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1 **Contrasting land use legacy effects on forest landscape dynamics in the Italian Alps and the Apennines**

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17

18 **Abstract**

19 Context: Land use legacies of human activities and recent post-abandonment forest expansion have
20 extensively modified numerous forest landscapes throughout the European mountain ranges. Drivers of
21 forest expansion and the effects of changes on ecosystem services are currently debated.

22 Objectives: i) to compare landscape transition patterns of the Alps and the Apennines (Italy), ii) to quantify
23 the dominant landscape transitions, and iii) to measure the influence of climatic, topographic and
24 anthropogenic driving factors.

25 Methods: Land cover changes and landscape pattern modifications were investigated at the regional (over
26 28 years, Alps and Apennines, Corine Land Cover dataset) and landscape scale (over 58 years, 8 Alpine and
27 8 Apennine sites, aerial images). The main driving factors of post-abandonment forest landscape dynamics
28 were assessed with a statistical modeling approach.

29 Results: Forest expansion was the dominant landscape transition at both Italian mountain ranges, with an
30 annual overall rate of 0.6%. Forest expansion was more extensive at lower elevations in the Apennines
31 where climate is less limiting and extensive abandoned croplands and pastures were available throughout
32 the study period. Distance from pre-existing forest edges in the Alps and elevation in the Apennines
33 emerged as the most important predictors.

34 Conclusions: Forest expansion is most rapid where areas of recent agricultural abandonment coincide with
35 favorable climatic conditions. Thus the prediction of forest landscape dynamics, in these mountain forests
36 with a long history of cultural use, requires knowledge of how the magnitude and timing of land use
37 changes intersect spatially and temporally with suitable conditions for tree establishment and growth.

38 **Keywords** Forest expansion, cultural landscape, historical ecology, aerial photographs, landscape structure,
39 land abandonment

40 **Acknowledgments**

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44 Introduction

45 Land use history exerts a strong long term legacy on forest landscapes, affecting their structure, spatial
46 pattern and associated ecosystem services (Bellemare et al. 2002; Ziter et al. 2017). At a landscape scale,
47 forests after their removal can be replaced by other land cover types such as crops, pastures, and
48 settlements, which in turn can lead to habitat fragmentation. Land abandonment has important effects on
49 natural disturbance regimes especially in regions with a long history of intensive human influence (Mantero
50 et al. submitted). After land abandonment, secondary succession can lead to forest expansion, a complex
51 dynamic process influenced by different factors such as climate, topography, seed availability and
52 anthropogenic variables. An important driver at local scales has proven to be the former land use intensity
53 and type (Walker et al. 2010). For example, abandoned cropland, with compacted soil due to former
54 plowing, may experience slower successional dynamics than former pastures (Dupouey et al. 2002). A pan-
55 European scale study over the last 25 years revealed that the most important landscape transitions are
56 urbanization and natural afforestation processes, both affecting landscape service provision (Van der Sluis
57 et al. 2019). In forest landscapes post-abandonment processes generally cause increased wildfire
58 frequency, extent and severity (e.g. Moreira 2001; Lloret 2002; Pausas et al. 2012) and a decrease in the
59 frequency and intensity of rockfall (Lopez-Saez et al. 2016; Farvaque et al. 2019) and avalanches
60 (Kulakowski et al. 2011; García-Hernández et al. 2017). Post-abandonment forest expansion can also cause
61 changes to major bio-geochemical cycles, soil properties and eco-hydrological processes (Pellis et al. 2019),
62 loss of biodiversity especially in semi-open areas with species-rich grasslands, and loss of cultural
63 landscapes (Otero et al. 2015; Hermoso et al. 2018; Ridding et al. in press). Secondary succession processes
64 can result in multiple pathways and abandoned lands may become more vulnerable to invasive species and
65 fire (Munroe et al. 2013).

66 For thousands of years, anthropogenic pressure over the Mediterranean basin has shaped the numerous
67 and diverse cultural landscapes (Naveh 1995). In many European rural areas abandonment after WW2 was
68 a widespread socio-economic process, causing large human migrations toward urban areas (MacDonald et
69 al. 2000; Poyatos 2003; Hocht et al. 2005). The decline of traditional agro-pastoral activities was
70 particularly intense in southern European mountains such as the Italian Alps and the Apennines. These two
71 mountain ranges, covering approximately 35% of the entire country of Italy (Vacchiano et al. 2017),
72 experienced a significant forest expansion after an extensive decrease of cultivated lands due to
73 depopulation (Falcucci et al. 2007). Forest cover in Italy shifted from 6 million ha in 1936 (Forest Map of the
74 Italian Kingdom) to 8.5 million ha in 1985 (First Italian Forest Inventory, IFNI85), and to 10.5 million in 2005
75 (Second Italian Forest Inventory, INFC05) and is currently estimated at about 11 million ha in 2015 with an
76 increase of 20% in the last 30 years (Ferretti et al. 2018). These estimates rely on a wide range of sources,
77 and studies using consistent datasets to quantify changes in land cover across broad areas prior to
78 widespread availability of satellite imagery are lacking. In the Italian and French Alps, the depopulation of
79 marginal lands started after the Industrial Revolution in approximately 1871 and, due to the two World
80 Wars, lasted through the 1950s (Batzing et al. 1996). Before the 1950s, grazing was widely distributed in
81 the Italian Alps where rangelands occupied 53% of the mountain areas (White 1950; Garbarino et al. 2013).
82 This was primarily cattle grazing in unfenced pastures. In the Apennines a human migration from mountain
83 areas toward the Adriatic and Tyrrhenian coastal areas occurred from 1951 to 1991, especially in the
84 northern and southern Apennines (Malandra et al. 2018; Vitali et al. 2018). Many of these areas were
85 subjected to former heavy exploitation for firewood and charcoal production, with wood pastures
86 occurring frequently at higher elevations. A national reforestation program to reduce slope erosion,
87 launched before WW2 and lasting until the 1970s, resulted in approximately 760,000 ha of new plantation
88 forests composed mainly of coniferous tree species (Piermattei et al. 2016).

89 In both mountain regions, forest expansion has occurred as a gap-filling process at lower elevations and as
90 an upward shift of treeline at higher elevations (Tasser et al. 2005). Forest expansion caused a direct

91 reduction of open areas, reduced the extent of forest-grassland ecotones, and led to decreases in species
92 diversity as well as culturally important landscapes (Falcucci et al. 2007; Petanidou et al. 2008). However,
93 there are fundamental differences in how forest expansion processes have unfolded in the two mountain
94 regions. A gentler topography and a favorable climate led to more intense deforestation in the Apennines,
95 creating open areas that were used as pastures and crops both at low and high elevations. However,
96 landscape mosaic simplification due to forest gap-filling processes mainly occurred at lower elevations. At
97 higher elevations, forest succession on abandoned croplands and grasslands led to a complex and
98 fragmented landscape (Malandra et al. 2019; Vitali et al. 2019). Understanding the underlying drivers of
99 forest landscape changes by comparing the different land use legacies of the Alps and the Apennines is a
100 fundamental step for more ecologically based landscape planning and management oriented toward
101 biodiversity conservation and other ecosystem services.

102 The Alps and the Apennines are large and highly representative areas for testing our three hypotheses on
103 mountain forest landscape changes in Italy over the last 60 years: 1) forest cover is increasing everywhere,
104 but with different patterns in the Alps and the Apennines; 2) pasture-to-forest is the dominant landscape
105 transition at high elevation; 3) historical forest cover (i.e. land use legacy) is the most important
106 environmental driver for predicting forest expansion today.

107

108 **Methods**

109 *Study area and sampling design*

110 Our multiscale research design was structured at two spatial scales (region and landscape) aimed at
111 comparing the two mountain ranges of Italy, the Alps (AL) and the Apennines (AP). They have similar total
112 length (1300 – 1350 km, respectively), but different geographic orientation (AL: from west-to-east across
113 northern Italy; AP: northwest-to-southeast from Liguria to Calabria). The two mountain ranges differ in
114 terms of climate, topography and land use history.

115 In AL, mean annual temperatures range from less than 0° to over 10° C, with very cold winters. Annual
116 precipitation ranges from 400 to over 3000 mm and summer dry periods are very rare (Isotta et al. 2013).
117 Metamorphic lithology with intrusive igneous rocks prevail in the inner sectors and sedimentary outcrops
118 dominate in the outer ones. Oak forests dominate at lower elevations whereas beech-silver fir (*Fagus*
119 *sylvatica* and *Abies alba*) forests prevail on mesic aspects of the montane zone, replaced by *Pinus sylvestris*
120 on xeric slopes. Coniferous forests with *Picea abies* (L.) H.Karst., *Larix decidua* Mill. and *Pinus cembra* L.
121 dominate the subalpine zone (Fauquette et al. 2018).

122 At AP, mean annual temperatures range from 6° to 10° C, and annual precipitation ranges from 730 to 877
123 mm, with a short and pronounced summer dry period at lower elevations (Blasi et al. 2014). The eastern
124 Mediterranean side (Adriatic) is generally more continental and humid than the western (Tyrrhenian) one.
125 The forest vegetation is largely dominated by broadleaf species of the Mediterranean and temperate
126 biomes. Xeric oak forests of *Quercus ilex* L., *Quercus pubescens* Willd., *Quercus cerris* L. and *Ostrya*
127 *carpinifolia* Scop. dominate at lower elevations and *Castanea sativa* Mill. the sub-montane zone. *Fagus*
128 *sylvatica*, locally mixed with *Abies alba*, largely dominates the montane zone up to treeline except for a few
129 locations in the central and southern sectors where natural pine forests occur, dominated by *Pinus mugo*,
130 *P. heldreichii*, or *P. nigra laricio*.

131 For our regional scale analyses, the study area in each mountain chain included all contiguous land above
132 500 m a.s.l. excluding those island polygons separated from the main mountain chains (Fig. 1). We obtained
133 two large areas of 52,002 km² (AL) and 44,615 km² (AP), where we assessed the land-cover changes (LCC)
134 for the 1990 – 2018 period, based on the Corine Land Cover (CLC Level 3, Feranec et al. 2016) raster maps

135 (100-m spatial resolution) after merging the original CLC categories into five larger groups: forest (FO),
136 grassland (GR), cropland (CR), urban (UR), and unvegetated (UV) (Table S1). We developed a transition
137 matrix for both regions by assessing changes for the five selected land-cover categories, allowing us to
138 compute the relative changes in AL and AP.

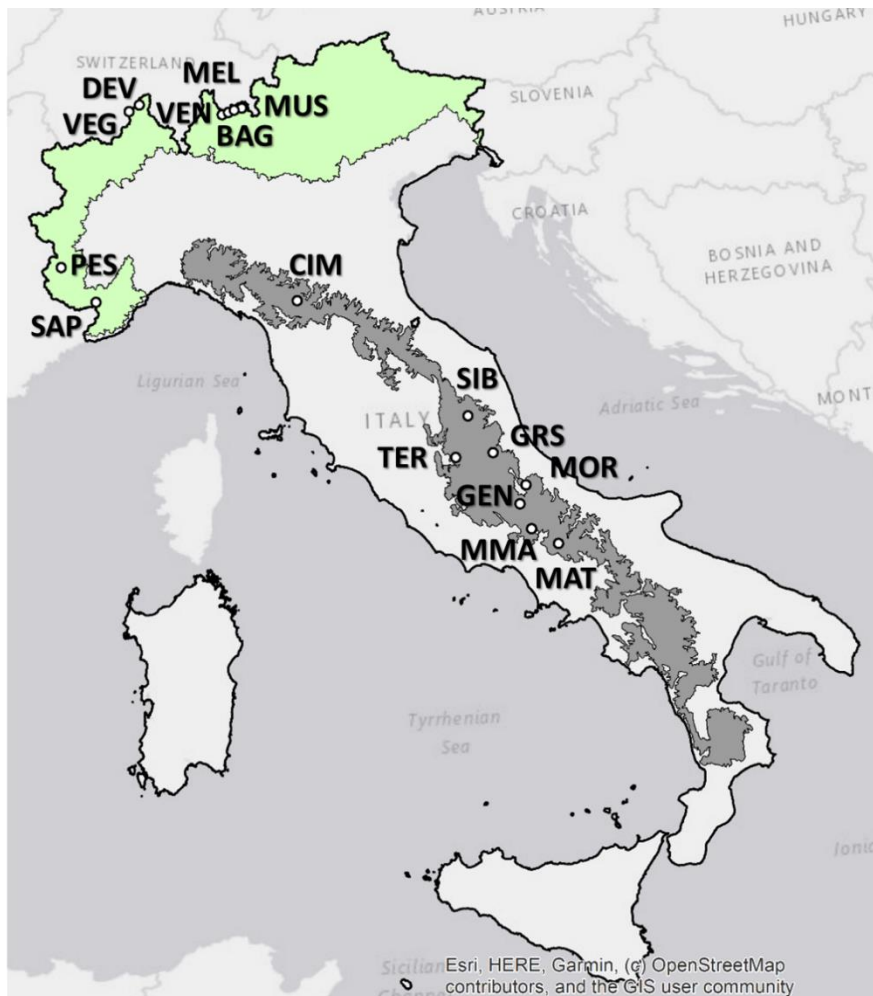
139 For our landscape-scale analyses, in each region (AL and AP) we selected 8 landscapes of variable extent
140 ranging from 6.3 to 16 km². These landscapes were selected and harmonized from previous projects and
141 unpublished data on land-use/land-cover changes in AL and AP (e.g. Garbarino et al. 2013; Malandra et al.
142 2019). The 16 study landscapes cover a total surface area of approximately 23,000 ha within an elevation
143 range of 500 – 2,600 m a.s.l., including all vegetation zones (Table 1). We adopted the altitudinal threshold
144 of 2,600 m a.s.l., as the potential alpine treeline (Caccianiga et al. 2008; Lingua et al. 2008; Garbarino et al.
145 2013) in order to limit the LUC analysis to the vegetated part of the 16 landscapes and to reduce the weight
146 of the ‘rock’ land cover category, which is uninformative for our research. Historical aerial photographs for
147 the years 1954-1962 (b/w, 1: 60,000 approximate cartographic scale, Italian Geographic Military Institute -
148 IGMI) were scanned and digitized at 800 ppi, with a mean spatial resolution of 1 m. The IGMI images were
149 georeferenced and orthorectified using PCI Geomatica 2012 software. Regularly distributed tie points were
150 used to co-register IGMI images with 2012 orthophotos (RGB, 0.5 m cell size, National Agency for Funding
151 in Agriculture - AGEA) resampled at 1 m. The average horizontal Root Mean Square Error (RMSE) was 23 m ±
152 2SD. We used the TINITALY DEM at 10 m spatial resolution (Tarquini et al. 2012) for orthorectification.

153 We applied a semi-automated object-based classification by combining the automated image segmentation
154 from eCognition software (scale factor 100, color factor 0.5) with on-screen photointerpretation of
155 segmented polygons (Garbarino et al. 2013). We then performed a supervised classification of the objects
156 based on an initial set of at least 10 training polygons for each category, selected through
157 photointerpretation. This was followed by a manual classification of the previously unclassified polygons.
158 Each polygon of the 32 land cover maps (16 landscapes × two time periods), was classified into five land
159 cover classes that were used for the regional-scale analysis: forest (FO), grassland (GR), cropland (CR),
160 urban (UR), unvegetated (UV). The UV category includes different land cover types such as rock, gravel,
161 sand, bare soil and sparse vegetation areas. The latter is a mosaic of sparse grasslands and barren
162 nonvegetated areas that are mostly located between 2000-3000 m a.s.l. A post-processing procedure on
163 the resulting 32 land cover maps was performed in a GIS environment to enforce consistency among the
164 two datasets. A minimum mapping unit (MMU) of 100 m² (Garbarino et al. 2011) was obtained by merging
165 smaller polygons with neighboring larger ones by using the ArcGIS tool ‘Eliminate’ (Malandra et al. 2019).
166 Merged polygons were rasterized at 1-m resolution and the resulting raster maps were smoothed using a
167 moving window (3 × 3) majority filter. We obtained the level of accuracy by randomizing 16 polygons/ha on
168 each map and classifying the objects visually using the same land cover categories adopted in the
169 automatic segmentation (Radoux and Bogaert 2017). Overall, the classification accuracy ranged from 78%
170 to 96% with a Cohen's Kappa coefficient between 0.67 and 0.93 (Table S2). The land cover change analysis
171 at the landscape scale provided 16 transition matrices that were combined to detect overall transitions and
172 differences between the AL and AP mountain regions. We converted the two transition matrices into two
173 transition diagrams showing gain, loss, net change and persistence for each land cover category (Cousins
174 2001). With the same dataset (Garbarino et al. 2019), we computed the relative contribution (in hectares)
175 of each land cover category to the transition to forest cover and we applied a Mann–Whitney test to
176 compare the medians of the two mountain ranges (AL and AP). The Mann–Whitney test was performed for
177 each of the five categories by using the 8 landscapes as sample size for each mountain range.

178 For our landscape-scale analyses, we assessed the main drivers affecting forest expansion in AL and AP
179 using a Random Forest (RF) model (Rodman et al. 2019). Specifically, we modelled the occurrence of a
180 transition to forest through a binary classification method using ‘mlr’ (Bischi et al. 2016) and ‘ranger’
181 (Wright & Ziegler 2017) R packages (R Core Team 2019). Given the unbalanced ratio between cells with a

182 transition to forest (minority class) and cells that remained unchanged (the majority class), we under-
183 sampled unchanged cells using a spatially random selection within each landscape using the 'spatialEco' R
184 package (Evans 2019). Before computing class transitions, we filtered out from the dataset the landscape
185 portions that were already forests in the past, and we downscaled gridded land cover maps from 1 m to 30
186 and 60 m resolutions using the majority class within each coarser cell. The coarser resolutions allowed us to
187 limit the influence of both co-registration and classification errors of aerial images on model predictions
188 and to evaluate the dependence of model predictions on the spatial scale. For our final analysis, we used
189 the 30 m resolution because RF models trained with data at 30 or 60 m resolutions produced very similar
190 outputs in terms of variables importance and trends of partial dependences. At a coarser resolution
191 prediction errors slightly increased (Table S3).

192



193

194 Figure 1. Location of the 16 landscapes (white circles) within the two Italian mountain regions: the Alps (light green)
195 and the Apennines (light grey) with a minimum elevation of 500 m a.s.l. For landscape codes see Table 1.

196

197 Because there was a substantial area of *Pinus* plantations within the forest cover of 2012 images in certain
198 landscapes of the Apennines, we removed plantation patches before computing class transitions in order to
199 model only natural forest dynamics.

200

201 Table 1. Environmental descriptors (Area = total surface area; El = mean elevation; Sl = mean slope; Te = mean annual
 202 temperature; Pr = mean annual precipitations; BD = mean distance from buildings; RD = mean distance from roads) of
 203 16 landscapes divided by mountain region (AL or AP).

Region	Landscape name	Landscape Code	Area (ha)	El (m a.s.l.)	Sl (°)	Te (°C)	Pr (mm)	BD (m)	RD (m)
AL	Bagni	BAN	1574.3	1788.1	36.0	3.9	1081.3	678.4	289.4
AL	Mello	MEL	1433.7	2045.8	34.1	1.7	983.7	935.0	359.8
AL	Sapè	SAP	959.4	1419.8	22.3	6.2	1001.9	343.8	178.4
AL	Pesio	PES	1599.2	1596.5	31.7	5.7	980.0	711.0	202.1
AL	Ventina	VEN	630.4	2248.6	30.1	0.1	900.8	1044.9	366.4
AL	Musella	MUS	921.4	2155.6	27.7	2.0	788.2	592.1	190.6
AL	Veglia	VEG	1407.2	2043.8	25.4	1.7	1045.1	815.3	205.1
AL	Devero	DEV	1570.0	2169.9	23.6	1.2	1138.6	1048.2	323.3
AP	Cimone	CIM	1593.4	1444.7	19.2	6.8	1352.5	349.3	111.6
AP	Sibillini	SIB	1601.3	1397.2	26.3	6.8	940.1	549.4	115.9
AP	Gran Sasso	GRS	1602.4	1577.5	27.2	6.3	872.3	1548.4	294.2
AP	Terminillo	TER	1600.7	1573.5	25.7	6.8	831.5	700.4	202.8
AP	Morrone	MOR	1603.7	1422.8	22.8	7.6	792.4	1447.3	276.9
AP	Genzana	GEN	1603.4	1234.7	22.4	8.2	789.1	664.7	228.5
AP	Monte Mare	MMA	1604.0	1149.7	24.1	9.0	780.9	1622.2	410.9
AP	Matese	MAT	1605.4	1112.5	22.5	9.2	694.7	481.4	163.4

204

205 We used several spatial predictors (Table S4) such as the distance from pre-existing forest edges,
 206 topographic variables (elevation, slope, heat load index *sensu* McCune and Grace 2002), climatic variables
 207 (mean annual temperature, annual precipitation), and anthropogenic variables (cost of movement,
 208 Euclidean distance to buildings, Euclidean distance to roads). We derived topographic variables from the 10
 209 m DEM and climatic variables from the ‘Climatologies at high resolution for the earth’s land surface areas’
 210 (CHELSA) v1.2 datasets at 30 arcsec (~1 km) spatial resolution (Karger et al. 2017). We computed the
 211 accumulated cost of movement across the terrain through the Tobler’s hiking function (on-path)
 212 implemented in the ‘movecost’ R package (Alberti 2019), using “buildings” in OpenStreetMap as starting
 213 locations. We applied two different approaches to obtain either the RF models predictions or the predictive
 214 performance estimates with a reduced bias. Specifically, we trained two RF models, one for AL and one for
 215 AP, using all the data and tuning hyper-parameters through an 8-fold spatial cross-validation (Brenning
 216 2012, Schratz et al. 2019). We used bias-reduced predictive performance estimates using two common
 217 measures in binary classification, the Brier score (Brier 1950) and the area under the receiver operating
 218 characteristics curve (AUC). These measures were averaged over a total of 800 RF models obtained through
 219 an 8-fold spatial cross-validation repeated 100 times using a nested 5-fold spatial cross-validation for
 220 hyper-parameters tuning (Lovelace et al. 2019, Schratz et al. 2019). For both strategies, we used a
 221 sequential model-based optimization approach in ‘mlrMBO’ R package (Bischl et al. 2017) to search for the
 222 optimal RF hyper-parameters (mtry, sample fraction and minimum node size) using 50 steps. The spatial
 223 cross-validation resampling technique was based on k-means clustering of observation coordinates and
 224 allowed us to geographically partition the data, thus maintaining the assumption of independence among
 225 training and test sets which would be violated in the case of randomly sampled observations due to the
 226 presence of spatial autocorrelation. We assessed variable importance from each model using the
 227 permutation method (Breiman 2001) and we employed partial dependencies (Friedman 2001; Goldstein et
 228 al. 2015) to interpret the marginal effect of each variable on the predicted probability of forest expansion.
 229 Specifically, we computed the average and the standard deviation of individual marginal effects obtained
 230 using all the observations in the dataset through the ‘generatePartialDependenceData’ function in ‘mlr’ R
 231 package.
 232
 233
 234
 235

236 **Results**

237 *Regional land cover changes – CORINE database*

238 Forest expansion occurred in both areas, but was greater in AL (+2,951 ha, +9 %) than in AP (+1023 ha, +3.7
 239 %) (Tab. 2). Cropland cover is generally stable whereas human infrastructures increased more at AP (+ 131
 240 ha, +34 %) than at AL (+256 ha, +19 %). Grasslands greatly decreased at both mountain ranges, but more at
 241 AL (-2784 ha, -39 %) than at AP (-1160 ha, -27 %), a pattern also observed for unvegetated areas (AP -56 ha,
 242 -19 %; AL -503ha, -9 %). Regional scale results (1990-2018) were weakly in agreement with the landscape
 243 scale results (1954-2012) as shown in the supplementary material (Table S5).

244

245 Table 2. Land cover categories surface areas expressed as a percentage of the total mountain area derived from the
 246 Corine Land Cover in the Alps (AL) and the Apennines (AP) throughout the years. Changes across the entire period
 247 (1990-2018) are indicated in the last two columns as absolute and relative percent values.

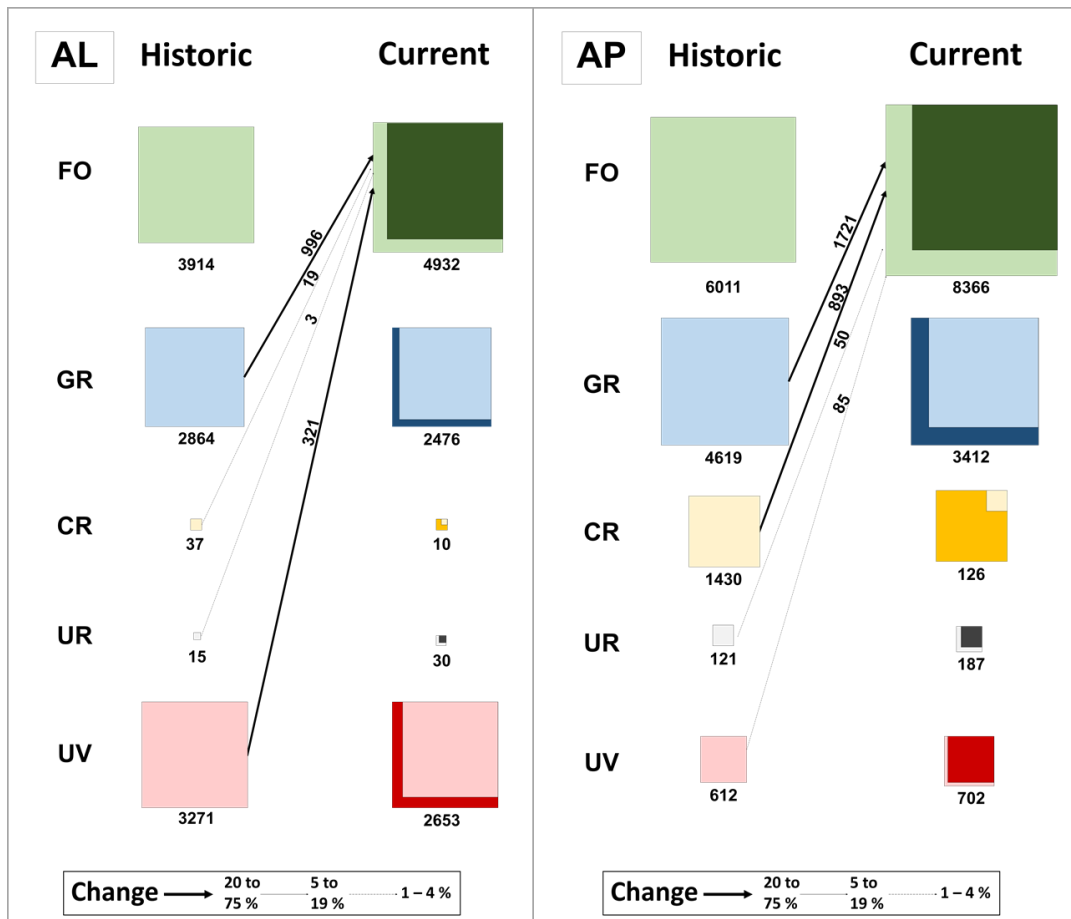
AL	1990	2000	2006	2012	2018	1990-2018 Change	
						<i>Absolute</i>	<i>Relative</i>
Forest (FO)	62.9	62.4	62.5	68.6	68.6	+5.7	+9.0
Grassland (GR)	13.8	14.9	14.6	8.5	8.5	-5.4	-38.8
Cropland (CR)	10.1	10.1	10.4	10.2	10.2	+0.2	+1.5
Urban (UR)	2.6	2.7	2.8	3.0	3.1	+0.5	+19.3
Unvegetated (UV)	10.6	9.9	9.7	9.6	9.6	-1.0	-9.2
AP	1990	2000	2006	2012	2018	1990-2018 Change	
						<i>Absolute</i>	<i>Relative</i>
Forest (FO)	62.5	62.5	62.3	64.8	64.8	+2.3	+3.7
Grassland (GR)	9.8	9.8	9.8	7.2	7.2	-2.6	-26.6
Cropland (CR)	26.2	26.1	26.2	26.3	26.3	+0.1	+0.5
Urban (UR)	0.9	1.0	1.1	1.1	1.2	+0.3	+34.2
Unvegetated (UV)	0.7	0.6	0.5	0.5	0.5	-0.1	-19.1

248

249 *Landscape transitions – aerial imagery*

250 Forest expansion also occurred at the landscape scale in both ranges. For all study landscapes combined,
 251 the mean annual forest surface area increment was 60 ha/year (0.5 %/year), and was slightly greater at AP
 252 (41 ha yr⁻¹ or 0.6% yr⁻¹) than at AL (19 ha yr⁻¹ or 0.4% yr⁻¹). Grasslands and Crops decreased in both areas,
 253 but a larger reduction of crops (CR) occurred at AP (Fig. 2). Unvegetated lands (UV) decreased only at AL,
 254 and urban infrastructures (UR) increased more at AP. The relative weights of CR and UR were historically
 255 higher at AP, whereas UV values were historically higher at AL. Forest expansion was mostly related to the
 256 “GR to FO” transition (66.3 % overall), but was greater at AL than at AP (74.4 % and 62.4 % respectively, Fig.
 257 3). The “CR to FO” transition was stronger at AP than AL (32.4 % and 1.4 %), whereas the opposite pattern
 258 was observed for the “UV to FO” transition (AL = 23.9 %, AP = 3.1 %). All land cover transitions to forest
 259 were significantly different between the two mountain ranges (Mann-Whitney test: GR, UR with $p < 0.05$;
 260 CR, UR with $p < 0.01$).

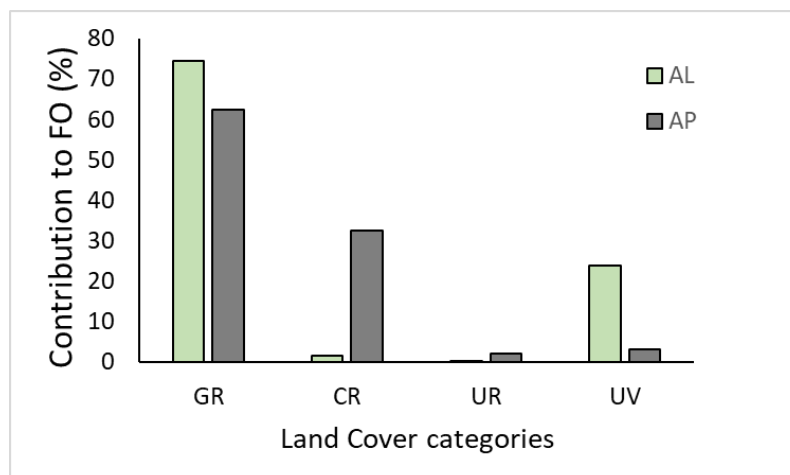
261



262

263 Figure 2. Area of land cover classes (ha) and land cover transitions from historic (1954-1962) to present time (2012) in
 264 the Alps (left panel) and the Apennines (right panel) study sites. Colored boxes refer to land cover categories with box
 265 size scaled to area: darker-colored inset boxes represent land cover class (LCC) persistence over time in the case of
 266 LCC increase (e.g. FO and UR categories); light-colored inset boxes represent persistence over time in the case of LCC
 267 decrease (e.g. GR, CR and UV). Transitions representing forest expansion are highlighted with arrows. Arrow thickness
 268 increases with magnitude of the land cover transition. The area converted to forest (ha) for each transition is reported
 269 in text above the arrow symbols.

270



271

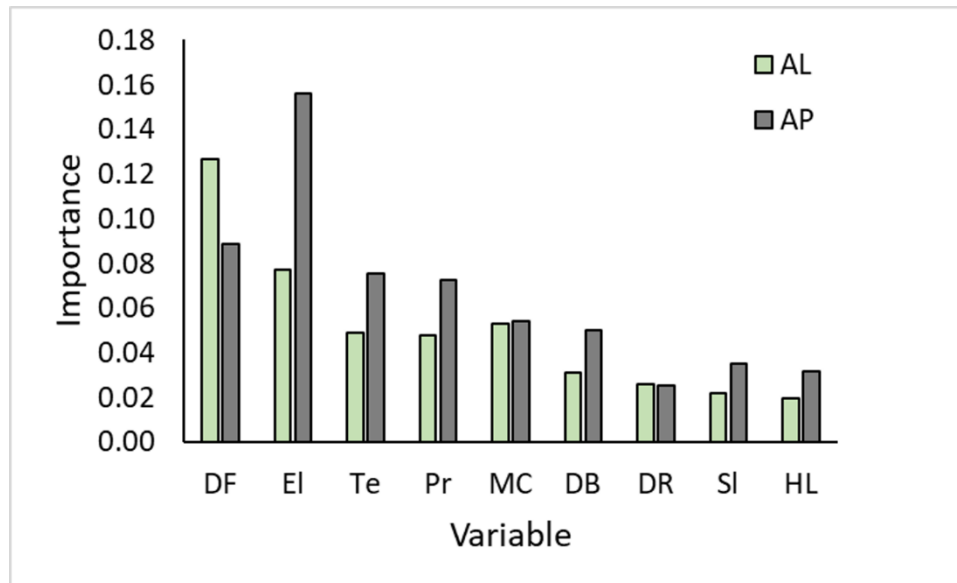
272 Figure 3. Percent contribution of each land cover category (GR = grassland, CR = cropland, UR = urban, UV =
 273 unvegetated) to forest expansion over the studied period (1954-2012).

274

275 *Drivers of forest expansion*

276 RF models derived for the forest expansion portion of the landscape indicate that the best predictor was
277 the distance from pre-existing forest edges (DF), particularly at AL (importance rate - $IR = 0.13$, Fig. 4).
278 Forest expansion is predicted also by elevation (EI), which is particularly important for AP mountain range
279 (AL: $IR = 0.08$, AP: $IR = 0.16$). Climatic variables such as mean annual temperature (Te) and precipitation (Pr)
280 were more important at AP ($IR = 0.07$ - 0.08). Anthropogenic impact proxy variables (MC, DB and DR) were
281 less influential in the models ($IR = 0.05$ - 0.03), but DB was more important for AP than for AL ($IR = 0.05$ and
282 0.03 respectively).

283

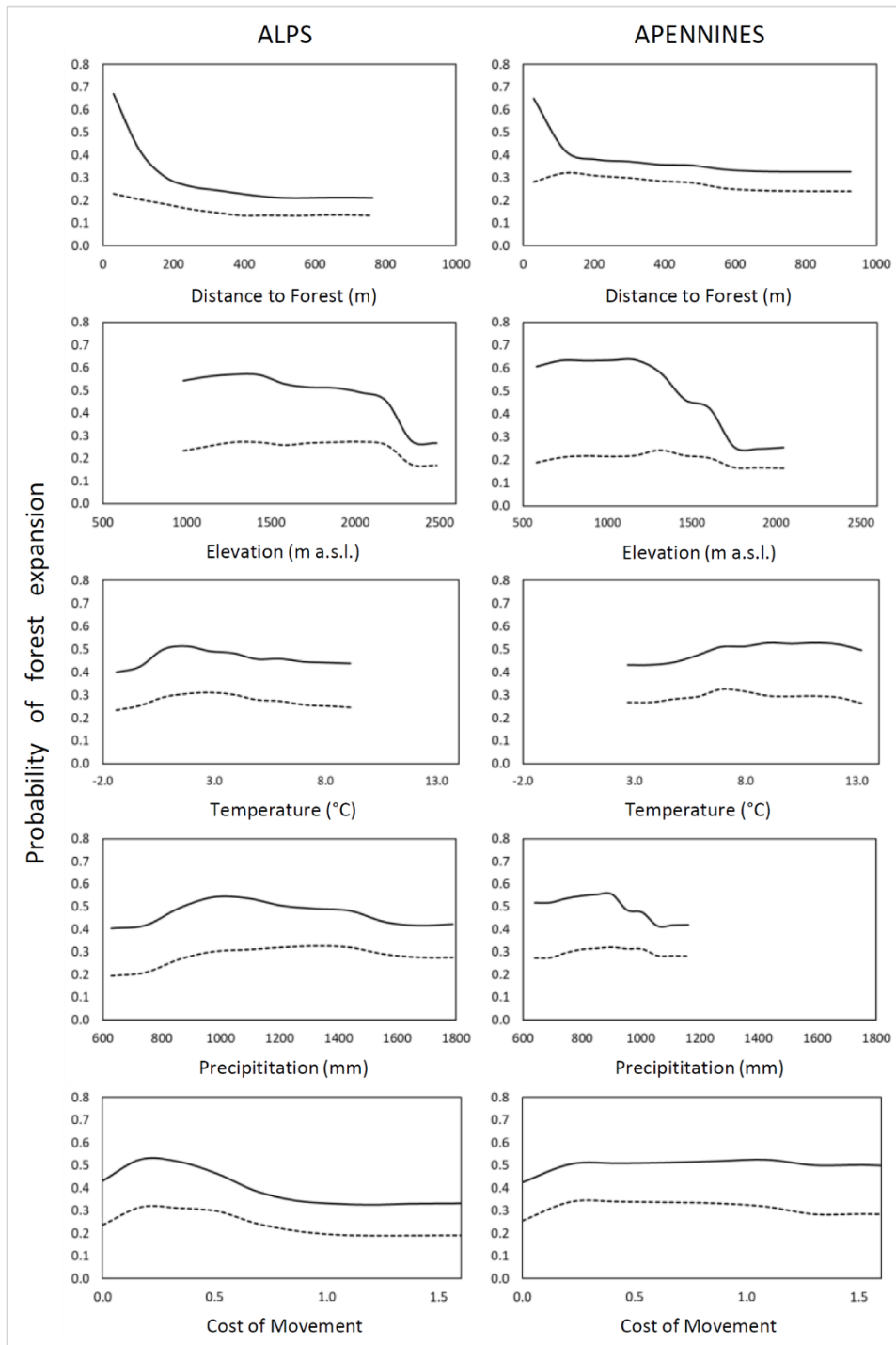


284

285 Figure 4. Importance rate of variables in random forest (RF) models of the Alps (light green) and the Apennines (grey).
286 Variables are: distance from pre-existing forest edges (DF), topographic (EI = elevation, SI = slope, HL = Heat load
287 index), climatic (Te = mean annual temperature, Pr = annual precipitation) and anthropogenic (MC = cost of
288 movement, DB = Euclidean distance to buildings, DR = Euclidean distance to roads).

289

290 At AL forest expansion probability was higher close to pre-existing forest edges, rapidly declining between 0
291 and 200 m and with a gradual decline between 200 and 800 m (Fig. 5). At AP the negative relationship
292 between forest expansion probability and distance to pre-existing forest edges featured a rapid decline
293 (from 0.6 to 0.4 of probability) between 0 and 150 m and a gradual decline between 150 and 900 m. The
294 effect of distance from pre-existing forest edges (DF) observed at AP exhibited a higher heterogeneity
295 compared to those observed at AL as highlighted by standard deviation computed at different values of the
296 predictor variable (Fig. 5). Forest expansion at AP was more likely to occur at lower elevations (500 – 1,000
297 m a.s.l.) and the probability abruptly decreased between 1,000 and 1,500 m a.s.l. A similar pattern was
298 observed at AL, but at higher elevations (1,000 – 2,000 m a.s.l.), with a clear decrease from 2,000-2,500 m
299 a.s.l. Relationships between forest expansion probability and annual temperature were generally weak,
300 although there was a slightly higher probability of forest expansion occurring between 2.5 - 7 °C at AL and
301 7.5 - 13 °C at AP.



302

303 Figure 5. Partial dependence plots showing relative influences of the five most important predictors on the probability
 304 of forest expansion, for the Alps (left) and the Apennines (right) across the respective input data ranges. We used all
 305 observations to train the model for computing the partial dependence function. Solid lines indicate the average over
 306 individual marginal effects of each selected explanatory variable (distance from pre-existing forest edges, elevation,
 307 temperature, precipitation and cost of movement) whereas dotted lines are the standard deviations of individual
 308 marginal effects.

309

310

311

312 Discussion

313 Forest expansion following land abandonment is a well-known process in mountain forests globally (Sitzia
314 et al. 2010). For example, forest expansion in mountainous populated areas has been recently detected in
315 South America (Nanni et al. 2019) and East Asia (Fang et al. 2014). Grazing decline and fire suppression
316 favored forest expansion and forest infilling in California, USA (Lydersen and Collins 2018) and recently rain
317 forest expansion into savanna has been detected in Australia (Ondeï et al. 2017). Natural forest expansion
318 is particularly evident and rapid in several mountain ranges of the World such as in the Mediterranean
319 Basin (e.g. Roura-Pascual et al. 2005; Niedrist et al. 2009; Weisberg et al. 2013) where agro-silvo-pastoral
320 traditional practices declined abruptly due to rural depopulation (Lasanta et al. 2017). However, there have
321 been few studies comparing post-abandonment forest expansion patterns among different regions or
322 mountain ranges (e.g. Tasser et al. 2007; Fontana et al. 2014).

323 By comparing the Italian Alps and the Apennines we found that environmental influences on forest
324 expansion processes were similar between the two regions. Our results for the observed time span (1954-
325 2012) indicate an overall forest area increase of 0.6% yr⁻¹ in the Italian Alps and Apennines. These values
326 match with the annual increments recently reported either for the entire Italian peninsula in 1985-2005
327 (0.3% yr⁻¹) and 2005-2015 (0.2% yr⁻¹) periods (RAF 2018) or for different sites of the Apennines (0.4 - 0.7%
328 yr⁻¹, Brachetti et al. 2012; Malandra et al. 2019). Similar rates are reported for other Alpine regions such as
329 Carnia (0.7% yr⁻¹), Tyrol (0.35% yr⁻¹) and South Tyrol (0.1% yr⁻¹) in the 1955-2000 period (Tasser et al. 2007).

330 Important differences between the two studied mountain ranges emerged from our analysis. Forest
331 expansion was more intense in the Apennines during the 1954-2012 period. This outcome could be related
332 to the different geographic layout of this mountain range and the wider latitudinal gradient that it
333 encompasses, providing warmer climate conditions more favorable for forest regeneration. Another
334 possible explanation for this pattern arises from the large differences in elevation gradients between the
335 two mountain ranges. Abandoned sites at lower elevations that were previously cultivated or grazed were
336 more common in the Apennines than in the Alps, and such sites experienced more rapid and extensive
337 forest expansion.

338 These differences highlight the importance of regional variation in climate and land use history for
339 understanding and predicting forest landscape change following agricultural abandonment. Differences in
340 land use history expressed by a mosaic of former croplands and pastures have important long-term
341 implications for post-abandonment forest establishment (Zimmermann et al. 2010). At the regional scale,
342 we found a greater reduction of grasslands in the Alps than in the Apennines, where we found a greater
343 increase of anthropogenic land cover types (mostly UR). The landscape transitions from grasslands,
344 croplands, and unvegetated lands to forests were by far the most relevant at our landscape scale of
345 analysis. Grassland-to-forest was the dominant shift in both mountain ranges due to a general decline of
346 traditional cattle grazing in mountain areas (e.g. Nagy et al. 2003). However, the two mountain ranges
347 differ in that the widespread transition from unvegetated areas (e.g. rocks and bare soil) to forest occurred
348 only in the Alps. Here, this transition is probably due to the tendency for coniferous treeline species (*Larix*
349 *decidua* and *Pinus cembra*) to invade higher-elevation, shrub-dominated and alpine plant community types
350 (Vittoz et al. 2008). On the other hand, *Fagus sylvatica* dominated treelines of the Apennines are less prone
351 to upward migration; forest expansion here was mostly the outcome of gap infilling processes (Vitali et al.
352 2018, Malandra et al. 2019). High elevation forests in the Apennines are dominated by *Fagus sylvatica*, a
353 strongly resprouting species, but with heavy seeds that disperse predominantly over short distances (Vitali
354 et al. 2017). Conifer species with greater long-distance seed dispersal ability occur at a few sites of the
355 central (*Pinus nigra* – Vitali et al. 2017; *Pinus mugo* – Dai et al. 2017) and southern (*Pinus heldreichii* – Vitali
356 et al. 2018) Apennines. Here, croplands-to-forest was the second most important transition, given the
357 possibility of growing a few rare food crops (e.g. potatoes, special cultivars of cereals, apples and chestnuts)

358 at higher elevations especially on southern or less exposed slopes (Bracchetti et al. 2012; Rovelli 2019). The
359 abandonment of upland traditional farming systems in the Apennines is one of the most important socio-
360 economic drivers of landscape degradation and biodiversity depletion (Farina 1995; Zimmermann et al.
361 2010).

362 Pre-existing forest edges emerged as a key land use legacy for future forest expansion both in the Alps
363 (Abadie et al. 2017; Tasser et al. 2007) and the Apennines (Malandra et al. 2019) with a greater importance
364 in the Alps. A general explanation for this is related to seed source availability and to the marginality of
365 ecotones such as forest edges. These are the first pastoral zones to be abandoned when grazing pressure is
366 reduced. The distance from pre-existing forest edges (years 1954-1962) appears a stronger driver in the
367 Alps where, because of harsh conditions at higher elevations, favorable microsites are necessary for tree
368 establishment. In the Apennines, where this variable was second in importance, high variability in effect
369 size is likely caused by heterogeneity among individual observations belonging to different landscapes. On
370 the other hand, in the Apennines, elevation was the most influential predictor variable with widespread
371 forest expansion having occurred at lower altitudes on slopes severely exploited prior to the analyzed time
372 span (1954-2012). The probability of forest expansion gradually decreased along an altitude gradient from
373 1,000 to 1,800 m a.s.l. in the Apennines, but decreased abruptly between 2,000 and 2,300 m a.s.l. in the
374 Alps. Forest expansion on former pastures and croplands was also faster at lower elevations in mountains
375 of southern Spain (Fernández et al. 2004).

376 The importance of land use legacy for forest landscape dynamics is emphasized by long-term studies
377 demonstrating that legacies may persist for decades, affecting current and future land cover changes
378 (Loran et al. 2017; Tasser et al. 2017). Human activities such as harvesting, grazing, fire and litter removal
379 when practiced for long time periods, may greatly affect current forest dynamics (Gimmi et al. 2013). The
380 role of land use legacies on past and future forest dynamics is typical of many southern European mountain
381 ranges shaped by historical anthropogenic disturbance regimes such as an intensive land use (Albert et al.
382 2008; Ameztegui et al. 2016). Our study confirms the importance of the location of pre-existing forest
383 edges as the legacy of centuries of human land use in mountain regions, as in other Mediterranean
384 mountain ranges such as the Pyrenees and the French Alps (Mouillot et al. 2005; Gartzia et al. 2016; Abadie
385 et al. 2017).

386 The differences observed in land cover change patterns between the Alps and the Apennines are not
387 surprising because of strong regional differences in climate, geology, topography, vegetation (e.g. treeline
388 species composition). Forest expansion by upward treeline rise or forest gap-filling processes has occurred
389 primarily on warmer and gentler slopes (e.g. southern exposure), whether in the Alps (Tasser et al. 2007;
390 Garbarino et al. 2013), the Apennines (Vitali et al. 2018) or the Pyrenees (Gartzia et al. 2016). More
391 favorable climate conditions and the greater availability of abandoned open areas make south-exposed low
392 elevation sites suitable areas for forest expansion. Ultimately, the more rapid rate of forest expansion in
393 the Apennines was linked to the greater availability of open areas given the more intense previous land
394 use. Tree encroachment on old pastures by secondary succession and on former unvegetated areas
395 through primary succession prevail in the Alpine region where climate change appears to have a strong
396 influence (Dirnböck et al. 2003; Gehrig-Fasel et al. 2007; Giorgi and Lionello 2008). Elevation plays a
397 fundamental role on forest dynamics at both landscape and stand scales in the Alps (Garbarino et al. 2009;
398 Kulakowski et al. 2011). Land use legacies such as conifer plantations and high elevation crop farming seem
399 stronger in the Apennines where climate change effects on forest expansion at high elevation appear
400 constrained by the unsuitability of *Fagus sylvatica* to migrate upwards due to its heavy seeds and its limited
401 ability to invade adjacent plant community types (Vitali et al. 2018, 2019). However, the limited transitions
402 from unvegetated areas to forests in the Apennines were also due to a combination of topographic
403 influences and the previous land use. In particular, several Apennines peaks with sandy or marl-sandy soils

404 are less topographically limited (mountain mass effect) so that unvegetated areas are rare and in the
405 absence of past human activities they are climatically suitable for forest dominance.

406 Forest expansion is an emerging and debated issue that requires accurate measurement and monitoring to
407 allow for proper management of current and future dynamics (Otero et al. 2015). There are two main
408 management strategies: i) passive management to support rewilding processes and limit human induced
409 landscape fragmentation and ii) active management to control and limit the negative effects of re-
410 vegetation processes (Lasanta et al. 2015). Negative effects of natural forest expansion include
411 simplification of landscape structure, decline of species diversity, increased risk of fire and soil erosion, and
412 the loss of cultural landscapes (Lasanta et al. 2015; Ferretti et al. 2019). A balance between conservation
413 through monitoring and active management of secondary succession dynamics (new forests) should be
414 attempted. A recent review, contrasting active management strategies with passive strategies allowing
415 forest secondary succession, found that the most efficient technique seems to be a combination of clearing
416 and extensive grazing (Lasanta 2019), maintaining high levels of landscape complexity and forest-meadow
417 edge.

418 With this study, by means of a standardized aerial image processing protocol we provided a robust dataset
419 that should be implemented with more comprehensive records (Garbarino et al. 2019). Quantitative
420 historical ecology with data on land use legacies can provide excellent information for ecosystem modelling
421 to predict forest landscape changes (Stürck and Verburg 2017).

422 We have shown that forest expansion in mountain ranges of Italy is controlled by land use legacies of pre-
423 existing forest edges, interacting with topography and climate. The Alps and the Apennines showed similar
424 landscape changes featuring grassland-to-forest transitions. However, the rate of forest expansion was
425 faster in the Apennines for the larger occurrence at lower elevations of old-fields recolonized by secondary
426 forests. In the Alps, climate and land use changes favored a widespread transition from unvegetated areas
427 to forest at higher elevations. Our results could be biased by the stronger mass effect on the Alps, and the
428 higher average elevation of alpine landscapes. A further limit in our approach is the spatial resolution
429 mismatch between the 1-km climate data resolution and the 30 m unit of forest expansion analysis. Thus,
430 the influence of climatic variables on forest expansion could be underestimated by our random forest
431 models.

432 Despite these limitations, our results demonstrate that post-abandonment forest expansion is a
433 widespread and ongoing process in Italian mountain forest landscapes. Future research should increase the
434 number of surveyed sites for increased sensitivity in comparing regional differences. It would be
435 informative to apply a land use change modeling approach. Predicting new landscape scenarios for Italian
436 mountain forests should account for the possible changes to disturbance regimes linked to climatic changes
437 (Vacchiano et al. 2017). The extensive forest cover that is blanketing large mountain areas has important
438 implications for habitat and biodiversity. Forest expansion in these mountain landscapes additionally leads
439 to an increase of fuel load continuity that increases the risk of wildfires, particularly in areas exposed to
440 severe drought stress increased by recent climate changes. Large wildfires have recently increased in
441 occurrence in mountain areas where they have historically been quite rare due to the prevalence of
442 managed pastures and farmlands. Other implications regard snowfall and accumulation regimes in
443 combination with soil erosion dynamics after forest fire. These issues are connected also to the naturalness
444 of these processes triggered by man in human-shaped landscapes. A no-management approach of the
445 successional processes would not guarantee, at least in the short term, a strictly natural forest
446 encroachment. A comprehensive overview and assessment of the multiple ecosystem services provided by
447 these complex and millenary landscapes is necessary in order to attempt a sustainable forest management
448 (Schulze and Schulze, 2010).

449

450 **Author contribution**

451 Matteo Garbarino conceived the experiment and all authors contributed to the study design. Material
452 preparation, data collection and analysis were performed by Matteo Garbarino, Donato Morresi, Emanuele
453 Sibona, Francesco Malandra, Alessandro Vitali and Carlo Urbinati. The first draft of the manuscript was
454 written by Matteo Garbarino and all authors commented on previous versions of the manuscript. All
455 authors read and approved the final manuscript.

456 **Data availability**

457 The datasets generated during and analyzed during the current study are available in the Figshare
458 repository, <https://doi.org/10.6084/m9.figshare.11409444.v1>

459

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- 686

687 **Supplemental material**

688 Table S1. Harmonization details of the Corine Land Cover legend in 5 land cover categories at AL and AP.

Corine Land Cover		Harmonization	
Code	Description	Code	Description
111	Continuous urban fabric	UR	Urban
112	Discontinuous urban fabric	UR	Urban
121	Industrial or commercial units	UR	Urban
122	Road and rail networks and associated land	UR	Urban
123	Port areas	UR	Urban
124	Airports	UR	Urban
131	Mineral extraction sites	UR	Urban
132	Dump sites	UR	Urban
133	Construction sites	UR	Urban
141	Green urban areas	UR	Urban
142	Sport and leisure facilities	UR	Urban
211	Non-irrigated arable land	CR	Cropland
212	Permanently irrigated land	CR	Cropland
213	Rice fields	CR	Cropland
221	Vineyards	CR	Cropland
222	Fruit trees and berry plantations	CR	Cropland
223	Olive groves	CR	Cropland
231	Pastures	GR	Grassland
241	Annual crops associated with permanent crops	CR	Cropland
242	Complex cultivation patterns	CR	Cropland
243	Land principally occupied by agriculture with ... natural vegetation	CR	Cropland
244	Agro-forestry areas	CR	Cropland
311	Broad-leaved forest	FO	Forest
312	Coniferous forest	FO	Forest
313	Mixed forest	FO	Forest
321	Natural grasslands	GR	Grassland
322	Moors and heathland	FO	Forest
323	Sclerophyllous vegetation	FO	Forest
324	Transitional woodland-shrub	FO	Forest
331	Beaches dunes sands	UV	Unvegetated
332	Bare rocks	UV	Unvegetated
333	Sparsely vegetated areas	UV	Unvegetated
334	Burnt areas	FO	Forest
335	Glaciers and perpetual snow	UV	Unvegetated
411	Inland marshes	UV	Unvegetated
412	Peat bogs	UV	Unvegetated
421	Salt marshes	UV	Unvegetated
422	Salines	UV	Unvegetated
423	Intertidal flats	UV	Unvegetated
511	Water courses	UV	Unvegetated
512	Water bodies	UV	Unvegetated
521	Coastal lagoons	UV	Unvegetated
522	Estuaries	UV	Unvegetated
523	Sea and ocean	UV	Unvegetated
999	NODATA	999	No Data

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690 Table S2. Classification accuracy (OA = overall accuracy, K = Cohen's Kappa coefficient) of 32 land cover
 691 maps (16 landscapes x 2 periods).

Landscape	Year	OA	K
BAG	1962	79	0.69
	2012	89	0.83
CIM	1954	82	0.73
	2012	93	0.88
DEV	1954	82	0.71
	2012	86	0.77
GEN	1954	78	0.70
	2012	78	0.67
GRS	1954	80	0.67
	2012	86	0.76
MAT	1954	86	0.77
	2012	94	0.88
MEL	1962	82	0.73
	2012	88	0.82
MMA	1954	91	0.83
	2012	90	0.68
MOR	1954	82	0.69
	2012	90	0.83
MUS	1961	82	0.72
	2012	89	0.84
PES	1954	88	0.78
	2012	86	0.73
SAP	1954	84	0.73
	2012	91	0.84
SIB	1954	86	0.80
	2012	86	0.77
TER	1954	85	0.75
	2012	84	0.71
VEG	1954	87	0.81
	2012	91	0.87
VEN	1961	94	0.89
	2012	96	0.93

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700 Table S3. Random forest model parameters used in this study (N transitions = number of forest expansion
 701 pixels, N total = number of total pixels in the model, OOB = Out-of-bag error or prediction error of RF
 702 models, Brier score = Brier predictive performance estimate, AUC = area under the receiver operating
 703 characteristics curve, K = Cohen's Kappa coefficient) computed for the AL and the AP datasets both at 30
 704 and 60 m spatial resolutions.

RF models	AL 30 m	AP 30 m	AL 60 m	AP 60 m
N transitions	14420	21027	3555	5184
N total	28900	40575	7082	10010
<i>OOB</i>	0.108	0.091	0.126	0.098
<i>Brier score</i>	0.175	0.151	0.182	0.132
<i>AUC</i>	0.829	0.854	0.819	0.849
<i>K</i>	0.482	0.519	0.483	0.510
<i>DF IR</i>	0.127	0.088	0.114	0.072
<i>EI IR</i>	0.077	0.156	0.065	0.168
<i>MC IR</i>	0.053	0.054	0.034	0.039
<i>Te IR</i>	0.049	0.076	0.033	0.051
<i>Pr IR</i>	0.048	0.073	0.028	0.060
<i>DB IR</i>	0.031	0.050	0.016	0.033
<i>DR IR</i>	0.026	0.025	0.014	0.012
<i>SI IR</i>	0.022	0.035	0.013	0.030
<i>HL IR</i>	0.019	0.032	0.011	0.023
<i>Total IR</i>	0.452	0.589	0.329	0.489

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707 Table S4. Description, data sources, spatial characteristics and usage rationale of explanatory and predictor
 708 variables used in the Random Forest models.

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Variable	Code	Type	Resolution	Data source	Description
Forest expansion	FE	Binary	10 m	LC map	Transition to forest occurrence
Elevation	EI	Topographic	10 m	DEM tinality	Gradient of site suitability
Slope	SI	Topographic	10 m	DEM tinality	Proxy of human pressure
Heat Load Index	HL	Topographic	10 m	DEM tinality	Gradient of site suitability
Precipitation	Pr	Climatic	1 km	CHELSA	Average annual precipitation
Temperature	Te	Climatic	1 km	CHELSA	Average annual temperature
Distance to Forests	DF	LU Legacy	10 m	LU 1954	Distance to former forest borders
Distance to Roads	DR	Anthropic	10 m	OSM	Euclidean distance from roads
Distance to Buildings	DB	Anthropic	10 m	OSM	Euclidean distance from buildings
Moving cost	MC	Anthropic	10 m	DEM - OSM	Cost of movement across the terrain

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716 Table S5. Land cover surface as a percentage of the total surface of AL and AP landscapes (10091 and 12830
717 ha, respectively) divided by regional (CORINE land cover) and landscape (aerial imagery) scales. For
718 comparison purposes, the considered study area is included within the borders of the 16 selected
719 landscapes.

Regional scale (CORINE LC)				Landscape (Aerial imagery)			
AL	1990	2018	Delta	AL	1954	2012	Delta
FO	51.58	53.02	1.44	FO	38.75	48.82	10.07
GR	16.16	4.43	-11.73	GR	28.35	24.51	-3.84
CR	0.81	0.55	-0.26	CR	0.37	0.10	-0.26
UR	0.04	0.04	0.00	UR	0.15	0.30	0.15
UV	31.40	41.96	10.55	UV	32.38	26.26	-6.11
AP	1990	2018	Delta	AP	1954	2012	Delta
FO	67.16	65.98	-1.18	FO	46.99	65.40	18.41
GR	16.03	10.82	-5.21	GR	36.11	26.67	-9.44
CR	3.52	3.66	0.14	CR	11.17	0.98	-10.19
UR	0.65	0.70	0.05	UR	0.95	1.47	0.52
UV	12.63	18.84	6.21	UV	4.78	5.49	0.70

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