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Carbon accrual rates, vegetation and nutrient dynamics in a regularly burned coppice woodland in Germany

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Abstract

Historically, large areas of forest in Europe were managed as coppice woodland to produce wood-based fuel for the smelting industry. We hypothesized that this practice produced a legacy effect on current forest ecosystem properties. Specifically, we hypothesized that the historical form of coppicing may have produced a legacy of elevated stocks of soil organic carbon (SOC), nutrients and black carbon (BC) in soil as fire was routinely used in coppiced woodland to clear land. We further hypothesized that these changes in soil properties would result in increased biodiversity. To test these hypotheses, we sampled the surface soil (0–5, 5–10 and 10–20 cm) from a chronosequence of forest sites found in the Siegerland (Germany) that had been coppiced and burned 1, 2, 3.5, 6, 8, 11 and 17 years before present. Mature beech and spruce forests (i.e., >60 years) were also sampled as reference sites: to provide a hint of what might occur in the absence of human intervention. We measured stocks of SOC, BC, NO_3-N, P, K, Mg, as well as cation exchange and water-holding capacity, and we mapped plant composition to calculate species richness and evenness. The results showed that coppicing in combination with burning soil and litter improved soil nutrient availability, enhanced biodiversity and increased SOC stocks. The SOC stocks and biodiversity were increased by a factor of three relative to those in the mature beech and spruce forests. This study shows that traditional coppicing practice may facilitate net C accrual rates of 20 t ha^{-1} yr^{-1} and maintain high biodiversity, indicating that aspects of traditional practice could be applied in current forest management to foster biodiversity and to mitigate climate change.

Keywords: biodiversity, black carbon, charcoal, soil nutrient stock, soil organic carbon stock, temperate coppice forest

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Introduction

Forest landscapes in Europe have undergone major changes in the last centuries. Beside extensive clearance and deforestation, coppice management was widely practiced, mainly in afforested areas (Ellenberg, 1988). The aim was to produce woody biomass harvested at intervals of 15–35 years (Ellenberg, 1988; Mcgrath *et al.*, 2015). Europe wide, up to 60% of total forest area has been under coppice management (i.e., coppice and coppice with standards; Stajic *et al.*, 2009; Mcgrath *et al.*, 2015). In Central and northwestern Europe, the practice of coppicing was abandoned between the mid-19th and the mid-20th centuries due to a declining market for coppice products (Mcgrath *et al.*, 2015). The percentage of the forest estate dedicated to fuel production via coppice declined most drastically after 1870 due to increased availability of coal (Banken *et al.*, 2007; Mcgrath *et al.*, 2015; Müllerová *et al.*, 2015). However, 15% of European forests are still coppiced, and the technique is being reintroduced mainly to produce woody biomass for the growing bioenergy sector in Europe (Bruckman *et al.*, 2011; Nerlich *et al.*, 2012; Mcgrath *et al.*, 2015). Also in Germany, coppicing was a common forest management technique between the 17th and the 19th centuries (Müller-Wille, 1980; Groß & Konold,

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2010). In some regions, for instance, the Siegerland, more than 50% of the total broadleaf forest was coppiced (Müller-Wille, 1980; Becker, 2002; Banken *et al*., 2007). Yet, little is known about the ecological legacy effects of past coppice management on current soil and vegetation properties. Understanding any potential legacy effects of traditional coppice management, might, for example, provide clues as to how we can optimize current forest management to maximize ecosystem service provision from intensively managed forest and woodlands.

Traditionally coppiced forests were used to cultivate woody perennials and agricultural crops and/or fodder plants: a type of agroforestry (Nerlich *et al*., 2012). The main aim of the coppice system developed in the Middle Ages was to produce fuel wood and charcoal for the local smelting industry (Nerlich *et al*., 2012; Mcgrath *et al*., 2015). Coppice regrowth was harvested regularly, every ~20 years (Nerlich *et al*., 2012). After the harvesting of woody biomass from these coppiced woodlands, the soils were generally hoed and burnt to clear litter and woody debris, which local farmers did to cultivate crops for at least one season on these recently cleared and burnt woodlands (i.e., 'Hauberg-wirtschaft'; Banken *et al*., 2007; Nerlich *et al*., 2012). The soils were prepared for cultivation of buckwheat (*Fagopyrum esculentum* Moench) or rye (*Secale cereale* L.) by hoeing and burning. The burning eliminated weeds as well as leaf litter and supplied ash that contained nutrients to soil, which facilitated at least one crop rotation (Nerlich *et al*., 2012). Immediately after this, coppicing and burning secondary succession began (Van Der Werf, 1991; Banken *et al*., 2007). Little is yet known about the relationships between plant successional changes of the coppiced and burned woodlands (Banken *et al*., 2007) to carbon accrual rates and the changes to soil nutrient status (Pregitzer & Euskirchen, 2004; Seedre & Chen, 2010) and other changes in soil properties. It is likely that past coppice management has left a legacy on current soil and vegetation properties (Banken *et al*., 2007). Likely, burning also introduced black carbon (BC) into the soil, which is resistant to microbial degradation (Preston & Schmidt, 2006).

In general, fire affects soil biological, chemical and physical properties (Certini, 2005; Forbes *et al*., 2006; Preston & Schmidt, 2006). The burning events cause an immediate loss and chemical transformation (i.e., polymerization and polycondensation) of organic matter (Johnson & Curtis, 2001; Wolf *et al*., 2013). Therefore, soils of fire-prone ecosystems are usually enriched with ash and BC (e.g., charcoal, pyrogenic C; Neary *et al*., 1999; Forbes *et al*., 2006), which affects, for instance, the cycling of nutrients (Certini, 2005), as well as postburn plant successions (Zackrisson *et al*., 1996; Deluca *et al*., 2006; Gundale & Deluca, 2007). Whether this enrichment of soil with BC favored the subsequent accumulation of soil organic carbon (SOC) is uncertain, but it is possible because BC can induce a priming effect that leads to higher or lower levels of SOC accumulation (Hamer *et al*., 2004; Wardle *et al*., 2008; Borchard *et al*., 2014; Glisczynski *et al*., 2016).

We hypothesized that coppice management combined with the use of fire to control understory regrowth would produce a legacy effect in European Forests (Mcgrath *et al*., 2015). Specifically, we hypothesized that this largely abandoned management practice would leave a legacy of elevated stocks of total SOC, BC and nutrients, as well as higher plant biodiversity. To test these hypotheses, we measured nonpyrolyzed SOC, BC, and soil nutrient stocks and also plant diversity and evenness across an age chronosequence of coppiced woodland in the open-air museum 'Historischer Hauberg Fellinghausen' in the Siegerland, Germany. In this open-air museum, a traditional coppice management unit is still in use and maintained (Becker, 2002; Banken *et al*., 2007; Müllerová *et al*., 2015). In this coppice management system, a small number of trees (i.e., <10% coverage ha^{-1}) are used as a seed source (i.e., standards) and cut less frequently, and weed regrowth is controlled with fire after harvest of the coppiced biomass to prepare soil for crop cultivation. This type of coppicing creates a mosaic pattern of plots across the landscape with vegetation at different stages of plant succession (Banken *et al*., 2007; Müllerová *et al*., 2015).

Materials and methods

Study areas

In this study, we investigated chemical soil properties and plant communities across a time series (i.e., chronosequence) of coppice age. The same dependent variables were measured in adjacent forests that are not actively managed as a type of experimental control. The sites comprised postburn coppice systems in seven age classes (i.e., seven sites that differed in time since coppicing and burning, i.e., 1, 2, 3.5, 6, 8, 11 and 17 years postcoppice). Additionally, we sampled spruce forests that established on abandoned coppice areas and also mature spruce and beech forests (stand ages >60 years) to compare coppicing with local forest ecosystems and to assess long-term effects on soil properties (Table 1). At each sampling location, botanical surveys were undertaken to re-assess and update the database of Banken *et al.* (2007). Distance between studied coppice plots at the open-air museum 'Historischer Hauberg Fellinghausen' was <500 m, and their distance to spruce forests was <6 km and to beech forests <10 km. The coppice and forest sites were located in the mountainous Siegerland (Germany) near the town of Kreuztal (Table 1).

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Table 1 Sampling sites, their vegetation formation, stand age, stand origin and stand management

Forest type	Site	Location (UTM*)	Vegetation†	Stand age (years)	Stand origin	Stand management
Coppice forest‡	C1	U32 427515, 5646316	Epilobio angustifolii-Digitalietum purpureae	1	Coppice	Natural regeneration
	C2	U32 427645, 5646177	Epilobio angustifolii-Digitalietum purpureae	2	Coppice	Natural regeneration
	C3	U32 427674, 5646137	Frangulo-Rubetum plicati	3.5	Coppice	Natural regeneration
	C4	U32 427777, 5646061	Frangulo-Rubetum plicati	6	Coppice	Natural regeneration
	C5	U32 427672, 5646441	Calluno-Sarothamnetum	8	Coppice	Natural regeneration
	C6	U32 427777, 5646254	Betulo pendulae-Quercetum roboris	11	Coppice	Natural regeneration
	C7	U32 427555, 5646834	Holco mollis-Quercetum	17	Coppice	Natural regeneration
Coppice to spruce forest	CS1	U32 430418, 5647681	Dicranaceae-Spruce forest	100	Coppice	Thinned/single stem
	CS2	U32 430857, 5648848	Dicranaceae-Spruce forest	121	Coppice	Thinned/single stem
	CS3	U32 431223, 5648623	Dicranaceae-Spruce forest	166	Coppice	Thinned/single stem
Spruce forest	S1	U32 431019, 5647147	Dicranaceae-Spruce forest	>60	Spruce forest	Thinned
	S2	U32 430962, 5647155	Dicranaceae-Spruce forest	>60	Spruce forest	Thinned
	S3	U32 431011, 5647111	Dicranaceae-Spruce forest	>60	Spruce forest	Thinned
Beech forest	B1	U32 429612, 5651532	Luzulo-Fagetum	>60	Beech forest	Thinned/single stem
	B2	U32 430576, 5652466	Luzulo-Fagetum	>60	Beech forest	Thinned/single stem
	B3	U32 430366, 5653275	Luzulo-Fagetum	>60	Beech forest	Thinned/single stem
	B4	U32 427827, 5653015	Luzulo-Fagetum	>60	Beech forest	Thinned/single stem
	B5	U37 430117, 5652338	Luzulo-Fagetum	>60	Beech forest	Thinned/single stem

*Geographic coordinate system: Universal Transverse Mercator (used receiver: GPS 72; Garmin Deutschland GmbH, Garching, Germany).

†Phytosociological classification (Schubert *et al.*, 2001).

‡Stands at the open-air museum 'Historischer Hauberg Fellinghausen'.

In the Siegerland, Cambisols have formed on lime-free sediments, which vary in their clay content, in hydrology (e.g., gleyic and stagnic properties) and in the abundance of rock fragments (e.g., skeletic). For this study, we minimized soil-related variability by selecting plots that occur only on Skeletic Dystric Cambisols and by restricting sampling to sites on similar landforms (i.e., middle slopes). In addition, we tried to minimize variations in soil texture to the main texture class: loam, although we had to accept variations within this texture class (i.e., coppice: silt loam; abandoned coppice: loam; spruce forest: silt loam; beech forest: sandy clay loam). The climate is temperate with mean annual temperature of 7.7 °C and mean annual precipitation of 1075 mm per annum (Banken *et al.*, 2007).

Sampling and analyses

The open-air museum 'Historischer Hauberg Fellinghausen' is managed as a complete unit of 24 ha (Banken *et al.*, 2007). Management of the open-air museum has produced a mosaic across the landscape of woodland plots in various stages of recovery postcoppice harvest. Hence, measurements from plots across the complete coppice management unit of the 'Historischer Hauberg Fellinghausen' were collected in a space-for-time substitution to constrain the effects of time (i.e., stand ages; Table 1) since last harvest/burn on soil properties and plant biodiversity (see also below, Botanical assessments and estimation of biomass C). At each of the 17 sampling sites (0.5 ha), five soil pits were dug. In each pit, 250 grams of soil sample was taken from 0–5 cm, 5–10 cm and at 10–20 cm depth (Table 1). For each sample site, the five soil samples from each depth increment (0–5, 5–10 and 10–20) were combined to produce a composite sample for each site*depth increment prior to chemical analysis to control for within-plot variability. To measure soil bulk density, additional soil samples were collected: At all 17 sampling sites and for all three soil depths, we collected five replicate soil samples using 100 cm^3 cores.

The composite soil samples used for chemical analysis were mixed, dried at 40 °C to a constant weight and subsequently sieved to <2 mm. Total contents of SOC and nitrogen (N) were analyzed by combustion using a C/N analyzer (NA 2000, Fisons Instruments, Ipswich, UK). The soils studied here showed pH values <6.5; thus, a correction of total C for calcareous C content was not required. Contents of aromatic C (i.e., BC) were measured using the benzene polycarboxyl acid (BPCA) method as described by Brodowski *et al.* (2005) and Kappenberg *et al.* (2016). As suggested by Brodowski *et al.* (2005), total BC contents were obtained by multiplying the BPCA yield with the conversion factor of 2.27. Here, we present SOC data that were corrected for measured BC content (SOC = total SOC − BC). Concentrations of plant-available

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potassium (K) and phosphorous (P) in soil were measured using the CAL-method (0.05 mol L^{-1} Ca-lactate, 0.05 mol L^{-1} Ca-acetate, acetic acid, pH 4.1, 1 : 20 w/v) (VDLUFA, 1991) and those of magnesium (Mg) and nitrate-nitrogen (NO_3-N) after extraction with 0.01 M $CaCl_2$. Soil pH values were measured after equilibration in 0.01 M $CaCl_2$ for 2 h. The potential cation exchange capacity (CEC) was determined following the Mehlich 100 method at pH 7.0 (Mehlich, 1942). To assess how much water can be stored in plant-available form, we assessed the water-holding capacity (WHC) as soil weight after 30 min of immersion in water minus the respective weight after 2 h of drainage (modified after Öhlinger, 1996). The 100 cm^3 samples were dried at 105 °C to constant weight and then used to determine dry soil bulk density (ρ_b), which was corrected for coarse fragments >2 mm (Schrumpf *et al*., 2011).

Carbon and nutrient stocks were determined for each soil depth layer (0–5, 5–10 and 10–20 cm) and summarized for each sampling site using Eqn (1):

$$\text{stock}_X = \sum_{j=1}^{n} (X_b \times \rho_b) \times Z \quad (1)$$

where stock_X represents stocks (SOC, BC, N [t ha^{-1}]; nutrients [kg ha^{-1}]; water-holding capacity [ML ha^{-1}]; CEC [$kmol_c$ ha^{-1}]) and acidity [$kmol_{H+}$ ha^1]), j represents soil depth layers (1, 2, 3), X_b are concentrations (SOC, N [g kg^{-1}]; nutrients [mg kg^{-1}]; water-holding capacity [L kg^{-1}]; CEC [$cmol_c$ kg^{-1}]), ρ_b is the bulk density corrected for coarse fragments (t m^{-3}), Z is the thickness of each soil layer (m), and n is the number of soil depth layers. Ideally, for comparing different management strategies and their impact on soil nutrient stocks, mass of the relevant surface layers needs to be accounted (e.g., hoeing lowers bulk density and increases the thickness of Ah horizon). In this study, we compare hoed and nonhoed soils; thus, we used Eqn (2) to standardize soil mass across sites prior to mass adjusted stock calculations (Ellert & Bettany, 1995):

$$Z_{\text{add}} = \frac{(M_{\text{equiv}} - M_{\text{surf}}) \times 0.0001\ \text{ha m}^{-2}}{\rho_{b\ \text{subsurf}}} \quad (2)$$

where Z_{add} represents additional thickness of subsurface layer to attain equivalent soil mass (m), M_{equiv} is mass of heaviest horizon (t ha^{-1}), M_{surf} represents soil mass in surface layers (t ha^{-1}), and $\rho_{b\ \text{subsurf}}$ is the bulk density of subsurface layer (t m^{-3}). Identical soil mass was calculated by adding soil mass in the surface layer (0–10) plus the soil mass stored in an additional (and variable) layer (Z_{add}) of the respective subsurface (10–20 cm) layer to attain an equivalent soil mass for all sampled sites (Ellert & Bettany, 1995).

Botanical assessments and estimation of biomass C

The plant communities, their richness and evenness of the sampling sites were surveyed in triplicate. Three 100 m^2 subplots in each 0.5 hectare sampling site were selected at the same elevation, and the distance between subplots was always <20 m. To assess variability in species composition between sites representing different stand ages and stand management strategies (i.e., time since harvest and burning of understory regrowth; Table 1), detailed plant surveys were performed following a stratified systematic sampling design (i.e., square triangle grid). This means that we restricted plant analyses to same elevation and degree of soil development within the study area.

The vegetation types were classified after Schubert *et al*. (2001) using JUICE 7.0 software (Tichy & Chytry, 2006) after surveying three representative areas of 100 m^2 of each coppice age class and forest type (Dierschke, 1994; Tremp, 2005) for their bryophytes (Frahm & Frey, 2004) and vascular plants (Aichele & Schwegler, 1998; Schulz, 2004; Rothmaler, 2005, 2007). To determine coverage and abundance of plant species, the Londo scale was used (Londo, 1976) that allowed multivariate statistical analyses (Leyer & Wesche, 2007). To determine species diversity, we calculated both species richness (i.e., number of species per 100 m^2) and species evenness (i.e., relative abundance of different species). To compute species evenness (E), we used Eqn (3):

$$E = D/R \quad (3)$$

where D is the Simpson's reciprocal index

$$D = 1/\sum [a_i/A]^2 \quad (4)$$

that accounts for area covered by each species (a_i) related to total area covered by all species (A) and, R is the richness.

Litter was collected from a fixed area within sampling site using a frame (400 cm^2) and dried at 60 °C to a constant weight to determine dry weight biomass. Dry biomass of the trees and coppiced branches (>150 cm) were calculated in accordance with Zianis *et al*. (2005) and for grasses and herbs (i.e., ground vegetation) following Bolte (1999). Zianis *et al*. (2005) and Bolte (1999) created empirical equations for aboveground biomass of coppiced trees based on diameter at breast height and coppice branch height. For the calculation of C stocks of living and dead biomass, we assumed a 50% C content for dry biomass (Joosten *et al*., 2004).

Data processing

We defined five different classes of land use, that is, the noncoppiced beech, spruce forests, abandoned coppice, sites with recently coppiced (1–6 years ago) and sites coppiced a long time ago (8–17 years ago). This allowed us to test statistical differences in soil and vegetation properties among these five classes, with a nonparametric analyses of variance (i.e., Kruskal–Wallis test) using STATISTICA software version 13 (StatSoft GmbH, Hamburg, Germany). In addition, we analyzed differences in soil chemistry and plant diversity using simple linear and nonlinear regression to investigate the changes along the coppice's chronosequence. To assess dependencies between soil properties, we used pairwise Spearman correlation. Nonparametric methods were chosen instead of analysis of variance because data distribution was non-normal.

Results

Soil properties

Coppicing and controlling plant regrowth with fire affected soil properties markedly (Fig. 1). During the first

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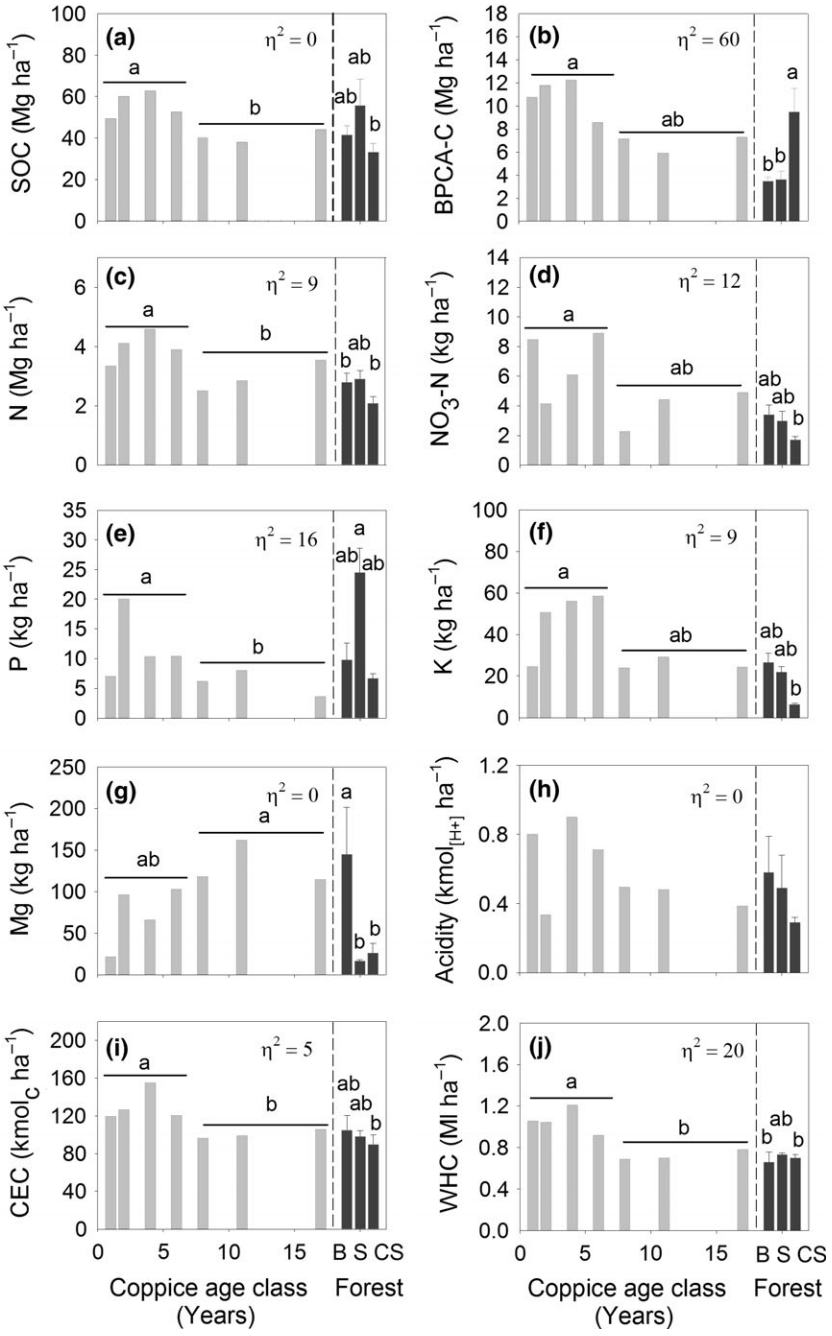


Fig. 1 Soil properties (i.e., soil organic carbon [SOC], black carbon [BC], total nitrogen [N], nitrate-nitrogen [NO3-N], phosphate-phosphorus [P], potassium [K], magnesium [Mg], acidity, cation exchange capacity [CEC] and water-holding capacity [WHC]) and their change within 17 years after coppice harvest and burning of understorey regrowth (i.e., gray bars; with stand ages of 1, 2, 3.5, 6, 8, 11 and 17 years). The soil properties were compared to soil properties of beech forests (B), spruce forest (S) and spruce forest cultivated on formerly coppiced land (CS). Effect size of burning (coppice vs. noncoppiced forests) on soil properties is indicated by η^2-values that indicate how important (in percentage terms) burning is in explaining variations in the data sets. Small letters indicate significant differences between forest types (Kruskal–Wallis test, $P < 0.05$). SOC data are total SOC after correction for BC content.

decade after harvest and soil burning, soil properties deviated strongly, with a gain in SOC after about 6 years, but thereafter approaching the SOC levels found in unaffected beech forests (Fig. 1a). The latter represents a type of baseline, which might be expected in an unmanaged woodland in this region (Banken *et al.*, 2007).

Black carbon stocks in the coppice forestry systems followed a temporal trend similar to that of the other

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nutrients (e.g., SOC and total N), being elevated in the first years after a burning but declining thereafter (Fig. 1b). A new equilibrium of BC content was reached approximately 6 years after last fire treatment and remained in spruce forests established on land coppiced more than 100 years ago (Fig. 1b). There was even a tendency of declining BC contents 8 years after burning, which rather is associated to varying charring efficiency during controlled burning or to site heterogeneity. However, the average BC stocks at coppiced sites (9.12 ± 0.46 Mg ha^{-1}) were significantly larger than those in the adjacent native beech (3.48 ± 0.37 Mg ha^{-1}) and spruce forests (3.65 ± 0.70 Mg ha^{-1}), which burned less frequently or even never (Fig. 1b). Thus, burning altered the quality of SOM substantially (Fig 1b; BPCA-C panel; $\eta^2 = 60$), as reflected by lower C/BC ratios (Kruskal–Wallis test, $P < 0.05$) for both recently coppiced forests (1–6 years ago: 5.26 ± 0.43; 8–17 years ago: 5.26 ± 0.43) and abandoned coppice (3.55 ± 0.85) relative to beech (11.97 ± 0.66) and spruce forests (15.39 ± 0.85).

Soil properties across the coppice age class sites can be grouped into three main patterns. The majority of soil properties reached a maximum 6 years after disturbance. First, there were moderate increases (<40% of initial values) of SOC, BC, N, CEC and WHC (Fig. 1a–c, i, j). These changes were correlated: The changes in the values for BC, N, CEC and WHC correlated significantly ($P < 0.05$) with those of SOC with R^2 values >0.8 ($R^2_{Spearman}$ BC = 0.94, $R^2_{Spearman}$ N = 0.91; $R^2_{Spearman}$ WHC = 0.87; $R^2_{Spearman}$ CEC = 0.95). Total N stocks were lower in abandoned coppiced sites than in those coppiced 1–6 years ago; yet, the C/N ratios were comparable across the coppice age classes ($P > 0.05$ according to the Kruskal–Wallis test; 1–6 years ago: 14.20 ± 0.92; 8–17 years ago: 14.13 ± 1.07; abandoned coppice: 15.96 ± 1.84). In contrast, the stocks of P and K responded differently to those of SOC, exhibiting a substantial initial increase (relative increases from initial values >100%, Fig. 1e, f). Moreover, and in contrast to available P and K, the availability of NO_3-N and acidity dramatically decreased within the first year after fire to minimum values measured in the second year after fire (Fig. 1d, h). Thereafter, values increased to levels assessed shortly after fire within 6 years after harvest and soil burning (Fig. 1d, h). Thereafter, the values decreased again to levels comparable to unmanaged beech forests (Fig. 1d, h). Both NO_3-N and acidity were positively correlated ($R^2_{Spearman} = 0.59$; $P < 0.05$) and thus formed the second group of correlated variables. The third pattern of response was found for Mg, which increased moderately within 17 years to levels found in beech forests (Fig. 1g).

Coppice succession

Coppicing induced a drastic loss of understorey biomass, which was followed by one year of cropping and then secondary succession (Table 1, Fig. 2a). In the same way, coverage of overstorey, understorey and ground cover varied across the coppice age classes due to increasing growth and biomass of the coppiced trees, which led to less coverage of early-successional light-demanding herbs and grasses (i.e., the ground cover; Fig. 2a). Secondary succession started with a short herb stage (i.e., *Epilobio angustifolii-Digitalietum purpureae*), which was followed by a stage in which shrub heath was abundant (i.e., *Frangulo-Rubetum plicati* and *Calluno-Sarothamnetum*), a wood-heath stage (i.e., *Betulo pendulae-Quercetum roboris*) and a birch and oak coppice woodland (i.e., *Holco mollis-Quercetum*; Fig. 2, Table 1).

The botanical changes during secondary succession also resulted in a shift, even not significant, in species

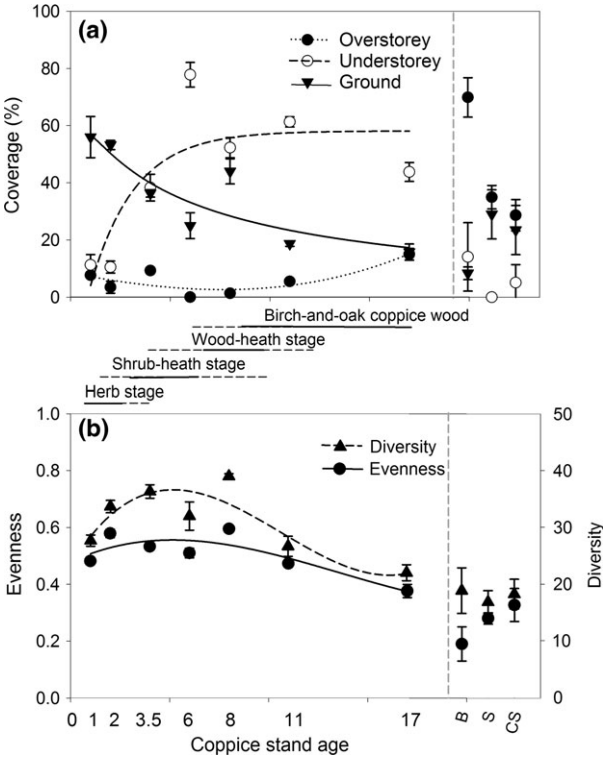


Fig. 2 Changing vegetation composition and plant community in a coppiced woodlands through time. Data are compared to vegetation surveyed in adjacent unmanaged beech forest (B), spruce forests (S) and spruce forest cultivated on formerly coppiced land (CS). Data are means ± the standard error ($n = 3$). Solid, dashed and dotted lines indicate regression curves with the following coefficients of determination: $R^2_{overstorey} = 0.71$ ($P = 0.24$); $R^2_{understorey} = 0.69$ ($P = 0.09$); $R^2_{ground} = 0.78$ ($P = 0.05$); $R^2_{diversity} = 0.77$ ($P = 0.17$) and $R^2_{evenness} = 0.72$ ($P = 0.23$).

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richness in the plots, which varied between 27 ± 2 and 39 ± 1 across the first decade of coppice regrowth. After 17 years of secondary succession, species diversity of coppice plots decreased to 22 ± 1, similar to the values found in nearby unmanaged beech and spruce forests (Fig. 2b). Similarly, evenness in the coppice forestry plots was highest during the first decade postburn (i.e., ranging from 0.47 ± 0.01 to 0.59 ± 0.01) and declined, although not significantly, to 0.38 ± 0.02 in plots 17 years postcoppice harvest. Yet, the evenness was still larger than the evenness found in nearby unmanaged forests where values of 0.28 ± 0.02 for spruce forests and 0.19 ± 0.06 for beech forests were typical (Fig. 2b).

Carbon stocks

Total C stocks of the coppice forest system (i.e., aboveground biomass plus total SOC) increased continuously across 17 years of regrowth (Fig. 3a), therewith also levelling out effects of heterogeneous BC inputs into the soils (Fig. 1b). The proportional increase in total C stock corresponded to an average annual net C accrual rate of 20 t ha^{-1} $year^{-1}$ (R^2 = 0.95; $P < 0.05$) during the 17-year regrowth period after burning. Already after 17 years, the coppice system contained as much C as adjacent mature beech and spruce forests (Fig. 3a). The majority of C was present in the coppice regrowth (branches): After 17 years of growth, there was finally less C stored in overstorey trees than in litter (Fig. 3a, b). Similarly, the contribution of different C pools to overall carbon storage changed due to regrowth patterns of tree branches after coppicing (Table 2; Figs 2a and 3b). During first two years after coppicing, there was as much organic C stored in litter than in aboveground biomass. Thereafter, it was mainly regrowth of the understory that increased C accumulation rate, and which finally elevated the proportion of C stored in aboveground biomass. Across the coppice age classes, there was thus an increased allocation of C to understorey biomass (Fig. 2a, b), while possibly small losses of soil C (Fig. 1a, b) led to lower contributions of soil organic C to the overall carbon C stock of the entire system (Fig. 3b).

Discussion

Our results show that coppicing and soil burning alters soil properties and increases biodiversity (Neary *et al.*, 1999; Certini, 2005; Müllerová *et al.*, 2015). Harvesting regrown branches from coppiced trees changes microclimatic conditions substantially and induces secondary succession (Chapin Iii *et al.*, 2011), which is in line with a previous study by Banken *et al.* (2007). Also the early-successional stages (i.e., short herb stage and shrub-heath stage) were more biodiverse than nearby unmanaged beech and spruce stands, which was already mentioned by Swanson *et al.* (2010). Thus, maintaining the landscape as a mosaic in which forest plots are in varying degrees of recovery postcoppicing may facilitate preservation of species and functional diversity (Banken *et al.*, 2007; Liman *et al.*, 2016), which is in line with findings for the Moravia region in the Czech Republic (Müllerová *et al.*, 2015).

In addition, across the succession and regrowth period of <20 years, coppiced woodlands accumulated as much C as nearby unmanaged beech and spruce forest. Measured biomass C in beech and spruce forests was in typical ranges for these forest types (Dieter & Elsasser, 2002; Lecointe *et al.*, 2006; Thuille & Schulze, 2006). However, the C storage in the studied (living museum) coppice stands was three to five times higher than for coppice stands described by Bruckman *et al.* (2011), but different stand densities, stump densities and number of branches per stump may explain differences. Even accounting for selective thinning performed to maintain

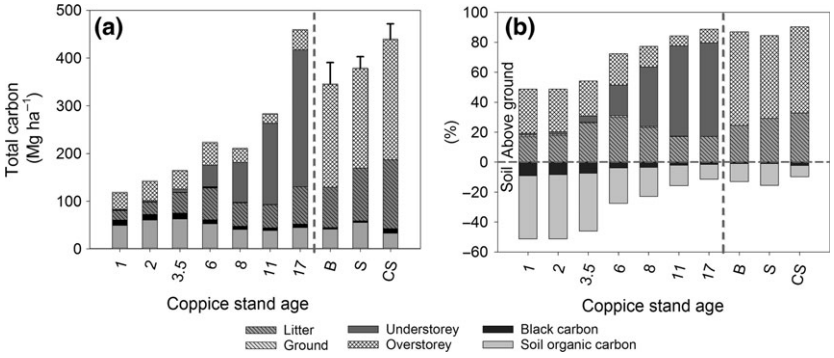


Fig. 3 Absolute (Figure a) and relative C stocks (Figure b) and their change over 17 years after coppice harvest and burning in a coppice forest system. Relative C stocks were calculated by ($C_{carbon\ pool}/C_{total\ carbon}$)*100. Carbon stocks were compared with adjacent mature beech forest (B), spruce forest (S) and spruce forest cultivated on formerly coppiced land (CS). Total carbon stocks are means ± the standard error (n = 3; see Table 1).

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Table 2 Stand characteristics of studied coppice forests (C), spruce forest on formerly coppiced land (CS), spruce forest (S) and beech forest (B)

		Coppice forest							Coppice to spruce forest			Spruce forest	Beech forest*
Stand characteristics	Unit	C1	C2	C3	C4	C5	C6	C7	CS2	CS3	CS4	S1 to S3	B1 to B5
Overstorey		*Quercus* ssp.							*Picea* ssp.				*Fagus* ssp.
Stand density	no. ha^{-1}	130	130	100	100	n.a.	70	100	333	167	167	433	255 ± 36
DBH	cm	29	29	32	35	n.a.	18	31	35	45	47	26	45 ± 3
		Betula ssp.											
Stand density	no. ha^{-1}	100	100	100	70	70	70	100	-/-	-/-	-/-	-/-	-/-
DBH	cm	18	16	17	18	18	19	22	-/-	-/-	-/-	-/-	-/-
Understorey		*Quercus* ssp.											
Stump density	no. ha^{-1}	400	200	366	333	566	466	633	-/-	-/-	-/-	-/-	-/-
Branches per stump	no. $stump^{-1}$	25	20	14	10	4	8	4	-/-	-/-	-/-	-/-	-/-
DBH	cm	0.2	0.5	1.0	3.0	4.4	5.4	6.6	-/-	-/-	-/-	-/-	-/-
		Betula ssp.											
Stump density	no. ha^{-1}	167	400	1065	333	500	366	266	-/-	-/-	-/-	-/-	-/-
Branches per stump	no. $stump^{-1}$	10	1†	1†	1†	4	4	3	-/-	-/-	-/-	-/-	-/-
DBH	cm	0.7	1.1	1.4	3.2	7.2	7.7	11.4	-/-	-/-	-/-	-/-	-/-
		Frangula spp.											
Stump density	no. ha^{-1}	266	33	n.a.	100	33	n.a.	n.a.	-/-	-/-	-/-	-/-	-/-
Branches per stump	no. $stump^{-1}$	13	11	n.a.	8	7	n.a.	n.a.	-/-	-/-	-/-	-/-	-/-
DBH	cm	0.7	1.1	n.a.	2.9	1.5	n.a.	n.a.	-/-	-/-	-/-	-/-	-/-

DBH, diameter at breast height; n.a., not applicable – no tree in unit.
*Banken *et al.* (2007), data shown as mean and their standard deviation.
†Natural regeneration from seedlings.

long-lived trees, our findings are reliable, because biomass C accumulation rates decline drastically after a stand age of about 20 years (Pregitzer & Euskirchen, 2004; Yang *et al.*, 2011). Moreover, succession may further alter soil properties, for example, enriching SOC and N-pools (Pérez & Moreno, 1998; Johnson & Curtis, 2001; Banken *et al.*, 2007).

Coppicing alters soil moisture and temperature, which are important drivers for soil organic matter (SOM) decomposition (Neary *et al.*, 1999). In addition, fire events rapidly affect quantity and quality of nutrients (Certini, 2005). Obviously, agrocoppicing and using fire to control ground vegetation are the main drivers of markedly altered nutrient (i.e., K, Mg, NO_3-N, P) stocks. However, the dynamic changes that occur during secondary succession could vary between sites differing in their abiotic properties (e.g., soil type, hydrology). Here, a fairly homogenous ecosystem was disturbed, which generally elevates availability of nutrients during the initial phase of secondary succession due to enhanced mineralization (Boerner, 1982; Banken *et al.*, 2007; Chapin Iii *et al.*, 2011), which is in line with our results for measured NO_3-N and P. Availability of K and Mg increased sequentially, K availability was highest 2–6 years after fire, and Mg availability was highest 11 years after harvest and fire, likely due to different dissolution coefficients of salts contained in ash (Ohno & Erich, 1990; Demeyer *et al.*, 2001).

The dissolution of ash should actually imply buffering effects on soil pH, but neither K nor Mg (i.e., corresponds to carbonate buffer; Demeyer *et al.*, 2001) correlated with soil pH. Yet, change of acidity levels followed temporal dynamics of NO_3-N, which is generally formed from NH_4-N via nitrification (Wan *et al.*, 2001), a process that releases protons (i.e., acidification). Sources of NH_4-N are assumed to be burning of litter and woody debris (Wan *et al.*, 2001; Banken *et al.*, 2007) and legumes (e.g., *Sarothamnus scoparius*; Johnson & Curtis, 2001; Banken *et al.*, 2007; Swanson *et al.*, 2010), which were abundant 8 years after coppicing (i.e., Calluno-Sarothamnetum plant community; Banken *et al.*, 2007). However, after 17 years, all studied nutrients equaled levels found in nearby unmanaged beech forests, and overall C stock was also comparable in the coppiced and unmanaged stands. Thus, the legacy effect of coppicing and burning understory on soil nutrient status in the Siegerland appears negligible. As a result, also changes in plant diversity and evenness were only transient in nature. Changes in SOC stocks correlated with CEC and WHC ($R^2_{Spearman}$ WHC = 0.87; $R^2_{Spearman}$ CEC = 0.95), which is known and used to improve soil quality (Lal, 2006). Furthermore, the stock of BC

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correlated with SOC, which implies that repeated coppicing and burning affect SOC quality over the long term (i.e., C/BC ratios varied significantly).

The studied coppice management system at least maintained SOC stocks, but repeated burning of soil and litter increased BC stocks and reduced C/BC ratios relative to those of beech forests, which represents the natural vegetation of this region (Bohn & Welß, 2003; Banken *et al*., 2007; Bolte *et al*., 2007). Notwithstanding that burning can increase BC stocks, their increase is hypothesized to be limited due to burning conditions (i.e., temperature and duration) and soil processes (e.g., turbation, BC degradation) that control inputs or losses of BC at the different sites (Forbes *et al*., 2006; Czimczik & Masiello, 2007; Santín *et al*., 2013). However, shortly after coppicing and burning, the stocks of total SOC were elevated. This temporal gain in total SOC may be explained by two factors (i) spatial heterogeneity within the forest ecosystems including heterogeneous input of BC or (ii) enrichment (e.g., root and litter derived) and protection of SOC (e.g., formation of chemically resistant C and stabilization of easily degradable C due to sorption on BC) during coppicing (Kuzyakov & Domanski, 2000; Yang *et al*., 2011; Borchard *et al*., 2014). Studying temporal dynamics using a space-for-time substitution can be problematic due to spatial variability of the target property, for example, total SOC. In this study, spatial variability may explain larger C stocks found in soils of patches with stand ages ≤6 years when compared to older stand ages. Nevertheless, the total SOC accrual clearly followed a temporal trend across the space-for-time chronosequence (Fig. 3), and the SOC stocks found were equal or even larger than those found in unmanaged beech forests, which is unusual for a disturbed forest system (Guo & Gifford, 2002). Consequently, we conclude that SOC enrichment was real, possibly promoted by ecophysiological processes (e.g., net primary productivity related to enrichment of SOC) in young coppice stand, and/or possibly favored by temporal increases in nutrient availability that resulted from ash input.

After harvest of coppice regrowth, soil was hoed, which incorporated litter and most upper topsoil into shallow soil layers that protected SOC during soil burning (Neary *et al*., 1999; Johnson & Curtis, 2001). However, any surplus of nonstabilized SOC should be prone to decomposition according to the soil's specific C-saturation capacity (Six *et al*., 2002; Schmidt *et al*., 2011). Nevertheless, during early succession, highly productive vegetation communities covered plots, which may have facilitated a SOC enrichment by increased root derived C inputs (Guo & Gifford, 2002; Swanson *et al*., 2010; Yang *et al*., 2011). Moreover, compared to beech forest systems, the studied coppice forest management system increased BC stocks and thereby possibly elevated SOC storage capacity in the mineral soil (Borchard *et al*., 2014), even if other authors found that added BC could potentially also accelerate litter and SOC decomposition (Wardle *et al*., 2008).

Conclusion

Our case study shows that coppiced forests can have higher biodiversity and accumulate within <20 years as much C as mature beech and spruce forests. These observations imply that it is possible to manage forests to maximize both biodiversity and C sequestration across landscapes. The increased BC stocks provide a stable soil C compound, which apparently did not prime decomposition of SOC as found for 'humus' (i.e., litter; Wardle *et al*., 2008), but which stabilized SOC in the mineral soil, so that over the long-term coppice management may promote the accumulation of soil C (Lehmann *et al*., 2008; Borchard *et al*., 2014). At present, managing understory regrowth by burning is unusual in Europe, but it is common practice in swidden cultivation in tropical regions (Van Vliet *et al*., 2012). The maintenance of coppice forests still requires regular work in the forest and a market for coppice products (e.g., biofuel), but coppicing may be a viable strategy to maintain biodiverse C-rich forests.

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