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Silviculture as a tool to support stability and diversity of forests under climate change: study from Krkonoše Mountains

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Abstract

In Europe, warming, droughts and the rise of extreme climate events have an increasing significant negative effect on forest stands. Therefore, it is necessary to create appropriate adaptation strategies of silviculture to mitigate the impacts of global climate change on forest ecosystems in Central Europe. The objectives of this paper were to evaluate stand production, structure and diversity on eight experimental research plots in the Krkonoše Mountains. Subsequently, three variants of management were compared in mixed stands at the age of 17 – 20 years originating from natural regeneration: A) control variant (stands before thinning), B) applied newly designed thinning in the context of climate change adaptation and C) simulative thinning from below. Number of trees decreased from 3,256 trees ha⁻¹ by 32% after adaptation thinning and by 36% after thinning from below. The basal area decreased in variant B by 22% and in variant C by 12%. Structural diversity and tree species richness increased after application of adaptation thinning, while decrease of diameter differentiation and total diversity was observed after thinning from below. Moreover, horizontal structure changed from aggregated spatial pattern to random distribution after the interventions, especially under adaptation thinning. The newly designed structuralizing adaptation thinning method seems to be a more suitable option in given habitat and stand conditions compared to the commonly performed thinning from below. In future, this issue will certainly require further close cooperation of forestry experts in order to arrive at optimal variants of solutions differentiated according to specific conditions.

Key words: thinning; forest adaptation; stand structure; timber production; Central Europe

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1. Introduction

Climate change substantially affects the geographical distribution of plant species worldwide (Bolte et al. 2010; Rigling et al. 2013; Scheffers et al. 2016). Most of the changes are associated with an increase in air temperature, decreasing rainfall during the growing season, and extreme climate events such as prolonged droughts, more frequent storms or uneven occurrence of precipitation (Allen et al. 2010; IPCC 2013; Spathelf et al. 2014). Global warming also increases the incidence of insect outbreaks, fungal pathogens and forest fires (Jönsson et al. 2009; Seidl et al. 2014; Bošela et al. 2018). These unfavourable conditions induce changes in tree species such as acclimatization and phenotypic plasticity, local adaptation, migration and mortality (Bussotti et al. 2015). For these reasons, forest ecosystem management deserves an increased attention (Knoke et al. 2008; Seidl et al. 2016; Dyderski et al. 2017).

The adaptation of forests to climate change is both a major challenge and a long-term task for European forestry (Lindner et al. 2010). Strategic decision-making in forest policy in Europe will have to rely on objective scientific knowledge with some uncertainty at local, regional and continental levels (Lindner et al. 2014; O'Connor et al. 2015). However, knowledge about the response of forest ecosystems to climate change is still insufficient or only partial (Pretzsch et al. 2014; Gazol et al. 2015). Forests of Central and Northern Europe are known to have increased their growth in the last century (Pretzsch et al. 2014). In contrast, many tree species suffer from drought stress at the southern edge of their current distribution in Europe and are likely to shift their span to the north and to higher altitudes (Rabasa et al. 2013; Saltré et al. 2014; Kozak & Parpan 2019). According to Hanewinkel et al. (2013), Norway spruce (*Picea abies* [L.] Karts.), the most commercial tree species in Europe, will only find

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appropriate conditions for growth in Central Europe at the highest elevations by the end of this century. Deciduous stands (beech and oak) should be dominant in Western, Central and Eastern Europe (Štefančík et al. 2018). Mainly Mediterranean oak species should be utilized in southern Europe (Moreno-Fernández et al. 2016). European beech (*Fagus sylvatica* L.), as ecologically and economically one of the most important native tree species in Europe, has experienced varied changes across the continent depending on the site quality and the altitudinal conditions of the environment (Piovesan et al. 2008; Aertsen et al. 2014; Podrázský et al. 2019). The situation was similar in the Czech Republic (Vinš et al. 1997) and in the study area of Krkonoše Mts., where it was originally an important dominant tree species (Vacek & Hejman 2012; Bulušek et al. 2016; Šimůnek et al. 2019). Bošela et al. (2016) report from the Carpathians in this respect that the increase of temperature in the growing season together with sufficient rainfall had a positive effect on the growth of beech. In contrast, Zimmermann et al. (2015) in Central Europe indicate the increasing temperature and intensity of droughts during the growing season as the main limiting factors for beech and warn against its cultivation in the area of interest.

The search for appropriate adaptive management practices in specific natural conditions should be based on structural and developmental knowledge of local natural forests in recent decades (Hemberet et al. 2012; Bošela et al. 2016; Vacek et al. 2019b). It will ordinarily be a set of cultivation measures (especially thinning and regeneration cutting) known from the past, which must be tested in changing specific habitats and stand conditions. O'Hara (2015) states that forestry should consider a number of new facts and opportunities when seeking ways of adaptive management, especially from the point of view of climate, invasive organisms, pests and introduced tree species. In particular, the continuous cover forests absorb climate extremes most effectively, improves carbon sequestration and effectively manage water (Pokorný 2013). It is however a question what species, age, spatial and genetic structure should we seek for to secure the permanence of quantity, quality and safety of production.

For ecological stability and production safety, forests should be differentiated in terms of species, spatial and age structure (Brang et al. 2014). In case of quality and quantity of timber production, it will depend mainly on the composition and structure of the forest (Coomes et al. 2014; Vacek et al. 2014), which are highly dependent on management methods (Vacek et al. 2015a). In mountain locations, forests can be managed by shelterwood silvicultural system, but it can also be a selection or partial cutting method or combination of both (Gallo et al. 2020). At the same time, highly important question is how to optimize the tending of existing stands (Štefančík 2015). Bošela et al. (2016) showed that the ways of tending the beech stands significantly influence not only the

quality and quantity of production, but also their vitality. Beech trees in the lower stand layers have not responded to an increase in temperature, suggesting that vertically diversified forests could be more stable under the conditions of warming climate. Primicia et al. (2015), on the other hand, states that trees with a high level of competition, especially in the lower layers, may be very sensitive to drought-heat stress in the future, as beech is known to be a sensitive species to temperature extremes in the initial stage of growth (Gallo et al. 2017).

Adequate species composition and form of mixing also play an important role in forest management under the unfavorable conditions of changing climate (Neuner et al. 2013; Vitali et al. 2018; Vacek et al. 2019a). Mixed forests are much more effective at mitigating the negative impacts of changing environmental conditions, especially in the cases of growth and survival (Albert et al. 2015; Pukkala 2018) and can even be more productive under climate change (Podrázský et al. 2014; Liang et al. 2016). In addition, the selection of a suitable species composition has a significant effect on carbon sequestration (Andivia et al. 2016; Cukor et al. 2017). In this respect, mixed forests have been more frequently established in Europe over the last two decades (Bravo-Oviedo et al. 2014; Forrester & Pretzsch 2015; del Río et al. 2016; Metz et al. 2016; Pretzsch & Biber 2016) and it is necessary to obtain exact knowledge about their management (Fahlvik et al. 2005; Agestam et al. 2006; Hynynen et al. 2011; Novák et al. 2017). Knowledge of thinning in various stand mixtures is not yet sufficient (Štefančík & Kamenský 2006; Štefančík 2010; Prévost & Gauthier 2012; Dhar et al. 2015; Drössler et al. 2015), especially in terms of stand growth, stability and vitality. Based on the described changes, the forest management practices should be significantly changed in following decades.

The aims of the presented study conducted in mixed forest stands (age of 17 – 20 years) in the western Krkonoše Mountains were to: (1) design a new optimized structuralizing thinning intervention in support of stand stability and diversity in the context of climate change adaptation, (2) determine basic stand characteristics, timber volume, structure (diameter, height) and diversity (species, structural, total) of mixed stands in the study area and (3) compare the state of the forest stands A) before the tending intervention, B) immediately after the model application of the newly designed structuralizing thinning intervention and C) after the simulative performance of the thinning from below.

2. Material and methods

2.1. Study area

The research was conducted in the western part of the Krkonoše Mountains close to the village Končiny, north part of the Czech Republic (GPS: 50°42'45"N, 15°27'31"E; Fig. 1). The study site belongs to the protec-

tion zone of **Krkonoše Mountains National Park** at elevation of 581–597 m a.s.l. and slope of 12–19° with east to north-eastern exposure. Annual temperature of the locality is 7.2 °C, and the annual precipitation varies around 940 mm. The highest amount of precipitation is in July (113 mm), the lowest in October (52 mm; Fig. 1). The warmest month is in July (16.5 °C), the coldest month in January (–2.2 °C). The growing season lasts about 130 days ($T_{\max} \geq 10$ °C) with an average temperature of 13.1 °C and a total precipitation of 530 mm. The average number of days with snow cover is 135 days and the number of days with precipitation ($P_{\max} \geq 1$ mm) is 125 days. The study area is qualified as a humid continental climate zone, characterized by warm to hot, humid summers and cold to severely cold winters (Dfb) according to the Köppen climate classification (Köppen 1931). According to the detailed Quitt climate classification (Quitt 1971), the area can be classified in the cold climatic region (sub-region CH 6). The parent rock consists mainly of slate, phyllites and schist. Modal cambisol is the predominant soil type.

In the area, 8 experimental research plots (ERP) in two forest stands (4 ERP in each stand) were established. Both forest stands originated from natural regeneration. The first studied stand (ERP 1–4) was composed of European beech (*Fagus sylvatica* L.), Sycamore maple (*Acer pseudoplatanus* L.), European ash (*Fraxinus excelsior* L.) and rowan (*Sorbus aucuparia* L.) of mean age 20 years. Second studied stand (ERP 5–8) was composed of main tree layer of silver birch (*Betula pendula* Roth.), rowan and beech of mean age 17 years under the standards of beech and silver fir (*Abies alba* Mill.) of mean age 51 years. In both stands Norway spruce (*Picea abies*

[L.] Karst.), European aspen (*Populus tremula* L.), wild cherry (*Prunus avium* L.) and goat willow (*Salix caprea* L.) were admixed tree species. Ash trees were weakened by *Hymenoscyphus fraxineus* (T. Kowalski) and attacks by *Hylesinus fraxini* (Panzer). In terms of phytocoenology, all ERP belonged to *Abieto-Fagetum acidophilum* (5K – Acidic Fir-Beech; Viewegh et al. 2003) and the alliance *Luzulo-Fagion sylvaticae* Lohmeyer et Tüxen in Tüxen 1954.

2.2. Data collection

Field-Map technology (IFER-Monitoring and Mapping Solutions Ltd.) was used to determine the structure of the tree layer of mixed forest stands on 8 ERP of 12.5 × 12.5 m in 2019. The positions of all trees with diameter at breast height (DBH) ≥ 4 cm were localized. The height of the live crown base and the crown projection were also measured in the tree layer, at least in 4 directions perpendicular to each other. Diameters of the tree layer were measured by a Mantax Blue metal calliper (Haglöf, Sweden) with an accuracy of 1 mm and heights were measured using a Vertex laser hypsometer (Haglöf, Sweden) with an accuracy of 0.1 m. According to the classification by Kraft (1884), the social position of the trees within a stand was assessed: 1 – predominant, 2 – dominant, 3 – co-dominant, 4 – sub-dominant, 5 – suppressed.

Subsequently, two thinning interventions were marked (trees selected for harvesting) in the stands on all the 8 ERP: (B) newly designed climate change adaptation thinning (hereinafter “adaptation thinning”) described in Table 1 and (C) thinning from below (harvested sub-dominant and suppressed trees) according to Kraft clas-

Fig. 1. Localization of mixed stands on experimental research plots Končiny 1–8 in Krkonoše Mountains National Park (protection zone) and mean monthly climatic values (1960–2018); map was made in software ArcGIS 10 (Esri).

sification with the support of natural tree species composition (preferentially spruce and beech at the expense of other tree species), which is applied in classical forest management on study site (Fig. 2); both tending methods were compared with the stands before the intervention (A). This new structuralizing adaptation thinning supporting admixed tree species was created in cooperation with the employees of Department of Silviculture, Faculty of Forestry and Wood Sciences, Czech University of Life Sciences in Prague, Czech Forestry Society (the largest forestry professional association in the Czech Republic), Forestry and Game Management Research Institute, Colloredo-Mannsfeld spol. s.r.o. (the largest private forest owner in the Czech Republic) and Jilemnice municipality forests. A form with 10 indicators was cre-

ated by main author according to references (Slodičák & Novák 2007; Cordonnier et al. 2013; Pokorný 2013; Forest Europe 2015; Vacek et al. 2019a) and subsequently the experts in silvicultural field (15 academic and practical persons) numbered the indicators according to the importance from 1 (min.) – 10 (max.) in relation to mitigate climate change. In order to simplify this thinning variant, the 3 least rated indicators were excluded from the evaluation. Trees selected for harvesting were documented in Field-Map technology. From January to February 2020, adaptation thinning was applied on all 8 ERP and immediate stand changes were compared with the simulative thinning from below.

Table 1. Considered suitable objectives with description and relevance of the newly designed climate change adaptation thinning.

Order of importance	Indicators of thinning choice	Description
1	Species composition addressing climate change	adaptable tree species (for drought, climate extremes), shift of forest vegetation zones, optimal stand mixtures in terms of ecological criteria (valency, susceptibility to damage, tolerance and adaptability of tree species)
2	Health status	resistance, vitality, foliation, abiotic and biotic damage
3	Species diversity	species richness (number of tree species), heterogeneity and evenness
4	Structural diversity	diameter and height (age) differentiation, vertical canopy, horizontal structure
5	Quality of production	silvicultural potential, stem quality (straightness, defects), crown quality (length, regularity, deflection)
6	Increment	timber, growth tendencies; periodic, mean and value increment
7	Continuous soil protection	canopy conservation, area and distribution of crown projections

Note: The criteria in the table should be applied in close relation to specific habitat and stand conditions.

Fig. 2. Vertical schema of mixed forest stand A) before thinning, B) after application of the climate change adaptation thinning and C) thinning from below; figure was made in software SIBYLA Triquetra 10 (Fabrika & Pretzsch, TUZVO).

2.3. Data analysis

The basic structure, diversity and production characteristics of the tree layer were evaluated by the SIBYLA Triquetra 10 forest growth simulator (Fabrika & Ďurský 2005). The PointPro 2.1 (Zahradník & Puš, ČZU) program was used to calculate the characteristics of the horizontal layout of the individuals on the plots. For the evaluation of the spatial pattern, the following indicators were computed: index of non-randomness (Pielou 1959; Mountford 1961), aggregation index (Clark & Evans 1954), index of cluster size (David & Moore 1954) based on the 10 quadrats and Ripley L -function (Ripley 1981). The test of the significance of deviations from the values expected for the random arrangement of the points was made using Monte Carlo simulations. The mean values of L -function were estimated as arithmetic means from L -functions computed for 999 randomly generated point structures. Species and structural diversity were evaluated by species richness (Margalef 1958), species heterogeneity (Shannon 1948), species evenness (Pielou 1975), Arten-profile index (Pretzsch 2006), diameter and height differentiation (Földner 1995), crown differentiation and total stand diversity (Jaehne & Dohrenbusch 1997; Table 2). Stand volume was calculated according to Petráš & Pajtlík (1991). The relative stand density index (SDI; Reineke 1933) and the crown closure (CC; Crookston & Stage 1999) were observed for each plot.

Table 2. Overview of indices describing the stand diversity and their common interpretation.

Criterion	Quantifiers	Label	Reference	Evaluation
Species diversity	Richness	D (Mi)	Margalef (1958)	minimum $D=0$, higher D = higher values
	Heterogeneity	H' (Si)	Shannon (1948)	minimum $H'=0$, higher H' = higher values
	Evenness	E (Pii)	Pielou (1975)	range 0 – 1; minimum $E=0$, maximum $E=1$
Horizontal structure	Index of non-randomness	α (P&Mi)	Pielou (1959); Mountford (1961)	mean value $\alpha=1$; aggregation $\alpha>1$; regularity $\alpha<1$
	Aggregation index	R (C&Ei)	Clark & Evans (1954)	mean value $R=1$; aggregation $R<1$; regularity $R>1$
	Index of cluster size	CS (D&Mi)	David & Moore (1954)	mean value $CS=0$; aggregation $CS>0$; regularity $CS<0$
Vertical structure	Arten-profile index	A (Pri)	Pretsch (2006)	range 0 – 1; balanced vertical structure $A<0.3$; selection forest $A>0.9$
Structure differentiation	Diameter dif.	TM_d (Fi)	Füldner (1995)	range 0 – 1; low $TM<0.3$; very high differentiation $TM>0.7$
	Height dif.	TM_h (Fi)		
Complex diversity	Stand diversity	B (J&Di)	Jaehne & Dohrenbusch (1997)	monotonous structure $B<4$; uneven structure $B=6-8$; very diverse structure $B>9$

Unconstrained principal component analysis (PCA) in Canoco 5 (Šmilauer & Lepš 2014) was used to analyze the relationships between stand structure, timber production and diversity of three variants of management. Data were log-transformed, centred and standardized before the analysis. The results of PCA were exported into the form of ordination diagram.

3. Results

3.1. Stand characteristics, structure and timber volume

Tree density ranged from 2,112 to 5,184 trees per ha⁻¹ before thinning (control variant A), after model adaptation thinning (variant B) number of trees was between 1,280 to 3,968 trees ha⁻¹ and after simulative thinning from below (variant C) it was from 1,408 to 3,008 trees ha⁻¹ (Table 3). Number of trees decreased from mean density of 3,256 trees ha⁻¹ by 32% after adaptation thinning (to 2,192 trees ha⁻¹) and by 36% after thinning from below (to 2,096 trees ha⁻¹). In variant A, basal area ranged

between 21.8 – 39.2 m² ha⁻¹ and stand volume reached 127 – 377 m³ ha⁻¹. After thinning interventions, stand volume decreased from 185 to 159 m³ ha⁻¹ in variant B (–14%) and to 175 m³ ha⁻¹ in variant C (–6%). The basal area decreased in variant B by 22% and in variant C by 12%. On the other hand, mean DBH increased by 8% after adaptation thinning, respectively by 16% after thinning from below. Similar positive trend after interventions was observed in stem volume and stand height. Mean height was higher in variant C (12.5 m) compared to variant A (10.8 m) and variant B (11.2 m). SDI decreased from 0.75 to 0.58 in variant B and to 0.63 in variant C. The highest differences among thinning variants were observed in tree density and mean height.

The diameter frequency on ERP 1–8 before thinning (variant A) showed visible left-skewed selection distribution, where beech dominated the overstorey and birch, beech and rowan mostly formed the understory (Fig. 3 and 4). Sycamore was evenly represented across diameter classes. Rowan was observed only in diameter to 12 cm. Birch was dominant in all diameter classes to 20 cm. After adaptation thinning number of trees decreased in classes

Table 3. Basic stand characteristics on experimental research plots 1 – 8 in A) control variant (before thinning), B) after applied climate change adaptation thinning and C) after simulative thinning from below.

ERP	Type	dbh [cm]	h [m]	v [m ³]	f	N [trees ha ⁻¹]	BA [m ² ha ⁻¹]	V [m ³ ha ⁻¹]	HDR	CC [%]	SDI
1	A	11.9	10.3	0.069	0.596	3,520	39.2	242	87.3	99.2	0.91
	B	13.8 ↗	11.4 ↗	0.100 ↗	0.584 ↗	2,112 ↘	31.4 ↘	212 ↘	83.1 ↘	99.2 ↘	0.68 ↘
	C	14.0 ↗	12.2 ↗	0.099 ↗	0.527 ↗	2,432 ↘	37.2 ↘	241 ↘	87.4 ↗	98.7 ↘	0.82 ↘
2	A	10.0	9.4	0.038	0.519	3,712	29.2	142	94.1	99.4	0.77
	B	11.0 ↗	9.5 ↗	0.051 ↗	0.563 ↗	2,304 ↘	21.7 ↘	118 ↘	86.8 ↘	98.6 ↘	0.55 ↘
	C	11.9 ↗	10.8 ↗	0.058 ↗	0.482 ↗	1,920 ↘	21.1 ↘	112 ↘	91.4 ↘	98.1 ↘	0.53 ↘
3	A	9.1	10.0	0.031	0.475	4,096	26.4	127	110.3	99.4	0.75
	B	9.3 ↗	10.0 ↘	0.035 ↗	0.509 ↗	2,752 ↘	18.5 ↘	95 ↘	107.6 ↘	98.9 ↘	0.52 ↘
	C	10.4 ↗	11.4 ↗	0.044 ↗	0.452 ↗	2,880 ↘	24.3 ↘	126 ↘	110.0 ↘	99.1 ↘	0.66 ↘
4	A	11.7	10.8	0.070	0.601	2,368	25.2	166	92.5	99.2	0.65
	B	11.1 ↘	10.5 ↘	0.065 ↘	0.634 ↘	1,728 ↘	16.4 ↘	112 ↘	94.9 ↘	97.4 ↘	0.45 ↘
	C	13.4 ↗	12.3 ↗	0.097 ↗	0.557 ↗	1,664 ↘	23.3 ↘	162 ↘	92.2 ↘	98.7 ↘	0.55 ↘
5	A	12.5	12.0	0.067	0.455	2,624	31.8	177	96.4	99.4	0.77
	B	13.9 ↗	12.8 ↗	0.091 ↗	0.469 ↗	1,664 ↘	25.1 ↘	152 ↘	92.2 ↘	98.9 ↘	0.58 ↘
	C	14.0 ↗	13.5 ↗	0.091 ↗	0.435 ↗	1,792 ↘	27.4 ↘	163 ↘	96.6 ↗	99.2 ↘	0.63 ↘
6	A	8.4	11.4	0.021	0.332	5,184	28.7	109	136.3	99.5	0.85
	B	8.7 ↗	11.7 ↗	0.023 ↗	0.336 ↗	3,968 ↘	23.6 ↘	93 ↘	134.5 ↘	99.4 ↘	0.69 ↘
	C	9.5 ↗	12.9 ↗	0.029 ↗	0.312 ↗	3,008 ↘	21.2 ↘	86 ↘	136.5 ↗	99.2 ↘	0.61 ↘
7	A	14.1	11.5	0.155	0.861	2,432	37.8	377	81.8	99.2	0.78
	B	16.3 ↗	12.4 ↗	0.215 ↗	0.829 ↗	1,728 ↘	35.3 ↘	372 ↘	76.3 ↘	98.5 ↘	0.70 ↘
	C	16.8 ↗	13.5 ↗	0.226 ↗	0.755 ↗	1,664 ↘	36.5 ↘	376 ↘	80.4 ↘	98.7 ↘	0.73 ↘
8	A	11.5	11.1	0.067	0.575	2,112	21.8	141	96.9	98.7	0.52
	B	12.8 ↗	11.6 ↗	0.091 ↗	0.610 ↗	1,280 ↘	16.1 ↘	117 ↘	91.1 ↘	95.2 ↘	0.41 ↘
	C	13.7 ↗	13.2 ↗	0.099 ↗	0.508 ↗	1,408 ↘	20.5 ↘	139 ↘	96.5 ↘	97.7 ↘	0.49 ↘

Notes: dbh – mean quadratic diameter at breast height, h – mean height, v – mean tree volume, f – form factor, N – number of trees per hectare, BA – basal area, V – stand volume, HDR – height to diameter ratio, CC – canopy closure, SDI – stand density index; changes: ↘ – decrease, ↗ – increase.

from 4 to 16 cm, while the center of thinning interest for variant C ranged from 4 to 8 cm. After thinning, the highest decrease in tree numbers was observed in first class 4–6 cm in both variants – in variant B by –448 trees ha^{-1} (–34%) and especially in variant C by –960 trees ha^{-1} (–74%). In variant B, the percentage of harvested trees was evenly distributed across diameter classes, while for variant C the percentage of harvested trees rapidly decreased with increasing diameter. In variant B, the most frequent diameter class have been still 4–6 cm, while it was moved to class 6–8 cm in variant C.

3.2. Biodiversity of tree layer

Before thinning (variant A), the main tree species on ERP 1–8 was birch (39%), followed by beech (21%), sycamore (11%), rowan (11%) and ash (9%; Fig. 5). After adaptation thinning (variant B) the highest change was observed in decrease (by 5%) of ash share due to weakening of trees by pests (see Study area). Share of birch decreased by 4%, while share of sycamore increased by 3% and beech by 2%. Share of all other tree species also increased. In variant C (thinning from below), share of

Fig. 3. Histogram of tree layer diameter structure according to main tree species in A) control variant (before thinning), B) after applied climate change adaptation thinning and C) after simulative thinning from below.

Fig. 4. Summary histogram of complete tree layer diameter structure in A) control variant (before thinning), B) after applied climate change adaptation thinning and C) after simulative thinning from below.

rowan decreased by 10% and willow by 4% due to their occurrence predominantly in the understory. On the other hand, an increase was observed in beech (by 2%) and especially in birch (by 8%). Higher mean species richness D was observed in variant B (0.573) and variant A (0.559) compared to variant C (0.346; Table 4). Together with species heterogeneity H' both indices showed high diversity compared to moderate diversity in variant C. No differences were observed in species evenness E among thinning variants indicating very high diversity (A – 0.721, B – 0.726, C – 0.736).

In terms of structural diversity, higher height differentiation TM_h was observed in stands after adaptation thinning (0.292) and in control variant (0.256) compared to thinning from below (0.212; Table 4). Similarly, lower diameter differentiation TM_d was observed in variant

Fig. 5. Tree species composition in A) control variant (before thinning), B) after applied climate change adaptation thinning and C) after simulative thinning from below; others tree species include: *Prunus avium*, *Abies alba* and *Populus tremula*.

Table 4. Tree layer biodiversity on experimental research plots 1–8 in A) control variant (before thinning), B) after applied climate change adaptation thinning and C) after simulative thinning from below.

ERP	Type	D [Mi]	H' [Si]	E [Pii]	α^* [P&Mi]	R* [C&Ei]	CS* [D&Mi]	A [Pri]	TM _d [Fi]	TM _h [Fi]	B [J&Di]
1	A	0.490	0.576	0.824	1.602	1.006	0.189	0.692	0.371	0.309	7.846
	B	0.523 ↗	0.540 ↘	0.773 ↘	1.303 ↘	1.141 ↗	−0.031 ↘	0.591 ↘	0.390 ↗	0.287 ↘	7.596 ↘
	C	0.385 ↘	0.560 ↘	0.930 ↗	1.140 ↘	1.020 ↗	0.116 ↘	0.719 ↗	0.344 ↘	0.220 ↘	6.740 ↘
2	A	0.852	0.797	0.883	1.677 ^A	0.913 ^A	0.086	0.756	0.308	0.242	9.111
	B	0.904 ↗	0.802 ↗	0.888 ↗	1.568 ↘	0.979 ↗	−0.053 ↘	0.689 ↘	0.340 ↗	0.277 ↗	8.866 ↘
	C	0.661 ↘	0.625 ↘	0.803 ↘	1.198 ↘	1.082 ↗	−0.088 ↘	0.717 ↘	0.306 ↘	0.225 ↘	7.840 ↘
3	A	0.721	0.604	0.715	1.619 ^A	0.980	0.190	0.672	0.334	0.244	8.139
	B	0.631 ↘	0.566 ↘	0.727 ↗	1.221 ↘	0.935 ↘	−0.060 ↘	0.673 ↗	0.354 ↗	0.256 ↗	7.422
	C	0.502 ↘	0.589 ↘	0.843 ↗	1.561 ↘	0.987 ↗	−0.060 ↘	0.710 ↗	0.306 ↘	0.205 ↘	6.887 ↘
4	A	0.772	0.571	0.676	1.622	1.009	0.239	0.664	0.408	0.310	8.107
	B	0.805 ↗	0.624 ↗	0.738 ↗	0.845 ↘	1.129 ↗	−0.085 ↘	0.609 ↘	0.457 ↗	0.367 ↗	8.378 ↗
	C	0.539 ↘	0.495 ↘	0.708 ↗	2.073 ^A	0.893 ^A	0.165 ↘	0.673 ↗	0.396 ↘	0.249 ↘	7.105 ↘
5	A	0.254	0.420	0.880	1.364	1.085	0.042	0.793	0.384	0.259	6.193
	B	0.270 ↗	0.421 ↗	0.894 ↗	1.259 ↘	1.209 ↗	−0.525 ^R ↘	0.761 ↘	0.406 ↗	0.261 ↗	5.866 ↘
	C	0.133 ↘	0.301 ↘	1.000 ↗	1.177 ↘	1.051 ↘	−0.215 ↘	0.786 ↗	0.344 ↘	0.229 ↘	4.356 ↘
6	A	0.351	0.334	0.555	1.549 ^A	0.962	0.239	0.551	0.261	0.197	5.787
	B	0.362 ↗	0.372 ↗	0.618 ↗	1.485 ↘	0.967 ↗	−0.185 ↘	0.558 ↗	0.238 ↘	0.183 ↘	5.533 ↘
	C	0.000 ↘	0.000 ↘	0.000 ↘	1.452 ↘	1.083 ↗	−0.210 ↘	0.495 ↘	0.210 ↘	0.117 ↘	3.310 ↘
7	A	0.513	0.497	0.711	1.192	1.133	−0.135	0.519	0.294	0.233	8.039
	B	0.537 ↗	0.502 ↗	0.714 ↗	0.809 ↘	1.296 ^R ↗	−0.384 ^R ↘	0.521 ↗	0.390 ↗	0.301 ↗	8.044 ↗
	C	0.270 ↘	0.472 ↘	0.989 ↗	1.535 ↗	1.021 ↗	−0.284 ↘	0.616 ↗	0.276 ↘	0.197 ↘	5.945 ↘
8	A	0.523	0.371	0.531	1.851 ^A	0.663 ^A	−0.334	0.510	0.324	0.267	6.924
	B	0.559 ↗	0.344 ↘	0.492 ↘	0.901 ↘	0.930 ↗	−0.559 ^R ↘	0.459 ↘	0.501 ↗	0.417 ↗	6.492 ↘
	C	0.276 ↘	0.295 ↘	0.618 ↗	1.353 ↘	0.750 ↗	−0.348 ↘	0.546 ↗	0.335 ↗	0.254 ↘	6.039 ↘

Notes: D – species richness, H' – species heterogeneity, E – species evenness, α – index of non-randomness, R – aggregation index, CS – index of cluster size, A – Arten-profile index, TM_d – index of diameter differentiation, TM_h – index of height differentiation, B – stand diversity index; changes: ↘ – decrease, ↗ – increase;

* statistically significant ($\alpha = 0.05$) for horizontal structure (^A – aggregation, ^R – regularity).

C (0.314) compared to variant B (0.383) and variant A (0.335). Structural differentiation was low to medium in all variants. No difference was found in vertical Arten-profile index A (A – 0.644, B – 0.606, C – 0.657) that showed high diversity. Complex diversity index B showed lower diversity in variant C (6.027) compared to other two variants (A – 7.518, B – 7.257).

Differences were observed also in horizontal structure. Index of non-randomness showed higher tendency to aggregation in stands before intervention compared to thinning variants. Similarly, other both indices showed the highest tendency to regular spatial pattern of trees in variant B. In the case of 5 ERP, horizontal structure moved from aggregated distribution to random, respectively from random to regular. Moreover, *L*-function showed shift from aggregated structure in variant A (from distance of 0.5 m) to prevailing random structure

in the other two variants, especially in adaptation thinning (Fig. 6).

3.3. Interactions between stand parameters, structure, diversity and management

The results of PCA are presented in an ordination diagram in Fig. 7. The first ordination axis explains 31.1% of data variability, the first two axes together explain 59.4% and the first four axes 84.7%. The x-axis illustrates the slenderness quotient (HDR) with crown closure (canopy) and the y-axis represents the mean stand height. Stand volume was positively correlated with mean diameter, tree volume and basal area, while these parameters were negatively correlated with tree density, HDR and index of non-randomness. Aggregation spatial pattern

Fig. 6. Spatial pattern of tree layer on experimental research plots 1–8 expressed by *L*-function in A) control variant (before thinning), B) after applied climate change adaptation thinning and C) after simulative thinning from below; the black line represents the *L*-function for real distances of trees; the bold gray line represents the mean course for random spatial distribution and the two thinner central curves represent 95% interval of reliability; when the black line of tree distribution on the ERP is under (below) this interval, it indicates a tendency of trees toward aggregation (regular) distribution; graphs made in software PointPro 2.1 (Zahradník & Puš, ČZU).

was increasing with increasing number of trees. Indices of diversity were positively correlated to each other. Mean height and species richness are the highest explanatory variable in ordination diagram compared to low explanation in SDI and Arten-profile index. Differences among all parameters were remarkable for ERP as marks of each records were relatively distant one from another whereas marks symbolizing variants of thinning were closer to each other, especially for adaptation thinning (variant B) and control variant A (before thinning). Before thinning, forest stands were characterized by high number of trees with aggregated structure, stands after applied adaptation thinning (variant B) had higher diameter and high differentiation and species evenness and the highest stand height was typical for simulative thinning from below (variant C). Generally, type of thinning had the highest effect (expected differences between plots) on mean height, species, structure and total diversity.

strategy to cope with climate change (Jactel et al. 2009; Kolström et al. 2011; Bravo-Oviedo et al. 2014; Reims et al. 2015). Emphasis is at the same time placed on the variability of mixed forests corresponding to the given habitat and stand conditions (Seidl et al. 2011; Rio et al. 2016; Králíček et al. 2017; Vacek et al. 2017a; Mikulénka et al. 2020). Mixed forest tending is more complicated and require higher forestry expertise and qualification than tending of homogenous stands due to the need of consistently respecting the characteristics of tree species, inter- and intra-specific competition in close relation to habitat conditions (Slodičák & Novák 2007; Poleno et al. 2009; Švec et al. 2015; Slanař et al. 2017).

Our results suggest that 33% of the trees, 22% of the basal area and 12% of the stand volume was harvested from the original number of 2,112 to 5,184 trees ha⁻¹ in case of adaptation thinning, and it was 36% of the trees, 12% of the basal area and 6% of stand volume in case

Fig. 7. Ordination diagram showing results of the PCA of relationships between variants of thinning (A – control before thinning, B – adaptation thinning, C – thinning from below), stand characteristics (Stand volume, Tree volume, Basal area, DBH, HDR, Height, Canopy, Tree number) and structural diversity (D , H' , E , A , TMD , TMh , α and B indices); Symbols indicate ● plots 1–8 and ■ variants of thinning (A, B, C).

4. Discussion

Climate change is currently one of the major challenges for the protection and management of forest ecosystems (Cotillas et al. 2009; Borys et al. 2016). It is therefore necessary to develop appropriate forestry strategies to mitigate the impact of global climate change on forest ecosystems (Chapin et al. 2001; Alvarez et al. 2016). In relation to ecological stability, it is then desirable to grow forests spatially, age and species differentiated (Brang et al. 2014). Promotion of mixed forests and species diversity is therefore often identified as a forest adaptation

of thinning from below. For comparison, slightly higher density (4,878 – 5,267 trees ha⁻¹) was found before the intervention in equally old (19 years) mixed spruce-beech stands in South Bohemia (Novák & Slodičák 2009). On the other hand, lower number of trees (1,566 – 2,240 trees ha⁻¹), such as stand volume (67 – 77 m³ ha⁻¹), was observed in mixed birch-dominated stands of age 20 years (Vacek et al. 2009). In parallel, there was an increase in average DBH by 8% and height by 4% for adaptation thinning and 16% in height and DBH for thinning from below. As in Waskiewicz et al. (2013),

the aim of the stand tending was to improve the stability of the stand and keep the desired species composition in the young mixture. In our case, similarly to Novák et al. (2017), it is a lower intensity of the intervention, however, the internal stability of the stands of studied mixtures has been demonstrably increased. In connection with the intensity of the intervention, Konôpka et al. (2008) showed that even with higher intensity of the intervention (40% of the removed basal area), the stability of the stand can be significantly increased.

More intensive opening of forest stands has a greater impact on habitat and stand conditions, as evidenced by many researches (Poleno 1993, 1994; Jaworski 2000; Štefančík 2006, 2015; Rio et al. 2016). These are primarily microclimatic and ecological conditions, which can have a significant impact on the stability, production, health and vitality of stands (Chroust 1997). In the context of major global climate change, a number of papers from the Mediterranean conditions cite, for example, Roig et al. (2005); Gea-Izquierdo et al. (2009); Guillemot et al. (2015); Rio et al. (2016), that stronger tending interventions improve the moisture conditions in the stand and consequently the growth of the stand can be increased compared to stands without any intervention. Wherefore, higher opening of the vegetation under adaptation thinning according should have a more favourable effect on microclimatic and ecological conditions in the monitored stands compared to the thinning from below as well as on the diversity and development of herb layer in relation to light availability (Axmannová et al. 2011; Hédli et al. 2017). It is also noted that the high intensity of tending leads to an increase in the vitality of dominant trees in the stand after severe drought-stress events (Misson et al. 2003; Kohler et al. 2010; Sohn et al. 2013). Also, Bošela et al. (2016) showed that methods of differentiated tending of beech stands significantly influenced their vitality in conditions of the Carpathians, where the average temperature in the growing season has been rising since 1950.

The level of stress from water deficit is strongly influenced also by the species composition of stands. Prior to tending interventions, the main species on the experimental plots were birch (39%), beech (21%), sycamore (11%), rowan (11%) and ash (9%). After adaptation thinning, the largest change in the species proportion was observed for ash as a result of damage by *Hylesinus fraxini* and *Hymenoscyphus fraxineus*, similarly to situation in other parts of the Krkonoše Mts. (Vacek et al. 2015b, 2017b). Furthermore, the prevailing proportion of birch trees decreased, while there was a percentage increase in beech, sycamore, willow, rowan and other admixed tree species (cherry, aspen, fir). In case of thinning from below, the proportion of birch, beech and sycamore increased on the expense of the proportion of rowan and willow due to minimal economic use. In

relation to tree species composition, researches showed a higher probability of drought stress in young spruce stands than in beech or other deciduous trees (Jelínek & Kantor 2001; Borken & Beese 2005; Novák & Slodičák 2009). From this point of view, this proposed thinning could be a suitable adaptation option to decrease the risk of drought stress in forest stands.

In terms of diversity, adaptation thinning caused a shift in horizontal structure from aggregated to random-to-regular spatial distribution of trees. Furthermore, there was an increase in the diameter and height differentiation of the stand and an increase in species richness compared to the thinning from below or control variant. Thinning from below caused significant overall decrease in diversity of the stand. Our simulative results are consistent with Fahlvik et al. (2015), Holmstrom et al. (2016) and Novák et al. (2017), who show a significant influence of stand tending in young mixed stands on their structure. It should be noted that both the type of tending and the mixing, including the horizontal and vertical structure of the stands, modify the use of aboveground and underground resources and affect the distribution of growth among the trees in the stand and thereby the way of competition (Pretzsch & Schütze 2014, 2015; Rio et al. 2014). More intensive tending intervention generally increases the growth capability of a stand and its vitality (Chroust 1997; Štefančík et al. 2018; Sharma et al. 2019). In case of adaptation thinning, we assumed that the new space created by tending intervention was necessary for the growth of promising future target trees, which will better resist the ongoing climate changes.

5. Conclusion

In this paper we tried to create a conceptual framework of the most important principles of stand tending, which take into account the negative effects of global climate change on forest stands on the example of Krkonoše Mts. The proposed thinning system offers various ways of management, especially in terms of optimizing ecological and economic criteria for selecting tree species. The results show that adaptation thinning seems to be a more suitable option in given habitat and stand conditions compared to the thinning from below. There was an increase in structural diversity and species richness in adaptation thinning which is heading to higher stability of forest stands. In addition, it is important to ensure that the proposed silvicultural measures can deliver the required social and environmental functions and services of forest ecosystems continuously and in the highest possible quantity and quality. As a next step, different thinning approaches for different climate scenarios and site conditions should be developed and tested in field conditions, in order to verify their validity and feasibility.

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