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Frequent coppicing deteriorates the conservation status of black alder forests in the Po plain (northern Italy)

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Highlights

- Alluvial forests with black alder are a priority conservation habitat in Europe
- We assess whether frequent coppicing is compatible with their conservation
- We measured the effect of time since coppicing on forest structure and diversity
- Coppicing simplified vertical stand structure and the herbaceous layer for 20-30 years
- Non-native species were significantly more abundant in recently coppiced areas.

[1] GV wrote the paper and carried out statistical analyses, FM and GC carried out field sampling and forest structure analyses, MFe designed and coordinated the research and carried out imagery analyses, MFr carried out soil sampling and analyses, RM provided input for study design, interpretation and discussions, and ML carried out phytosociological analyses.

Abstract

Alluvial forests with black alder are a priority conservation habitat in Europe. In the Po plain, black alder is traditionally managed by coppicing with frequent rotations. This study aims to ascertain whether such management is compatible with habitat conservation, by measuring the effect of time since coppicing on forest structure and plant species composition across different layers.

We compared the effects of three treatments, each thrice replicated: recent (10-20 years), medium (20-30 years) and old coppice (>40 years). In all nine stands we measured basal area, tree and regeneration density, mean tree diameter and height, dominance by alder, species richness, Shannon diversity, and the number of ruderal and non-native species. Significant differences in dendrometric variables, species richness, diversity, and percent cover by chorotype were assessed for treatment effects by two-way ANOVA.

Frequently coppiced stands had a lower basal area, mean tree size, and volume, a more simplified vertical structure, a lower cover of the herbaceous layer and higher bare soil cover due to harvesting disturbance, a significantly lower cover by typical woodland *Fraxinetalia* species, and a significantly higher frequency and cover of non-native species.

Our study showed that frequent coppicing worsened the conservation status of black alder forests in the study area, simplified stand structure, deteriorated species composition, and increased the spread of non-native and ruderal plant species. Such negative effects persisted even 20-30 years after cutting. We recommend amending the current legislation and introducing mandatory Implications Assessment procedures everywhere alder forests are susceptible to be impacted in a similarly negative way.

Keywords: coppice, floodplain forests, forest management, Habitat Directive, non-native species, plant diversity, understory

1. Introduction

Black alder (*Alnus glutinosa* (L.) Gaertn.) is a tree species of riparian and water-logged habitats that is naturally widespread from mid-Scandinavia to southern Europe (Kajba et al. 2003). It forms pure stands on periodically submerged sites, while it mixes with ash (*Fraxinus excelsior* L.), maples (*Acer pseudoplatanus* L. and *Acer platanoides* L.) and oaks (mostly *Quercus robur* L.) on riverside and plateau sites (Dethioux 1974), where its intolerance to shading and lower groundwater tables reduce its ability to compete (Claessens et al. 2010).

Black alder grows between sea level and 1,300 on the Alps (Shaw et al. 2014). It is largely indifferent to soil parent material, but it requires precipitation above 510 mm per year and high water saturation (McVean 1953), and a high degree of atmospheric humidity throughout its reproductive cycle. When the water-table sinks below the surface during summer, tree growth increases but seedlings may suffer from drought (McVean 1953). The tree is able to fix atmospheric nitrogen in symbiotic root nodules (Bond et al. 1954), and its litter increases nitrogen and phosphorous content of the soil (Moiroud 1991, Giardina et al. 1995). The species has a maximum lifespan of 100 to 160 years (Claessens et al. 2010). It reaches sexual maturity at age 3-30, when it starts producing seeds with mast pulses every 3-4 years (Dethioux 1974). Seeds are dispersed by water or wind (up to 150 m: McVean 1955, but usually within 30 m: Funk 1990). However, regeneration occurs mostly from vegetative reproduction, e.g. in linear flood populations (Koop 1987, Deiller et al. 2003). Regeneration from seed is usually scattered and it occurs under favorable establishment conditions, e.g., on low-lying alluvial land or on former meadows (Douda et al. 2009). Seedlings require a higher light intensity than those of larger-seeded trees (McVean 1956); it was found that natural regeneration of black alder is not possible under the canopy of a mature stand (Tapper 1993), except in openings larger than 0.1 ha (Claessens et al. 2010). The regeneration of black alder also depends on the frequency and intensity of disturbance (e.g. browsers, floods, or forest harvesting) (Pokorný et al. 2000; Wolf et al. 2004), and on the abundance of herbs that may compete with the seedlings (McVean 1956).

Due to their specific hydrological regime and rare occurrence, black alder forests and carrs are considered an endangered forest community in Europe (Ellenberg 1996). Alluvial forests with black alder and ash are a priority habitat of Community interest listed in the Annex I of the Habitats Directive 92/43/EEC as 91E0* – Alluvial forests with *Alnus glutinosa* and *Fraxinus excelsior* (*Alno-Padion*, *Alnion incanae*, *Salicion albae*). These forests are highly important for the conservation of a great number of typical woodland and floodplain plant species (Claessens 2003), particularly when interspersed in an agricultural matrix. Despite being often small and fragmented (Schnitzler 1994), black alder forests are often characterized by a high richness in herbaceous species (Brown et al. 1997). Beyond plant diversity, black alder forests support other ecosystem services as well, such as water filtration and purification in waterlogged soils (Peterjohn and Correll 1984), flood control and riverbank stabilization (Piégay et al. 2003).

Currently, these forests represent less than 1% of the forest cover in most European countries (Claessens et al. 2010) due to both land use changes such as conversion to non-native tree plantations or agricultural land, or to environmental changes related to human activities, e.g. land draining, impact of industrial areas, negative selection in favor of more valuable timber species such as oak and ash, and the introduction of non-native species (EEA 2012). For these reasons, the conservation status of 91E0* habitat is currently “unfavourable inadequate” or “unfavourable bad” (Kremer et al. 2015).

In the Po plain, black alder is traditionally managed by coppicing, with rotations of 10 to 30 years due to the fact that the potential for vegetative regeneration from stumps declines at 60-80 years of age (Kapustinskaite 1960). Private ownership usually prevails in floodplain forests, with the consequence of creating a mosaic of small but intense and frequent cuttings, which can deteriorate habitat conservation and spatial continuity. Moreover, floodplain forests are highly vulnerable to plant invasions due to the frequent and intense natural disturbances, to their linear nature which facilitates long-distance species dispersal, and to intensive human pressure (Richardson et al. 2007). In this perspective, the question arises whether such management is compatible with habitat conservation.

This study aims to ascertain the effect of time since coppicing on the conservation status of black alder stands, as measured by (a) forest composition, structure, and biomass, and (b) species composition and naturalness of the herbaceous layer, e.g., the relative frequency of ruderal and non-native herbaceous species.

2. Study area

The study was conducted in the Natura 2000 site "IT1110021 - Laghi di Ivrea" (Figure 1), a 1600-ha Site of Community Importance (SCI) at the center of the 500-km^2 Ivrea Morainic Amphitheater (IMA). Mean annual temperature and annual precipitation are 12.5°C and 1002 mm, respectively (years 1921-2000) (Andreone et al. 2001). The bedrock is a juxtapositions of three metamorphic units (eclogitic micaschists, basic granulites and vulcanites) as a result of uplift and underplating during the Tertiary Alpine orogenesis (Johnson 1973). The Morainic Amphitheater was constructed between the Pleistocene and the Last Glacial Maximum (Carraro et al. 1974). Thereafter, small lakes formed in the gaps between secondary moraines, but most later evolved into peat bogs or were artificially drained. Such low-elevation sites are characterized today by Endoaquepts or Haplosaprists soils (Piazzi et al. 2007). The latter is predominant in peat and raised bogs, where the sapric organic material has an extremely slow hydraulic conductivity and C/N ratios may be as high as 45.

The site hosts 11 habitat types of the EU Habitats Directive (1992/43/EEC Annex I), among which the priority habitat 91E0* covers 59 hectares. A total of 32 plant and animal species of the EU Nature Directives (1992/43/EEC and 2009/147/EC Annex II) (Natura 2000 Network Viewer 2016). Anthropic pressure has caused the number of plant species to decline from 179 to 160 species between 1950 and 2005; at least 12 non-native plant species were reported in the area so far (Minuzzo et al. 2005; Lonati et al. 2014).

Forests are mostly owned by small private owners. Between January 2012 and June 2015, 40 silvicultural treatments were authorized across 8 hectares of 91E0* forests inside the site; 10% of this area was treated by thinning, 30% by coppicing, and 60% by contemporary cutting of the coppice and high forest layers (Regione Piemonte 2016a).

3. Methods

We designed the study as a chronosequence of stands coppiced in three different times: recent (10-20 years, TR1), medium (20-30 years, TR2) and old coppicing (>40 years, TR0). To do so, we preliminarily assigned one of such treatments to all forest stands classified as 91E0* habitats (according to Andreone et al. 2001) within the study area, based on the analysis of repeated aerial images (years 1954, 1975, 1979, 1994-1996, 1998-1999, 2007, 2009). The images were orthorectified and georeferenced, then visually classified into forested / nonforested categories, and differentiated to obtain age ranges for each forest stand. Age classes were subsequently confirmed by field surveys and exploratory increment core sampling. Only stands belonging to the association *Carici remotae-Fraxinetum* Koch ex Faber 1926 (alliance *Alnion incanae* Pawłowski in Pawłowski and Wallisch 1928) and already existing in year 1954 were considered for further analysis, i.e., waterlogged stands of the alliance *Alnion glutinosae* Malcuit 1929 and secondary stands on former non-forested land were filtered out.

Following superposition to cadastrial stand maps, we identified three independent study areas where all three elements of the chronosequence could be found in stands less than 100 m apart from one another, in order to minimize site differences between treatments and counter pseudoreplication. The only three areas where this condition was met in all the SCI are indicated in Figure 1. A total of nine stands (i.e., 3 study areas x 3 treatments) were selected for analysis; stands were at a constant elevation (about 240 m a.s.l.) and had a mean size of 1120 m^2.

In spring 2015, in each stand we randomly established a circular sampling plot (radius = 10 m) where we recorded species, frequency, diameter at breast heigh (dbh), origin (seed or sucker) and height of all adult trees with dbh >=7.5 cm. We also recorded species, frequency, origin, and height of all juvenile trees (dbh <7.5 cm) in a concentric 6-m radius circular plot. From plot data we computed common descriptors of stand structure (species composition, number of trees per hectare, basal area, quadratic mean diameter, average and top height, percent trees originated from seed) and compared them across treatments by Mann-Whitney test.

Within each sampling plot we randomly established five understory subplots (radius = 2 m) where we visually assessed percent cover of upper tree (height >15 m), lower tree (height between 5 and 15 m), upper shrub (height between 1.3 and 5 m), lower shrub, herbaceous and bare soil layers, and assigned cover-abundance scores (Braun-Blanquet 1932) to all plant species by each layer. At the center of each understory subplot, we measured canopy cover by taking a hemispherical photograph at 1m height above the ground. Hemispherical photographs were shot in Nikon .NEF format at 400 ISO with a 6 Megapixel Nikon D70S equipped with a Samyang 8mm f/3.5 aspherical IF MC Fisheye Lens set at shutter priority (time = 1/500 s). Canopy cover was determined by averaging the ratio of white to total image pixels obtained by each of four global thresholding algorithms (Otsu 1979; Huang and Wang 1995; Yen et al. 1995; Li and Tam 1998) for the Fiji image analysis software (Schindelin et al. 2012), applied to the blue band of each photograph.

In order to check for the absence of significant edaphic differences, three topsoil samples were extracted from the center of each understory subplot at a depth of 0-10 cm. All samples were air-dried and sieved (< 2 mm). Total carbon (corresponding to total organic carbon, TOC, thanks to the absence of carbonates) and nitrogen (TN) were analyzed by dry combustion with a CN elemental analyzer (CE Instruments NA2100, Rodano, Italy).

For each understory subplot we computed species richness (total number of species), the Shannon diversity index, and the number of non-native species based on chorotype (according to Celesti-Grapow et al. 2009). We associated a phytosociological optimum (according to Aeschimann et al. 2004) to each vascular plant species at the class level, including all subordinated syntaxa (Lonati et al. 2013; Orlandi et al. 2016; Pittarello et al. 2016) (Online Resource 1), and computed the number of species belonging to the following phytosociological groups, listed in order of increasing conservation value: ruderals (classes *Artemisietea vulgaris* and *Bidentetetea tripartitae*), tall herbs (classes *Filpendulo-Convolvuletea* and *Molinio-Arrhenatheretea*), shrubs (classes *Crataego-Prunetea*, *Franguletea* and *Salicetea purpureae*), and European alder forests (order *Fraxinetalia*). After converting cover-abundance data to mid-percent values ('+' in 0.3%; '1' in 2.8%; '2a' in 10.0%; '2b' in 20.5%; '3' in 38.0%; '4' in 63.0%; '5' in 88.0%) (Tasser and Tappeiner 2005), we computed total percent cover of non-native species and of each phytosociological group for each subplot.

All variables were log- (for continuous data) or arcsin- transformed (for percent data) to ensure homoskedasticity and normality of error distributions, and assessed for differences as a function of treatment by 2-way ANOVA with study area as a random factor and using Tukey's Honest Significant Difference (HSD) post-hoc test. All analyses were carried out in the SPSS 20.0 statistical environment (IBM Corporation 2011).

4. Results

All stands exhibited high tree densities (1000 to 2300 trees per hectare), with a large variability within treatments but a thinning trend as time since coppicing increased (Figure 2). Conversely, basal area, mean diameter, and volume increase with time since coppicing, up to yields of 450 m^3 ha^{-1} in stands harvested >40 years before sampling. Seedling density was highly variable, between 260 and 7000 per hectare, mostly originated from seed and predominantly by ash (64%), but decreased with increasing stand age (Figure 3). Canopy cover declined with increasing stand age (83%, 78%, and 74% respectively in TR1, TR2 and TR0, $p < 0.01$ with mixed-model ANOVA), possibly due to a structural change from a dense coppice to a high-forest with larger but sparser trees.

In all treatments, alder occupied preferentially the dominant vegetation layer, while ash was found in all layers. However, frequent coppicing simplified vertical stand structure. Relative to TR1 and TR2, old coppices showed a differentiation in two distinct tree layers, the upper dominated by alder, and the lower by ash (Table 1). In both medium and recently coppiced stands, dominant trees were still competing with each other within the same vegetation layer (<15 m height), even after 20-30 years. Moreover, recently coppiced stands (TR1) had a lower herbaceous and higher bare soil cover, likely due to the use of machines to transport harvested woods in the stand (as tracks observed in the field clearly showed).

Soils were rich in organic carbon and showed presence of gley. TOC%, TN%, and C/N ratio varied in the range of 4.5 to 19.3, 0.36 to 1.59, and 11.4 to 14.7, respectively, but they did not differ significantly between treatments (Table 2). Therefore, we ruled out topographic or edaphic effects in determining understory species composition.

Recently coppiced stands (TR1) had a higher total species richness, and TR1 and TR2 a higher Shannon diversity, than undisturbed stands (Table 3). However, this did not result in an increased naturalness of plant species composition. In fact, the number of *Fraxinetalia* species (*Carex remota*, *Carex brizoides*, *Carex pendula*, *Impatiens noli-tangere*, *Ribes rubrum*, *Equisetum telmateja*) remained substantially low (3 species per plot) and unchanged along the chronosequence, while their cover decreased significantly from 80% in TR0, to 65% in TR2, and 49% in TR1.

Conversely, the frequency and cover of non-native species were significantly higher in both medium and recently coppiced stands than in TR0 (Table 3). All 10 non-native species sampled in the study areas (*Acer negundo, Acer palmatum, Juglans regia, Ligustrum sinensis, Paulownia tomentosa, Robinia pseudoacacia, Lonicera japonica, Parthenocissus quinquefolia, Duchesnea indica, Oxalis fontana*) were found only in the two latter stages of the chronosequence (Appendix 1). These areas also showed a significantly higher number or ruderal nitrophilous species (*Urtica dioica, Alliaria petiolata, Geum urbanum, Geranium robertianum, Oplismenus undulatifolius, Galeopsis pubescens, Polygonum hydropiper*), which lower the biodiversity value of alder stands and whose cover was relatively high (10%) even 20-30 years after treatment. Also the number and cover of species more typically occurring in mown and fertilized meadows or in hydrophylous herb communities (*Poa trivialis, Rubus caesius, Humulus lupulus, Filipendula ulmaria, Lythrum salicaria*), rather than in alder forests, were higher in recently and medium vs. old coppices (36% and 41% in TR1 and TR2, 5% in TR0). The same was true also for shrub species (*Viburnum opulus, Cornus sanguinea, Corylus avellana, Crataegus monogyna, Euonymus europaeus, Ligustrum vulgare, Frangula alnus, Salix cinerea*), as they can take advantage from higher light levels in recently opened gaps. Shrub cover was still high (41%) after 20-30 years from coppicing, showing the magnitude and long lasting legacy of silvicultural treatments on stand structure and light conditions.

5. Discussion

5.1 Forest structure and dynamics

The yield of black alder in Europe at age 80 is between 500 and 1000 m^3 ha^{-1} (Lockow 1995; Sopp 1974). Old coppice stands included in this study approached the lower end of this range. The average yield of black alder in the forest district where the study was carried out was 148 m^3 ha^{-1} (Bertani et al. 2003), which is indicative of the relative rarity of undisturbed mature alder forests in the area. Contrary to mountain areas, management of coppices in lowland sites of the Po plain is still quite active, due to their high accessibility and the predominantly private ownership.

In our study, seedling density decreased with increasing stand age. This confirms that alder regeneration by seed is difficult in closed stands, as seedlings are very intolerant of shading and rank herbaceous vegetation (Latham and Blackstock 1998). Herbaceous competitors (e.g. *Carex* spp.) were more abundant soon after coppicing (see below) and may have prevented alder seedling development (McVean 1956). Should alder seedlings take advantage of the higher light levels from canopy opening, establishment would decline soon after coppicing (Ash and Barkham 1976). At the same time, intense coppicing with few or no standards favored ash regeneration, both from seed and from faster sprout growth. Ash seedlings and sprouts may grow very slowly if in shade for several years, but are able to respond with rapid growth when gaps appear in the canopy (Merton 1970; Tapper 1992, 1993; Gatsuk et al. 1980). Once in the canopy, ash is able to overgrow and suppress alder through competition for light (Tapper 1996).

Finally, biomass, mean tree size, and complexity of vertical structure (i.e., number of tree layers) declined with increasing coppicing frequency. These effects persisted up to 30 years after treatment, and may result in a loss of habitats and lower spatial heterogeneity of resources (Motta et al. 2015) that can hamper diversity and ecosystem stability. A more diverse array of tree sizes and the existence of multiple vertical layers instead may provide a greater number of potential ecological niches for a wide array of relevant forest biota, such as invertabrates, birds, lichens (e.g. Müller et al. 2005; Czeszczewik and Walankiewicz 2006; Nascimbene et al. 2013; Negro et al. 2015).

5.2 Effects of coppice management on habitat conservation

Many herb species typically occurring in woodland communities are perennials and can persist throughout the coppice cycle. Under a coppice regime, the relatively constant and saturated set of niches that is found in high forests is replaced by a variety of others, which are filled by species capable of survival in a relatively wider range of ecological circumstances (van der Werf 1991). Sometimes, the greatest threat to plant diversity is the abandonment of coppicing, whereby many open-habitat species are slowly outshadowed under the ever denser growing crowns (Baeten et al. 2009; Negro et al. 2015).

However, when the conservation target is the forest habitat, as in the case of black alder remnants, too frequent coppicing can be detrimental. The first agent of damage is harvesting machinery, which may induce compaction, rutting and stripping, irreversibly reduce soil fertility and drainage, and alter species composition, especially on loamy soils (Decocq et al. 2005). Altered drainage may then produce habitat deterioration for other species of conservation interest such as aquatic invertebrates, insects and amphibians (e.g., *Pelobates fuscus insubricus*: Andreone et al. 2004). Second, frequent coppicing alters light and moisture regimes, which facilitates ruderal, nitrophilous, tall herbs and non-native species (Funk et al. 2008) that can outcompete woodland species of interest. The post-coppicing communities analyzed by this study contained species adapted to many types of canopy and soil disturbance and relatively high light levels such as ruderal species, nitrophilous species (e.g., *Poa trivialis*, *Urtica dioica*, *Rubus caesius*) (Honnay et al. 1999; De Keersmaeker et al. 2004), tall herbs, and shrubs. In undisturbed stands, germination of these species is prevented by low light levels and the abundant leaf litter (Sydes and Grime 1981). But if this vegetation component increases due to the high coppicing frequency, it may progressively hamper specialist woodland herbs by competitive exclusion (Hipps et al. 2005). Canopy removal during coppicing may increase the amount of solar radiation reaching the soil, the quantity and composition of light, the temperature, humidity, evaporation and mineralization rates. Seeds of many open-habitat species require a high temperature regime for germination and it is apparent that this occurs only when there is no vegetation present to cast shade, such as in the first two years after coppicing. Moreover, the decrease of C and N in recently coppiced stands might indicate an effect of coppicing frequency on organic matter recycling and mineralization. Soils of undisturbed forests have been previously found to be richer in C and N than in intensely harvested ones (Johnson and Curtis 2001; Finér et al. 2003), including in alder carr (Honnay et al. 1999; Verheyen et al. 1999; Dzwonko 2001; Falkengren-Grerup et al. 2006; Orczewska 2009). This may result from a more prolonged accumulation of litter that, in the case of alder, has a high N content and rate of decomposition (Karkanis 1975; Pereira et al. 1998). However, also the opposite was found, i.e., a higher N availability in regularly harvested stands, due to a higher organic matter turnover rate (Covington 1981; Kimmins 1987; Koerner et al. 1997; Keersmaeker et al. 2004). Canopy removal increases soil temperature (Carlson and Groot 1997; Pennock and Kessel 1997; Hashimoto and Suzuki 2004), decreases soil water content (Ma et al. 2013) and shortens saturation periods, therefore promoting a faster mineralization in harvested stands. Such different findings may depend on regional climate (Yin et al. 1989), soil fertility, former agricultural use (Compton and Boone 2000), its duration, and time since abandonment.

The picture is further complicated by invasion of non-native species. Changes in the hydrologic cycle of humid forests, induced e.g. by upstream water use, canopy cover changes, or climate change, can further facilitate invasion by species adapted to drier conditions (Huston 2004). Moreover, floodplain forests remnants can be more vulnerable to plant invasions due to their small size and high perimeter-to-area ratio, which facilitates species colonization from the margins and makes any canopy disturbance a potential threat to native species. Among the non-native species found by this study in alder stands, some have a well-documented high degree of invasiveness, e.g., *Acer negundo* L., *Robinia pseudoacacia* L., *Lonicera japonica* Thunb. (Regione Piemonte 2015a), while for some others this is less certain. However, also the latter ones have been previously reported in other areas of Piedmont region, indicating that their naturalization potential and invasiveness are likely higher than expected, e.g., *Ligustrum sinense* (Lonati et al. 2014; Soldano et al. 2015) or *Paulownia tomentosa* (Selvaggi 2014). Once established, non-native species can persist due to their high regeneration potential (also by vegetative reproduction), high degree of adaptation to disturbances, allelopathy, and influx of non-native seeds in the soil seed bank (Lorenzo et al. 2013; Gioria and Pyšek 2015).

For all these reasons, we recommend amending the current legislation and introducing mandatory Implications Assessment procedures everywhere alder forests are susceptible to be impacted by coppicing in the negative ways suggested by our analyses. Should coppice need to be maintained, best silvicultural practices should include higher retention of living and dead biomass, longer rotations (Della Rocca et al. 2014), and cessation of all drainage activities, as a high water level inhibits the vigorous growth of expansive, nutrient-demanding species and reduces the competitive exclusion of woodland flora by such herbs. Promoting shadier conditions in the forest floor may also limit the expansion of ruderal and non-native plant species, and facilitate the immigration and establishment of typical woodland herbs (Orczewska 2009).

6. Conclusion

Many species and habitats associated with natural forested floodplains have disappeared from most of Europe. Current silvicultural practices, especially in small private woodlots, may increase pressures on biodiversity and ecosystem conservation. Our study showed that frequent coppicing has negative consequences for the conservation of black alder forests in the western part of the Po Plain (northern Italy), inducing a simplification of stand structure, a deterioration of species composition, and the spread of non-native plant species. Such negative effects persisted even 20-30 years after cutting, suggesting that the cumulative effects of repeated frequent coppicing would be characterized by even harsher impacts.

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(https://www.change.org/p/salviamo-la-ricerca-italiana) for the increase of research funding in Italy to the levels requested by the EU Lisbon strategy.

Conflict of Interest

The authors declare that they have no conflict of interest.

Appendices

Additional supporting information in the online version of this article (see “Supplementary Material”) contains the following: ESM_1 - List of native and non-native species, phytosociological optimum (according to Aeschimann et al. 2004), species frequency (%), and minimum and maximum cover (cover-abundance scores according to Braun-Blanquet 1932)

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Tables

Table 1. Percent cover (mean ± standard error) of different vegetation layers (all species, black alder and ash) by treatment. Different letters indicate significant differences between treatments at $p < 0.10$ (ANOVA with Tukey's HSD test).

% cover of vegetation layer	TR1 (10-15 years)		TR2 (20-30 years)		TR0 (>40 years)	
Upper tree	0 ± 0.0	a	0 ± 0.0	a	82 ± 2.3	b
Lower tree	83 ± 1.8	b	87 ± 1.2	b	19 ± 1.7	a
Upper shrub	21 ± 4.2	b	16 ± 2.1	ab	10 ± 1.4	a
Lower shrub	41 ± 3.1	c	26 ± 2.4	b	14 ± 1.4	a
Herbaceous	78 ± 3.4	a	93 ± 0.9	b	91 ± 1.1	b
Bare soil	22 ± 3.4	b	7 ± 0.9	a	9 ± 1.1	a
Alnus glutinosa						
Total	45 ± 3.0	a	61 ± 4.5	b	78 ± 3.3	c
Upper tree	0 ± 0.0	a	0 ± 0.0	a	78 ± 3.3	b
Lower tree	45 ± 3.0	b	61 ± 4.5	c	0 ± 0.0	a
Upper shrub	0 ± 0.3	a	0 ± 0.0	a	0 ± 0.0	a
Lower shrub	-		-		-	
Fraxinus excelsior						
Total	66 ± 5.5	b	55 ± 6.7	ab	48 ± 3.7	a
Upper tree	0 ± 0.0	a	0 ± 0.0	a	3 ± 1.6	b
Lower tree	44 ± 4.7	b	30 ± 5.0	a	21 ± 2.2	a
Upper shrub	18 ± 3.5	a	16 ± 2.2	a	11 ± 1.4	a
Lower shrub	4 ± 1.2	a	9 ± 1.6	b	12 ± 1.3	b

Table 2. Soil data by treatment in the study area (means ± standard error). Different letters indicate significant differences between treatments at $p < 0.10$ (ANOVA with Tukey's HSD test).

Variable	TR1 (8-10 years)	TR2 (20-30 years)	TR0 (>40 years)
TN%	0.8 ± 0.08 a	0.9 ± 0.07 a	1.0 ± 0.07 a
TOC%	10.5 ± 1.05 a	11.8 ± 0.84 a	13.0 ± 0.81 a
C/N	12.6 ± 0.24 a	12.8 ± 0.17 a	12.9 ± 0.2 a

Table 3. Diversity, richness, and cover of functional groups (mean ± standard error) by treatment. Different letters indicate significant differences between treatments at $p < 0.05$ (ANOVA with Tukey's HSD test).

% cover of vegetation layer	**TR1 (10-15 years)**		**TR2 (20-30 years)**		**TR0 (>40 years)**	
Total richness	15.4 ± 0.50	b	12.7 ± 0.67	a	11.3 ± 0.69	a
Shannon index (H)	3.0 ± 0.06	c	2.7 ± 0.09	b	2.1 ± 0.05	a
Species number						
Non-native species	1.7 ± 0.27	c	0.9 ± 0.19	b	0.0 ± 0.00	a
Ruderal species	1.3 ± 0.12	b	1.1 ± 0.19	b	0.5 ± 0.13	a
Hydrophilous tall herb species	2.3 ± 0.21	b	3.1 ± 0.22	b	2.1 ± 0.09	a
Shrub species	2.8 ± 0.31	b	1.4 ± 0.21	a	1.9 ± 0.36	a
Fraxinetalia species	3.0 ± 0.22	a	2.9 ± 0.19	a	3.0 ± 0.24	a
Percent cover						
Non-native species	15 ± 2.0	c	4 ± 1.2	b	0 ± 0.0	a
Ruderal species	15 ± 2.4	c	10 ± 2.4	b	0 ± 0.2	a
Tall herb species	36 ± 4.6	b	41 ± 4.6	b	5 ± 1.0	a
Shrub species	16 ± 2.4	b	6 ± 1.4	a	3 ± 0.9	a
Fraxinetalia species	49 ± 3.4	a	65 ± 3.3	b	80 ± 2.9	c

Figure captions

Fig. 1 Location of the study areas in Piedmont, Italy (left: blue – Special Protection Areas, red – Sites of Community Importance, green – Ramsar sites) and within the Site of Community importance “Laghi di Ivrea” (right).

Fig. 2 Stand structural variables in the study areas by treatment. TR0: old (>40 years), TR1: medium (10-20 years), TR2: recent coppice (20-30 years). Different letters indicate significant differences between treatments at $p < 0.10$ (Mann-Whitney test).

Fig. 3 Regeneration (individuals per hectare) in the study areas by treatment. TR0: old (>40 years), TR1: medium (10-20 years), TR2: recent coppice (20-30 years). Different letters indicate significant differences between treatments at $p < 0.10$ (Mann-Whitney test).

revised Figure 1

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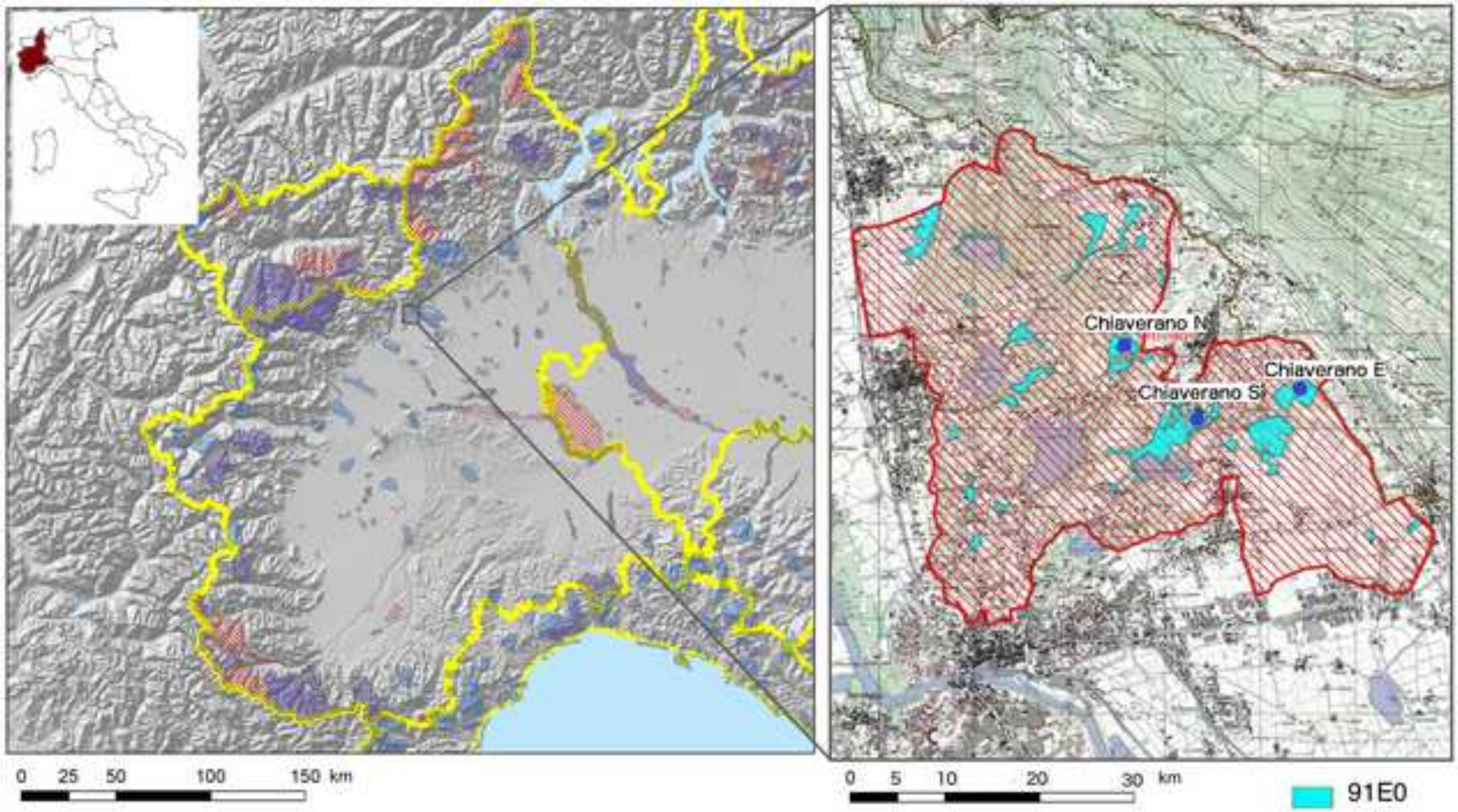

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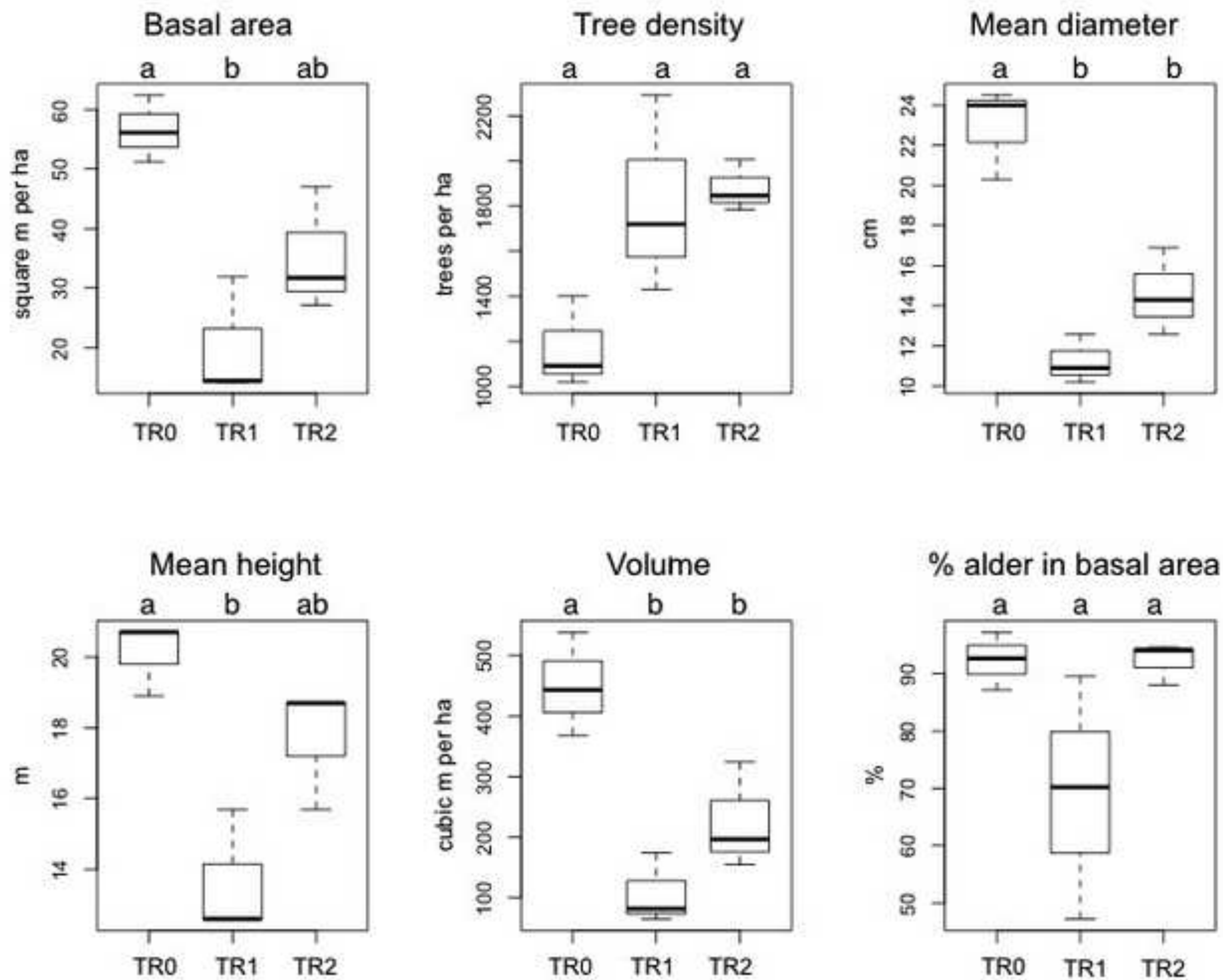

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