UNIVERSITÀ POLITECNICA DELLE MARCHE

Doctoral School in Agriculture, Food and Environmental Sciences

XXXII cycle

Coordinator: Prof. Bruno Mezzetti

Land use/cover shifts and wildfires as drivers of mountain forest landscape dynamics in the Apennines (Italy)

Doctoral dissertation

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November 2016 – October 2019

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Abstract

This dissertation is a compilation of 2 published original articles (Chapters 2-3), 1 article in progress (Chapters 4) and some bridging chapters. Each research chapter stands alone, containing its own introduction, materials and methods, results and discussion, conclusions and appendices. Specifically, the material in chapter 2 is an article published in *Forests*, chapter 3 an article in *Regional Environmental Change* in collaboration with researchers from the University of Nevada (Reno-USA) and University of Turin (Italy). Finally, chapter 4 is an article in progress with the collaboration of researchers from University of California (Berkeley-USA) after a visiting research position in 2019.

This research aimed to study landscape dynamics along the Italian peninsula, due to land use/cover shifts and recent wildfires, within the contest of climate change. Land use changes were mainly focused on the mountain areas along the whole Apennines range, whereas the landscape effects of two recent and extreme fire seasons were assessed along the Italian peninsula and the two major islands (Sardinia and Sicily).

Land-use science usually adopts the case-study approach to investigate landscape changes, therefore we considered a meta-analysis an appropriate and preliminary tool for summarizing general patterns and heterogeneous findings from several case-studies over a large geographic area. Apennines landscapes have undergone significant transformations throughout the last century due to national and regional socio-economic changes. Out of 51 published articles and different databases that referred to 57 case-studies, we explored heterogeneous data sets, adopting a stepwise approach to select the case-studies including: i) a general overview of the main studies, ii) an analysis of the study sites features and of land-use/cover transitions and iii) a landscape pattern analysis. We standardized the processing methods and obtained a new set of homogeneous data for a comparative analysis. After pre-processing of the selected papers to reduce data heterogeneity, we computed *ex-novo* common landscape metrics. We obtained digital images subjected to automatic segmentation. Most case studies were conducted in Central and Southern Apennines and 83% were at local scale, 77% applied change detection and only 38% included both change detection and landscape spatial pattern analysis. We found a clear trend of forest expansion (+78%) and the decrease of croplands (-49%) and grasslands (-19%) but not significant changes in landscape spatial patterns.

In a second article, we used a replicate landscape approach and a systematic sampling design for quantifying changes at regional scale. Previous studies of land-use change in forest landscape were conducted in relatively small areas and used heterogenous sampling protocols. We investigated land-cover changes and landscape configurational shifts comparing different slope aspects and altitudinal zones and assessed the main drivers of the non-forest/forest dynamics. At each of the 10 representative sites located along the Apennines, we selected two paired study landscapes (NE vs. SW

slopes) each with a surface of 16 km^2. We applied object-based classification to aerial photos of 1954 and 2012 and produced 40 land-cover maps. Computing land-cover transitions and key landscape metrics and applying multivariate statistics and binomial generalized linear models (GLMs), we measured the overall landscape changes. The Apennines landscape mosaics lost structural complexity at lower elevation with shrubs and trees encroachment in abandoned pastures whereas a diffuse fragmentation of historical grasslands was detected at higher elevation due to development of woody vegetation patches beyond the forest-grassland ecotone. Forest expansion occurred more rapidly at lower elevations, on steeper slopes, and closer to existing forests and cultivated areas. A replicate landscape approach proved useful for quantifying changes to forest cover and landscape structure along complex gradients of topography and land-use history, following a diffuse agro-pastoral abandonment.

Finally, we studied the occurrence, the severity and the landscape effects of two major forest fire seasons in peninsular Italy and Sicily and Sardinia islands. This part is a preliminary draft of a manuscript in progress to be submitted shortly to an indexed international journal. This study provides descriptive results on fire characteristics (e.g. extension, severity) of 113 forest fires that occurred in two exceptionally dry years (2007, 49 fires and 2017, 64 fires). An analysis of drought was carried out in both years in the central and southern Italian regions, using the self-calibrated Palmer Drought Severity Index. We used burned area products provided by the MODIS database and selected all forest fires larger than 100 ha. We collected Landsat TM and OLI multi-spectral images of selected areas and computed the Relative difference NBR index (RdNBR) for each event. We applied a semi-automatic procedure (automatic segmentation and manual classification) to extract the fire perimeters and to assess fire severity. Drought conditions were more severe in 2017 especially in central Italy but large forest fires (>100 ha) occurred more in southern regions, mainly in July and August. The average fire surface area was larger in 2007 and shrublands and broadleaf forests were the land categories more widely burned. Approximately 2400 ha burned in both years (2% of total combined burned area). We did not detect relevant differences in severity of combined 2007 vs. 2017 fires, but conifers and shrublands had the highest values in both years. A predictive model will be applied to define the landscape sensitivity to fire, their main drivers and the severity of forest fires.

CHAPTER 1

General introduction

Besides recent disputes about the role of human activities on global change, many scientists agree upon the concept of "Anthropocene", a new geological era severely shaped by human footprint (Steffen et al. 2007). Among the numerous related topics, climate change and land use/cover changes (LULCC) are dynamic processes occurring at multiple spatio/temporal scales (Houghton et al. 2003; Foley et al. 2005) affecting synchronously biosphere and society (Skole et al. 1997; Gillanders et al. 2008). Mean global air temperature has increased of 1.52 °C since 1850 and a warmer climate is likely to raise the frequency of heat waves and drought periods affecting differently various ecosystems (IPCC SRCCL 2019). Nature and extent of LULCC need to be quantified for assessing their interaction with ecosystem processes, biogeochemical cycles, biodiversity, and climate (Turner et al. 1994, Goldewijk 2001). Historical land use is widely considered a fundamental constraining factor shaping current land cover (Gimmi et al. 2008; Garbarino et al. 2013) and constraining future landscape response to environmental change (Foster et al. 1998). We consider necessary to study LULCC at different spatial and temporal scales (Ramankutty et al. 2002; Pongratz et al. 2008) taking advantage of large amount of historical maps recently available and the improvements and increased remotely sensed data (Gutman et al. 2004; Paudel et al. 2016).

In Europe, ecosystems and biota coevolved under anthropic pressure, generating the so-called cultural landscapes (Naveh 1995). However, in many European mountain landscapes, the decline of agro-silvo-pastoral activities progressively decreased and induced secondary ecological succession, featuring forest expansion or forest gap filling (Améztegui et al. 2010) in former grasslands or crops. Broad-scale abandonment caused vegetation homogenization (Lasanta-Martineza et al. 2005) and loss of landscape heterogeneity (Bracchetti et al. 2012). Scientific literature on LULCC in Italy has increased rapidly in the last decades (Malandra et al. 2018) but a large share of it concerns the Alps rather than the Apennines, due to the transnational layout of the alpine range (Zimmermann et al. 2010; Romano & Zullo 2016). Nonetheless, the Apennines are the second largest mountain range of Italy, extending along the peninsula for over 1200 km and covering, with the Alps, about 35% of the total surface area of Italy (Vacchiano et al. 2016). The harsh morphology together with climate variations and the millenary human pressure have shaped the forest landscape throughout time creating a vegetational mosaic and increasing biodiversity (Vacchiano et al. 2016). A notable migration from mountain/rural to valley/urban areas occurred especially after World War II and caused: i) a diffuse abandonment of silvo-pastoral activities (Falcucci et al. 2007; Bakudila et al. 2015), ii) a widespread transition from grasslands and croplands to shrublands and forests and iii) an overall decrease of landscape

heterogeneity (Peroni et al. 2000; Cimini et al. 2013). However, most of these Apennines studies are single cases and used very heterogeneous sampling protocols making cross comparison unfeasible (Malandra et al. 2018). Therefore, we engaged in a research adopting a homogenous sampling design and rigorous analytical methods in order obtain consistent results on landscape change and its drivers at multiple spatial scales.

The widespread process of natural forest expansion/filling in former grasslands caused controversial effects according to conservation, landscape and forest ecologists (Sitzia et al. 2010). In climatic sensitive areas like the Mediterranean basin, such expansion contributes to increase the fire risk due to the larger fuel load availability and its spatial continuity (Pausas & Fernandez Munoz 2012; Barros et al. 2014). In fact, in a context of climate warming, this can greatly increase frequency, extension and severity of wildfires (San-Miguel-Ayanz et al. 2013; De Rigo et al. 2017). Forest recovery dynamics after high fire severity and fire regime changes are important research topics for providing appropriate management strategies for fire prevention and landscape restoration (Catry et al. 2012). Forest fires are an ecological disturbance triggered by natural events (e.g. lightnings) or diverse human activities (e.g. accident/negligence or arson) (Pausas & Vallejo 1999). In the Italian peninsula and major islands, as in other Mediterranean areas, human ignition is very relevant and depends on the socio-economic context of the region concerned (Ganteaume et al. 2013). Here, landscapes experienced low-to-moderate-severity fires in former times, but the increasing amount of woody fuel, the climate warming trend together with some socio-cultural variables are changing the threshold of fire risk. Stand replacing events are likely to increase in frequency and extent (Vacchiano et al. 2017) like those that occurred in 2007 and 2017 in Italy. Research and modelling these aspects could help predict the future trend and help explain related shifting disturbances such as wildfires. These conditions should induce land planners, managers and fire control agencies to shift their prevention, control and mitigation measures from local to landscape or regional scale.

CHAPTER 2

70 years of land use/land cover changes in the Apennines (Italy)

This chapter has been revised and accepted for online publication on Sept. 7th, 2018:

Malandra F.[1], Vitali A.[1], Urbinati C.[1], Garbarino M.[2], 2018. 70 years of land use/land cover changes in the Apennines (Italy): a meta-analysis. Forests. DOI: https://doi.org/10.3390/f9090551

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Abstract

Land-use science usually adopts a case-study approach to investigate landscape change processes, so we considered a meta-analysis an appropriate tool for summarizing general patterns and heterogeneous findings across multiple case-studies over a large geographic area. Mountain landscapes in the Apennines (Italy) have undergone significant variations in the last century due to regional and national socio-economic changes. In this work, we reviewed 51 manuscripts from different databases and examined 57 case-studies. We explored heterogeneous data sets, adopting a stepwise approach to select the case-studies: Step 1) a general overview of the main studies; Step 2) an analysis of the features of the study sites and of land-use/cover transitions; Step 3) a landscape pattern analysis. We standardized the processing methods to obtain a new set of homogeneous data suitable for comparative analysis. After some pre-processing of the selected paper due to the broad heterogeneity of the data, we calculated common landscape metrics ex novo. We obtained digital images used to perform automatic segmentation with eCognition Developer 64 software. Our review indicated that most case studies were in Central and Southern Italy, 83% were examined at local scale, 77% carried out change detection, but only 38% included both change detection and landscape spatial pattern analysis. The results revealed a clear trend of forest expansion (+78%) and the reduction of croplands (-49%) and grasslands (-19%). We did not find significant changes in the landscape spatial patterns.

Keywords: LULCC; review; silvo-pastoral systems; new forests, cultural landscapes; crop land abandonment.

Introduction

Climate change and land use/land cover changes (LULCC) are considered to be drivers of global importance (Houghton 2003; Foley et al. 2005) affecting society and biosphere systems (Skole et al. 1997; Gillanders et al. 2008). LULCC is commonly perceived to be a local process (Lambin et al. 2006), although the nature and extent of changes occurring at broader scales (regional and global) should be quantified to understand human-driven dynamics on the earth's surface (Turner et al. 1994; Goldewijk 2001). The impact of humans on the biosphere has been so significant that scientists recently introduced the "Anthropocene" concept whereby planet Earth is shifting to a new geological epoch in which human activities are severely altering the natural environment (Steffen et al. 2007). It is very important to study landscape changes at different spatial and temporal scales (Ramankutty et al. 2002; Pongratz et al. 2008) and to provide more standardized approaches given the wide diversity of methods and data sources used (Rozenstein et al. 2011). In recent decades, improvements in remote sensing (RS) techniques and the increased availability of RS data (Gutman et al. 2004) facilitated the analysis of LULCC worldwide (Paudel et al. 2016). In anthropogenic ecoregions like the Mediterranean basin, LULCC are of extreme interest because they can affect the conservation of "cultural landscapes" (Naveh 1995; Godone et al. 2014). In Italy for example, the literature on LULCC has increased in the last three decades. Scientific works are available for the Alps in particular, a transnational mountain range studied by Italian, French, Swiss and Austrian researchers (Zimmermann et al. 2010; Romano & Zullo 2016). The Apennines - the main peninsular orographic system - have also merited attention due to the significant landscape changes since World War II. The Alps and Apennines together cover about 35% of the entire surface area of Italy (Vacchiano et al. 2016). Both have been affected by the presence and activities of man throughout the millennia, but given their lower elevation, greater accessibility and the more intensive livestock transhumance, the Apennines show clearer signs of change. The dwindling human presence and pastoral activities in many mountain areas has progressively triggered secondary ecological successions that have changed the physiognomy of the landscape in recent decades and altered its structure and functions. A significant study conducted over 50 years (1950-2000) at national scale in 6 main geographic areas found a significant increase in forest cover and a decrease in agricultural cover, explained by the population decrease in both mountain systems (Falcucci et al. 2007).

The Apennines were heavily exploited for firewood, charcoal production and wood pasture for many centuries until around 1960. National reforestation programs started before WW2 to reduce the severe slope erosion in mountainous and hilly areas and continued in the 1950s, and 1960s in particular, adding about 760,000 ha of plantations (mainly conifer tree species) in the Apennines (Piermattei et al. 2016; Vacchiano et al. 2016). Meanwhile rural population migration towards urban and industrial

areas resulted in the general abandonment of rural activities and mountain settlements (De Sillo et al. 2012; Frate et al. 2014; Bakudila et al. 2015) causing natural forest gap-filling and natural reforestation dynamics (mainly broadleaf species). The reduction of forest-grassland ecotones also resulted in a reduction in plant species diversity (Gartzia et al. 2014), contributing to the disappearance of "cultural landscapes" (Zagaria et al. 2017).

The increasing percentage of land subject to these changes in land use/ land cover and resulting changes in ecological landscape patterns led to more detailed studies being carried out to quantify and describe these changes and their effects on mountain ecosystems more clearly. The interdisciplinary nature of LULCC studies and increasing amount of published literature can hamper the efficient tracking of the latest updates and possible connections among the different studies (Wang & Feng 2011). Several LULCC studies are single case studies providing local results and not suited towards generalization at regional or national level. In order to compare local case studies, they have to be fit into a common framework to analyze processes using the same methodological and conceptual approaches (Van Vliet et al. 2016). Therefore, the aims of this review were: i) to collect all the existing literature concerning the LULCC in the Apennines, and ii) conduct a meta-analysis of selected studies to detect possible common patterns over the past 70 years. This work has been challenging due to the wide heterogeneity of sources, geographic locations and extent of study areas and different methods applied (Falcucci et al. 2007).

Materials and Methods

Literature search and data extraction

We carried out a search using ISI Web of Science, Elsevier Scopus databases and Google Scholar up to December 2017 to analyze the published Italian and English literature on LULCC in the Apennines and adjacent areas, using the following keyword combinations: land use change* OR land cover change* OR landscape dynamics* AND Apennines*. We included studies carried out in Eastern Sicilian mountains considered to be the orographic prolongation of the Apennines (Finetti 2005). We included reports, book chapters, proceedings of Italian and international conferences, PhD thesis and other grey literature. The selection of research studies included in the meta-analysis was based on three criteria: i) location of study sites in the Apennines; ii) availability of land use/land cover transition data in a suitable time interval; iii) availability of data on forest-cover categories. We did not consider elevation as a selection criterion.

We adopted a stepwise approach for case studies eligible for the meta-analysis: i) studies with comparable descriptive information (Step 1); ii) studies reporting suitable topographic, climatic and anthropogenic impact data and/or that provide LULC data (Step 2); iii) studies providing thematic maps suitable for landscape pattern analysis (Step 3) (Table 1).

Table 1 - List of case studies ordered by sequential ID number. Source type: IA indexed article; NA non-indexed article; PR conference proceeding; GL grey literature; TH doctoral thesis; BC book chapter. The selection step/s used for the different analyses: summary review (1), study site description and LULC transitions analysis (2), landscape pattern analysis (3). The case study code is the site name abbreviation. ROM = Romagnese; OLT = Oltrepò Pavese; PEV = Perino valley; SRB = Samoggia River Basin; BTO = Borgo Tossignano; MRW = Magra river watershed; MCP = Massa Carrara province; MTV = Mt. Vigese; LRB = Lamone river basin; MOS = Moscheta; CSM = Cutigliano-San Marcello; CAR = Cardoso; SPA = San Paolo in Alpe; LMF = Lama forest; EMR = Emilia Romagna region; PHI = Pisan hills; GAR = Gargonza; SIP = Siena province; PDO = Poggio dell'Olmo; ACQ = Acqasanta Terme; LAG = Laga; MIC = Micigliano; RPP = Rieti province; RIE = Rieti province; ATV = Aterno valley; SEP = S. Eufemia and Pacentro; MAM = Majella massif; SIM = Simbuini mountains; CEV = Cervara valley; COM = Collemeluccio-Montedimezzo; LEA = Lepini mountains A; MOM= Monti del Matese; TAB = Taburno; CDA = Conca di Avellino; HAV = High Agri Valley; AGV = Agri valley; PNP = Pollino National Park; SMR = Sila mountain range; SSB = Serra San Bruno; NEB = Nebrodi; ETN = Etna; MAD = Madonie; FCA = Forests of Campania; LEB = Lepini mountains B; LEC = Lepini mountains C; PRE = Premilcuore; COR = Corniolo; CAS = Castagno d'Andrea.

ID	Reference	Source type	Selection step/s	Study area code	Area covered (ha)	Time range (years)	LULC classes(n)	Landscape metrics(n)
1	Assini et al. 2014	IA	1-2-3	ROM	650	54	7	4
2	Filipponi et al. 2012	PR	1-2	OLT	650	54	5	0
3	Vincini et al. 2003	GL	1	PEV	22000	45	1	0
4	Brath et al. 2006	IA	1-2	SRB	17800	37	7	0
5	Mariano et al. 2008	PR	1-2	BTO	-	48	3	2
6	Farina 1995	BC	1	MRW	-	40	6	0
7	Farina 1991	NA	1	MCP	-	40	0	1
8	Pezzi et al. 2011	IA	1-2-3	MTV	617	51	11	0
9	Benini et al. 2010	IA	1-2-3	LRB	12318	27	11	6
10	Agnoletti 2002	BC	1-2-3	MOS	830	46	19	0
11	Bonavita et al. 2007	NA	1-2	CSM	12634	59	6	0
12	Agnoletti et al. 2007	IA	1-2	CAR	1054	48	21	3
13	Argenti et al. 2006	NA	1-2	SPA	214	42	8	0
14	Vazzano et al. 2011	NA	1	LMF	1854	27	28	0
15	Campiani et al. 2001	GL	1	EMR	-	18	18	0

16	Bertacchi & Onnis 2004	BC	1-2	PHI	900	47	1	0
17	Agnoletti 2002	BC	1-2-3	GAR	267	46	14	0
18	Geri et al. 2010	IA	1-2-3	SIP	-	46	3	6
19	Rocchini et al. 2006	IA	1-2-3	PDO	440	44	7	4
20	Bracchetti et al. 2012	IA	1-2	ACQ	16800	51	8	3
21	Cimini et al. 2013	NA	1	LAG	143	19	15	0
22	Pelorosso et al. 2007	TH	1-2	MIC	3619	45	5	0
23	Pelorosso et al. 2009	IA	1-2	RPP	-	40	7	5
24	Pelorosso et al. 2007	TH	1-2	RIE	-	40	8	0
25	Peroni et al. 2000	IA	1	ATV	4000	41	5	6
26	Van Gils et al. 2008	IA	1	SEP	8700	28	5	0
27	Palombo et al. 2013	IA	1	MAM	14440	53	1	0
28	De Sillo et al. 2012	IA	1-2-3	SIM	35000	50	9	8
29	Piovesan et al. 2005	IA	1	CEV	-	49	6	0
30	Frate et al. 2014	IA	1	COM	25000	57	3	8
31	Smiraglia et al. 2007	IA	1-2-3	LEA	11294	46	14	1
32	Corso et al. 2010	PR	1-2	MOM	2297	43	8	3
33	Napolitano et al. 2003	PR	1-2	TAB	5300	44	10	2
34	Fichera et al. 2012	IA	1-2-3	CDA	57355	50	4	4
35	Simoniello et at. 2015	IA	1-3	HAV	72500	24	14	8
36	Romano et al. 2007	IA	1-2	AGV	35669	43	10	9
37	Gargano et al. 2011	IA	1	PNP	74000	14	3	1
38	Nicolaci et al. 2014	IA	1-2	SMR	130200	71	7	0
39	Modica et al. 2012	IA	1-2-3	SSB	4035	51	9	7
40	La Mela Veca et al. 2016	IA	1-3	NEB	437	57	6	0
41	La Mela Veca et al. 2016	IA	1	ETN	422	57	6	0
42	La Mela Veca et al. 2016	IA	1	MAD	527	57	6	0
43	Migliozzi et al. 2010	PR	2	FCA	-	-	-	-
44	Smiraglia et al. 2007	IA	2	LEB	-	-	-	-
45	Smiraglia et al. 2007	IA	2	LEC	-	-	-	-
46	Pelleri et al. 2003	PR	2	PRE	-	-	-	-

47	Pelleri et al. 2003	PR	2	COR	-	-	-	-
48	Pelleri et al. 2003	PR	2	CAS	-	-	-	-
49	Falcucci et al. 2007	IA	-	-	-	-	-	-
50	Salvati et al. 2015	IA	-	-	-	-	-	-
51	Falcucci et al. 2008	IA	-	-	-	-	-	-
52	Marchetti et al. 2012	NA	-	-	-	-	-	-
53	Pompei et al. 2007	TH	-	-	-	-	-	-
54	Corona et al. 2005	NA	-	-	-	-	-	-
55	Corona et al. 2008	IA	-	-	-	-	-	-
56	Massimi & Tubito 1998	GL	-	-	-	-	-	-
57	Munafò et al. 2015	GL	-	-	-	-	-	-

We first excluded the case studies that did not provide clear descriptive information such as geographic coordinates of the area or the type of analysis carried out (Step 1). The name, location and surface of study sites, time-period of analysis, material used (aerial photos, satellite imageries, field data), type of overall land-use/cover categories, type of forest classes, and presence and type of computed landscape metrics were all considered to be necessary to compare the case studies. Selected sites were plotted (Figure 1) using the available coordinates or those assigned to the centroid of each study-area. We ended up with 42 study sites out of the initial 57 (from 51 documents) after step 1 of the selection process.

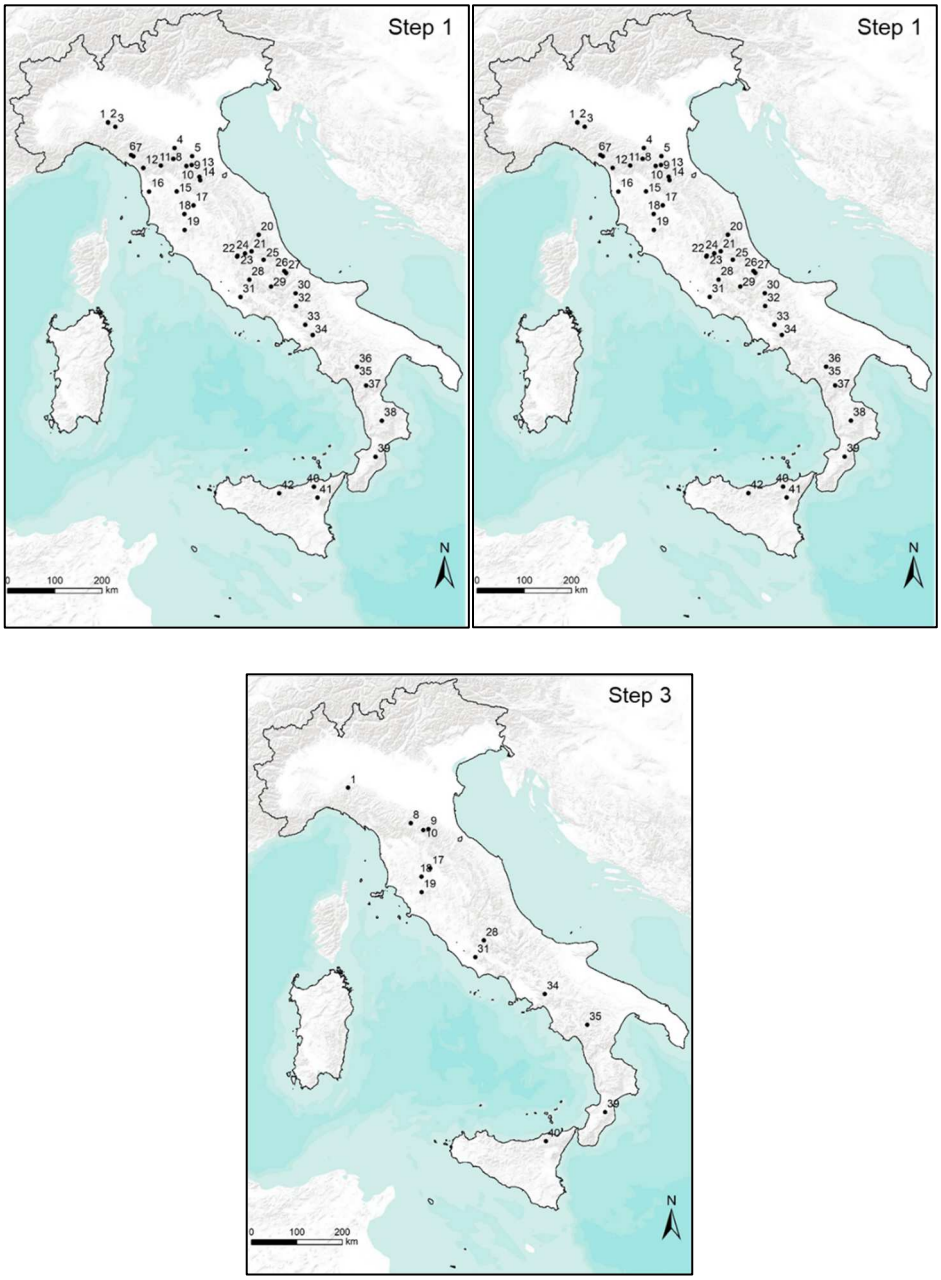

Figure 1 - Location of the selected study sites used for each of the three selection steps. Step 1 included studies with comparable descriptive information; Step 2 included studies reporting suitable site and anthropogenic data and/or providing LULC data; Step 3 included studies providing thematic maps suitable for landscape pattern analysis.

Study-site features

After further filtering, we reduced the list to 32 case studies (Step 2). We excluded four case studies (ID 23, 46, 47, 48) from the study-site description due to the lack of information but considered them for the land-use/cover transition analysis. Study areas were digitally scanned from article maps and then geo-referenced in a GIS environment. We drew a circular area around the centroid, approximating the extension of each study site and a buffer area to include the entire surface area. We extracted: i) zonal statistics referring to topographic, climatic and anthropogenic variables over the circular buffer : mean elevation and slope (DEM 20 m – ISPRA); ii) mean temperature and precipitation (WorldClim (Fick & Hijmans 2017); iii) mean density of gridded livestock (cattle, goats and sheep) from the Livestock Geo-Wiki database (Robinson et al. 2014); iv) population density in 1951 and 2011 (Population Census of National Institute of Statistics); v) mean road density (National Geoportal database); vi) road-distance median (Italian National Geoportal) for a homogeneous description of the study sites. Real livestock data was scarce so the current presence of livestock was detected using models (livestock raster grids). Population density is given by weighted mean of inhabitants/km^2 of all municipalities included in the buffer areas (both for 1951 and 2011) and the extension of each municipality. Road density was calculated by including both national and provincial roads in the buffer areas.

Land-use/Land cover transitions

The analysis of transitions required the available data to be standardized since it was too heterogeneous as it had been classified with different land-use categories and processed using various analytical approaches. We therefore standardized the processing methods to obtain a new set of homogeneous data suitable for making comparisons. Firstly, we merged similar LULC categories and obtained 8 homogeneous ones: forests (only broadleaf), shrublands, grasslands (dense and sparse grasslands, pastures and meadows), croplands (all arable lands), unvegetated lands (bare soils, water surfaces and rocks), urban (infrastructures in general: towns, houses, private gardens, roads and industrial plants), orchards (fruit tree plantations and groves) and plantations (generally conifer plantations). We tried to include all LULC transitions calculated in the selected studies into two-time intervals: "old" between 1948 and 1968 and "new" between 2000 and 2012. We calculated absolute (ha) and relative changes (%). We then focused on broadleaf forest cover to compare the dynamics of different study sites and the annual percentage rate of forest expansion was added to the general transition analysis.

Landscape pattern analysis

We selected 13 papers (step 3) and calculated common landscape metrics ex novo after some pre-processing due to the wide heterogeneity of methods and indices used in the single studies. We obtained digital images of all the maps included in the selected articles and performed an automatic segmentation with eCognition Developer 64 software (scale parameter = 100, color parameter = 5). Each segmented map was imported as vector data into a GIS environment and geo-referenced using ground control points (GCPs) from Bing-aerial web maps. The vector data was manually classified according to the eight land-use/cover categories identified for the transition analysis without altering the classification of the original sources. We ended up with two homogeneous geo-referenced maps (past and present) for each study site corresponding to the original maps from the relative papers.

Each map was rasterized into ASCII (American Standard Code for Information Interchange) format (20-m resolution). A total of 26 raster files (13 "old" and 13 "new") were imported into the Fragstat 4.2 software program (McGarigal & Marks 1994) and processed to calculate 23 landscape indices. We set the configuration of the moving window used for the metric computation applying the 8-cell neighborhood rule for all the 26 raster files, making results comparable (Zatelli et al. 2019). We selected five landscape metrics excluding those with high intercorrelation (Riitters et al. 1995): PAD patch density (number of patch/100 hectares), MPS mean patch size (hectares), MSI mean shape index, AGI aggregation index and SDI Simpson's diversity index. We explored landscape structure differences over time using a Wilcoxon paired test with the medians of index variations.

Results

Literature search and data extraction

We collected 51 papers (including 57 case studies) comprising 34 scientific articles (27 indexed and 7 non-indexed) and 17 other works (reports, theses, proceedings and book chapters) published between 1991 and 2016 (Table 1). Most of these works (78%) were published between 2005 and 2016 (Figure 2). 83% of the studies are at local scale and refer to single sites but with greatly varying sizes (from hundreds to thousands of hectares); 12% applied the LULCC analysis at regional scale (e.g. Brath et al. 2006) and only 5% at national scale, e.g. Falcucci et al. 2007 (Figure 3a). The study sites are mostly located in the Central (37.5%) and Southern (37.5%) Apennines, with 25.0% in the Northern Apennines (Figure 3b).

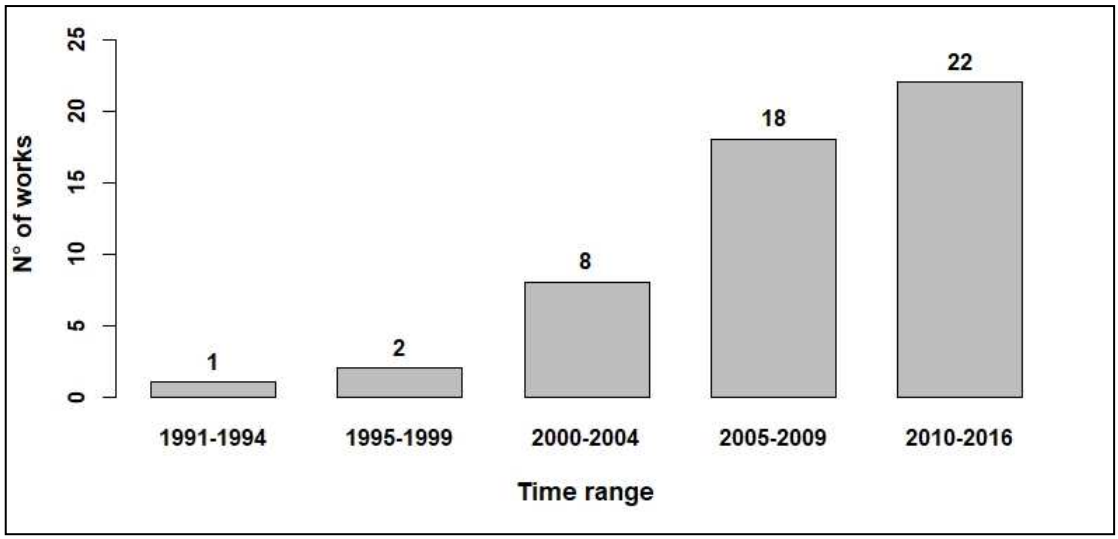

Figure 2 - National and international literature published between 1991 and 2016 on LULCC in the Apennines.

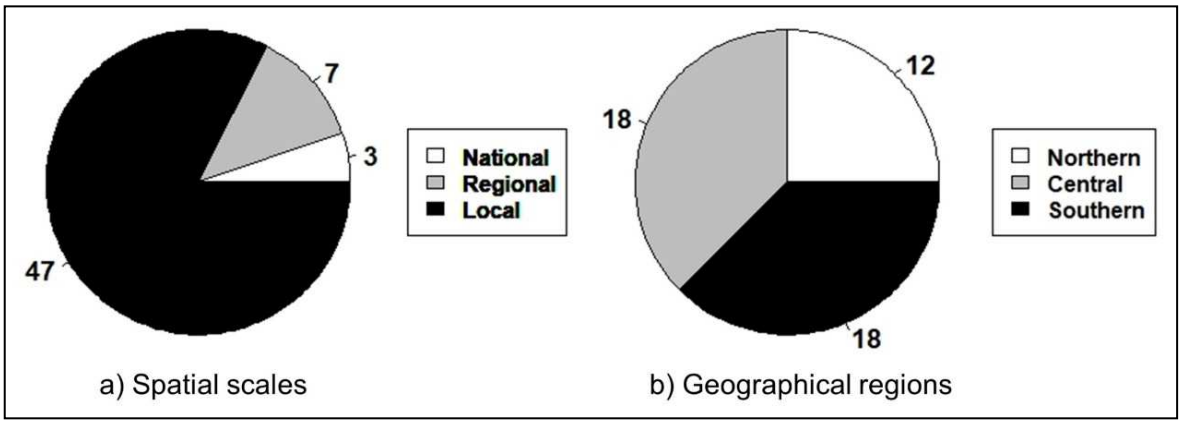

Figure 3 - (a) Breakdown of studies at national, regional and local scale; (b) number of case studies with available or inferable geographic locations in Italy.

Furthermore, 77% of the studies showed change detection, 40% carried out a landscape pattern analysis through landscape metrics but only 38% of the studies included both change detection and landscape metric analyses. Anthropogenic drivers such as population and grazing data were only considered in 7 studies, the mean temporal extent of the change detection was 46 years, and the most common spatial resolution was a minimum mapping unit (MMU) of 100 m^2. The studies including landscape metrics analysis used one to nine metrics. The variables used most frequently were the number of patches, mean patch size, Shannon's diversity index and the landscape shape index. The most common keywords were "land-cover change" (47%) followed by "forest cover dynamics" (15%) and "vegetation patterns" (11%).

After the first selection step, we retained 42 case studies (Figure 1), mainly (64%) located in the mountainous areas of the Apennines. Different data sources were used for the LULCC analyses: 66% used aerial photos, 57% theme-based maps, 20% historical maps and 19% satellite imagery. 30% of

the papers used DEMs, but only 7% added ground control points (GCPs). Temporal frequencies used for change detections varied from two to seven chrono-sequences: the most common years were 1954 (16 studies), 1960 (6); 1990 (6); 2000 (10). The extension of the study areas ranged from 143 to 130,200 ha, excluding a few at regional/national-scale. The LULC categories varied considerably in relation to local differences: eight on average and three of them specifically related to forest or woodland types.

Study-site features

We extracted topographic, climatic and anthropogenic variables from 28 case studies (Step 2) (Table S1). Mean elevation ranged largely from 56 m to 1442 m a.s.l. and mean slope was between 7° and 33°. The most common livestock was sheep, followed by cattle and goats. Human population density decreased from 1951 to 2011 in 71% of selected sites, with a mean change of -9 inhabitants/km^2 (± 26 SD).

Land-use/cover transitions

The most significant landscape change was the broadleaf forest (Fo) expansion, increasing from 36.5% to 54.4% of average cover (Figure 4). In the past, the minimum was 0% (SPA) and the maximum 71.1% (CSM). These values now range from 13% (LEA) to 85.1% (CDA). Average grasslands cover decreased from 21.9% to 12.4%. The highest value in the past was 74.6% (ROM) and is currently 41.4% (MOM). Cropland cover decreased from 20.9% to 13.2%, whereas shrub cover increased slightly from 7.4% to 7.8% (Figure 4). Unvegetated, urban, orchard and plantation classes are outliers due to the high variability. However, an average of the past and present percentages show that Un is practically unchanged (around 4%). Urban areas showed a remarkable increase, doubling from 0.7% to 1.7%, along with conifer plantations which increased from 1.9% to 3.5%. Orchards decreased from 6.3% to 4.0%. Orchard, shrubland and unvegetated land covers (ha) were not statistically different at the critical value of 0.05 (Wilcoxon paired test with medians of "old" vs "new" covers).

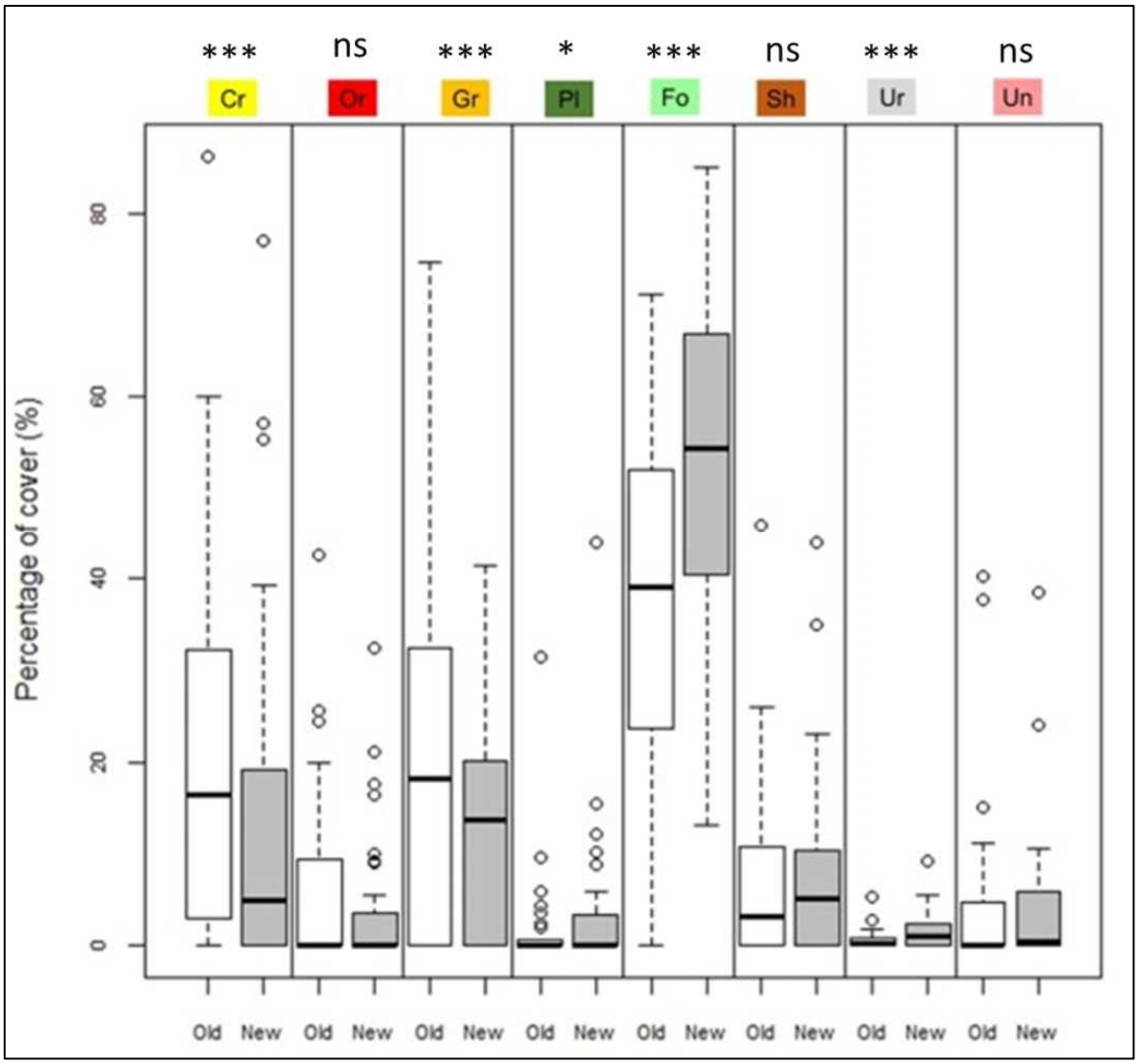

Figure 4 - Boxplot showing percentage of land-use/cover of each class in both time-periods of analysis (Old = White, New = Grey). Each boxplot comprises the cover data of the 32 case studies. Vertical lines separate each category. Labels at the top of the boxes refer to class names: Cr = Cropland (yellow), Or = Orchards (red), Gr = Grassland (orange), Pl = Plantation (dark green), Fo = Forest (green), Sh = Shrubland (brown), Ur = Urban (grey), Un= Unvegetated (pink). * = p-value < 0.05, ** = p-value < 0.01, *** = p-value < 0.001, ns = not significant (Wilcoxon paired test to compare "Old" and "New" covers for each category).

Considering the average relative change (averaged values of each study site) (Figure 5), Fo increased by 78.0 % (+7114 ha average, +18.6 ha min and +73,427 ha max). The rate of forest change varied from 0.16 %/year. (CSM) to 4.75 %/yr (CAR) with a mean value of 1.01 %/year. Grasslands and croplands decreased by 19.1% (-2982 ha) and 48.5% (-3621 ha) respectively. Shrublands showed very high variability in both directions but there was an overall increase of 125.4%. Urban cover tripled (+301.5%, +427 ha), whereas orchards decreased by 29.6% (-622 ha). Conifer plantations expanded by +47.9% (+507 ha) but only in eight sites.

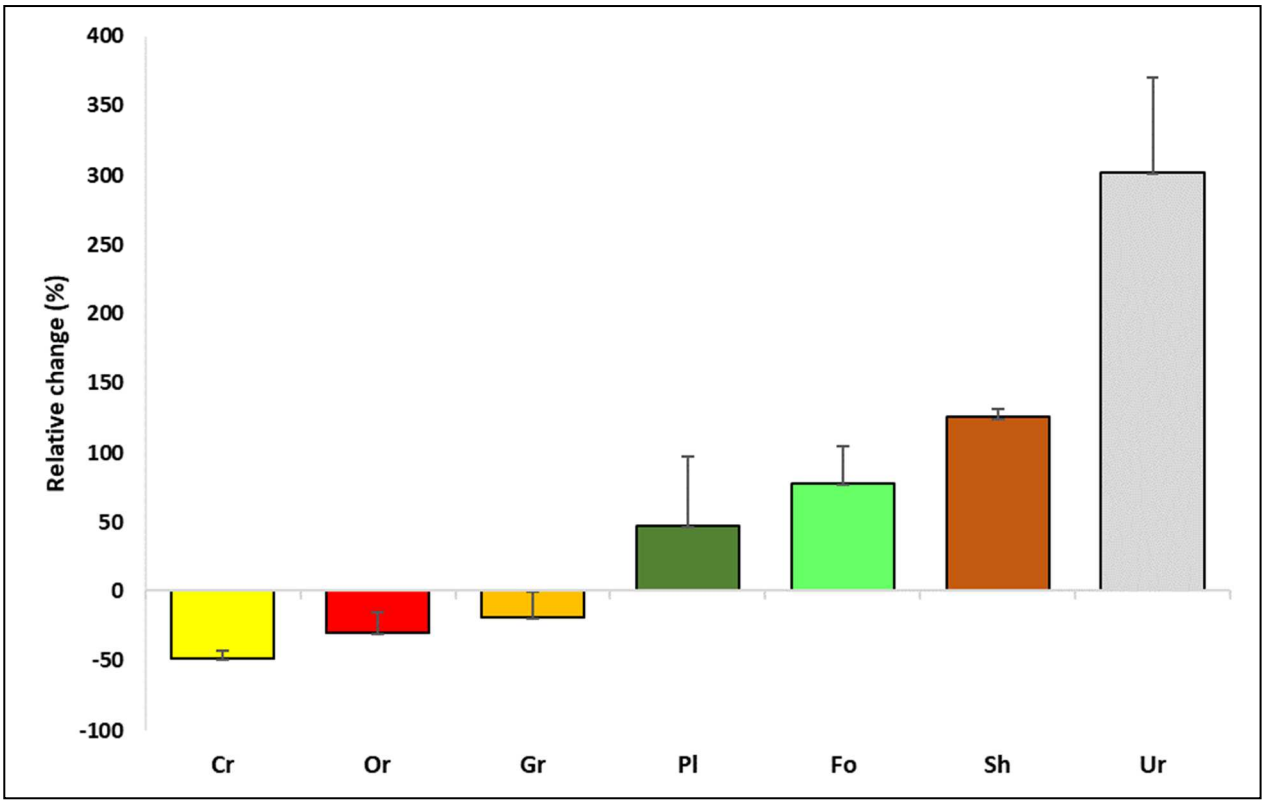

Figure 5 - Relative change (%) of LULC categories over time. Study site changes are averaged and plotted as bar plots with standard error whiskers. The unvegetated class was excluded as an outlier. Cr = Cropland (yellow), Or = Orchards (red), Gr = Grassland (orange), Pl = Plantation (dark green), Fo = Forest (green), Sh = Shrubland (brown), Ur = Urban (grey).

Landscape pattern analysis

Landscape patterns were analyzed on 13 sites (Step 3). No significant differences (Wilcoxon paired test) were observed at the critical p-value of 0.05 testing "old" vs "new" groups of metrics. Averaged PAD increased through time (from 37.2 to 43.9 patch/100 ha), whereas MPS and MSI decreased slightly. SDI remained stable over time and AGI increased slightly (Figure 6).

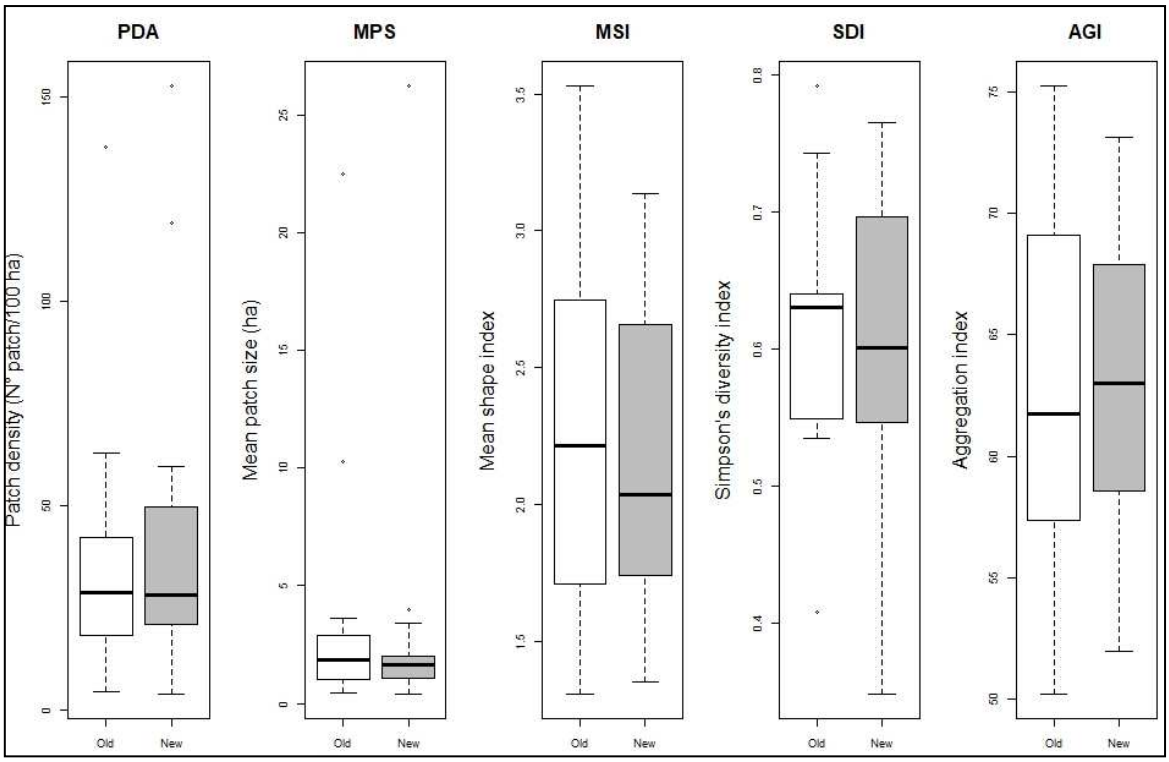

Figure 6 - Boxplot showing mean value of landscape pattern indices in the two time-intervals (white = past, grey = present). Each boxplot comprises values of metrics of the 13 case studies (Step 3). Horizontal lines are median values and circles are outliers. PAD = patch density (number of patches/100 hectares), MPS = mean patch size (hectares), MSI = mean shape index, SDI = Simpson's diversity index, AGI = aggregation index.

Discussion

The opportunities to measure and show LULCC are increasing rapidly thanks to developments in remote sensing platforms and sensors, GIS software and access to specific databases. This also applies to areas like the Apennines which have often played second fiddle to the Alps. We did not find any literature before 1991, but interest in this area has been increasing since natural reforestation in rural and mountain areas has become so important, requiring a review of existing land management policies and habitat conservation strategies. New forest expansion has significant implications in terms of wood and non-wood products, carbon sequestration, slope erosion control, biodiversity conservation, recreation opportunities and the value of ecosystem services in general (Vazquez-Quintero et al. 2016). Moreover, LULCC are closely linked to climate change affecting the extent, intensity and frequency of forest disturbances, such as wildfires (Pausas et al. 2012; Bebi et al. 2016).

The results of this study indicate increasing interest in LULCC analyses, especially in the Central Apennines (Argenti et al. 2006; Rocchini et al. 2006; Pelorosso 2007). These studies focused on both

forested areas and agro-ecosystems (cropland and orchard areas) in addition to more extensive vegetation areas (shrubs and grasslands). Research was carried out in both mountainous areas (65% of the cases) and at lower-elevation sites (< 600 m a.s.l.) (35%). Landscape metrics computation is usually necessary to comprehend change detection more easily, and the processes and patterns of landscape cover transitions. About 77% of the whole studies reviewed included the change-detection analysis but only 38% of them dealt with landscape pattern analysis. Since the first national planimetric and stereoscopic aerial photos coverage was carried on in 1954-1956 (G.A.I. flight), most of the studies dealt with those years (Bracchetti et al. 2012; La Mela Veca et al. 2016). Previous temporal studies depended on different sources such as local aerial photos, historical maps or local land registers (Pezzi et al. 2011).

The landscape investigated is mainly in mountain areas so that the non-forest/forest transition was the most significant ecological process covered in the material reviewed. Broadleaf forests (natural forest) (Fo) were found to have expanded by 78% compared to the 48% expansion of planted forests (Pl). These two forest categories are generally very distinct due to their different origins. Conifer forests largely comprise *Pinus* spp. Stands and were planted throughout the 1900s for erosion control of overgrazed, steep slopes and to give employment to (Falcucci et al. 2007; Vacchiano et al. 2016; Piermattei et al. 2016). Broadleaf forest cover increased after agro-pastoral land was abandoned, especially after World War II. Permanent or temporary population migration from the mountains towards coastal and urban areas was common in the Mediterranean basin, and continues to this day (Bakudila et al. 2015; Falcucci et al. 2008; Vitali et al. 2017). The shift from extensive to intensive agricultural systems caused widespread abandonment of crops and grasslands, especially around mountain settlements (Pelorosso et al. 2009; De Sillo et al. 2012; Campagnaro et al. 2017). A decrease in population density (-9 inhabitants/km^2) occurred between 1951-2011. Natural secondary successions are common when anthropogenic pressure is reduced (Gartzia et al. 2014) and occurred in former grasslands and croplands (Bracchetti et al. 2012; Torta 2004) triggering shrub (Assini et al. 2014) and tree species (Ruhl et al. 2005) encroachment. These processes were also observed in other Mediterranean mountain areas in Greece (Petanidou et al. 2008), France (Roura-Pascual et al. 2005) and Spain (Poyatos et al. 2003; Ameztegui et al. 2010; Gartzia et al. 2014). Our review confirmed that crops decreased by an overall average of 49%, pastures decreased by an overall average of 19% and shrub cover increased by 125%. Similar processes and trends occurred in the Alps even though the landscapes and plant species are different (Gellrich et al. 2007; Tappeiner et al. 2008). The recent decrease in livestock grazing in mountain areas and the subsequent abandonment of grasslands is widespread (Falcucci et al. 2007; Pelorosso et al. 2009). However, it is not easy to quantify this process since data is generally scarce. One of the few recorded analyses in the Central Apennines showed a

30.3% reduction in cows and Mediterranean buffalos and a 32.5% reduction in sheep and goats over a 40-year time span (Pelorosso et al. 2009). Our step 2 data showed that the average number of sheep (18.2 heads/km^2) is now higher. The invasive nature of natural forests at the expense of former crop areas and pastures contributed to the disappearance of "cultural landscapes" such as transhumance trails (De Aranzabal et al. 2008; Orlandi et al. 2016; Zagaria et al. 2017), shaped by the co-evolution of human activities with the ecosystems and biota over thousands of years (Debussche et al. 1999). Another significant change, although in relative terms, is the expansion of urban areas and infrastructure which has increased by 302% in the last 60-70 years. This process has a twofold explanation: one linked to the widespread dispersion of houses and infrastructures at all levels of elevation, and the second to the immediate detection of these elements due to the increasingly high quality of aerial images.

Further interpretation of these socio-environmental processes can be assumed from the changes in landscape patterns. The interpretation of landscape metrics can be challenging due to the numerous variables considered (e.g. type of site, land-use classes and area extension). Moreover, the literature (Step 3) showed that landscape pattern analyses are highly fragmented. In general, the Apennines landscape could be expected to have lost some heterogeneity due to less human pressure, resulting in a more cohesive structure than in the past (Peroni et al. 2000), however landscape indices calculated with standardized data and methods revealed non-significant differences between the past and the present landscape structures (Figure 6). The result showed a slight overall decrease in landscape diversity (SDI) and an increase in same-class patch aggregation (AGI) in a few articles that carried out accurate analyses (Geri et al. 2010; Bracchetti et al. 2012). However, the heterogeneous nature of the study sites, data sources (e.g. resolution), and methods used could have biased the analyses, which would suggest that further direct tests should be carried out. The overall simplification processes of the Apennines landscape as suggested by the literature does not exclude local increases in specific mosaic fragmentation due to initial forest and shrub encroachment in grasslands and unvegetated areas. Similar dynamics are also described in the inner valleys of the western and central Italian Alps, with an increase in patch density and a decrease in mean patch area in most sites, with a corresponding slight reduction of landscape diversity (Garbarino et al. 2013).

Conclusions

This meta-analysis aimed to check the state of the art of existing studies on LULCC in the Apennines and find possible common patterns of landscape transition. We reviewed national and international

literature available in different databases and tried to standardize published and non-published datasets to provide comparable results. Case studies were selected according to three hierarchical steps based on the type and availability of information. Case studies were carried out at various elevations along the Apennines, especially in Central Italy. Authors adopted different analysis methods, generally using aerial photos but also other remotely-sensed data. The main process detected was the natural expansion of broadleaf (natural) forest on former grasslands and croplands caused by significant socio-economic changes. We detected ongoing landscape simplification occurring in inner mountain areas, but further analysis is necessary to confirm the intensity and rate of this process. These types of reviews that combine studies on large geographic areas to detect multi-scale changes in human-shaped environments are helpful in finding trade-offs between LULCC dynamics (Munteanu et al. 2014). They also play a crucial role in the development of common management strategies and predicting future scenarios (Van Vliet et al. 2016).

Supplementary materials

Table S1 - Zonal statistics of 28 case studies computed on a circular areas buffer approximating the size of the entire study site. Topographic variables: ELEVation, SLOpe. Climatic variables: TEMPerature, PRECipitation. Anthropic variables: CATTLE, GOAT, SHEEP, POPulation density on 1951, POPulation density on 2011, Road DEnsity and Road Distance median.

Ca-St ID	SITE	ELEV [m a.s.l.]	SLO [°]	TEMP [°C]	PREC mm	CATTLE [head/km²]	GOAT [head/km²]	SHEEP [head/km²]	POP51 [inh/km²]	POP11 [inh/km²]	Road DE [km/ha]	RoadD median [N]
		DEM-ISPRA		WorldClime		Livestock Geo_wiki			ISTAT		National Geoportal	
1	ROM	348	7	15	651	14	1	110	19	17	No road	No road
2	OLT	919	15	9	1023	9	0	0	78	27	No road	No road
4	SRB	317	13	12	854	15	1	4	98	123	6.7	481
5	BTO	1169	18	9	847	11	1	23	54	42	5.1	849
8	MTV	722	23	10	954	4	0	1	60	33	10.8	297
9	LRB	581	22	11	903	5	1	4	25	12	4.2	796
10	MOS	706	31	10	977	3	1	3	32	13	No road	No road
11	CSM	942	24	9	906	3	2	12	95	56	7.4	660
12	CAR	566	32	11	902	1	3	2	87	41	5.3	577
13	SPA	867	33	9	907	0	0	0	51	28	No road	No road
16	PHI	56	7	15	844	2	1	22	104	83	10.1	444
17	GAR	500	14	12	778	3	0	1	104	96	19.6	179
18	SIP	331	9	13	735	4	0	34	68	60	8.1	601
19	PDO	884	14	11	626	7	0	14	34	17	No road	No road
20	ACQ	729	25	12	848	3	1	51	68	33	17.1	940
22	MIC	1126	30	9	835	2	3	2	23	11	3.2	1279
24	RIE	706	17	12	846	11	1	24	87	88	6.2	680
28	SIM	1169	20	9	833	6	1	7	56	41	3.7	1094

31	LEA	286	10	14	915	16	2	15	107	164	3.8	1031
32	MOM	1442	25	8	746	10	2	51	38	21	1.6	1612
33	TAB	736	20	12	717	18	10	38	146	128	5.7	640
34	CDA	577	15	13	731	5	1	12	317	383	9.5	439
36	AGV	805	12	12	751	20	7	40	51	40	6.1	626
38	SMR	1208	15	10	881	4	5	5	61	52	5	740
39	SSB	890	10	12	914	4	8	9	151	118	4.2	836
43	FCA	983	23	11	756	0	0	0	48	55	No road	No road
44	LEB	677	18	12	854	4	3	7	114	127	No road	No road
45	LEC	796	20	12	842	13	3	18	110	116	0.2	3483

CHAPTER 3

Patterns and drivers of forest landscape change in the Apennines (Italy)

This chapter has been revised and accepted for online publication on July 6th, 2019:

Malandra F.[1], Vitali A.[1], Urbinati C.[1], Weisberg Peter J.[2], Garbarino M.[3], 2019. Patterns and drivers of forest landscape change in the Apennines range, Italy. Regional Environmental Change. DOI: https://doi.org/10.1007/s10113-019-01531-6

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

Keywords: Land use change; Mountain forest landscape; Landscape mosaic; Reforestation; Apennines; Forest regeneration

Introduction

Land-use change in mountain landscapes

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

Human pressure and consequent landscape modifications

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

Farmland abandonment and forest expansion in the Apennines (Italy)

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

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

Materials and methods

Study areas

Our ten study areas are all located within the Apennines along 4.30° of latitude (about 660 km), extending from North-East (NE) to South-West (SW) between 38-45° N and 8-17° E. They encompass a region 1200 km in length and 40-200 km of width, from the Ligurian sea to the Calabrian tip. The study areas encompass several mountain peaks higher than 2000 m a.s.l., from Mt. Cimone (North) to Mt. Pollino (South) together with comparable land covers along the altitudinal gradient and suitable for a change-detection analysis. The highest elevation is Corno Grande (2914 m a.s.l.) of the Gran Sasso massif in the central Apennines (Figure 1). Most of the study areas are included in the European Union Natura 2000 network of protected sites: about 78.5% of the analyzed areas is in the European Union Natura 2000 network of protected sites. Mean annual temperatures range from 6.2 to 10.0 °C and annual precipitation ranges from 730 to 877 mm. NE slopes (Adriatic side) are in general more continental than SW slopes (Tyrrhenian side), whereas precipitation is greatest for NE slopes. At each study area, we analyzed two paired study landscapes (NE and SW aspect), each extending for 16 km^2. The 20 study landscapes cover a total surface of approximately 32,000 ha within an elevation range of 347-2500 m a.s.l., including all vegetation zones, from hilly (< 600-m a.s.l.) to alpine.

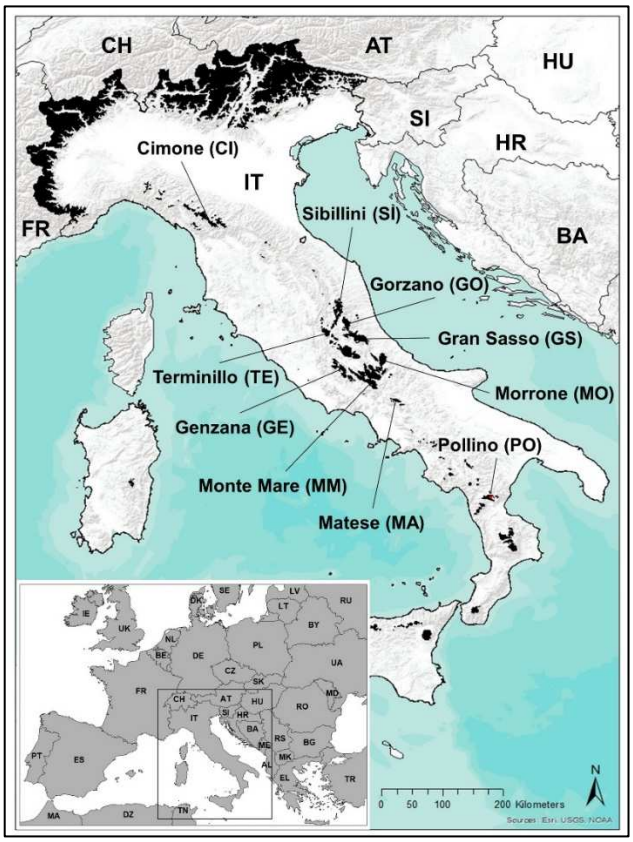

Figure 1 – Geographic distribution of the 10 study areas selected along the Apennines. Areas in black have elevation > 1500 m a.s.l.

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

Image analysis

We collected, processed and analyzed two types of aerial imagery: i) 1954-1955 flight aerial photos (b/w, 1 m cell size) from IGMI (Italian Geographic Military Institute) GAI (Italian Aerial Group); ii) 2010-2014 orthophotos from AGEA (National Agency for Funding in Agriculture) (RGB, 0.5 m cell size). For Mt. Pollino only, we processed 1948 IGMI b/w photos and 2003 AGEA orthophotos. Here we refer to 1954 for older aerial photos (1948, 1954, 1955) and to 2012 for newer ones (2003, 2010-2014). Several IGMI 1948 images were scanned at 1200 DPI, mosaicked and resampled at 1 m resolution. Mt. Pollino is a representative southern location, where peaks > 2000 m a.s.l. are very rare. Historical GAI aerial photos were orthorectified using the AGEA orthophotos and a 20-m resolution DTM (ISPRA – Italian Institute for Environmental Protection and Research) as reference data. We used PCI Geomatica 2012 software for geometric correction of historical images (mean RMSE overall = 23 m ± 2 SD; mean RMSE for Mt. Pollino = 82 m ± 2 SD). To facilitate the comparison between historical and recent aerial photographs, we resampled the higher resolution (0.5 m) AGEA images to 1 m as for the IGMI images. We applied a semi-automatic object-based classification by combining the automatic segmentation through eCognition software (scale factor 100, color factor 0.5) with on-screen photointerpretation of segmented polygons (Garbarino et al. 2013). For the 40 land-cover maps (20 landscapes x 2 time periods) each polygon was classified into 9 land-cover classes: bf (broadleaf forest), cf (conifer forest); sh (shrubland), dg (dense grassland dense), sg (sparse grassland), or (orchard, vineyards, other tree groves), cr (cropland, herbaceous crops in general), un (unvegetated, bare soil and water bodies), ur (urban, buildings and infrastructures).

The 40 land-cover maps (see examples in Figure S2, S3) were post-processed in ArcGIS 10.4 software so as to enforce consistency among the two datasets (Figure S1). This two-step process aimed for a minimum mapping unit (MMU) of 100 m^2. At first, the polygons with surface area < 100 m^2 were

merged with neighboring larger ones by using the ArcGIS tool "Eliminate". After a rasterization of vector data (1-m resolution) the raster maps were smoothed by using a moving-window (3 x 3) majority filter (Jensen et al. 2001). Overall classification accuracy (Figure S1 – table insertion) ranged from 70% (Morone SW 1954) to 96% (Gorzano NE 2012) with a K coefficient between 62% (Cimone SW 1954) and 92% (Gorzano NE 2012). For validation data, we randomized 100 points on each map and classified them visually using the same land-cover categories adopted in the automatic segmentation.

Data analysis

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

To analyze 1954-2012 landscape patterns of the 20 study landscapes, we computed suitable landscape and class metrics from each raster image using the FRAGSTATS 4 statistical package (McGarigal and Marks 1994). We set the configuration of the moving window used for the metric computation applying the 8-cell neighborhood rule for all raster files (Zatelli et al. 2019). We selected five metrics (patch density PD, patch area mean AREA_mn, mean shape index SHAPE_mn, contagion index CONTAG and Simpson's diversity index SIDI) for the analysis after excluding other metrics that were highly correlated (Pearson's $r > 0.8$) (Riitters et al. 1995) and ecologically redundant (Tischendorf 2001). We ordered our residual 36 study landscapes through multivariate ordination using principal components analysis (PCA) based on a main matrix of the five landscape metrics, indirectly related to a secondary matrix of environmental variables (elevation, slope, temperature and precipitation) and anthropogenic variables (population density and urban cover). PCA was performed with the statistical package PC-ORD 7. The statistical significance of the ordination analysis was tested using a Monte Carlo permutation method based on 10,000 runs with randomized data.

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

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

We explored the main drivers of broadleaf forest transitions by rescaling the spatial resolution of land-cover raster maps (1 m) to the minimum resolution of topographic variables (DEM 10 m TINITALY) (Tarquini et al. 2012). We then used in the analysis only rescaled pixels with a minimum cover threshold of 51% of a single dominant category. We limited the analysis to those categories more prone to a transition to broadleaf forest: sparse grassland (sg), dense grassland (dg), unvegetated land (un) and shrubland (sh). We built a transition map for each landscape and from the whole dataset we extracted only the pixels showing potential shift from non-forest to forest. For each transition (e.g. shrubland to forest), we calculated a Boolean map indicating transient pixels (1 = forest cover in 2012) and non-transient pixels (0 = non-forest cover in 2012). These binomial values were obtained by the response variable (reforestation) in the models. We fitted binomial generalized linear models (GLMs) to predict the transition to forest cover as a function of three topographic variables (elevation, slope, north-eastness index) and three land-cover variables (proximity to former forest, proximity to former cropland and proximity to former urban area). We ranked all the potential models according to the Akaike Information Criterion (AIC) and then selected the most parsimonious models showing the lowest AIC value (Burnham & Anderson 2002). We also used the Akaike weights (Wi) of each model to measure the conditional probability of the candidate model with the greatest empirical support. All GLMs were run with the R software (R Core Team 2018), using the 'glm' function of the package *stats*. We performed model selection using the *MuMIn* package (Bartón 2017). We checked for collinearity of predictors using the 'vif' function of package *rms*.

Results

Land use change and landscape features

Concerning land-cover transitions, 40.7% (> 13,000 ha) of the total surveyed area of the Apennines changed land-cover class (Figure 2 and table S2). Land-cover categories with net increases included broadleaf forests with the highest increment (4452 ha, +34%), conifer forests (1064 ha, +114%), shrubland (180 ha, +16%) and urban areas (109 ha, +46%). Negative transitions predominantly occurred in croplands (-2237 ha, -76%), orchards (-174 ha, -72%), dense grasslands (-2251 ha, -33%) and sparse grasslands (-1204 ha, -22%) (Figure 2 and table S2).

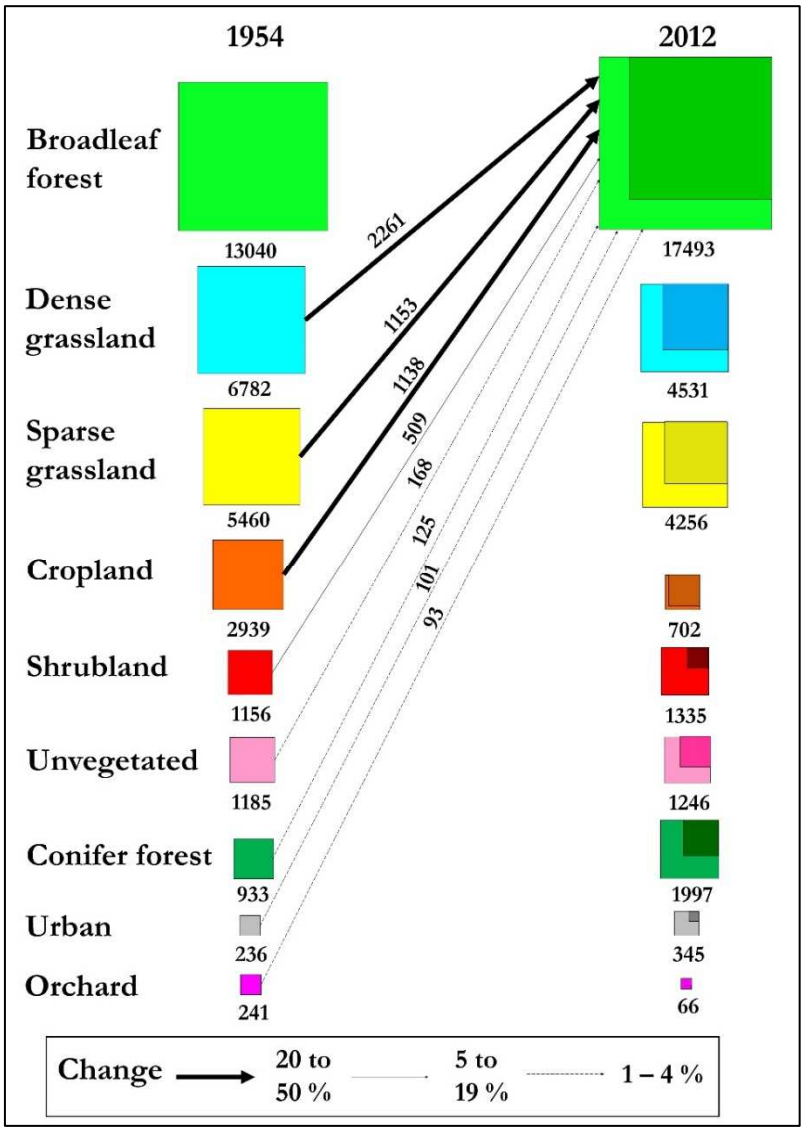

Figure 2 – Area of land-cover classes (ha), and land-cover transitions from past to present in the Apennines study sites. Light-colored boxes are size-scaled land-cover categories. Darker-colored inset boxes represent the relative unchanged surfaces (persistence) of each land-cover class over time. Transitions to broadleaf forests are

highlighted with arrows. Arrow thickness increases with magnitude of land-cover changes. The figures above the arrows are hectares of lands converted to broadleaf forests (modified from Cousins 2001).

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

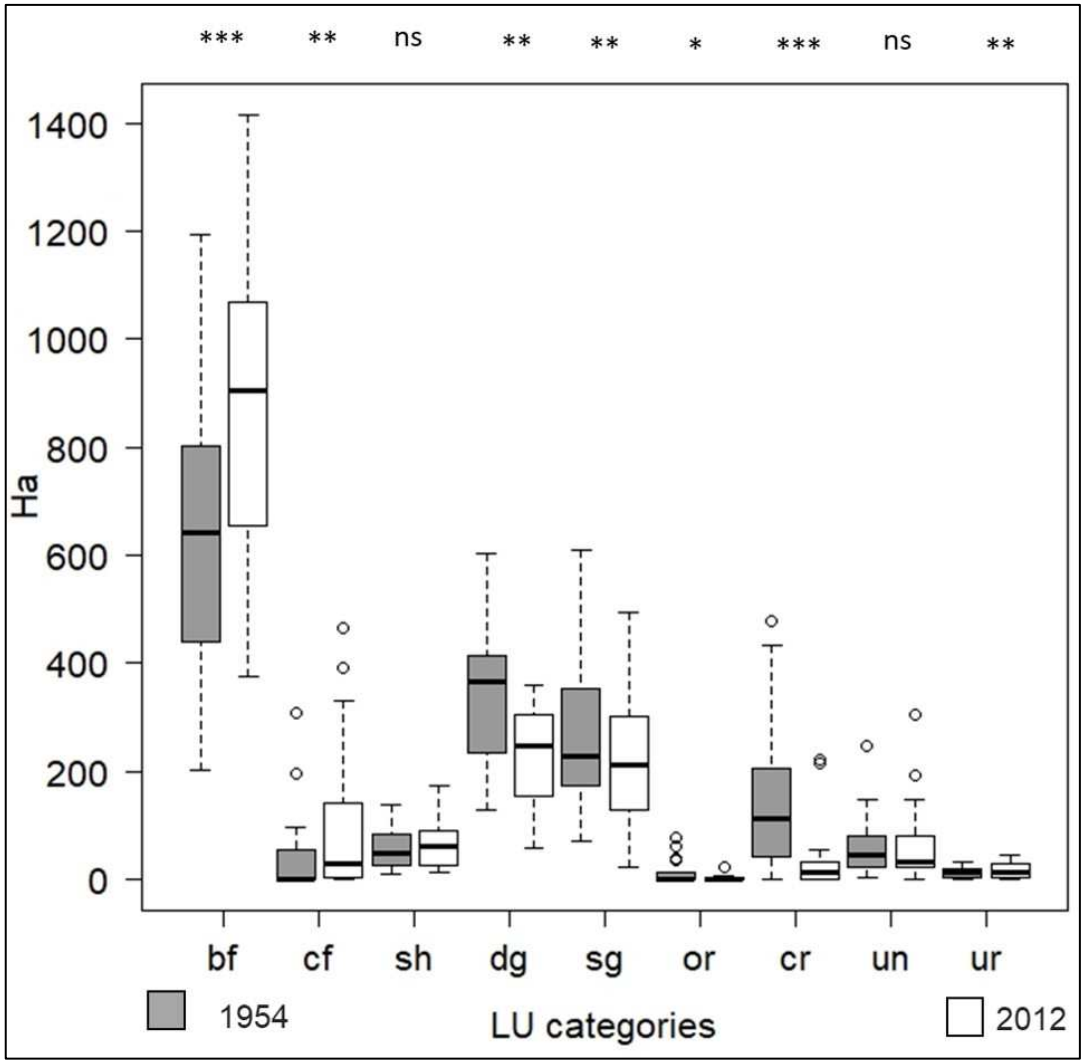

Figure 3 - Mean distribution of land-cover categories (hectares) for the two time periods (1954 and 2012) across 20 replicate study landscapes: broadleaf forest (bf), conifer forest (cf), shrubland (sh), dense grassland (dg),

sparse grassland (sg), orchard (or), cropland (cr), unvegetated (un), urban (ur). Horizontal lines are median values and circles are outliers. * = p-value < 0.05, ** = p-value < 0.01, *** = p-value < 0.001, ns = not significant (Wilcoxon paired test to compare 1954 and 2012 covers for each category).

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

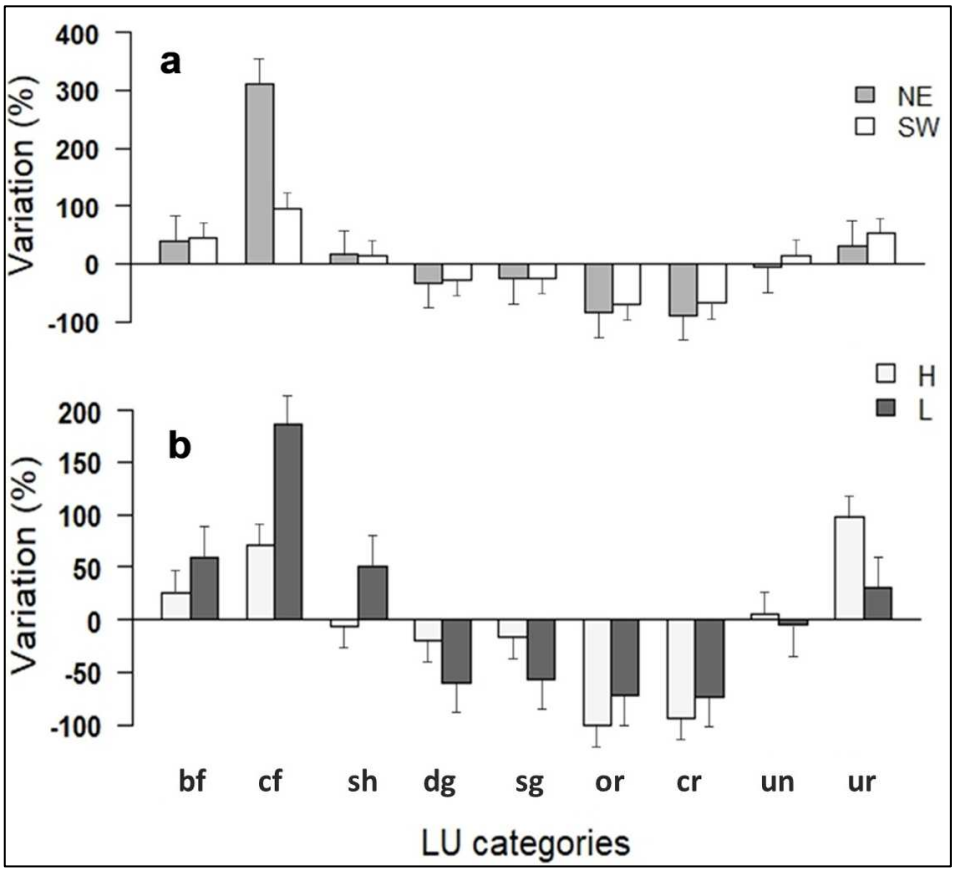

Figure 4 – Relative change (%) of land-cover categories in the 18 study landscapes: a) by main slope aspects (NE vs SW) and b) by elevation (H > 1300 m a.s.l. vs L < 1300 m a.s.l.). Error bars show standard errors. Broadleaf forest (bf), conifer forest (cf), shrubland (sh), dense grassland (dg), sparse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

Relative land cover changes varied significantly across an elevational threshold (above and below 1300 m a.s.l.) (Figure 4b). All forest types increased dramatically more at lower than higher elevations:

broadleaf 2832 ha, +59% vs. 1537 ha, +26% and conifer 719 ha, +186% vs 351 ha, +70%. Shrubland increased only at lower elevation (208 ha, +50%), but maintained similar cover values at higher elevations. A similar reduction trend occurred between dense grassland and sparse grassland at lower and higher elevations (dg = -59% and sg = -57% at lower elevation; dg = -21% and sg = -17% at higher elevation). Moreover, agricultural cover (crops) experienced a greater relative reduction at higher (around -359 ha, -93%) than at lower elevation (-1862 ha, -73%) (Table S5, S6).

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

Landscape pattern change

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

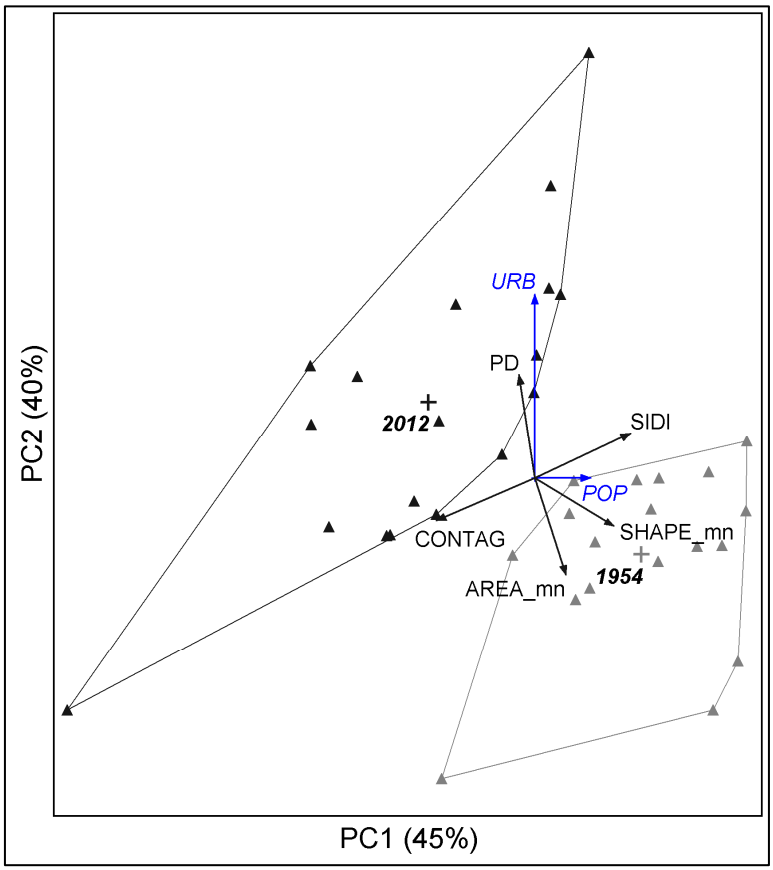

Figure 5 - Principal components analysis of the 36 Apennines forest landscapes covered with this study. Gray and black triangles are site scores in 1954 and 2012 landscapes, respectively and both included within convex hulls. Symbols (+) are centroids of convex hulls. Linear vectors indicate linear correlations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

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

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

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

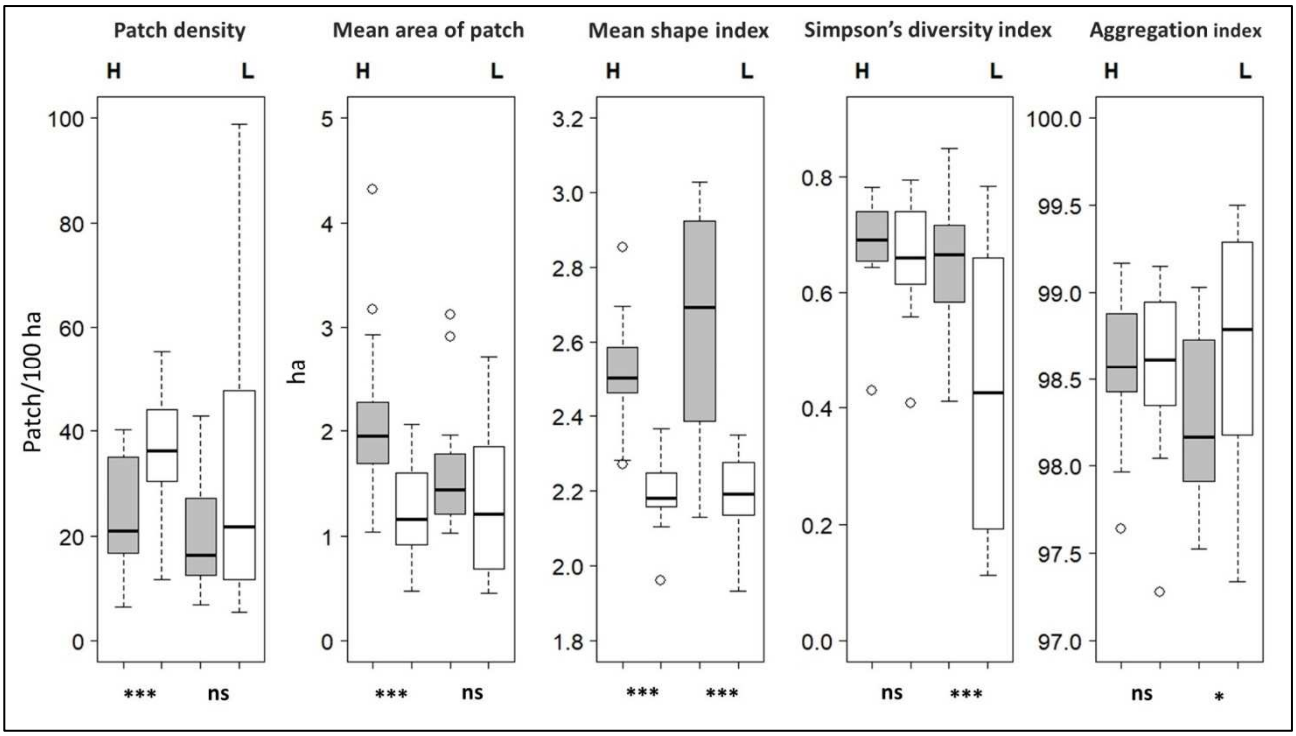

Figure 6 - Mean distribution of landscape metrics for the two time periods (1954 = gray boxes, 2012 = white boxes) and elevation level (H = High, L = Low) across the 18 study landscapes: patch density; mean patch area; mean shape index; Simpson's diversity index; aggregation index. Horizontal lines are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

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

Forest landscape change in broadleaf forests

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

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

Contribution to bf	Global * (%)	NE (%)	SW (%)	H (%)	L (%)
Conifer forest	2.3	0.5	4.1	4.0	1.0
Shrubland	9.2	11.3	7.1	11.5	8.0
Dense grassland	40.7	42.4	38.0	51.3	33.5
Sparse grassland	20.8	19.8	19.9	23.7	17.5
Orchard	1.7	0.5	3.3	0.1	2.9
Cropland	20.5	21.3	21.7	4.0	32.5
Unvegetated	3.0	2.8	3.4	4.5	2.2
Urban	1.8	1.5	2.4	0.9	2.4
Broadleaf forest mean annual absolute increment (ha/year)	75.4	41.7	33.7	26.5	48.8
Broadleaf forest mean annual relative increment (%/year)	0.6	0.6	0.6	0.4	0.8

The binomial GLM analysis highlighted that for the model that accounted for the transitions of all land-cover categories to broadleaf forest (all - bf) the best supported model included all six predictor variables (Table 2). New broadleaf forests expanded in proximity to former bf, at lower elevations, on steeper slopes and far from urban settlements. Other models, built on different transition types, showed that transitions to new broadleaf were primarily associated with proximity to old broadleaf forest, and secondarily associated with lower elevations. Transitions from both cropland and unvegetated lands

were positively associated with slope, the second most important variable in our models. We also observed a strong positive influence of distance from urban areas for transitions from both croplands and sparse grasslands to broadleaf forest. Transition from dense grassland to broadleaf forest occurred mainly on steeper slopes and closer to old croplands.

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

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

Discussion

LUCs are affecting forest cover dynamics worldwide with significant local differences. In some areas increasing farming and logging caused forest fragmentation and/or deforestation, whereas in many others the rural marginality determined opposite transitions, with secondary forests invading abandoned croplands and pastures (Rudel et al. 2005; Rey Benayas 2007). Following periods of extensive forest clearing to increase farming and livestock grazing, post-abandonment natural reforestation occurred in Mediterranean and temperate biomes of Europe and North America (Flinn and Vellend 2005). In some tropical areas, abandoned croplands are shifting to second growth forest over a longer time (Florentine and Westbrooke 2004). There are examples in Oceania (Endress and

Chinea 2019), Puerto Rico (Lugo and Helmer 2004) and eastern Africa (Chapman and Chapman 1999) and even in semi-arid regions of Argentina (Basualdo et al. 2018). In mountain areas of the Mediterranean basin land cover dynamics are faster as reported in the Alps (Tasser et al. 2005; Niedrist et al. 2009), the Carpathians (Kuemmerle et al. 2009; Weisberg et al. 2013), the Pyrenees (Metailiè and Paegelow 2005; Roura-Pascal et al. 2005), in Greece (Petanidou et al. 2008) and Spain (De Aranzabal et al. 2018). Similarly, the Apennines have experienced dramatic land-cover change and forest expansion dynamics over a 60-year period (1954-2012) (Malandra et al. 2018), affecting almost half the total land surface over a broad elevation range. Unlike other studies in this region (e.g. Benini et al. 2010), we have extended the analysis to the entire Apennine range, selecting study landscapes around the most important mountain groups (> 2000 m a.s.l.). The standardized protocol for image processing of aerial photography enhanced the output precision, confirmed by the high validation scores. We also diversified the analysis according to slope exposure (NE vs SW) and elevation (>/< 1300 m a.s.l.) to assess relationships between land-cover change and forest-dynamics with potential orographic drivers (Améztegui et al. 2010). An additional focus on the land-cover dynamics of broadleaved forests helped to further develop inferences about drivers of land-cover change.

Land use change and topographic factors

The overall forest cover increase, regardless of the scale of analysis, is very close to the 35-48% forest cover increased reported for the Apennines by more local studies (Rocchini et al. 2006). The average annual forest expansion rate (0.5 %/year) is also quite similar to that found by other authors (0.4-0.7%/year) in different sectors of the Apennines (Bracchetti et al. 2012). In general, shrubland is expected to be the most dynamic land-cover class (Gartzia et al. 2014) and to expand considerably after the withdrawal of agro-pastoral management. However, on the studied landscapes, the observed increase in shrubland cover was relatively moderate. This could be the balanced result of two land-cover transitions occurring simultaneously: one from existing shrublands to broadleaf forest and the other from existing grasslands to shrublands (Malavasi et al. 2018). Additionally, another possible reason might be the direct transition from grassland to forest. The loss of grasslands in the Apennines is an evident landscape process of recent decades: livestock grazing declines in mountain regions of central Italy between 1961 and 2000 were estimated at approximately 30% for cattle and 33% for sheep and goats (Pelorosso et al. 2009), although reliable data on pastoralism are often scarce or incomplete (Falcucci et al. 2007). The abandonment of grasslands and croplands largely influenced the observed land-cover transitions, with notable differences for aspect and elevation. The more favorable topography and climate conditions of the Apennine SW slopes favored farming and livestock grazing. The greater human pressure induced more relevant land-cover changes following land abandonment (Vitali et al. 2017). On these slopes, where the population density is higher than on NE ones, grasslands

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

Landscape mosaic shift driven by land abandonment

We observed dramatic changes in landscape mosaic structure occurring over the 60-year period, suggesting a shift mostly driven by patch shape simplification and patch density increase. However, there were contrasting trends of landscape configurational changes between lower and higher elevations. Bracchetti et al. (2012) reported a more homogeneous landscape matrix in the Central Apennines, with decreases over time in shape and diversity indices. Similarly, our results suggested an overall simplification of the landscape mosaic (Geri et al. 2010a), mostly at lower elevations. At lower-elevation, abandonment of farming and grazing activities followed by natural forest infilling caused a more homogeneous landscape mosaic. Woody species encroachment in former grasslands is likely to be driving local fragmentation at higher elevations and throughout the region. The forest recolonization of grassland-ecotones at high elevation, and the in-filling of open areas and forest gaps at low elevation, have both led to an increase of forest patch size through time. This process was globally described in a review paper, summarizing changes in landscape metric behavior in rural mountain and hill landscapes after abandonment processes (Sitzia et al. 2010). Common trends of mean patch area increase were detected although changes in patch density were inconsistent across studies. Even in the Apennines, similar processes have been discussed (Assini et al. 2014). In the

Central Apennines, Bracchetti et al. (2012) detected an increasing mean patch area and a decreasing density of woodland patches, rapidly merging into fewer larger patches. They found that this coalescence after tree colonization and woodland expansion is a very fast process.

Driving forces of secondary succession

The processes of broadleaf forest expansion were altitude-dependent. At high-elevation, secondary forests mostly derive from former grasslands; whereas, at low-elevation, contributions to secondary forests were distributed among different land cover classes. This is likely due to the land-cover composition in 1954. Topographic variables, such as slope aspect, have strongly conditioned land and forest use in the Apennines (Vitali et al. 2017). The large-scale removal of forests on SW slopes, occurred in ancient times, today provides the greater potential for forest expansion after abandonment. In Europe, a rural depopulation of 17% between 1961 and 2010 (FAOSTAT 2010) induced extensive land abandonment and forest expansion. In the Apennine municipalities comprising our studied landscapes, the national census data reported a relevant population decrease. This process however exhibited differences according the two main slope aspects and elevation zones, with clear effects in forest cover transitions. Attempts to correlate forest increase to population change have not always been successful, given the geographic scale of analysis and the lack of appropriate demographic records (e.g. number of active farmers or forest workers) (Vitali et al. 2017). However, our study encompassing the entire Apennines range shows a strong negative correlation between the population of mountain municipalities and forest cover. All GLM models identified the distance from existing broadleaf forest as an important proximate driver of forest expansion (Abadie et al. 2017). This derives from the species capacity of propagule dispersal which usually occurs in the vicinity to seed sources (Nathan & Muller-Landau 2000). Similar influences of proximity to pre-existing forests have been found by several authors in the central Pyrenees (e.g. Gartzia et al. 2014). Grassland to forest transitions occurred farther from existing settlements. Anthropogenic variables including distance to old cropland and urban areas can negatively influence reforestation, since shrubland transition to forest often occurs in long-abandoned areas. In the Apennines these anthropogenic variables are often more relevant drivers of transitions from shrubland to broadleaf forest than physiographic variables such as slope angle and aspect.

Conclusion

The main goal of this work was to develop a generalizable model of Apennine landscapes changes at the regional scale, through targeted sampling of replicate study landscapes within key environmental

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

Supplementary materials

Table S1 – Main environmental features of the 20 study landscapes. Topographic (ELEVation and SLOPe) and climatic (TEMPerature and PRECipitation) variables are averaged for each landscape. Dominant lithology is expressed as percentage of the following types: A: limestone; B: sandstone; C: marl; D: alluvial deposit; E: dolomite. Broadleaf forest cover is averaged at the most recent year (2012), except for the Pollino landscape (2003). New images: 2003, 2010-2014.

Site Name	Site Code	Lat [°]	Long [°]	ELEV [m a.s.l.]	SLOP [°]	Lithology [%]	TEMP [°]	PREC [mm]	Broadleaf Forest Cover [%]
Data sources	►	WGS84	WGS84	DEM20 ISPRA	DEM20 ISPRA	Italian Geological Service	WorldClim 2.0	WorldClim 2.0	AGEA New images
Cimone NE	Cine	44.22	10.70	1448.7	17.9	B: 92%	6.2	866.5	67.2
Cimone SW	Cisw	44.15	10.68	1427.9	19.0	B: 90%	6.3	866.1	53.6
Sibillini SW	Sisw	42.91	13.18	1400.1	26.0	A: 100%	7.5	877.8	40.4
Sibillini NE	SIne	42.81	13.30	1343.5	24.7	B: 64%; A: 36%	8.3	871.5	39.1
Gorzano NE	Gone	42.64	13.44	1523.9	25.8	B: 100%	6.8	860.6	67.0
Gorzano SW	Gosw	42.62	13.36	1528.1	29.0	B: 50%; C: 50%	6.9	857.0	48.6
Terminillo NE	Tene	42.48	13.04	1420.0	31.8	A: 71%; C: 29%	7.1	821.9	62.5
Terminillo SW	Tesw	42.46	12.98	1574.5	25.1	A: 100%	6.7	813.0	61.4
Gran Sasso NE	Gsne	42.44	13.52	1445.8	22.7	C: 39%; B: 33%	8.0	854.6	52.6
Gran Sasso SW	GSsw	42.50	13.52	1581.7	27.2	A: 56%; D: 36%	6.7	856.8	23.4
Morrone NE	Mone	42.13	13.97	1423.5	22.4	A: 56%; D: 28%	7.8	829.1	39.2
Morrone SW	Mosw	42.10	13.95	1124.0	25.4	A: 91%	9.2	831.3	35.4
Genzana NE	Gene	41.93	13.93	1537.9	22.8	A: 100%	7.1	813.2	60.9
Genzana SW	Gesw	41.93	13.87	1235.7	22.2	A: 73%; D: 16%	8.9	820.6	41.9

Monte Mare NE	Mmne	41.65	14.02	1148.2	23.9	E: 35%; A: 32%	9.5	793.4	88.5
Monte Mare SW	MMsw	41.62	13.98	1363.5	20.9	A: 70%; B: 11%	8.3	788.7	59.5
Matese NE	Mane	41.47	14.39	1116.1	21.7	A: 96%	10.0	730.8	66.7
Matese SW	Masw	41.44	14.35	1403.1	23.5	A: 100%	8.3	739.4	41.4
Pollino W	Pow	39.93	16.15	1638.3	23.1	A: 100%	7.6	851.3	75.2
Pollino E	Poe	39.92	16.21	1799.0	16.8	A: 100%	6.5	856.0	69.9

Table S2 – General transition (ha) matrix for land-cover changes over a 60-year (1954-2012), aggregated across 20 replicate landscapes. Land-cover classes are expressed with ID and abbreviation: cr (cropland, herbaceous crops in general), bf (broadleaf forest), cf (conifer forest), ur (urban, buildings and infrastructures), dg (dense grassland), un (unvegetated, bare soil and water bodies), or (orchard, vineyards, other tree groves), sh (shrubland), sg (sparse grassland).

LU classes ID	LU classes	1 cr	2 bf	3 cf	4 ur	5 dg	6 un	7 or	8 sh	9 sg	Tot 1954
1	cr	577	1138	176	102	370	43	22	331	180	2939
2	bf	33	11943	267	59	292	72	7	88	279	13040
3	cf	0	125	742	6	17	5	0	11	27	933
4	ur	13	101	19	54	28	4	2	8	7	236
5	dg	18	2261	323	64	2544	179	5	305	1083	6782
6	un	1	168	33	4	127	547	0	26	278	1185
7	or	57	93	2	21	18	2	30	12	5	241
8	sh	1	509	122	6	112	29	1	250	127	1156
9	sg	2	1153	313	28	1023	365	0	305	2271	5460
Tot 2012		702	17493	1997	345	4531	1246	66	1335	4256	**31973**

Table S3 – General transition matrix for land-cover changes (in hectares) occurred from 1954 to 2012, aggregated across 18 replicate landscapes with NE slope aspect. Mt. Pollino landscapes were excluded from this analysis.

LU classes ID	LU classes	1 cr	2 bf	3 cf	4 ur	5 dg	6 un	7 or	8 sh	9 sg	Tot 1954
1	cr	79	603	31	20	181	3	2	174	31	1124
2	bf	10	5862	63	24	134	34	1	57	90	6274
3	cf	0	14	89	1	0	1	0	1	0	105
4	ur	3	41	7	13	14	1	0	5	1	85
5	dg	4	1201	142	35	1388	123	0	173	498	3565
6	un	0	79	10	2	94	295	0	7	165	653
7	or	4	13	0	2	4	0	1	2	0	27
8	sh	0	319	9	4	45	10	0	104	39	530
9	sg	1	561	81	10	453	151	0	95	681	2032
Tot 2012		102	8694	431	111	2313	618	4	617	1505	**14395**

Table S4 – General transition matrix for land-cover changes (hectares) occurred from 1954 to 2012, aggregated across 18 replicate landscapes with SW slope aspect. Mt. Pollino landscapes were excluded from this analysis.

LU classes ID	LU classes	1 cr	2 bf	3 cf	4 ur	5 dg	6 un	7 or	8 sh	9 sg	Tot 1954
1	**cr**	498	532	146	82	188	40	19	151	148	1804
2	**bf**	23	4034	198	34	114	21	6	20	73	4523
3	**cf**	0	101	645	6	15	1	0	7	8	782
4	**ur**	9	59	12	41	14	3	2	3	6	150
5	**dg**	14	928	171	28	958	51	5	121	410	2686
6	**un**	1	83	22	3	33	249	0	19	107	518
7	**or**	53	80	2	19	14	2	29	10	5	214
8	**sh**	1	175	111	2	63	18	1	141	79	590
9	**sg**	1	486	223	17	529	208	0	194	1451	3110
Tot 2012		600	6478	1530	232	1929	593	62	666	2288	**14377**

Table S5 – General transition matrix for land-cover changes (hectares) occurred from 1954 to 2012, aggregated across 18 replicate landscapes at HIGH elevations (> 1300 m a.s.l.). Mt. Pollino landscapes are excluded from analysis.

LU classes ID	LU classes	1 cr	2 bf	3 cf	4 ur	5 dg	6 un	7 or	8 sh	9 sg	Tot 1954
1	cr	16	81	62	7	86	3	0	74	54	384
2	bf	5	5475	96	27	184	34	0	34	114	5968
3	cf	0	81	391	5	11	1	0	6	6	501
4	ur	0	18	7	17	15	1	0	1	2	63
5	dg	4	1042	115	44	2072	166	0	141	813	4397
6	un	0	91	8	2	123	449	0	22	273	968
7	or	0	2	0	0	0	0	0	0	0	2
8	sh	0	234	33	4	98	25	0	204	106	703
9	sg	0	481	141	18	902	335	0	177	1947	4003
Tot 2012		25	7505	853	125	3491	1015	0	658	3315	**16988**

Table S6 – General transition (ha) matrix for land-cover changes over a 60-year time period, aggregated across 18 replicate landscapes at LOW elevations (< 1300 m a.s.l.). Pollino landscapes were excluded from this analysis.

LU classes ID	LU classes	1 cr	2 bf	3 cf	4 ur	5 dg	6 un	7 or	8 sh	9 sg	Tot 1954
1	cr	561	1053	115	95	283	40	22	251	124	2545
2	bf	28	4420	165	32	64	21	7	43	50	4830
3	cf	0	33	342	1	4	0	0	2	2	385
4	ur	13	77	12	37	13	3	2	7	5	167
5	dg	15	1087	198	18	274	8	5	152	96	1854
6	un	1	72	20	2	5	95	0	5	8	207
7	or	57	93	2	21	18	2	30	12	5	241
8	sh	1	260	88	2	10	3	1	42	11	417
9	sg	2	566	163	9	80	24	0	111	184	1139
Tot 2012		678	7663	1104	217	751	196	66	625	486	**11785**

Table S7 – Land-use transitions in hectares (and %) occurred from 1954 to 2012 at each landscape.

Var	cr Ha (%)	bf Ha (%)	cf Ha (%)	ur Ha (%)	dg Ha (%)	un Ha (%)	or Ha (%)	sh Ha (%)	sg Ha (%)
Cine	-155(-80)	265(32)	48(558)	9(55)	-89(-22)	-25(-43)	0(-)	-9(-29)	-49(-70)
Cisw	-56(-51)	249(41)	82(26)	4(21)	-132(-40)	-13(-32)	-1(-54)	-61(-75)	-76(-73)
Gene	-109(-100)	230(30)	13(-)	-1(-4)	-108(-28)	-5(-14)	0(-)	-4(-3)	-21(-11)
Gesw	-420(-97)	248(59)	83(431)	15(50)	-83(-42)	45(31)	-40(-66)	73(144)	74(31)
Gone	-14(-100)	299(38)	0(-)	3(341)	-56(-14)	-19(-38)	0(-)	-64(-50)	-152(-69)
Gosw	-151(-86)	319(69)	49(-)	0(2)	106(46)	-27(-94)	-1(-45)	-35(-40)	-265(-44)
Gsne	0(-)	163(24)	0(-)	0(-)	-244(-60)	56(23)	0(-)	-5(-5)	27(15)
GSsw	-263(-55)	130(53)	116(181)	24(171)	-128(-69)	63(75)	-73(-94)	132(310)	-5(-2)
Mane	-203(-94)	278(35)	27(-)	21(95)	-111(-32)	11(136)	-7(-68)	44(169)	-64(-37)
Masw	-106(-33)	157(31)	0(-)	5(81)	-36(-12)	11(15)	1(165)	-26(-56)	-10(-3)
Mmne	-129(-97)	256(22)	3(-)	-1(-10)	-74(-47)	4(64)	-13(-98)	-25(-66)	-27(-32)
MMsw	-100(-86)	376(65)	0(-)	0(6)	-206(-45)	-2(-4)	1(-)	34(154)	-107(-30)
Mone	-252(-96)	427(211)	0(-)	-8(-95)	-125(-28)	-13(-15)	0(-)	61(580)	-93(-16)
Mosw	-51(-55)	213(60)	271(138)	8(54)	-186(-50)	-24(-49)	-14(-37)	-53(-38)	-169(-49)
Slne	-89(-81)	297(90)	173(180)	-5(-21)	-316(-53)	-48(-34)	-4(-89)	112(438)	-123(-46)
Sisw	-63(-90)	110(20)	135(68)	16(117)	30(23)	13(58)	-29(-80)	24(38)	-240(-46)

Tene	-75(-87)	201(25)	58(-)	3(418)	-134(-35)	0(0)	0(-)	-28(-55)	-29(-11)
Tesw	1(-)	147(17)	10(-)	6(24)	-126(-27)	5(18)	0(-)	-17(-30)	-29(-17)
Poe	0(-)	69(6)	-17(-36)	0(1370)	-142(-49)	23(219)	0(-)	4(22)	60(33)
Pow	-12(-100)	8(0)	6(-)	2(669)	-101(-43)	-3(-62)	0(-)	12(75)	84(61)

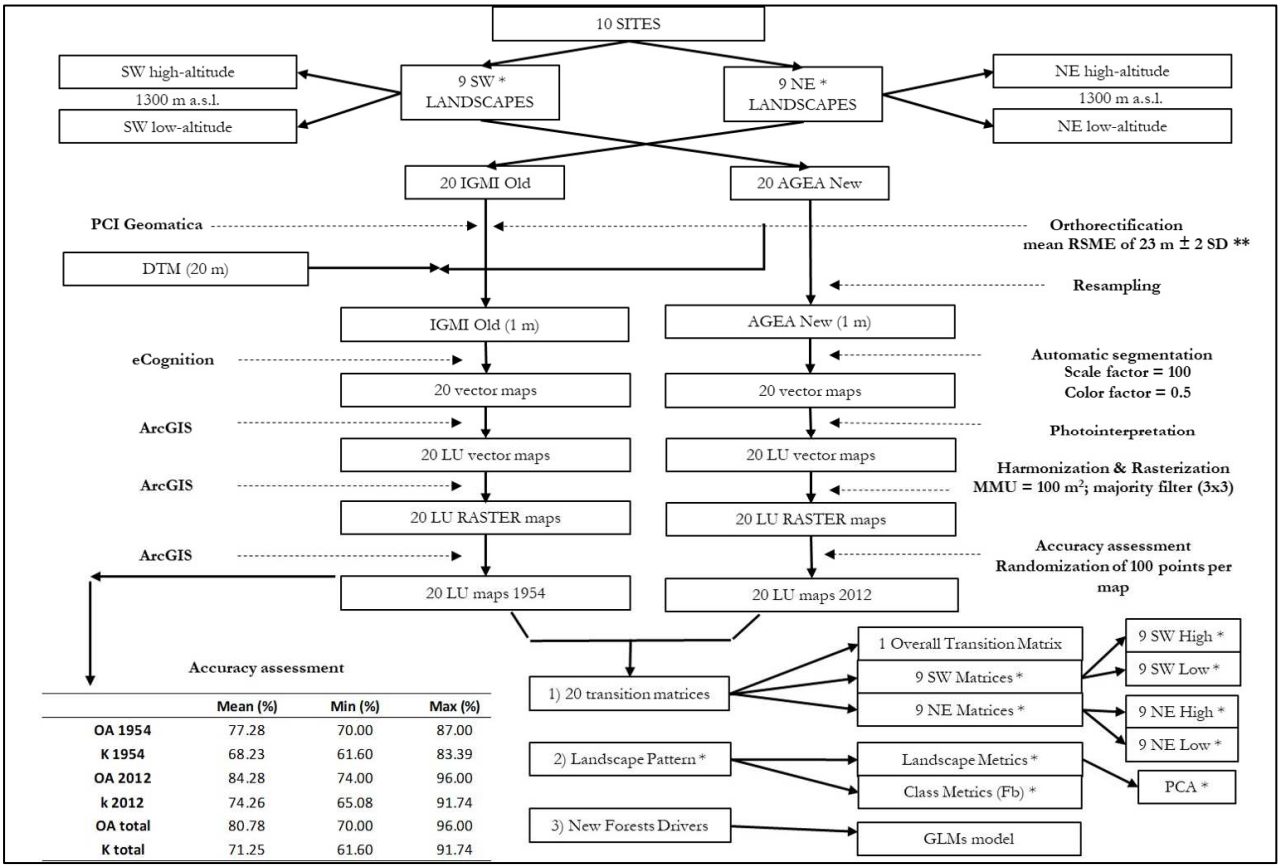

Figure S1 – Workflow of image processing for the land cover change analysis in the Apennines. In the accuracy assessment table, acronyms refer to: overall accuracy (OA); kappa coefficient (K). * = M. Pollino landscapes were excluded from analysis. ** = mean RMSE of 82 m ± 2 SD for the M. Pollino landscape orthorectification.

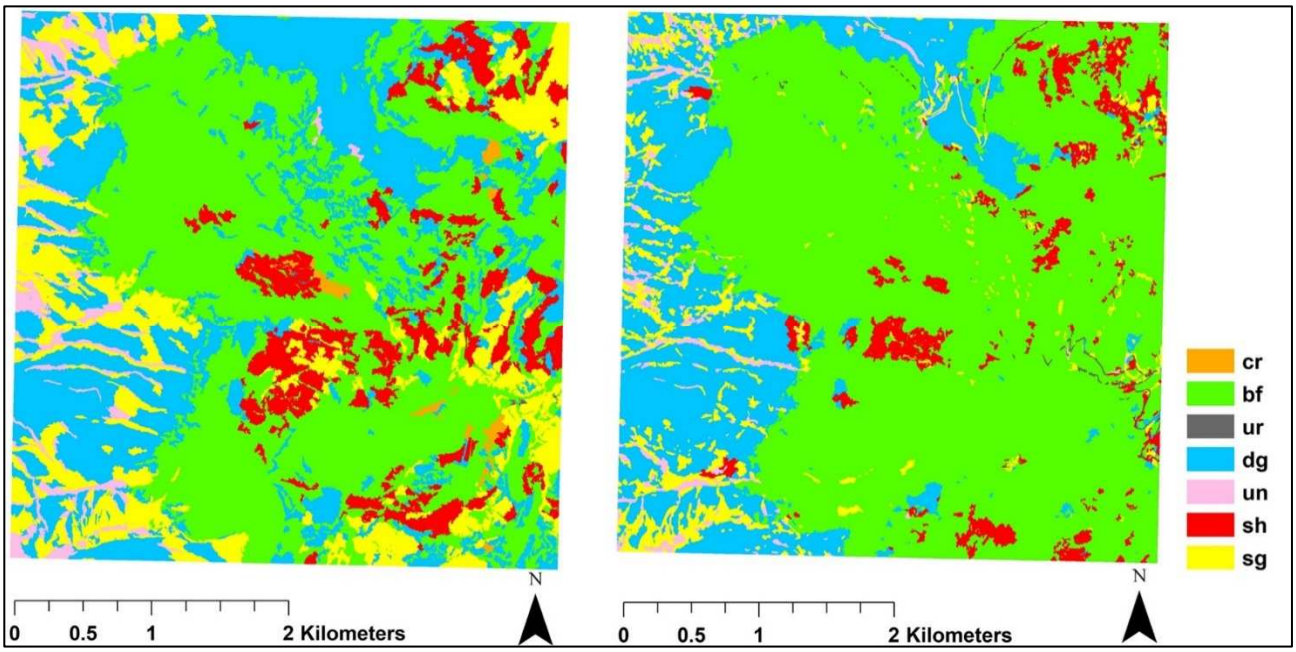

Figure S2 – Example of thematic maps of land cover (1954 and 2012) for Gorzano North-East landscape (Gone). Land-cover classes are expressed with abbreviation: cr (cropland, herbaceous crops in general), bf (broadleaf forest), cf (conifer forest), ur (urban, buildings and infrastructures), dg (dense grassland), un (unvegetated, bare soil and water bodies), or (orchard, vineyards, other tree groves), sh (shrubland), sg (sparse grassland).

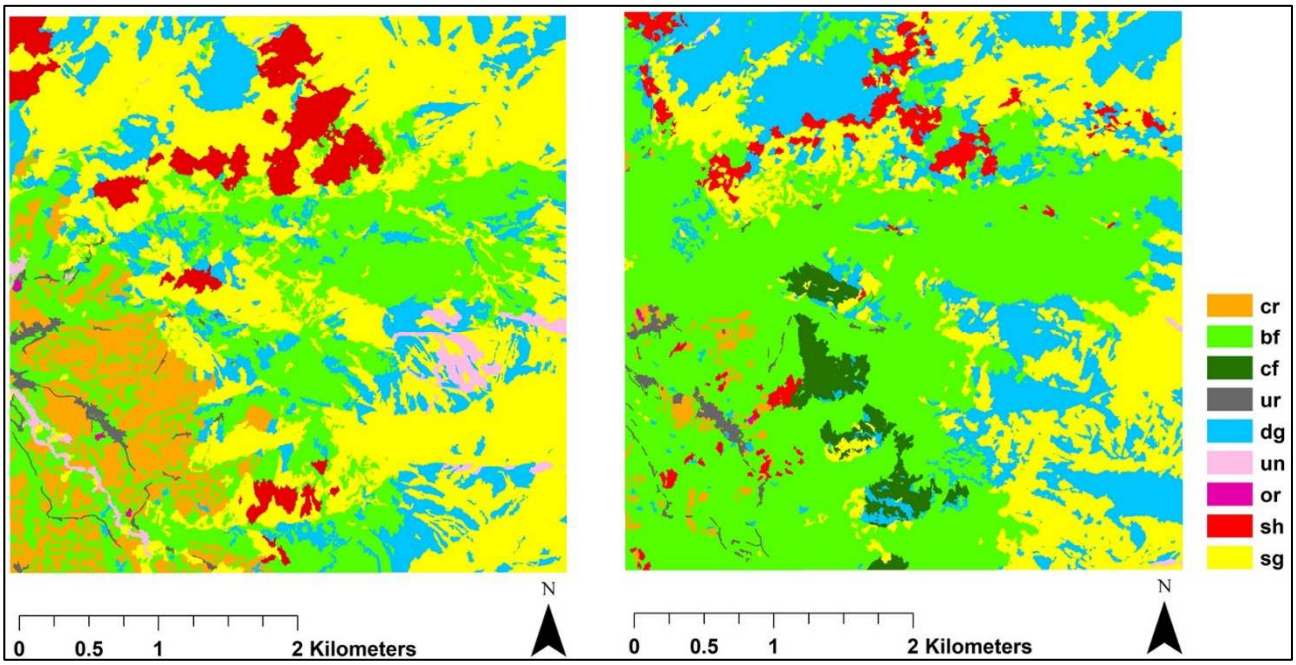

Figure S3 – Example of thematic maps of land cover (1954 and 2012) for Gorzano South-West landscape (Gosw). Land-cover classes are expressed with abbreviation: cr (cropland, herbaceous crops in general), bf (broadleaf forest), cf (conifer forest), ur (urban, buildings and infrastructures), dg (dense grassland), un (unvegetated, bare soil and water bodies), or (orchard, vineyards, other tree groves), sh (shrubland), sg (sparse grassland).

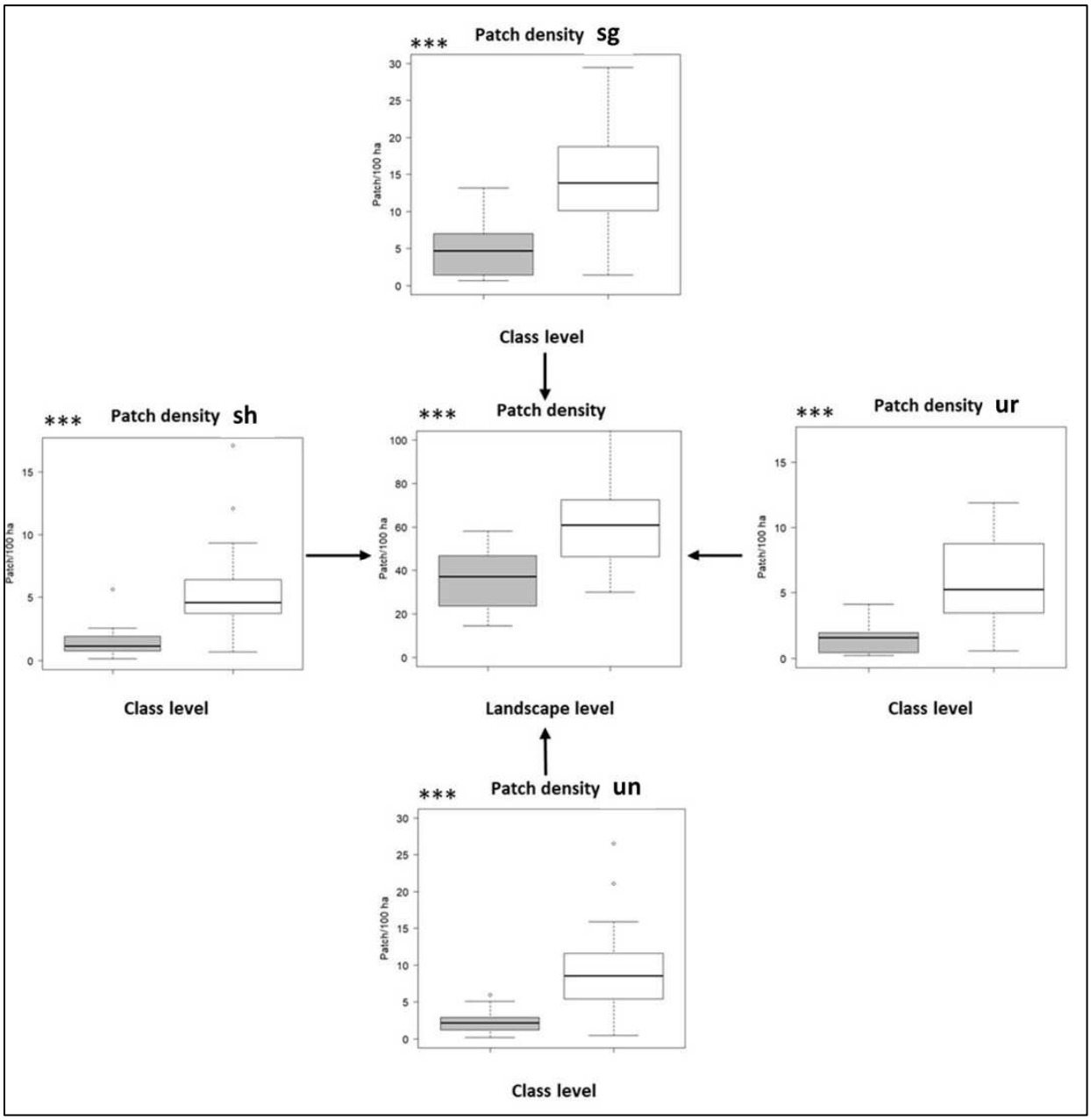

Figure S4 – Mean distribution of patch density variation over time (1954 = gray boxes, 2012 = white boxes) across 18 replicate landscapes: the central boxplot refer to landscape-level patch density whereas the four external boxplots represent class-level patch density of 4 LC categories that contributed to the landscape-scale index (arrows to the central box). Land-cover classes are abbreviated as follows: sh (shrubland), sg (sparse grassland), ur (urban), un (unvegetated land). Horizontal lines inside boxes are the median values and circles are outliers. * = p-value < 0.05, ** = p-value < 0.01, *** = p-value < 0.001, ns = not significant (Wilcoxon paired test to compare 1954 and 2012 indices).

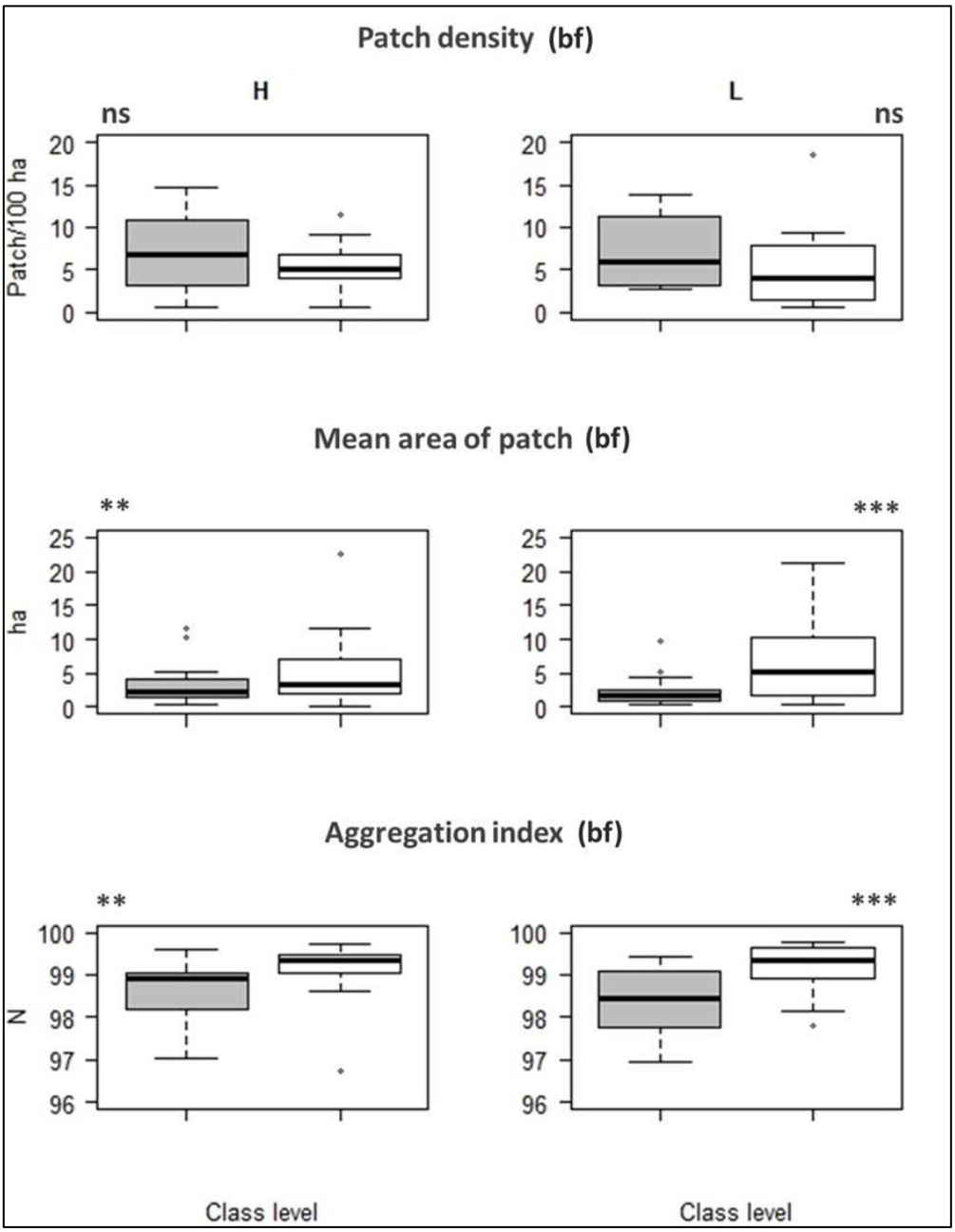

Figure S5 – Mean distribution of class metrics (broadleaf forest = bf) over time (1954 = gray boxes, 2012 = white boxes) and according elevation zones (H = High, L = Low) across the 18 replicate landscapes: patch density; mean patch area; aggregation index. Horizontal lines inside boxplots are median values and circles are outliers. * = p-value < 0.05, ** = p-value < 0.01, *** = p-value < 0.001, ns = not significant (Wilcoxon paired test to compare 1954 and 2012 indices for each metric).

CHAPTER 4

Wildfires and landscape dynamics: effects of two extreme fire seasons in Italy

The material was presented at the XI National Congress of the Italian Society of Forest Ecology and Silviculture (SISEF) held in Palermo 12-15 November 2019. Abstract ID: #c12/206

A printable version of this chapter with improvements and extended analysis is in progress and will be submitted to an international indexed journal

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Abstract

Recent changes in both climate and land use/cover in the Mediterranean basin are expected to be synchronous drivers for fire-regime shifts, also occurring in the Italian peninsula where fire is one of the most common forest disturbances. We studied wildfires occurrence, their spatial distribution and severity in two dry years in peninsular and insular Italy, the geographic area with a prevailing summer fire season. This chapter is a first draft of a manuscript in progress to be submitted shortly to an indexed international journal. The study provides first results on fire characteristics (e.g. extension, severity) of 113 forest fires that occurred in two exceptionally dry years (2007, 49 fires and 2017, 64 fires). We first assessed the differences of drought conditions using the self-calibrated Palmer Drought Severity Index and then classified fire severity based on land cover types of burned areas. We used burned area products provided by the MODIS database and selected all forest fires larger than 100 ha. We collected Landsat TM and OLI multi-spectral images of selected areas and computed the Relative difference NBR index (RdNBR) for each event. We applied a semi-automatic procedure (automatic segmentation and manual classification) to extract the fire perimeters and to assess fire severity. Drought conditions were more severe in 2017 especially in central Italy but large forest fires occurred more in southern regions, mainly in July and August. The average fire surface area was larger in 2007 and shrublands and broadleaf forests were the land categories more widely burned. Approximately 2400 ha burned in both years (2% of total 2007 and 2017 burned areas). We did not detect relevant differences averaging severity of combined 2007 vs. 2017 fires, but conifers and shrublands reached the highest values in both years. Next step will be to apply a predictive model to assess the landscape sensitivity to fire, its main drivers and severity changes.

Introduction

Fire regimes have changed throughout the history in many landscapes worldwide (Guyette et al. 2002; Pausas & Keeley 2009), especially in relation to changes of human activities and climate (Marlon et al. 2008). Humans played a determinant role in modifying landscapes and fuel load and continuity (Pausas and Keeley 2009). The increase of tree density as a result of fire suppression and reforestation actions have increased the build-up of woody fuels and consequently altered fire regimes (Covington & Moore 1994; Moritz 2003; Pausas et al. 2008; Stephens et al. 2013). Fire intensity and regimes shift can derive also from changes in fuel flammability depending on vegetation moisture controlled both at short (i.e. 1-hour) and medium (i.e. seasonal) term by the variations of temperature, precipitations and relative humidity. Wildfires can be largely dependent on other weather parameters, such as short-term droughts which are, with wind speed, the main drivers of the spread rate and propagation direction of fires (Rothermel, 1972; Flannigan et al. 2009).

In the last decades, the use of satellite optical remote sensing for mapping wildfire distribution and severity (Gitas et al. 2012) and tracking fire regime changes has increased significantly. After a fire in a burned area, a drastic reduction of chlorophyll content together with an increase of bare soil and the alteration of soil moisture occurs and the spectral characteristics of the residual vegetation change with an increase in the visible (except for green) and diminution in the near infra-red region of the electromagnetic spectrum (Escuin et al. 2008). For this reason, remote sensing is considered a valuable tool for mapping burned areas and assessing fire severity, offering adequate spectral and spatial resolution. Multi-spectral imagery provided by the United States (US) Landsat program can be considered as the most valuable source of time-series data to study vegetation-response to disturbances at a landscape scale (Frolking et al. 2009; Morresi et al. 2019). Multi-band satellite imagery allows extraction of a series of spectral indices combining the red, near-infrared and short-wave infrared regions in order to discriminate burned areas: for example, the NBR (Normalized Burn Ratio) (Key & Benson 1999) has been successfully tested and used in the discrimination of burned areas and severity in the Mediterranean-basin region (Escuin et al. 2008).

The Mediterranean basin commonly hosts some of the most severe and extended wildfires, making up the 90% of the total European burned area (Chuvieco 2009). Fire is an integrated component of Mediterranean ecosystems since at least the Miocene (Dubar et al. 1995) and there is evidence of climatic change mainly in relation to increases in temperatures (Esteban-Parra et al. 2003) with obvious consequences in the fire regime (Pausas 2004). Recent changes of climate and land use/cover in the Mediterranean basin are expected to be synchronous drivers for fire-regime shifts (Bajocco et al. 2010). Temperature anomalies (Bedia et al. 2014) and summer droughts (Dimitrakopoulos et al. 2011) are

critical in explaining fire occurrences but, mostly in the Mediterranean basin, the interaction between climate and land-use/cover changes is responsible in raising the frequency, area and severity of wildfires (San Miguel-Ayanz et al. 2013; Hernandez at al. 2015; San Miguel-Ayanz et al. 2018). The increase in wildfire incidence in such highly populated area introduces also the high risk of direct damages to human lives, activities and infrastructures. Nearly 200 million people are living in just 5 countries: Greece, Italy, France, Spain and Portugal (EPRS 2019), all featuring an extensive wildland-urban interface (WUI), where the risk of fire spread is extremely high. In these countries, the rapid industrialization of the 20th century caused relevant socio-economic changes and a progressive but irreversible migration from rural to urban areas and an extensive decline of traditional agro-silvo-pastoral practices (Blondel & Aronson 2000). Reduction of forest management and woody vegetation expansion in marginal lands increased fuel loads (Moreira et al. 2011) and wildfire risk (Hernandez et al. 2015), which, together with increasing temperature and drought stress, will presumably continue to increase in the future (de Rigo et al. 2017). Such change requires adaptation strategies to avoid more devastating effects such as those occurred in these last few years in California (Stephens et al. 2018; Williams et al. 2019).

In the Apennines, the second largest mountain range of Italy located in a Mediterranean context, fire is a major disturbance in the summer, especially at lower-mid elevations. In the period between 1980 and 2017, the average annual burned area in Italy was 107 357 ha and the annual mean frequency of fires was 9 121 (San Miguel-Ayanz et al. 2018). Due to improvements in fire policies and prevention (FAO, 2010), the five-year average burned area in Italy decreased from 147 150 ha in 1980–1989 to 72 945 ha in 2010–2017, except for two climatically exceptional years (2007 and 2017) when 227 730 ha and 161 987 ha of surface area burned, respectively. In 2007 total precipitations were 16% less and mean temperatures were 1.2 °C higher than average. Positive thermal anomalies were detected in the whole country from January to August; April was the month with the greatest positive thermal anomaly in northern and central regions, whereas June in southern ones (ISPRA 2007). The occurrence of previous prolonged drought periods, higher summer temperatures and strong winds enhanced the spread of large wildfires similarly to those occurred in Greece during the same period (Camia & Amatulli 2009). Similarly, in 2017 precipitation was 22% lower and temperatures were higher by 1.3 °C than average. Concerning February-August period, 2017 has been the hottest year of the historical series, with the greatest positive thermal anomaly (around +2 C° on average), with peaks in June for central (+3.8 °C) and southern regions (+3.1 °C) (ISPRA 2017). Wildfires occur in central, southern and insular Italy generally in late-spring and summer period whereas in late winter and spring in the Alps. Future scenarios predict with return period analysis that Italy will experience the highest annual increase of extreme climatic events (Moriondo et al. 2006) that are likely to affect wildfire frequency

and severity. This study is aimed to assess wildfire occurrence, spatial distribution and severity in two dry years in peninsular and insular Italy, the geographic area with a summer fire season, selecting 113 large wildfires (≥100 ha) that occurred in 2007 and 2017 and creating a fire dataset using remote sensing techniques. In a second phase, a model-approach will be used to detect and predict fire severity in dry years for the entire study region. More detailed modelling of wildfire occurrence and severity in the context of climate change is needed to develop appropriate planning strategies to reduce the negative effects of wildfire and more accurately manage the forest recovery in mountain areas.

Materials and Methods

Study areas

The study area includes a large part of Italy extending between 44-37° N and 8-17° E, from Liguria to Calabria, and the two major islands (Sicily and Sardinia) for a total of 209 198 Km^2 (Figure 1). This is a homogeneous region for fire seasonality (summer) and for land cover change dynamics that occurred in last decades after the inner-area abandonment and human migration towards the plains and coasts. Broadleaf forests naturally expanded in former grasslands and croplands and mountain landscapes underwent an overall pattern simplification from these land use changes (Peroni et al. 2000; Cimini et al. 2013; Malandra et al. 2019).

Figure 1 – The study area (green line) includes the peninsular and insular regions of Italy excluding the Alpine regions due to the different fire seasonality.

The area has very heterogeneous morphological and geolithological features. The Italian peninsula is completely surrounded by the Mediterranean Sea and comprises some of the most varied and scenic landscapes. It has mainly hills and mountains with very limited and narrow plain areas along the coasts in river valleys. Most of the mountain ranges are formed from the uplift of Paleozoic igneous and primarily marine sedimentary rocks. Prevailing substrata are mainly terrigenous, clastic and carbonate, with interspersed igneous and metamorphic outcrops (Blasi et al. 2014).

Climatically, the area belongs to the Mediterranean and the Temperate division including the Tyrrhenian, Adriatic and Apennine eco-provinces (Blasi et al. 2014). The Mediterranean division has mean annual temperatures above 13°C and annual precipitation from 500 to 1400 mm. Rainfall decreases in summer with 2-3-month of aridity. The Temperate division features mean annual temperatures often exceeding 10°C. Annual precipitation is irregular but mainly with a bimodal distribution (relative peaks in autumn and spring), with snow in late winter and scarce rainfall in summer (Blasi et al. 2014). North-eastern slopes (Adriatic side) are in general more continental and rainy than South-western ones (Tyrrhenian side) (Malandra et al. 2019). Forests cover more than 35% of the national surface and more than 50% in some regions (e.g. Liguria, Toscana, Umbria, Sardegna). Broadleaf forests are approximately 68.0% of the total forest cover, followed by conifer forests (about 13.5%) and mixed forests (9.7%) (RAF ITALIA - Rapporto sullo stato delle foreste e del settore forestale in Italia, 2019).

Forest fires occur along the entire Italian peninsula, from north to south, in the Alps (mainly in late winter or spring especially in the Western sector), in the Apennines and on the main islands. Within the study area, fires have higher frequency and larger extension in the southern regions (Sicilia, Sardegna, Calabria and Campania) and generally occur during the summer season (June, July, August and rarely September) (Michetti & Pinar 2019). For the 1970-2017 period, the forest area cumulatively burned is about 2.2 M ha (Pettenella et al. 2017). A large part of all the forest fires occurs in the Apennines or pre-Apennines area (Corona et al. 2014) and the causes of ignition are largely human-induced (more than 99%) (Lovreglio et al. 2012).

Drought analysis

We divided the study area in Central regions (Liguria, Emilia-Romagna, Toscana, Umbria, Marche, Lazio, Abruzzo, Lazio, Molise) and Southern regions (Campania, Puglia, Basilicata, Calabria, Sardegna, Sicilia) of Italy and assessed drought conditions in both target years (2007 and 2017) using monthly self-calibrated Palmer Drought Severity Index (scPDSI) from the Climate Explorer grid

(https://climexp.knmi.nl/start.cgi) (Wells et al. 2004). We compared the monthly scPDSI values in both years with the 1950-2017 mean monthly values. We also computed the variation range extracting 1950-2017 mean standard deviation values for each month.

Wildfire selection and images preprocessing

We searched for large wildfires in the study area using MODIS Collection 5.1 Level 3 (monthly burned area products - MCD45) for 2007 and MODIS Collection 6 Level 3 (monthly burned area products - MCD64A1) for 2017. In a GIS environment, we first filtered wildfires larger than 100 ha. After the assessment of occurrence date (in MODIS burned area products) of these selected wildfires, we collected Landsat scenes selecting the least cloud-contaminated acquisitions (<10% of cloud cover) available during the vegetation growing season (02 June - 06 September). We collected 59 Landsat TM and OLI images (30x30 m spatial resolution) covering the whole study area. We downloaded images taken before and one year after each fire event. Landsat data are available at the USGS Earth Reso [1] Observation and Science (EROS) Center Science Processing Architecture (ESPA) On-Der Interface already processed in surface reflectance (Level-2 Science Products). Landsat 5 TM-surface reflectance products were generated by the USGS using the Landsat Ecosystem Disturbance Adaptive Processing System (LEDAPS) (Masek et al. 2006) while Landsat 8 OLI images processing was based on the Landsat Surface Reflectance Code (LaSRC) (Vermote et al. 2016). We masked cloud, cloud shadows and water bodies with Fmask 3.3 tool in each acquisition (Zhu et al. 2015). From the wide range of vegetation indices directly provided by the USGS, we selected the surface-reflectance NBR index (Normalized Burn Ratio) for fire-perimeter and burn-severity assessment. NBR [1] exploits near-infrared (NIR) and short-wave infrared (SWIR2) bands to explore spectral response of an area in a specific vegetative status (Key & Benson 1999):

$$\frac{NIR - SWIR2}{NIR + SWIR2} \qquad [1]$$

Fire perimeter and burn severity assessment

We extracted fire perimeter and assessed spatial distribution of burn severity for each wildfire producing in a GIS environment the Relative difference NBR (RdNBR). The difference NBR (dNBR) [2] was computed subtracting pre-fire NBR to post-fire NBR (Key 2006; Key & Benson 2006) and relativized dividing the index by the square root of the pre-fire NBR [3] (Miller & Thode 2007). The implementation of pre-fire condition of vegetation with a relative version of dNBR proved useful for a more accurate assessment and a more direct comparison of severity between fires across space and time (Miller & Thode 2007). An extended assessment of remotely sensed burn severity was performed analyzing spectral changes caused by the fire during the following vegetative season.

$$dNBR = ((NBRprefire - NBRpostfire) * 1000) - dNBRoffset \quad [2]$$

$$RdNBR = \frac{dNBR}{\sqrt{|NBRprefire|}} \quad [3]$$

We adopted a semiautomatic method to extract fire perimeter from fire scars. We applied an automatic segmentation using eCognition Developer 64 8.9 software (Gitas et al. 2004) and a manual on-screen classification (Key and Benson 2006) of burned segmented polygons. We then corrected perimeters through on-screen digitization of burned pixels in post-fire Landsat TM/OLI images, using false-color composites (RGB = SWIR2, NIR, Red), as misclassification between unchanged and low-severity pixels occurred frequently (Morresi et al. 2019). For the fire severity assessment, we calibrated dNBR with a $dNBR_{offset}$, which was computed averaging dNBR values of undisturbed pixels located close to fire perimeters in order to account for differences due to phenology or precipitation between the pre- and post-fire images. Such pixels belong to 2-3 land-cover categories which burned in the corresponding fire (Key 2006; Fernandez-Garcia et al. 2018; Morresi et al. 2019). Finally, we filtered and selected those wildfires with a presence of at least 50% of forest-related land cover types (broadleaf, conifer and mixed forest, heterogeneous areas with complex agriculture-forest systems, shrubland, sparse vegetation and grassland) within perimeters, intersecting Corine Land Cover 2006 (CLC06) for 2007 and Corine Land Cover (CLC12) for 2017 fires. Figure S1 and S2 (see Supplementary materials section) provide examples of severity maps (RdNBR) for two 2007 and 2017 fire events.

We extracted burned surface area (ha), perimeter length (km), coordinates of the fire centroid (WGS84UTM33N) and regions of fire occurrence. We attributed a location with the closest toponym and an identification code (ID) was specified to each fire detected in 2007 and 2017 (Supplementary materials – Table S1). We used a Digital Elevation Model 30x30 (DEM SINAnet ISPRA) to determine topographic features of the fire surface area and CLC06 (2007 fires) and CLC12 (2017 fires) to check land-cover types of the burned areas. We analyzed land covers of reburned areas in three different time-steps (2006 = CLC06, 2012 = CLC12, 2018 = CLC18). Finally, we associated land-cover and fire severity of each event in 2007 and 2017 for further comparison.

Results

Intra-annual scPDSI trend showed that conditions were drier in both years than the 1950-2017 mean. 2017 was generally drier than 2007 in the central regions, especially in the summer (June -4.4; July -4.8;

August -5.0), than in southern ones (June -2.7; July -2.9; August -3.0). 2007 was drier in southern regions only in January and February (Figure 2).

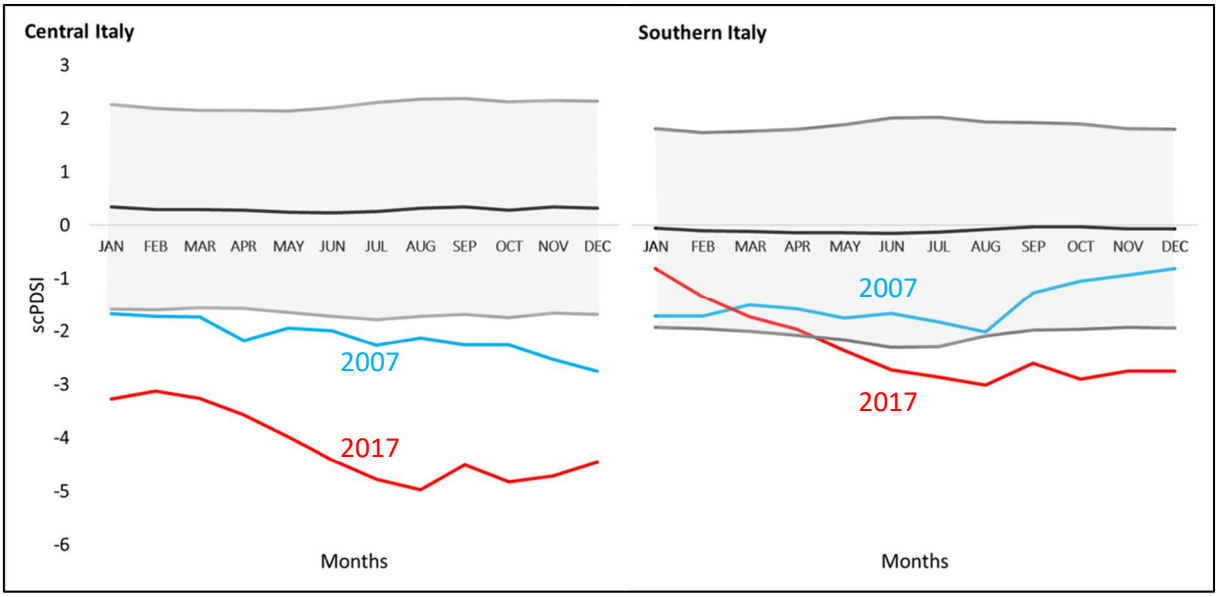

Figure 2 - Monthly scPDSI values in 2007 (blue lines) and 2017 (red lines) in central vs. southern regions compared to the 1950-2017 drought trend mean (black line) and standard deviations (gray lines).

Our dataset consists of 113 large wildfires: 49 occurred in 2007 and 64 in 2017 (Figure 3 – Table S1).

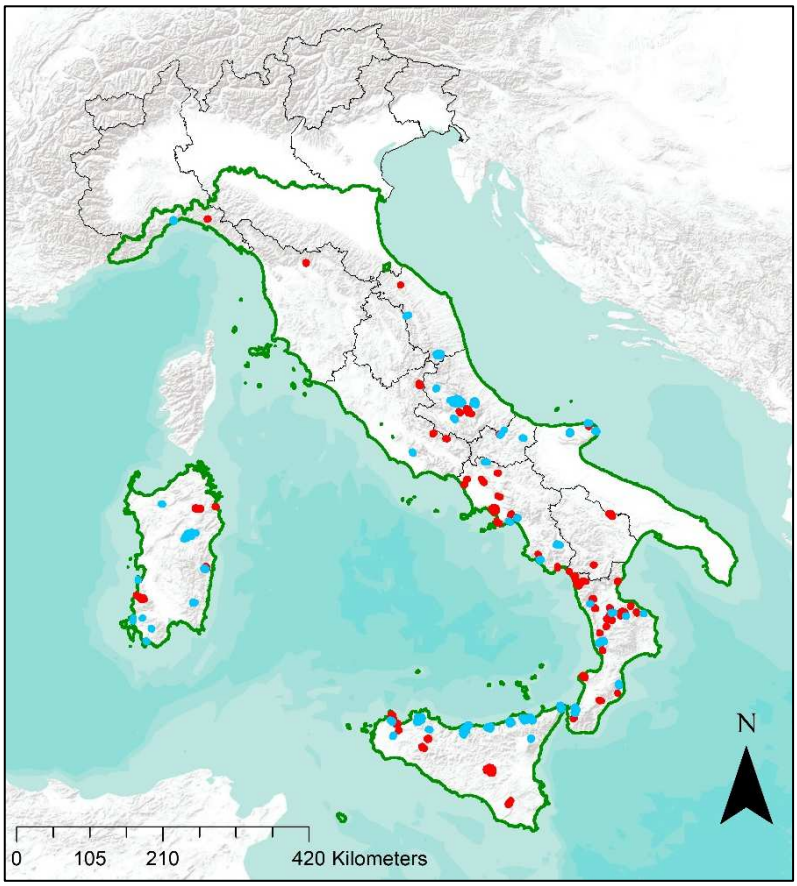

Figure 3 – Geographical distribution of 2007 (blue) and 2017 (red) wildfires within the study area.

Total burned area in 2007 and 2017 wildfires were 56 765 ha and 45 800 ha, respectively. The mean burned area was 1158 ha (± 1587, min = 121 ha, max = 7474 ha) in 2007 and 716 ha (±861, min = 108 ha, max = 3742 ha) in 2017. The largest fire was 7474 ha detected near Nuoro (Sardegna) in 2007 and the smallest was 108 ha near Camini (Calabria) in 2017 (Table S1). Fires are mostly located in southern Italy, especially in Calabria (n=34), Sicilia (n=22), Campania (n=16) and Sardegna (n=13). Sicilia was the most disturbed region with a total of 30 769 ha burned in the two years (Table 1).
Most frequently (n=41/113), the selected forest fires belong to the 251-500 ha class and about 50% were between 250 and 750 ha (Figure 4). Nearly all fires occurred in the summer, especially in July (47% in 2007 and 53% in 2017) and in August (39% in 2007 and 45% in 2017) (Figure 5).

Wildfires occurred in a wide range of altitude (0-1978 m a.s.l.) and slope (0-76°), but mean elevation was 547 (±302) m a.s.l. in 2007 and 567 (±279) m a.s.l. in 2017, whereas mean slope was 19.2° (±6.5°) in 2007 and 20.2° (±6.5°) in 2017. Main slope aspect varied largely across the study area, but fires were mainly on warmer exposures. About 43% of the total burned area was comprised in nature conservation areas (e.g. UE Natura 2000 network and National or Regional protected areas).

Table 1 - Total number of fires and burned area per year and region of the study area. Regions are listed by decreasing overall burned area.

Region	Fire (count) 2007	Burned area (ha) 2007	Fire (count) 2017	Burned area (ha) 2017
Sicilia	11	17977	11	12791
Calabria	10	5392	24	12227
Sardegna	9	11141	4	5311
Abruzzo	6	11142	3	3438
Campania	5	3278	11	6538
Puglia	3	2498	3	2495
Marche	2	3645	1	172
Liguria	1	873	1	236
Lazio	1	312	3	1759
Basilicata	0	0	2	501
Molise	1	507	0	0
Toscana	0	0	1	331
TOT	49	56765	64	45799

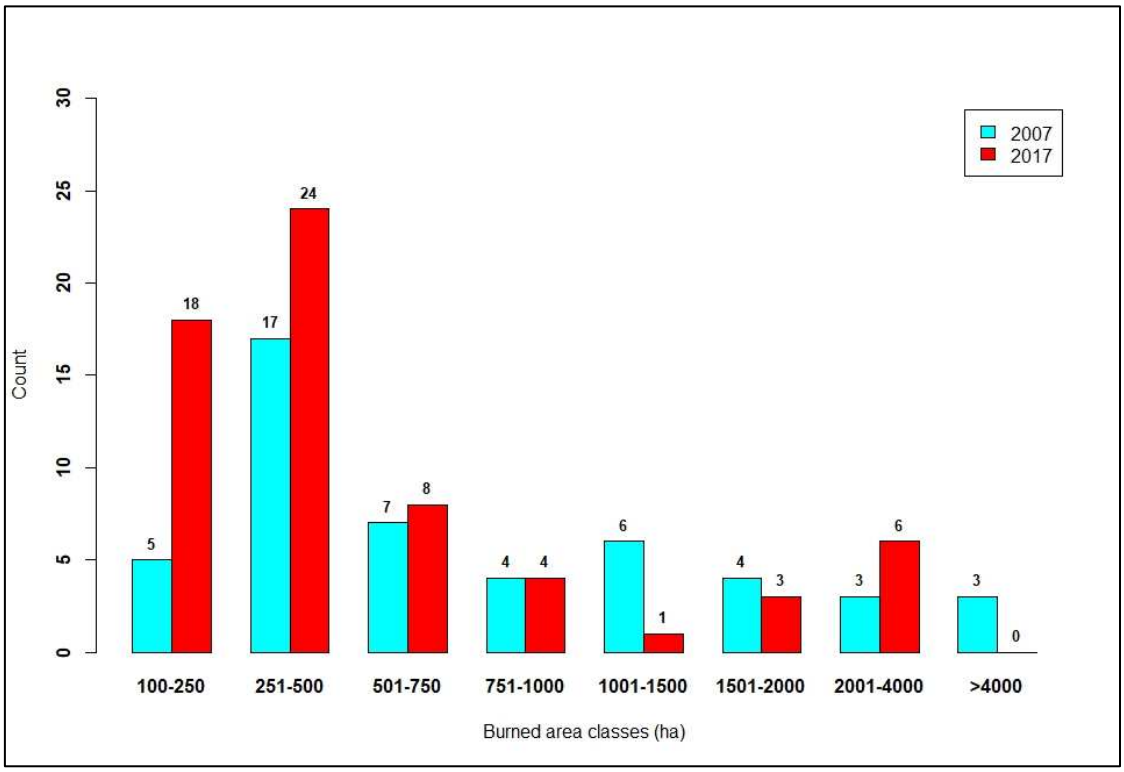

Figure 4 – Frequency distribution of 2007 (blue) and 2017 (red) fires in classes of burned area (ha). Labels over bars are the relative fire counts.

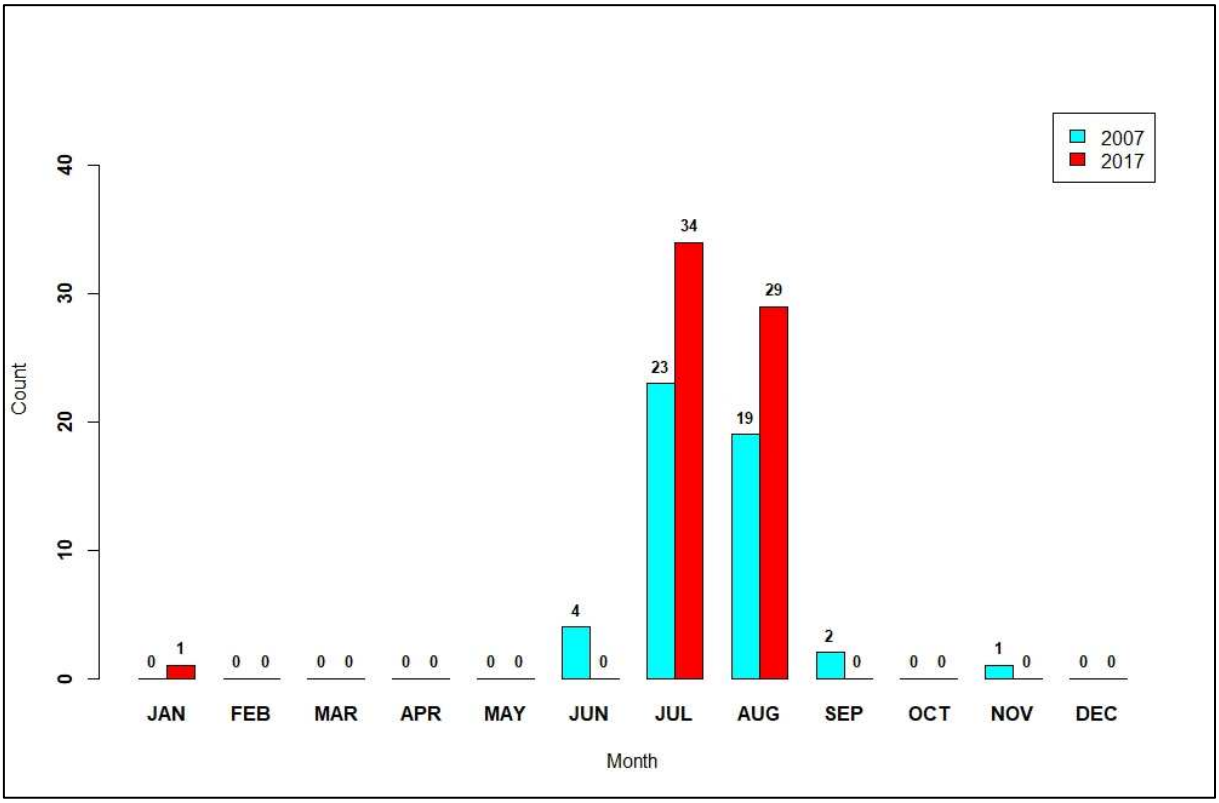

Figure 5 – Fire occurrence on corresponding months in 2007 (blue) and 2017 (red). Labels upon bars show relative counts.

In the two years, 30.1% of the burned area was shrubland (31 590 ha), 19.8% broadleaf forest (20 755 ha), 13.5% grassland (14 204 ha), 9.2% conifer forest (9657 ha) and 5.2% mixed forests (5478 ha) (Table 2 – Figure 6). Shrublands, grasslands and broadleaf forests share similar percentage in 2007 and 2017, with some differences on cropland (8.3% in 2007 and 3.4% in 2017), heterogeneous areas with agriculture and forest (17.7% in 2007 and 7.3% in 2017), conifer forest (5.7% in 2007 and 13.4% in 2017) and mixed forest (3.1% in 2007 and 7.7% in 2017) (Table 2 – Figure 6). Forest-shrub land-covers burned more in 2017 (73.3%) than in 2007 (56.6%), whereas more agro-pastoral cover-types burned in 2007 (42.4% vs. 26.4% in 2017).

Only in 2% of the total area (2400 ha) fires seemed to have occurred in both years. Here we detected similar land-cover shares one year before (2006) and five years after (2012) the 2007 fires: for instance, shrubland (Shb) were 57.1% (2006) and 69.4% (2012) of total reburned area and broadleaf forest (Brf) 12.3% and 13.0%. More evident variations occurred one year after (2018) the second fire: a decrease in shrublands (69.4% in 2012 and 42.3% in 2018), in conifer forest (from 4.2% in 2012 to 0.2% in 2018) and an increase in sparse vegetation (0.3% in 2012 to 37.8% in 2018) occurred (Table 3).

Table 2 – Burned area each year and for land cover category. **Urb** = Urban, **Orc** = Orchard, **Crp** = Cropland, **Agf** = Heterogeneous areas with agriculture and forest, **Shb** = Shrubland, **Svg** = Sparse vegetation area, **Grs** = Grassland, **Brf** = Broadleaf forest, **Cof** = Conifer forest, **Mxf** = Mixed forest.

	2007		2017		Total burned	
Land cover categories	Ha	%	Ha	%	Ha	%
Urb	512	0.9	151	0.3	663	0.6
Orc	368	0.6	783	1.6	1151	1.1
Crp	4710	8.3	1627	3.4	6337	6.0
Agf	10062	17.7	3517	7.3	13579	12.9
Shb	16654	29.3	14935	31.0	31590	30.1
Svg	627	1.1	916	1.9	1543	1.5
Wtb	17	0.0	5	0.0	22	0.0
Grs	8338	14.7	5866	12.2	14204	13.5
Brf	10547	18.6	10208	21.2	20755	19.8
Cof	3228	5.7	6429	13.4	9657	9.2
Mxf	1760	3.1	3718	7.7	5478	5.2

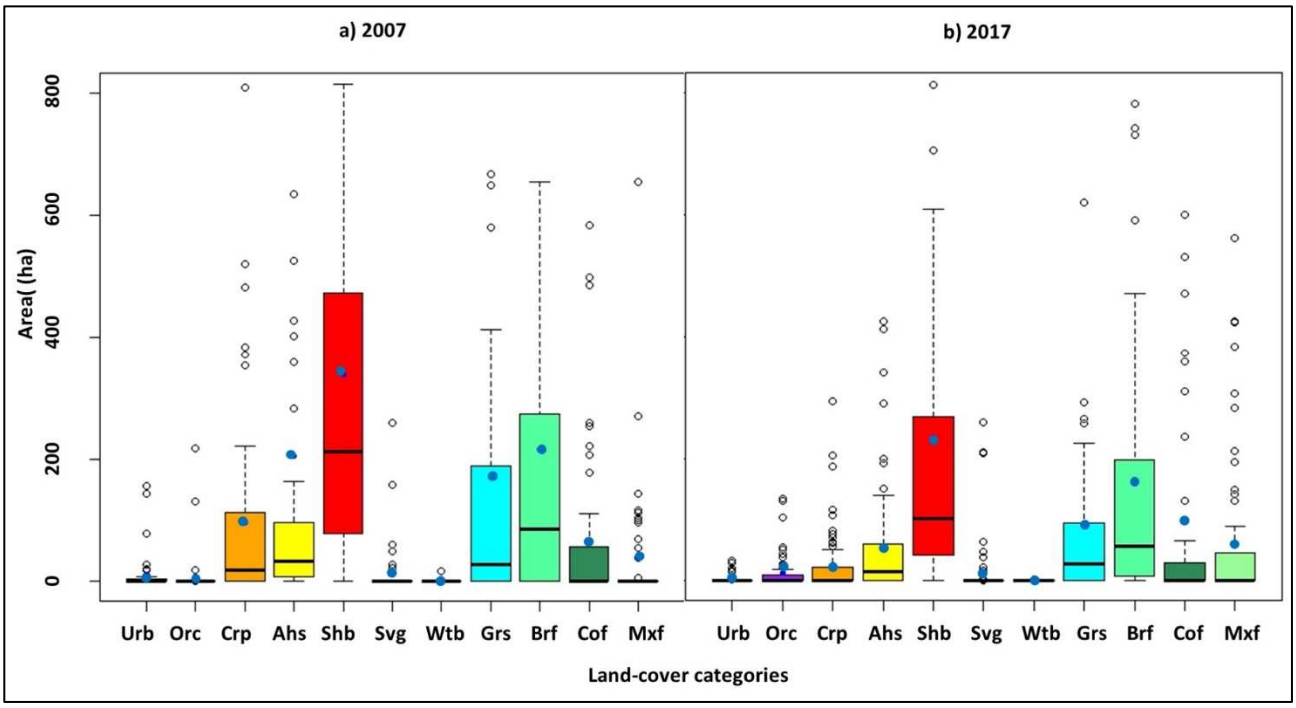

Figure 6 - Mean area (ha) of land cover types burned in the two years. Median values (horizontal bold lines), mean values (blue circles) and outliers (white circles) are showed. **Urb** = Urban, **Orc** = Orchard, **Crp** = Cropland, **Agf** = Heterogeneous areas with agriculture and forest, **Shb** = Shrubland, **Svg** = Sparse vegetation, **Wtb** = Water body, **Grs** = Grassland, **Brf** = Broadleaf forest, **Cof** = Conifer forest, **Mxf** = Mixed forest. Y-axis maximum value was limited to 800 ha to facilitate data comparison.

Table 3 – Share of variation of land-cover categories in twice burned areas, Corine Land Cover one year before (2006), five years after (2012) the first fire (2007) and one year after (2018) the second fire (2017). **Urb** = Urban, **Orc** = Orchard, **Crp** = Cropland, **Agf** = Heterogeneous areas with agriculture and forest, **Shb** = Shrubland, **Svg** = Sparse vegetation area, **Grs** = Grassland, **Brf** = Broadleaf forest, **Cof** = Conifer forest, **Mxf** = Mixed forest

CLC Categories	2006 %	2012 %	2018 %
Urb	1.0	1.0	0.8
Orc	0.5	0.5	0.0
Crp	0.1	0.4	0.0
Agf	17.7	4.6	6.0
Shb	57.1	69.4	42.3
Svg	0.0	0.3	37.8
Grs	7.0	6.7	1.2
Brf	12.3	13.0	11.7
Cof	4.1	4.2	0.2

Mxf	0.0	0.0	0.0

In terms of fire severity, mean values of RdNBR are overall similar, slightly higher in 2007 (276 ± 261) than in 2017 (246 ± 242) as shown by mean RdNBR values (red dotted lines) in pixel distribution histogram (Figure 7).

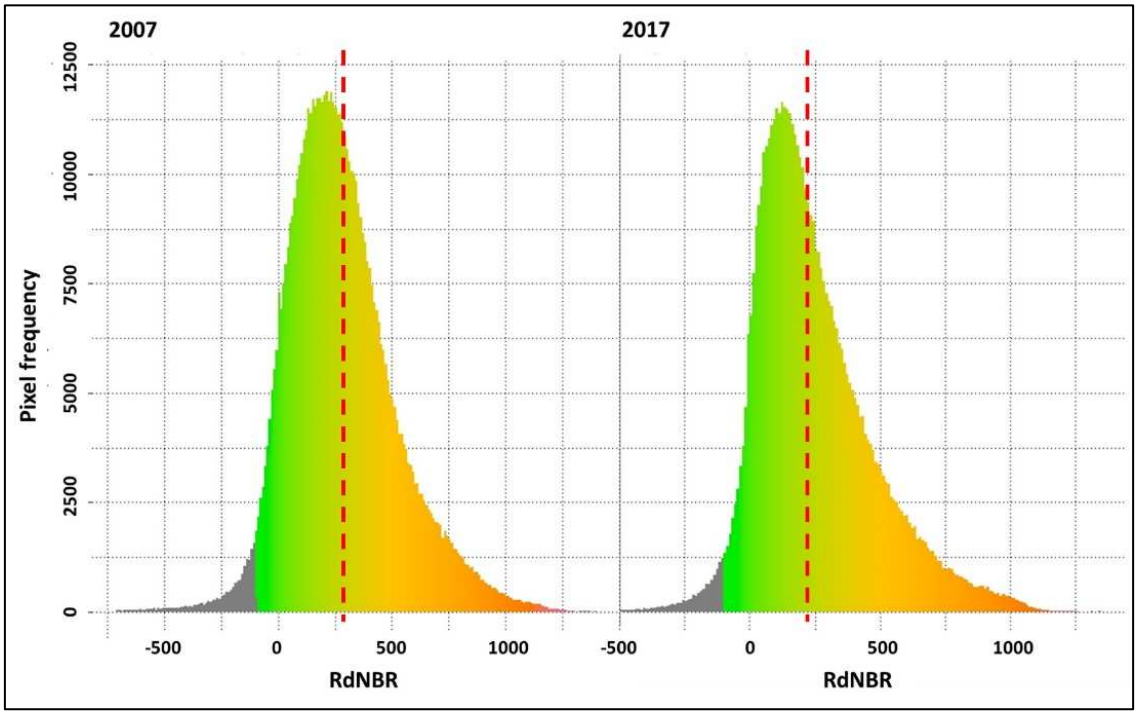

Figure 7 – Frequency distribution of RdNBR values of pixels in 2007 (left) and 2017 (right) fires. Higher values of RdNBR indicate increasing severity (green-yellow-red scale color). Dotted red lines show the mean RdNBR values of 2007 (276) and 2017 (246) fires.

Comparing fire severity and land cover, conifer forests have the higher RdNBR mean value (357 ± 269), followed by shrubland (293 ±245), mixed forests (288 ± 249) and broadleaf forest (271 ± 244). (Figure 8).

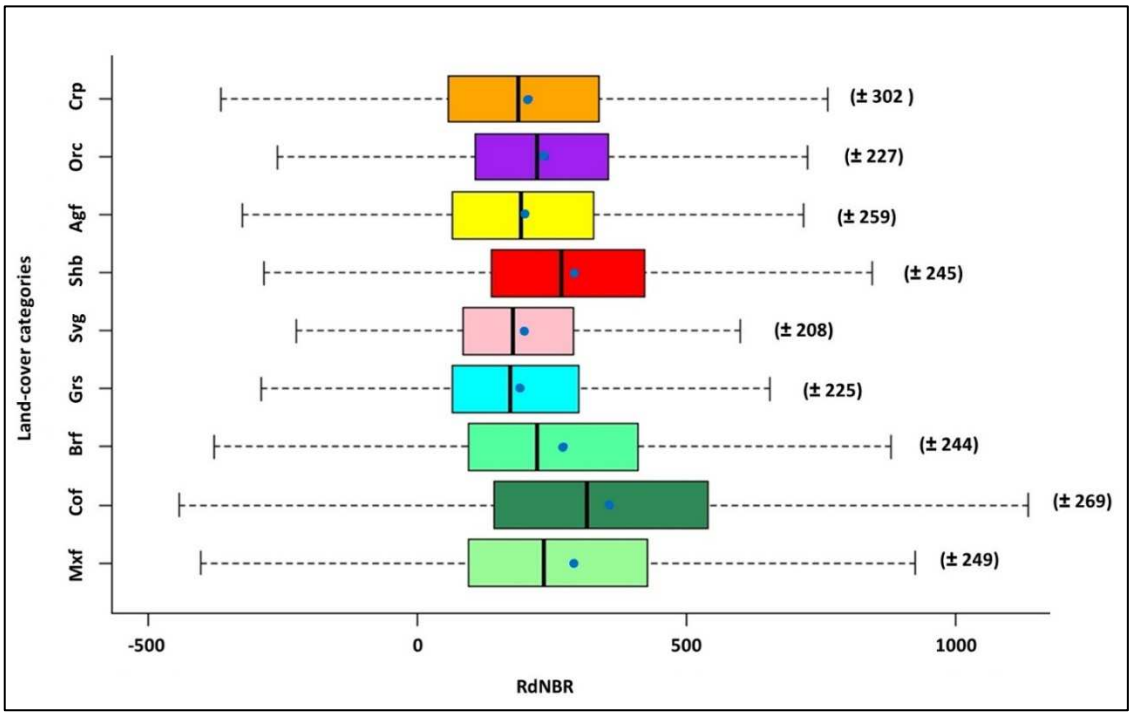

Figure 8 – RdNBR values of each land-cover category for all of the 113 fires. Vertical bold lines are the median values. Blue dots are mean values. Outliers are not showed. **Orc** = Orchard, **Crp** = Cropland, **Agf** = Heterogeneous areas with agriculture and forest, **Shb** = Shrubland, **Svg** = Sparse vegetation area, **Grs** = Grassland, **Brf** = Broadleaf forest, **Cof** = Conifer forest, **Mxf** = Mixed forest.

Discussions

Many ecosystems worldwide are experiencing a shift in fire regime in consequence of land use/cover and climate change (Guyette et al. 2002; Stephens et al. 2013; Borchers Arriagada et al. 2019; Gorbatenko et al. 2020; Lizundia-Loiola et al 2020). Especially in the Mediterranean basin, the climatic shift, the accumulation of biomass load and the increase in fuel continuity are contributing to increase the frequency, surface and severity of wildfires (San Miguel-Ayanz et al. 2013; Hernandez at al. 2015; San Miguel-Ayanz et al. 2018).

At national scale both selected years (2007 and 2017) were considerably drier than the mean of the 1950-2017 period, with higher temperatures and less precipitation (+1.2 °C and -16% in 2007; +1.3 °C and -22% in 2017) (ISPRA 2007; ISPRA 2018). Drought conditions were more severe in central regions in 2017. A prolonged drought period and higher summer temperatures were likely the main drivers of the higher rate of wildfires. Pausas & Fernandez-Munoz (2012) suggested that wildfires are currently less fuel-limited and more drought-driven than 30 years ago in western Mediterranean basin.

Indeed, exploring European fire records (San Miguel-Ayanz et al. 2018), the burned area in Italy since 2000 was much higher in extremely dry years such as 2007 and 2017. Since the mean size of a forest fire in Italy is approximately 12 ha (Pettenella & Corradini 2018), due to the strong effort given to the fire extinction actions (land and airborne), we considered only large and mega wildfires (> 100 ha and > 500 ha respectively) since severe climatic condition can increase their incidence (San-Miguel-Ayanz et al. 2013), with possible catastrophic damages in terms of human casualties and economic losses. We found more numerous large forest fires in 2017 although the overall burned area was similar in both years (56 765 ha in 2007 and 45 800 in 2017) due to averaged larger wildfires in 2007. These wildfires were more widespread in southern regions and occurred in July and August, as also indicated by EFFIS-JRC fire annual reports for Italy. Some policy changes that occurred after 2016 on the national fire-defense organization may have influenced fire activity at national scale. Specifically, the role of the Italian police force (Corpo Forestale dello Stato: CFS) on forest management has been reduced as a consequence of a new law (ddl Madia: 7/8/2015, n.124), which has mutated the governance of fire-fighting actions (Michetti & Pinar 2018). Therefore, the interplay between a drought-driven weather pattern and concurrent changes in fire-policy governance could have affected recent wildfires occurrence.

The analysis of land-cover types burned in the two years provided information about large fire selectivity in forest ecosystems. The 64% of total burned area was shrubland and all types of forests (mostly broadleaf). The co-occurrence of shrubs-forests cover, common in landscape mosaics of many Mediterranean ecosystems, was fostered by secondary succession in former grasslands and croplands after agro-pastoral activities abandonment (Gartzia et al. 2014; Vacchiano et al. 2016; Malandra et al. 2019) and is very prone to burn after prolonged dry conditions. In addition, shrubland is a very variable land-cover type worldwide in terms of species composition, fuel characteristics and structure (Cowling et al. 1996) and, together with conifer forests, is very fire sensitive in the Mediterranean Basin. Several studies in southern Europe suggested that shrublands and some conifer stands are the most prone to fire (Moreira et al. 2009; Barros et al. 2014). Our results are coherent in terms of fire selectivity: shrub-forest covers (shrublands, broadleaf, conifer and mixed forests) burned more in 2017 (73%) than in 2007 (57%) due to the drier conditions. Agro-pastoral covers (cropland, orchards, heterogeneous areas with agriculture and forest, grasslands and sparse vegetation areas) burned more in 2007 (42% and 26% respectively). The precision of land cover analysis can be limited by the type of data available at national scale (Corine Land Cover), which is designed as a pan-European data set and not as a national one. The spatial resolution and classification structure have been conceived to fit a broad range of environmental regions of European Union and not singles countries (Bossard et al. 2000).

Nonetheless, the CLC data are widely used by researchers (Mancini et al. 2017; Amos et at. 2019) and are a suitable tool to compare land covers in large burned areas.

The use of a relative measure of spectral change (RdNBR) was crucial to assess burn severity in heterogeneous landscapes with different vegetation types and to compare fire-response of several burned areas in space and time. An absolute measure of change (dNBR) could lead to incorrect estimates of burn severity, especially in landscape with several land-cover types featuring different pre-disturbance amount of chlorophyll in the biomass load (Miller and Thode 2007). Overall, we did not detect a significant difference of mean severity combined all the fires of 2007 and 2017, but only a slightly higher mean value for 2007. This is not surprising considering that similar land-cover types burned and weather patterns were globally comparable in both years. Conifer forests are the land-cover type with the highest mean severity (RdNBR = 357), followed by shrubland (RdNBR = 293) and mixed forests (RdNBR = 288). It is well known that burn severity assessed with remotely sensed NBR is primarily controlled by vegetation type (Miller & Thode 2007). Conifer stands are more flammable than broadleaf forests because of their higher content of resins and terpenes (Nord & Green 1977) and the lower foliage moisture content (Rowe & Scotter 1973). Therefore, burn severity is expected to be higher in conifer/mixed than in broadleaf forest, as detected by other studies in Mediterranean-basin ecosystems (Fernandes et al. 2010). Shrub cover, if extended on large portions of ecosystems, plays an active role in fire spread and generally shows a high rate of severity due to its high proneness to burn. However, severity assessment is strictly related to the method applied: we assessed spectral changes caused by fire during the following vegetative season, defined as an extended assessment of remotely sensed burn severity. This method considers first- and second-order effects caused by fire and includes delayed survivorship and mortality of vegetation (Key 2006). Such approach is generally used for forest and some shrubland fire (Vogelmann et al. 2011), but it can underestimate the burn severity of some land-cover types (e.g. shrublands), since a limited amount of regrowth of herbs/shrubs is common in the following vegetative season (Key 2006).

Conclusions

These preliminary results show similar features and effects of forest fires occurred in two very dry years in the peninsular and insular Italy. The amount and quality of the dataset call for further analysis: specifically, we will develop predictive models suitable for two main goals: i) to investigate, among numerous physical and anthropic factors, the most influential drivers of fire severity in dry and hot years at regional scale, including in the model topography (elevation, slope, aspects, etc.), land-cover

types (vegetation) and anthropic variables (infrastructures, human pressure, etc.); ii) to forecast the impact of wildfires on forest ecosystems in the context of climate change (in drier and warmer years). A predictive generalized linear mixed model or linear mixed effect model should be suitable to reach these goals, including years (2007 and 2017) and fire ID as random effects. The integration in the model of human factors affecting fire spread and severity will be crucial since the increasing fire activity is also determined by drivers involving ignition sources (mostly human-induced) and land management (land use, human activities, etc.). Moreover, in southern European countries (Portugal, Greece, Italy, etc.), the expansion of wildland-urban interfaces (WUIs) and the related increasing likelihood of dangerous fire events within them is raising concern in policy, land managers and scientific community for the direct ecological and socio-ecological implications. Thus, the output of the modelling approach will prove useful to develop appropriate planning strategies to reduce the negative effects of wildfire in a climate-change scenario and more accurately manage the forest recovery in mountain areas.

Supplementary materials

Table S1 - List of localities and main features of wildfires used for the analysis. Fire events are sequentially numbered and ordered by decreasing latitude. The name assigned to the fire event is that of the closest official toponym.

ID	Name	Acronym	Year	Region	x UTM33N	y UTM33N	Area ha	Perimeter km
1	Moconesi	MOC17	2017	Liguria	39226.2	4937356.3	235.6	24.8
2	Cogoleto	COG07	2007	Liguria	-9893.6	4935097.4	873.4	26.4
3	Montale	MON17	2017	Toscana	180453.6	4874486.6	330.6	20.4
4	Cesane	CES17	2017	Marche	316299.3	4842416.5	172.3	11.1
5	Cancelli	CAN07	2007	Marche	326245.2	4798309.3	533.9	41.3
6	Roccafluvione	ROC07	2007	Marche	371226.4	4742086.8	3110.8	94.9
7	Antrodoco	ANT17	2017	Lazio	343862.2	4698787.8	1149.1	34.6
8	L'Aquila	AQU07	2007	Abruzzo	367904.0	4693378.4	356.7	24.0
9	Navelli	NAV07	2007	Abruzzo	396950.4	4674546.3	7349.3	217.2
10	Lettomanoppello	LET07	2007	Abruzzo	422894.1	4672789.3	2260.7	82.7
11	Morrone	MOR17	2017	Abruzzo	413781.7	4660716.9	2775.3	84.2
12	Prezza	PRZ17	2017	Abruzzo	401304.8	4659378.5	310.1	21.8
13	Ortona Dei Marzi	ODM07	2007	Abruzzo	393833.0	4649811.7	461.7	33.4
14	Peschici	PES07	2007	Puglia	586412.9	4643470.0	781.9	29.8
15	Peschici	PES17	2017	Puglia	586828.6	4638929.0	257.2	20.3
16	Valle del Trigno 1	VT107	2007	Abruzzo	465597.0	4633349.8	140.9	15.4
17	Vieste	VIE07	2007	Puglia	597003.7	4632119.7	710.2	46.3
18	Cagnano Varano	CAG17	2017	Puglia	560031.6	4630074.2	529.6	24.3
19	Cagnano Varano	CAG07	2007	Puglia	560056.2	4629714.7	1006.4	45.2

20	Vico nel Lazio	VNL17	2017	Lazio	363892.5	4628545.0	397.9	24.2
21	Valle del Trigno 2	VT207	2007	Abruzzo	460546.1	4627204.8	572.2	28.0
22	Schiavi d'Abruzzo	SDA17	2017	Abruzzo	459226.4	4626860.2	352.9	24.8
23	Montorio nei Frentani	MNF07	2007	Molise	492567.8	4621941.6	506.7	22.0
24	Sora	SOR17	2017	Lazio	382584.5	4620722.8	212.2	21.5
25	Bassiano	BAS07	2007	Lazio	333763.4	4601127.6	311.7	24.0
26	Valle Agricola	VAG07	2007	Campania	439233.5	4587328.6	490.9	46.0
27	Faicchio	FAI17	2017	Campania	456266.2	4571494.2	266.4	18.5
28	Monte Massico	MMA17	2017	Campania	411659.7	4562586.1	241.8	16.9
29	Pignataro Maggiore	PMA17	2017	Campania	434639.5	4560217.3	494.6	35.7
30	Monte Petrinp	MPE17	2017	Campania	408238.0	4555517.6	435.1	28.4
31	Polvica	POL17	2017	Campania	458147.6	4537656.5	399.4	22.8
32	Chiaramonti	CMO07	2007	Sardegna	-26519.1	4526888.0	327.9	36.1
33	Porto Ottiolu	POT17	2017	Sardegna	50536.6	4523122.4	374.5	17.1
34	Alà dei Sardi Ludurru	ASL17	2017	Sardegna	25156.6	4520148.6	1965.2	69.0
35	Vesuvio	VES17	2017	Campania	451367.3	4518908.2	3168.4	101.6
36	Gravina di Puglia	GPU17	2017	Puglia	618530.5	4511655.4	1708.6	47.8
37	Roccapiemonte	RPI17	2017	Campania	475189.0	4511530.4	410.9	26.8
38	Castiglione dei Genovesi	CDG17	2017	Campania	483148.5	4507504.6	174.6	17.2
39	Colline di Giovi	CGI07	2007	Campania	484223.3	4507087.1	480.9	41.4
40	Cetara	CET07	2007	Campania	473047.6	4501920.3	365.8	25.1
41	Monte Faito	MFA17	2017	Campania	456292.4	4500313.6	593.5	45.4
42	Nuoro	NUO07	2007	Sardegna	12355.5	4482054.1	7473.5	187.4
43	Teggiano	TEG07	2007	Campania	542946.3	4468581.9	1429.7	46.1
44	Pattano	PTN17	2017	Campania	514116.5	4454511.7	146.6	14.8
45	Monte Corice	MCO07	2007	Campania	517280.2	4446427.1	510.4	39.4
46	Episcopia	EPI17	2017	Basilicata	594227.5	4439363.6	291.5	19.3
47	Policastro Bussentino	PBU17	2017	Campania	541917.3	4436298.1	206.7	21.6
48	Tortolì	TOR17	2017	Sardegna	36939.4	4435729.2	838.8	45.3
49	Arzana	ARZ07	2007	Sardegna	35097.3	4433708.7	1131.0	34.5
50	Acquafredda	AQF17	2017	Basilicata	558741.7	4430162.8	209.8	10.0
51	Tortora	TTA17	2017	Calabria	567052.8	4422844.2	498.3	32.1
52	Capo Frasca	CFR07	2007	Sardegna	-60962.2	4417392.5	220.4	13.6
53	Tortora Marina	TMA17	2017	Calabria	568939.3	4416909.7	504.4	45.2
54	Papasidero	PAP17	2017	Calabria	578411.0	4416090.9	260.2	11.8
55	Trebisacce	TRE17	2017	Calabria	628424.9	4415770.5	270.6	17.7
56	Mormanno	MOM17	2017	Calabria	582525.4	4415391.0	250.3	11.3
57	San Nicola Arcella	SNA17	2017	Calabria	572654.9	4410740.2	997.0	56.8
58	Arbus	ARB17	2017	Sardegna	-57199.9	4392178.2	2133.0	111.5
59	Mottafollone	MOT17	2017	Calabria	593090.9	4390358.4	470.3	29.7
60	Villasalto	VSA07	2007	Sardegna	18942.0	4385120.7	703.4	26.8
61	Sangineto	SGI07	2007	Calabria	589173.7	4383570.5	260.0	19.3
62	Cropalati	CRO17	2017	Calabria	647312.7	4380434.3	243.4	21.1
63	San Demetrio Corone	SDC17	2017	Calabria	617188.3	4377461.7	248.8	19.3
64	San MArco Argentano	SMG17	2017	Calabria	595767.1	4376994.1	451.1	23.9
65	Pietrapaola	PIP17	2017	Calabria	655836.0	4370830.8	279.5	23.1
66	Acri	ACR07	2007	Calabria	621122.1	4370648.1	865.5	53.6

67	Longobucco	LON17	2017	Calabria	635061.1	4369475.4	2815.2	107.7
68	Terravecchia	TER07	2007	Calabria	665623.0	4369291.6	187.3	14.4
69	Acri	ACR17	2017	Calabria	617743.8	4368395.1	165.3	11.9
70	Ortiano	ORT07	2007	Calabria	640337.0	4365599.4	365.9	22.8
71	Carbonia	CAR07	2007	Sardegna	-54567.6	4362763.1	208.9	11.6
72	Rose	ROS17	2017	Calabria	616746.3	4360870.7	1946.5	82.9
73	Portovesme	PVE07	2007	Sardegna	-68157.3	4360279.8	484.5	35.2
74	Rovito	ROV17	2017	Calabria	612138.3	4350329.3	209.7	13.4
75	Santadi	SAN07	2007	Sardegna	-41674.6	4347569.6	296.6	16.2
76	Domanico	DOM17	2017	Calabria	602649.9	4341306.9	186.7	17.2
77	Capo Teulada	CTE07	2007	Sardegna	-49018.7	4329501.1	295.0	19.1
78	Martirano Lombardo 2	ML217	2017	Calabria	606047.3	4329149.1	407.0	31.8
79	Altilla Grimaldi	AGR07	2007	Calabria	606277.1	4329103.0	1610.2	66.0
80	San Mango d'Aquino	SAQ07	2007	Calabria	601508.9	4326598.5	353.8	28.6
81	Martirano Lombardo 1	ML117	2017	Calabria	604026.0	4325483.2	287.8	16.2
82	Gizzeria	GIZ17	2017	Calabria	606633.1	4316290.2	244.3	12.5
83	Tropea	TRO17	2017	Calabria	578752.4	4280246.9	250.2	19.7
84	Zaccanopoli	ZAC17	2017	Calabria	580775.2	4277426.2	411.3	33.3
85	Badolato 1	BA107	2007	Calabria	629910.4	4269078.5	121.5	10.1
86	Badolato 2	BA207	2007	Calabria	630491.0	4265713.1	451.4	28.7
87	Camini	CAM17	2017	Calabria	628365.2	4254847.5	107.8	8.4
88	Canolo	CAN17	2017	Calabria	603330.6	4244040.7	411.8	26.6
89	Monte Ciccia	MCI07	2007	Sicilia	547351.5	4233890.4	1318.0	77.6
90	Solano	SOL07	2007	Calabria	567890.7	4232860.1	829.0	52.1
91	Messina	MES17	2017	Sicilia	547448.9	4232375.7	641.4	32.6
92	Cerasi	CER07	2007	Calabria	567101.1	4225995.0	347.5	33.8
93	Lo Zingaro	LZI17	2017	Sicilia	304167.3	4219148.4	3742.4	70.0
94	San Martino delle Scale	SMS17	2017	Sicilia	342913.0	4219129.4	583.9	31.8
95	Mosoforra	MOS17	2017	Calabria	565272.5	4218311.9	309.1	20.0
96	Patti	PAT07	2007	Sicilia	503453.6	4217703.2	1666.0	94.8
97	Sant'Angelo in Brolo	SAB07	2007	Sicilia	493938.9	4217239.1	1303.0	104.0
98	San Martino delle Scale	SMS07	2007	Sicilia	343091.8	4216261.6	2794.6	147.8
99	Scopello	SCO07	2007	Sicilia	303427.3	4214574.7	1116.5	43.3
100	San Marco d'Alunzio	SMA07	2007	Sicilia	474323.4	4212505.3	1511.4	71.6
101	Castellammare del Golfo	CGO17	2017	Sicilia	311730.8	4210089.2	651.4	26.6
102	Santo Stefano di Camastra	SSC07	2007	Sicilia	444380.5	4204933.4	1921.7	83.3
103	Marineo	MAR07	2007	Sicilia	357960.3	4202239.0	301.3	20.8
104	Cefalù	CEF07	2007	Sicilia	409037.8	4201629.1	5087.4	157.1
105	Segesta Calatafini	SEC17	2017	Sicilia	313762.0	4200797.4	138.2	11.2
106	Vita	VIT07	2007	Sicilia	305711.2	4193111.4	402.5	29.8
107	Castiglione di Sicilia	CDS07	2007	Sicilia	504046.2	4189172.5	554.9	25.8
108	Corleone	COR17	2017	Sicilia	356012.9	4188941.1	686.0	28.9
109	Passopisciaro	PPI17	2017	Sicilia	504079.5	4188934.3	288.9	15.6
110	Bisacquino	BIS17	2017	Sicilia	348342.7	4176412.1	953.4	36.2
111	Piazza Armerina 2	PA217	2017	Sicilia	440169.5	4144786.0	654.8	31.0
112	Piazza Armerina 1	PA117	2017	Sicilia	446298.9	4143963.5	3610.4	106.9
113	Chiaramonte Gulfi	CGU17	2017	Sicilia	473084.2	4097003.8	840.5	77.1

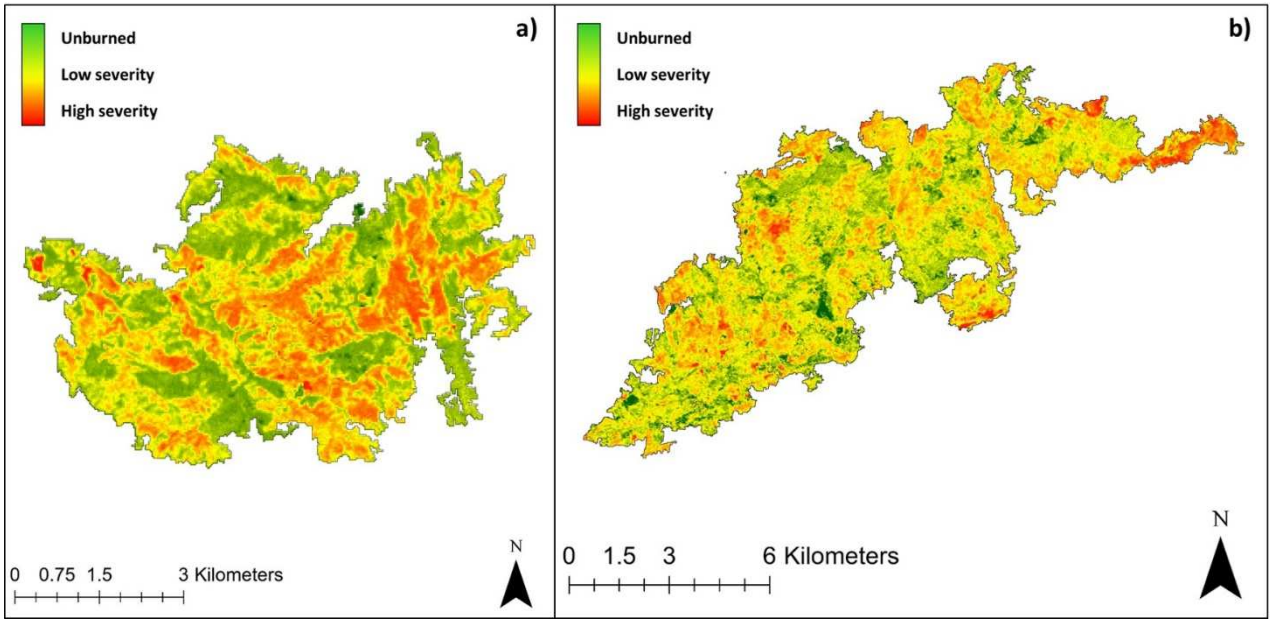

Figure S1 – Example of severity maps for a) Roccafluvione 2007 fire (ROC07 – Area = 3110.8 ha) and b) Nuoro 2007 fire (NUO07 – Area = 7473.5 ha). Green-yellow-red scale color defines the increasing severity (RdNBR) from unburned area (green) to high severity areas (red) within fire perimeter.

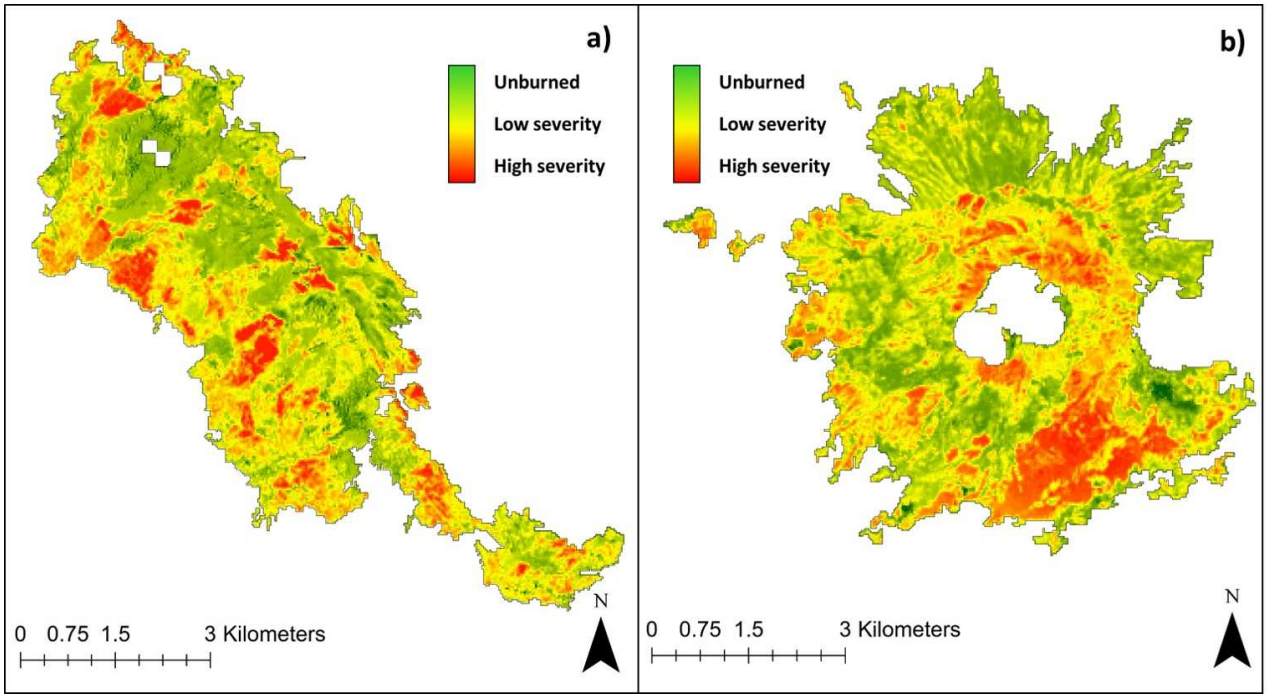

Figure S2 – Example of severity maps for a) Morrone 2017 fire (MOR17 – Area = 2775.3 ha) and b) Vesuvio 2017 fire (VES17 – Area = 3168.4 ha). Green-yellow-red scale color defines the increasing severity (RdNBR) from unburned area (green) to high severity areas (red) within fire perimeter.

CHAPTER 5

General conclusions

The research conducted during the three-years of doctorate focused especially on forest landscape changes and land use/cover shifts that occurred through the last decades in the Apennines and the possible influence of these changes on wildfires frequency and intensity. The study area is the second largest Italian mountain range that lacked a standardized research approach to assess land use dynamics due to the fragmentation of local case studies. We think to have contributed to this issue with the two published works presented in Chapter 2 and 3.

The significant socio-economic changes that occurred through 70 years caused similar dynamics along and across the Apennines: broadleaf forest-cover greatly increased by encroaching into former grasslands and croplands, with similar gains at NE and SW slope aspects but significantly greater at lower elevation, on steeper slopes and closer to existing forests. Landscape changes included two contrasting dynamics: simplification at lower elevation due to tree establishment and infilling in abandoned agro-pastoral lands and fragmentation of historical grasslands at higher elevation due to development of woody vegetation patches beyond the forest-grassland ecotone.

The increased fuel load due to woody vegetation expansion after reforestation and natural encroachment, together with warmer and especially drier conditions, could be a factor of increased fire risk. This second part of the study (Chapter 4) was presented at a recent national congress of the Italian Society of Forest Ecology and Silviculture and provided details on the effects of fires in the two most severe seasons (2007 and 2017). More work is in progress. Large forest fires are more frequent in southern regions. The average fire area was larger in 2007 in shrublands and broadleaf forests that were the land categories more widely burned in both years. We did not detect very relevant differences between fire features in 2007 and 2017, but conifers and shrublands were the most severely affected in both years. Climate change models are predicting an increase of fire hazard across the whole Mediterranean basin; therefore, we need predictive models to forecast fire severity in forest ecosystems at broader scales, especially in dry years. Forecasting wildfire hazard changes and impacts on forest ecosystems is essential for appropriate management strategies of fire prevention and control and of forest recovery. This will contribute to control or reduce the devastating effects of large forest fires on human lives, infrastructures and environmental resources.

In conclusion, I hope to have added a contribution to increase comprehension of forest landscape changes and their drivers in the Apennines range, the backbone of the Italian peninsula. The work done will also merge into a comprehensive and comparative study on land use/cover changes occurred

in Alps and the Apennines, in progress with colleagues of the University of Turin. The large effort to assemble from scratch a rich, homogenous and suitable fire database slowed down the application of appropriate models to the data extracted for predicting fire spread and severity in land cover and climate change scenario. The experience gained during these three years in Italy and abroad and the availability of large and partly unexplored data provide ideal conditions to continue and deepen my research activities beyond this PhD experience.

CHAPTER 6

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