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

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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

36 drivers affecting post-agricultural forest dynamics. We selected two paired study landscapes (North-East
37 vs. South-West) of 16 km² for each of 10 sites located along the entire range. We applied object-based
38 classification to aerial photography from 1954 and 2012, resulting in 40 land-cover maps. We assessed: i)
39 overall landscape changes by computing land-cover transitions; ii) landscape patterns through key
40 metrics; iii) reforestation dynamics through multivariate statistics and binomial generalized linear models
41 (GLMs). Apennine landscape mosaics experienced structural simplification at lower elevation due to tree
42 establishment in abandoned pastures, but a diffuse fragmentation of historical grasslands at higher
43 elevation due to development of woody vegetation patches beyond the forest-grassland ecotone. Forest
44 expansion occurred more rapidly at lower elevations, on steeper slopes, and closer to existing forests and
45 cultivated areas. A replicate landscape approach proved useful for quantifying changes to forest cover
46 and landscape structure along complex gradients of topography and land-use history, following a diffuse
47 agro-pastoral abandonment.

48

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

51

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58 **INTRODUCTION**

59

60 **Land-use change in mountain landscapes**

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

68

69 **Human pressure and consequent landscape modifications**

70 In Europe, mountain areas have been deeply transformed by human presence (Debussche et al. 1999;
71 Geri et al. 2010a) so that ecosystems and biota coevolved under anthropic pressure, generating the so-
72 called “cultural landscapes” (Naveh 1995). However, during the last century, European mountain
73 landscapes have experienced a progressively decreasing intensity of human impacts (Debussche et al.
74 1999) due to the decline of small-scale agriculture, pastoralism and forest utilization, especially in areas
75 of marginal productivity for agriculture (Chauchard et al. 2007). The increasing abandonment of
76 mountain and rural areas, often triggered by the decline of livestock grazing (MacDonald et al. 2000;
77 Fernández et al. 2004), induced a natural expansion of forest cover arising from secondary succession or
78 gap filling in pre-existing woodlands (Améztegui et al. 2010). Natural reforestation is a heterogeneous
79 and site-dependent process (Garbarino et al. 2013) that is driven by topographic, climatic and socio-
80 economic factors (Debussche et al. 1999). The future landscape structure depends on how such processes
81 interact over time. These are common dynamic processes observed across Europe, from the Spanish (De
82 Aranzabal et al. 2008), and French Pyrenees (Roura-Pascual et al. 2005), to the Greek mountains
83 (Petanidou et al. 2008), as well as in the Alps (Tasser et al. 2005) and Carpathians (Weisberg et al. 2013).
84 Biodiversity loss and structural simplification are commonly reported outcomes of land abandonment in
85 Mediterranean mountain ecosystems such as the Apennines (Falcucci et al. 2007; Petanidou et al. 2008).
86 Worldwide, the effects of farmland abandonment on biodiversity is still debated. Some researchers
87 consider it a threat and others an opportunity for habitat regeneration. In various regions of the world,
88 both negative and positive effects are reported (Plieninger et al. 2014; Queiroz et al. 2014).

89

90 **Farmland abandonment and forest expansion in the Apennines (Italy)**

91 The Apennines are the second largest mountain range of Italy, extending along the peninsula for over
92 1200 km. Strongly heterogeneous natural features have interacted with human pressure to shape the
93 forest landscape mosaic, which is very rich in plant biodiversity. During the late Holocene (after ca. 6000

94 years BP), the Apennines forest landscape was dominated by broadleaf forests, intensively coppiced and
95 extensively converted to cropland or rangeland until the 1950's (Vacchiano et al. 2017). Coniferous
96 forests are naturally present at only a few sites, but between the 1930's and 1980's, approximately 1 million
97 hectares of pine and spruce forest were planted to reduce the severe slope erosion induced by former
98 over-exploitation of steep mountain slopes (Vacchiano et al. 2017). Moreover, the outlawing of
99 sharecropping and tenant farming in the 1950's caused a diffuse abandonment of resource use in marginal
100 areas and a severe depopulation in mountain municipalities (Falcucci et al. 2007; Bakudila et al. 2015).
101 This in turn led to widespread forest expansion into abandoned grasslands and croplands (Cimini et al.
102 2013) and an overall decrease of landscape heterogeneity (Peroni et al. 2000).

103 Previous analyses of LUC in the Apennines have been implemented for relatively small localities and
104 have used varying sampling protocols, such that they are often not directly comparable (Malandra et al.
105 2018). To better understand the influence of LUC on landscape structure at the regional scale, we
106 conducted a land-cover change analysis of the entire Apennine range with a homogeneous sampling
107 design and a rigorous method of image analysis, using 20 replicate mountain landscapes. Our goals were:
108 i) to identify the most important land-cover transitions over the 60-year period, at the two prevailing
109 slope exposures (North-East vs South-West) and at lower and higher elevations ($>/< 1300$ m a.s.l.); ii)
110 to measure the mosaic shifts that occurred at each landscape over time along elevational gradients; and
111 iii) to detect the main drivers (natural or human-induced) affecting forest cover change. We hypothesized
112 that natural reforestation would occur mainly in mountain areas where the decrease of agro-pastoral
113 activities is associated with favorable site conditions. Assuming the same abandonment rate, we expected
114 that at lower elevation sites, where mean annual temperatures are higher and the growing season longer,
115 reforestation should be more relevant. Moreover, we expected forest expansion to be significantly greater
116 on warmer SW slopes, subjected to more intensive past land use and providing more suitable conditions
117 for natural reforestation after abandonment (Vitali et al. 2017).

118

119 MATERIAL AND METHODS

120

121 Study areas

122 Our ten study areas are all located within the Apennines along 4.30° of latitude (about 660 km), extending
123 from North -East (NE) to South-West (SW) between 38-45° N and 8-17° E. They encompass a region
124 1200 km in length and 40-200 km of width, from the Ligurian sea to the Calabrian tip. The study areas
125 encompass several mountain peaks higher than 2000 m a.s.l., from Mt. Cimone (North) to Mt. Pollino
126 (South) together with comparable land covers along the altitudinal gradient and suitable for a change-
127 detection analysis. The highest elevation is Corno Grande (2914 m a.s.l.) of the Gran Sasso massif in the
128 central Apennines (Figure 1). Most of the study areas are included in the European Union Natura 2000
129 network of protected sites: about 78.5% of the analyzed areas is in the European Union Natura 2000
130 network of protected sites. Mean annual temperatures range from 6.2 to 10.0 °C and annual precipitation
131 ranges from 730 to 877 mm. NE slopes (Adriatic side) are in general more continental than SW slopes
132 (Tyrrhenian side), whereas precipitation is greatest for NE slopes. At each study area, we analyzed two
133 paired study landscapes (NE and SW aspect), each extending for 16 km². The 20 study landscapes cover
134 a total surface of approximately 32000 ha within an elevation range of 347-2500 m a.s.l., including all
135 vegetation zones, from hilly (< 600-m a.s.l.) to alpine.

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

143

144 Image analysis

145 We collected, processed and analyzed two types of aerial imagery: i) 1954-1955 flight aerial photos (b/w,
146 1 m cell size) from IGMI (Italian Geographic Military Institute) GAI (Italian Aerial Group); ii) 2010-2014
147 orthophotos from AGEA (National Agency for Funding in Agriculture) (RGB, 0.5 m cell size). For Mt.
148 Pollino only, we processed 1948 IGMI b/w photos and 2003 AGEA orthophotos. Here we refer to 1954
149 for older aerial photos (1948, 1954, 1955) and to 2012 for newer ones (2003, 2010-2014). Several IGMI
150 1948 images were scanned at 1200 DPI, mosaicked and resampled at 1 m resolution. Mt. Pollino is a
151 representative southern location, where peaks > 2000 m a.s.l. are very rare. Historical GAI aerial photos
152 were orthorectified using the AGEA orthophotos and a 20-m resolution DTM (Ispra – Italian Institute
153 for Environmental Protection and Research) as reference data. We used PCI Geomatica 2012 software
154 for geometric correction of historical images (mean RMSE overall = 23 m \pm 2 SD; mean RMSE for Mt.
155 Pollino = 82 m \pm 2 SD). To facilitate the comparison between historical and recent aerial photographs,
156 we resampled the higher resolution (0.5 m) AGEA images to 1 m as for the IGMI images. We applied a
157 semi-automatic object-based classification by combining the automatic segmentation through eCognition
158 software (scale factor 100, color factor 0.5) with on-screen photointerpretation of segmented polygons
159 (Garbarino et al. 2013). For the 40 land-cover maps (20 landscapes x 2 time periods) each polygon was
160 classified into 9 land-cover classes: bf (broadleaf forest), cf (conifer forest); sh (shrubland), dg (dense
161 grassland dense), sg (sparse grassland), or (orchard, vineyards, other tree groves), cr (cropland, herbaceous
162 crops in general), un (unvegetated, bare soil and water bodies), ur (urban, buildings and infrastructures).
163 The 40 land-cover maps (see examples in Figure S2, S3) were post-processed in ArcGIS 10.4 software
164 so as to enforce consistency among the two datasets (Figure S1). This two-step process aimed for a
165 minimum mapping unit (MMU) of 100 m². At first, the polygons with surface area < 100 m² were merged
166 with neighboring larger ones by using the ArcGIS tool “Eliminate”. After a rasterization of vector data
167 (1 m resolution) the raster maps were smoothed by using a moving-window (3 x 3) majority filter (Jensen
168 et al., 2001). Overall classification accuracy (Figure S1 – table insertion) ranged from 70% (Morone SW
169 1954) to 96% (Gorzano NE 2012) with a K coefficient between 62% (Cimone SW 1954) and 92%

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

172

173 **Data analysis**

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

185 To analyze 1954-2012 landscape patterns of the 20 study landscapes, we computed suitable landscape
186 and class metrics from each raster image using the Fragstats 4 statistical package (McGarigal and Marks
187 1994). We selected 5 metrics (patch density PD, patch area mean AREA_mn, mean shape index
188 SHAPE_mn, contagion index CONTAG and Simpson's diversity index SIDI) for the analysis after
189 excluding other metrics that were highly correlated (Pearson's $r > 0.8$) (Riitters et al., 1995) and
190 ecologically redundant (Tischendorf 2001). We ordered our residual 36 study landscapes through
191 multivariate ordination using principal components analysis (PCA) based on a main matrix of the five
192 landscape metrics, indirectly related to a secondary matrix of environmental variables (elevation, slope,
193 temperature and precipitation) and anthropogenic variables (population density and urban cover). PCA
194 was performed with the statistical package PcORD 7. The statistical significance of the ordination analysis
195 was tested using a Monte Carlo permutation method based on 10000 runs with randomized data.

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

204 To assess land abandonment in the forest landscapes of the Apennines, we also used demographic data
205 from the national population census carried out for each municipality every ten years (Vitali et al. 2017).
206 From the complete dataset (ISTAT - Population Census 1871-2011) we used the interval 1951-2011
207 (ISTAT 1951, 2011) subtracting the population densities (inhabitants/km²) averaged over the two years
208 for the municipalities included in the selected study landscapes.

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

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

228

229 RESULTS

230

231 Land use change and landscape features

232

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

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

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

251 Relative land cover changes varied significantly across an elevational threshold (above and below 1300 m
252 a.s.l.) (Figure 4b). All forest types increased dramatically more at lower than higher elevations: broadleaf
253 2832 ha, +59% vs. 1537 ha, +26% and conifer 719 ha, +186% vs 351 ha, +70%. Shrubland increased
254 only at lower elevation (208 ha, +50%), but maintained similar cover values at higher elevations. A similar

255 reduction trend occurred between dense grassland and sparse grassland at lower and higher elevations
256 (dg = -59% and sg = -57% at lower elevation; dg = -21% and sg = -17% at higher elevation). Moreover,
257 agricultural cover (crops) experienced a greater relative reduction at higher (around -359 ha, -93%) than
258 at lower elevation (-1862 ha, -73%) (Table S5, S6).

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

265

266 **Landscape pattern change**

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

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

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

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

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

294

295 **Forest landscape change in broadleaf forests**

296 The observed increase of broadleaf forests (Figure 2 and table 1) was derived mainly from secondary
297 successions occurring in grassland (61.5%), cropland (20.5%) and shrubland (9.2%). The influence of
298 slope aspect was weak, whereas elevation appeared a more relevant factor given that grassland to forest
299 transitions were greater at higher than at lower elevation (75% vs. 51%) and cropland to forest transitions
300 were greater at lower than at higher elevation (33% vs. 4%, respectively). In general, lower-elevation
301 landscapes showed more dynamic forest expansion. The overall bf mean annual increment over the 58-
302 year period was 0.59 % (SW 0.62%, NE 0.56%, L 0.80% and H 0.40%). Within the 60-year time interval,
303 the mean population density decreased significantly by 34% overall (Wilcoxon Test: $W = 3$, p -value $<$
304 0.001). Comparing among slope aspects, we found that population density decreased similarly in the NE
305 and SW municipalities (-22 vs. -20 inhabitants/km² respectively). We found that population density
306 (inhabitants/km²) and forest expansion (ha) were negatively correlated (Pearson's $r = -0.64$).

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

317

318 **DISCUSSION**

319

320 LUCs are affecting forest cover dynamics worldwide with significant local differences. In some areas
321 increasing farming and logging caused forest fragmentation and/or deforestation, whereas in many others
322 the rural marginality determined opposite transitions, with secondary forests invading abandoned
323 croplands and pastures (Rudel et al. 2005; Rey Benayas 2007). Following periods of extensive forest
324 clearing to increase farming and livestock grazing, post-abandonment natural reforestation occurred in
325 Mediterranean and temperate biomes of Europe and North America (Flinn and Vellend 2005). In some
326 tropical areas, abandoned croplands are shifting to second growth forest over a longer time (Florentine
327 and Westbrooke 2004). There are examples in Oceania (Endress and China 2019), Puerto Rico (Lugo
328 and Helmer 2004) and eastern Africa (Chapman and Chapman 1999) and even in semi-arid regions of
329 Argentina (Basualdo et al. 2018). In mountain areas of the Mediterranean basin land cover dynamics are
330 faster as reported in the Alps (Tasser et al. 2005; Niedrist et al. 2009), the Carpathians (Kuemmerle et al.
331 2009; Weisberg et al. 2013), the Pyrenees (Metailiè and Paegelow 2005; Roura-Pascal et al. 2005), in
332 Greece (Petanidou et al. 2008) and Spain (De Aranzabal et al. 2018).

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

343

344 **Land use change and topographic factors**

345 The overall forest cover increase, regardless of the scale of analysis, is very close to the 35-48% forest
346 cover increase reported for the Apennines by more local studies (Rocchini et al. 2006). The average
347 annual forest expansion rate (0.5 %/year) is also quite similar to that found by other authors (0.4-
348 0.7%/year) in different sectors of the Apennines (Bracchetti et al. 2012).

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

355 The loss of grasslands in the Apennines is an evident landscape process of recent decades: livestock
356 grazing declines in mountain regions of central Italy between 1961 and 2000 were estimated at
357 approximately 30% for cattle and 33% for sheep and goats (Pelorosso et al. 2009), although reliable data
358 on pastoralism are often scarce or incomplete (Falcucci et al. 2007). The abandonment of grasslands and

359 croplands largely influenced the observed land-cover transitions, with notable differences for aspect and
360 elevation. The more favorable topography and climate conditions of the Apennine SW slopes favored
361 farming and livestock grazing. The greater human pressure induced more relevant land-cover changes
362 following land abandonment (Vitali et al. 2017). On these slopes, where the population density is higher
363 than on NE ones, grasslands shifted more slowly to other land cover types. At NE exposure farming and
364 grazing decreased faster or even disappeared at higher elevations.

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

381

382 **Landscape mosaic shift driven by land abandonment**

383

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

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

402

403 **Driving forces of secondary succession**

404 The processes of broadleaf forest expansion were altitude-dependent. At high-elevation, secondary
405 forests mostly derive from former grasslands; whereas, at low-elevation, contributions to secondary
406 forests were distributed among different land cover classes. This is likely due to the land-cover
407 composition in 1954. Topographic variables, such as slope aspect, have strongly conditioned land and
408 forest use in the Apennines (Vitali et al. 2017). The large-scale removal of forests on SW slopes, occurred
409 in ancient times, today provides the greater potential for forest expansion after abandonment. In Europe,

410 a rural depopulation of 17% between 1961 and 2010 (FAOSTAT 2010) induced extensive land
411 abandonment and forest expansion. In the Apennine municipalities comprising our studied landscapes,
412 the national census data reported a relevant population decrease. This process however exhibited
413 differences according the two main slope aspects and elevation zones, with clear effects in forest cover
414 transitions.

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

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

429

430 **CONCLUSION**

431 The main goal of this work was to develop a generalizable model of Apennine landscapes changes at the
432 regional scale, through targeted sampling of replicate study landscapes within key environmental strata.
433 Since the 1950's, following a period of widespread depopulation and land abandonment, the Apennines
434 have experienced an overall forest expansion (Vacchiano et al. 2017; Malandra et al. 2018). Forest cover
435 gains were similar at the two main exposures (NE and SW), but significantly greater at lower elevation

436 (below 1300 m a.s.l.). We quantified the importance of several key land-cover change drivers such as
437 distance from pre-existing forest, elevation, slope angle and distance from previous croplands. Landscape
438 structural complexity was reduced at lower elevations and experienced an inverse process of
439 fragmentation at higher elevations through time. The withdrawal of traditional agro-silvo-pastoral
440 practices in marginal lands observed in the Apennines is widespread in most European mountain areas
441 (Roura-Pascal et al. 2005; Petanidou et al. 2008; Weisberg et al. 2013; Mallinis et al. 2014; Campagnaro et
442 al. 2017; De Aranzabal et al. 2018). The combined approach of using areal changes of land-use/land-
443 cover and landscape metrics to quantify landscape pattern dynamics appeared a suitable method to infer
444 driving factors of variability and to understand their ecological effects (Geri et al. 2010b; Campagnaro et
445 al. 2017). Moreover, appropriate management actions and suitable regional policy strategies should be
446 implemented in these transient areas to prevent further decline (MacDonald et al. 2000). Extended spatio-
447 temporal lags for this type of analyses provide suitable data for developing land-use models, facilitating
448 the prediction of more reliable landscape changing scenarios and forest dynamics trends, useful tools for
449 land management and landscape restoration.

450

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455

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634

635 **TABLES AND FIGURES**

636

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

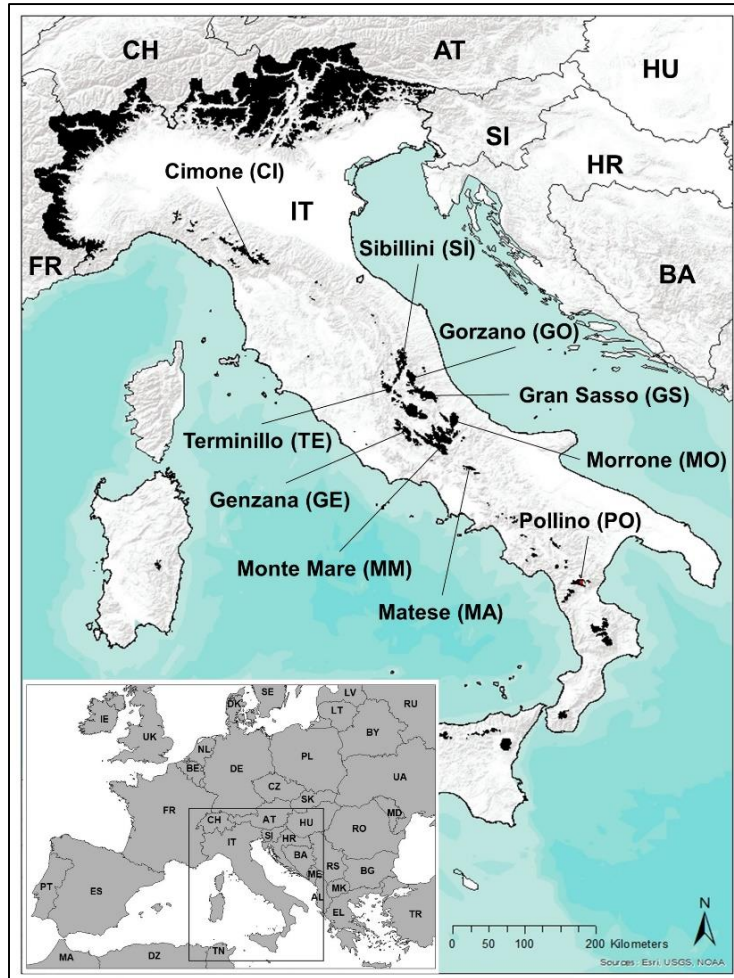
Contribution to bf	Global *	NE	SW	H	L
	(%)	(%)	(%)	(%)	(%)
Conifer forest	2.3	0.5	4.1	4.0	1.0
Shrubland	9.2	11.3	7.1	11.5	8.0
Dense grassland	40.7	42.4	38.0	51.3	33.5
Sparse grassland	20.8	19.8	19.9	23.7	17.5
Orchard	1.7	0.5	3.3	0.1	2.9
Cropland	20.5	21.3	21.7	4.0	32.5
Unvegetated	3.0	2.8	3.4	4.5	2.2
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

642

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

Transition types	Unchanged pixels (0)	Changed pixels (1)	Parameters (z-value)	W_i
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

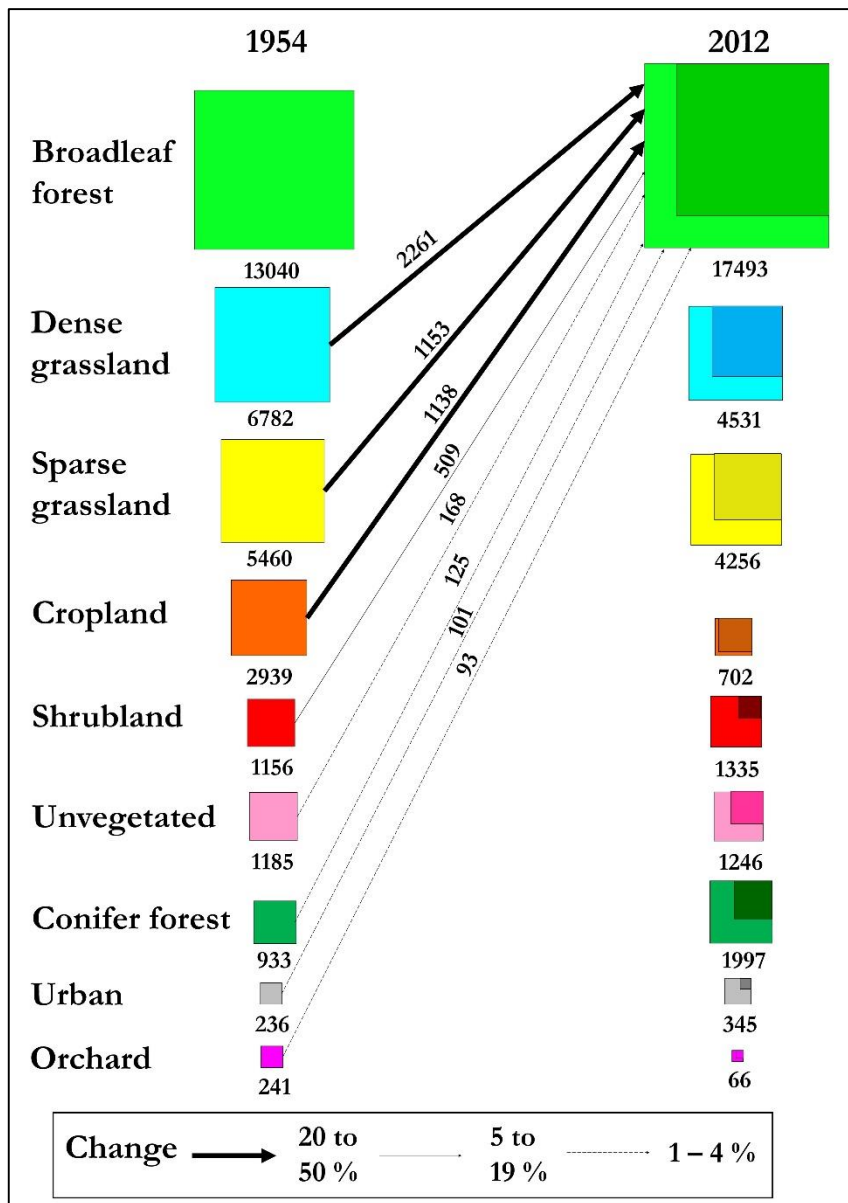
652



653

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

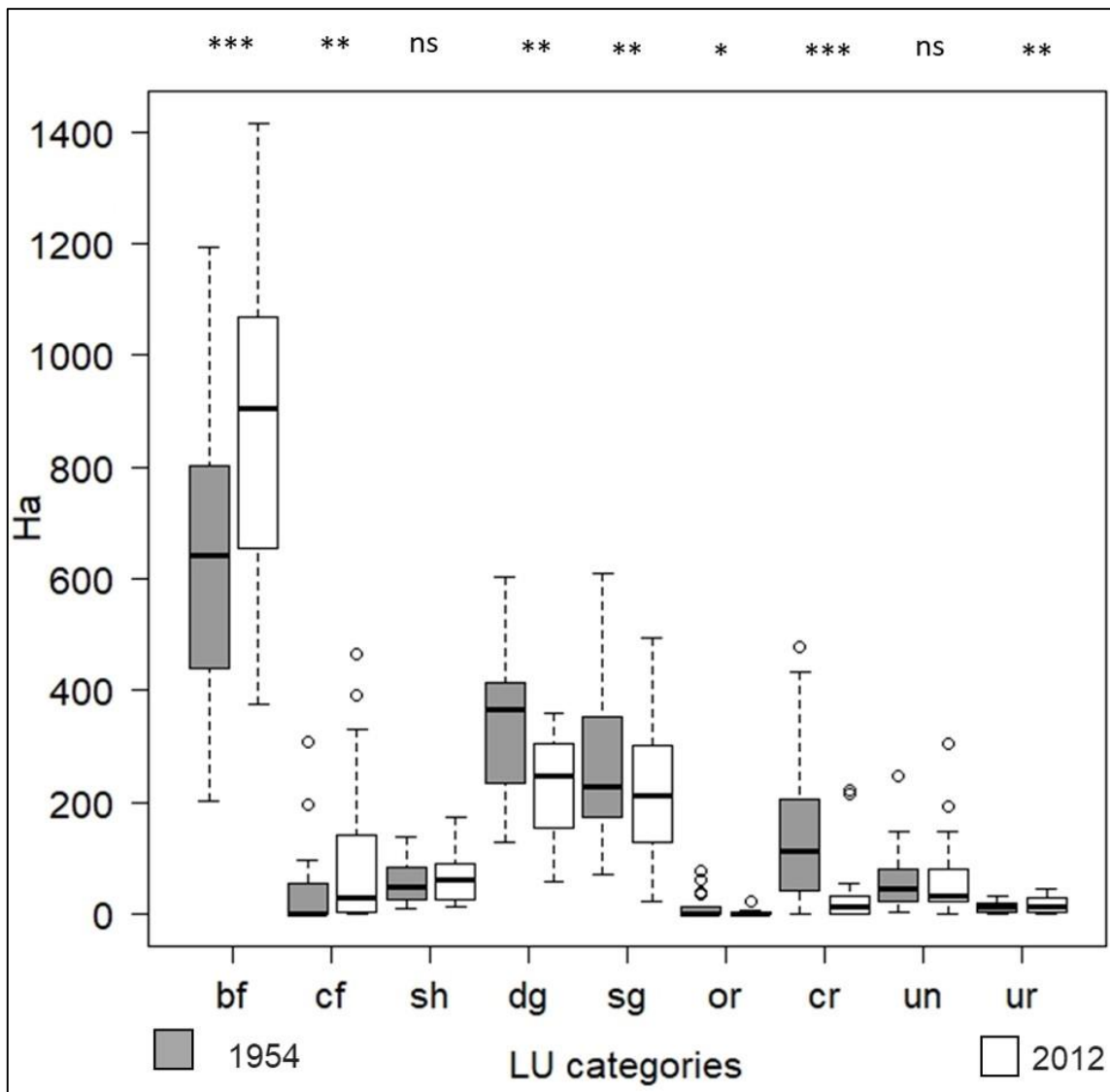
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659 **Figure 2** – Area of land-cover classes (ha), and land-cover transitions from past to present in the
660 Apennines study sites. Light-colored boxes are size-scaled land-cover categories. Darker-colored inset
661 boxes represent the relative unchanged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

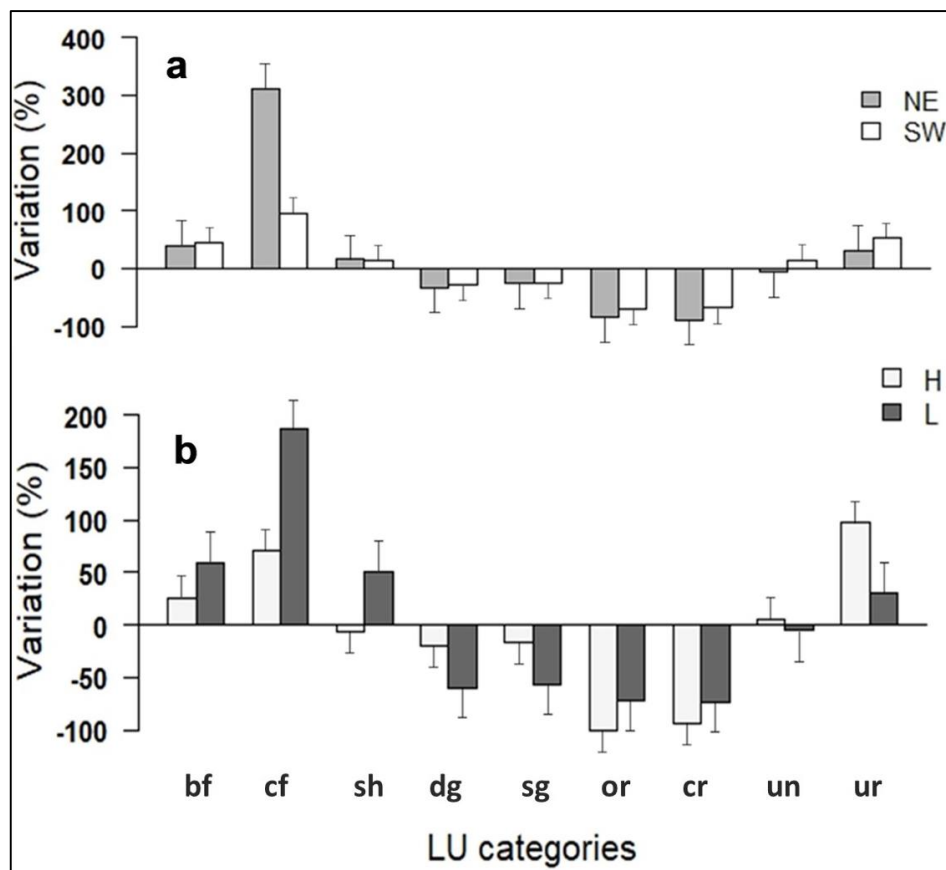
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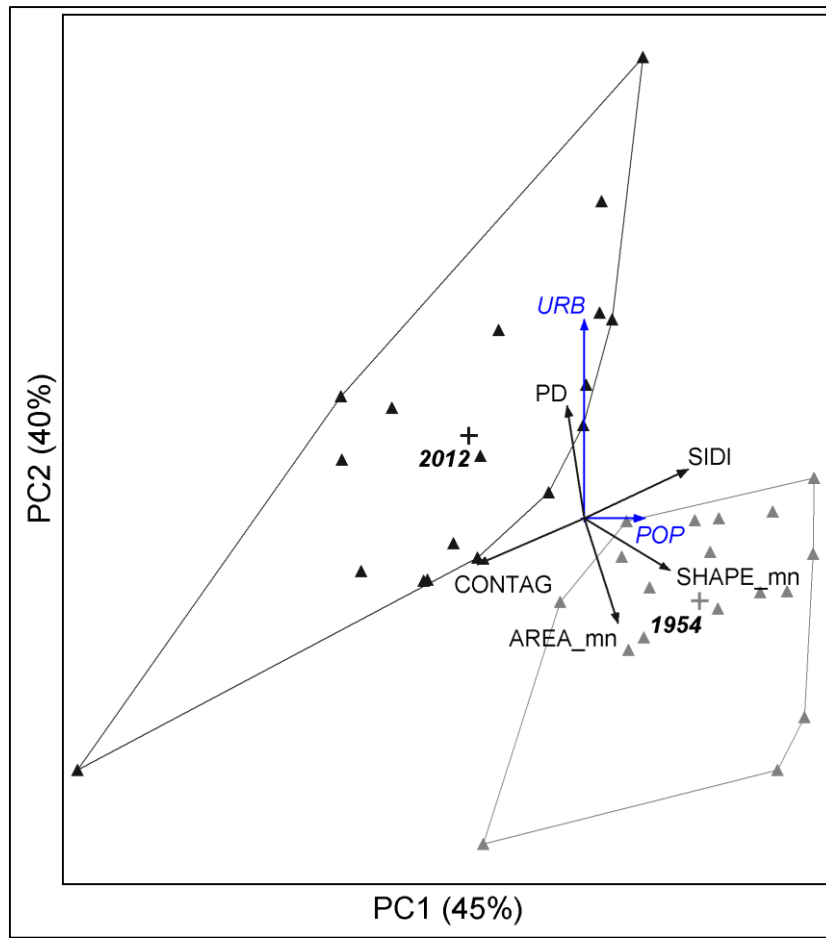
667

668 **Figure 3** - Mean distribution of land-cover categories (hectares) for the two time periods (1954 and 2012)
669 across 20 replicate study landscapes: broadleaf forest (bf), conifer forest (cf), shrubland (sh), dense
670 grassland (dg), sparse grassland (sg), orchard (or), cropland (cr), unvegetated (un), urban (ur). Horizontal
671 lines are median values and circles are outliers. * = p-value < 0.05, ** = p-value < 0.01, *** = p-value <
672 0.001, ns = not significant (Wilcoxon paired test to compare 1954 and 2012 covers for each category).



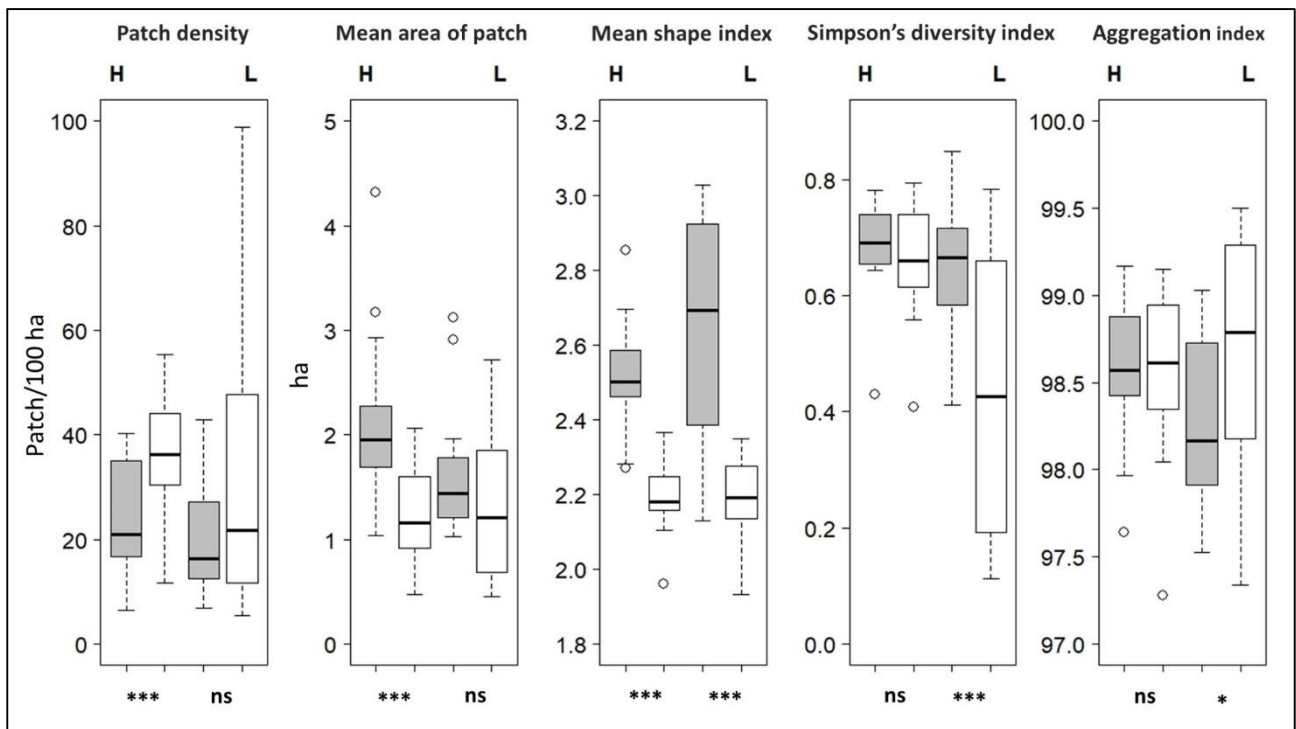
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675 **Figure 4** – Relative change (%) of land-cover categories in the 18 study landscapes: a) by main slope
 676 aspects (NE vs SW) and b) by elevation (H > 1300 m a.s.l. vs L < 1300 m a.s.l.). Error bars show standard
 677 errors. Broadleaf forest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



679

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



688

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

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