPE_mn, contagion index CONTAG and Simpson's diversity index SIDI) for the analysis after excluding other metrics that were highly correlated (Pearson's $r > 0.8$) (Riitters et al., 1995) and ecologically redundant (Tischendorf 2001). We ordered our residual 36 study landscapes through multivariate ordination using principal components analysis (PCA) based on a main matrix of the five landscape metrics, indirectly related to a secondary matrix of environmental variables (elevation, slope, temperature and precipitation) and anthropogenic variables (population density and urban cover). PCA was performed with the statistical package PcORD 7. The statistical significance of the ordination analysis was tested using a Monte Carlo permutation method based on 10000 runs with randomized data.

Moreover, we explored the statistical distribution of the 5 landscape metrics over the 1954-2012 period, comparing high and low elevation belts, but substituting contagion index with aggregation index (AI). Using the latter, each class is weighted by its proportional area in the landscape becoming more suitable when comparing the two paired elevation belts with different surface areas. Then, we calculated three representative class metrics (patch density, mean patch area, and aggregation index) for a more focused analysis of changes to the broadleaf forest class. We applied the Wilcoxon paired test to assess statistical differences in median values of the metrics between the two exposures and between the two elevational ranges.

To assess land abandonment in the forest landscapes of the Apennines, we also used demographic data from the national population census carried out for each municipality every ten years (Vitali et al. 2017). From the complete dataset (ISTAT - Population Census 1871-2011) we used the interval 1951-2011 (ISTAT 1951, 2011) subtracting the population densities (inhabitants/km^2) averaged over the two years for the municipalities included in the selected study landscapes.

We explored the main drivers of broadleaf forest transitions by rescaling the spatial resolution of land-cover raster maps (1 m) to the minimum resolution of topographic variables (DEM 10 m TINITALY) (Tarquini et al. 2012). We then used in the analysis only rescaled pixels with a minimum cover threshold of 51% of a single dominant category. We limited the analysis to those categories more prone to a transition to broadleaf forest: sparse grassland (sg), dense grassland (dg), unvegetated land (un) and shrubland (sh). We built a transition map for each landscape and from the whole dataset we extracted only the pixels showing potential shift from non-forest to forest. For each transition (e.g. shrubland to forest), we calculated a Boolean map indicating transient pixels (1 = forest cover in 2012) and non-transient pixels (0 = non-forest cover in 2012). These binomial values were obtained by the response variable (reforestation) in the models. We fitted binomial generalized linear models (GLMs) to predict the transition to forest cover as a function of three topographic variables (elevation, slope, north-eastness index) and three land-cover variables (proximity to former forest, proximity to former cropland and proximity to former urban area). We ranked all the potential models according to the Akaike Information

Criterion (AIC) and then selected the most parsimonious models showing the lowest AIC value (Burnham and Anderson 2002). We also used the Akaike weights (*Wi*) of each model to measure the conditional probability of the candidate model with the greatest empirical support. All GLMs were run with the R software (R Core Team 2018), using the 'glm' function of the package *stats*. We performed model selection using the *MuMIn* package (Bartón 2017). We checked for collinearity of predictors using the 'vif' function of package *rms*.

RESULTS

Land use change and landscape features

Concerning land-cover transitions, 40.7% (> 13000 ha) of the total surveyed area of the Apennines changed land-cover class (Figure 2 and table S2). Land-cover categories with net increases included broadleaf forests with the highest increment (4452 ha, +34%), conifer forests (1064 ha, +114%), shrubland (180 ha, +16%) and urban areas (109 ha, +46%). Negative transitions predominantly occurred in croplands (-2237 ha, -76%), orchards (-174 ha, -72%), dense grasslands (-2251 ha, -33%) and sparse grasslands (-1204 ha, -22%) (Figure 2 and table S2).

The mean landscape percentage of forest cover is above 54% and is largely dominated by broadleaf forests (bf) (>50%, Table S1) that experienced 51% of the overall change that occurred. Broadleaf forest is the land-cover category with the highest range of variability among studied landscapes within each time period (Figure 3) followed by dense and sparse grasslands. Conifer forests, orchards, croplands and unvegetated lands appeared more stable through time but have the greatest share of outlier sites with the greatest cover differences. Differences in the areal coverage of land cover types from 1954 to 2012 are statistically significant for all categories except for shrubland and unvegetated land.

Conifer forest showed the greatest percent increases in land cover at NE aspects (327 ha, +312%) rather than SW aspects (748 ha, +96%) (Figure 4a). Broadleaf forest also increased but to a lesser degree and similarly for both aspects (1954 ha, +43% at SW and 2420 ha, +39% at NE). Urban areas increased twice as much at SW aspects (81 ha, +54%) compared to NE aspects (25 ha, +29%). Croplands and orchards had largely decreased through time (70-100%) but at similar rates across slope aspects (Table S3, S4).

Relative land cover changes varied significantly across an elevational threshold (above and below 1300 m a.s.l.) (Figure 4b). All forest types increased dramatically more at lower than higher elevations: broadleaf 2832 ha, +59% vs. 1537 ha, +26% and conifer 719 ha, +186% vs 351 ha, +70%. Shrubland increased only at lower elevation (208 ha, +50%), but maintained similar cover values at higher elevations. A similar reduction trend occurred between dense grassland and sparse grassland at lower and higher elevations (dg = -59% and sg = -57% at lower elevation; dg = -21% and sg = -17% at higher elevation). Moreover, agricultural cover (crops) experienced a greater relative reduction at higher (around -359 ha, -93%) than at lower elevation (-1862 ha, -73%) (Table S5, S6).

Land-cover change varied in magnitude among the studied landscapes (Table S7). However, land-cover change seemed not to vary consistently along the latitudinal gradient. Broadleaf forest expanded in all studied landscapes with the highest increment at Morrone NE (427 ha, +211%). Morrone SW was the landscape most extensively reforested with conifers by 2012 (466 ha). Sibillini NE (-316 ha, -53%) and Gorzano SW (-265 ha, -44%) lost the greatest cover of dense/sparse grassland respectively. Furthermore, Terminillo SW was the only landscape with no agricultural loss.

Landscape pattern change

First (PCA1) and second (PCA2) axis accounted for 45% and 40% of the total variance respectively (Monte Carlo test, $p < 0.05$) (Figure 5). The first principal component was strongly correlated with mean shape index, contagion index and Simpson's diversity index (respectively $r = 0.73$, $r = -0.90$ and $r = 0.89$), whereas patch density ($r = 0.90$) and mean patch area ($r = -0.84$) were strongly correlated with the second principal component (Figure 6).

The PCA biplot shows a clear separation of studied landscapes through time (1954 – 2012). The direction of change is towards a simplification of patch shape associated with smaller and more numerous patches, along with an increase of spatial aggregation of patches. Overall landscape diversity (SIDI) decreased over time, whereas population density of rural areas decreased, and urban areas increased.

Changes in landscape structure varied with elevational zone (Figure 6). At high elevations, patch density significantly increased whereas mean patch area decreased (Figure 6). At low elevations, patch density and mean patch area did not change through time ($p>0.05$). Shape index decreased significantly at both elevation levels. Diversity (SIDI) decreased significantly at low elevation, whereas aggregation of patches increased. Conversely at high elevations, diversity and patch aggregation did not change across years.

Results of landscape metrics computed at the two elevations showed overall dynamics of mosaic simplification at lower elevation and an increase in landscape fragmentation at higher elevation. Fragmentation was indicated by the observed increase in patch density, that was principally driven by an increased in number of patches in high-elevation land categories, such as shrubland (+4.1 patch/100 ha), sparse grassland (+9.3 patch/100 ha) and unvegetated land (+7.4 patch/100 ha) (see Figure S4 in Supplementary Material).

Class metrics were analyzed for the broadleaf forest category to highlight forest mosaic shifts occurring at the two elevation levels across time. Forest patch density slightly decreased at high and low elevation even if the old and new medians were not significantly different (Figure S5). Mean area of forest patches generally increased (+5.3 ha), more at low elevation than high elevation (respectively +7.7 ha and +1.9 ha average) along with aggregation index, which had the greatest increase at low elevation (+0.8 average). Thus, low-elevation landscapes appeared to experience a greater forest mosaic simplification than was the case for higher elevations.

Forest landscape change in broadleaf forests

The observed increase of broadleaf forests (Figure 2 and table 1) was derived mainly from secondary successions occurring in grassland (61.5%), cropland (20.5%) and shrubland (9.2%). The influence of slope aspect was weak, whereas elevation appeared a more relevant factor given that grassland to forest transitions were greater at higher than at lower elevation (75% vs. 51%) and cropland to forest transitions were greater at lower than at higher elevation (33% vs. 4%, respectively). In general, lower-elevation landscapes showed more dynamic forest expansion. The overall bf mean annual increment over the 58-year period was 0.59 % (SW 0.62%, NE 0.56%, L 0.80% and H 0.40%). Within the 60-year time interval, the mean population density decreased significantly by 34% overall (Wilcoxon Test: $W = 3$, p-value < 0.001). Comparing among slope aspects, we found that population density decreased similarly in the NE and SW municipalities (-22 vs. -20 inhabitants/km^2 respectively). We found that population density (inhabitants/km^2) and forest expansion (ha) were negatively correlated (Pearson's $r = -0.64$).

The binomial GLM analysis highlighted that for the model that accounted for the transitions of all land-cover categories to broadleaf forest (all - bf) the best supported model included all six predictor variables (Table 2). New broadleaf forests expanded in proximity to former bf, at lower elevations, on steeper slopes and far from urban settlements. Other models, built on different transition types, showed that transitions to new broadleaf were primarily associated with proximity to old broadleaf forest, and secondarily associated with lower elevations. Transitions from both cropland and unvegetated lands were positively associated with slope, the second most important variable in our models. We also observed a strong positive influence of distance from urban areas for transitions from both croplands and sparse grasslands to broadleaf forest. Transition from dense grassland to broadleaf forest occurred mainly on steeper slopes and closer to old croplands.

DISCUSSION

LUCs are affecting forest cover dynamics worldwide with significant local differences. In some areas increasing farming and logging caused forest fragmentation and/or deforestation, whereas in many others the rural marginality determined opposite transitions, with secondary forests invading abandoned croplands and pastures (Rudel et al. 2005; Rey Benayas 2007). Following periods of extensive forest clearing to increase farming and livestock grazing, post-abandonment natural reforestation occurred in Mediterranean and temperate biomes of Europe and North America (Flinn and Vellend 2005). In some tropical areas, abandoned croplands are shifting to second growth forest over a longer time (Florentine and Westbrooke 2004). There are examples in Oceania (Endress and Chinea 2019), Puerto Rico (Lugo and Helmer 2004) and eastern Africa (Chapman and Chapman 1999) and even in semi-arid regions of Argentina (Basualdo et al. 2018). In mountain areas of the Mediterranean basin land cover dynamics are faster as reported in the Alps (Tasser et al. 2005; Niedrist et al. 2009), the Carpathians (Kuemmerle et al. 2009; Weisberg et al. 2013), the Pyrenees (Metailiè and Paegelow 2005; Roura-Pascal et al. 2005), in Greece (Petanidou et al. 2008) and Spain (De Aranzabal et al. 2018).

Similarly, the Apennines have experienced dramatic land-cover change and forest expansion dynamics over a 60-year period (1954-2012) (Malandra et al. 2018), affecting almost half the total land surface over a broad elevation range. Unlike other studies in this region (e.g. Benini et al. 2010), we have extended the analysis to the entire Apennine range, selecting study landscapes around the most important mountain groups (> 2000 m a.s.l.). The standardized protocol for image processing of aerial photography enhanced the output precision, confirmed by the high validation scores. We also diversified the analysis according to slope exposure (NE vs SW) and elevation (>/< 1300 m a.s.l.) to assess relationships between land-cover change and forest-dynamics with potential orographic drivers (Améztegui et al. 2010). An additional focus on the land-cover dynamics of broadleaved forests helped to further develop inferences about drivers of land-cover change.

Land use change and topographic factors

The overall forest cover increase, regardless of the scale of analysis, is very close to the 35-48% forest cover increased reported for the Apennines by more local studies (Rocchini et al. 2006). The average annual forest expansion rate (0.5 %/year) is also quite similar to that found by other authors (0.4-0.7%/year) in different sectors of the Apennines (Bracchetti et al. 2012).

In general, shrubland is expected to be the most dynamic land-cover class (Gartzia et al. 2014) and to expand considerably after the withdrawal of agro-pastoral management. However, on the studied landscapes, the observed increase in shrubland cover was relatively moderate. This could be the balanced result of two land-cover transitions occurring simultaneously: one from existing shrublands to broadleaf forest and the other from existing grasslands to shrublands (Malavasi et al. 2018). Additionally, another possible reason might be the direct transition from grassland to forest.

The loss of grasslands in the Apennines is an evident landscape process of recent decades: livestock grazing declines in mountain regions of central Italy between 1961 and 2000 were estimated at approximately 30% for cattle and 33% for sheep and goats (Pelorosso et al. 2009), although reliable data on pastoralism are often scarce or incomplete (Falcucci et al. 2007). The abandonment of grasslands and croplands largely influenced the observed land-cover transitions, with notable differences for aspect and elevation. The more favorable topography and climate conditions of the Apennine SW slopes favored farming and livestock grazing. The greater human pressure induced more relevant land-cover changes following land abandonment (Vitali et al. 2017). On these slopes, where the population density is higher than on NE ones, grasslands shifted more slowly to other land cover types. At NE exposure farming and grazing decreased faster or even disappeared at higher elevations.

At lower elevation, human influence is generally higher and successional dynamics are expected to be faster than at higher elevation, where soil and climate conditions are less favorable (Körner 2007). At high-elevation sites, livestock grazing was more widespread and favored the conservation of grasslands through time, but the transition to other land covers (shrubland or forest) was slower. Low-elevation studied landscapes showed a larger cover reduction, probably facilitated by faster successional processes under less severe environmental conditions. Post-abandonment forest expansion in grasslands and croplands was indeed greater at low-elevation sites also in mountain areas of southern Spain (Fernández et al. 2004). In the central Pyrenees, woody plant encroachment into both types of grasslands was observed to be greater at lower elevations and progressively less intense at increasingly higher elevations (Gartzia et al. 2016). Widespread secondary succession to woody plant species following agricultural abandonment is supported by numerous other studies in European mountain systems, including the Apennines (Rocchini et al. 2006; Palombo et al. 2013). The landscape mosaic of the Apennines is changing also under the effect of the urban area expansion (Falcucci et al. 2007). In general, we observed that urban cover increased especially at higher elevations, due to the higher concentration of tourist resort infrastructures. Nonetheless these results could be biased by the higher detectability of human infrastructures in more recent aerial photos.

Landscape mosaic shift driven by land abandonment

We observed dramatic changes in landscape mosaic structure occurring over the 60-year period, suggesting a shift mostly driven by patch shape simplification and patch density increase. However, there were contrasting trends of landscape configurational changes between lower and higher elevations. Bracchetti et al. (2012) reported a more homogeneous landscape matrix in the Central Apennines, with decreases over time in shape and diversity indices. Similarly, our results suggested an overall simplification of the landscape mosaic (Geri et al. 2010a), mostly at lower elevations. At lower-elevation, abandonment of farming and grazing activities followed by natural forest infilling caused a more homogeneous landscape mosaic. Woody species encroachment in former grasslands is likely to be driving local fragmentation at higher elevations and throughout the region.

The forest recolonization of grassland-ecotones at high elevation, and the in-filling of open areas and forest gaps at low elevation, have both led to an increase of forest patch size through time. This process was globally described in a review paper, summarizing changes in landscape metric behavior in rural mountain and hill landscapes after abandonment processes (Sitzia et al. 2010). Common trends of mean patch area increase were detected although changes in patch density were inconsistent across studies. Even in the Apennines, similar processes have been discussed (Assini et al. 2014). In the Central Apennines, Bracchetti et al. (2012) detected an increasing mean patch area and a decreasing density of woodland patches, rapidly merging into fewer larger patches. They found that this coalescence after tree colonization and woodland expansion is a very fast process.

Driving forces of secondary succession

The processes of broadleaf forest expansion were altitude-dependent. At high-elevation, secondary forests mostly derive from former grasslands; whereas, at low-elevation, contributions to secondary forests were distributed among different land cover classes. This is likely due to the land-cover composition in 1954. Topographic variables, such as slope aspect, have strongly conditioned land and forest use in the Apennines (Vitali et al. 2017). The large-scale removal of forests on SW slopes, occurred in ancient times, today provides the greater potential for forest expansion after abandonment. In Europe, a rural depopulation of 17% between 1961 and 2010 (FAOSTAT 2010) induced extensive land abandonment and forest expansion. In the Apennine municipalities comprising our studied landscapes, the national census data reported a relevant population decrease. This process however exhibited differences according the two main slope aspects and elevation zones, with clear effects in forest cover transitions.

Attempts to correlate forest increase to population change have not always been successful, given the geographic scale of analysis and the lack of appropriate demographic records (e.g. number of active farmers or forest workers) (Vitali et al. 2017). However, our study encompassing the entire Apennines range shows a strong negative correlation between the population of mountain municipalities and forest cover.

All GLM models identified the distance from existing broadleaf forest as an important proximate driver of forest expansion (Abadie et al. 2017). This derives from the species capacity of propagule dispersal which usually occurs in the vicinity to seed sources (Nathan and Muller-Landau 2000). Similar influences of proximity to pre-existing forests have been found by several authors in the central Pyrenees (e.g. Gartzia et al. 2014). Grassland to forest transitions occurred farther from existing settlements. Anthropogenic variables including distance to old cropland and urban areas can negatively influence reforestation, since shrubland transition to forest often occurs in long-abandoned areas. In the Apennines these anthropogenic variables are often more relevant drivers of transitions from shrubland to broadleaf forest than physiographic variables such as slope angle and aspect.

CONCLUSION

The main goal of this work was to develop a generalizable model of Apennine landscapes changes at the regional scale, through targeted sampling of replicate study landscapes within key environmental strata. Since the 1950's, following a period of widespread depopulation and land abandonment, the Apennines have experienced an overall forest expansion (Vacchiano et al. 2017; Malandra et al. 2018). Forest cover gains were similar at the two main exposures (NE and SW), but significantly greater at lower elevation (below 1300 m a.s.l.). We quantified the importance of several key land-cover change drivers such as distance from pre-existing forest, elevation, slope angle and distance from previous croplands. Landscape structural complexity was reduced at lower elevations and experienced an inverse process of fragmentation at higher elevations through time. The withdrawal of traditional agro-silvo-pastoral practices in marginal lands observed in the Apennines is widespread in most European mountain areas (Roura-Pascal et al. 2005; Petanidou et al. 2008; Weisberg et al. 2013; Mallinis et al. 2014; Campagnaro et al. 2017; De Aranzabal et al. 2018). The combined approach of using areal changes of land-use/land-cover and landscape metrics to quantify landscape pattern dynamics appeared a suitable method to infer driving factors of variability and to understand their ecological effects (Geri et al. 2010b; Campagnaro et al. 2017). Moreover, appropriate management actions and suitable regional policy strategies should be implemented in these transient areas to prevent further decline (MacDonald et al. 2000). Extended spatio-temporal lags for this type of analyses provide suitable data for developing land-use models, facilitating the prediction of more reliable landscape changing scenarios and forest dynamics trends, useful tools for land management and landscape restoration.

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TABLES AND FIGURES

Table 1 – The relative proportion (%) of each land-cover category in transition to new broadleaf forest, for all topographic settings combined ("global"), by slope aspect (NE vs SW) and by elevation zone (H vs L). The last two rows report the mean bf absolute annual increment (ha/years) and relative annual increment (%/year), within the entire time span (58 years). * = Pollino landscapes are included.

Contribution to bf	Global * (%)	NE (%)	SW (%)	H (%)	L (%)
Conifer forest	2.3	0.5	4.1	4.0	1.0
Shrubland	9.2	11.3	7.1	11.5	8.0
Dense grassland	40.7	42.4	38.0	51.3	33.5
Sparse grassland	20.8	19.8	19.9	23.7	17.5
Orchard	1.7	0.5	3.3	0.1	2.9
Cropland	20.5	21.3	21.7	4.0	32.5
Unvegetated	3.0	2.8	3.4	4.5	2.2
Urban	1.8	1.5	2.4	0.9	2.4
Broadleaf forest mean annual absolute increment (ha/year)	75.4	41.7	33.7	26.5	48.8
Broadleaf forest mean annual relative increment (%/year)	0.59	0.56	0.62	0.40	0.80

Table 2 - Binomial generalized linear models fitted to transition to forest as a function of topographic and cover variables. Wi is the relative Akaike weight, referring to the relative empirical support for each of the models shown compared to other models (not shown) considered within each transition type. Transition types express potential transition to broadleaf forest. Distance to old broadleaf (Distbf), distance to old cropland (Distcr), distance to old urban (Distur); elevation (Elev), slope (Slop) and North-Eastness index (Nes). Transition type acronyms refer to: all land-cover categories converted to broadleaf (all – bf); sparse grassland to broadleaf (sg – bf); dense grassland to broadleaf (dg – bf); unvegetated land to broadleaf (un – bf); shrubland to broadleaf (sh – bf). P-values of model parameters are < 0.01. * = p-value < 0.05, ** = p-value < 0.1.

Transition types	Unchanged pixels (0)	Changed pixels (1)	Parameters (z-value)	Wi
all – bf	1104073	470706	- 358 Distbf - 185 Elev + 93 Slop - 55 Distcr + 52 Distur - 2 Nes*	0.75
sg - bf	398120	110144	- 197 Distbf - 149 Elev + 75 Distur - 40 Slop - 37 Distcr - 2 Nes*	0.81
cr - bf	182797	111510	- 144 Distbf + 84 Slop - 67 Elev + 40 Distur + 9 Distcr - 2 Nes**	0.71
dg - bf	393909	190710	- 221 Distbf - 133 Elev - 50 Distcr + 35 Slop + 18 Distur	0.66
un - bf	73921	13501	- 79 Distbf + 27 Slop - 26 Elev - 23 Distcr + 5 Distur - 2 Nes**	0.64
sh - bf	55326	44841	- 97 Distbf - 49 Elev - 31 Distcr - 25 Distur + 2 Slop**	0.49

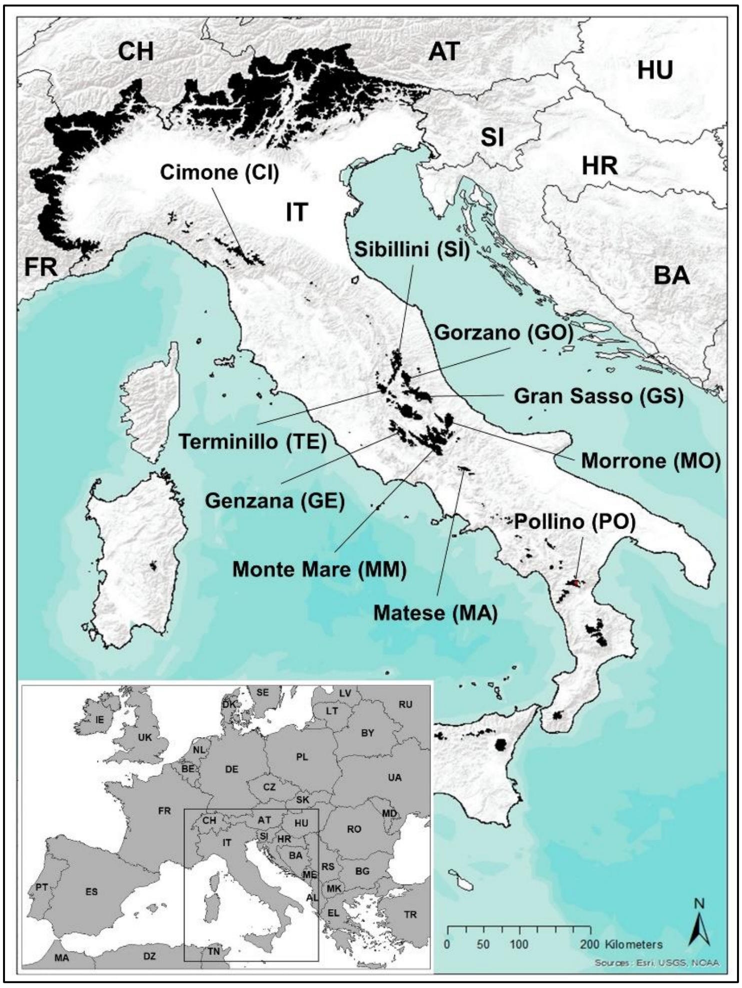

Figure 1 – Geographic distribution of the 10 study areas selected along the Apennines. Areas in black have elevation > 1500 m a.s.l.

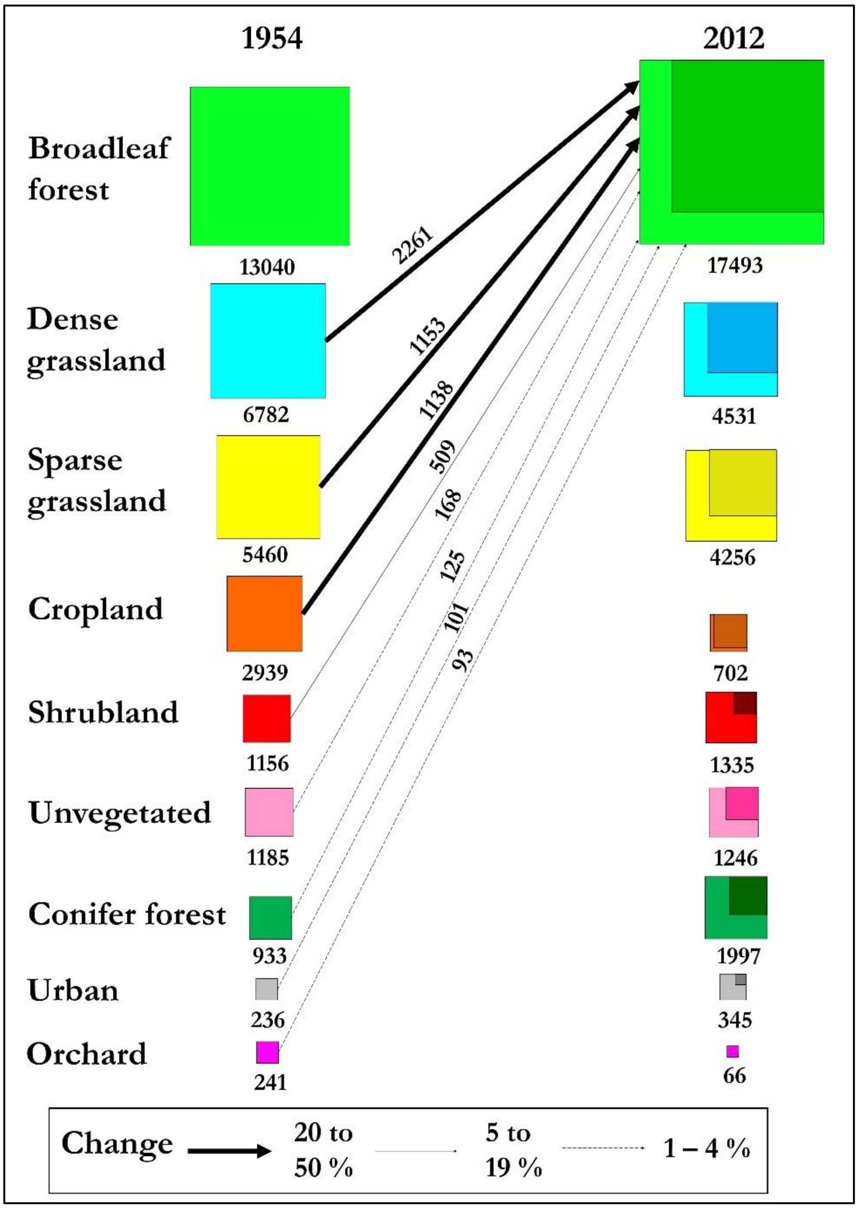

Figure 2 – Area of land-cover classes (ha), and land-cover transitions from past to present in the Apennines study sites. Light-colored boxes are size-scaled land-cover categories. Darker-colored inset boxes represent the relative unchanged surfaces (persistence) of each land-cover class over time. Transitions to broadleaf forests are highlighted with arrows. Arrow thickness increases with magnitude of land-cover changes. The figures above the arrows are hectares of lands converted to broadleaf forests (modified from Cousins 2001).

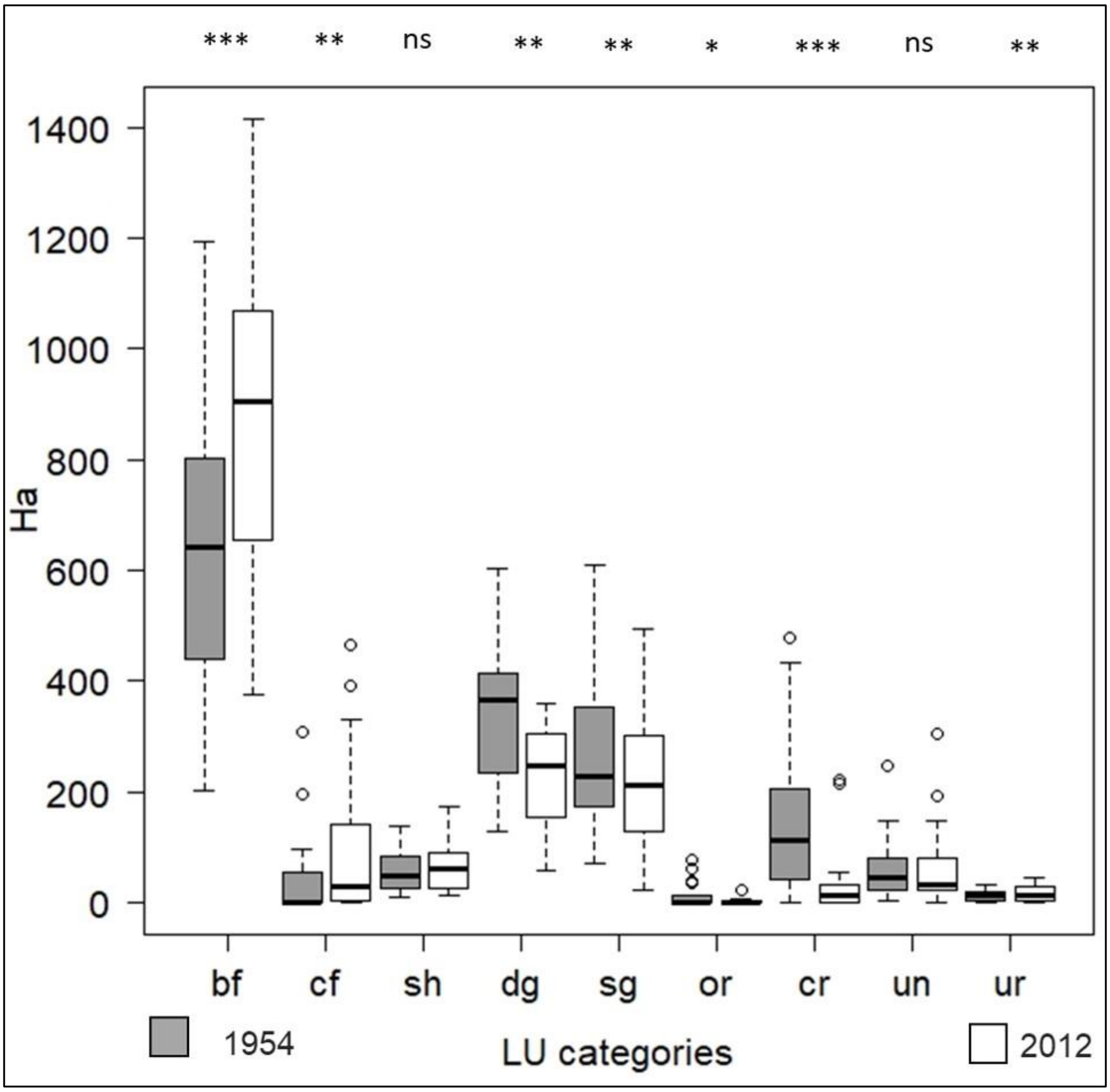

Figure 3 - Mean distribution of land-cover categories (hectares) for the two time periods (1954 and 2012) across 20 replicate study landscapes: broadleaf forest (bf), conifer forest (cf), shrubland (sh), dense grassland (dg), sparse grassland (sg), orchard (or), cropland (cr), unvegetated (un), urban (ur). Horizontal lines are median values and circles are outliers. * = p-value < 0.05, ** = p-value < 0.01, *** = p-value < 0.001, ns = not significant (Wilcoxon paired test to compare 1954 and 2012 covers for each category).

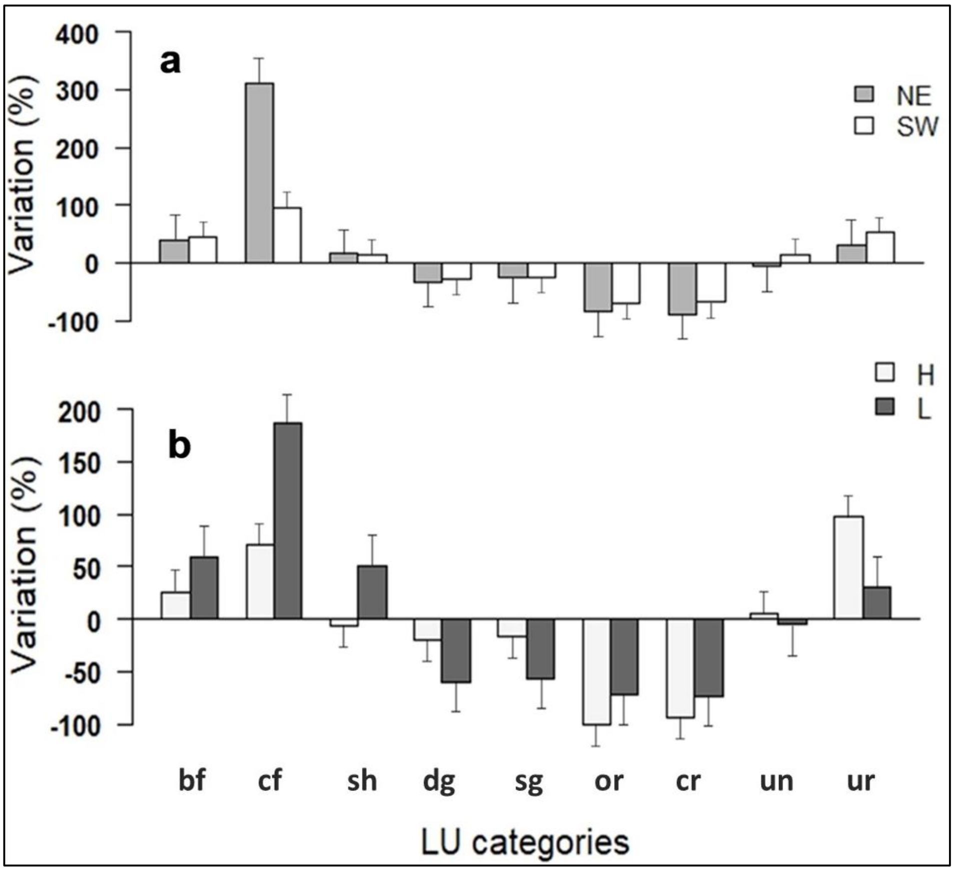

Figure 4 – Relative change (%) of land-cover categories in the 18 study landscapes: a) by main slope aspects (NE vs SW) and b) by elevation (H > 1300 m a.s.l. vs L < 1300 m a.s.l.). Error bars show standard errors. Broadleaf forest (bf), conifer forest (cf), shrubland (sh), dense grassland (dg), sparse grassland (sg), orchard (or), cropland (cr), unvegetated (un), urban (ur).

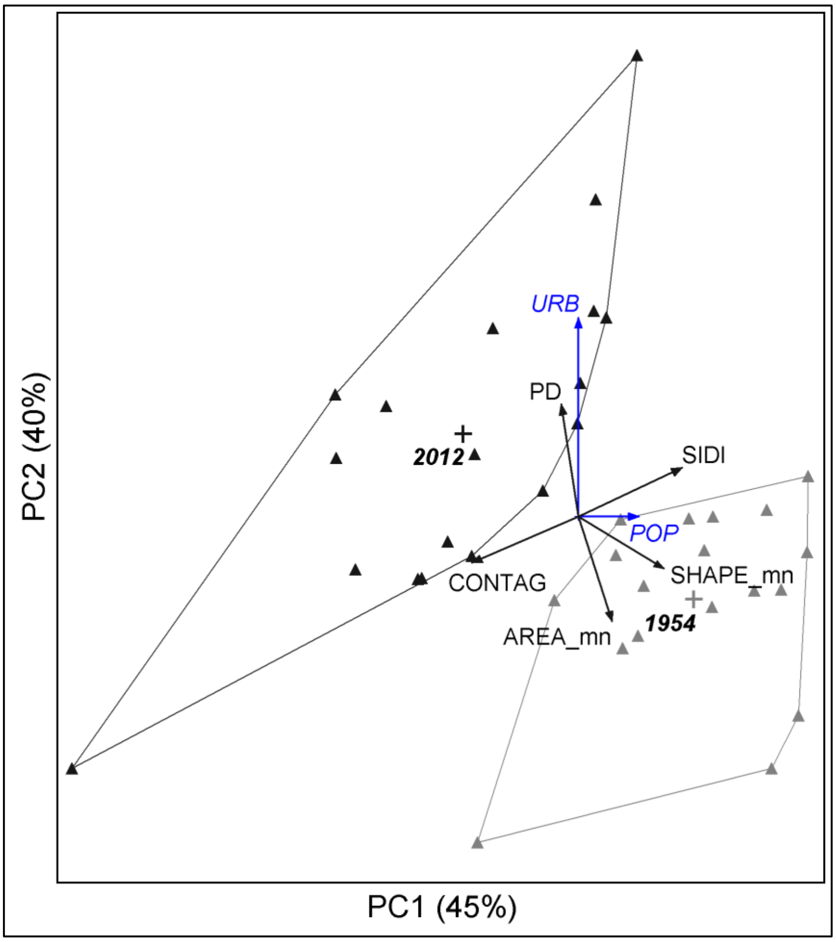

Figure 5 - Principal components analysis of the 36 Apennines forest landscapes covered with this study. Gray and black triangles are site scores in 1954 and 2012 landscapes, respectively and both included within convex hulls. Symbols (+) are centroids of convex hulls. Linear vectors indicate linear correlations of environmental variables with PCA axes. Arrows are landscape structure variables (black) and anthropogenic variables (blue). PD (Patch density); CONTAG (Contagion index); AREA_mn (mean patch area); SHAPE_mn (mean shape index); SIDI (Simpson's diversity index); URB (Urban settlements); POP (population density).

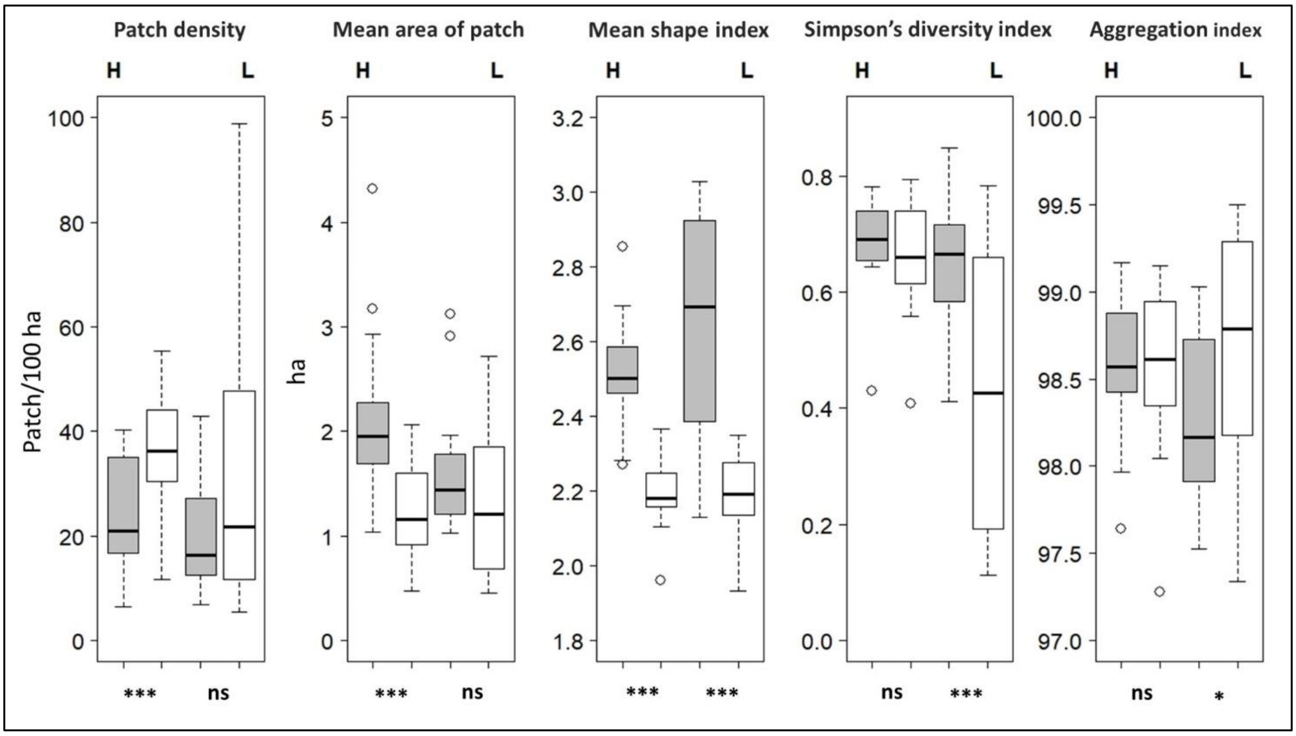

Figure 6 - Mean distribution of landscape metrics for the two time periods (1954 = gray boxes, 2012 = white boxes) and elevation level (H = High, L = Low) across the 18 study landscapes: patch density; mean patch area; mean shape index; Simpson's diversity index; aggregation index. Horizontal lines are the median values and circles are outliers. * = p-value < 0.05, ** = p-value < 0.01, *** = p-value < 0.001, ns = not significant (Wilcoxon paired test to compare 1954 and 2012 indices for each metric).