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6

7 **SCALE DEPENDENT EFFECTS OF COPPICING ON THE SPECIES POOL**
8 **OF LATE-SUCCESSIONAL BEECH FOREST**
9

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32 **Running head:** Scale dependent effects of coppicing on diversity
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43 **Abstract**

44 **Question:** We study the effect of traditional coppicing and its abandonment on community
45 structure and plant diversity at various scales. The aim is to compare active and abandoned beech
46 coppices in terms of: (i) structural features (ii) total, understory and overstory plant diversity across
47 a range of spatial scales (iii) species richness of beech forest specialist species.

48 **Location:** Monti Sibillini, Central Apennines, Italy.

49 **Methods:** We applied a multiscale approach working in parallel at fine scale (individual
50 management units, 0.5-1 ha), and at coarser scale (the whole forest patch). Two forests patches
51 were selected according to management regimes (active, 1-40 years, and abandoned coppicing, >40
52 years) in each of the two study areas (Bolognola and Ussita). Individual management units were
53 sampled with 400mq plots; each forest patch was represented by 20 plots arranged in grids.

54 **Results:** We found significant differences in structural features between coppiced and abandoned
55 forests. At plot scale, only the number of beech forest specialist species was significantly higher in
56 abandoned plots. At the forest patch scale, the Bolognola actively coppiced forests hosts many more
57 species than the abandoned ones, but an opposite result was found at Ussita. The abandoned forests
58 show a constant higher beech forest specialist species richness at Ussita, while in Bolognola we
59 found just the opposite. The managed forests host more overstory species than abandoned ones in
60 both areas. Surprisingly the total beech forest specialist species richness was higher in the coppiced
61 plots with respect to the abandoned ones. The species assemblages were more similar between
62 patches having different management regimes within the same area, than between patches having
63 the same management regime across different areas.

64 **Conclusions:** This material provides a novel contribution to the study of species diversity patterns
65 in this forest system, suggesting the importance of a multiple scale approach in forest diversity
66 studies. The beech forest specialist species can largely persist in a heterogeneous coppice landscape,
67 where abandoned stands are mixed with stands under a regular coppicing. The results can link the
68 existing knowledge of beech coppice diversity between landscape- and plot-scale, helping to
69 address new conservation planning.

70

71

72 **Keywords:** Scale-dependence; landscape; alpha diversity; beta diversity; Central Apennines; forest
73 stand structure

74 **Introduction**

75 In Mediterranean and temperate Europe, coppicing still represents a widespread method of forest
76 resource exploitation (Peterken 1981; Rackham 2008). This management system relies on the
77 resprouting capacity of some tree species after coppicing. Some of the trees are not cut during
78 coppicing (the so called “standards”) and serve as a seed sources and forest floor protection. These
79 standards are left in the coppice with density dependent planned on different variables. Then,
80 standards are cut on a longer rotation. The rotation period of cutting in Mediterranean coppices is
81 about 15-30 years. Traditionally, coppice systems provide firewood and charcoal but are also
82 grazing areas for livestock (Peterken 1981; Amorini & Fabbio 2001).

83

84 In Italy, about 3.9 million ha of forests were managed with coppicing (Ciancio et al. 2006). In the
85 central-Appennines the woodlands dominated by *Fagus sylvatica* L. were largely coppiced until the
86 last century (Pavari 1999). In the last decades, in parallel with socio-economic changes (e.g.
87 abandonment of the countryside, see Antrop 2004; Agnoletti 2007; Rackham 2008), the coppicing
88 significantly declined in Mediterranean Europe (Barbero et al. 1990; Evans & Barkham 1992;
89 Coppini & Hermanin 2007), and a significant demise of this forest management is also documented
90 for temperate Europe (Van Calster et al. 2007; Hedl et al. 2010). This has made the process of
91 coppicing abandonment a very relevant ecological process in most of southern and Mediterranean
92 Europe.

93

94 In the abandonment of coppiced oak woodlands, Debussche et al. (2001) observed a cover decrease
95 in lower height classes of woody stems and an increase in cover and height of the uppermost woody
96 layer, resulting into a simplified vertical structure due to the competition for light. Similarly, a
97 pronounced dominant canopy cover shift was observed from shrubs to trees by Van Calster et al.
98 (2008) in a *Fagus sylvatica* dominated high forest. The regeneration process after logging of a
99 coppiced forest often causes a decrease of plant diversity at the stand scale (Debussche et al. 2001;
100 Mason & MacDonald 2002; Decocq et al. 2004; Bartha et al. 2008). In a mixed forest of Central
101 Europe (mainly oak-hornbeam forest), Kopecky et al. (2013) demonstrated that the dominant
102 process after abandonment of coppicing was the non-random extinction of light-demanding species,
103 leading to an impoverished, temporally nested plant community structure.

104

105 The consequences of coppice abandonment is in line with a worldwide simplification of forest
106 structure and function (McKinney & Lockwood 1999; Ares et al. 2010), generating a progressive
107 taxonomic impoverishment and homogenization of the woodland vegetation (Keith et al. 2009). In
108 light to these evidences, ecological studies need to explore these issues because plant species
109 assemblages change rapidly under human impact (Hedl et al. 2010; Kopecky et al. 2013).

110

111 Although the herbaceous layer contains more than 80% of the total plant species richness, this layer
112 is often an under-considered component of forest ecosystems (Gilliam 2007). However, its effects
113 on tree productivity and long-term stability of the ecosystem have been demonstrated, as well as its
114 influence on the composition, microbiological processes and nutrient cycling of the organic layer
115 (Clement 2001; Gilliam 2007; Salemaa et al. 2008). Herbaceous species also contribute
116 significantly to biodiversity and particularly important for this are those plants which are late
117 successional beech forest species (beech forest “specialist species”). These species usually
118 characterize the forest mature phases and ancient woodlands (Gilliam 2007; Bartha et al. 2008),
119 adapted to the relatively stable environmental conditions of forests (Hermy et al. 1999). The role of
120 forest management in preserving the *specialist species* pool at different spatial scales is still poorly
121 evaluated (Bartha et al. 2008). These species can be used as indicators of biodiversity, and in this
122 respect are important functional components for their habitat and resource specificity (Gilliam
123 2007). Recent studies performed in beech forests of the Italian Apennines (Bartha et al. 2008;
124 Campetella et al. 2011) demonstrated that, at the stand scale, the abundance contribution of the
125 beech forest *specialist species* doubled after about 40–60 years after coppicing.

126

127 Species diversity depends considerably on spatial scale (Podani et al. 1993; Palmer & White 1994;
128 Crawley & Harral 2001). Diversity studies usually target coarse (landscapes – several km²) or small
129 (plots – m²) scales (Brockway 1998; Chiarucci et al. 2008a; Bartha et al. 2008) and therefore “in-
130 between” scales are often unexplored. This is a significant gap because management practices are
131 often carried out at intermediate scales and this is likely to have an important effect on species
132 diversity. Moreover, forest fragmentation represents a widespread process (Wulder et al. 2009;
133 Albuquerque & Rueda 2010), and consequently species diversity analysis at the forest patch level is
134 of crucial importance. Although plant biodiversity has become a key element of Central European
135 forest management policy, there is still a limited number of studies examining the influence of
136 coppice abandonment on biodiversity (Kopecky et al. 2013) and very few of them used large dataset
137 (Battles et al. 2001). Information on Mediterranean parts of Europe is largely missing and a study
138 which compares coppice forest units to abandoned ones can contribute to disentangle the ecological
139 processes behind the forest coppice abandonment.

140

141 The present study investigates plant species diversity in relation to coppice abandonment, on a
142 range of spatial scales, from the plot to the whole stand. Our overall objective was to compare the
143 structure and diversity of coppiced and abandoned beech forests both at the plot (20 m x 20 m) and
144 the entire forest patch (minimum size of 80 ha) scale in the Italian Apennine, by using a well-
145 balanced probabilistic sampling design. Our specific research questions were to compare active and

146 abandoned beech coppices in terms of: (i) structural features (ii) total, understory and overstory
147 plant diversity across a range of spatial scales (iii) species richness of beech forest *specialist*
148 *species*.

149

150

151 **Materials and methods**

152 *Study area*

153 The study area is located in the central-Appennines, in the Monti Sibillini National Park (southern
154 Marche Region, Italy). The area belongs to the bio-climatic ‘warm’ *Fagus* forest zone (800-1000 m
155 to 1500 m a.s.l., as proposed in the central Appenines by Piovesan et al. (2005), where the mean
156 annual precipitation ranges between 1100 and 1400 mm and the mean annual temperature is
157 between 8 and 12°C (Biondi & Baldoni 1995). The bedrock is mainly Mesozoic and Tertiary
158 limestone. The soils (Skeleti-Calcaric Phaeozems and Calcari-Humic Leptosols) are weakly
159 structured, shallow and have low water holding capacity, with medium-high content of carbonates
160 (10-40%) loamy texture, and pH ranging from neutral to sub-alkaline (ASSAM 2006). According to
161 the phytosociological classification, these beech forests belong to two associations of the alliance
162 *Geranio striati-Fagion*, i.e. the *Cardamino kitaibelii-Fagetum sylvaticae* and the *Lathyro veneti-*
163 *Fagetum sylvaticae*.

164

165 *Sampling design*

166 The study was based on a well-balanced sampling design. Several criteria were used to select forest
167 patches differing only in current management regime, namely “active coppice” or “abandoned
168 coppice”, but maintaining all the other ecological factors as constant as possible. To do this, a forest
169 management map was used (IPLA - Istituto per la Pianta da Legno e l’Ambiente 2001). Four
170 forest patches were selected in two different areas, with management regimes (active and
171 abandoned coppicing) in each area. Each forest patch was located on limestone bedrock on NW-N-
172 NE facing slopes, and had a minimum size of 80 *ha*, considered enough to represent the mean forest
173 patch size occurring in the region. The two abandoned coppice patches mostly derived from the
174 abandonment of coppices or by the coppice conversion into high forest. One pair of management
175 regime (“active’ / “abandoned”) was in the Bolognola municipality, the other in the Ussita
176 municipality.

177 To represent the plant species composition and structural attributes at the plot and forest patch
178 scales without sampling bias (Chiarucci 2007), a probabilistic sampling design was realized by
179 placing 20 *m* x 20 *m* plots in the nodes of a grid with cells of 200 *m* x 200 *m*, oriented in N-S and E-
180 W directions (Figure 1). This distance was chosen to sample the highest level of heterogeneity,
181 since the single size of units coppiced in a single event is usually between 0.5-2 *ha* (Canullo et al.

182 2009). The grid was virtually overlaid on each of our selected forest patches starting by a random
183 point. The south east corner of each plot was positioned at each grid node, after its location by
184 Garmin Vista Etrex GPS.

185 To minimize edge effects, plots were placed at a minimum distance of 20 m from roads or forest
186 edges. Plots falling into non-*Fagus facies* were placed elsewhere or rejected. Since the resulting
187 number of plots in each forest patch was between 20 and 23, they were standardized to 20 by
188 removing those plots located in most marginal sites. This resulted into a sample made of 20 plots in
189 each of the four forest patches, two for each area and each 2 management regimes (active and
190 abandoned).

191 In each plot, all vascular plant species were recorded and their cover visually assessed. For
192 structural data, in each plot the cover of emergent trees (individuals with crown emerging over the
193 dominant tree layer), dominant trees (trees composing the crown dominant layer), subordinate trees
194 (trees > 5 m, with crown totally or partially dominated from the upper canopies), shrubs (woody
195 individuals < 5 m), herbs, total plant cover, deadwood cover (i.e. coarse woody debris, stumps),
196 litter and bare soil were also visually assessed. The first four variables mentioned above are
197 considered as vertical physiognomic forest layers. Each individual plant with DBH > 2 cm was
198 censused, taxonomically identified and grouped into one physiognomic forest layer. In each plot all
199 the individuals released as “standards” were also recorded and their height (we used Vertex
200 hypsometer, Haglof Inc., Madison, Miss.) and DBH were measured. Field works were carried out
201 from June to August 2008.

202 Beech forest *specialist species* (Bartha et al. 2008) were classified considering their common habitat
203 preference, taking into account their similar phytocoenological role (Borhidi 1995; Nagaike et al.
204 2003), as determined by regional synecology and their actual role in the local floras (assigned from
205 Pignatti 1982, 1998; Pignatti et al. 2004; Ballelli et al. 2005). The complete list of the beech forest
206 *specialist species* identified in this study is provided in the Supplementary material 1.

207

208 *Statistical analysis*

209 The diversity of woody species at the plot scale was quantified by using the Shannon index (H') on
210 the basis of the number of individuals with DBH > 2 cm.

211 The effects of management and forest patches on the structural variables (number of standards,
212 standard diameter, standard height, emergent trees density, dominant trees density, subordinated
213 trees density, shrubs density, total woody density, total vegetation cover, herb layer cover,
214 deadwood cover) and diversity variables (vascular species richness, overstory species richness,
215 understory species richness, beech forest *specialist species* richness, Shannon index H') was
216 analyzed by nested ANOVA statistics (Linear Mixed Models), with management as a fixed factor
217 and forest patches nested within the management type. A correction for multiple testing was carried

218 out to avoid inflating the significance of individual tests ($p \leq 0.001$).
219 We addressed the problem of the spatial autocorrelation of error, as this can result in spurious
220 relationships (type I error). For this purpose, we tested for autocorrelation of the models error by
221 using a randomization test. Namely, we plotted the experimental variogram of the models' residuals
222 and produced a 'random envelope' by permuting the coordinates of residual values 999 times while
223 re-computing the variogram at each step. The maximum and minimum value of the semi-variance
224 obtained for each lag provided the 'random envelope'. If the observed variogram values were not
225 within such envelope, this was taken as evidence of autocorrelation and therefore we fitted models
226 with exponentially or spherically correlated spatial error to account for spatial non-independence of
227 the residuals.

228 We pooled plots belonging to different forest patches to obtain the diversity measures at the scale of
229 the entire forest. The species richness (total-, understory-, overstory-) was compared by sample
230 (plot) rarefaction curves (Gotelli & Colwell 2001), obtained by using the analytical formula
231 (Kobayashi 1974; Chiarucci et al. 2008b).

232 To test the variability in community composition among sites and between management regimes,
233 diversity was multiplicatively partitioned (Baselga 2010) according to the multi-level sampling
234 hierarchy: plot, forest patch, municipality, forest management type. Forest patches were defined on
235 basis of the two management regimes and were tested for difference by comparing variation in
236 species composition between patches within the same municipality and, between patches within the
237 same forest management type that belonged to different municipalities. To partition the total
238 diversity of vascular plants, we calculated different measures of forest diversity (Table 1). The beta-
239 diversity was calculated according to the Whittaker multiplicative formula, $\beta = \gamma/\alpha$
240 (Whittaker 1972; Baselga 2010). All the analyses were performed in R environmental (R Core
241 Team 2014), using the "vegan" package (Oksanen et al. 2015).

242

243 **Results**

244 *Forest stand structure at the plot scale*

245 A total of 5.446 individuals of woody species with $DBH > 2 \text{ cm}$ were censused. As expected, the
246 mean standards height was significantly higher in the abandoned coppices than in active ones
247 (Table 2), while the density of emergent trees and shrubs were significantly higher in active
248 coppices. As an effect of nested forest areas, standards density, standards diameter and standards
249 height were higher in the abandoned ones (Table 2). Interestingly, total woody individuals density
250 with $DBH > 2 \text{ cm}$ showed no significant difference among the two management types (Table 2).

251 Results for the variables describing the overstory and understory layers evidenced (Table 2) that the
252 two management regimes had similar plant cover (almost 90%, on average). The analysis of the
253 spatial autocorrelation of error shows that all the observed variogram values occurred within the

254 random envelope, supporting the absence of autocorrelation among the different sampling units in
255 the adopted sampling design (see Supplementary material 2).

256

257 *Diversity - at the plot scale*

258 A total of 291 vascular plant species were found (more than the 10 % of the flora of Marche region,
259 see Conti et al 2005) in the 80 plots. The mean species richness of vascular plants, the Shannon
260 index of diversity of woody individuals and the species richness of understory plants did not differ
261 between the two management types and the two areas. On the other hand, the species richness of
262 overstory species resulted statistically different between treatments and areas, being higher in active
263 coppices and in the Bolognola area (Table 3). The mean species richness of beech forest specialists
264 was significantly higher in the abandoned plots as compared to the forest still actively managed
265 (Table 3).

266

267 *Diversity - at forest patch scale*

268 The rarefaction curves for the 20 plots in each forest patch (Figure 2) do not suggest an asymptotic
269 accumulation of total species richness. At Bolognola, the actively coppiced forest hosts many more
270 species than the abandoned one, while an opposite pattern, even if with a lower difference, was
271 found at Ussita. The pattern of overstory species was similar in both areas, with the managed forests
272 hosting many more species than abandoned ones, while the patterns of understory species showed
273 similar difference as the total species. The patterns of beech forest *specialist species* was rather
274 complex and divergent between the two areas. At Ussita, abandoned coppices had constantly higher
275 species richness of beech specialists at all the sample sizes (from 1 to 20 plots), while at Bolognola
276 abandoned coppices had higher species richness of beech specialists only at smaller sample sizes
277 (up to 10 plots) and managed coppices had higher species richness at larger sample sizes.

278

279 *Diversity partitioning*

280 Multiplicative partitioning of beta diversity into hierarchical levels showed discordant results in the
281 two study areas in term of number of species and species composition. In fact, at the plot scale, the
282 number of species was higher in abandoned patch than actively coppiced forest in Ussita, but it was
283 higher in actively coppiced forest than in abandoned forest in Bolognola. On the other hand the
284 variability in species composition within each patch was higher within coppiced forest in Bolognola
285 but little higher within abandoned forest in Ussita (Table 4).

286 At patch level, although there were more species in coppiced patches than abandoned ones, the
287 variability in species composition between patches within the same municipality was slightly lower
288 than the variability measured between patches within the same forest management type (Table 5).
289 In other words, species assemblage were more similar between patches having different forest

290 management regimes within the same area, than between patches having the same forest
291 management regime across different areas.

292 The same trend was shown by considering differences in beta diversity at the scale of forest
293 management type and at the scale of municipality. Beta diversity was slightly higher between the
294 two municipalities than between the two forest management types (Table 6).

295

296 **Discussion**

297 *Forest structure*

298 The results of this paper support the hypothesis that management intensity influences forest
299 structure as already reported in a number of studies (Nagaike et al. 2003; Decocq et al. 2004;
300 Sullivan et al. 2009). The density of emergent trees was found to be higher in actively coppiced
301 forests compared to abandoned ones, as it was expected in a coppice management-system. In
302 recently coppiced stands, the released “standards” are the only plant physiognomically recognizable
303 as trees and they widely contribute to generate this layer. Later, due to a shift of tree layers driven
304 by competition for light (Rademacher et al. 2004), the formerly emergent standard trees merge into
305 the newly growth dominant layer. The higher density of subordinated trees in the abandoned forests,
306 indicates the appearance of a layer formed by those trees that remained dominated during the
307 competition for light.

308 Other studies indicate that during the process of coppice regeneration, forest structural
309 heterogeneity decreases, with upper forest layers becoming dominant (Debussche et al. 2001) and
310 shrubs declining (Van Calster et al. 2008). The “loss” of shrubs means a type of simplification
311 despite of the higher subordinated tree density. The various regenerative phases of the various
312 management units forming forest patches in the Monti Sibillini result in a complex and
313 heterogeneous picture of structural characteristics of those forests. The lack of differences between
314 actively managed and abandoned coppices in the total density of woody overstory species (DBH
315 >2cm), may reflect the fact that abandonment is still relatively recent.

316 The persistence of a woody species in coppiced forests is connected to its resprouting ability (Bond
317 & Midgley 2001) and this is the biological trait on which coppicing is based in the whole
318 Mediterranean forests for centuries (Montes et al. 2004; Pinto-Correia and Vos 2004). As a legacy
319 from past land use, these results are probably due to the common origin of the compared forest
320 patches (i.e. also the abandoned forest patches included previously active coppiced stands).

321

322 *Forest species diversity*

323 At the plot scale, actively managed and abandoned coppice stands do not significantly differ neither
324 in total plants species richness nor in under- and overstory species richness. Our results seem to
325 contradict several studies indicating a decrease in richness with maturing coppices at the plot scale

326 (Debussche et al. 2001; Manson & Macdonald 2002; Decocq et al. 2004; Gondard & Romane 2005;
327 Bartha et al. 2008). A shift from species-rich assemblages of a relatively open and low-nutrient
328 forest to an impoverished plant assemblage of a closed-canopy abandoned forest was also reported
329 by Kopecky et al. (2013). Even in the nearby Siena region, plant richness at the plot scale has been
330 found to decrease in more mature forests, and even if this result was not specifically focused on the
331 abandonment of coppices the basic mechanism was that (Amici et al. 2013). The explanation for
332 this contradiction lies in the small size of coppiced management units studied and in the large scale
333 heterogeneity in the successional age of the forests. At the same time, the multi scale probabilistic
334 sampling here performed in four ecologically homogeneous forests, each characterized by an actual
335 precise management orientation, played an important role to understand the effects of scale in the
336 resulting species diversity patterns. In fact, the multi scale approach here adopted offers a totally
337 different view with respect to the observations made at a single spatial scale (e.g., a plot) and a
338 single regenerative forest phase. The small size of the management units (i.e. forest parcels
339 coppiced by local owners: 1.28-1.53 ha \pm 2.03 – 3.46, Canullo et al. 2009) in the investigated forest
340 patches, makes different regeneration stages co-occurring at the same time, favoring considerable
341 heterogeneity. Consequently, the variability of species richness at plot scale can hide real
342 differences in diversity of the entire forest. This calls for further studies focusing on a range of
343 spatial scales, above the simple plot scale (Podani et al. 1993; Gotelli & Colwell 2001; Chiarucci &
344 Bonini 2005). On the other hand, the results of this study evidenced contrasting results at larger
345 scale with the forest patch generated by still active coppicing in Bolognola being more complex and
346 including diverse set of patches which appears to promote plant species richness and composition
347 due to spatial turnover, even in the case of the beech *specialist species*. Both forest patches located
348 in Ussita resulted less complex (lower beta diversity with respect to the Bolognola ones), with a
349 reduced gamma diversity (50 species less), but with a higher species richness in the abandoned
350 forest patch for all the considered plant groups (total, herb layer and beech forest *specialist species*).

351

352 While it was not expected to find a higher beech forest *specialist species* richness in coppiced
353 patches because of the contribution of species preferring open habitat (i.e. non-forest-species,
354 Bartha et al. 2008), which are favored by coppicing and disturbance, it is somewhat surprising to
355 find a higher species richness of beech forest specialist at larger scales (thus a larger species pool)
356 in the active coppice Bolognola with respect to the abandoned coppice. The difference was really
357 minimal (33 vs. 29) but it suggests the possibility that a rich species pool of beech forest specialists
358 can also be maintained in coppices under active management. This outcome seems to be not in line
359 with other literature outcomes, but it should be noticed that most of the previous results were
360 achieved at a plot or stand scale, without an upscaling as it has done here. For example, Nagaike et
361 al. (2003) and Decocq et al. (2004), found a decrease of forest *specialist species* after silvicultural

362 treatments at the plot scale. Furthermore, Bartha et al. (2008) report a linear positive regression for
363 beech *specialist species* with coppice age. Our finding on beech forest *specialist species* could be
364 explained by the fact that when larger scales are considered these actively managed coppices
365 represent very heterogeneous landscapes, due to the contemporary presence of different
366 regeneration stages in a rather fine scale patchiness. This can maintain a heterogeneous structure of
367 the forests and consequently a broader spectrum of ecological niches is available for plants,
368 including many of those plants which are linked to mature beech forest stages. In fact, Bartha et al.
369 (2008) reported a remnant of beech forest *specialist species* also at young stages. Mechanistic
370 insights of a former study on the functional traits of understory forest species in the same area
371 (Campetella et al. 2011) pointed out that beech forest *specialist species* may meet their habitat
372 requirements already at a successional age of about 14 years after coppicing. Thus, the change of
373 scale perspective leads to the view that despite a reduction of beech forest *specialist species* can be
374 observed at the plot scale, when the focus is on a larger scale this is not valid any more. Practically,
375 the species richness of this group of species is decreased at the alpha diversity level (plot scale) but
376 not at the gamma diversity level (species pool), and this results into a higher beta diversity, i.e. a
377 more heterogeneous distribution in the landscape. In practical terms, these results can be explained
378 by the presence of older coppice stands randomly occurring within the active coppice forest patches
379 in the Bolognola context. There the active forest patch could be considered as a mosaic of
380 contiguous plots in which we can find early cuts with older coppice stands. This is depending on the
381 type of fragmented property, private in most of cases, and to the fact that, during the last decades,
382 due to lack of active management different coppices have been almost abandoned or managed with
383 lower frequency (Amorini & Fabbio 2001; Ciancio et al. 2006; Canullo et al. 2009). This helps to
384 generate a more complex mosaic of forest stands likely to maintain a larger pool of *specialist*
385 *species* (with narrow niches).

386 On the other hand, the smaller pool of beech forest *specialist species* in abandoned patch of
387 Bolognola (accumulation curve reduced the increasing rate at an effort of 14-16 sampled plots) is
388 probably due to the homogeneity of the forest stands, characterized by a more closed-canopy and a
389 consequent impoverished flora, more oriented on shade adapted species (i.e. lower availability of
390 light for the understory species, including the beech forest specialist ones; see also Rogers et al.
391 2008; Keith et al. 2009).

392 The higher overstory species richness in the coppiced plots is the only convergent result between
393 the two Municipality contexts (areas), which can be explained by a higher availability of woody
394 species niches provided by the coppicing disturbance on the canopy. In fact, also the multiplicative
395 partitioning of beta diversity emphasized that Municipalities (areas) provide a more important effect
396 on the species diversity distribution with respect to the management itself (i.e. beta diversity higher
397 between the two municipalities than between the two forest management types). This is another

398 unexpected result (distance between forest stands is a stronger driver than forest management in
399 influencing species composition), probably produced by the significant effects of different forest
400 practices that are implemented by the “local woodcutters” for the maintenance of coppicing.
401 Despite the same management system in both municipalities, the adoption of different criteria in the
402 selection of coppiced parcels (influencing also their spatial distribution) and of standards in terms of
403 number and distribution (as our results demonstrated: 12.65 in Ussita, and 7.35 in Bolognola in 400
404 m²), can be relevant to support the obtained result in species composition similarity.
405 These results must be inspected more deeply in the future, trying to shed light onto the socio-
406 economic and ecological causes of the observed habitat mosaic. The marginalization of forest areas
407 in the Apennines calls for conservation action plans to sustain the considerable biodiversity
408 heritage. Scale, in this respect, is an important factor. We have contributed to show that statements
409 about the effect of forest management on diversity should always define the spatial scale they refer
410 to, as measures considering only plot-level data might result in misleading conclusions. Our results
411 emphasize that analyses should extend toward to the landscape scale in order to better inform and
412 assist conservation management planning.

413

414 *Practical consequences for forest management and ecosystem function*

415 Despite the local relevance of the study and the limitation in addressing general conclusions at
416 broader geographical scales, we consider the present study as an important contribution in
417 understanding the influence of the management type on species diversity.

418

419 Coppice is an important component of the traditional multifunctional management of forests in the
420 Mediterranean area (Pinto Correia and Vos 2004), but still a fervent debate is active to evaluate the
421 future and the sustainability of such forest management (Ciancio et al. 2006). The importance of
422 coppice management, as a driver of the forest landscape diversification, is reported by different
423 authors (Decoq et al. 2004; Coppini and Hermanin 2007; Kopecky et al. 2013; Müllerová et al.
424 2015).

425

426 Several studies have demonstrated that coppices with standards under a regular felling regime have
427 some negative effects, such as soil erosion, changes in bulk density or soil porosity, changes of pH
428 levels towards soil acidification, depletion of the soil organic matter and nutrient pools (Dyck and
429 Skinner 1990; Alban and Perala 1990; Staaf and Olsson 1994; Corona et al. 1996; Worrel and
430 Hampson 1997). In the long term, these changes have been associated with losses of plant species
431 diversity as well (Gilliam and Roberts 1995), so care should be taken in deeming coppicing
432 'sustainable'. According to our results, however, the coppice with standards in the study area seems
433 to be sustainable in terms of the preservation of species diversity on a larger spatial scale. In fact,

434 the present study demonstrates that even the group of species more linked to mature forest stages
435 (beech forest *specialist species*) can largely persist in a coppice landscape, if this is managed
436 ensuring a certain heterogeneity with relatively small coppiced stands and a high variability of
437 coppice age.

438 The small abandoned patches within the coppice landscape can play a crucial role as propagulum
439 sources and their importance should not be discounted (Gimona et al. 2011). To improve its
440 sustainability, coppicing should be planned in a larger scale management, in which the coppice
441 stands are identified as parts of a heterogeneous mosaic, where more developed abandoned stands
442 are mixed with others under a regular felling regime. Such a solution would be likely to have
443 environmental benefits (e.g. in terms of ecosystem services) and provide a useful diversification of
444 wood production and of forest work and its related professional skill (Coppini and Hermanin 2007),
445 hence enhancing the social sustainability of this management system. This suggestion is also
446 consistent with Kopecky et al. (2013) finding that too large a shift to closed-canopy forest
447 management endangers the diversity of European lowland forests. According to our results, trade-
448 offs between open and closed-canopy species might be less than expected at the landscape scale, as,
449 in the studied system, even the forest specialists pool might be maintained in a coppiced landscape
450 mosaic.

451 This introduces more interesting reflections about the biodiversity consequences of forest
452 management at intermediate spatial scales. For such reasons, our results link the existing knowledge
453 of beech coppice diversity from landscape- and plot-scale and can help to address new management
454 issues and conservation planning on local territorial scale. In the uncertain but certainly variable
455 future, due to, for example, climatic and land use changes (Millar et al. 2007), forest management
456 needs to be more resilient (i.e. active adaptive management). In this context, understanding how to
457 promote biodiversity, given its insurance value, is likely to become more important to help sustain
458 desired states of dynamic ecosystem in the face of uncertainty and surprise (e.g Hobbs et al. 2006).

459

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467 **References**

- 468 Agnoletti, M. 2007. The degradation of traditional landscape in a mountain area of Tuscany during
469 the 19th and 20th centuries: Implication for biodiversity and sustainable management. *Forest*
470 *Ecology and Management* 249: 5-17.
- 471 Alban, D.H., Perala, D.A. 1990. Impacts of Aspen timber harvesting on soils. In: Gessel, S.P.,
472 Lacate, D.S., Weetman, G.F., Powers, R.F. (Eds.) *Proceedings of the Seventh North American*
473 *Forest Soils Conference on Sustained Productivity of Forest Soils*. University of British
474 Columbia, Vancouver, BC, pp. 377–391.
- 475 Albuquerque, F.S., Rueda, M. 2010. Forest loss and fragmentation effects on woody plant species
476 richness in Great Britain. *Forest Ecology and Management*. 260: 472–479.
- 477 Amici, V., Santi, E., Filibeck, G., Diekmann, M., Geri, F., Landi, S., Scoppola, A., Chiarucci, A.
478 2013. Influence of secondary forest succession on plant diversity patterns in a Mediterranean
479 landscape. *Journal of Biogeography* 40: 2335–2347.
- 480 Amorini, E., Fabbio, G. 2001. La gestione del bosco ceduo nelle aree protette. *Informatore*
481 *Botanico Italiano* 33: 164-168.
- 482 Antrop, M. 2004. Landscape change and the urbanization process in Europe. *Landscape Urban*
483 *Plan*. 67: 9-26.
- 484 Ares, A., Neill, A.R., Puettmann, K.J. 2010. Understory abundance, species diversity and functional
485 attribute response to thinning in coniferous stands. *Forest Ecology and Management* 260: 1104–
486 1113.
- 487 ASSAM (Agenzia Servizi Settore Agroalimentare delle Marche). 2006. Suoli e Paesaggi delle
488 Marche. Errebi Grafiche Ripesi, Falconara Marittima, Ancona.
- 489 Ballelli, S., Lucarini, D., Pedrotti, F. 2005. Catalogo dell’Erbario dei Monti Sibillini di Vittorio
490 Marchesoni. *Braun Blanquetia* 38: 1-259.
- 491 Barbero, M., Bonin, G., Loisel, R., Quézel, P. 1990. Changes and disturbances of forest ecosystems
492 caused by human activities in the western part of the Mediterranean basin. *Vegetatio* 87: 151-
493 173.
- 494 Bartha, S., Merolli, A., Campetella, G., Canullo, R. 2008. Changes of vascular plant diversity along
495 a chronosequence of beech coppice stands, central Apennines, Italy. *Plant Biosystem* 142: 572 –
496 583.
- 497 Baselga, A. 2010. Partitioning the turnover and nestedness components of beta diversity. *Global*
498 *Ecology and Biogeography* 19: 134–143.
- 499 Battles, J.J., Shlisky, A.J., Barrett, R.H., Heald, R.C., Allen-Diaz, B.H. 2001. The Effects of forest
500 management on plant species diversity in a sierran conifer forest. *Forest Ecology and*
501 *Management* 146: 211-222.
- 502 Biondi, E., Baldoni, M. 1995. A possible method for geographic delimitation of phytoclimatic

503 types: With application to the phytoclimate of the Marche region of Italy. *Doc. Phytosociol.* 15:
504 15-28.

505 Bond, W.J., Midgley J.J. 2001. The persistence niche: ecology of sprouting in woody plants. *Trends*
506 *in Ecology and Evolution* 16: 45-51.

507 Borhidi, A. 1995. Social behaviour types, the naturalness and relative ecological indicator values of
508 the higher plants in the Hungarian flora. *Acta Botanica Hungarica* 39: 97-181.

509 Brockway, D.G., 1998. Forest plant diversity at local and landscape scales in the Cascade
510 Mountains of southwestern Washington. *Forest Ecology and Management* 109: 323-341.

511 Campetella, G., Botta-Dukát, Z., Wellstein, C., Canullo, R., Gatto, S., Chelli, S., Mucina, L.,
512 Bartha, S. 2011 - Patterns of plant trait-environment relationships along a forest succession
513 chronosequence. *Agriculture EcosystemEnvironment* 145: 38- 48.

514 Canullo, R., Campetella, G., Merolli, A. 2009. Gestione integrata degli aspetti ecologici e
515 paesaggistici degli ecosistemi montani: il paesaggio delle faggete. In Corrado R., Scotti M. (eds)
516 *Laboratorio del paesaggio: indirizzi e progetti per le aree protette marchigiane*, Il lavoro
517 editoriale (Progetti editoriali srl), Ancona: 63-88.

518 Chiarucci, A. 2007. To sample or not to sample? That is the question ... for the vegetation scientist.
519 *Folia Geobotanica* 42: 209-216.

520 Chiarucci, A., Bacaro, G., Rocchini, D. 2008a. Quantifying plant species diversity in a Natura 2000
521 network: Old ideas and new proposals. *Biological Conservation* 141: 2608-2618.

522 Chiarucci, A., Bacaro, G., Rocchini, D., Fattorini, L. 2008b. Discovering and rediscovering the
523 sample-based rarefaction formula in the ecological literature. *Community Ecology* 9: 121-123.

524 Chiarucci, A., Bonini, I. 2005. Quantitative floristics as a tool for the assessment of plant diversity
525 in Tuscan forests. *Forest Ecology and Management* 212: 160-170.

526 Ciancio, O., Corona, P., Lamonaca, A., Portoghesi, L., Travaglini, A. 2006. Conversion of clearcut
527 beech coppices into high forests with continuous cover: A case study in central Italy. *Forest*
528 *Ecology and Management* 224: 235-240.

529 Clement, C. 2001. The ecological importance of understory herbaceous plants. In: The Effects of
530 Land-Use change on the biodiversity of the Highlands Plateau: A Carolina Environmental
531 Program Report: 32-42.

532 Conti, F., Abbate, G., Alessandrini, A, Blasi, C. 2005. An annotated checklist of the Italian flora.
533 Roma, Palombi.

534 Coppini, M., Hermanin, L. 2007. Restoration of selective beech coppices: A case study in the
535 Apennines (Italy). *Forest Ecology and Management* 249: 18-27.

536 Corona, P., Iovino, F., Lucci, S. 1996. La gestione dei sistemi forestali nella conservazione del
537 suolo. *EM-Linea Ecologica* 4: 4-15.

538 Crawley, M.J., Harral J.E. 2001. Scale dependence in plant biodiversity. *Science* 291: 864-868.

- 539 Debussche, M., Debussche, G., Lepar, J. 2001. Changes in the vegetation of *Quercus pubescens*
540 woodland after cessation of coppicing and grazing. *Journal of Vegetation Science* 12: 81-92.
- 541 Decocq, G., Aubert, M., Dupont, F., Alard, D., Saguez, R., Wattez-Franger, A., De Foucault B.,
542 Deleis-Dusollier, A., Bardat, J. 2004. Plant diversity in a managed temperate deciduous forest:
543 understory response to two silvicultural systems. *Journal of Applied Ecology* 41: 1065-1079.
- 544 Dyck, W.J., Skinner, M.F. 1990. Potential productivity decline in New Zealand radiata pine forests.
545 In: Gessel, S.P., Lacate, D.S., Weetman, G.F., Powers, R.F. (Eds.) *Proceedings of the Seventh*
546 *North American Forest Soils Conference on Sustained Productivity of Forest Soils*. University of
547 British Columbia, Vancouver, BC, pp. 318–329.
- 548 Evans, M.N., Barkham, J.P. 1992. Coppicing and natural disturbance in temperate woodlands – a
549 review. In *Ecology and management of coppice woodlands*. Buckley, G.P. (Eds.), London, 79-
550 99.
- 551 Gimona, A., Polhill, J.G., Davies, B. 2011. Sinks, sustainability and conservation incentives. In Liu,
552 J., Hull, V., Morzillo, A., Wiens, J., (Eds.) *Sources, Sinks and Sustainability*. Cambridge
553 University Press. pp. 155-178.
- 554 Gilliam, F.S., Roberts, M.S. 1995. Forest management and plant diversity. *Ecological Society of*
555 *America, Ecological Applications*, 5: 911-912.
- 556 Gilliam, F.S. 2007. The ecological significance of the herbaceous layer in temperate forest
557 ecosystems. *Bioscience* 57(10): 845-858.
- 558 Gondard, H., Romane, F. 2005. Long-term evolution of understory plant species composition after
559 logging in chestnut coppice stands (Cevennes Mountains, southern France). *Annals of Forest*
560 *Science* 62: 333-342.
- 561 Gotelli, N.J., Colwell, R.K. 2001. Quantifying biodiversity: procedures and pitfalls in the
562 measurement and comparison of species richness. *Ecology Letters* 4: 379-391.
- 563 Hédli, R., Kopecký, M., Komárek, J. 2010. Half a century of succession in a temperate oakwood:
564 from species-rich community to mesic forest. *Diversity and Distribution* 16: 267-276.
- 565 Hermy, M., Honnay, O., Firbank, L., Grashof-Bokdam, C., Lawesson, J.E. 1999. An ecological
566 comparison between ancient and other forest plant species of Europe, and the implications for
567 conservation. *Biological Conservation* 91: 9–22.
- 568 Hobbs, R., et al. 2006. Novel ecosystems: theoretical and management aspects of the new
569 ecological world order. *Global Ecology and Biogeography* 15: 1–7.
- 570 IPLA, 2001. I tipi forestali delle Marche. Inventario e carta forestale della regione Marche. Regione
571 Marche, Diffusioni Grafiche. Torino.
- 572 Keith, S.A., Newton, A.C., Morecroft, M.D., Bealey, C.E. Bullock, J.M. 2009. Taxonomic
573 homogenization of woodland plant communities over 70 years. *P Roy. Soc. Lond. B. Bio.* 276:
574 3539–3544.

- 575 Kobayashi, S. 1974. The species–area relation I. A model for discrete sampling. *Researches on*
576 *Population Ecology* 15: 223–237.
- 577 Kopecky, M., Hédl, R., Szabo, P. 2013. Non-random extinctions dominate plant community
578 changes in abandoned coppices. *Journal of Applied Ecology* 50: 79–87.
- 579 McKinney, M.L., Lockwood, J.L. 1999. Biotic homogenization: a few winners replacing many
580 losers in the next mass extinction. *Trends in Ecology and Evolution* 14: 450–453.
- 581 Mason, C.F., Macdonald S.M. 2002. Responses of ground flora to coppice management in an
582 English woodland – a study using permanent quadrats. *Biodiversity and Conservation* 11: 1773-
583 1789.
- 584 Millar, C.I., Stephenson, N.L., Stephens, S.L. 2007 Climate change and forests of the future:
585 managing in the face of uncertainty. *Ecological Applications* 17(8): 2145–2151.
- 586 Montes, F., Canellas, I., Del Rio, M., Calama, R., Montero, G. 2004. The effects of thinning on the
587 structural diversity of coppice forests. *Annals of Forest Science* 61: 771-779.
- 588 Müllerová, J., Hédl, R., Szabó, P. 2015. Coppice abandonment and its implications for species
589 diversity in forest vegetation. *Forest Ecology and Management* 343: 88-100.
- 590 Nagaike, T., Kamitani, T., Nakashizuka T. 2003. Plant species diversity in abandoned coppice
591 forests in a temperate deciduous forest area of central Japan. *Plant Ecology* 166: 145-156.
- 592 Oksanen, J., Blanchet, F.G., Kindt, R., Legendre, P., Minchin, P.R., O'Hara, R.B., Simpson, G.L.,
593 Solymos, P. 2015. Vegan: community ecology package. – R package ver. 2.2-1, [http://cran.r-](http://cran.r-project.org)
594 [project.org](http://cran.r-project.org), <https://github.com/vegandevs/vegan>.
- 595 Palmer, M.W., White, P.S. 1994. Scale dependence and the species-area relationship. *The American*
596 *Naturalist* 144: 717–740.
- 597 Pavari, A. 1999. Il problema dei boschi cedui in Italia. *Monti e Boschi* 6: 17-26.
- 598 Peterken, G.F. 1981. *Woodland Conservation and Management*. Chapman and Hall, London.
- 599 Pignatti, G., Terzuolo P.G., Varese, P., Semerari, P., Lombardi, V.N. 2004. Criteri per la
600 definizione di tipi forestali nei boschi dell'Appennino meridionale. *Forest@* 1: 112-127.
601 Available:<http://ww.sisef.it/> Accessed Jan 2007 04.
- 602 Pignatti, S. 1982. *Flora d'Italia*. Bologna. Edagricole.
- 603 Pignatti, S. 1998. *I boschi d'Italia*. Sinecologia e biodiversità. Torino. UTET.
- 604 Pinto-Correia T., Vos W. 2004. Multifunctionality in Mediterranean landscapes – past and future.
605 In: Jongman R. (Ed.) *The New Dimensions of the European Landscape*, Wageningen EU Frontis
606 Series, Springer.
- 607 Piovesan, G., Biondi, F., Bernabei, M., Di Filippo, A., Schirone, B. 2005. Spatial and altitudinal
608 bioclimatic zones of the Italian peninsula identified from a beech (*Fagus sylvatica* L.) tree-ring
609 network. *Acta Oecologica* 27: 197-210.
- 610 Podani, J., Czárán, T., Bartha, S. 1993. Pattern, area and diversity: the importance of spatial scale in

611 species assemblages. *Abstracta Botanica* 17: 37-51.

612 R Core Team 2015. R Foundation for Statistical Computing, Vienna, Austria.

613 Rackham, O. 2008. Ancient woodlands: modern threats. *New Phytologist* 180: 571–586.

614 Rademacher, C., Neuert, C., Grundmann, V., Wissel C., Grimm, V. 2004. Reconstructing
615 spatiotemporal dynamics of Central European natural beech forests: the rule-based forest model
616 BEFORE. *Forest Ecology and Management* 194: 349-368.

617 Rogers, D.A., Rooney, T.P., Olson, D., Waller, D.M. 2008. Shifts in southern Wisconsin forest
618 canopy and understorey richness, composition, and heterogeneity. *Ecology* 89: 2482–2492.

619 Salemaa M., Derome J., Nojd P. 2008. Response of boreal forest vegetation to the fertility status of
620 the organic layer along a climatic gradient. *Boreal Environment Research* 13: 48-66.

621 Staaf, H., Olsson, B.A. 1994. Effects of slash removal and stump harvesting on soil water chemistry
622 in a clearcutting in SW Sweden. *Scandinavian Journal of Forest Research* 9: 305–310.

623 Sullivan, T.P., Sullivan, D.S., Lindgren, P.M.F., Ransome, D.B. 2009. Stand structure and the
624 abundance and diversity of plants and small mammals in natural and intensively managed
625 forests. *Forest Ecology and Management* 258: 127-141.

626 Van Calster, H., Baeten, L., Deschrijver, A., Dekeersmaeker, L., Rogister, J., Verheyen, K., Hermy,
627 M. 2007. Management driven changes (1967–2005) in soil acidity and the understorey plant
628 community following conversion of a coppice-with-standards forest. *Forest Ecology and
629 Management* 241: 258–271.

630 Van Calster, H., Baeten, L., Verheyen, K., De Keersmaeker, L., Dekeyser, S., Rogister J. E.,
631 Hermy, M. 2008. Diverging effects of overstorey conversion scenarios on the understorey
632 vegetation in a former coppice-with-standards forest. *Forest Ecology and Management* 256:
633 519–528.

634 Whittaker, R.H. 1972. Evolution and measurement of species diversity. *Taxon* 21: 213-251.

635 Worrel, R., Hampson, A. 1997. The influence of some forest soils—a review. *Forestry* 70 (1): 61–
636 85.

637 Wulder, M.A., White, J.C., Andrew, M.E., Seitz, N.E., Coops, N.C. 2009. Forest fragmentation,
638 structure, and age characteristics as a legacy of forest management. *Forest Ecology and
639 Management* 258: 1938-1949.

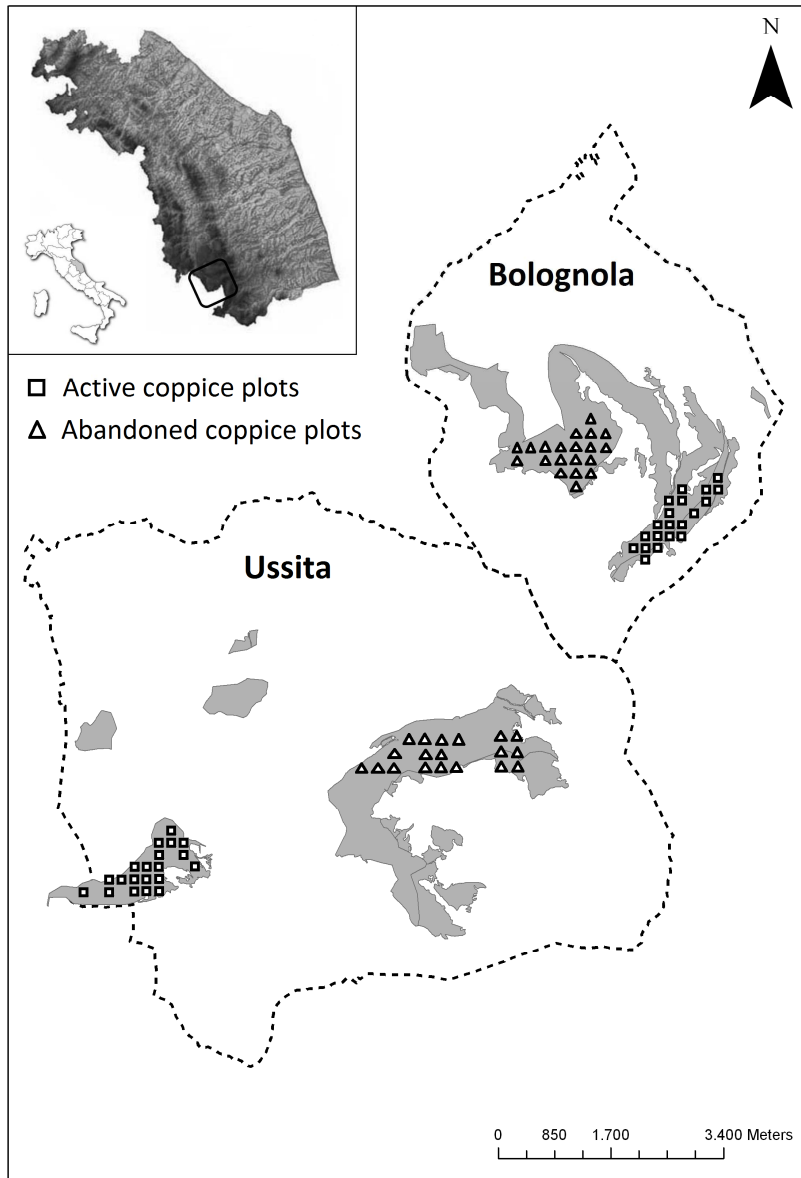
640 **Fig. 1.** Sampling design applied in the Beech forest patches of the study area near Bolognola and
641 Ussita.

642

643 **Fig. 2.** Species rarefaction curves for total species richness, overstory species richness, understory
644 richness and beech forest specialist richness, in each forest patch and according to the relative
645 municipalities (BOL=Bolognola; USS=Ussita). Vertical lines are confidence intervals. Empty
646 triangles=coppiced forest, black triangles=abandoned forest.

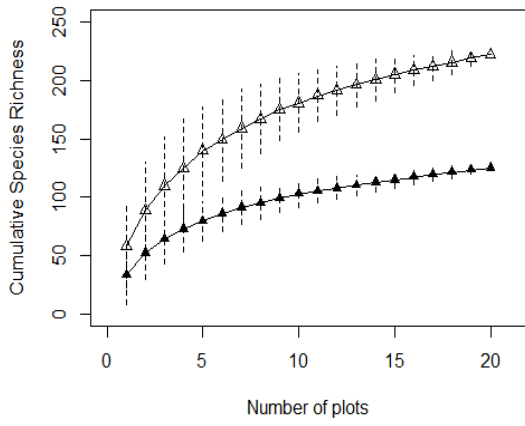
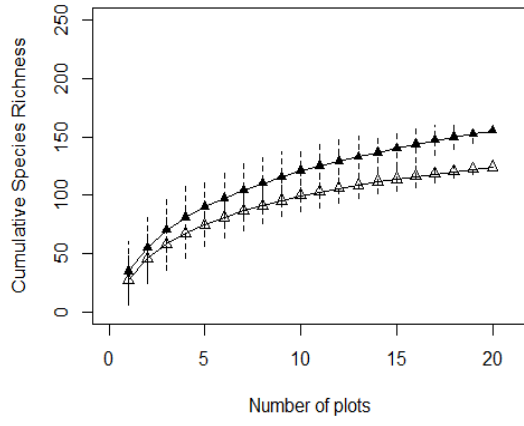
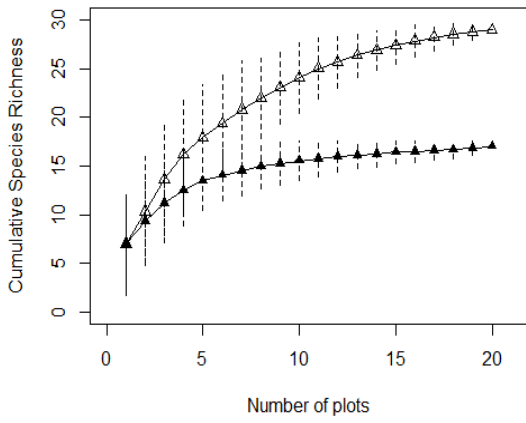
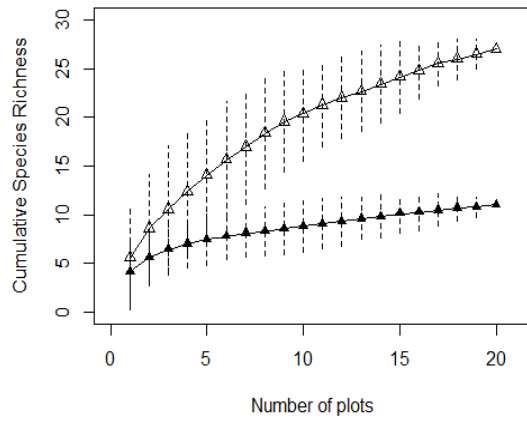
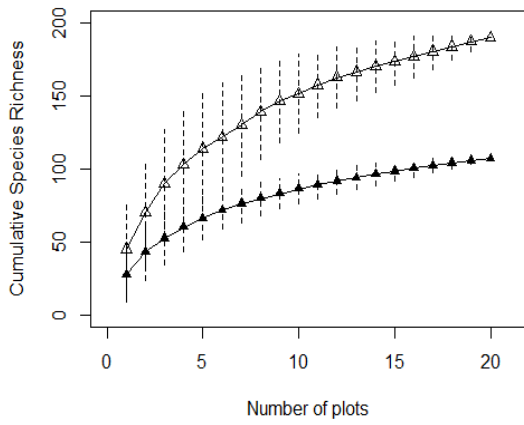
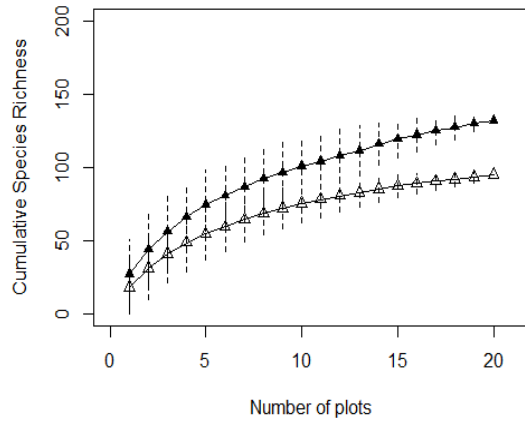
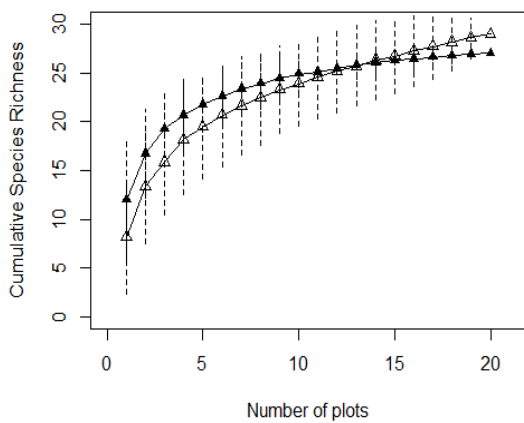
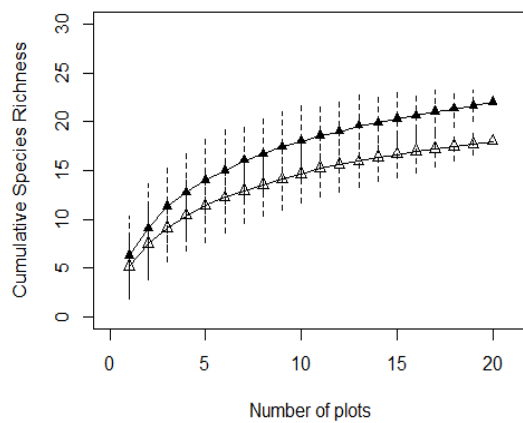
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648 **Figures**



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650 Figure 1 -

BOL TOTAL RICHNESS**USS TOTAL RICHNESS****BOL OVERSTORY****USS OVERSTORY****BOL UNDERSTORY****USS UNDERSTORY****BOL SPECIALIST****USS SPECIALIST**

652 Figure 2 -

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655 **Tables**

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658 Table 1 - Partitioning of the total diversity of vascular plants according to the multi-level sampling
659 hierarchy: plot, forest patch, municipality, forest management type.
660

	Sampling hierarchy	Measurement	Municipalities	Forest management types
Alpha	Plot	Mean n. of species per plot	Same	Same
	Patch	Mean n. of species per patch	Same	Different
	Patch	Mean n. of species per patch	Different	Same
	Patch	Mean n. of species per patch	Different	Different
	Forest management type	Mean n. of species per management type	Different	Same
	Municipality	Mean n. of species per municipality	Same	Different
Beta	Plot	Beta diversity among the plots	Same	Same
	Patch	Beta diversity among the patches	Same	Different
	Patch	Beta diversity among the patches	Different	Same
	Patch	Beta diversity among the patches	Different	Different
	Forest management type	Beta diversity between forest management types	Different	Same
	Municipality	Beta diversity between municipalities	Same	Different
Gamma	Plot	Cumulative n. of species recorded by the plots	Same	Same
	Patch	Cumulative n. of species recorded by the patches	Same	Different
	Patch	Cumulative n. of species recorded by the patches	Different	Same
	Patch	Cumulative n. of species recorded by the patches	Different	Different
	Forest management type	Cumulative n. of species recorded by the forest management types	Different	Same
	Municipality	Cumulative n. of species recorded by the municipalities	Same	Different

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693 Table 2 - Mean values of selected variables characterizing released standards, woody individuals
 694 (DBH>5cm) and stand layers in the forest patches according to management type (plot scale 20 m x
 695 20 m).
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Management Forest patch (Municipalities)	Active Coppice		Abandoned		Sign. of Management	Sign. of Area
	Bol (n=20)	Uss (n=20)	Bol (n=20)	Uss (n=20)		
Mean standards number	7.35ab (±4.17)	11.15abc (±6.27)	6.47a (±5.13)	12.65c (±6.76)	0.759	0.000
Mean standards diameter	20.09a (±6.68)	23.63c (±11.21)	29.36b (±10.59)	20.60a (±8.09)	0.078	0.000
Mean standards height	12.41a (±3.07)	12.55a (±3.60)	20.06b (±4.59)	11.58c (±3.90)	0.000	0.000
Number of emergent trees	4.80a (±3.60)	4.20a (±6.04)	1.95b (±2.28)	1.25b (±1.20)	0.001	0.451
Number of dominant trees	41.20a (±33.20)	26.25a (±14.42)	29.00a (±10.36)	36.6a (±24.86)	0.859	0.987
Number of subordinate trees	20.15a (±22.58)	24.25ab (±18.39)	26.95a (±15.44)	23.55ab (±20.34)	0.482	0.483
Number of shrubs	14.80a (±21.96)	8.65ab (±10.03)	1.60b (±2.72)	2.55b (±2.96)	0.000	0.627
Total woody individuals	80.95a (±52.06)	60.41a (±31.69)	59.51a (±21.94)	63.95a (±38.86)	0.223	0.747
Total vegetation cover (%)	91.35a (±6.53)	88.20a (±5.40)	87.55a (±5.85)	85.5a (±10.08)	0.046	0.158
Herb layer cover (%)	22.20a (±17.60)	6.24b (±11.55)	12.86ab (±11.49)	15.48ab (±14.81)	0.986	0.306
Deadwood cover (%)	2.95a (±2.19)	2.97a (±1.62)	8.25b (±5.40)	3.25a (±2.22)	0.000	0.000

Significant differences (p<0.001) generated both by the management and the nested forest patches were assessed according to ANOVA statistics (Linear Mixed Models). Significant differences between single forest patches are indicated by letters (post hoc Bonferroni). Standard deviation values are reported in parenthesis. (Bol=Bolognola Municipality; Uss=Ussita Municipality; n=number of relevees)

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Table 3 - Mean values of species richness and Shannon index (H') characterizing the forest patches according to management type (plot scale 20 m x 20 m).

Management Forest patch (Municipalities)	Active Coppice		Abandoned		Sign. of Management	Sign. of Area
	Bol (n=20)	Uss (n=20)	Bol (n=20)	Uss (n=20)		
Mean vascular species n.	55.05a (±17.08)	28.35b (±10.66)	34.95b (±12.60)	35.30b (±13.00)	0.068	0.023
Mean vascular overstory species n.	7.15a (±2.79)	5.70ab (±2.77)	5.85ab (±2.41)	3.90b (±1.70)	0.006	0.002
Mean vascular understory species n.	47.90a (±17.37)	22.65b (±10.22)	29.10b (±11.13)	31.35b (±12.73)	0.151	0.163
Mean specialist species n.	8.80a (±3.12)	5.15c (±1.69)	12.40b (±3.29)	6.25c (±2.14)	0.000	0.000
Shannon (H') woody individuals	1.09a (±0.45)	0.96a (±0.55)	0.93a (±0.54)	0.74a (±0.53)	0.109	0.216

Significant differences (p<0.001) generated both by the management and the nested forest patches were assessed according to ANOVA statistics (Linear Mixed Models). Significant differences between single forest patches are indicated by letters (post hoc Bonferroni). Standard deviation values are reported in parenthesis. (Bol=Bolognola Municipality; Uss=Ussita Municipality; n=number of relevés)

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Table 4 - diversity partitioning between plots within each forest patch.

Municipality	Forest patch management type	Gamma	Mean alpha plot	Beta plot
USSITA (USS)	abandoned	155	35.25	4.40
	coppiced	124	28.35	4.37
BOLOGNOLA (BOL)	abandoned	125	34.95	3.58
	coppiced	222	55.05	4.03

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Table 5 - diversity partitioning between patches within the same municipality (a), the same forest patch management type (b), between different municipalities and forest management types (c).

(a)			
PATCH (within the same municipality)	Gamma	Mean alpha patch	Beta patches
USS-coppiced/USS-abandoned	192	139.50	1.38
BOL-coppiced/BOL-abandoned	239	173.50	1.38

(b)			
PATCH (within the same forest management type)	Gamma	Mean alpha patch	Beta patches
BOL-coppiced/USS-coppiced	246	173	1.42
BOL-abandoned/USS-abandoned	196	140	1.40

(c)			
PATCH (between different municipalities and forest management type)	Gamma	Mean alpha patch	Beta patches
BOL-abandoned/USS-coppiced	173	124.50	1.39
USS-abandoned/BOL-coppiced	265	188.50	1.41

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Table 6 - diversity partitioning between forest management types (a) and between municipalities (b).

(a)		
Total Richness	Mean alpha forest management type	Beta forest management type
291	221	1.32

(b)		
Total Richness	Mean alpha municipality	Beta municipality
291	215.5	1.35

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