Campetella Giandiego, Canullo Roberto, Gimona Alessandro, Garadnai Janos, Chiarucci Alessandro, Giorgini Daniele, Angelini Elia, Cervellini Marco, Chelli Stefano, Bartha Sandor (2016): Scale-dependent effects of coppicing on the species pool of late successional beech forests in the central Apennines, Italy. APPLIED VEGETATION SCIENCE 19:(3) pp. 474-485.

SCALE DEPENDENT EFFECTS OF COPPICING ON THE SPECIES POOL OF LATE-SUCCESSIONAL BEECH FOREST

Campetella Giandiego, Canullo Roberto, Gimona Alessandro, Garadnai Janos, Chiarucci Alessandro, Giorgini Daniele, Angelini Elia, Cervellini Marco, Chelli Stefano, Bartha Sandor.

Running head: Scale dependent effects of coppicing on diversity
Abstract

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

Location: Monti Sibillini, Central Apennines, Italy.

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

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

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

Keywords: Scale-dependence; landscape; alpha diversity; beta diversity; Central Apennines; forest stand structure
Introduction

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

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

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

The consequences of coppice abandonment is in line with a worldwide simplification of forest structure and function (McKinney & Lockwood 1999; Ares et al. 2010), generating a progressive taxonomic impoverishment and homogenization of the woodland vegetation (Keith et al. 2009). In light to these evidences, ecological studies need to explore these issues because plant species assemblages change rapidly under human impact (Hedl et al. 2010; Kopecky et al. 2013).
Although the herbaceous layer contains more than 80% of the total plant species richness, this layer is often an under-considered component of forest ecosystems (Gilliam 2007). However, its effects on tree productivity and long-term stability of the ecosystem have been demonstrated, as well as its influence on the composition, microbiological processes and nutrient cycling of the organic layer (Clement 2001; Gilliam 2007; Salemaa et al. 2008). Herbaceous species also contribute significantly to biodiversity and particularly important for this are those plants which are late successional beech forest species (beech forest “specialist species”). These species usually characterize the forest mature phases and ancient woodlands (Gilliam 2007; Bartha et al. 2008), adapted to the relatively stable environmental conditions of forests (Hermy et al. 1999). The role of forest management in preserving the specialist species pool at different spatial scales is still poorly evaluated (Bartha et al. 2008). These species can be used as indicators of biodiversity, and in this respect are important functional components for their habitat and resource specificity (Gillam 2007). Recent studies performed in beech forests of the Italian Apennines (Bartha et al. 2008; Campetella et al. 2011) demonstrated that, at the stand scale, the abundance contribution of the beech forest specialist species doubled after about 40–60 years after coppicing.

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

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

Study area

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

Sampling design

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

To represent the plant species composition and structural attributes at the plot and forest patch scales without sampling bias (Chiarucci 2007), a probabilistic sampling design was realized by 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-W directions (Figure 1). This distance was chosen to sample the highest level of heterogeneity, since the single size of units coppiced in a single event is usually between 0.5-2 ha (Canullo et al.
2009). The grid was virtually overlaid on each of our selected forest patches starting by a random point. The south east corner of each plot was positioned at each grid node, after its location by Garmin Vista Etrex GPS.

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

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

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

\textbf{Statistical analysis}

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

The effects of management and forest patches on the structural variables (number of standards, standard diameter, standard height, emergent trees density, dominant trees density, subordinated trees density, shrubs density, total woody density, total vegetation cover, herb layer cover, deadwood cover) and diversity variables (vascular species richness, overstory species richness, understory species richness, beech forest specialist species richness, Shannon index $H'$) was analyzed by nested ANOVA statistics (Linear Mixed Models), with management as a fixed factor and forest patches nested within the management type. A correction for multiple testing was carried
out to avoid inflating the significance of individual tests (p≤0.001).

We addressed the problem of the spatial autocorrelation of error, as this can result in spurious relationships (type I error). For this purpose, we tested for autocorrelation of the models’ error by using a randomization test. Namely, we plotted the experimental variogram of the models’ residuals and produced a ‘random envelope’ by permuting the coordinates of residual values 999 times while re-computing the variogram at each step. The maximum and minimum value of the semi-variance obtained for each lag provided the ‘random envelope’. If the observed variogram values were not within such envelope, this was taken as evidence of autocorrelation and therefore we fitted models with exponentially or spherically correlated spatial error to account for spatial non-independence of the residuals.

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

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

**Results**

**Forest stand structure at the plot scale**

A total of 5,446 individuals of woody species with DBH > 2 cm were censused. As expected, the mean standards height was significantly higher in the abandoned coppices than in active ones (Table 2), while the density of emergent trees and shrubs were significantly higher in active coppices. As an effect of nested forest areas, standards density, standards diameter and standards height were higher in the abandoned ones (Table 2). Interestingly, total woody individuals density with DBH > 2 cm showed no significant difference among the two management types (Table 2).

Results for the variables describing the overstory and understory layers evidenced (Table 2) that the two management regimes had similar plant cover (almost 90%, on average). The analysis of the spatial autocorrelation of error shows that all the observed variogram values occurred within the
random envelope, supporting the absence of autocorrelation among the different sampling units in
the adopted sampling design (see Supplementary material 2).

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

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

Diversity partitioning
Multiplicative partitioning of beta diversity into hierarchical levels showed discordant results in the
two study areas in term of number of species and species composition. In fact, at the plot scale, the
number of species was higher in abandoned patch than actively coppiced forest in Ussita, but it was
higher in actively coppiced forest than in abandoned forest in Bolognola. On the other hand the
variability in species composition within each patch was higher within coppiced forest in Bolognola
but little higher within abandoned forest in Ussita (Table 4).
At patch level, although there were more species in coppiced patches than abandoned ones, the
variability in species composition between patches within the same municipality was slightly lower
than the variability measured between patches within the same forest management type (Table 5).
In other words, species assemblage were more similar between patches having different forest
management regimes within the same area, than between patches having the same forest management regime across different areas.

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

Discussion

Forest structure

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

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

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

Forest species diversity

At the plot scale, actively managed and abandoned coppice stands do not significantly differ neither in total plants species richness nor in under- and overstory species richness. Our results seem to contradict several studies indicating a decrease in richness with maturing coppices at the plot scale.
(Debussche et al. 2001; Manson & Macdonald 2002; Decocq et al. 2004; Gondard & Romane 2005; Bartha et al. 2008). A shift from species-rich assemblages of a relatively open and low-nutrient forest to an impoverished plant assemblage of a closed-canopy abandoned forest was also reported by Kopecky et al. (2013). Even in the nearby Siena region, plant richness at the plot scale has been found to decrease in more mature forests, and even if this result was not specifically focused on the abandonment of coppices the basic mechanism was that (Amici et al. 2013). The explanation for this contradiction lies in the small size of coppiced management units studied and in the large scale heterogeneity in the successional age of the forests. At the same time, the multi scale probabilistic sampling here performed in four ecologically homogeneous forests, each characterized by an actual precise management orientation, played an important role to understand the effects of scale in the resulting species diversity patterns. In fact, the multi scale approach here adopted offers a totally different view with respect to the observations made at a single spatial scale (e.g., a plot) and a single regenerative forest phase. The small size of the management units (i.e. forest parcels coppiced by local owners: 1.28-1.53 ha ±2.03 – 3.46, Canullo et al. 2009) in the investigated forest patches, makes different regeneration stages co-occurring at the same time, favoring considerable heterogeneity. Consequently, the variability of species richness at plot scale can hide real differences in diversity of the entire forest. This calls for further studies focusing on a range of spatial scales, above the simple plot scale (Podani at al. 1993; Gotelli & Colwell 2001; Chiarucci & Bonini 2005). On the other hand, the results of this study evidenced contrasting results at larger scale with the forest patch generated by still active coppicing in Bolognola being more complex and including diverse set of patches which appears to promote plant species richness and composition due to spatial turnover, even in the case of the beech specialist species. Both forest patches located in Ussita resulted less complex (lower beta diversity with respect to the Bolognola ones), with a reduced gamma diversity (50 species less), but with a higher species richness in the abandoned forest patch for all the considered plant groups (total, herb layer and beech forest specialist species).

While it was not expected to find a higher beech forest specialist species richness in coppiced patches because of the contribution of species preferring open habitat (i.e. non-forest-species, Bartha et al. 2008), which are favored by coppicing and disturbance, it is somewhat surprising to find a higher species richness of beech forest specialist at larger scales (thus a larger species pool) in the active coppice Bolognola with respect to the abandoned coppice. The difference was really minimal (33 vs. 29) but it suggests the possibility that a rich species pool of beech forest specialists can also be maintained in coppices under active management. This outcome seems to be not in line with other literature outcomes, but it should be noticed that most of the previous results were achieved at a plot or stand scale, without an upscaling as it has done here. For example, Nagaike et al. (2003) and Decocq et al. (2004), found a decrease of forest specialist species after silvicultural
treatments at the plot scale. Furthermore, Bartha et al. (2008) report a linear positive regression for beech specialist species with coppice age. Our finding on beech forest specialist species could be explained by the fact that when larger scales are considered these actively managed coppices represent very heterogeneous landscapes, due to the contemporary presence of different regeneration stages in a rather fine scale patchiness. This can maintain a heterogeneous structure of the forests and consequently a broader spectrum of ecological niches is available for plants, including many of those plants which are linked to mature beech forest stages. In fact, Bartha et al. (2008) reported a remnant of beech forest specialist species also at young stages. Mechanistic insights of a former study on the functional traits of understory forest species in the same area (Campetella et al. 2011) pointed out that beech forest specialist species may meet their habitat requirements already at a successional age of about 14 years after coppicing. Thus, the change of scale perspective leads to the view that despite a reduction of beech forest specialist species can be observed at the plot scale, when the focus is on a larger scale this is not valid any more. Practically, the species richness of this group of species is decreased at the alpha diversity level (plot scale) but not at the gamma diversity level (species pool), and this results into a higher beta diversity, i.e. a more heterogeneous distribution in the landscape. In practical terms, these results can be explained by the presence of older coppice stands randomly occurring within the active coppice forest patches in the Bolognola context. There the active forest patch could be considered as a mosaic of contiguous plots in which we can find early cuts with older coppice stands. This is depending on the type of fragmented property, private in most of cases, and to the fact that, during the last decades, due to lack of active management different coppices have been almost abandoned or managed with lower frequency (Amorini & Fabbio 2001; Ciancio et al. 2006; Canullo et al. 2009). This helps to generate a more complex mosaic of forest stands likely to maintain a larger pool of specialist species (with narrow niches).

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

The higher overstory species richness in the coppiced plots is the only convergent result between the two Municipality contexts (areas), which can be explained by a higher availability of woody species niches provided by the coppicing disturbance on the canopy. In fact, also the multiplicative partitioning of beta diversity emphasized that Municipalities (areas) provide a more important effect on the species diversity distribution with respect to the management itself (i.e. beta diversity higher between the two municipalities than between the two forest management types). This is another
unexpected result (distance between forest stands is a stronger driver than forest management in influencing species composition), probably produced by the significant effects of different forest practices that are implemented by the “local woodcutters” for the maintenance of coppicing. Despite the same management system in both municipalities, the adoption of different criteria in the selection of coppiced parcels (influencing also their spatial distribution) and of standards in terms of number and distribution (as our results demonstrated: 12.65 in Ussita, and 7.35 in Bolognola in 400 m²), can be relevant to support the obtained result in species composition similarity.

These results must be inspected more deeply in the future, trying to shed light onto the socio-economic and ecological causes of the observed habitat mosaic. The marginalization of forest areas in the Apennines calls for conservation action plans to sustain the considerable biodiversity heritage. Scale, in this respect, is an important factor. We have contributed to show that statements about the effect of forest management on diversity should always define the spatial scale they refer to, as measures considering only plot-level data might result in misleading conclusions. Our results emphasize that analyses should extend toward to the landscape scale in order to better inform and assist conservation management planning.

Practical consequences for forest management and ecosystem function

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

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

Several studies have demonstrated that coppices with standards under a regular felling regime have some negative effects, such as soil erosion, changes in bulk density or soil porosity, changes of pH levels towards soil acidification, depletion of the soil organic matter and nutrient pools (Dyck and Skinner 1990; Alban and Perala 1990; Staaf and Olsson 1994; Corona et al. 1996; Worrel and Hampson 1997). In the long term, these changes have been associated with losses of plant species diversity as well (Gilliam and Roberts 1995), so care should be taken in deeming coppicing 'sustainable'. According to our results, however, the coppice with standards in the study area seems to be sustainable in terms of the preservation of species diversity on a larger spatial scale. In fact,
the present study demonstrates that even the group of species more linked to mature forest stages (beech forest specialist species) can largely persist in a coppice landscape, if this is managed ensuring a certain heterogeneity with relatively small coppiced stands and a high variability of coppice age.

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

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

Acknowledgements

We would like to thank Sula Vanderplank and Camilla Wellstein, for their comments on a previous version of this manuscript. This study was partially funded by the “Ministero per le Politiche Agricole e Forestali” and by the Nature Reserve of Torricchio.
References


Podani, J., Czárán, T., Bartha, S. 1993. Pattern, area and diversity: the importance of spatial scale in
species assemblages. *Abstracta Botanica* 17: 37-51.


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

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

Figures
# Tables

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

<table>
<thead>
<tr>
<th>Sampling hierarchy</th>
<th>Measurement</th>
<th>Municipalities</th>
<th>Forest management types</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Alpha</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Plot</td>
<td>Mean n. of species per plot</td>
<td>Same</td>
<td>Same</td>
</tr>
<tr>
<td>Patch</td>
<td>Mean n. of species per patch</td>
<td>Same</td>
<td>Different</td>
</tr>
<tr>
<td>Patch</td>
<td>Mean n. of species per patch</td>
<td>Different</td>
<td>Same</td>
</tr>
<tr>
<td>Patch</td>
<td>Mean n. of species per patch</td>
<td>Different</td>
<td>Different</td>
</tr>
<tr>
<td>Forest management type</td>
<td>Mean n. of species per management type</td>
<td>Different</td>
<td>Same</td>
</tr>
<tr>
<td>Municipality</td>
<td>Mean n. of species per municipality</td>
<td>Same</td>
<td>Different</td>
</tr>
<tr>
<td><strong>Beta</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Plot</td>
<td>Beta diversity among the plots</td>
<td>Same</td>
<td>Same</td>
</tr>
<tr>
<td>Patch</td>
<td>Beta diversity among the patches</td>
<td>Same</td>
<td>Different</td>
</tr>
<tr>
<td>Patch</td>
<td>Beta diversity among the patches</td>
<td>Different</td>
<td>Same</td>
</tr>
<tr>
<td>Patch</td>
<td>Beta diversity among the patches</td>
<td>Different</td>
<td>Different</td>
</tr>
<tr>
<td>Forest management type</td>
<td>Beta diversity between forest management types</td>
<td>Different</td>
<td>Same</td>
</tr>
<tr>
<td>Municipality</td>
<td>Beta diversity between municipalities</td>
<td>Same</td>
<td>Different</td>
</tr>
<tr>
<td><strong>Gamma</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Plot</td>
<td>Cumulative n. of species recorded by the plots</td>
<td>Same</td>
<td>Same</td>
</tr>
<tr>
<td>Patch</td>
<td>Cumulative n. of species recorded by the patches</td>
<td>Same</td>
<td>Different</td>
</tr>
<tr>
<td>Patch</td>
<td>Cumulative n. of species recorded by the patches</td>
<td>Different</td>
<td>Same</td>
</tr>
<tr>
<td>Patch</td>
<td>Cumulative n. of species recorded by the patches</td>
<td>Different</td>
<td>Different</td>
</tr>
<tr>
<td>Forest management type</td>
<td>Cumulative n. of species recorded by the forest management types</td>
<td>Different</td>
<td>Same</td>
</tr>
<tr>
<td>Municipality</td>
<td>Cumulative n. of species recorded by the municipalities</td>
<td>Same</td>
<td>Different</td>
</tr>
<tr>
<td>Management</td>
<td>Active Coppice</td>
<td>Abandoned</td>
<td>Sign. of Management</td>
</tr>
<tr>
<td>----------------</td>
<td>-----------------</td>
<td>-------------------</td>
<td>---------------------</td>
</tr>
<tr>
<td>Forest patch (Municipalities)</td>
<td>Bol (n=20)</td>
<td>Uss (n=20)</td>
<td>Bol (n=20)</td>
</tr>
<tr>
<td>Mean standards number</td>
<td>7.35ab (±4.17)</td>
<td>11.15abc (±6.27)</td>
<td>6.47a (±5.13)</td>
</tr>
<tr>
<td>Mean standards diameter</td>
<td>20.09a (±6.68)</td>
<td>23.63c (±11.21)</td>
<td>29.36b (±10.59)</td>
</tr>
<tr>
<td>Mean standards height</td>
<td>12.41a (±3.07)</td>
<td>12.55a (±3.60)</td>
<td>20.06b (±4.59)</td>
</tr>
<tr>
<td>Number of emergent trees</td>
<td>4.80a (±3.60)</td>
<td>4.20a (±6.04)</td>
<td>1.95b (±2.28)</td>
</tr>
<tr>
<td>Number of dominant trees</td>
<td>41.20a (±33.20)</td>
<td>26.25a (±14.42)</td>
<td>29.00a (±10.36)</td>
</tr>
<tr>
<td>Number of subordinate trees</td>
<td>20.15a (±22.58)</td>
<td>24.25a (±18.39)</td>
<td>26.95a (±15.44)</td>
</tr>
<tr>
<td>Number of shrubs</td>
<td>14.80a (±21.96)</td>
<td>8.65ab (±10.03)</td>
<td>1.60b (±2.72)</td>
</tr>
<tr>
<td>Total woody individuals</td>
<td>80.95a (±52.06)</td>
<td>60.41a (±31.69)</td>
<td>59.51a (±21.94)</td>
</tr>
<tr>
<td>Total vegetation cover (%)</td>
<td>91.35a (±6.53)</td>
<td>88.20a (±5.40)</td>
<td>87.55a (±5.85)</td>
</tr>
<tr>
<td>Herb layer cover (%)</td>
<td>22.20a (±17.60)</td>
<td>6.24b (±11.55)</td>
<td>12.86ab (±11.49)</td>
</tr>
<tr>
<td>Deadwood cover (%)</td>
<td>2.95a (±2.19)</td>
<td>2.97a (±1.62)</td>
<td>8.25b (±5.40)</td>
</tr>
</tbody>
</table>

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=Bologna Municipality; Uss=Usita Municipality; n=number of relevees)
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).

<table>
<thead>
<tr>
<th>Management</th>
<th>Active Coppice</th>
<th>Abandoned</th>
<th>Sign. of Management</th>
<th>Sign. of Area</th>
</tr>
</thead>
<tbody>
<tr>
<td>Forest patch (Municipalities)</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Bol (n=20)</td>
<td>28.35b (±10.60)</td>
<td>34.95b (±12.60)</td>
<td>35.30b (±13.00)</td>
<td>0.068 0.023</td>
</tr>
<tr>
<td>Uss (n=20)</td>
<td>5.70ab (±2.77)</td>
<td>5.85ab (±2.41)</td>
<td>3.90b (±1.70)</td>
<td>0.006 0.002</td>
</tr>
<tr>
<td>Bol (n=20)</td>
<td>22.65b (±10.22)</td>
<td>31.35b (±12.73)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Uss (n=20)</td>
<td>5.15c (±1.69)</td>
<td>6.25c (±2.14)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Mean vascular species n.</td>
<td>55.05a (±17.08)</td>
<td>34.95b (±12.60)</td>
<td>35.30b (±13.00)</td>
<td>0.068 0.023</td>
</tr>
<tr>
<td>Mean vascular overstory species n.</td>
<td>7.15a (±2.79)</td>
<td>5.85ab (±2.41)</td>
<td>3.90b (±1.70)</td>
<td>0.006 0.002</td>
</tr>
<tr>
<td>Mean vascular understory species n.</td>
<td>47.90a (±17.37)</td>
<td>31.35b (±12.73)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Mean specialist species n.</td>
<td>8.80a (±3.12)</td>
<td>12.40b (±3.29)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Shannon ($H'$) woody individuals</td>
<td>1.09a (±0.45)</td>
<td>0.74a (±0.53)</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

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

<table>
<thead>
<tr>
<th>Municipality</th>
<th>Forest patch management type</th>
<th>Gamma</th>
<th>Mean alpha plot</th>
<th>Beta plot</th>
</tr>
</thead>
<tbody>
<tr>
<td>USSITA (USS)</td>
<td>abandoned</td>
<td>155</td>
<td>35.25</td>
<td>4.40</td>
</tr>
<tr>
<td></td>
<td>coppiced</td>
<td>124</td>
<td>28.35</td>
<td>4.37</td>
</tr>
<tr>
<td>BOLOGNOLA (BOL)</td>
<td>abandoned</td>
<td>125</td>
<td>34.95</td>
<td>3.58</td>
</tr>
<tr>
<td></td>
<td>coppiced</td>
<td>222</td>
<td>55.05</td>
<td>4.03</td>
</tr>
</tbody>
</table>
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)  

<table>
<thead>
<tr>
<th>PATCH (within the same municipality)</th>
<th>Gamma</th>
<th>Mean alpha patch</th>
<th>Beta patches</th>
</tr>
</thead>
<tbody>
<tr>
<td>USS-coppiced/USS-abandoned</td>
<td>192</td>
<td>139.50</td>
<td>1.38</td>
</tr>
<tr>
<td>BOL-coppiced/BOL-abandoned</td>
<td>239</td>
<td>173.50</td>
<td>1.38</td>
</tr>
</tbody>
</table>

(b)  

<table>
<thead>
<tr>
<th>PATCH (within the same forest management type)</th>
<th>Gamma</th>
<th>Mean alpha patch</th>
<th>Beta patches</th>
</tr>
</thead>
<tbody>
<tr>
<td>BOL-coppiced/USS-coppiced</td>
<td>246</td>
<td>173</td>
<td>1.42</td>
</tr>
<tr>
<td>BOL-abandoned/USS-abandoned</td>
<td>196</td>
<td>140</td>
<td>1.40</td>
</tr>
</tbody>
</table>

(c)  

<table>
<thead>
<tr>
<th>PATCH (between different municipalities and forest management type)</th>
<th>Gamma</th>
<th>Mean alpha patch</th>
<th>Beta patches</th>
</tr>
</thead>
<tbody>
<tr>
<td>BOL-abandoned/USS-coppiced</td>
<td>173</td>
<td>124.50</td>
<td>1.39</td>
</tr>
<tr>
<td>USS-abandoned/BOL-coppiced</td>
<td>265</td>
<td>188.50</td>
<td>1.41</td>
</tr>
</tbody>
</table>
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               |