



## 4 Conservation

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# Conservation of Coppice and High Forest Management within the Natura 2000 Network – A Review

Peter Buckley and Jenny Mills

## ABSTRACT

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The Natura 2000 network protects some of the most threatened species and habitats in the European Union, of which forests account for about 50% of the total designated area. This paper examines the broad habitat preferences of the terrestrial species listed in Annexes of the Birds and Habitats Directives, of which a majority are associated with non-forest habitats. By comparison, European red lists and the various country and regional level lists of species of principal importance contain many more species and species groups than the Directive Annexes. Foresters are likely to use a much narrower suite of species, often based only on the Annexes, when setting practical conservation targets for woodlands.

Achieving the objective of ‘favourable conservation status’, as required by the Directives, should apply equally to the designated forest habitat types and their listed specialist species.

European Commission literature describes these habitats in terms of their typical tree, shrub and herbaceous species, although in practice a mixture of iconic and specialist Annex species may be used for making conservation assessments. Recognising the value of traditional coppice and its long anthropogenic history can be considered a valid reason for conservation in itself, but this form of management is now in serious decline all over Europe. High forests and old growth habitats, together with their associated species, also have equal claims for protection under the Natura 2000 network. Given the difficulty of simultaneously achieving species and habitat targets in the context of both early and late-successional aspects of forest conservation, we consider different silvicultural strategies that may achieve wider biodiversity benefits in the forest environment.

### Key words

Natura 2000, Birds and Habitat Directives Annex species, forest habitat type, indicator species, coppice, silvicultural system

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## INTRODUCTION

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Some of the most valued and threatened species and habitats in Europe are protected within the Natura 2000 network under the Birds Directive (European Commission 1979) and the Habitats Directive (European Commission 1992). The latter Directive targets more than 230 ‘habitat types’ and 1500 animal and plant species for conservation in its various Annexes, many of which are rare, threatened or endemic. They include 303 animals, 586 plants (Habitats Directive Annex II, HDII) and more than 190 birds (Birds Directive Annex I, BDI). For a further 400 species and sub-species listed in Annex IV of the Habitats Directive (HDIV), which includes many that are also listed in HDII, a strict protection regime must be applied across their entire natural range in the European Union (EU), both within and outside Natura 2000 sites.

Approximately 375,000 km<sup>2</sup> of forests are included in the Natura 2000 Network, representing around 50% of its total area and 21% of the total forest resource in the EU (European Commission 2015). A large proportion of this forest would undoubtedly have been coppiced in the past: based on the average of 24 European countries, up to 15% of the area is presently classified as coppice, together with a probably much greater extent of neglected or converted former coppices (Buckley and Mills, 2015). Considering the large protected area and the strong emphasis given to conserving the threatened biodiversity of forest ecosystems within the EU, one would anticipate that a high proportion of Directive-protected species would be found in, or be dependent on, forested habitats. To discover whether this is the case, the habitat preferences of species listed in BDI

and HDII were investigated. The contribution that the traditional forestry techniques of coppicing and pollarding can bring to the protection of biodiversity in Natura 2000 sites was also considered.

Many of the species on the BDI and HDII lists are species of conservation concern, judged as vulnerable or under threat by the International Union for Conservation (IUCN). We consider the composition of different taxa making up these lists, their endemism, threat status, and their preferences for forest habitats or other, more open ones. In the case of forest and woodland habitats, the definition of ‘favourable conservation status’, as applied by the Bird and Habitat Directives to both habitats and species, especially more ‘typical’ species as well as the Natura 2000 species, depends on the ability of different forest management regimes to conserve them. Here we focus initially on traditional coppice forest management, a widespread but now rapidly disappearing silvicultural practice in Europe, and the implications that abandonment or conversion to high forest might have for protecting habitats and species. At the same time we consider what additional protected species niches high forest systems might provide. Finally, we discuss management strategies that might deliver combinations of both early and late-successional growth stages, and which may serve to increase species diversity in forested landscapes.

## METHODS: ALLOCATING BROAD HABITAT PREFERENCES TO SPECIES

Using the HDII and BDI Annexes, each protected species was allocated to one or a number of broad habitat types, using the hierarchical classification proposed by the European Environment Agency (EUNIS) (European Environment Agency n.d.). The EUNIS species browser (<http://eunis.eea.europa.eu/species.jsp>) lists the ‘most preferred habitats’ in its quick facts for nearly all of these species. These, excluding fish, were allocated to the 10 EUNIS hierarchical habitats (<http://eunis.eea.europa.eu/habitats-code-browser.jsp>) described in Table 1. If no habitats were listed for a species on the EUNIS database, the world IUCN Red List species details (<http://www.iucnredlist.org/details/>) were consulted.

When not listed in either database, it was recorded in the ‘Insufficient data’ column, except for fewer than 10 cases where information was taken, for example, from Wildscreen ARKive (<http://www.arkive.org>), EEA Eionet (<https://www.eionet.europa.eu>), Joint Nature Conservation Committee (<http://jncc.defra.gov.uk>), Environment Directorate General of the European Commission ([http://ec.europa.eu/environment/index\\_en.htm](http://ec.europa.eu/environment/index_en.htm)) and Birdlife International (<http://www.birdlife.org>)

While recording this data, it was also noted if a species was on the IUCN Red List and if it was an endemic.

Table 1. Summary of 10 broad habitat types and their descriptions, based on the hierarchical classification proposed by the European Environment Agency (EUNIS)

1	<b>Marine</b>	Marine habitats: fully saline, brackish or almost fresh. Includes marine littoral habitats including tidal saltmarshes; marine littoral habitats and strandlines; waterlogged littoral saltmarshes and associated saline or brackish pools.
2	<b>Coastal</b>	Habitats are those above spring high tides, including coastal dunes and wooded coastal dunes, beaches and cliffs. Supra-littoral habitats include strandlines, moist and wet coastal dune slacks and dune-slack pools.
3	<b>Inland surface waters</b>	Non-coastal fresh or brackish waterbodies (rivers, streams, lakes and pools, springs), including their littoral zones. Also constructed waterbodies (canals, ponds, etc.) supporting semi-natural communities and seasonal waterbodies.
4	<b>Mires, bogs and fens</b>	Wetlands, with the water table at or above ground level for at least half of the year, dominated by herbaceous or ericoid vegetation. Includes inland saltmarshes and waterlogged habitats where the groundwater is frozen.
5	<b>Grasslands</b>	Dry or only seasonally wet land with >30% vegetation cover. Dominated by grasses and other non-woody plants, including mosses, macro-lichens, ferns, sedges and herbs. Includes semiarid steppes, successional weedy vegetation and managed grasslands (e.g. recreation fields and lawns).
6	<b>Heathland</b>	Dry or only seasonally inundated land with >30% vegetation cover. Includes tundra; heathland dominated by shrubs or dwarf shrubs not above 5m tall. Also shrub orchards, vineyards, hedges, climatically-limited dwarf trees (krummholz) >3 m high, <i>Salix</i> and <i>Frangula</i> carrs.
7	<b>Woodland</b>	Dominated by trees over 5m, with a canopy cover of at least 10%. Includes lines of trees, coppices, tree nurseries, plantations and fruit and nut tree orchards. Includes <i>Alnus</i> and <i>Populus</i> swamp woodland and <i>Salix</i> . Excludes <i>Corylus avellana</i> scrub and <i>Salix</i> and <i>Frangula</i> carrs.
8	<b>Sparsely vegetated</b>	Habitats with less than 30% vegetation cover which are dry or only seasonally wet. Includes caves and passages including underground waters and disused underground mines, and habitats with permanent snow and surface ice.
9	<b>Cultivated</b>	Habitats maintained solely by frequent tilling or recently abandoned arable land and gardens.
10	<b>Constructed</b>	Primarily human settlements, buildings, industrial developments, transport networks and waste dumps. Includes artificial saline and non-saline waters with wholly constructed beds or heavily contaminated water, virtually devoid of plant and animal life.

## RESULTS

### Species groupings

We calculate that 80-90% of BDI and HDII species are also registered on the International Union for Conservation of Nature Red List of Threatened Species (IUCN 2015), which classifies species on the basis of their relative extinction risk, consistent with their need for protection (Fig. 1). Relative to their species numbers, plants, birds and mammals are well represented, but some taxa, such as the arthropods, have received less attention, with under 50% of HDII species recorded on the world Red List, perhaps reflecting the relative scarcity of specialists dealing with this numerous group. Moreover, the species chosen for protection under HDII and HDIV are subject to taxonomic, geographic and aesthetic bias, with preferences given to larger, iconic species, but also including many that are widespread (Cardoso 2012).

This bias is evident in the relative dominance of vertebrates compared with very few in the arthropod group, which in turn is biased towards Lepidoptera and Coleoptera, while completely lacking large insect Orders such as Diptera and Hymenoptera. Although plant species make up the largest group in HDII, only 32 bryophytes and no fungi or lichens are included (Orlikowska et al. 2016).

### Endemicity and threat status

Listing of HDII species is heavily influenced by their endemic status. Overall, 415 primarily terrestrial species or subspecies (41.7%) are strict endemics, i.e. restricted to one EU country or to Macaronesia. Plants and molluscs have the highest share of endemic taxa (63.8% and 48.3% respectively), with reptiles and amphibians intermediate and breeding birds

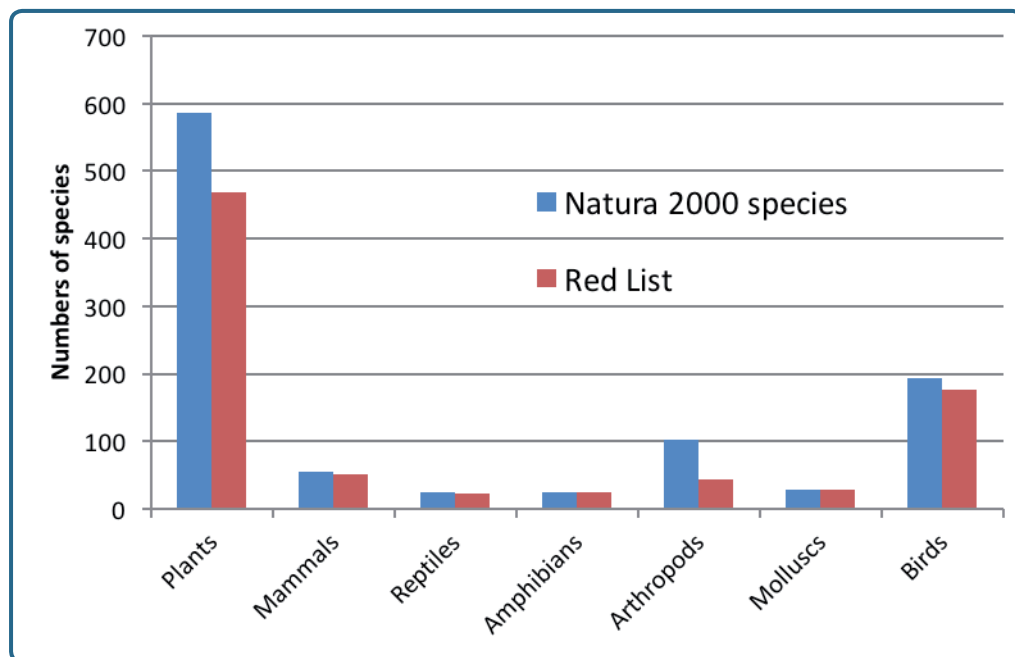


Figure 1. Numbers of BDI and HDII species and on the world IUCN Red List present in each taxonomic group, excluding fish

the lowest (4.9%) (Fig. 2). The low number of arthropods (8.3%) almost certainly reflects an incomplete assessment of this very diverse group. Macaronesian plant species, being by definition full endemics, make up over a quarter of all HDII plants, while of the non-Macaronesian plants, 55.9% are also strictly endemic.

Nearly half of BDI and HDII species (48%) fell into the threatened categories (critically endangered, endangered, vulnerable and near-threatened) on the world Red List.

The figures were (Fig. 3):

- 87% for reptiles,
- 68% for molluscs,
- 55% for plants,
- 52% for amphibians
- 43% for mammals,
- 36% for arthropods
- 21% for birds.

While reptiles, molluscs and plants were relatively more threatened, many mammals, amphibians and birds were of 'least concern' on the IUCN World Red List, but when viewed in a narrower European context, several species may be perceived as more threatened.

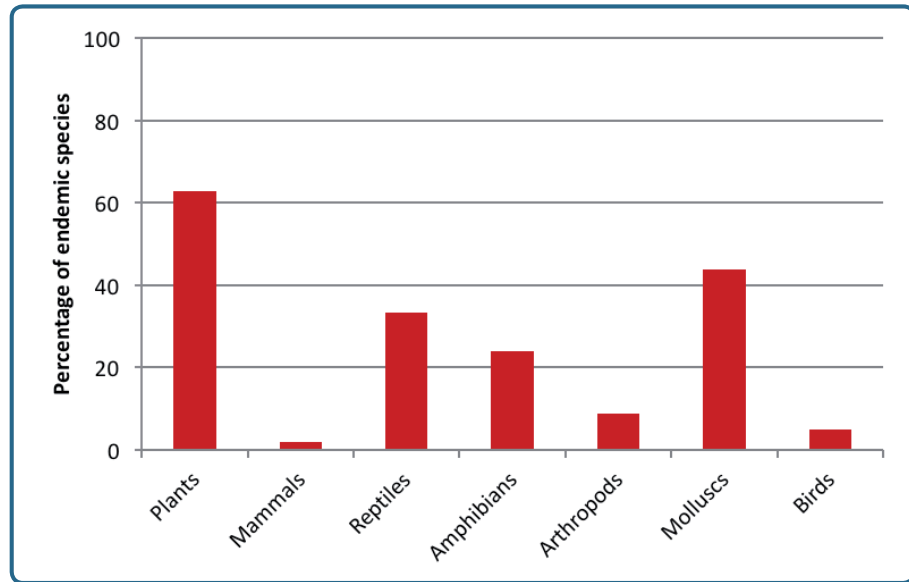


Figure 2. Percentage of BDI and HDII endemic species, by taxonomic groups (excluding fish)

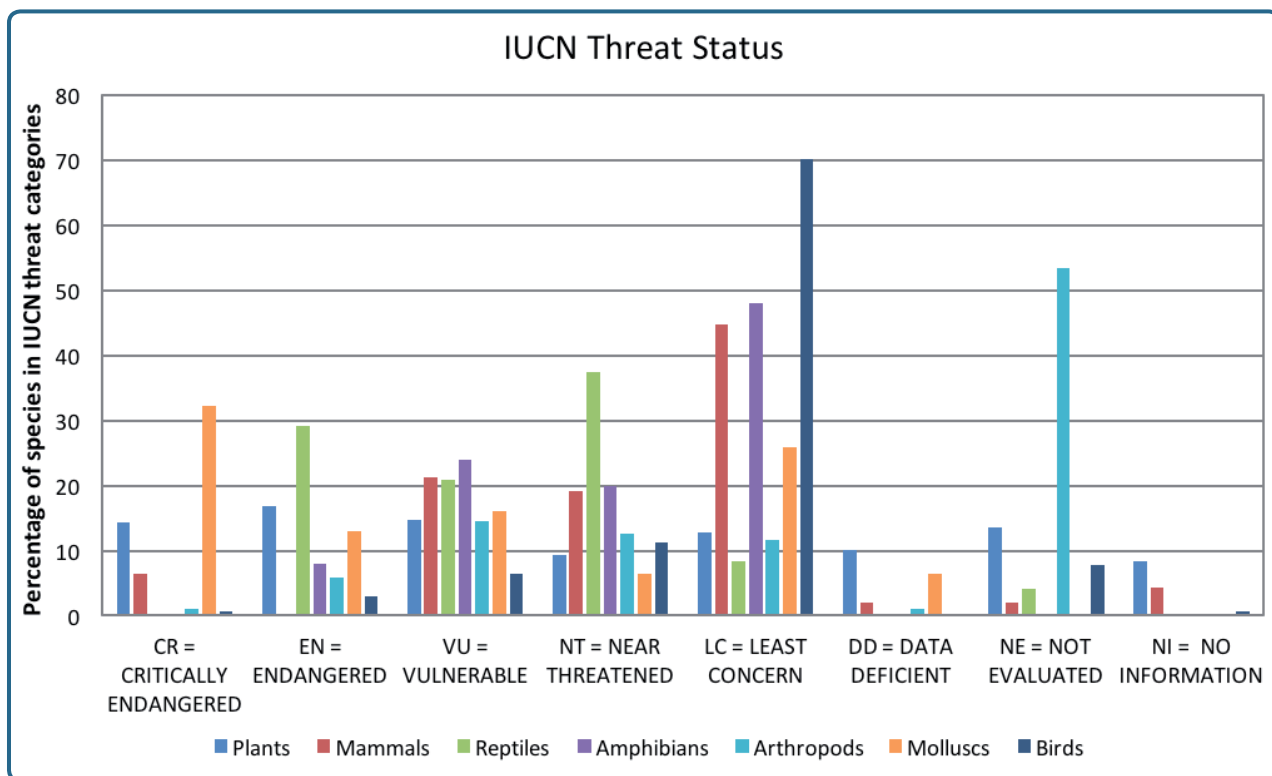


Figure 3. Percentage of BDI and HDII species from different terrestrial taxonomic groups in the IUCN world list categories

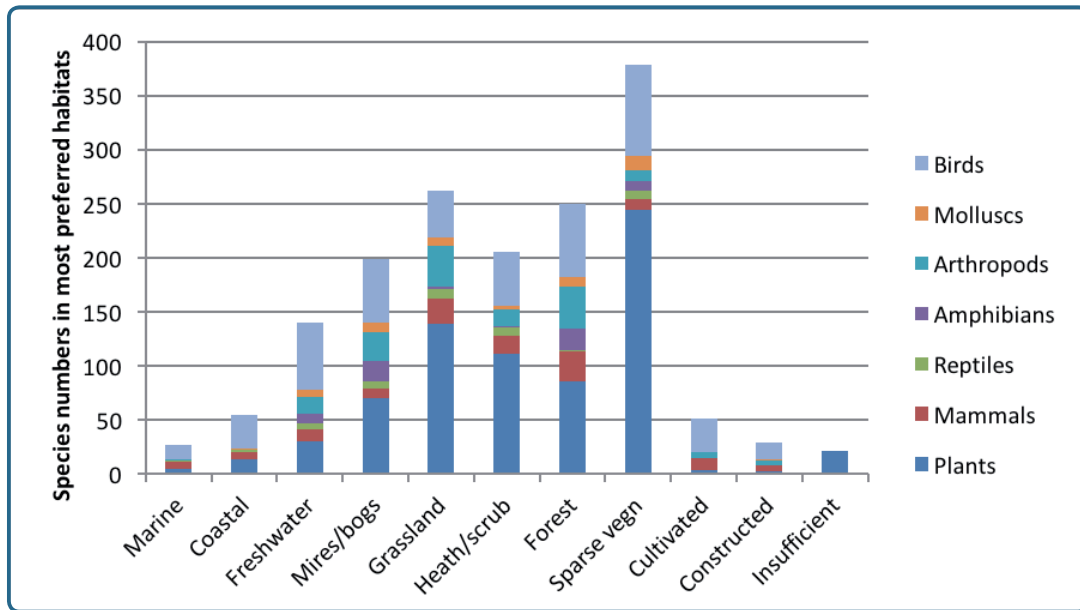


Figure 4. Numbers of BDI and HDII species occurring in different EUNIS habitat types

### Habitat distributions of protected species

The most frequent preferred species habitats were in sparsely vegetated habitats, with grasslands, forests, heathlands and wetlands intermediate, and relatively few in marine, coastal, cultivated and construction sites (Fig. 4). Several plant species were given preferred habitat status in sparse vegetation, although many could also be categorised more specifically as species of sand dunes, cliffs, tundra and alpine habitats. Of particular interest was the ‘forest and woodland’ category, which contained relatively balanced proportions of the different taxonomic groups compared with other categories, including a comparatively high number of arthropods, amphibians and mammals, although relatively fewer plants and reptiles than in other open habitats. As forests cover such a large part of the Natura 2000 network, it is not surprising that they shelter a large number of Directive-protected species. Collectively, however, the great variety of more open habitats (e.g. sparse vegetation, grassland, heath, etc.) contain significantly more. The vast majority of these BDI and HDII species appeared to be associated with non-forest or relatively open conditions.

### Spatial hierarchies of protected species

Lists of rare species tend to become more refined as the area of interest narrows. A hierarchical gradient taken from the IUCN world perspective, diminishing in scope for Natura 2000 and the European Red Lists, and further to the more localised level of countries and regions, shows that species lists of principal conservation importance often tend to become more focused and lengthier (Table 2). In separate European countries and regions, protected species lists are generally focused more at this level than at the BDI and HDII Annex level: those species relatively widely distributed at a European level effectively become ‘rarer’ at a local level, and therefore more notable. Compared with the BDI or HDII species annexes, European Red Lists contain many more species, often more than three times the number. This is particularly obvious for invertebrate Red Lists of dragonflies (Kalkman et al. 2010), saproxylic beetles (Nieto and Alexander 2010), non-marine molluscs (Cuttlelod et al. 2011), butterflies (van Swaay et al. 2011) and bees (Nieto et al. 2014). At a national level the picture is even more variable: in Britain, for example, as would be expected



from this country's size and its history of glacial impoverishment, the numbers of vascular plants, mammals, reptiles and amphibians were lower than the equivalent BDI and HDII annexes and European red lists, but a greater effort has been made to cover non-vascular

plants, invertebrates, fungi and lichens. In other countries and regions, such as France, Estonia and Flanders, the same tendency to specialise within some of the broader taxonomic groups is seen (Table 2).

Table 2. Numbers of terrestrial species (mostly terrestrial or freshwater) selected at different spatial levels for biodiversity conservation: the IUCN world red list, the BDI and HDII, the IUCN European red lists, UICN French red lists, the UK Biodiversity Action Plan, Estonian protected species and Flanders red lists

Taxonomic group	HDII and BDI species on IUCN world red list <sup>1</sup>	HDII and BDI species (Natura 2000) <sup>2,3</sup>	European red list (EU27)	France red lists	Britain – species of principal importance <sup>4</sup>	Estonian protected species <sup>5</sup>	Flanders red lists
<b>Vascular plants</b>	412	554	*1750 <sup>6</sup>	1018 <sup>7,8</sup>	382	215	1152 <sup>9</sup>
<b>Non-vascular plants</b>	1	32	* †		552	46	
<b>Mammals</b>	45	47	179 <sup>10</sup>	99 <sup>11</sup>	25	18	65 <sup>12</sup>
<b>Total invertebrates</b>	75	135			597	52	
Dragonflies	11	11	*134 <sup>13</sup>		3	5	64 <sup>14</sup>
Saproxylic beetles	9	17	*408 <sup>15</sup>		10	3	19 <sup>16</sup>
Molluscs	31	31	*1805 <sup>17</sup>		29	4	
Lepidoptera	14	38	421‡ <sup>18</sup>	253‡ <sup>19</sup>	195	10	72‡ <sup>20</sup>
Bees	0	0	1900 <sup>21</sup>		44	18	
<b>Reptiles</b>	23	24	128 <sup>22</sup>	35 <sup>23</sup>	6	5	6 <sup>24</sup>
<b>Amphibians</b>	25	25	82 <sup>25</sup>	35 <sup>23</sup>	4	11	16 <sup>24</sup>
<b>Fungi/lichens</b>	0	0	0		782	97	
<b>Birds</b>	150	162	399 <sup>26</sup>	345 <sup>27</sup>	105	93	200 <sup>28</sup>

\*to be completed by 2018 † bryophytes only ‡ butterflies only

<sup>1</sup>IUCN (2015) <sup>2</sup>European Commission (1992) <sup>3</sup>European Commission (1979) <sup>4</sup>Natural Environment and Rural Communities (NERC) Act (2006a, 2006b), Nature Conservation (Scotland) Act (2004) <sup>5</sup>Riigi Teataja (2014a, 2014b) <sup>6</sup>Bilz et al. (2011) <sup>7</sup>UICN France et al. (2012) <sup>8</sup>UICN France et al. (2009) <sup>9</sup>Van Landuyt et al. (2006) <sup>10</sup>Temple and Terry (2007) <sup>11</sup>UICN France et al. (2009) <sup>12</sup>Maes et al. (2014) <sup>13</sup>Kalkman et al. (2010) <sup>14</sup>De Knijf (2006) <sup>15</sup>Nieto and Alexander (2010) <sup>16</sup>Thomaes et al. (2015) <sup>17</sup>Cuttelod et al. (2011) <sup>18</sup>van Swaay et al. (2010) <sup>19</sup>UICN France et al. (2014) <sup>20</sup>Maes et al. (2011) <sup>21</sup>Nieto et al. (2014) <sup>22</sup>Cox and Temple (2009) <sup>23</sup>UICN France et al. (2015) <sup>24</sup>Jooris et al. (2012) <sup>25</sup>Temple, Cox (2009) <sup>26</sup>Birdlife International (2015) <sup>27</sup>UICN France et al. (2011) <sup>28</sup>Devos et al. (2004).



At forest species protection level, Britain's state forestry service (the Forestry Commission) has produced a web-based decision support system for its managers dealing with Habitats and Rare, Priority, and Protected species (HaRPPS). This provides information on about 123 woodland species, including:

- 25 mammals,
- 37 birds,
- 4 herptiles,
- 21 invertebrates,
- 13 vascular plants and
- 23 fungi and lower plants (Forest Research 2011),

allowing forest managers to predict which species might be present in a given area and to test the impact of forest operations on them.

Although the British lists of species of principal importance for conservation cover all habitats, including forests (NERC Act 2006a,b; Nature Conservation (Scotland) Act 2004) there are big disparities with HaRRPS for different taxonomic groups: mammals, birds and herptiles are well covered, whereas vascular plants, bryophytes, liverworts and invertebrates are not (Fig. 5). Practising forest managers should be able to identify iconic animals and birds in their well-protected groups, but are less likely to have specialist knowledge of some invertebrates, fungi, vascular and non-vascular plants.

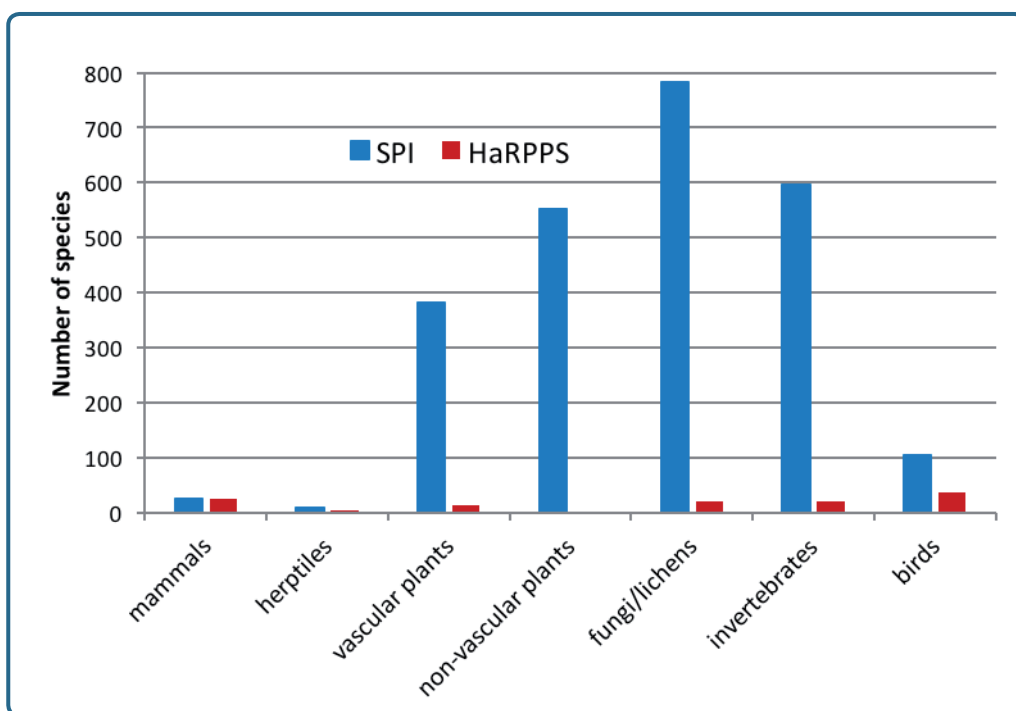


Figure 5. Numbers of species of principal importance (SPI) in Britain by taxonomic group, relative to that of the Forestry Commission's information system for use in woodland habitats (Habitats and Rare, Priority and Protected Species HaRPPS) (Forest Research 2011)

### Favourable conservation status

The emphasis placed on rare or iconic species is not always effective in promoting species diversity, as the overriding issue for forest species is fundamentally the protection of their habitat and its quality. However, when compiling the Standard Data Forms for the designation of Natura 2000 sites, agencies tend to focus on rare species, irrespective of whether they are only a fraction of a metapopulation that extends beyond the boundary of the protected area (Battisti and Fanelli 2014). In fact, in terms of ecological integrity, achieving a 'favourable conservation status', a legal requirement of Natura 2000 designation, applies to any 'typical species' of a HDI habitat (Rees et al. 2013). The Directive applies equally to the habitat, which must be stable or increasing and likely to sustain its structure and function for the foreseeable future. The reality is that only 15% of the protected forest habitats in the EU are reported as being in a favourable condition due to multiple factors, such as fires, disease, browsing, pollution, urbanisation, etc., but mainly to forest and plantation management, such as the removal of dead and dying trees (European Commission 2015). Among the human activities reported on Standard Data Forms, agriculture and forestry were associated with more than 86% of a sample of Natura 2000 sites, of which forestry activities affected 59% (Tsiafouli et al. 2013). Many broadleaved forest HDI habitats described as 'Temperate Forests of Europe' in the European Commission's Interpretation Manual of European Union Habitats EU28 (European Commission 2013) have the potential to be coppiced, based on the re-sprouting potential of the dominant trees (Mairota et al. 2016), although most is

now high forest. The summary descriptions of each forest habitat type are of essentially widespread or characteristic plant species (Table 3), including several relatively common herbs and grasses, which depend on the forest margins and the more frequently open canopies that could be provided by coppice management. Very few HDII species (i.e. rarities and endemics) are listed. When this suite of 'typical', widespread species is present, it follows that a 'favourable conservation status' is more likely to be achieved for rarer ones.

To support the regular monitoring of Natura 2000 sites a range of species specialists associated with long-term anthropogenic management of their forest habitat could be identified, as recognised by the Habitats Directive (Epstein et al. 2015). Such 'indicator species' would not necessarily be rare endemics or HDII species, but could represent several taxa, including vascular plants, bryophytes, wood-decaying fungi, epiphytic lichens, saproxylic beetles and land snails (Nordén et al. 2014). Some of these are more properly indicators of traditional high forest or old growth, but many ancient woodland 'indicator plants' with limited dispersal characteristics (*sensu* Hermy et al. 1999; Verheyen et al. 2003; Kimberley et al. 2013) are also associated with former coppice habitats; Decocq et al. (2005) even suggested that they might be better labelled 'coppice-woodland species'. In northwest Germany, Schmidt et al. (2014) listed 67 ancient woodland indicator plants, most of them typical of closed forests, but with 13% preferring forest edges and clearings, while Pellisier et al. (2013) identified 40 'core' and 38 'periphery' forest species based on a large database of over 1800 forest patches in northern France.

Aesthetic as well as biodiversity criteria can be taken into account in species protection. In the Zurich Canton of Switzerland, aesthetic criteria were involved in an action plan to restore the typical flora (from a target list of 172 species) associated with 'light' or open-canopied forests, which was carried out on a portion of the total forest area of 47,500 ha (Bürgi et al. 2010). The areas selected were based on an analysis of the target species and forest management practices, recognising not only anthropocentric history but also the ecological continuity of coppice habitats within the region, much in the spirit of the Habitats Directive.

### Provision for coppice specialists

Traditional coppice management, often based on regular short rotations over centuries, has produced a habitat for species that are adapted to the dynamic of rapidly altering light, temperature and hydrological regimes (Peterken 1993, Rackham 2003, Szabo 2010). These regular, intense pulses of disturbance tend to boost the diversity in both the ground flora and shrub layers (Ash and Barkham, 1976; Decocq et al. 2004; Brunet et al. 2010; Verheyen et al. 2012; Campetella et al. 2016). The transient woodland structure produced is important for many songbirds that forage and nest in young growth, as well as for other open-ground

**Table 3.** Species with frequencies of 10% (2/20) or more that are named in the summaries of 20 different forest habitat types from the 'Forests of Temperate Europe' (Annex 1 code 9100); the list is based on 26,433 Natura 2000 sites where at least 100 sites are devoted to each forest habitat type

<b>Trees</b>	/20		/20		/20
<i>Quercus petraea</i>	8	<i>Acer tartaricum</i>	3	<i>Ilex aquifolium</i>	2
<i>Fagus sylvatica</i>	6	<i>Betula pubescens</i>	3	<i>Populus nigra</i>	2
<i>Quercus cerris</i>	6	<i>Fraxinus angustifolia</i>	3	<i>Populus tremula</i>	2
<i>Quercus robur</i>	6	<i>Fraxinus excelsior</i>	3	<i>Quercus pyrenaica</i>	2
<i>Carpinus betulus</i>	5	<i>Euonymus verrucosus</i>	3	<i>Sorbus domestica</i>	2
<i>Acer campestre</i>	4	<i>Picea abies</i>	3	<i>Taxus baccata</i>	2
<i>Sorbus torminalis</i>	4	<i>Quercus pubescens</i>	3	<i>Tilia tomentosa</i>	2
<i>Tilia cordata</i>	4	<i>Acer pseudoplatanus</i>	2	<i>Ulmus glabra</i>	2
<i>Abies alba</i>	3	<i>Alnus glutinosa</i>	2	<i>Ulmus minor</i>	2
<b>Shrubs</b>	/20		/20		/20
<i>Euonymus verrucosus</i>	3	<i>Frangula alnus</i>	2	<i>Vaccinium myrtillus</i>	2
<i>Ligustrum vulgare</i>	3	<i>Pyrus pyraster</i>	2		
<i>Buxus sempervirens</i>	2	<i>Ruscus aculeatus</i>	2		
<b>Herbaceous Species</b>	/20		/20		/20
<i>Carex montana</i>	4	<i>Anemone nemorosa</i>	2	<i>Hieracium sabaudum</i>	2
<i>Dentaria</i> spp.	4	<i>Buglossoides purpureocaerulea</i>	2	<i>Lathyrus niger</i>	2
<i>Festuca heterophylla</i>	3	<i>Carex michelii</i>	2	<i>Luzula forsteri</i>	2
<i>Knautia drymeia</i>	3	<i>Cyclamen purpurascens</i>	2	<i>Molinia caerulea</i>	2
<i>Potentilla micrantha</i>	3	<i>Galium schultesii</i>	2	<i>Potentilla alba</i>	2
<i>Pteridium aquilinum</i>	3	<i>Galium sylvaticum</i>	2	<i>Pulmonaria mollis</i>	2
<i>Tanacetum corymbosum</i>	3	<i>Helleborus odorus</i>	2	<i>Tamus communis</i>	2

foragers (Camprodon and Brotons 2006; Fuller 2012). After coppicing, the resulting sunny and warm microclimate creates suitable conditions for a range of butterflies, macromoths and other invertebrates (e.g. Sparks et al. 1996; Fartmann et al. 2013; Horák et al. 2014), which take advantage of increased understorey flowering and abundant sources of pollen and nectar.

While many thermophilic and opportunistic species are cosmopolitan, others are more restricted to the coppice habitat. They include many vascular plants tolerant of intermittent shading, accompanied by a large insect biomass dependent on flowers and young foliage (Warren and Key 1991; Greatorex-Davies and Marrs 1992). In order to maintain viable populations, sufficient canopy openings and forest margins must be present, whether created anthropogenically or by a natural disturbance dynamic. Some beneficiaries that are specialists of the coppice habitat are considered of high conservation value: there are examples of conservation coppicing carried out expressly to support a single species or group of species. Examples are rare butterfly populations such as the Scarce Fritillary (*Euphydryas maturna*) and many others that are not necessarily listed in HDII and HDIV (e.g. van Swaay et al. 2006; Kobayashi et al. 2010; Fartman et al. 2013; Dolek et al. 2018). Very low densities of standards in coppice, covering as little as 10-15% of the stand, have been recommended in order to maintain open conditions for butterfly conservation (Clarke et al. 2011). Coppicing may also be maintained specifically for other iconic species such as the hazel grouse (*Bonasa bonasia*), where coppice provides a substitute for its optimum forest habitat of shrub layers in gaps of old-growth forests (Kajtoch et al. 2012), for migrant songbirds that nest and forage in scrub (e.g. *Sylvia* species), and small mammals such as the hazel dormouse (*Muscardinus avellanarius*) (Ramakers et al. 2014; Sozio et al. 2016).

Many other species also benefit from the openings created by coppicing. However, in long-neglected or converted coppice stands, plant species diversity and some red-listed herb layer species tend to diminish rapidly (Van Calster et al. 2008a; Kopecky et al. 2013; Vild et al. 2013; Müllerová et al. 2015). In formerly grazed and coppiced sub-continental oak forest in the Czech Republic, these declining and endangered species tended to persist in locations with high light availability and relatively higher pH (Roleček et al. 2017). Similarly, in comparing vegetation data from still-active selection coppices with beech-dominated high forests in the Banat region in Romania, the coppices were slightly more diverse, containing thermophilous and non-forest species more typical of more open grassland habitats, although they were similar in herb species richness to high forests (Šebesta et al. 2017). The re-application of traditional forest management practices may be able to reverse successional tendencies in long-abandoned or converted former coppices. In lowland thermophilous oak forest, restoration of a litter-raking treatment effectively increased the richness and cover of both forest and dry grassland species over a 5-year period (Douda et al. 2017). The restoration of canopy thinning, analogous to coppicing, in a long abandoned ancient coppice-with standards woodlands, has been shown to potentially support and revive light-demanding woodland floras (Vild et al. 2013) and also to increase the functional diversity responses of plant and ground-dwelling spider communities (Šipoš et al. 2017).

Several researchers have shown that vascular plants in the herb layer of beech forests were marginally more diverse in managed stands or after disturbance at the plot level, compared with unmanaged stands, later to decline with neglect (e.g. Schmidt 2005; Bartha et al. 2008; Garadnai et al. 2010; Mölder et al. 2014). At the patch level, Campetella et al. (2016) showed

that a rich species pool of specialist plants associated with beech forest in the Central Apennines could be maintained under active management, i.e. within a landscape mosaic comprising different woodland development stages. In the same region, Scolastrì et al. (2016) found that beech forests, whether classified as old coppice-with-standards or as high forest, contained many heliophilous plants indicative of past light regimes, as well as many shade-tolerant, understorey species typical of 9210\* Apennine beech forests with *Taxus* and *Ilex* recognised in the European Commission's Habitat Directive Interpretation Manual (European Commission, 2013). Cervillini et al. (2017) considered that with canopy cover stabilising between 10 and 16 years, approximately 10 years before coppice harvesting, many such specialists of shaded beech forests were able to persist.

### Conversion to high forest

Coppices gradually change their biological character when they are abandoned or are converted into high forests. Several long-term studies have investigated the vegetational and edaphic changes resulting from this transition in European forests (Debussche et al. 2001; Peterson 2002; Decocq et al. 2004, 2005; Van Calster et al. 2007, 2008b; Baeten et al. 2009; Verheyen et al. 2012; Kopecký et al. 2013; Verstraeten et al. 2013; Becker et al. 2016). Most of these recorded a decline in species-richness of the tree, shrub and herb layers, with homogenisation increasing under the shade cast by a developing canopy, together with increases in shade-tolerant, vernal and eutrophic species.

Changes in the vegetation, such as increasing tree cover, may be happening in parallel with coppice abandonment, frequently detected in signals of eutrophication and acidification resulting from increased atmospheric deposition (Verheyen et al. 2012), as well as potential climate change. Peterson (2002), investigating

a chronosequence of sample plots in ageing coppice in Denmark (median age = 40 years), suggested that increasing shade, together with the build-up of acidifying litter, tended to reduce species density and to favour clonal forest species. In Belgium, Van Calster et al. (2007) also reported increases in soil acidity in coppice-with-standards undergoing conversion to high forest from 1967-2005, at least partly explained by the poor litter quality under canopies of *Fagus sylvatica* and *Quercus robur*. In recordings made over an interval of 50 years, Verstraeten et al. (2013) found that the species pool of understorey herbs in former coppice-with standards generally declined, as did Ellenberg light indicator values, while those for nitrogen availability increased. The high input of atmospheric deposition within this period shifted the plant community towards a more N-demanding and shade-tolerant type.

In Germany, similar observations were made by Becker et al. (2016) in coppice-with-standards woodlands which had been in conversion for c. 100 years. They recorded decreases in species richness, accompanied by increases in nitrophilic and shade-tolerant species over a recording interval of 41 years, although the legacy of coppicing was still evident in the composition of the tree, shrub and herb layers, suggesting that the influence of former management could persist for more than a century. In beech-dominated forest that had formerly been under a coppice-with-standards regime, Heinrichs and Wolfgang (2017) detected relatively more homogenisation over time in those understorey communities situated on dry, nutrient-poor and sun-exposed slopes, which tended to lose light-demanding, drought tolerant and oligotrophic species, compared with a more mesic forest community, which tended to gain in generalist species. A more recent resurvey interval, with a baseline set in the 1990s, detected similar increases in nitrophilous and mesotrophic



light-demanding species in formerly coppiced thermophilous oak forests in SW Poland (Reczyńska and Świerkosz 2016). However, in this case an increase in plant biodiversity and an inferred decrease in soil pH occurred over the 20-year interval, coinciding with major reductions in sulphur emission levels between 1960 and 2000. Other drivers of change were declining soil moisture and increased ungulate grazing.

### Provision for other forest habitats

Notwithstanding the apparent lack of deadwood for saproxylic niches in coppices, it has been pointed out that some are capable of maintaining microhabitats such as dendrothelms and mould cavities in old coppice stools, pollards or standard trees (Lassauce et al 2012; Vandekerckove et al. 2016, Larrieu et al. 2016). Microhabitats in ageing stands of trees are key components of biodiversity – for example tree cavities will benefit several mammals, birds, arthropods, but also fungi, bryophytes and lichens, including several obligate saproxylic beetles listed in Annex II of the Habitats Directive such as *Limoniscus violaceus*, *Osmoderma eremitica*, *Cerambyx longicorn* and *Lucanus cervus*. As stands age and amounts of deadwood increase, old coppices may even have the potential to allow saproxylic species to re-colonise. In the medium term at least, they may favour species with a preference for sun-exposed wood (Vandekerckhove et al. 2016).

The reductions in herb-layer diversity commonly observed in unmanaged forests do not apply to many other species groups. A meta-analysis of European forest literature found a marginally wider species diversity in unmanaged forests compared with managed ones, the differences increasing with time since abandonment (Paillet et al. 2010). Management tended to favour light-demanding understorey vascular plants, ruderals and competitive species, whereas bryophytes, lichens, fungi, saproxylic beetles and carabids,

more dependent on closed-canopy, benefited from abandonment. However, the way in which high forests are managed may considerably affect the biodiversity of species requiring longer rotations. A systematic Biodiversity Exploratory Project on beech high forests in Germany actually found a greater species diversity in managed forests compared with unmanaged ones, but the former contained higher average amounts of deadwood, possibly accounting for a higher diversity of specialist deadwood beetles, mosses and lichens (Müller et al. 2015).

Conversely, in three European biogeographical regions Zehetmair et al. (2015a,b) found no differences between commercially exploited Natura 2000 sites and matching non-Natura 2000 stands of 9130 *Asperulo-Fagetum* forest in terms of their densities of forest-dwelling bats or beetle diversity (including saproxylic species). This suggested Natura 2000 status alone would not make the stands more ecologically effective, especially for encouraging late succession species, and that additional conservation efforts were needed in these designated stands. This would require more deadwood, both standing and fallen, retention of ‘habitat trees’ with microhabitats such as cavities and bark pockets, and mature, living trees as potential recruits. Current forest certification schemes and local forest administration rules increasingly advocate such conservation measures, but non-selective and intensive harvesting practices in many forest types still tends to remove senescent trees and reduce deadwood (Larrieu et al. 2016). This is particularly the case in actively managed coppice woodland with few, if any, mature trees, except in ageing stands that are no longer exploited.

In another forest type, old thermophilic oak forests, canopy openness favoured saproxylic species (fungi, lichens, beetles, ants, bees and wasps), inferring that coppice and wood pasture could maintain their populations in more open conditions (Horák et al. 2014). Similarly,

in lowland oak forest in southern Moravia, canopy openness favoured an optimum diversity of spiders (Košulič et al. 2016), although these authors suggested that small-scale disturbances created by conservation thinning and selective harvesting, rather than extensive coppicing, could adequately maintain the various successional stages required. In old-growth, predominantly beech forest in the Czech Republic Horák et al. (2016) also found that saproxylic beetle richness was positively influenced by canopy openness, as well as by the quantity of deadwood, whereas saproxylic fungi species responded more to canopy closure, deadwood quantity and higher levels of humidity. The higher temperatures under more open canopies might also partially compensate for a lack of deadwood (Schulze et al. 2016). Deadwood and old-growth conditions equally benefit the diversity of bird and bat communities. Cavity-nesting birds, as well as gleaner bats, were positively associated with standing deadwood in a study comparing managed and unmanaged stands of both lowland and upland forests in France (Bouvet et al. 2016). More nesting and feeding opportunities were available when microhabitats such as cavities and cracks were abundant, but insectivore birds, which require more open forests with well-developed shrub layers, were negatively affected by high densities of living trees.

Clearly, a range of forest age-classes or patches at a landscape scale would help to optimise their species diversity. While British literature tends to emphasise the benefits of young growth associated with coppice for birds, both European and North American studies emphasise the merits of later stand development for this same taxonomic group, perhaps reflecting the fact that Britain has relatively fewer old-growth stands (Quine et al. 2007). Thus, some balance between the extent of open and closed forests should deliver the maximum biodiversity for all taxa.

## Strategies to increase biodiversity

What other forms of silviculture might mirror the biodiversity associated with coppice management? Clear-cutting routines, which create abundant open space after harvesting, have aspects in common with a coppice cycle, although in coppice the canopies generally recover faster through vegetative regeneration and are also harvested earlier. Contrasting with traditional coppice-with-standards, the more frequent harvests in forests undergoing selective cutting may actually disadvantage the ancient woodland flora by causing greater disturbance (Decocq et al. 2005). In another context, the type of timber-harvesting practice, whether clear-cutting, thinning or selective, had relatively little effect on understorey plant diversity in temperate North American forests (Duguid and Ashton 2013). However, in this case selective cutting did increase plant species diversity compared with unharvested controls, possibly because the frequency of interventions increased the opportunity for early successional ruderals to co-exist with late successional perennials, analogous to the situation in harvested traditional coppices in Europe.

High forests, if neglected or managed along continuous cover, selection, or close-to-nature forestry lines, are far less likely to sustain large populations of light-demanding, thermophilic species, unless disturbance is sufficiently frequent and on a scale large enough to trigger patches of young growth across the landscape. In a comparison of intensively managed shelterwoods in Germany with the more extensive felling practices in Romania, where a period of self-thinning was followed by clear-cutting, Schulze et al. (2014) suggested that shelterwoods were probably less effective in promoting a wider biodiversity. At a practical level, some forest owners might prefer the simplicity of a clear-cutting routine to more intricate, close-to-nature management designed to optimise



stand structure, species composition, amounts of deadwood and habitat trees for conservation (Borrass, 2014).

The few studies directly comparing managed and unmanaged forests have tended to agree that veteran trees and deadwood should be retained in order to support a full biodiversity of species, because the disintegration phase in forest development generally provides the highest biodiversity (Winter and Brambach 2011). If a few trees are allowed grow to large diameters, e.g. for more than 150 years, they will increasingly provide the cavities, dendrothelms, bark cracks and fungal sporophores that are missing in younger stands. For beech-fir forests Larrieu et al. (2012) recommended conserving 10-20% of the forest area as veteran trees, retaining at least some individuals of >70cm diameter; similarly, for beech forests, Gossner et al. (2013) suggested retaining 'habitat' trees of >50cm diameter.

Since coppice rotations are far too short to allow trees to enter the disintegration phase, longer rotations incorporating significant amounts of young growth could be achieved in irregular and strip shelterwoods, wood pastures and standards within the coppice. Standards could potentially provide some microhabitats and deadwood, but are traditionally felled at relatively young biological ages, typically at 100 years or less (Matthews 1989; Harmer and Howe 2003), and would need to be retained for longer if their full biodiversity potential were to be realised. Larrieu et al. (2012; 2016) considered that intervals of 50 years without harvesting in coppice-with-standards was insufficient to reach tree-bearing microhabitat densities approaching those of old-growth forests; double this period was more likely to achieve it. Large diameters of deadwood, favoured by many saproxylic beetles, can coexist within relatively open and sunny conditions in coppices and wood-pastures (Seibold et al. 2015; Sebek et al.

2015). Rather longer standard tree rotations of 125 years have been recommended by others for conservation reasons, covering 20-25% of the area (Hopkins and Kirby 2007). A greater proportion of older trees within coppice is provided by the 'single tree orientated silviculture' method advocated by Manetti et al. (2016), in which low densities of target trees within the coppice are selected (e.g. 100 ha<sup>-1</sup>) and thereafter favoured by frequent thinning of their immediate neighbours, until they become valuable timber trees. This system produces a varied horizontal and vertical canopy structure comprising isolated trees, thinned stools and unmanaged coppice, although the crop trees are destined to be harvested when biologically still young, at merchantable size. Another silvicultural technique is to manage groups of standards as mini-high forests, embedded within the coppice stand (Mairota et al. 2016).

Standing and lying deadwood accumulation is strongly linked to biodiversity; the larger pieces providing a stable and enduring environment for the larvae of large-bodied beetles (Gossner et al. 2013). In European forests, a deadwood threshold of the order of >20-50m<sup>3</sup> ha<sup>-1</sup> has been suggested as necessary to support a high diversity of saproxylic organisms (Müller and Büttler 2010; Lachat et al. 2013). However, a significant patch-scale threshold of >300 m<sup>3</sup> ha<sup>-1</sup> was found in old-growth, mixed-montane forests in the Czech Republic, more than twice the level recommended by Müller and Butler (2010) for this type of forest (Horák et al. 2014). In south-eastern Germany, both the quantity and the diversity of deadwood (in contrasting sunny and shady situations) were found to be important drivers of saproxylic beetle assemblages in a mixed montane broadleaved/coniferous forest (Seibold et al. 2016). An extensive review of biodiversity within European beech forests by Brunet et al. (2010) concluded that the general sensitivity of species groups to shelterwood

management roughly followed the order:

- herbaceous plants
- < soil macrofungi
- < ground dwelling arthropods
- < land snails
- < saproxylic fungi
- < hole nesting birds and saproxylic insects
- < epiphytic lichens and bryophytes
- < epixylic bryophytes,

a further argument for retaining a proportion of veteran trees in order to fully represent the saproxylic and epiphytic species. Shortening rotation lengths, as in the increased exploitation of wood energy in aged coppices, could negatively impact saproxylic biodiversity if 'habitat trees' are not retained (Lassauce et al. 2013).

To optimise conservation objectives, it is frequently suggested that older trees and old-growth features should be deliberately interspersed amongst conventional forest cycles - an ideal situation would be a mosaic of different forest structures and ages at a landscape or regional scale. Several authors cited conservation measures using variable retention harvests, in which patches of unharvested 'tree islands', or '*îlots de sénescence*', are connected by a network of 'deadwood corridors', set within a productive, multi-aged forest matrix (Vandekerckove et al. 2013; Mason and Zapponi 2015; Larrieu et al. 2016). High density patches of mature trees would theoretically provide a more humid microclimate for fungi, bryophytes and lichens than would the spatially separated trees in a conventional coppice-with-standards arrangement. The best places for retaining veteran trees are likely to be within forest patches possessing a long history of continuity (Brin et al. 2016). Deadwood could also be retained in situ as part of regular harvesting, where the particular tree species may also be important. Gossner et al. (2016) suggested that leaving some larger-sized logs of subordinate

trees such as *Carpinus betulus* behind on the forest floor could help to conserve saproxylic beetle diversity more effectively than would leaving larger amounts of dominant species, such as beech.

A study by Winter and Brambach (2011) showed that uniformly managed forests were less diverse in the number of different forest growth stages that they represented than their equivalent in matched forest reserves. A landscape mosaic consisting of different forest types and ages might be expected to provide habitats for far more species than one type more uniformly managed (Schulte et al. 2006). Interacting patchworks, networks, and gradients within the landscape will ultimately determine forest conservation and biodiversity (Forman 1995; Lindenmayer and Franklin 2002). If, on the other hand, a whole landscape were given over to the small-scale dynamics of close-to-nature silviculture, this would tend to reduce overall beta-diversity and homogeneity in forest structure (Decocq et al. 2005). Building in increased structural diversity, using a variety of systems - clear-felling, shelterwood cutting, group selection, single tree selection, etc. - would offer greater complexity from a silvicultural point of view (Schall and Ammer 2013).

## CONCLUSIONS

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The Natura 2000 network uses criteria of species rarity and endemism to represent Europe's threatened biodiversity. This is also true at international, national and regional levels, where priority species and some habitats are given special conservation and protection status. With the emphasis on the protection of rare and threatened species, this appears to be more of a bottom-up exercise than one based on the habitat type (Maiorano et al. 2015). The former is a fine filter, whereas the latter, though a coarse filter, could nevertheless be regarded as a surrogate for the presence of notable and rare species. However, the Natura 2000 system can be said to provide a positive 'umbrella' for many groups of non-Annex species, with some exceptions such as amphibians and reptiles (European Commission 2016; van der Slijs et al. 2016).

An intimate knowledge of habitat requirements is needed to manage and maintain healthy populations and to balance the claims of several competing species. However, the Natura 2000 exercise will always be incomplete: many taxonomic groups have yet to be assessed or updated, as can be seen from the continuous revision of the European Red Lists and priority species lists used by different countries. In particular, invertebrates (such as arachnids and molluscs), soil fauna, bats and small mammals have poor representation. Taking one example, only 17 saproxylic beetles are listed on HDII whereas 407 appear on the EU27 Red List, 57 (14%) of which are in the threatened categories. Many are still 'data deficient', with more waiting to be assessed, some of which will likely be found to be threatened (Niето and Alexander 2010) (Table 2).

Although the HDII list is in serious need of revision and regular updating (Hochkirch et al.

2013), this is likely to remain a long-term project. A recent EU Working Document on the two Natura 2000 Directives found that they were indeed 'fit for purpose' in achieving the broader framework of EU Biodiversity policy. While it could be argued that more improvements in species coverage and alignment with international agreements would be desirable, these could generate uncertainty, leading to delays in the full implementation of the Directives while increasing costs and decreasing legal certainty (Milieu et al. 2016).

Comparatively few Natura 2000 species are 'coppice' specialists, but these and more generalist species have an important role to play. Götmark (2013) suggested that, depending on forest size and objectives, four types of conservation management strategies should be combined:

- 1) minimal intervention, which could eventually apply to coppices that are no longer managed;
- 2) traditional management, based on historical research, such as coppicing and pollarding;
- 3) non-traditional management, for example to promote old-growth characteristics, though this is not applicable to most coppices, or a particular composition of tree species; and
- 4) management specifically to promote threatened, indicator and other species.

A silvicultural portfolio embracing the extremes of all successional stages, from coppicing of young trees through to old growth, best promises to enhance diversity at a landscape level. Forestry certification schemes currently set standards for tree retention and deadwood, but some also acknowledge the contribution to biodiversity of traditional forest management, such as coppicing and pollarding. A review of the impacts of forestry practices in Britain and

Ireland found that most improvements to forest biodiversity resulted from the temporary open space after harvesting, or through permanent open space, often associated with the road and ride network (Bellamy and Charman 2012). Given the potentially huge array of species comprising forest biodiversity, young growth alone cannot provide niches for all of them, whereas, in coppicing it can be used to promote

iconic species as well as cosmopolitan ones. Other species, including many of those listed in the Annexes of Natura 2000, depend on high forest structures and old growth by combining different forest development stages. Overall biodiversity will only increase if both the protected and 'typical' species of forest habitats are given equal scrutiny.

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# The Status of Coppice Management within Forested Natura 2000 Sites

Paola Mairota and Peter Buckley

Most forest habitats that are listed for their nature conservation importance in the Habitats Directive of the European Union and the Bern Convention have been modified for centuries by human intervention. It is well documented that many forests throughout Europe were traditionally coppiced (cf. Piussi & Redon 2001; Kirby & Watkins 2015), thus influencing the woodland ecology not only at the stand level, but at wider spatial (landscape) and temporal scales, creating specific communities that are often the focus of nature conservation initiatives. As such, coppice management falls within the scope of the Habitats Directive (Council Directive 92/43/EEC; European Commission 2003; Loidi & Fernandez-Gonzalez 2012). However, this form of silvicultural system has become obsolete in many of the EU28 countries, particularly those in the north and east, whereas in others it is still very relevant to the country's economy (Figure 1). Nowadays, the trend towards non-intervention in coppice stands, or their conversion to high forest, is the de facto approach within areas protected for conservation.

In order to examine prevailing attitudes towards coppicing within sites designated under the Natura 2000 framework as Sites of Community Importance or Special Areas of Conservation (SCIs or SACs), a study was carried out within the framework of the EuroCoppice COST Action FP1301 to examine the relevant Site Management Plans (SMPs) in six participating countries. The aim was to sample the extent to which different countries recognised coppicing activities, and what extent they considered alternative options that might better secure the conservation status of the habitat in question (The full study is available in the open source iForest article Mairota et al. 2016a). These six countries (Belgium, Czech Republic, Estonia,

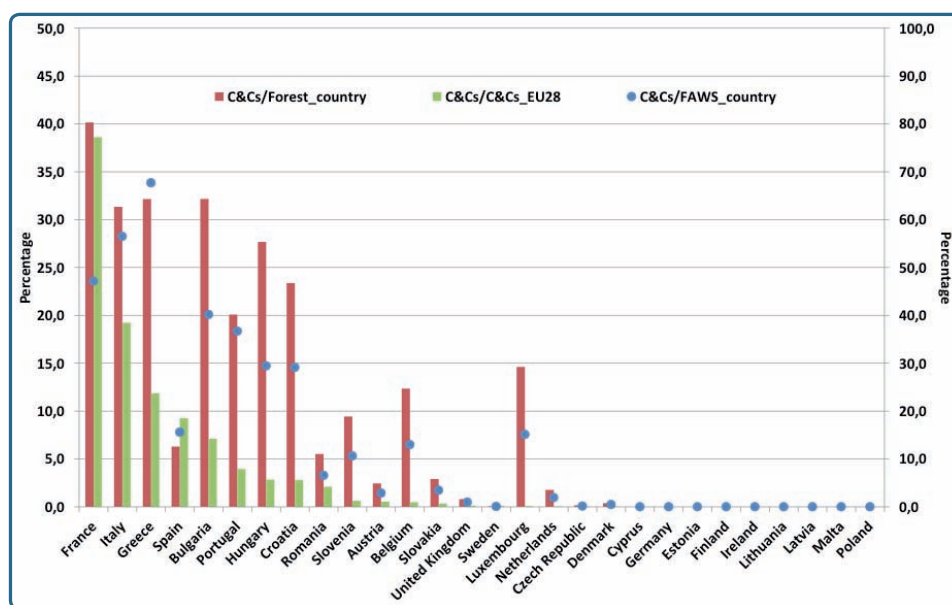


Figure 1. Left axis: Share of simple coppice (C) and coppice with standards (Cs) woodlands over the forest area of the country (C&Cs/Forest-country), and share of country simple coppice and coppice with standards in the EU (C&Cs/C&Cs EU); Right axis: Share of simple coppice/coppice with standards woodlands over the forest available for wood supply in the country (C&Cs/FAWS country) (Processed from UNECE-FAO 2010 data)

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Germany, Italy and the United Kingdom), represent a range of EU Biogeographical Regions, including both small and large regions, different administrative systems (centralized to devolved) and greatly differing amounts of forest cover. In addition, a sub-national level (at either the NUTS1 or NUTS2 regional scale) was chosen to review Natura 2000 Site Management Plans (SMPs) for three of these countries (Germany, Italy and the United Kingdom).

The share of Natura 2000 area in the sample countries is comparable to the EU28 terrestrial average, which is 14.6%. Of this, 73.9% is protected under the SCIs and SACs of the Habitats Directive, while the remainder falls under the Birds Directive. However, progress in formulating SMPs in compliance with the Habitats Directive's recommendations varies widely between the EU countries, as is mirrored in the six sample countries. In Italy there are a number of NUTS2 regions without enforced, or even envisaged SMPs, but here compliance to the Directive is ensured by collective conservation measures for those habitat types belonging to the same biogeographical zone (IT-D4 Friuli Venezia Giulia), or macro-environmental category (IT-C1 Piemonte and IT-F4 Puglia).

As a general tendency, it appears that a greater proportion of forest areas were designated as SCIs/SACs than many other habitats. The majority (68 %) of the 78 Annex I forest habitat types recognised by the Habitats Directive have the potential to be coppiced, i.e. the dominant species is capable of resprouting. This ability varies among the main forest habitat categories (i.e. 9000 'Forests of Boreal Europe', 9100 'Forests of Temperate Europe', 9200 'Mediterranean deciduous forests', 9300 'Mediterranean sclerophyllous forests', 9400 'Temperate mountainous coniferous forests', 9500 'Mediterranean and Macaronesian mountainous coniferous forests')

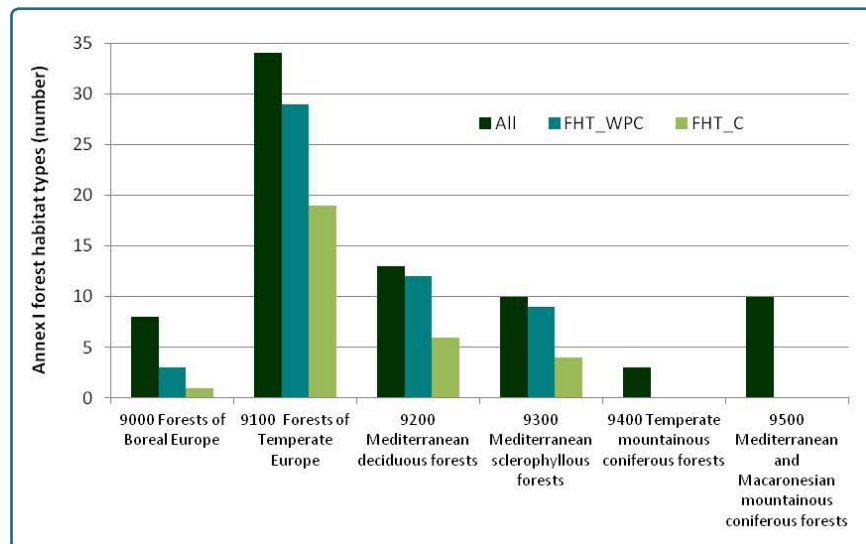


Figure 2. Distribution of forest habitat types in the main forest categories according to the Habitats Directive and incidence of both forest habitat types with potential for coppice and forest habitat types for which coppice is reported in the sample countries.

FHT\_WPC: forest habitat types with potential for coppice;  
FHT\_C: Forest habitat types which have been coppiced historically.

Europe', 9100 'Forests of Temperate Europe', 9200 'Mediterranean deciduous forests', 9300 'Mediterranean sclerophyllous forests') (Figure 2).

In the sample countries, 38% of the habitat types were considered to have been coppices in the past, with more and more evidence to this effect being reported (e.g. Madera et al. 2017). However, coppicing is no longer allowed in Estonia (where non-intervention is the current management strategy in protected areas), while it is only allowed for research purposes in the Czech Republic. Management prescriptions for coppices in SCIs/SACs tend to be rather strict in Italy (detailing specific aspects such as coupe size, rotation length, number of standards, standard age category, sporadic tree species release and canopy cover). Conversely, coppicing done to conserve particular target species is still practised in parts of the United Kingdom and Germany. Similar signs of a strict conservation interest have in fact also been noted in Italy (Negro et al. 2014), where a debate has recently begun between the Italian chapter of Pro Silva

(a Europe-wide association of silviculturists) and two national scientific societies dealing with vegetation science (SISV) and forest ecology (SISEF).

A closer look was taken at a number of SCI/SAC management plans (172 SMPs, 51% of those available) of five administrative regions in three sample countries (IT-E2 Umbria and ITF-4 Puglia (NUTS2), UK-J, South East England and UK-L Wales (NUTS1), and DE-B Rhineland-Palatinate (NUTS1). This revealed that coppice management was rarely encouraged and that conversion to high forest was often thought desirable. While the justification for this view was seldom provided, other than in generic/anecdotal terms, it was frequently argued that high forest could achieve higher financial returns, or that high forest, regenerating from seed, was the more 'natural' condition. That being said, no scientific study has thus far convincingly demonstrated that a high forest/wilderness state could achieve a more 'favourable conservation status' than that provided by coppice in most SCI/SAC forest habitats (European Commission 2013). On the other hand, a number of studies have provided increasing evidence of the importance of coppice in promoting biodiversity through its provision of open habitats (e.g. Garadnai et al. 2010, Mölder, 2010, Müllerová 2015).

SMPs generally addressed the notable species listed in Annex II where they occurred within the habitat, but were less concerned with other species that might benefit from coppice management (Buckley and Mills 2015). This is in spite of the Habitat Directive's aim to protect the habitat per se, with its array of characteristic (but not necessarily rare) species; in this case, species that are frequently associated with the mosaic of age classes created by coppice woods or coppice-with-standards.

Another common feature was that, notwithstanding differences in the amount of detail

required by the individual regional authorities dealing with SMPs, these plans were often rather descriptive or aspirational documents and provided no comprehensive management prescriptions or schedules. Their utility as the first level of a cascade process for integrated landscape/forest planning (sensu Baskent & Keles 2005) is therefore very limited. This is concerning, because decisions to abandon coppice at the stand level, or to select another (high forest) silvicultural solution, has a strong impact on forest landscape structure and functioning and could affect some key elements of biodiversity. A number of technical practices, such as the group selection of standards or single tree silviculture, when combined with non-intervention and conversion to high forest, have the potential to increase forest landscape micro- and macro-heterogeneity (Cf. Mairota et al. 2016b). This is a desirable objective in order to maintain high levels of beta-diversity in the long run (e.g. Hunter 1990, Buckley 1992, Fuller & Warren 1993, Mairota & Piussi 2006, Chiarucci et al. 2008, Garadnai et al. 2010, Kopecký et al. 2013 and Buckley & Mills 2015).

A case can be made for a more balanced approach to forest management (combining coppice, high forest and non-intervention), as this appears most likely to revive and maintain specific forest landscape habitats and site conditions, as well as revitalise local economies. Overcoming socio-economic factors and, especially, the cultural factors behind SMP strategies and attitudes is necessary. One factor that may become important is the increasing demand for wood for energy (Mantau et al. 2010, UN-ECE-FAO 2011). In response to the EU Renewable Energy Directive 2009/28/EC and in compliance with the Framework Program for the Forestry Sector, Horizon 2020 should improve the transparency of wood-fuel flows in agreement with the EU 995/2010 Timber Regulation.



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# Prevention of Soil Erosion and Rockfall by Coppice and High Forest – A Review

Peter Buckley, Christian Suchomel, Christine Moos and Marco Conedera

## INTRODUCTION

An important regulating ecosystem service of forests is their ability to protect against natural hazards such as soil erosion and rockfall, particularly on steep slopes. The ability to provide this service strongly depends on the forest structure and condition (e.g. Dorren et al. 2007, Imaizumi et al. 2008, Fuhr et al. 2015, Moos et al. 2017). With coppice, however, the question remains whether clear-cutting might actually exacerbate slope erosion, and if, in their abandoned or converted state, coppice stools could eventually become unstable and prone to collapse. In such a case, the risk of rockfall may be enhanced (Radtke et al. 2014).

At higher altitudes in the European mountain regions of Switzerland, Austria, Slovenia, Italy, Cyprus and Spain, coniferous forest species such as Norway spruce (*Picea abies*), silver fir (*Abies alba*) and European larch (*Larix decidua*) predominate in protection forests, while broad-leaved species with innate coppicing ability are more prevalent at lower altitudes. These include European beech (*Fagus sylvatica*), oak (*Quercus* spp.), chestnut (*Castanea sativa*), lime (*Tilia* spp.), maple (*Acer* spp.), ash (*Fraxinus* spp.), hazel (*Corylus avellana*), whitebeam and wild service tree (*Sorbus* spp.), hornbeam (*Carpinus betulus* L.), hop hornbeam (*Ostrya carpinifolia*), and black locust (*Robinia pseudoacacia*) (Jancke et al. 2013). Beech in particular may reach as far as the upper timberline (1600-2000m asl) in the Alps, as in southern Switzerland (Geschi 2014), or in Slovenia (Perret et al. 2015).

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Tree cover increases rainfall interception and transpires away soil moisture, thereby reducing runoff, so that a continuous or semi-continuous canopy may give good slope protection. Standing and lying trees can slow down, deviate, or stop falling rocks, and thus reduce their propagation and intensity (Perret et al. 2004, Dorren et al. 2007). By adopting appropriate forms of silviculture and eco-engineering, these forests can permanently reduce the risks to human life and property, although in extreme cases the trees may have to be supplemented or replaced by civil engineering and bioengineering solutions (Dorren et al. 2005, Dorren et al. 2007). From one point of view the high stem densities in coppice form strong physical barriers and extensive rooting networks (Gerber and Elsener 1998) and can re-grow rapidly after cutting, when parts of the root system may remain alive. On the other hand, abandoned coppices on slopes can develop a large aerial biomass relative to their root system (Conedera et al. 2010), which in time may cause stool instability and uprooting (Vogt et al. 2006). On more gentle farmland slopes in lowland regions, where the soil surface may be periodically exposed by arable cultivation, one alternative might be to grow short-rotation coppice stands of *Populus*, *Alnus* and *Robinia* to protect against soil erosion (Petzold et al. 2014).

The goal of this paper is to give an overview on the effect of coppice stands on risks induced by erosion, landslides and rockfall and to discuss management strategies aiming at high protection capacity of these forests.

## 1. The role of tree canopies

Trees intercept and transpire moisture, as well as increasing both water infiltration into the soil and the water storage capacity, thus delaying levels of soil saturation that could cause incipient slope stability (Forbes and Broadhead 2011). The level of this effect strongly depends on the type of vegetation (e.g. forest structure, species composition) and season (Anderson et al. 1976). While harvesting removes the coppice canopy, the probability of slope failure will depend upon the frequency of cutting, the amount of litter and brash left behind, and the presence of unharvested trees (Piusi and Puglisi 2012). Remaining tree roots tend to increase infiltration by increasing soil pore formation and forming networks that facilitate a faster drainage than if no channels were present (Vergani and Graf 2016). The recovering canopy of the transpiring crop may also reduce excessive soil moisture and, therefore, the risk of surface instability, although in cool, temperate regions where precipitation usually exceeds evapotranspiration, the advantages may be small. Nevertheless, soil loss resulting from forest harvesting can become an issue at slope gradients above 8-9° and it increases significantly above 20°, when major landslides and debris flows are likely to occur (Borrelli et al. 2016).

## 2. Root reinforcement

Shallow landslides occurring on slopes carry earth, mud, clay and other debris; they are generally less than 2m deep (Rickli and Graf 2009, Sidle and Bogaard 2016) and are often triggered by heavy rainfall or earthquakes. Tree rooting forms a fibrous reinforcement, increasing the soil shear strength: in general, the coarse roots (>10 mm diameter) act as anchors or soil nails, while fine to medium roots (0.01-10 mm

diameter) tend to reinforce and 'pin' together the soil profile (Stokes et al. 2009). We can distinguish basal root reinforcement along a potential slip surface, lateral root reinforcement at the margins of the landslides, and stiffening effects of soil under tension and compression (Mao et al. 2012, Schwarz et al. 2015, Cohen and Schwarz 2017). These effects are mainly influenced by root density, root tensile strength and depth of rooting. The glue-like exudates of root mycorrhizae provide additional soil strength by contributing to the formation of soil aggregates (Bronick and Lal 2005). In an investigation of a steep slope revegetated 25 years earlier by hydroseeding and supplementary planting of grey alder (*Alnus incana*) and purple osier willow (*Salix purpurea*), Burri et al. (2009) showed that soil aggregate stability approached that of a nearby mature ('climax') beech forest on a similar incline. In coppices, a window of susceptibility to erosion begins when roots start to decay after cutting, and persists until new woody vegetation and root growth is achieved.

Slopes also appear to influence root morphology, with the larger roots orientated uphill and assisting soil anchorage, as observed in downy oak (*Quercus pubescens*) and manna ash (*Fraxinus ornus*) by Chiatante et al. (2003). Di Iorio et al. (2005) found the same tendency in maiden (uncoppiced) trees of downy oak, growing on slopes ranging from 14 - 34°, where the first-order laterals tended to cluster asymmetrically, in an upslope direction, and to form resistant I-beam cross-sections. This adaptive root architecture emphasizes the resistance of these up-slope roots to pullout, counteracting the turning moment that tall, upright tree stems of abandoned coppice stools are constantly subject to. A study of managed and abandoned chestnut coppices in northern Italy, situated on slopes of 13 - 35°, showed denser but shallower

rooting in the 0 - 50 cm soil profile of a currently managed stand compared with overaged stands (Bassanelli et al. 2013). This may have been influenced by the renewal of the root systems after each coppicing event, although there was less soil depth than in the abandoned coppice sites. The study showed that root tensile strength was not affected by abandonment, but simulation modelling suggested that slopes of  $>35^\circ$  were intrinsically unstable and likely to lead to shallow landslides, particularly those with high levels of soil moisture saturation. These authors concluded that maintaining a regular coppice cycle was essential to prevent shallow landslides occurring on steep slopes. On the other hand, Dazio et al. (2018) suggested that aging chestnut coppice stands in southern Switzerland tended to provide progressively more root reinforcement, owing to an increasing proportion and absolute number of coarse roots.

The roots of different tree species appear to react differently to coppicing. In birch (*Betula* spp.) coppice, Bédéneau and Pagès (1984) found that medium to coarse ( $>5$  mm diameter) roots were the same age as the stool, suggesting that the old root system remained intact, whereas in chestnut the roots were freshly regenerated (Dazio et al. 2018). The latter also seems to hold true for beech (Amorini et al. 1990, Bagnara and Salbitano 1998) and maple (Lees 1981) but not for some *Eucalyptus* species, which tended to keep their original root systems after cutting (Riedacker, 1973, Wildy and Pate 2002). It seems likely that the drastic reduction of carbohydrate resources resulting from stem loss forces the plant to direct its energies into shoot production, with root development (especially that of coarse roots) lagging behind. This is exacerbated when short rotations are applied; in a hybrid poplar plantation, for example, coppicing caused the plants to use carbohydrates stored in the roots for the new stem growth, potentially inhibiting rooting (Lee 1978, Bédéneau and Auclair 1989).

The amount of rooting, and particularly the development of structural coarse roots, has particular implications for coppice. In maiden trees and in old coppice, there is some evidence that the ratio of coarse to fine roots increases over time, whereas younger coppice tends to be more dependent on fine rooting (Montagnoli et al. 2012, Di Iorio et al. 2013). Laboratory and field pullout tests (Giadrossich et al. 2013, Vergani et al. 2016) have been used to estimate the tensile force of root bundles, which also clearly demonstrate a power law relationship between root diameter and tensile force. Root reinforcement can be estimated using a number of different models, most recently by the Root Bundle Model (RBM) (Schwarz et al. 2013), which uses a Weibull survival function to account for mechanical variability and the relative contributions of different combinations of coarse and fine roots. Simulations show that coarse roots are disproportionately influential in effecting root reinforcement - the maximum tensile force of a single root of 50 mm diameter being the equivalent of more than 500, 1 mm diameter roots (Vergani et al. 2017).

Trees that root relatively deeply, such as European ash (*Fraxinus excelsior*), *Quercus* spp., aspen (*Populus tremula*) and alder (*Alnus glutinosa*) give better soil anchorage, especially when species with different root forms are mixed together (Rayner and Nicoll 2012). With an increasing ratio of coarse to fine roots developing within a tree crop over time, we might expect that root reinforcement, and consequently soil stability, would also increase as coppices are converted, or gradually develop into high forests. In over-mature coppice crops, coarse roots will also extend outwards from the stool, stabilising a greater surface area than would be the case of recently cut coppice, which is more dependent on its finer roots (Dazio et al. 2018). On the other hand, by virtue of their very high stem densities, many coppices may reinforce the soil surface with their rooting as effectively as



high forests. Breaking forces, taking into account root diameter, are also quite variable between species: for example, Vergani et al. (2012) found that beech roots were almost twice as resistant as larch (*Larix decidua*) and spruce. The order was beech (84N) > sycamore (65N) > hop-hornbeam (56N) > ash (47N) > larch (46N) > sweet chestnut (44N) > Norway spruce (40N).

When the shear zone lies below rooting depth, particularly on relatively impermeable clays liable to slope instability, the reinforcing effect of roots is expected to be negligible (van Beek et al. 2005). However, the hydrological regulation under a forest may have a positive influence on soil stability. When coppices on slopes are cut, a potential problem could arise if the rate of decay of the original root system is not compensated by the rapid regrowth of fine and coarse roots, or if the interval between harvesting and root regrowth is prolonged. New roots may not counterbalance the decay of the old root system in those species that tend to renew their roots after coppicing, lowering root reinforcement (Vergani et al. 2017). However, some coarse roots can take several years to decay and this may provide a sufficient interval of protection from the risk of shallow landslides. In felled beech stands in Northern Tuscany, Preti (2013) found that root tensile strength declined in a roughly linear fashion, at 11% per year for a total decay time of c. 9 years. This work also predicted that deforested slopes could be liable to shallow landslides within a decade of tree death, a period in which heavy rain- or snowfall events could easily occur. Silvicultural treatments could mitigate this risk, for example by extending the rotation period, as this might raise the level of root reinforcement and conserve soil resources (Rubio and Escudero 2003). Standard trees retained among the coppice could also provide pockets of permanent anchorage when the coppice is cut. Finally, uneven-aged or selective coppicing will maintain a permanent

canopy and therefore reinforce rooting. In many situations, however, conversion of coppice to high forest can be extremely expensive and demanding compared to the default option of abandonment, or even coppicing on a short rotation (Vergani et al. 2017).

Uprooting of abandoned chestnut coppice (>50 years) was also investigated by Vogt et al. (2006) in the southern Swiss Alps on slopes of 20 - >30°. The uprooted stems were taller and larger, with the probability of overturning increasing on steeper slopes, particularly in hollows and gullies. To avoid large trees becoming unstable due to their increasing gravitational load, the authors recommended re-coppicing or thinning within the coming 30 years. Being more vulnerable to windthrow, the surface scars created by uprooting might form starting points for erosion. However, Conedera et al. (2010) did not consider this to be a long term issue, because any gaps were likely to be filled by forest regeneration in due course. Although surcharge resulting from the weight of overaged stools has also been suggested as a factor likely to cause shallow landslides and a reason for continued coppicing, this has been largely discounted (Stokes et al. 2008, Vergani et al. 2017).

### 3. The barrier effect

On very steep slopes exceeding 30° in the source (or release) area of rockfall, the protective effect of trees can actually be negative (Dorren et al. 2007) if, by swaying to and fro in the wind, they act as levers to loosen and tear open the soil profile (Frehner et al. 2005). On the other hand, apart from tree roots binding the soil surface together, they may intrude into rock fissures and also promote the decomposition of rocks by organic acids (Frehner et al. 2007).

Both in the areas of transit (usually on >30° inclines) and deposition (<30° inclines), the protective effect of forests against falling rocks is basically due to the barrier effect of



Figure 1. Trees acting as barriers on a steep slope (Photo: Christian Suchomel)

standing and lying trees (Figure 1). Collisions with trees slow down or stop rocks, with sparse forests offering less protection than dense stands (Foetzki et al. 2004, Dorren et al. 2007). The main parameters influencing the degree of protection are: the forest density (number of stems  $\text{ha}^{-1}$ ), the diameter distribution of the trees, the tree species' specific energy dissipative capacity, the length of the forested part of the slope, the block volume and the block's kinetic energy (Dorren et al. 2005, Moos et al. 2017). It is often suggested that only rocks  $< 2 \text{ m}^3$  can be halted by single trees, but there are some examples from the Alps where rocks up to  $20 \text{ m}^3$  have been halted (Dorren et al. 2007, Ernst 2017). Several studies have shown that the basal area, i.e. the total surface covered by tree stems in a given area, is a good indicator of the protective effect of forests against rockfall (Berger and Dorren 2007, Dupire et al. 2016, Moos et al. 2017). Not only large diameter trees ( $> 36 \text{ cm}$ ), but also small trees can stop larger blocks ( $> 1 \text{ m}^3$ ), provided that part of the kinetic energy has already been dissipated. Thus, coppices stands may offer sufficient protection against larger blocks when combined with larger trees on the upper part of a slope (Dorren et al. 2005).

A study by Dupire et al. (2016) used the rockfall algorithm Rockyfor3D (Dorren 2012) to generate simulations of the rockfall hazard in 3886 forest plots in the French Alps, based on sloping terrain of  $20^\circ$  or more. Using

measures of the plot basal area and the mean tree diameter, they were able to calculate the minimum length of forest to needed to obtain a reduction of 99% in rockfall hazard. The study found that coppices dominated by deciduous *Fagus sylvatica* and *Quercus* spp. were the most effective stands in this respect, compared with pure coniferous stands of *Pinus* spp. and *Larix decidua*. Stands with high stem densities, high basal areas and greater biological and structural diversity were the most efficient, with the presence of a large number of trees being more important than lower densities of thicker trees.

Again using the RockyFor3D simulation model of rockfall (Dorren 2012), Fuhr et al. (2015) assessed the protection efficiency of pure and mixed uneven-aged stands dominated by beech, silver fir and Norway spruce along a maturity gradient. 'Young' stands with the highest stem densities gave the best protection against  $1\text{-}2 \text{ m}^3$  rocks, but even the neglected 'sub-adult' and 'mature' stands had tree densities of  $> 500 \text{ ha}^{-1}$ . The 'mature' stands, containing some individuals up to 220-260 years old and a significant number of very large trees ( $> 77.5 \text{ cm DBH}$ ) still offered high levels of protection, particularly against the larger sizes of rocks. Recently logged plots were considered much less effective, as the low-cut stumps could act as springboards, rather than obstacles, for the falling rocks. Moreover, mature stands contained high volumes of deadwood, including snags, which increased the roughness of the forest floor and, after modifying the simulation model to consider this, the stopping distance of large rocks was reduced by 28%. Radtke et al. (2014) recommended a slight extension of the coppice cycle in broadleaved mixed stands dominated by *Ostrya carpinifolia* and *Fraxinus ornus*, arguing that 25-year coppice forests gave better protection than young coppice, while beyond 40 - 50 years of age many stools tend to lose stability or break apart.

#### 4. Spatial arrangement of coppices

Coppice stems may be dense and clustered, with the multiple stems per stool in young stands tending to confer more protection than sparser, older stands with fewer stems per stool. A high stem density can reduce many risks (Ringebach 2013), but in unmanaged stands the declining stem density, through natural self-thinning, decreases the probability of rock collisions. This could be balanced to some extent by the increasing diameter and mechanical resilience of older trees, unless they are more prone to rot, as well as by the build-up of high volumes of deadwood in unmanaged stands. Older stems have thicker, more absorbent and energy-dissipating bark with which to resist rockfall and are more likely to arrest larger boulders with less stem damage. The higher stem densities associated with young stems may be effective against smaller ( $<0.25 \text{ m}^3$ ) rock sizes (Omura and Marumo 1988, Cattiau et al. 1995). Working in coppice stands of *Orno-Ostryetum* forest in northern Italy, Radtke et al. (2014) concluded that overaging did not adversely affect their protection function, at least for stands  $<60$  years old, although the gaps between stools were generally larger. They also found that in theory, a random distribution of stems had a higher protective effect than clustered distributions because the gaps between coppice stools decreased the likelihood of tree impacts. In a test case on Apennine coppice, the average distance between tree/boulder contacts (ADC), a measure of the energy absorbed by a forest structure, needed to be adjusted upwards from a theoretical single-stem arrangement so as to account for the higher rates of energy dissipation by coppiced trees (Ciabocco et al. 2009). They suggested that management based on the now-obsolete coppice selection system, where some stems are retained on individual stools at each cutting, or coppices with large reserves or standards, could give good rockfall protection.

Radtke et al. (2014) also found that the protective effect against large rocks was still one-third greater in the overaged coppice stands than the equivalent site without significant tree cover immediately following coppicing, provided that a few standard trees remained.

Ciabocco et al. (2009) conducted a series of impact tests on fresh beech stems (3-10 cm DBH) using a reinforced 84 kg concrete pendulum bob, swung to impact with clamped, single coppice stems. As expected, this demonstrated that mechanical resistance increased with stem diameter and lessened with the height of impact. However, it was surmised that highly flexible young coppice stems, generally of smaller diameter than those in mature forests, could decelerate boulders effectively and that the clumping of stems on stools could act as additional small retention fences. Although probably limited in their ability to protect against rocks  $>1 \text{ m}^3$ , simultaneous impacts against more than one stem on the same stool could effectively trap rocks between them (Figure 2). Nevertheless it was uncertain whether this multi-stemmed coppice structure produced a greater protective effect. Furthermore, the basal sweep of stems associated with slopes, resulting from growth stresses that form tension wood, could weaken them against impacts.

The history and spatial pattern of rockfall was investigated by Favillier et al. (2015) on sub-montane broadleaved forest on slopes of



Figure 2. Rock caught in a coppice stool (Photo: Christian Suchomel)



25 - 39° in the Vercors massif of the French Alps. An exhaustive analysis of wounds and bark scarring on the stems of individual trees and coppice stools revealed, as expected, a high incidence of impacts from rockfall near the top of the release zone, at frequencies of <20 years, as well as laterally in topographic depressions, which tended to funnel any rockfall. At 150 m downslope, the frequency of the damage interval fell to >40 years. Favillier et al. (2015) also demonstrated that the fast-growing downy oak, with its thicker bark, might be capable of absorbing more impact energy with less damage than an Italian maple (*Acer opalus*) of similar age. In a rockfall corridor in the French Alps, Stokes et al. (2005) showed that beech suffered less from stem breakage, wounding and uprooting than did the other species tested. Through winching experiments to break or uproot a tree, they found that beech was twice as resistant as silver fir and three times more than Norway spruce, which tended to uproot. In similar experiments, Dorren et al. (2005) ranked species in the following order of energy of dissipation: pedunculate oak (*Quercus robur*) >beech >sycamore >silver fir > larch/Norway spruce. There was a strong exponential relationship between stem DBH and the amount of energy dissipated from an impacting rock. Such differences could be attributed to the different xylem structure of the broadleaves, which can make them more resistant to splitting and deformation, and their greater number of roots that are anchored at a greater depth.

## 5. Silvicultural comparisons

In the southern Italian Apennines, Ferretti et al. (2014) developed a Synthetic Index of Protection (SIP) against soil erosion to compare the efficiency of different types of canopy of tree species, shrub and herbaceous layers, based on their respective interception values. Taking this (and slope angle) into account, they determined the most suitable silvicultural treatments

providing a continuous canopy cover. Beech selection coppices, in which some stems were always retained on the stools, provided good protection, as did the conversion to an uneven-aged beech high forest structure, although both options were costly. With Turkey/downy oak forest cover, the alternatives were:

- a) to continue coppicing,
- b) to convert to high forest via a shelterwood system, or
- c) to retain about 50 standards ha<sup>-1</sup> along with the coppice (Ferretti et al. 2014).

The authors suggested making very small felling coupes, predicated on getting good natural regeneration, either from seedlings or coppice resprouting. Becker et al. (2013) argued that on steep slopes, small diameter coppice poles of low volume were both uneconomic and technically difficult to harvest. They suggested that on dry, steep slopes of up to 16.7°, slow-growing stands of oak could be grown on longer rotations (50-80 years) in order to produce a more profitable mass per unit ratio. High quality trees could be retained as standards (at densities of 20-30 ha<sup>-1</sup>) to be harvested after two coppice rotations (100-160 years), while some poorer-quality trees could be left to die back naturally and become 'habitat trees'. Steeper slopes would require more expensive methods to be employed, such as cable harvesting.

The relatively small stem sizes associated with coppice might be considered most appropriate in the deposition zone of slopes, at a point where the slope incline eases and most travelling rocks have been slowed by impacts on trees further up in the transit zone. Although regrowth of coppices after cutting is rapid, the same practice of restricting felling coupes to 40m in the fall line is commonly advocated (Dorren et al. 2015). Pure coppice stands are only recommended in areas with short transit area slopes of less than 75 m length (Frehner

et al. 2005). Coupe sizes of 0.5 ha or more were less likely to give protection, since a weakened root reinforcement might allow loose rocks to reach their maximum velocity when travelling through the felling coupe. It was therefore recommended to keep clear cuts small and well-distributed throughout the whole protection area, with maximum widths of 20 m on

steep slopes regularly prescribed. In the case of preventing shallow landslides, as opposed to rockfall, Vennetier et al. (2014) also recommended limiting clear cuts to <0.5 ha, certainly <1 ha, or adopting a selection silviculture to protect the soil, pointing out that the increased cutting intensity, as in coppicing for fuelwood, might exacerbate the risk of erosion.

## CONCLUSIONS

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After coppicing, stands regrow quickly and soon achieve stem densities of a critical diameter, which are able to withstand soil erosion and minor rockfalls, as well as recover quickly from stem wounding and breakages. As the stems of traditional, in-rotation coppices rarely exceed 15 - 20 cm DBH, their protection function tends to be limited for rocks greater than 1 m<sup>3</sup> (Jancke et al. 2009). With abandonment, and increasing stem size, there is always the risk of stools being uprooted on unstable steep slopes during high winds or due to soil oversaturation, although the same would equally apply to mature high forest crops. Overaged coppice stands will eventually self-thin, increasing their stool spacing, but Fuhr et al. (2015) showed that old stands were able to retain moderate stem densities, as well as some trees large enough to intercept large blocks of c. 5 m<sup>3</sup>, while the high volumes of deadwood presented additional barriers.

By maintaining high stem densities, active coppicing does appear to provide an effective protection service against rockfall. As many former coppice forests develop into high forests, either through conversion or abandonment, they often retain the high stem densities that tend to reduce rockfall hazard (Dupire et al. 2016). Coppice harvests are also likely to be more economic in the deposition zone, below the steeper slopes, and may still be more cost-effective than converting the stand to a high forest structure. Coppicing also promotes

strong lateral rooting reinforcement against soil shear, with many broadleaves tending to have deep roots. The 'retention fences' resulting from multiple stems on the same stool may be more effective in trapping rocks than discrete, single stems of equivalent diameter, especially if rocks impact more than one stem simultaneously, although this may be counterbalanced by the clumped stem distributions forming large gaps between stools.

Beech and several other broadleaves also have roots with a stronger tensile strength than those of conifers, their frequent competitors in mountain situations; for a given DBH their stems are also more able to dissipate rockfall energy. It is not clear, however, to what extent root reinforcement retains its effectiveness immediately after cutting, before canopy cover is re-established. Conversion or abandonment of coppices on very steep slopes does not necessarily impair their protection services. Most evidence points to high forests as being inherently more stable structures with respect to soil erosion, due to their greater amount of coarse rooting compared with coppice. Hence the abandonment of coppicing on vulnerable slopes may not adversely affect the ability to regulate shallow landslides, and may actually increase soil stabilisation, especially in the case of those tree species that need to renew their root system immediately after harvesting. However, in the special case of river banks and gullies,

which are liable to debris flows during floods, managed coppice can avoid the overturning of large stems and their transport down swollen rivers (Rudolf-Miklau and Hübl 2010).

Since abandoned and over-mature coppices are even-aged, they will eventually break up synchronously. Under these circumstances, and particularly in the slow-growing conditions of mountain habitats, there may be insufficient naturally-seeded regeneration to take over the protection function of root reinforcement, especially if large gaps form. Thus, several

authorities advocate only clearing small coupes at a time, or uneven-aged/group selection systems, which rely on small canopy openings that fill with natural regeneration. All of this assumes the presence of relatively few domestic or wild browsing animals, as the fresh shoots on a coppice stool and natural seedling regeneration are both equally vulnerable. If coppicing operations are to be continued on slopes, protection can be enhanced by keeping gap sizes to a minimum, retaining standards and ensuring natural regeneration.

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# Historical Coppicing and its Legacy for Nature Conservation in the Czech Republic

Radim Hédľ

## INTRODUCTION

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I wrote this contribution having in mind a twofold perspective on coppicing: a historical one and an ecological one. The logic connecting these two otherwise distinct views is that the long-term presence, or even dominance in some regions of coppice management in the Czech Republic has influenced both past and current forest ecosystems. And vice versa, the historical range of coppicing has been largely determined by ecological factors. One cannot fully understand one aspect without the other. Despite being so widespread an activity, ranging from the prehistory up to the first half of the 20<sup>th</sup> century, coppicing has been deliberately and entirely abandoned in the past decades. The research devoted to historical and ecological

aspects of former coppice management partly aims to restore it for conservation and production purposes. In the Czech Republic, this process is just begun - nevertheless, this is stunning progress compared to the situation less than two decades ago, when coppicing was completely absent from nature conservation handbooks (e.g. Míchal and Petříček 1999) and not even mentioned in forestry. Up until now, several research projects directly or indirectly focusing on coppicing have been completed, or are still running, and a growing interest among the conservationists can be clearly observed. As foresters tend to be much more conservative as a whole, the future of coppicing restoration for wood production remains somewhat less promising.

## BRIEF HISTORICAL PERSPECTIVE ON COPPICING FROM THE MIDDLE AGES TO THE 20<sup>TH</sup> CENTURY

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Coppicing was a widespread management system in the Czech lands (Bohemia, Moravia and Silesia) at least since the Late Middle Ages. Its historical range strongly correlates with the extent of lowlands (150 to about 500 m a.s.l.), which occupy roughly one-third to half of the country area. This correlation is apparently because the lowlands are the most fertile, and hence the most densely populated areas of the Czech Republic since prehistory. Coppicing was a primary source of fuel energy, so the constant production of fuelwood was of high societal

concern, at least until it was replaced by fossil fuels at some time during the 19<sup>th</sup> century.

Forest has always been relatively scarce in the lowlands of the Czech Republic. Only sites least favourable for agriculture, such as slopes or stony soils, were left to forest management. This could explain why coppicing, an intense and effective fuelwood production system of the past, prevailed in the lowlands. Two noteworthy examples, illustrating which factors historically played a role in decision making with regard to the forest management type,

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were presented by Szabó and Hédl (2013). Coppicing was clearly preferred where the natural conditions allowed. Non-timber forest systems, including wood pasture, were probably applied only in the relatively less suitable situations. To fully understand the driving factors for particular types of management in the past would, however, require further research. This situation was typical for central and northern Bohemia, southern Moravia and adjacent parts of Silesia (Figure 1). The structure of forest vs. agricultural land use remained conservative for centuries in these regions, with crop fields predominating and forest areas being relatively small (Mackovčín et al. 2011).

In Moravia and Silesia, the proportion of coppicing systems within all types of forest management can be relatively precisely established for the 18<sup>th</sup> and 19<sup>th</sup> centuries. This information has been obtained through extensive research, using all available archival material for the region ([http://longwood.cz/?page\\_id=165](http://longwood.cz/?page_id=165)). The share of coppicing in Moravia and Czech Silesia ranged from zero to 100% in individual cadastres (civil parishes), showing a strongly uneven pattern. In the densely populated lowlands, the proportion of coppicing on all forest systems was typically more than 50%, often 80–100% (apart from quite significant areas without forest). In contrast, forested uplands had little or no coppicing management and in the transitional belts the coppicing proportion varied between zero to about 30–40%. Interestingly, the corresponding geographic pattern of coppicing in Moravia could be traced back to the Middle Ages (14<sup>th</sup> century), pointing to the long-term stability of coppicing systems for at least six centuries (Szabó et al. 2015).

In Bohemia, the western part of the Czech Republic, no reliable data for a similarly detailed mapping of the historical coppicing area exists. Land use and management data from the so-called Stable Cadastre, a land use survey of the 1820s–1840s, was rewritten long after the survey and in an unsystematic manner (P. Szabó, pers. comm.). Although this information is now freely available on the internet (<http://archivnimapy.cuzk.cz/uazk/pohledy/archiv.html>), further critical research is required in order to construct a detailed map of the historical coppicing for Bohemia. Nevertheless, approximations can be made: a map of the historical area of coppicing in the whole Czech Republic has recently been published by Maděra et al. (2017), which confirms that coppicing prevailed in the lowlands of both Bohemia and Moravia (Figure 1).

A map of coppice forests for 1947, presented in the above-cited paper, shows a very similar pattern, indicating a persistence of coppice at time when there was no active coppicing in the country any more. The leading researchers in forestry at that time emphasised the negative

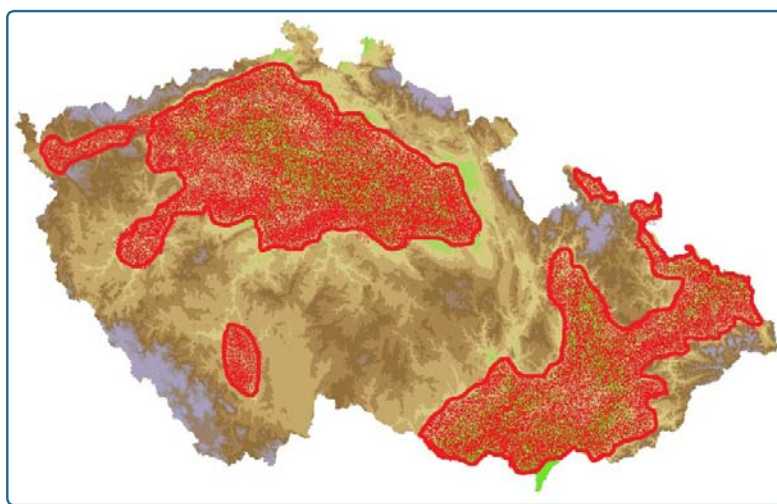


Figure 1. The approximate historical range of coppice forests in the Czech Republic, based on historical research by P. Szabó and his colleagues and published maps by Maděra et al. (2017). The area with significant historical coppicing (outlined in red) correlates with the lower altitudes (150 to 500 m a.s.l.). Current coppicing restoration work (not shown) is confined to no more than seven small-scale sites (situation in 2018).

aspects of coppicing, instead proposing methods for converting the remaining coppices into high forest (e.g. the special issue of *Lesnictví [Forestry]* devoted to coppicing, 1957/2). Probably the last deliberate coppicing activity was performed shortly before the WW II. Studies using tree-rings and archival resources (maps

and written documents) confirm the story of gradual coppicing abandonment over the past two centuries in Děvín, one of the most significant sites with historical coppicing in the Czech Republic (Altman et al. 2013, Müllerová et al. 2014). The last regular coppicing was applied there in 1935/1937.

## LEGACY OF HISTORICAL COPPICING AND EFFECTS OF COPPICING ABANDONMENT IN TODAY'S FORESTS

The legacy of historical coppice management in forests of the Czech Republic has yet to be published. Persistent effects of past coppicing management in the present forest ecosystems has so far received only little attention. To the author's knowledge, there has been no systematic study of the effects of past coppicing on abiotic (e.g. soil chemistry) or biotic properties of forest ecosystems. The latter includes the distribution of individual species and communities, as well as patterns in biodiversity. Why would this knowledge be worth the attention of researchers, conservationists and forest managers?

The approach is similar to other studies on the legacy of past land use. Several studies have shown a marked legacy of ancient land

use on soil properties and biotic communities (reviewed by Hermy and Verheyen 2007). These legacy effects could be somewhat more complex (and subtle) than coarse transitions from agricultural land to forest. However, they may be at least partly responsible for the current distribution of oak (Maděra et al. 2017) or the biodiversity of forest understory vegetation (Figure 2). Unpublished research by Hédl et al. shows that 19<sup>th</sup> century coppicing in Moravia significantly explains current species richness at the plant community level. Plots in cadastres with the 19<sup>th</sup> century coppicing show a higher number of vascular plant species than in plots where coppicing was absent. However, the contribution of coppicing, independent from other factors, is relatively low. At a still broader perspective, patterns of the historical coppicing (outlined above) largely coincide with the potential vegetation (after Neuhäuslová et al. 1998). Oak and oak-hornbeam forests are the types of potential vegetation prevalent in areas where coppicing once dominated. One must keep in mind that the natural conditions largely correlate with land use and partly with management types, so statements about the net effects of coppicing on the actual or potential distribution of species or ecological communities require careful differentiation.

On the other hand, changes in biodiversity and composition following coppicing abandon-



Figure 2. Coppicing in the Děvín Nature Reserve, Pálava, showed positive effects on flowering of herb species of forest understory, such as *Primula veris*.



ment are relatively well documented. Related research is based on two types of evidence: recent resurveys of vegetation plots, recorded at times shortly after the coppicing abandonment, and comparisons of sites with varying or contrasting parameters of environmental conditions, resembling the situation in active coppices. Both types of studies were performed in forests with historically prevalent coppicing in southern Moravia and central Bohemia. Several groups of organisms were targeted in these studies: vascular plants and their communities (Hédl et al. 2010, Kopecký et al. 2013, Müllerová et al. 2015), butterflies (Benes et al. 2006, Freese et al. 2006), epigeic invertebrates (Spitzer et al. 2008) and saproxylic beetles (Vodka et al. 2009, Vodka and Cizek 2013). Paradoxically, historical coppicing has sometimes been associated with extant, but declining populations of some species (Konvicka et al. 2008, Roleček et al. 2017), whereas research has shown that past coppicing may not be responsible for these changes (Szabó 2013), especially not for the long-term survival of the studied populations. Generally, coppicing and wood pasture, along with other non-forestry uses such as litter raking, could have comparable effects on biodiversity (e.g. Vild et al. 2015, Chudomelová et al. 2017, Douda et al. 2017).

Summarising the published studies from sites in the Czech Republic, the main conclusion would be that the coppicing abandonment has led to a decline in biodiversity. This concerns the species-rich deciduous lowland forests, where coppicing was the dominant forest management system up to the first half of the 20<sup>th</sup> century. The decline affected both individual species requiring forest habitats with frequent canopy opening and the ecological communities where species richness decreased and homogenization of species assemblages was documented. Remaining knowledge gaps concern the effects of coppicing abandonment on other groups



Figure 3. Coppicing restoration in the Na Voskopě Nature Reserve, Bohemian Karst. Clearings have been made to monitor the ability of coppiced individuals to resprout and the effects on biodiversity.

of organisms, namely those requiring shadier conditions and biomass accumulation.

Largely motivated by the alarming results of the above-cited studies, some coppicing has been restored in the past decade in order to promote vanishing biodiversity. At present, seven sites (some of them with several sub-sites) have so far been restored to traditional coppicing (Figure 3). The total extent of these sites hardly exceeds a few hectares, and most of them are found in protected areas, including natural reserves and national parks. Nonetheless, plans aim to restore at least a hundred hectares, pooling all sites. Results from freshly restored coppices showed positive effects on species and/or functional diversity of various taxonomic groups (Vild et al. 2013, Sebek et al. 2015, Šipoš et al. 2017, Hédl et al. 2017); the last case cited is of a newly established traditional coppice on former agricultural land. An important feature of coppicing restoration studies is that they capture the effects of one-time canopy opening rather than the long-lasting effects of coppice management. Several coppicing cycles would have to be run to assess the actual effects on ecological communities under the current environmental conditions.



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