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(Article begins on next page)



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

3

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13

14 **Highlights**

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

20

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.

21 **Abstract**

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

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

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

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

43

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

46

47 **1. Introduction**

48

49 Black alder (*Alnus glutinosa* (L.) Gaertn.) is a tree species of riparian and water-logged
50 habitats that is naturally widespread from mid-Scandinavia to southern Europe (Kajba et al.
51 2003). It forms pure stands on periodically submerged sites, while it mixes with ash
52 (*Fraxinus excelsior* L.), maples (*Acer pseudoplatanus* L. and *Acer platanoides* L.) and oaks
53 (mostly *Quercus robur* L.) on riverside and plateau sites (Dethioux 1974), where its
54 intolerance to shading and lower groundwater tables reduce its ability to compete (Claessens
55 et al. 2010).

56

57 Black alder grows between sea level and 1,300 on the Alps (Shaw et al. 2014). It is largely
58 indifferent to soil parent material, but it requires precipitation above 510 mm per year and
59 high water saturation (McVean 1953), and a high degree of atmospheric humidity throughout
60 its reproductive cycle. When the water-table sinks below the surface during summer, tree
61 growth increases but seedlings may suffer from drought (McVean 1953). The tree is able to
62 fix atmospheric nitrogen in symbiotic root nodules (Bond et al. 1954), and its litter increases
63 nitrogen and phosphorous content of the soil (Moiroud 1991, Giardina et al. 1995). The
64 species has a maximum lifespan of 100 to 160 years (Claessens et al. 2010). It reaches sexual
65 maturity at age 3-30, when it starts producing seeds with mast pulses every 3-4 years
66 (Dethioux 1974). Seeds are dispersed by water or wind (up to 150 m: McVean 1955, but
67 usually within 30 m: Funk 1990). However, regeneration occurs mostly from vegetative
68 reproduction, e.g. in linear flood populations (Koop 1987, Deiller et al. 2003). Regeneration
69 from seed is usually scattered and it occurs under favorable establishment conditions, e.g., on
70 low-lying alluvial land or on former meadows (Douda et al. 2009). Seedlings require a higher

71 light intensity than those of larger-seeded trees (McVean 1956); it was found that natural
72 regeneration of black alder is not possible under the canopy of a mature stand (Tapper 1993),
73 except in openings larger than 0.1 ha (Claessens et al. 2010). The regeneration of black alder
74 also depends on the frequency and intensity of disturbance (e.g. browsers, floods, or forest
75 harvesting) (Pokorný et al. 2000; Wolf et al. 2004), and on the abundance of herbs that may
76 compete with the seedlings (McVean 1956).

77

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

90

91 Currently, these forests represent less than 1% of the forest cover in most European countries
92 (Claessens et al. 2010) due to both land use changes such as conversion to non-native tree
93 plantations or agricultural land, or to environmental changes related to human activities, e.g.
94 land draining, impact of industrial areas, negative selection in favor of more valuable timber
95 species such as oak and ash, and the introduction of non-native species (EEA 2012). For

96 these reasons, the conservation status of 91E0* habitat is currently “unfavourable inadequate”
97 or “unfavourable bad” (Kremer et al. 2015).

98

99 In the Po plain, black alder is traditionally managed by coppicing, with rotations of 10 to 30
100 years due to the fact that the potential for vegetative regeneration from stumps declines at 60-
101 80 years of age (Kapustinskaite 1960). Private ownership usually prevails in floodplain
102 forests, with the consequence of creating a mosaic of small but intense and frequent cuttings,
103 which can deteriorate habitat conservation and spatial continuity. Moreover, floodplain
104 forests are highly vulnerable to plant invasions due to the frequent and intense natural
105 disturbances, to their linear nature which facilitates long-distance species dispersal, and to
106 intensive human pressure (Richardson et al. 2007). In this perspective, the question arises
107 whether such management is compatible with habitat conservation.

108

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

113

114 **2. Study area**

115

116 The study was conducted in the Natura 2000 site "IT1110021 - Laghi di Ivrea" (Figure 1), a
117 1600-ha Site of Community Importance (SCI) at the center of the 500-km² Ivrea Morainic
118 Amphitheater (IMA). Mean annual temperature and annual precipitation are 12.5°C and 1002
119 mm, respectively (years 1921-2000) (Andreone et al. 2001). The bedrock is a juxtapositions
120 of three metamorphic units (eclogitic micaschists, basic granulites and vulcanites) as a result

121 of uplift and underplating during the Tertiary Alpine orogenesis (Johnson 1973). The
122 Morainic Amphitheater was constructed between the Pleistocene and the Last Glacial
123 Maximum (Carraro et al. 1974). Thereafter, small lakes formed in the gaps between
124 secondary moraines, but most later evolved into peat bogs or were artificially drained. Such
125 low-elevation sites are characterized today by Endoaquepts or Haplosaprists soils (Piazzi et
126 al. 2007). The latter is predominant in peat and raised bogs, where the sapric organic material
127 has an extremely slow hydraulic conductivity and C/N ratios may be as high as 45.

128

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

135

136 Forests are mostly owned by small private owners. Between January 2012 and June 2015, 40
137 silvicultural treatments were authorized across 8 hectares of 91E0* forests inside the site;
138 10% of this area was treated by thinning, 30% by coppicing, and 60% by contemporary
139 cutting of the coppice and high forest layers (Regione Piemonte 2016a).

140

141 **3. Methods**

142

143 We designed the study as a chronosequence of stands coppiced in three different times: recent
144 (10-20 years, TR1), medium (20-30 years, TR2) and old coppicing (>40 years, TR0). To do
145 so, we preliminarily assigned one of such treatments to all forest stands classified as 91E0*

146 habitats (according to Andreone et al. 2001) within the study area, based on the analysis of
147 repeated aerial images (years 1954, 1975, 1979, 1994-1996, 1998-1999, 2007, 2009). The
148 images were orthorectified and georeferenced, then visually classified into forested /
149 nonforested categories, and differentiated to obtain age ranges for each forest stand. Age
150 classes were subsequently confirmed by field surveys and exploratory increment core
151 sampling. Only stands belonging to the association *Carici remotae-Fraxinetum* Koch ex
152 Faber 1926 (alliance *Alnion incanae* Pawłowski in Pawłowski and Wallisch 1928) and
153 already existing in year 1954 were considered for further analysis, i.e., waterlogged stands of
154 the alliance *Alnion glutinosae* Malcuit 1929 and secondary stands on former non-forested
155 land were filtered out.

156

157 Following superposition to cadastral stand maps, we identified three independent study areas
158 where all three elements of the chronosequence could be found in stands less than 100 m
159 apart from one another, in order to minimize site differences between treatments and counter
160 pseudoreplication. The only three areas where this condition was met in all the SCI are
161 indicated in Figure 1. A total of nine stands (i.e., 3 study areas x 3 treatments) were selected
162 for analysis; stands were at a constant elevation (about 240 m a.s.l.) and had a mean size of
163 1120 m².

164

165 In spring 2015, in each stand we randomly established a circular sampling plot (radius = 10
166 m) where we recorded species, frequency, diameter at breast height (dbh), origin (seed or
167 sucker) and height of all adult trees with dbh ≥ 7.5 cm. We also recorded species, frequency,
168 origin, and height of all juvenile trees (dbh < 7.5 cm) in a concentric 6-m radius circular plot.
169 From plot data we computed common descriptors of stand structure (species composition,
170 number of trees per hectare, basal area, quadratic mean diameter, average and top height,

171 percent trees originated from seed) and compared them across treatments by Mann-Whitney
172 test.

173

174 Within each sampling plot we randomly established five understory subplots (radius = 2 m)
175 where we visually assessed percent cover of upper tree (height >15 m), lower tree (height
176 between 5 and 15 m), upper shrub (height between 1.3 and 5 m), lower shrub, herbaceous and
177 bare soil layers, and assigned cover-abundance scores (Braun-Blanquet 1932) to all plant
178 species by each layer. At the center of each understory subplot, we measured canopy cover
179 by taking a hemispherical photograph at 1m height above the ground. Hemispherical
180 photographs were shot in Nikon .NEF format at 400 ISO with a 6 Megapixel Nikon D70S
181 equipped with a Samyang 8mm f/3.5 aspherical IF MC Fisheye Lens set at shutter priority
182 (time = 1/500 s). Canopy cover was determined by averaging the ratio of white to total image
183 pixels obtained by each of four global thresholding algorithms (Otsu 1979; Huang and Wang
184 1995; Yen et al. 1995; Li and Tam 1998) for the Fiji image analysis software (Schindelin et
185 al. 2012), applied to the blue band of each photograph.

186

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

192

193 For each understory subplot we computed species richness (total number of species), the
194 Shannon diversity index, and the number of non-native species based on chorotype
195 (according to Celesti-Grapow et al. 2009). We associated a phytosociological optimum

196 (according to Aeschimann et al. 2004) to each vascular plant species at the class level,
197 including all subordinated syntaxa (Lonati et al. 2013; Orlandi et al. 2016; Pittarello et al.
198 2016) (Online Resource 1), and computed the number of species belonging to the following
199 phytosociological groups, listed in order of increasing conservation value: ruderals (classes
200 *Artemisietea vulgaris* and *Bidentetetea tripartitae*), tall herbs (classes *Filpendulo-*
201 *Convolvuletea* and *Molinio-Arrhenatheretea*), shrubs (classes *Crataego-Prunetea*,
202 *Franguletea* and *Salicetea purpureae*), and European alder forests (order *Fraxinetalia*). After
203 converting cover-abundance data to mid-percent values ('+' in 0.3%; '1' in 2.8%; '2a' in
204 10.0%; '2b' in 20.5%; '3' in 38.0%; '4' in 63.0%; '5' in 88.0%) (Tasser and Tappeiner
205 2005), we computed total percent cover of non-native species and of each phytosociological
206 group for each subplot.

207

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

213

214 **4. Results**

215

216 All stands exhibited high tree densities (1000 to 2300 trees per hectare), with a large
217 variability within treatments but a thinning trend as time since coppicing increased (Figure 2).
218 Conversely, basal area, mean diameter, and volume increase with time since coppicing, up to
219 yields of 450 m³ ha⁻¹ in stands harvested >40 years before sampling. Seedling density was
220 highly variable, between 260 and 7000 per hectare, mostly originated from seed and

221 predominantly by ash (64%), but decreased with increasing stand age (Figure 3). Canopy
222 cover declined with increasing stand age (83%, 78%, and 74% respectively in TR1, TR2 and
223 TR0, $p < 0.01$ with mixed-model ANOVA), possibly due to a structural change from a dense
224 coppice to a high-forest with larger but sparser trees.

225

226 In all treatments, alder occupied preferentially the dominant vegetation layer, while ash was
227 found in all layers. However, frequent coppicing simplified vertical stand structure. Relative
228 to TR1 and TR2, old coppices showed a differentiation in two distinct tree layers, the upper
229 dominated by alder, and the lower by ash (Table 1). In both medium and recently coppiced
230 stands, dominant trees were still competing with each other within the same vegetation layer
231 (<15 m height), even after 20-30 years. Moreover, recently coppiced stands (TR1) had a
232 lower herbaceous and higher bare soil cover, likely due to the use of machines to transport
233 harvested woods in the stand (as tracks observed in the field clearly showed).

234

235 Soils were rich in organic carbon and showed presence of gley. TOC%, TN%, and C/N ratio
236 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
237 differ significantly between treatments (Table 2). Therefore, we ruled out topographic or
238 edaphic effects in determining understory species composition.

239

240 Recently coppiced stands (TR1) had a higher total species richness, and TR1 and TR2 a
241 higher Shannon diversity, than undisturbed stands (Table 3). However, this did not result in
242 an increased naturalness of plant species composition. In fact, the number of *Fraxinetalia*
243 species (*Carex remota*, *Carex brizoides*, *Carex pendula*, *Impatiens noli-tangere*, *Ribes*
244 *rubrum*, *Equisetum telmateja*) remained substantially low (3 species per plot) and unchanged

245 along the chronosequence, while their cover decreased significantly from 80% in TR0, to
246 65% in TR2, and 49% in TR1.
247
248 Conversely, the frequency and cover of non-native species were significantly higher in both
249 medium and recently coppiced stands than in TR0 (Table 3). All 10 non-native species
250 sampled in the study areas (*Acer negundo*, *Acer palmatum*, *Juglans regia*, *Ligustrum sinensis*,
251 *Paulownia tomentosa*, *Robinia pseudoacacia*, *Lonicera japonica*, *Parthenocissus*
252 *quinquefolia*, *Duchesnea indica*, *Oxalis fontana*) were found only in the two latter stages of
253 the chronosequence (Appendix 1). These areas also showed a significantly higher number or
254 ruderal nitrophilous species (*Urtica dioica*, *Alliaria petiolata*, *Geum urbanum*, *Geranium*
255 *robertianum*, *Oplismenus undulatifolius*, *Galeopsis pubescens*, *Polygonum hydropiper*),
256 which lower the biodiversity value of alder stands and whose cover was relatively high (10%)
257 even 20-30 years after treatment. Also the number and cover of species more typically
258 occurring in mown and fertilized meadows or in hydrophyllous herb communities (*Poa*
259 *trivialis*, *Rubus caesius*, *Humulus lupulus*, *Filipendula ulmaria*, *Lythrum salicaria*), rather
260 than in alder forests, were higher in recently and medium vs. old coppices (36% and 41% in
261 TR1 and TR2, 5% in TR0). The same was true also for shrub species (*Viburnum opulus*,
262 *Cornus sanguinea*, *Corylus avellana*, *Crataegus monogyna*, *Euonymus europaeus*, *Ligustrum*
263 *vulgare*, *Frangula alnus*, *Salix cinerea*), as they can take advantage from higher light levels
264 in recently opened gaps. Shrub cover was still high (41%) after 20-30 years from coppicing,
265 showing the magnitude and long lasting legacy of silvicultural treatments on stand structure
266 and light conditions.

267

268 **5. Discussion**

269

270 5.1 Forest structure and dynamics

271 The yield of black alder in Europe at age 80 is between 500 and 1000 m³ ha⁻¹ (Lockow 1995;
272 Sopp 1974). Old coppice stands included in this study approached the lower end of this
273 range. The average yield of black alder in the forest district where the study was carried out
274 was 148 m³ ha⁻¹ (Bertani et al. 2003), which is indicative of the relative rarity of undisturbed
275 mature alder forests in the area. Contrary to mountain areas, management of coppices in
276 lowland sites of the Po plain is still quite active, due to their high accessibility and the
277 predominantly private ownership.

278

279 In our study, seedling density decreased with increasing stand age. This confirms that alder
280 regeneration by seed is difficult in closed stands, as seedlings are very intolerant of shading
281 and rank herbaceous vegetation (Latham and Blackstock 1998). Herbaceous competitors (e.g.
282 *Carex* spp.) were more abundant soon after coppicing (see below) and may have prevented
283 alder seedling development (McVean 1956). Should alder seedlings take advantage of the
284 higher light levels from canopy opening, establishment would decline soon after coppicing
285 (Ash and Barkham 1976). At the same time, intense coppicing with few or no standards
286 favored ash regeneration, both from seed and from faster sprout growth. Ash seedlings and
287 sprouts may grow very slowly if in shade for several years, but are able to respond with rapid
288 growth when gaps appear in the canopy (Merton 1970; Tapper 1992, 1993; Gatsuk et al.
289 1980). Once in the canopy, ash is able to overgrow and suppress alder through competition
290 for light (Tapper 1996).

291

292 Finally, biomass, mean tree size, and complexity of vertical structure (i.e., number of tree
293 layers) declined with increasing coppicing frequency. These effects persisted up to 30 years

294 after treatment, and may result in a loss of habitats and lower spatial heterogeneity of
295 resources (Motta et al. 2015) that can hamper diversity and ecosystem stability. A more
296 diverse array of tree sizes and the existence of multiple vertical layers instead may provide a
297 greater number of potential ecological niches for a wide array of relevant forest biota, such as
298 invertebrates, birds, lichens (e.g. Müller et al. 2005; Czeszczewik and Walankiewicz 2006;
299 Nascimbene et al. 2013; Negro et al. 2015).

300

301 *5.2 Effects of coppice management on habitat conservation*

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

309

310 However, when the conservation target is the forest habitat, as in the case of black alder
311 remnants, too frequent coppicing can be detrimental. The first agent of damage is harvesting
312 machinery, which may induce compaction, rutting and stripping, irreversibly reduce soil
313 fertility and drainage, and alter species composition, especially on loamy soils (Decocq et al.
314 2005). Altered drainage may then produce habitat deterioration for other species of
315 conservation interest such as aquatic invertebrates, insects and amphibians (e.g., *Pelobates*
316 *fuscus insubricus*: Andreone et al. 2004). Second, frequent coppicing alters light and moisture
317 regimes, which facilitates ruderal, nitrophilous, tall herbs and non-native species (Funk et al.
318 2008) that can outcompete woodland species of interest. The post-coppicing communities

319 analyzed by this study contained species adapted to many types of canopy and soil
320 disturbance and relatively high light levels such as ruderal species, nitrophilous species (e.g.,
321 *Poa trivialis*, *Urtica dioica*, *Rubus caesius*) (Honnay et al. 1999; De Keersmaecker et al.
322 2004), tall herbs, and shrubs. In undisturbed stands, germination of these species is prevented
323 by low light levels and the abundant leaf litter (Sydes and Grime 1981). But if this vegetation
324 component increases due to the high coppicing frequency, it may progressively hamper
325 specialist woodland herbs by competitive exclusion (Hipps et al. 2005). Canopy removal
326 during coppicing may increase the amount of solar radiation reaching the soil, the quantity
327 and composition of light, the temperature, humidity, evaporation and mineralization rates.
328 Seeds of many open-habitat species require a high temperature regime for germination and it
329 is apparent that this occurs only when there is no vegetation present to cast shade, such as in
330 the first two years after coppicing. Moreover, the decrease of C and N in recently coppiced
331 stands might indicate an effect of coppicing frequency on organic matter recycling and
332 mineralization. Soils of undisturbed forests have been previously found to be richer in C and
333 N than in intensely harvested ones (Johnson and Curtis 2001; Finér et al. 2003), including in
334 alder carr (Honnay et al. 1999; Verheyen et al. 1999; Dzwonko 2001; Falkengren-Grerup et
335 al. 2006; Orczewska 2009). This may result from a more prolonged accumulation of litter
336 that, in the case of alder, has a high N content and rate of decomposition (Karkanis 1975;
337 Pereira et al. 1998). However, also the opposite was found, i.e., a higher N availability in
338 regularly harvested stands, due to a higher organic matter turnover rate (Covington 1981;
339 Kimmins 1987; Koerner et al. 1997; Keersmaecker et al. 2004). Canopy removal increases soil
340 temperature (Carlson and Groot 1997; Pennock and Kessel 1997; Hashimoto and Suzuki
341 2004), decreases soil water content (Ma et al. 2013) and shortens saturation periods, therefore
342 promoting a faster mineralization in harvested stands. Such different findings may depend on

343 regional climate (Yin et al. 1989), soil fertility, former agricultural use (Compton and Boone
344 2000), its duration, and time since abandonment.

345

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

362

363 For all these reasons, we recommend amending the current legislation and introducing
364 mandatory Implications Assessment procedures everywhere alder forests are susceptible to be
365 impacted by coppicing in the negative ways suggested by our analyses. Should coppice need
366 to be maintained, best silvicultural practices should include higher retention of living and
367 dead biomass, longer rotations (Della Rocca et al. 2014), and cessation of all drainage

368 activities, as a high water level inhibits the vigorous growth of expansive, nutrient-demanding
369 species and reduces the competitive exclusion of woodland flora by such herbs. Promoting
370 shadier conditions in the forest floor may also limit the expansion of ruderal and non-native
371 plant species, and facilitate the immigration and establishment of typical woodland herbs
372 (Orczewska 2009).

373

374 **6. Conclusion**

375

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

384

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393 in Italy to the levels requested by the EU Lisbon strategy.

394

395 **Conflict of Interest**

396 The authors declare that they have no conflict of interest.

397

398 **Appendices**

399 Additional supporting information in the online version of this article (see “Supplementary
400 Material”) contains the following: ESM_1 - List of native and non-native species,
401 phytosociological optimum (according to Aeschimann et al. 2004), species frequency (%),
402 and minimum and maximum cover (cover-abundance scores according to Braun-Blanquet
403 1932)

404

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650 **Tables**

651

652 **Table 1.** Percent cover (mean \pm standard error) of different vegetation layers (all species,
 653 black alder and ash) by treatment. Different letters indicate significant differences between
 654 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)	
	Mean \pm SE	Letter	Mean \pm SE	Letter	Mean \pm SE	Letter
Upper tree	0 \pm 0.0	a	0 \pm 0.0	a	82 \pm 2.3	b
Lower tree	83 \pm 1.8	b	87 \pm 1.2	b	19 \pm 1.7	a
Upper shrub	21 \pm 4.2	b	16 \pm 2.1	ab	10 \pm 1.4	a
Lower shrub	41 \pm 3.1	c	26 \pm 2.4	b	14 \pm 1.4	a
Herbaceous	78 \pm 3.4	a	93 \pm 0.9	b	91 \pm 1.1	b
Bare soil	22 \pm 3.4	b	7 \pm 0.9	a	9 \pm 1.1	a
<i>Alnus glutinosa</i>						
Total	45 \pm 3.0	a	61 \pm 4.5	b	78 \pm 3.3	c
Upper tree	0 \pm 0.0	a	0 \pm 0.0	a	78 \pm 3.3	b
Lower tree	45 \pm 3.0	b	61 \pm 4.5	c	0 \pm 0.0	a
Upper shrub	0 \pm 0.3	a	0 \pm 0.0	a	0 \pm 0.0	a
Lower shrub	-		-		-	
<i>Fraxinus excelsior</i>						
Total	66 \pm 5.5	b	55 \pm 6.7	ab	48 \pm 3.7	a
Upper tree	0 \pm 0.0	a	0 \pm 0.0	a	3 \pm 1.6	b
Lower tree	44 \pm 4.7	b	30 \pm 5.0	a	21 \pm 2.2	a
Upper shrub	18 \pm 3.5	a	16 \pm 2.2	a	11 \pm 1.4	a
Lower shrub	4 \pm 1.2	a	9 \pm 1.6	b	12 \pm 1.3	b

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Table 2. Soil data by treatment in the study area (means \pm 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 \pm 0.08 a	0.9 \pm 0.07 a	1.0 \pm 0.07 a
TOC%	10.5 \pm 1.05 a	11.8 \pm 0.84 a	13.0 \pm 0.81 a
C/N	12.6 \pm 0.24 a	12.8 \pm 0.17 a	12.9 \pm 0.2 a

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Table 3. Diversity, richness, and cover of functional groups (mean \pm 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 \pm 0.50	b	12.7 \pm 0.67	a	11.3 \pm 0.69	a
Shannon index (H)	3.0 \pm 0.06	c	2.7 \pm 0.09	b	2.1 \pm 0.05	a
Species number						
Non-native species	1.7 \pm 0.27	c	0.9 \pm 0.19	b	0.0 \pm 0.00	a
Ruderal species	1.3 \pm 0.12	b	1.1 \pm 0.19	b	0.5 \pm 0.13	a
Hydrophilous tall herb species	2.3 \pm 0.21	b	3.1 \pm 0.22	b	2.1 \pm 0.09	a
Shrub species	2.8 \pm 0.31	b	1.4 \pm 0.21	a	1.9 \pm 0.36	a
<i>Fraxinetalia</i> species	3.0 \pm 0.22	a	2.9 \pm 0.19	a	3.0 \pm 0.24	a
Percent cover						
Non-native species	15 \pm 2.0	c	4 \pm 1.2	b	0 \pm 0.0	a
Ruderal species	15 \pm 2.4	c	10 \pm 2.4	b	0 \pm 0.2	a
Tall herb species	36 \pm 4.6	b	41 \pm 4.6	b	5 \pm 1.0	a
Shrub species	16 \pm 2.4	b	6 \pm 1.4	a	3 \pm 0.9	a
<i>Fraxinetalia</i> species	49 \pm 3.4	a	65 \pm 3.3	b	80 \pm 2.9	c

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682 **Figure captions**

683

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

687

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

691

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

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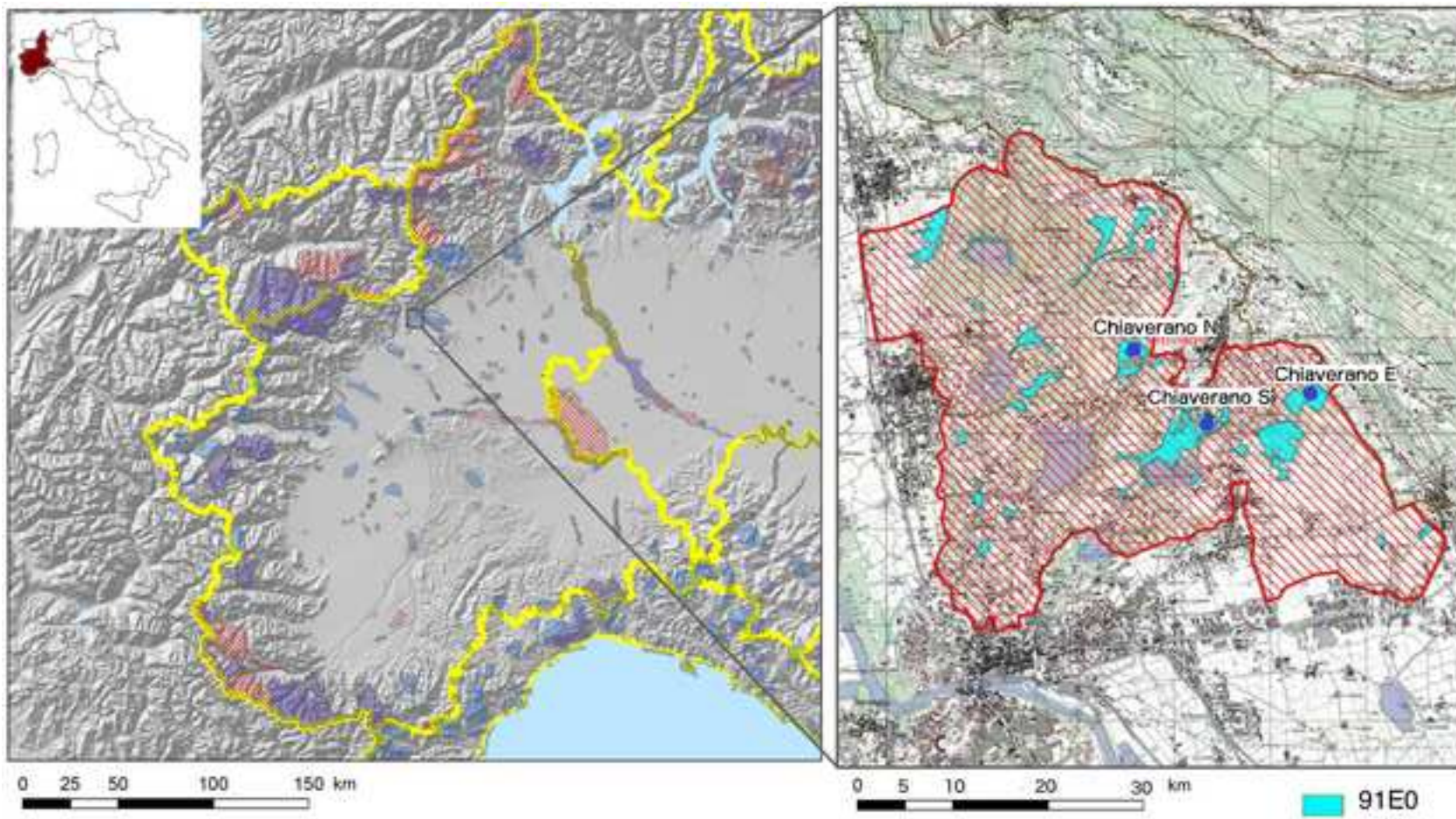


Figure 2

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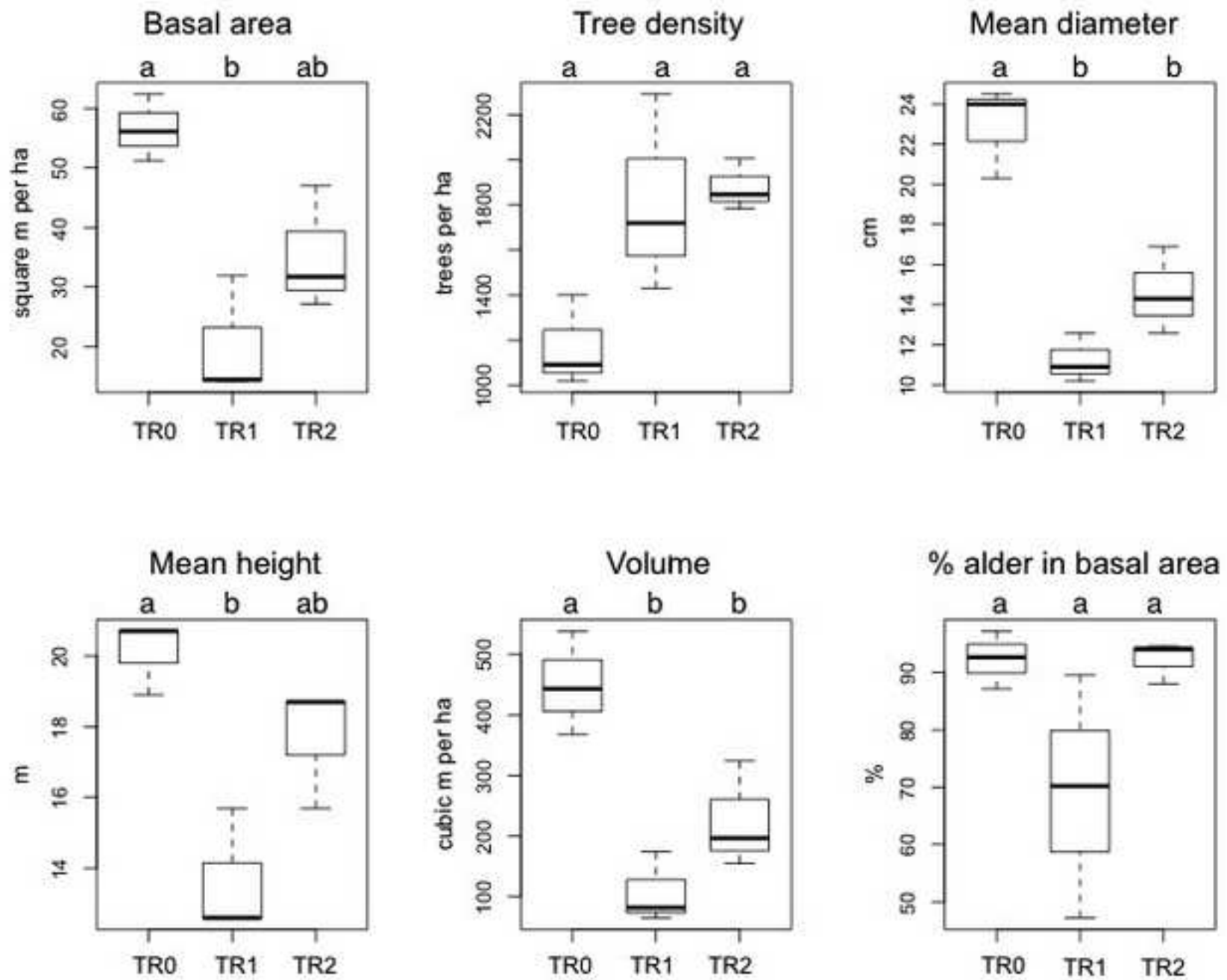
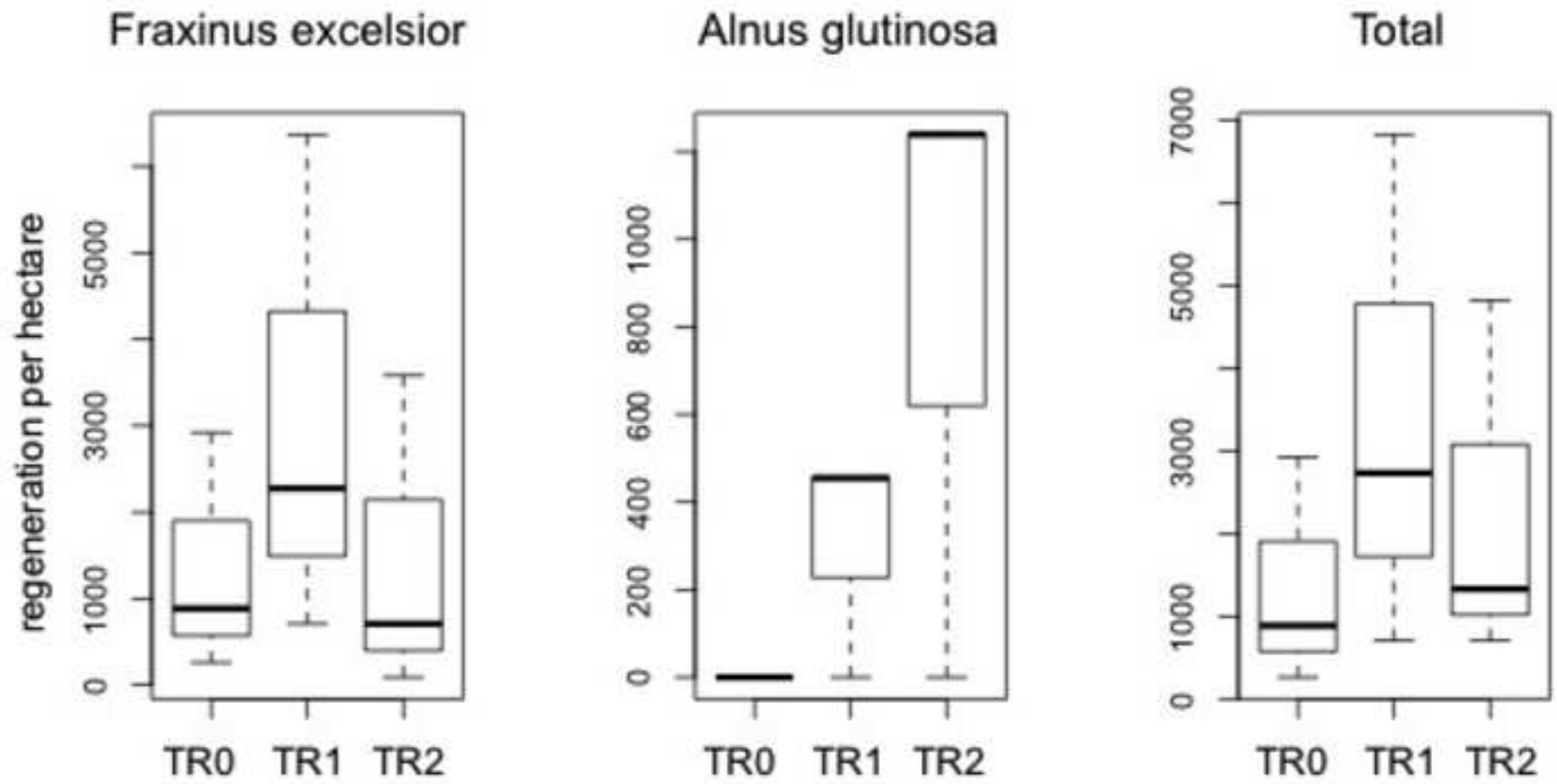


Figure 3
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