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EDITORIAL

Editorial to the thematic issue:
Silviculture and forest production under climate change

Dear readers,

It is with great honour that we present to you this thematic issue "Silviculture and forest production under climate change" with the contributions of unique research work dealing with the current environmental problems. The ongoing global climate change poses a challenge for various sectors, including forestry. On the one hand, forest ecosystems are exposed to and threatened by climate change and, on the other hand, forests have the ability to influence the course of climate change, by regulating the water regime and air quality, carbon sequestration or reducing climate extremes. It is therefore important not to see climate change only as a threat causing large-scale forest disturbances and economic recession, but also as an opportunity for innovative approaches in forest management, protection and silviculture.

Climate change gradually and significantly affects the ability of forests to fulfil their functions and provide ecosystem services to the human society. In recent years, there has been a constant increase in average air temperature in Central Europe, increasing inequality in the distribution of precipitation, increasing frequencies of storms and torrential rains followed by floods. Evaporation is generally increasing because of higher temperatures while available moisture in forest ecosystems is significantly decreasing. The result is an increase in the incidence of long-term droughts, followed by an increase in stress factors for forest tree species and also the occurrence of forest fires, in extreme cases. The simultaneous effect of these factors reduces the vitality of forest stands and positively affects the population dynamics of insect pests and increases the occurrence of fungal pathogens. In the conditions of Central Europe, Norway spruce is a particularly vulnerable tree species, but, actually, all our main commercial tree species are threatened. In the Czech and Slovak Republics, unplanned salvage logging has accounted for ca 65% of the total timber harvest in the last 10 years as a result of the above-mentioned factors, of which 80% to 85% were the coniferous tree species.

Depending on the habitat conditions and the specifics of the stand, climate change may also have a positive effect. Tree growth and the stand production can be positively affected by increased CO_2 concentrations in the air. The result is therefore a change of habitat potential for silviculture of particular tree species or their mixtures. As a result of climate change, habitat conditions are shifting into lower forest vegetation zones. This significantly worsens the conditions particularly for species with high demands on soil and air humidity. For these reasons, it will be necessary to apply various adaptation measures, based on a change in the species composition with the aim of creating mixed stands that show high stability and economic viability at the same time. Simultaneously, it will be necessary to replace declining spruce stands in the lower and middle elevation zones with more resistant species, such as European larch, silver fir and Scots pine. In deciduous trees, a higher proportion of economically highly valuable species, such as oaks, maples and wild cherry, is expected. We must also not forget the significant potential of the introduced tree species, such as Douglas fir, northern red oak, ponderosa pine and black pine, including new species that have not been used in forestry in Central Europe so far. On the other hand, caution is advised, as climate change may accelerate the spread of invasive tree species such as *Ailanthus altissima*, with a negative impact on native tree species. However, the desired change in the tree composition based on an increasing diversity will require higher initial costs for artificial regeneration and tending interventions. In genetically valuable stands, natural forest regeneration is one of the most important adaptation measures. When establishing mixed stands, group mixing is generally recommended; prolongation of the regenerative period and the use of continuous-cover silvicultural methods lead to the creation of more stable structurally differentiated stands. The general economic goal that is realistically achieved in the natural conditions of Central Europe is the structure of a forest mixed in groups, spatially and age-wise significantly structured with tree species corresponding to the expected climate development.

Successful implementation of these measures requires a change in decision-making in forest management, comprehensively considering changing natural conditions, economic indicators and environmental limits of forest cultivation. What our forests will look like for future generations and what the resulting impact of climate change will be on forestry is in the hands of forest managers, depending on supportive forestry research and (we must not forget) publishing scientific papers in the Central European Forestry Journal.

This monothematic issue contains 7 papers including one review paper **"Assisted migration vs. close-to-nature forestry: what are the prospects for tree populations under climate change?"** The authors reviewed biological background and limits of mechanisms by which tree populations may cope with climate change: adaptation by natural selection, gene flow, epigenetic phenomena and phenotypic plasticity, as well as forest management strategies which rely on these mechanisms. It was argued that maintaining genetic diversity is important in the long-term view but natural selection cannot ensure sufficiently rapid response to environmental change. On the other hand, epigenetic memory effects may change adaptively relevant traits within a single generation, while close-to-nature forestry practices are the basic requirement to make use of epigenetics.

The article **"Substitution of Norway spruce for Douglas-fir: changes of soil microbial activities as climate change induced shift in species composition"** is a case study, based on an assumption that this species is a convenient substitution for declining Norway spruce stands under current climatic change (extreme drought, bark beetle outbreak). In the presented study, the respiration rate and nitrification intensity were compared in the stand parts dominated by Norway spruce and Douglas-fir in comparable site conditions. It was concluded that admixture of Douglas-fir will accelerate and promote microbial activities, resulting in faster litter decomposition (increased potential and actual respiration activity) and more pronounced nitrogen dynamics (nitrification).

The above-mentioned tree species was also an object of the paper **"Litterfall nutrient return in thinned young stands with Douglas fir"** dealing with both the amount and properties of litterfall in 18-year-old to 20-year-old stands with Douglas fir in 2011. Two experimental plots were Douglas fir dominated and one showed ca. 20–30% share of Scots pine at the beginning of observation. The experiments consisted of two treatments such as unthinned control and 50–62% trees' density reduction accounting for 43–59% basal area reduction. It was concluded that young Douglas fir dominated stands react to very heavy thinning by a reduced amount of litterfall and a quicker rate of litter decomposition. This effect showed similar trend when compared to native conifers (Norway spruce, Scots pine).

The chestnut species is also considered to be in danger of extinction due to the effects of climate change, particular diseases (e.g. *Cryphonectria parasitica*) and land mismanagement. This is the crucial statement of the original article **"Soil erosion in Mediterranean chestnut tree plantations at risk due to climate change and land abandonment"**. This study quantifies soil losses in areas under tree canopies and open areas considering two plantations of different age; planted in the 1960s and 1990s. The results showed that the open areas and the youngest plantation show higher soil losses than the areas under the trees and the oldest plantation.

Another tree species with higher tolerance to dry periods and to a lower soil nutrient content is **"Northern red oak (*Quercus rubra* L.) as a species suitable for the upcoming seasons with frequent dry periods".** The aim of this study was to assess the regeneration of Northern red oak and its growth in the first years after planting under given climatic conditions of recent years in an area characterized by longer dry periods. It also assesses the value of the Lang's rain factor to evaluate local climatic conditions. The results of surveys on research plots with sessile oak and red oak plantations confirmed that in areas with higher temperatures and lower precipitation as expressed by Lang's rain factor, Northern red oak can be successfully grown, showing better resistance to dry climatic conditions than domestic oaks in both growth and field performance first years after planting.

In the context of climate change, the crucial question is how silvicultural treatments should be modified, in order to reach favourable conditions for initiating natural regeneration of forest stands. Therefore, the last two articles dealt just with this issue. The aim of the first study **"Does shelterwood regeneration on natural Scots pine sites under changing environmental conditions represent a viable alternative to traditional clear-cut management?"** was to evaluate the influence of the basal area reduction (stand densities: clear-cut, 0.4., 0.6 and 0.8) and soil preparation (milling cutter, forestry mulcher, brush rake and control variant without soil preparation) on the regeneration of Scots pine in the conditions of natural pine sites (Northern Bohemia). Based on the research it was concluded that the best outcome of natural regeneration was achieved after seed cut decreasing the stand density to 0.4. This was manifested by sufficient seed numbers, highest regeneration density, superior seedling characteristics and positive effect on soil moisture and temperature characteristics. Additionally, shelterwood regeneration on natural Scots pine sites was recommend as a promising silvicultural approach and may in the future become an important tool in mitigating negative effects of climate change.

The last, but not the least important paper **"Silviculture as a tool to support stability and diversity of forests under climate change: study from Krkonoše Mountains"** is focused on appropriate adaptation strategies in 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. The results showed that structural diversity and tree species richness increased when adaptation thinning had been applied, while a decrease in 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 the given habitat and stand conditions compared to the commonly performed thinning from below.

We thank all authors for their contributions and professional approach to presenting their research areas. We also believe that these interesting papers grouped in a monothematic issue will receive sound feedback from the Central European forestry community.

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REVIEW PAPER

Assisted migration vs. close-to-nature forestry: what are the prospects for tree populations under climate change?

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Abstract
Climate change is currently perceived as the most important challenge faced globally by ecosystems and human society. The predicted changes of temperature and precipitation patterns are expected to alter the environmental conditions to which forest trees in Europe are adapted, and expose them to new pests and pathogens. This would unavoidably lead to a huge loss of ecosystem services provided to society, and at the local scale may potentially endanger the very existence of forests. In this study, we reviewed biological background and limits of mechanisms by which tree populations may cope with climate change: adaptation by natural selection, gene flow, epigenetic phenomena and phenotypic plasticity, as well as forest management strategies, which rely on these mechanisms. We argue that maintaining genetic diversity is important in the long-term view but natural selection cannot ensure sufficiently rapid response to environmental change. On the other hand, epigenetic memory effects may change adaptively relevant traits within a single generation, while close-to-nature forestry practices are the basic requirement to make use of epigenetics. Assisted migration, as a frequently suggested mitigation option, relies primarily on the knowledge gained from provenance research; the review analyses potential pitfalls of this strategy. We suggest that all approaches, i.e., leaving a part of forests without management, close-to-nature forestry, and transfer of forest reproductive materials from sources presumably adapted to future climates are combined across the landscape in an integrative manner.

Key words: adaptation; epigenetics; phenotypic plasticity; provenance; climate change mitigation

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Editor: Zdeněk Vacek

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1. Introduction

Under rapidly changing environments, living organisms can choose between two strategies to survive: migration into more suitable habitats, and local persistence through adaptation (Aitken et al. 2008). The third option (if it is to be considered an option) is extinction. These potential strategies apply to tree populations as well, and the ongoing climate change makes them highly up-to-date, as the scenarios of climate development display sometimes a very dramatic picture of future environments (IPCC 2014; Ripple et al. 2019). Moreover, trees represent edificators of forest ecosystems; the future of tree populations determines the fate of a broad spectrum of forest-dwelling organisms.

Forestry practice largely mimics processes going on in natural forest ecosystems. Consequently, from the point of view of a forester, the choice of strategies in the face of climate change is in fact identical: the forester can rely on natural capacities of tree species to cope with warming and drying climate and try to enhance them through sylvicultural interventions in forest stands, or try to influence genetic composition of forest stands by the choice of forest reproductive materials (FRM) for reforestation (Konnert et al. 2015). Both strategies are currently intensively discussed. The former requires utilization of natural ecological and microevolutionary processes as applied in close-to-nature (or continuous-cover) forestry, which learns from the dynamics of natural old-growth forests (ProSilva 2012; Schütz et al. 2016). This concept is, however, not free of controversies: on one hand, it has become the main management approach in several countries such as Slovenia or Switzerland (Schütz et al. 2016), on the other hand, it was labelled as a non-concept and doubts were expressed about its applicability (O'Hara 2016). The same applies to large-scale FRM transfers: although being most often suggested by expert community as a remedy against climate change, their perception by general public is not always positive (St-Laurent et al.

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2018), and their associated risks and uncertainties are mentioned (Montwe et al. 2018; Silvestro et al. 2019).

Whatever is the personal preference of a particular forester, recent development of forest condition in Central Europe clearly demonstrated that continuing business-as-usual unavoidable results in large-scale destruction of forests, especially (but not solely) even-aged monospecific stands of conifers. In this opinion paper we review risks and prospects of both above-mentioned strategies under the view of heritable variation (genetic or epigenetic) and its use in the mitigation of climate change effects on forest ecosystems.

2. Capacity of natural populations to cope with climate change and its limits

There are several features of forest trees as a group that favour their persistence in changing environments: mostly large population sizes, high fecundity, plasticity and generally spatially extensive dispersal (Aitken et al. 2008; Anderson 2016). Only generation turnover, typically very long compared to most herbs, is a disadvantageous characteristic in this context.

In natural ecosystems without human influence, populations have used two strategies to persist in their habitats. In long term, organisms adapt to environmental variation (both spatial and temporal) through natural selection. At a short temporal scale, they respond to environmental fluctuation through phenotypic plasticity. However, nature does not know time constraints. In contrast, a forester needs to take into account not only the time scale at which both these strategies operate, but also needs to consider practical issues such as their costs in terms of the loss of forest functions, which is at stake. Maintaining broad genetic variation in order to preserve the potential for evolutionary adaptation by natural selection is a strategy requiring timeframes of many generations to be effective, which in case of trees means centuries, maybe millennia. Even if the gene targeted by selection is dominant, the between-generation change of allele frequencies is small unless selection is extremely strong (Falconer & Mackay 1996). Moreover, selection may be hindered by trade-offs between fitness-related traits (Sthultz et al. 2009; Darychuk et al. 2012), which are typically polygenic (Jermstad et al. 2001), meaning that trade-offs among loci, epistatic interactions etc. further complicate the evolutionary response to such a complex selection pressure as climate change.

Gene flow is another mechanism which makes predictions about buffering capacity of natural populations against climate change uncertain. Marker studies based on paternity analysis or model approaches (e.g., Two-Gener; Smouse 2001) yielded mean dispersal distances around tens or hundreds of meters (for overview, see Savolainen et al. 2007). However, the distribution of dispersal distances is highly skewed with a significant right tail; empirical assessment of pollen flow showed that in anemophilous species such as pines, pollen can be transferred viable over hundreds of kilometres (Lindgren et al. 1995; Kremer et al. 2012). Reproductive success of such pollen is probably not big because of differences in flowering phenology and competition with local pollen; therefore, genes are mostly transferred in a stepping-stone manner, resulting in large-scale clines of gene frequencies and the resulting phenotypic traits produced by isolation by distance (Gömöry et al. 2012a; Bošeľa et al. 2016). The same applies to pollen flow among altitudinal belts in the mountains: physical distance is mostly not a problem for pollen flow but differences in timing of flowering associated with altitudinal gradients of temperatures constrain direct gene exchange between populations at contrasting elevations (Brunet et al. 2012; Holliday et al. 2016). The effect of gene flow on average fitness of local populations, irrespective of its extent, is difficult to assess. In the case of strong local adaptation, the effect is expected to be detrimental: incoming genes may be maladaptive, and in the offspring generation breakdown of co-adapted gene complexes may occur (Charlesworth & Willis 2009; Lindtke et al. 2012). On the other hand, under unstable environments, gene flow may bring in useful genetic variants available in the species' gene pool but absent in a local population. However, the net effect is unknown and there is virtually no experimental evidence on this issue.

In spite of this, forest trees have used the strategy of adaptation by selection to cope with environmental change in their history, as evidenced by the existence of local races, ecotypes, and more generally, by intraspecific adaptive genetic variation documented in provenance trials. On the other hand, provenance trials have also documented another typical feature of tree populations, namely the adaptation lag (Mátyás 1996): adaptational optimum is seldom attained in natural populations, which shows the limits of this strategy.

Phenotypic plasticity, the alternative strategy, is a vital feature of forest trees as long-lived organisms, allowing them to cope with environmental fluctuations during their long life span. Phenotypic plasticity can be defined as the capacity of a given genotype to render different phenotypes under different environmental conditions (Valladares et al. 2006). This implies that the organism is capable to modify its developmental pathway in response to environmental stimuli. There is a wide variety of cellular and molecular mechanisms, which may potentially be responsible for the ability of a given set of genes to produce different phenotypes under different environments, including differential transcription or translation levels, allosteric switches and post-translational modifications of proteins, metabolic control, changes of signaling pathways etc. (Schlichting & Smith 2002; Nicotra et al. 2010). A part of these mechanisms is associated with chemical modification of the hereditary material (the DNA itself or protein constituents of the chromatin

– histones) or with transposon activation, expansion of short repeats etc. – all these mechanisms do not change the sense of the information contained in genes but affect the expression level of a particular gene or genes. Some of these modifications are transferred to daughter molecules during the DNA replication and, consequently, appear in the 'hereditary outfit' of the offspring: either they are associated with the change of the DNA sequence (e.g., transposon-associated changes) and hence should be classified as genetic, or they represent just chemical changes (cytosine methylation) and then constitute the basis of epigenetic inheritance.

Naturally, this definition of phenotypic plasticity leaves considerable flexibility in deciding what processes and phenomena the term comprises, both in terms of the size of phenotypic effect and the persistence of this effect during the ontogeny of an organism. A more stringent concept restricts phenotypic plasticity to developmental processes when developmental pathway is largely irreversibly set in early ontogenetic stages and ultimately results in a stable phenotype. A looser view includes any environment-induced phenotypic expression changing after the environmental stimulus disappears, or even gene regulatory processes that may have no gross phenotypic effects (DeWitt & Scheiner 2004). Whatever concept a practical forester adopts, the crucial issue is how broad the reaction norm of tree populations is. Whether the reaction of a tree species to changed environment is associated with a visible modification of the phenotype is not as important as the ability to survive and reproduce under new conditions.

While the above strategies were associated with the persistence in current habitats, tree populations may also migrate when their current sites become no more suitable for survival. Signs of such range shifts can already be observed (Woodall et al. 2009). Migration rates are the most important factor for the success of this strategy. Unfortunately, our knowledge of migration rates largely relies on the Holocene colonization, while it is not fully clear whether the speed of the Holocene spread of trees was governed primarily by the speed of Holocene warming or the dispersal capacity of trees (cf. Huntley et al. 1995). Nevertheless, the disparity between the observed migration rates and the rate of displacement of tree habitats under the ongoing climate change is considerable. Feurdean et al. (2013) estimated maximum migration rates for early-successional species such as birch or Scots pine to 225 – 540 m yr^{-1}, while for climax trees such as silver fir or beech they range between 115 and 385 m yr^{-1} (of course, depending also on the mode of dispersal, animal or wind). In fact, in a fragmented landscape and under a strong ungulate pressure the recent colonization rates may even be considerably smaller. In contrast, even moderate climate change scenarios predict latitudinal (i.e. horizontal) shift of the isotherm by 3 km yr^{-1} (Mátyás 2007); obviously, tree migration cannot keep pace with the speed of habitat changes. Moreover, postglacial colonization occurred in non-disturbed landscapes. Range fragmentation typical for most of Europe substantially reduces the migration speed, as both the size of source populations producing propagules and availability of target sites colonized by these propagules are reduced (Iverson et al. 2004). In mountainous countries such as Slovakia, evasion into higher elevations may be an additional option. However, the speed of the vertical shift of the isotherm (estimated at 11.5 m yr^{-1}; cf. Mátyás 2007) may be prohibitive for altitudinal migration for several tree species as well.

3. What are the options for forestry practice?

Although maintaining broad genetic variation in tree populations remains an important task for practical forestry, migration to new sites or selection of appropriate genetic variants seem to be too slow processes to allow adaptation to rapid climate change in organisms having as long generation turnover as trees. In contrast, heritable phenotypic change through chemical modification of the hereditary material resulting in changed gene expression patterns can appear very rapidly. Vegetative phenology is a good example of an important adaptive trait whose control may be at least partly epigenetic. In European beech (*Fagus sylvatica*), phenological responses to climate reflect local adaptation but at the same time, significant genotype-by-environment interactions indicate that they are phenotypically plastic (Kramer et al. 2017). In conifers, timing of spring flushing and autumn growth cessation, and consequently frost hardiness are largely determined by day length and temperature during seed development (Skrøppa 1994; Johnsen et al. 2005). Epigenetic carryover effect may cause changes of phenology from the parental generation to the offspring even for provenances transferred over several degrees of latitude: in Norway, budset dates of the offspring of Norway spruce (*Picea abies*) introduced from Germany and Austria a century ago were shown to resemble more the local Norwegian families than families newly imported from Central Europe (Skrøppa et al. 2009). Such newly-acquired expression pattern may remain stable for further generations (genetic assimilation; cf. Pigliucci et al. 2006). A similar effect may be induced by the climate of the site of juvenile development (Gömöry et al. 2015). In terms of climate change, such behaviour is promising, as it means that physiological adaptation does not necessarily require genetic change, and generation turnover makes the population fit to the changed climate. On the other hand, epigenetic memory was proved in forest trees only in association with phenology and frost resistance; its importance related to growth, drought tolerance and other climate-relevant traits has not been explicitly confirmed. Observations in provenance experiments, where identical provenances are planted on climatically contrasting sites, indicate that phenotypic plasticity in drought-related traits such as resistance to xylem embolism is extensive and induced by the local climate (Wortemann et al. 2011); this is not a proof but still an indication for a potential role of epigenetics in drought resistance. The use of the epigenetic memory to make the future tree generation accommodated to new climate requires, however, that both reproduction and juvenile growth must occur at the site where the new climate appears, under as natural conditions as possible. This implies that the developing seeds and seedlings do not leave the site even temporarily, so it is not practically feasible with reforestation. Even when the offspring is planted back in the forest stand where the seeds were collected, the nursery for raising the plants is always placed elsewhere. Of course, leaving the reproduction completely to natural processes (as it is done in nature reserves or other types of wilderness) is an option, but not for commercial or managed forests, where timber production is one of the expected benefits. Active sylvicultural management does not contradict natural regeneration; nevertheless, not all sylvicultural systems are equally suitable in terms of potential epigenetic effects. Approaches resulting in synchronized natural regeneration on large plots such as uniform shelterwood system or strip fellings performed on broad strips lead to quite unnatural environments for developing seedlings just during juvenile stages of ontogeny (Matthews 1989; Puettmann et al. 2009). Of course, macroclimate fits even in this case; the question is to what extent microclimate of a large open area may differ from that of a closed stands and thus adversely affect the offspring generation.

In addition to epigenetics, sylviculture also affects genetics: sylvicultural measures may enhance as well as hamper evolutionary processes such as gene flow and natural selection by affecting simultaneously species demography and local environmental conditions (Finkeldey & Ziehe 2004). Even though a sufficiently rapid adaptation to climate change through natural selection of appropriate genetic variants is not realistic, maintaining or increasing genetic diversity as a basis for the stability of tree populations in response to environmental fluctuations remains important goal of sylvicultural management in a long-term view (Lefèvre et al. 2014).

Assisted migration is another option to accommodate forest tree populations to changed climate, currently broadly advocated by many forest geneticists and breeders (Mátyás 1994; Aitken & Whitlock 2013; Williams & Dumroese 2013; Konnert et al. 2015; Aitken & Bemmels 2016). Assisted migration includes assisted gene flow, i.e., intentional translocation of individuals (in any stage of ontogeny) within the natural range of a species, and assisted colonization, i.e., translocation of individuals outside the range, which implies a change of species composition of forest stands at the target site (Aitken & Whitlock 2013). Both approaches involve the transfer of FRM (in the case of assisted gene flow also movement of pollen can be considered, but is rarely practiced except in the form of supplemental pollination in seed orchards). While assisted gene flow relies on non-local provenances of native tree species, assisted colonization also includes the use of non-native tree species.

The choice of appropriate provenances is a crucial aspect of the FRM-transfer strategy. In spite of a rapid development in the field forest genomics during the last two decades (Neale & Kremer 2011; Plomion et al. 2016), the application of the results of genomic studies in setting the FRM transfer rules is still associated with many caveats (cf. Lind et al. 2018). Until physiological mechanisms underlying phenotypic traits relevant in the climate-change context are known and their controlling genes are identified (while genetic dissection of phenotypic traits in general is still in a very initial phase), practical applicability of genomics is very limited. In practice, the current reflections about provenance choice under climate change are based on climate matching: the search for provenances growing currently in climates, which are expected to occur at the reforestation site in the future. The climate envelope approach, traditionally applied in assessment of species climate sensitivity (Bolte et al. 2009) can in principle be applied also at the population level.

Consequently, the most reliable information source for guiding assisted migration constitute common-garden experiments, where the performance of different populations under particular climates is explicitly tested. The importance of provenance trials for guiding the FRM transfer was recognized early (Mátyás 1994). Provenance recommendations have sometimes been based on anecdotal experience with the performance of particular provenances in simple small-scale trials. However, nowadays range-wide experiments comprising tens of provenances replicated across several climatically diverse test sites are available for most commercially or ecologically important tree species. Such trials can be used in two ways. A specific way is determining the environment where a particular provenance performs best or choosing the optimum provenance for a particular site. A conceptually broader way is the identification of patterns of climate-related responses, which allows setting generalized rules for provenance transfer. Two primary approaches have been developed for this purpose: general transfer functions and population response functions (Rehfeldt et al. 1999; Aitken et al. 2008). From the point of view of methodology, both approaches relate some fitness-related trait (typically survival, height growth or biomass production) of the translocated population to translocation rate, i.e., the climatic (or geographic) distance between the site of origin and the site of plantation ('ecological distance' or 'ecodistance'; cf. Mátyás 1994), and look for the climatic optimum, i.e. the rate of transfer at which the whole provenance set often combined across different experiments (general transfer function) or a particular provenance (population response function) performs best. These approaches allow provenance recommendations also under climate change, provided a reliable climate prediction for a particular planting site is available.

Even though the advantages of common-garden experiments for the assessment of future behavior of tree populations under climate change are obvious, there are still several problematic aspects of this approach (see Konnert et al. 2015). One is associated with the methodology of provenance research. There are no universal guidelines for the collection of seedlots representing provenances, which implies that the material used does not necessarily properly represent the genetic setup of the maternal stand (too few maternal trees, unbalanced family sizes, seed collection during a poor-crop year). Further, plants are typically raised together in a nursery. Optimized microclimatic and soil conditions as well as other nursery practices reduce the playground for natural selection, and cause epigenetic imprinting (Gömöry et al. 2015) which may influence the performance of provenances during later ontogenetic stages. The same applies to outplanting: typically, wide spacing, weed control and fencing against deer are used, which all may distort the assessment of survival rates as an important fitness measure. The selection of provenances and planting sites is not necessarily random: often provenances performing well in older trials or conspicuous by other relevant properties are systematically offered into international experiments (e.g., Polish well-growing Norway spruce provenance Istebna or Slovenian late-flushing beech provenance Idrija), and ecologically marginal planting sites are avoided. Another set of problems is associated with the measurement of trials. Most studies are based on trial assessments at juvenile age. The measurements of height growth as an 'obligatory' parameter gradually become technically complicated with increasing age, not to mention physiological traits such as gas exchange or photosynthesis. Studies at the age of 40+ years exist but are rare (e.g., Gömöry et al. 2012b); however, even in this case it remains questionable whether they can be extrapolated to the age of stand rotation. Finally, modelling of tree–climate relationship requires relevant and reliably assessed climatic characteristics of both plantation sites and sites of origin. Currently, climate variables at fine resolution can easily be acquired from public sources such as WorldClim, ClimateEU or EuroCordex (Fick and Hijmans 2017; http://tinyurl.com/ClimateEU; https://www.euro-cordex.net/). Nevertheless, as interpolated data they are by principle loaded by a certain error. There is also a certain dilemma about the choice of appropriate climate proxies: on one hand, general climatic variables such as mean annual temperatures or annual precipitation totals may poorly reflect actual driving forces of climatic adaptation; on the other hand, more sophisticated climatic indices may be difficult to be calculated and applied in practice except in computer-based decision-support systems. Moreover, provenance choice for assisted migration requires a reliable prediction of future climates. Even if there was a general consensus on mathematical modeling of the development of climate, a realistic climatic scenario would require knowledge about greenhouse gas emissions, which depends on future decisions of the human society.

In any case, obligatory permanent keeping records of the origin of any FRM used for reforestation is a basic prerequisite of further assessment of the effectiveness of assisted migration. The current practice of abandoning records after the expiration of forest management plans does not allow evaluation of experience with the performance of the artificially established forest stands.

4. Conclusion

As explained above, all strategies of mitigation of climate change effects are loaded by uncertainties and risks of their own. In such a situation risk diversification is the only reasonable solution. Economic theory advises not to keep all assets in the same pocket. Biology should advise to do the same, and combine all approaches (Messier et al. 2019). First, 'mother nature' should be left to do its job in a part of forests, i.e., a reasonably large part of forests needs to be strictly excluded from any active management; mainly to leave space for natural forces to act, but also to preserve a possibility to learn from natural processes. Of course, what is 'reasonably large part' may be a matter of discussion; for sure it should comprise all current forest reserves and all existing old-growth fragments, even when they are not yet under legal protection, and a part of managed forests in large-scale protected areas such as national parks can also be converted into wilderness. Second, sylvicultural treatments should vary across the landscape to create alternating areas of structurally diverse forest stands with long regeneration periods, differentiated by age, and areas with shorter rotation periods with lower disturbance risks and allowing for faster species turnover (Brang et al. 2014; Spathelf et al. 2015). Spatially varying cutting regimes contribute to the formation of variable ecological niches and impose variable selective pressures on trees; consequently, foster genetic diversity (Fady et al. 2016). Naturally, practical application must conform to the condition of each particular forest stand. Continuous cover forestry cannot be applied everywhere, but for sure it can be applied at much larger scale than currently. Third, the knowledge gained from translocation experiments must be applied in forestry practice in spite of all uncertainties, which means transfer of FRM from warmer and drier regions. In a mountainous country such as Slovakia this may not necessarily mean import of FRM but also altitudinal transfer. In any case, assisted migration should primarily focus on transfer of provenances of native species rather than introduction of non-native species. Again, risk reduction implies the use of multiple provenances and species (Konnert et al. 2015). Forestry legislation needs to be adapted to allow these measures. In Slovakia, the first step was accomplished by adopting a new Forest Code no. 355/2019, which introduced the concept of close-to-nature forestry. Now, it is time to amend the legislation on forest reproductive material, in Slovakia as well as elsewhere in Central Europe, to allow FRM transfer on larger scales.

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ORIGINAL PAPER

Substitution of Norway spruce for Douglas-fir: changes of soil microbial activities as climate change induced shift in species composition – a case study

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Abstract

Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco) is the most common utilized introduced tree species in the temperate zone, planted in many European countries. Also, in the Czech Republic it represents the most planted exotic species as well. Its planted area exceeds 6,000 ha and represents 0.25% of the forest area of the country. In the last years, this species is supposed as convenient substitution for declining Norway spruce (*Picea abies* [L.] Karst.) stands under current climatic change (extreme drought, bark beetle outbreak). The changed species composition affects also soil dynamic processes, including litter decomposition and transformation. In the presented study, the respiration rate and nitrification intensity were compared in the stand parts dominated by Norway spruce and Douglas-fir in comparable site conditions (410 m a.s.l., 650 mm, 8 °C, gleyic Luvisol, fresh oak-fir site with *Oxalis acetosella,* age 97 years). Analyses were performed in laboratory conditions by standard methods determining actual (basal) and potential respiration activity and intensity of ammonia and nitrate ion production. Admixture of Douglas-fir considerably intensified decomposition activities (soil respiration) and profoundly increased the nitrification rate in the surface humus and uppermost mineral soil layer – Ah horizon. Increased interest in the Douglas-fir silviculture needs increased care for forest soil dynamics with respect of nitrogen balance.

Key words: forest soils; soil microbiology; soil respiration; ammonification; nitrification

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1. Introduction

Soil microbial activity is an important indicator of the soil organic matter dynamics. Its role in the site quality indication has been recognized since early history of the modern soil science; it indicates changes in soil dynamics more evidently and early, compared to soil chemical characteristics (Seifert 1957). Among others, the activity of soil micro-organisms results in the formation of differentiated humus forms (Green et al. 1993; Ponge 2003). Decomposition and transformation processes are supported especially by the quality and amount of the litter (above- as well as below-ground) and by root exsudates, determining the composition and activity of soil organism communities (Prescott & Grayston 2013). The conditions for organic matter dynamics are affected particularly by the species composition of the forest ecosystems and by their age and spatial structure (Augusto et al. 2002). Even the natural disturbances and silvicultural treatments influence the microbial activities of the soils (Svoboda & Podrázský 2005).

Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco) is a species with high commercial importance in the World as well as in the Europe, widely used in the forest management of many countries. On the contrary, in the Czech Republic it is cultivated only on area, exceeding slightly 6,000 ha, which represents approximately 0.25% of the stand area (Beran 2018), despite increasing interest of the forestry practice. Its potential is much higher in the country on the other hand. Only its proper use as the site improving and stabilizing tree species accordingly to recent legislation could result in an area increase to some 5.7 – 6.2% of the forested lands, which accounts for 149,616 to 163,713 ha and 25.3 – 28.6 mil. EUR yr^{-1} of value increment (Pulkrab et al. 2014). In this study mentioned, the cultivation of Douglas-fir in monocultures, prevailing in the Western Europe, is excluded. The production potential of Douglas-fir was documented substantially to consider its use in the future, also its environmental effects were partly analyzed; for summary see e.g. Martiník (2003), Kubeček et al. (2014) and

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Mondek & Baláš (2019), in other countries e.g. Augusto et al. (2002), Budde (2006) and Riedl et al. (2019). The selection of proper provenances is of great importance (Beran 1993, 1995; Eilmann et al. 2013; Petkova et al. 2014; Popov 2014; Kšír et al. 2015, 2019), as well as the corresponding seed material management (Martiník & Palátová 2012; Kuneš et al. 2017). Also, the natural regeneration of this species is very successful in convenient conditions (Bušina 2007; Kantor et al. 2010). On the other hand, Douglas-fir is suitable for afforestation of agricultural landscapes (Cukor et al. 2017a).

Douglas-fir has been planted at the territory of the School Training Forest Kostelec nad Černými lesy since 1880, occupying the area around 10.5 ha (Hart et al. 2010); its production was analyzed here in some cases (Tauchman et al. 2010). On the contrary, some publications indicate certain risk for the nitrogen dynamics in forest ecosystems. Douglas-fir litter dynamics resulted in some cases in the increased nitrogen mineralization, especially nitrification, and the danger of nitrate leaching can be expected (e.g. Zeller et al. 2010). Despite this indication, Douglas-fir is supposed to be a very convenient substitution for Norway spruce at lower altitudes (Temperli et al. 2012, Podrázský et al. 2012), especially under ongoing climatic change (Allen et al. 2010; Vacek et al. 2016; Šimůnek et. al. 2019) causing large-scale spruce decline in Czech Republic (Vacek et al. 2019a; Toth et al. 2020). Douglas-fir is more tolerant to extreme droughts than Norway spruce (Vitali et al. 2017). Moreover, spruce sensitivity to extreme climatic events was also documented in the last decades in others studies compared to other tree species (Králíček et al. 2017; Vacek et al. 2019c; Vančura et al. 2020). In relation to climatic change, Douglas-fir plays important role in carbon sequestration in soils (Antisari et al. 2015; Cukor et al. 2017b).

Aim of the presented study is the comparison of the respiration, ammonification and nitrification activities under Douglas-fir and Norway spruce at comparable site. Two hypotheses were postulated:

1) Douglas-fir is capable of producing litter, which is more easily decomposable compared to Norway spruce; it is supposed to be indicated by higher respiration activity of soil microbes in the upper soil layers,
2) Humus forms developed under Douglas-fir dominance show more intense nitrogen dynamics, manifesting itself especially with high nitrification potential.

2. Material and methods

The study area is typical for relatively large areas of the Central Bohemia region, the altitude ranges between 350 – 450 m, the average annual temperature 7.5 – 8.5 °C, precipitation 550 – 650 mm. The study site is located in a stand on the territory of the School Training Forest Kostelec nad Černými lesy. Sites are acid to fresh, soils are Luvisols and Luvic Cambisols, 3rd – 4th vegetation altitudinal zone. In the studied stand (410 m a.s.l., 650 mm, 8 °C, gleyic Luvisol, Forest Site group 4O1 – fresh oak-fir site with *Oxalis acetosella* – Viewegh (2003), age 97 years) the standing volume accounted for 830 – 1 030 m^3 per ha, depending on the Douglas fir share (14 – 30% Nr. of trees, 32.4 – 42.4% of basal area, 36.6 – 58.3% of stand volume). Natural regeneration occurs after chemical weed control reaching 16,000 – 31,000 trees per ha. Seedlings disappear at older age in the relatively dark stand (Hart et al. 2010; Remeš et al. 2010).

Microbiological soil characteristics were studied in the same stand parts, where the soil chemistry was evaluated before (see Ulbrichová et al. 2014). Soil samples were taken Nov 27 2011 and processed using standard methods (see Svoboda & Podrázský 2005). Soils were sampled in pure larger stand parts with dominance of both Norway spruce and Douglas-fir. Under each species, two bulk samples from F_2+H and Ah horizons were taken, each bulk sample from 5 particular sites (Table 1). From each bulk sample 4 subsamples were separated for any analysis in the laboratory described below.

Table 1. Bulk samples taken in the stands of Douglas-fir and Norway spruce.

Species Horizon	Douglas-fir	Norway spruce	Douglas-fir	Norway spruce
F_2+H	G1	G3	G5	G7
Ah	G2	G4	G6	G8

Analyses were done in the Laboratory Tomáš, located at FGMRI, Research Station Opočno. In particular samples the following analyses were performed:

Biological activity actual: from each bulk sample, 4 subsamples of 50 g of air-dried soil (mineral horizon) or 25 g (surface humus horizon) were placed in 1 liter flask, watered by determined amount of distilled water. In the flask, a cup with 25 ml of 0.1 N NaOH was located, capturing respired CO_2. In the next 5 days, after each 24 h period the diluted CO_2 was analyzed by titration.

Biological activity potential: half amount of both substrates was sprayed with 12.5% glucose solution in the amount of 4 ml for 50 g of substrate. Incubation is 24 h at 25 °C. In both cases, the amount of captured CO_2 is determined by titration.

Nitrification: the weight of substrate samples is 50 g (mineral soil) or 25 g (humus horizon), moisturized by distilled water and incubated in the thermostatic chamber 14 days at 25 °C. The space inside was saturated by vapor all the time of incubation. The same amount of substrates was stored in the freezer for control measurement. The incubated samples as well as the control sample are leached by potassium sulphate, filtered, and the ammonia and nitrate nitrogen is determined after that. The ammonia ions are determined in the solution of sulphuric acid colorimetrically with Nessler´s solution, the nitrate nitrogen is determined directly by nitrate-electrode.

The SW Statistica 10.0 (Statsoft, Tulsa) was used for data processing, the non-parametric test Kolmogorov-Smirnov was applied for data analysis.

3. Results

Results indicated highly significant differences in the soil respiration activity, both actual and potential (Table 2). The substrates from the Douglas-fir stand parts show significantly faster decomposition and the microbial communities are much more active in F_2+H and Ah horizons compared to Norway spruce.

Table 2. Respiration activity actual (AA) and potential (PA) in the stand of Douglas-fir and Norway spruce in the horizons F_2+H and Ah (mg CO_2 / 100 g dry soil per 24 h).

Species	Douglas-fir	Norway spruce	Douglas-fir	Norway spruce
Horizon	F_2+H		Ah	
AA 1.day	199.0720 a	95.8345 b	22.5567 a	6.07858 b
AA 2.day	162.3819 a	80.7918 b	14.5320 a	2.86152 b
AA 3.day	140.1887 a	62.8726 b	15.6585 a	4.52673 b
AA 4.day	137.0985 a	57.4448 b	13.2878 a	3.13458 b
AA 5.day	153.4415 a	76.3180 b	12.8356 a	8.43414 a
PA 1.day	541.1018 a	271.4150 b	108.4001 a	29.91750 b

Notes: Different indexes indicate statistically significant differences at ($p<0.001$) between corresponding horizons; AA – actual respiration activity, PA – potential respiration activity.

Similarly, the nitrogen dynamics is more intense under Douglas-fir, compared to Norway spruce (Table 3). The soil moisture values were comparable in both stand parts at the time of sampling. In the hologranic layer, the contents of soil ammonia cations were lower in the stand parts under Douglas-fir before incubation, this difference was not significant. Contents of the nitrate anions were comparable. Both nitrogen forms showed lower content values in organomineral horizons (Ah) under spruce before incubation, it represented roughly one-half in the case of ammonia nitrogen, one-tenth in the case of the nitrate one.

These differences changed after incubation. Under spruce, the increased ammonification intensity led to increase of ammonia cations in both studied horizons. In addition, nitrification was supported, the contents of nitrate nitrogen was cca doubled after the period of incubation.

Different situation was documented under Douglas-fir stand parts. The content of ammonia cations decreased cca by 50 % in both studied horizons, on the contrary, the content of nitrate nitrogen multiplied during the incubation. The differences in the content of nitrogen forms under particular tree species after incubation are shown in the Fig. 1 – 2. The case study confirmed high potential of Douglas-fir compared to Norway spruce to support soil microbial activity resulting in potential carbon and nitrogen mineralization.

4. Discussion

The obtained results confirmed both hypotheses formulated, i.e. significant increase of respiration as well as of nitrification activity in the humus forms under Douglas-fir after incubation. Faster litter decomposition, mineralization and higher activity of microbial communities can be supposed due to presence of Douglas-fir compared to Norway spruce.

The microbial communities are determined especially by environmental conditions (temperature, soil moisture, geology - bedrock) primarily, secondary effects were detected for tree species (litter), as documented by Augusto et al. (2002) and Brockett et al. (2012). The

Table 3. Soil nitrogen dynamics in the soil substrates under Douglas-fir and under Norway spruce.

Species Horizon		Soil moisture	N–NH_4^+ before incubation	N–NH_4^+ after incubation	N–NO_3^- before incubation	N–NO_3^- after incubation
		%		mg kg^{-1} D.M.		
D.–fir	F_2+H	45	82.8	37.4 a	24.6	101.0 a
D.–fir	Ah	18	43.4	28.1 a	11.4	29.0 a
N. spruce	F_2+H	42	101.2	134.0 b	21.1	47.0 b
N. spruce	Ah	16	26.0	46.4 a	1.2	3.3 b

Notes: Different indexes indicate statistically significant differences at ($p < 0.001$) between corresponding horizons.

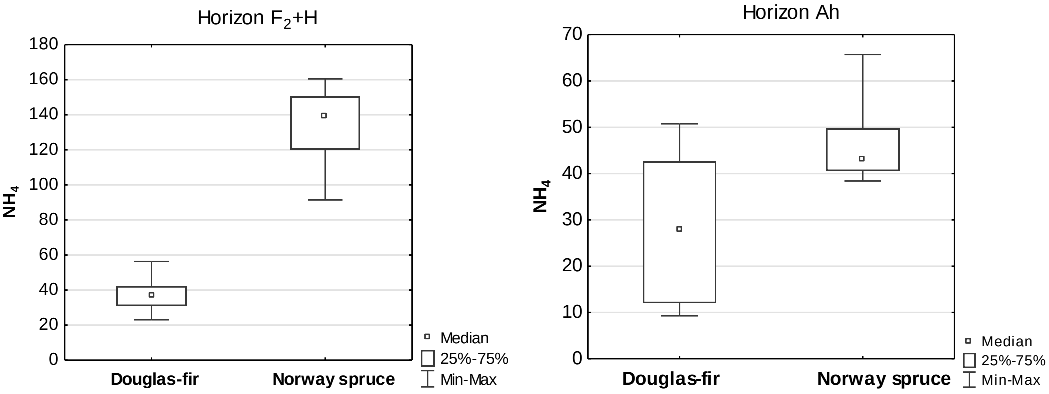

Fig. 1. Contents of ammonia nitrogen in the horizon F_2+H (left) and Ah (right) under Douglas-fir and Norway spruce after incubation (mg kg^{-1}).

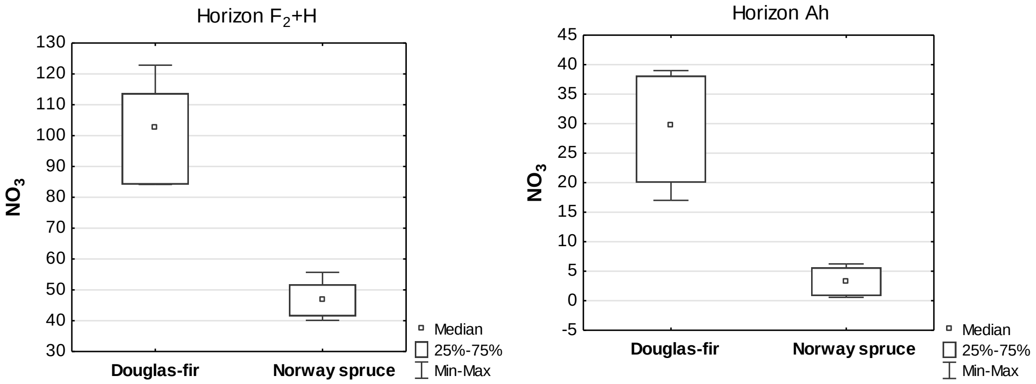

Fig. 2. Contents of nitrate nitrogen in the horizon F_2+H (left) and Ah (right) under Douglas-fir and Norway spruce after incubation (mg kg^{-1}).

latter authors detected also higher N–availability in the Douglas-fir dominated tree communities. In our case, environmental conditions determined by local microclimate were the same and the soil moisture was similar too (Table 3). The differences in the soil characteristics were, therefore, a result of different tree species (litter) effects dominantly.

Changes of tree species composition of forest ecosystems offer rapid modification of the soil organic matter turnover and SOM quality (Antisari et al. 2015). These authors described changes in the carbon and nutrient cycling as well as in the microbial activity as they compared native European beech with Douglas-fir stands growing at higher elevations (1 000 – 1 300 m a.s.l.) of the Tuscanian Apeninnes. Antisari et al (2015) concluded also higher basal respiration in the upper mineral soil under Douglas-fir compared to beech, whereas slower C–mineralization and C–turnover were shown on the other side, resulting in higher C–accumulation under Douglas-fir. They ascribed this fact to the more recalcitrant and hydrophobic composition than that of the epipedon under beech. Less favorable humus forms under Douglas-fir compared to beech reported Martiník (2003) and Menšík et al. (2009). Similarly, increasing C–accumulation and soil stabilization with ageing (80 – 100 years) of Douglas-fir described in the same Italian region Antisari et al. (2018). In older stands of this species, lower availability of nitrogen can be observed due to increasing C/N ratio. From this we can concluded, that long term N–mobilisation due to increased N–mineralisation and fixation in biomass can lead to N–shortage in the ecosystem.

Prietzel & Bachmann (2012) studied the changes in humus form status and sequestration of SOM (OC – organic carbon) and N in the set of stands, documenting shift from Norway spruce and Scots pine towards Douglas-fir and European beech forests. They also confirmed in most cases significant decrease of surface humus C/N ratio in the order spruce, pine > mixture of spruce or pine with Douglas-fir > Douglas-fir > beech. Forest floor OC and N stocks decreased in the same order. Trends in the mineral soil were opposite, as for OC stock and especially in the mineral N amount. Mixture of Douglas-fir and spruce was documented as an effectively increasing C and N sequestration; the benefits of mixtures are also in the CO_2 fixation. Mixed forest stands are a very good tool for forest stability enhancement (Mondek et al. 2019) and for high quality timber supply (Palátová et al. 2017; Riedl et al. 2019). Higher nitrification under Douglas-fir compared to Norway spruce documented also Malchair & Carnol (2009). On the contrary, humus form microbial activity was lower in Douglas-fir compared to broadleaved (birch) stands (Welke & Hope (2005). The proper stand composition and management is a vital prerequisite for sufficient stand nutrition. Mixing of tree species is important also in the case of Douglas-fir absence in the stand mixture in relation to higher diversity, stability and also production compared to monospecific forest stands (Pretszch et. al. 2010; Vacek et al. 2019b; Mikulenka et al. 2020; Prokůpková 2020). This is much more important while this species is included (Šrámek et al. 2019). Increased interest in silviculture of Douglas-fir has to be connected with establishing of site corresponding species mixture. Under these conditions, Douglas-fir can be a proper substitution for site-degrading conifers in the period of climate change (Mondek & Baláš 2019).

5. Conclusions

The results confirmed differences in the litter decomposition and mineralization activities between stand parts dominated by Norway spruce and Douglas-fir. It can be supposed, that admixture of Douglas-fir will accelerate and promote microbial activities, resulting in faster litter decomposition (increased potential and actual respiration activity) and more pronounced nitrogen dynamics (nitrification). These trends can explain lower surface humus amount and its more favorable pedochemical characteristics. Proper use of this knowledge demands cultivation of Douglas-fir in site-corresponding mixtures of tree species. Substitution of coniferous stands with site-corresponding mixtures is supposed to improve forest stability and vitality in the changing environmental conditions.

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ORIGINAL PAPER

Litterfall nutrient return in thinned young stands with Douglas fir

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Abstract

The use of Douglas fir (DF) is on the increase in the Czech Republic. This tree species shows a good production and also a beneficial impact on soil to some extent. We studied both amount and properties of litterfall in 18 to 20-year-old stands with DF in 2011. Two experimental plots were DF-dominated and one showed ca. 20 – 30% share of Scots pine (SP) at the beginning of observation. The experiments consisted of two treatments such as unthinned control and 50 – 62% trees' density reduction accounting for 43 – 59% basal area reduction. Litterfall was collected using litter traps of 0.25 m^2 area in 3 – 4 traps per treatment. Forest floor L and F were taken in 2011 and 2018 to investigate the development of their amount after thinning. Both whole-period sum and mean annual litterfall were reduced after thinning compared to controls. The mean annual litterfall totaled ca. 3 t ha^{-1} in 20-year-old DF–dominated stands. This amount represents an annual nutrient return of 30 – 40 kg N, 1 – 3 kg P, 3 – 5 kg K, 12 – 30 kg Ca and 1 – 2 kg Mg per one hectare. The reduction of the annual litterfall was more pronounced in DF–dominated stands. All thinned plots showed increased decomposition rates reflected in lower total L+F amounts in both DF–dominated plots whereas unthinned plots accumulated more L+F at the end. The mixed DF–SP plot showed reduced L and increased F layer amounts in both unthinned and thinned treatments with only minor change to L+F sum between 2011 and 2018.

Key words: litterfall; forest floor; thinning; nutrients; Douglas fir

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1. Introduction

Forest ecosystems face both climate changes and various levels of anthropogenic disturbance. At many sites, Norway spruce stands exhibit a die-off due to lack of precipitation and following bark beetle outbreak. Foresters are, therefore, forced to find new silviculture approaches frequently; for example tree species conversion. At many sites, domesticated non-native trees such as Douglas fir (hereinafter referred to as DF), which have proved their capabilities, are beneficial. Although DF has been connected with central European forestry for more than hundred years, only several thousand hectares are managed (0.28% of current species composition) in the Czech Republic. This situation is likely to be attributable to fear of wider use of non-native tree species. However, the risk of wrong decision is minimized by intensive research and long-term verification of recommended silvicultural measures in forest practice in the last decades.

Besides its outstanding production (Kantor et al. 2001; Podrázský et al. 2013), also soil improving effects of litterfall were observed (Thomas & Prescott 2000; Podrázský & Remeš 2005, 2008; Podrázský & Kupka 2011; Ulbrichová et al. 2014) compared to other conifers. Although some information on accumulation and decomposition of DF litterfall have been published (Menšík et al. 2009; Podrázský et al. 2009), long-term investigation into year-by-year amounts and quality of litterfall including its relation to thinning were needed. In the Czech Republic, the use of DF is a rising issue which is reflected in number of projects (Kubeček et al. 2014; Slodičák et al. 2014) and resulted in establishment of many experimental plots. Thinning improves stands' stability (e.g. Settineri et al. 2018) and reduces stress resulting from overstocking (Chase et al. 2016) which can lead to a greater mortality during the decline events such as drought (Livingston & Kenefic 2018). Thinning can also help improve microbial conditions in organic horizons such as L and F (Wang et al. 2019).

The objectives of this study was to monitor litterfall amounts changes after thinning and analyze its nutrient contents in order to estimate an impact of thinning on nutrient return over longer time compared to previously published studies.

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2. Material and methods

Three thinning experiments with DF were established in spring 2011 (Table 1) in 18 to 20-year-old stands. DF dominated in the species compositions totally (Obo) or other species were admixed – mainly Norway spruce or Scots pine in the experiment Pol1 or Pol2, respectively. Stands were established by artificial (Obo) and natural (Pol1 and Pol2) regeneration. All sites are located in the Eastern part of Bohemia and belong to Colloredo-Mansfeld family estate. Mean annual temperature is 9.6 °C and mean annual amount of precipitation is 520 mm in the region of interest.

Each experiment consisted of two plots (with area of 0.04 ha), one was control unthinned stand and second one was managed using a very heavy thinning from below (Fig. 1). Thinning intensity was 62% on the basis of number of trees (N) and 59% on the basis of removed basal area (G) in Pol1. Two thinnings were done in Pol2. First at the start of observation period (50% of N and 49% of G removed) and second was realized 7 years later at the age of 24 years (24% of N and 43% of G removed). On the experiment Obo, 57% of N representing 43% of G was removed by first thinning at the age of 20 years.

Litterfall was collected using 3 – 4 litter traps (with an area of 0.25 m^2 each) per partial plots installed within stands in February (Obo) and October (Pol1, Pol2) 2011. The samples were taken twice to fourth times per year until October 2018.

At both the beginning (autumn 2011) and the end (autumn 2018) of observation period, forest-floor humus horizons (data from L = fresh litter and F = fermented layer were analyzed in this study) were investigated quantitatively in comparative plots (C – control and T – thinned) of all experiments. The samples were taken using steel frames (25 × 25 cm) to demarcate an area for collecting all enclosed material. Comparative plots and also observed stands are relatively small and homogenous. Nevertheless, the number of samples was gradually increased (three samples per plot in 2011 and six samples per plot in 2018).

All samples were dried, first under conditions of open air, later in a laboratory oven at 80 °C, and dry samples were subsequently weighed. Nutrient content in litterfall was assessed from composite samples from each comparative plot (after mineralization by mineral acids). Total nitrogen concentration was analyzed using Kjehldahl procedure and phosphorus concentration was determined colorimetrically. An atomic absorption spectrophotometer was used to determine total potassium concentration by flame emission, and calcium and magnesium by atomic absorption after addition of lanthanum. Data were analyzed using a descriptive statistics and we use paired t-test for total litterfall samples from three control and thinned plots.

Table 1. Characteristics of experimental plots at the start (2011) and at the end (2018) of litterfall observation by number of trees (N) and by basal area (G).

Plot	Abbreviation	Coordinates	Elevation slope, aspect	Soil	Site*	Year	Age [years]	Species composition [%] by N						Species composition [%] by G					
								DF	NS	SP	OK	BI	EL	DF	NS	SP	OK	BI	EL
Polanky 1 – Control	Pol1 C	50.205924 16.028731	260 m 10° WSW	dystric cambisol on gravel-sand sediment	*Fageto – Quercetum cidophilum*	2011	18	80	18	1	0.5	0.5		82	14	1	2	1	
						2018	25	80	17	1.5	0.8	0.7		77	18	2	2	1	
Polanky 1 – Thinned	Pol1 T					2011	18	88	9	3				96	4	0			
						2018	25	89	11					95	5				
Polanky 2 – Control	Pol2 C	50.20615 16.028933	265 m 10° WSW	dystric cambisol on gravel-sand sediment	*Fageto – Quercetum acidophilum*	2011	18	74	3	23				55	3	42			
						2018	25	66	3	31				32	3	65			
Polanky 2 – Thinned	Pol2 T					2011	18	51	2	47				35	1	64			
						2018	25	55	3	42				35	1	64			
Obora – Control	Obo C	50.24797 16.119434	345 m 6° WNW	dystric cambisol on sandy marlstones to spongilitic, sporadically silicificated claystones	*Querceto – Fagetum illimerosum trophicum*	2011	20	89	11					98	2				
						2018	27	100						100					
Obora – Thinned	Obo T					2011	20	94					4	96					4
						2018	27	97					3	98					2

*according to Viewegh et al. (2003), Tree species codes: DF – Douglas fir, NS – Norway spruce, SP – Scots pine, BI – silver birch, OK – sessile oak, EL – European larch (see Jenkins et al. 2012).

3. Results

Thinning regimes (Fig. 1) led to lower amounts of annual litterfall in DF-dominated plots Pol1 and Obo compared to unthinned control (Fig. 2). As for the whole-period, total sums of litterfall were also reduced between 2011 – 2018; the thinned treatment values were found to be close to those of control values in mixed DF–SP plot Pol2 (Table 2). The reduction amounted to 1.2 t (5%) whereas both DF–dominated plots showed even less dry mass amounting 3.9 (8%) and 6.0 t (10%) in Pol1 and Obo plots respectively. Mean annual litterfall was 170 kg lower in Pol2, 563 kg in Pol1 and 773 kg in Obo compared to their unthinned treatments. Differences (long-term lower amount of litterfall under thinned stands compared control stands) were 15% (p–value 0.10).

The less dry mass return the lower nutrient return. The only slightly higher N and P return in thinned Pol2 (Fig. 3) was found due to slightly higher, though insignificantly, concentrations of these nutrients. As for the litterfall nutrient concentrations, N oscillated between 0.8 – 1.3% -, Ca 0.4 – 0.8% -, P 0.04 – 0.10%, K 0.09 – 0.16% and Mg 0.04 to 0.06% which accounted for an annual litterfall return amounting 30 – 50 kg of N, 2 – 3 kg of P, 3 – 5 kg of K, 12 – 30 kg of Ca and 1 – 2 kg of Mg per one hectare of young DF–dominated young stands (Table 2).

We also observed, that litterfall turnover under young DF stands was relatively fast (Fig. 4). Although litterfall amounted 20 – 30 t per hectare (Fig. 4) between 2011 and 2018, dry mass of forest floor horizons L+F of only two control plots (Pol1 and Obo) was ca 4 t per hectare higher compared to the thinned treatments. Effect of thinning was observed on the upper forest floor layers as all thinned plots showed increased decomposition rates reflected in lower total L+F amounts in both DF-dominated plots and also in reduced L and increased F layer in the mixed DF–SP Pol2 plot (Fig. 4).

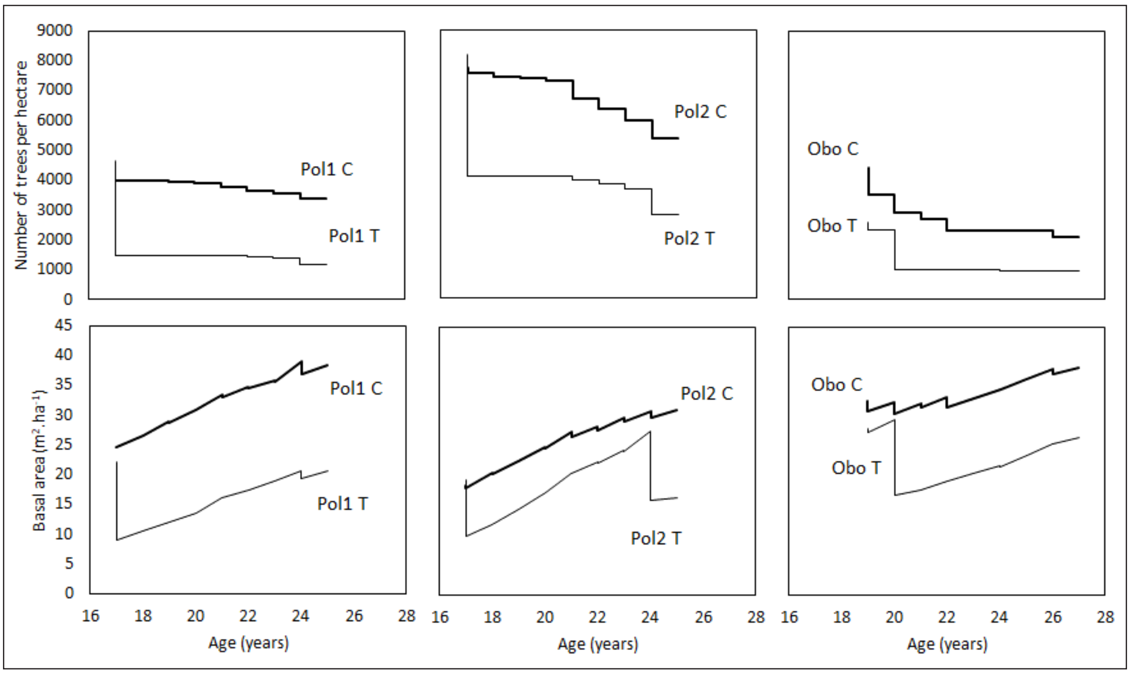

Fig. 1. Number of trees (above) and basal area (below) on experimental plots with litterfall observation (for plot designation see Table 1).

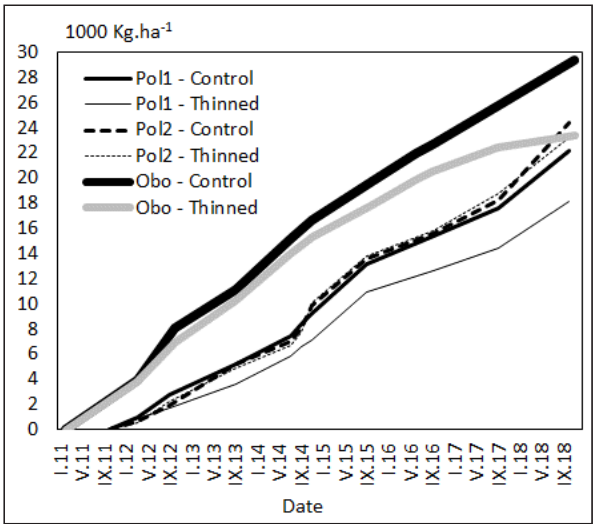

Fig. 2. Cumulative amount of litterfall dry mass in the period between 2011 – 2018 (for plot designation see Table 1).

Table 2. Amount of dry-mass and nutrients in litterfall under young Douglas-fir stands (for plot designation see Table 1).

	Plot	Dry mass	Nutrients				
			N	P	K	Ca	Mg
Amount for observation period 2011 – 2018 [kg ha^{-1}]	Pol1 C	22 121	293.0	20.4	30.5	103.0	14.0
	Pol1 T	18 177	220.3	13.7	25.3	84.4	11.8
	Pol2 C	24 419	188.0	10.6	29.9	112.2	12.4
	Pol2 T	23 227	209.5	11.9	22.4	86.3	9.5
	Obo C	29 373	394.0	26.9	41.8	240.7	16.3
	Obo T	23 383	312.4	21.2	30.5	165.8	12.1
Amount for mean annual litterfall [kg ha^{-1}]	Pol1 C	3 160	41.9	2.9	4.4	14.7	2.0
	Pol1 T	2 597	31.5	2.0	3.6	12.1	1.7
	Pol2 C	3 488	26.9	1.5	4.3	16.0	1.8
	Pol2 T	3 318	29.9	1.7	3.2	12.3	1.4
	Obo C	3 790	50.8	3.5	5.4	31.1	2.1
	Obo T	3 017	40.3	2.7	3.9	21.4	1.6

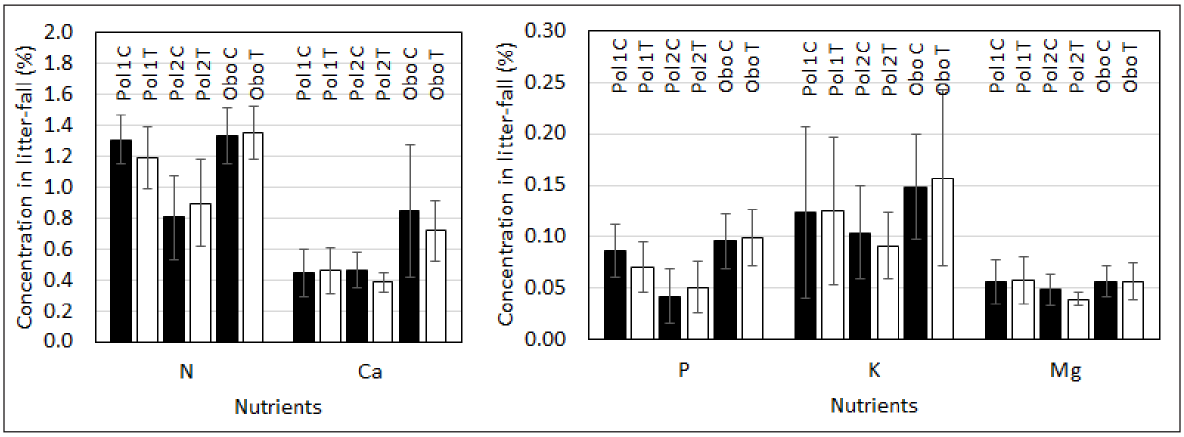

Fig. 3. Concentrations (mean with S.D.) of nutrients (left – N and Ca, right – P, K and Mg) in litterfall under young Douglas-fir stands (for plot designation see Table 1).

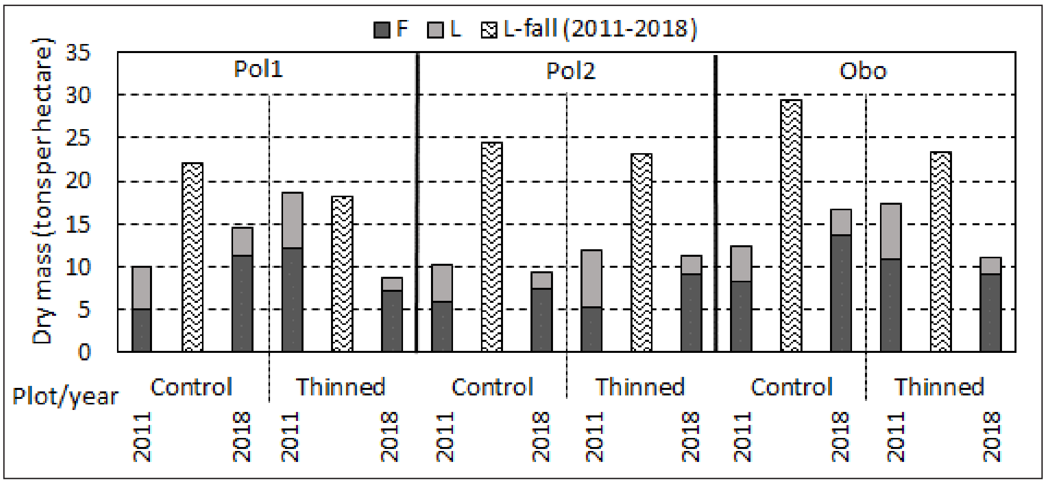

Fig. 4. Change of dry-mass accumulation in horizons L and F under differently thinned young Douglas-fir stands in connection with litterfall dry mass for the same period (for plot designation see Table 1).

4. Discussion

As for the DF litterfall amount, 20-year-old stands were reported as shedding between 1.4 and 2.5 t ha^{-1} of dry mass annually (Turner & Long 1975; Binkley et al. 1984). In 40-year-old and older stands, annual dry mass litterfall ranged between 1.4 – 3.5 t ha^{-1} (Will 1959; Mcshane et al. 1983; Fried et al. 1990; Longdoz et al. 2000; Berg et al. 2001). This, however, does not mean that litterfall increase mainly with the age. On the other hand the litterfall seems to be strongly related to basal area (Novák et al. 2014) which increases with the age; G is also strongly related to the total stand biomass of DF (Ponette et al. 2001). Besides G, also other stand production characteristics such as wood volume, above-ground biomass and mean annual increment correlated strongly with mean annual litterfall (Erkan et al. 2018). Maguire (1994) reported more released necromass through branch mortality from larger DF trees and denser plots with DF. In our 17 – 25-year-old experiments, the annual litterfall amounted to ca. 3 t ha^{-1} which was attributable to differences in stand density and G. For example Turner & Long (1975) reported 1.4 t ha^{-1} litter that fell off annually from the 23-year-old, thinner (650 trees ha^{-1}) stand with G amounting 9.7 m^2 ha^{-1} whereas 22-year-old, much denser (2 756 trees ha^{-1}) stand with higher G (42 m^2 ha^{-1}) showed 2.5 t ha^{-1} of litterfall (Binkley et al. 1984).

Although differences among tree species litterfall have been described (e.g. Augusto et al. 2002). Observed amounts of DF litterfall are consistent with those ones shown in the Czech native commercial conifers (Kacálek et al. 2018). It is also in accordance with Hansen et al. (2009), who reported no significant differences among litterfall amounts of tree species such as Norway spruce, Sitka spruce, Douglas fir, European beech and common oak in common garden experiment in Denmark. Hansen et al. (2009) concluded that previously reported large variability in forest floor accumulation should primarily be attributed to differences in litter decomposition.

The monitoring time span is a crucial prerequisite for getting reliable data as year-on-year amounts of falling-off litter vary which is confirmed e.g. by Will (1959) or Trofymow et al. (1991). The fluctuating values are also attributable to climate oscillations. As for the annual nutrient return, our values amounting ca. 30 – 50 kg N, 1 – 3 kg P, 3 – 5 kg K, 12 – 30 kg Ca a 1 – 2 kg Mg are similar to ranges reported for DF stands by other authors (Will 1959; Turner 1981; Fried et al. 1990; Trofymow et al. 1991).The litterfall amount itself is not, however, the most important from forest nutrition point of view. Density reduction of stands with higher basal area would result in more water available for trees and also in higher nutrient release (or lower nutrient immobilization) from decomposing needle litter (Bueis et al. 2018) which is a positive effect to the site though the total amounts of litter are reduced.

Thinning reduces litterfall amount (del Río et al. 2017) as the stand density is controlled deliberately. This can be expected if the pre-thinned stands are homogeneous. Accordingly, we observed higher differences (lower litter-fall due to thinning) in DF-dominated plots (Pol1 and Obo) compared to plot Pol2 with higher initial density, higher share of other species (Scots pine) and lower thinning intensity (by G). The trees left on site use more light, additional water and more available nitrogen (Chase et al. 2016) to increase the diameter increment and enlarge crowns thus closing the canopy again which increases litterfall gradually (Roig et al. 2005; Erkan et al. 2018). Trofymow et al. (1991) reported 15-year reduction of DF litterfall after removal of 2/3 of G at the age of 25 years. DF old-growth stands are expected to reduce foliage due to humidity stress and severe air temperature which also reduces transpiration rate and increase soil moisture (Dong et al. 2018). Therefore, the role of thinning, which emulates natural loss of needles, is to help stands cope with the climatic extremes.

Our results did not confirm the fear of the excessive accumulation of raw litter in forest floor under DF-dominated stands. The upper two horizons (L+F) accumulated 9 – 19 t ha^{-1} of dry mass. Similar (3 – 15 t ha^{-1}, Šarman 1982) and even higher (25 – 26 t ha^{-1}, Podrázský et al. 2006) results were reported for Norway spruce, or European beech where it amounted 14 – 18 t ha^{-1} (Podrázský & Viewegh 2005). Relatively quick decomposition observed in our experiment was additionally accelerated by thinning (confirmed on two experimental plots with lower initial density, lower share of other species and higher thinning intensity). It was in accordance with results of Wright (1957) and Wilhelmi (1988) for Norway spruce or Blanco et al. (2008) for Scots pine. In those studies, there were reported greater both litterfall and forest floor accumulation in unthinned or in (only) light-thinned stands. How this effect is reflected in the stocks of nutrients in the forest floor und upper soil horizons, it needs further and more detailed research and we included it into following analyses in our experiments.

5. Conclusion

Young DF-dominated stands react to very heavy thinning by lower amount of litterfall and quicker rate of litter decomposition. This effect showed similar trend compared to native conifers (Norway spruce, Scots pine). Thus DF can be recommended as the commercial species in the Czech Republic conditions which do not pose any excessive risk if used as admixed species with native broadleaves or conifers.

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ORIGINAL PAPER

Soil erosion in Mediterranean chestnut tree plantations at risk due to climate change and land abandonment

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Abstract

Chestnut cultivation is a type of traditional centennial exploitation in southern areas of Extremadura. Currently, chestnut is in danger of extinction due to the effects of climate change, some diseases (*e.g. Cryphonectria parasitica*) and land mismanagement. The high temporal variability of rainfall leads to soil erosion in chestnut cultivation. New forms of management are proposed to try, among other things, to reduce soil losses. This study quantifies soil losses in areas under tree canopies and open areas considering two different age plantations; 1990s and 1960s. To achieve the proposed goal 18 erosion plots were installed as paired plots under tree canopies and open areas in both plantations. The total amount of rainfall per event, tree cover, bare soil, runoff coefficient and sediment concentration were also measured in every plot. The results showed that the highest percentage of bare soil (> 45%) coincides with the period of greatest tree cover (> 75%). The open areas and the youngest plantation showed soil losses higher than the areas under the tree and the oldest plantation. In addition, soil losses increase as higher percentages of bare soil are recorded. We conclude that the size of the trees and a better soil stability in older plantations help reduce soil losses.

Key words: soil losses; land management; rural areas; traditional crops; sustainability

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1. Introduction

There are many theories relating to the origin of chestnut trees but no clear evidence has appeared yet about the existence of this species in Europe. Some authors are frequently considering that a natural origin of chestnut trees exists in eastern Europe (Martín et al. 2007). Something similar happens in the Iberian Peninsula with the natural origin of this specie. The hypothesis most accepted has been an introduction by the human bounded to the stage of the Roman Empire during the reconquest (Conedera & Krebs 2007). Nevertheless, some recent researches have shown the presence of chestnut trees before to the Roman era (Pulido et al. 2007).

In Europe, the chestnut tree can be found in many countries. France occupies the first position followed by Italy and Spain (Conedera et al. 2016). Also, the presence of this specie involves that the sweet chestnut (*Castanea sativa*) industry have a great economic potential in some countries in Europe (Centeno et al. 1998). The chestnut tree appears in Iberian Peninsula in almost all territory (Conedera et al. 2016), especially in areas in which accumulated rainfall is above to 1,000 mm y^{-1} within an altitudinal range between 500 and 1,200 m, with a cultivated area of 36,682 ha. The largest area of chestnut trees is in Galicia (24,592 ha) followed by Andalusia (8,856 ha) and Extremadura (2,481 ha) according to the Ministry of Agriculture, Fisheries and Food in 2018. In Extremadura, the chestnut trees are mainly located in mountain areas due to weather conditions.

The chestnut trees crop has had a large tradition in Europe since Roman period until today (Conedera et al. 2004), hence the extension of this crop occupies the most varied range of landscapes and soils. Acid soils are the best pedological type for the development of chestnuts since it has low clay content and a high percentage of organic matter (Berrocal 1998). Currently, traditional

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*Corresponding author. Jesús Barrena-González, e-mail: jesusbarrena@unex.es grafted and ungrafted species (e.g. *reboldos chestnut)* coexist in the region of Extremadura (Martín et al. 2017). Chestnut trees, like other arboreal species, suffer from diseases that involve a huge degradation of the plant and, in many cases, death. For many years the chestnut trees have been suffering two types of diseases, mainly in Extremadura, like chestnut blight (*Cryphonectria parasitica*) and the ink disease (*Phytophthora cinnamomi*) (González et al. 2013). Moroever the continuous abandonment of crops in rural areas (MacDonald et al. 2000; Renwick et al. 2013) and the context of climate change in the southern areas, the threats of disappearance is multiplied considerably.

But, can abandonment of chestnut cultivation prove beneficial in other environmental aspects? Considering that the chestnut trees are located on slopes more than 15%, with bare soil most of the year coupled with previous agricultural activities, the reduction of some erosive processes may be effective as shown in other studies (García-Ruiz 2010). In these areas where there has been a human substitution of shrub species such as *Cistus ladanifer*, re-colonization is the most probable scenario in case agricultural activities is abandoned.

At this time, when there seems to be a high awareness of the effects that climate change (Meira Cartea 2007) can have, we should rethink the forms of land use in each of the possible environments in which all agricultural activities are developed. One of the main lines of action is to reduce soil erosion rates. This action appears as one of the United Nations Sustainable Development Goals (Griggs et al. 2013; Keesstra et al. 2016) in the face of the progress of desertification processes and events of torrential rains increasingly common (Dregne 1986; Millán et al. 2005).

There are many published works on the quantification and monitoring of soil erosion in agricultural land (Pimentel et al. 1995; Lal 2001; Montgomery 2007). Often, these studies try not only to quantify soil losses but also aspects such as their effects on multiple soil functions (Doran & Zeiss 2000; Carter 2002). However, all the results and conclusions obtained do not seem to have penetrated much of the agricultural community in the Mediterranean basin (Carvalho et al. 2002; Cerdà et al. 2017). Frequently, land management still has the same unsustainable characteristics as decades ago. Even this type of unsustainable management also has no impact on political decisions. Public interventions about degradation processes in many places are usually minimal or nil (Requier-Desjardins et al. 2011).

Studies on soil erosion have been carried out to date in almost all possible agricultural systems, whether agricultural or livestock use. There are many works on soil erosion in olive groves (Fleskens & Stroosnijder 2007; Gómez et al. 2009; Kairis et al. 2013), vineyards (Rodrigo-Comino 2018), fruit trees (Cerdà et al. 2016; Keesstra et al. 2016), or in agrosilvopastoral systems (Schnabel 1997; Rubio-Delgado et al. 2018). However, studies that address soil erosion in chestnut crops are very few. There are some in Kazakhstan (Shokparova et al. 2014), and other works that only mention the existing problem.

The risk of chestnut disappearance in southern areas due to climate change is increasingly likely. If this situation is increased by the location of the chestnut crop on very steep slopes, there is a need to quantify and monitor the progression of these types of crops in Extremadura.

The main aim of this work is to quantify the losses by soil erosion in a farm of chestnut trees located on a hillside. The secondary objectives are to compare the amount of loss soil in chestnut with different ages (30 – 35 and 60 – 65 years) and to analyze the difference between areas under trees and in open areas. Our hypothesis is tree cover plays a crucial role preventing soil erosion.

2. Materials and Methods

2.1. Study area

The mean altitude of study area ranges from 437 to 1,437 m a.s.l. (Fig. 1A and 1B) and the mean slope of 16° (Fig. 2). The soil type that predominates in the study area can be referred to as an Entisol Orthent (USDA Taxonomy). It is a characteristic soil of recently eroded surfaces, located on unstable slopes that do not allow soil development. For this study, the chestnut plantation selected is located in the natural region of the Villuercas (Extremadura, SW Spain), in the heart of the Geopark Villuercas-Ibores-Jara (UNESCO World Heritage Site). It is a place where this type of crops is traditionally predominant over others (such as the cultivation of the olive grove), but where the lack of information on soil losses is very significant.

The chestnut trees are located in a field 3 km from the municipality of Navezuelas (Cáceres), known by the name of El Pilangrón. This farm was selected for presenting the characteristic patterns of this type of plantation in the surrounding area, i.e. steep slopes typical of the predominant Appalachian relief. The total annual average rainfall reaches 1,000 mm and the average annual temperature is 16.6 °C (Guadalupe meteorological station, REDAREX, 39°23'13'' N, 05°20'53'' W). The climate is Mediterranean of type Csa according to the Köppen climate classification (Köppen & Geiger 1930), characterized by cold months like January with an average temperature of 7 °C and very hot months like July with an average temperature of 26 °C. December (150.0 mm) and August (7.9 mm) are the wettest and driest months, respectively.

Trees were planted in the 1960s and 1990s (Fig. 1C) following a spatial pattern of 12 × 12 meters of distance between them to facilitate the different tasks that are practiced on the farm. In Navezuelas, since the nineteenth century the feet of chestnut trees are of the reboldos chestnut variety in which the graft of the *verata* variety is later performed (synonymous with *Injerta de*

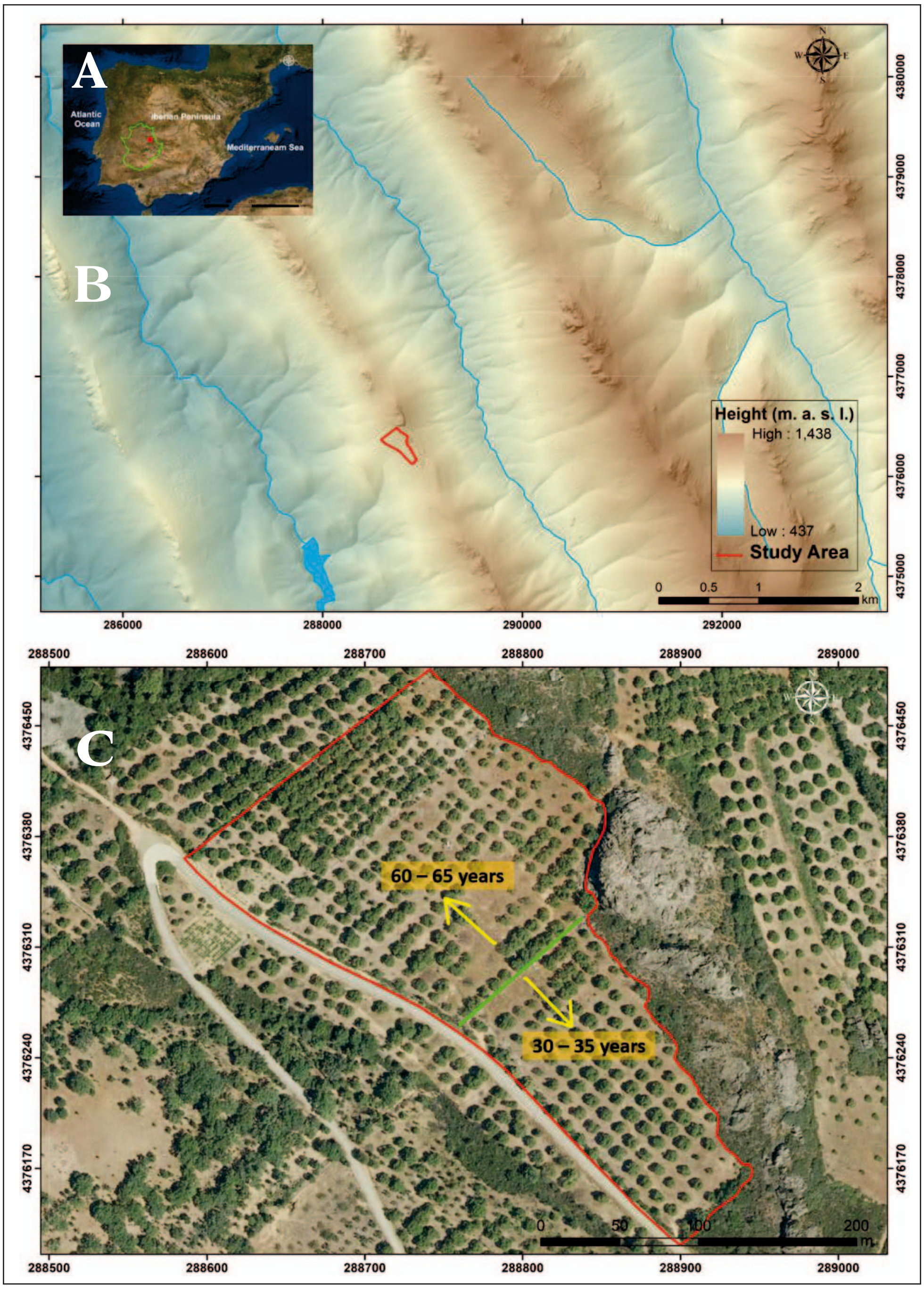

Fig. 1. Location of the study area (A), topographic elevations and changes (B) and distribution of chestnut trees plantation (C).

Navezuelas). The age at which chestnut trees start to produce varies between grafted trees and those that are not. Normally, ungrafted trees produce chestnuts in around 10 years while the graft starts in 5 or 6 years. Traditionally, reboldos chestnuts are grafted between 15 or 20 years old in Navezuelas. However, this kind of graft it has worse consequences than if the graft is done when the trees are 2 or 3 years old. This is mainly due to the fact that wounds caused by external agents (e.g. broken branches) are more difficult to heal on the feet of *reboldos* chestnuts that have been grafted later (personal communication with the owner).

2.2. Experimental design

For the experimental design, the two different types of chestnut age were considered [i.e., chestnuts planted in 1960s (A) and 1990s (B)]. A total of 18 erosion plots were installed randomly in pairs and taking into account two locations: under tree canopies and in open areas. In A: 4 under two canopies and 4 in open areas; in B: 6 under three canopies and 4 in open spaces. Erosion plots have a size of 2 m^2, 2 m in length and 1 m in width (Figs. 3 and 4). The height of the metallic material that acts as the edge of the plot is 10 cm high. Each plot in its lower

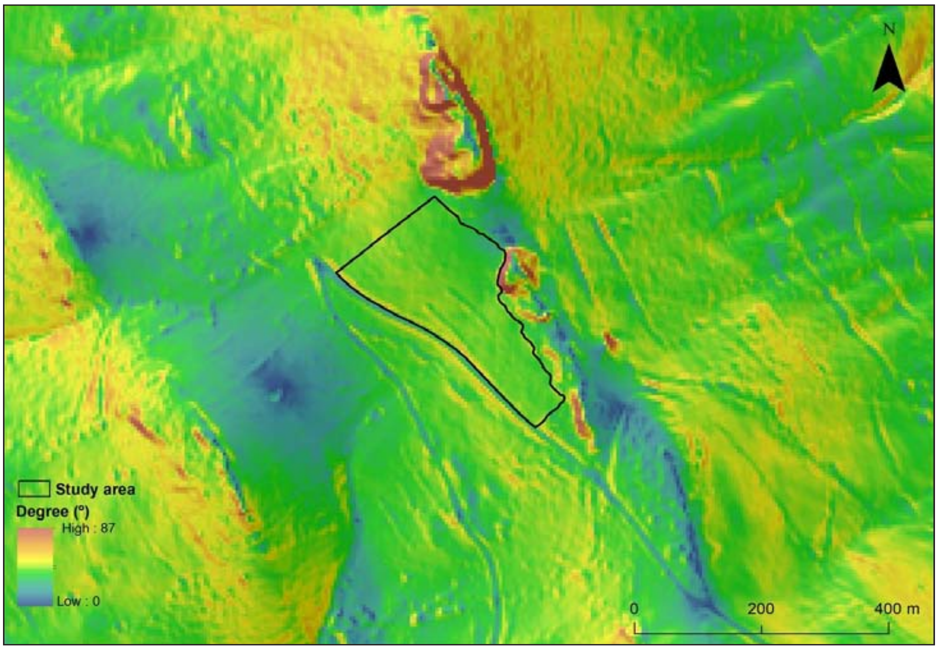

Fig. 2. Slope map of the study area and surroundings.

area has a collector that connects with a 25 litres bottle where the amount of water and deposits generated by the rain event is stored.

Measurements are made after each rain event. In each of the measurements, the accumulated rain is recorded in the pluviometer installed in the farm and it is using the data of Guadalupe meteorological station too. The percentage of bare soil and the percentage of tree cover in each plot was quantified through a visual estimate (Antoneli et al. 2018). Quantification measurements of bare soil and tree cover are always carried out by the same person. Water with or without sediments accumulated in the 25 litres carafe is deposited in precipitation vessels that are subsequently taken to the laboratory to quantify

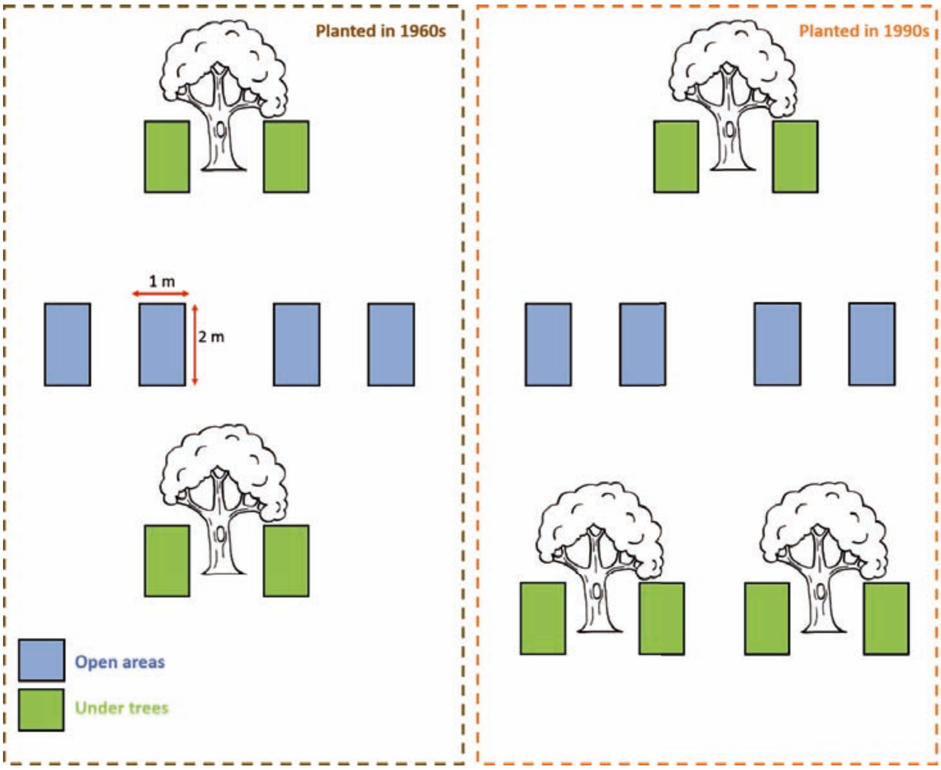

Fig. 3. Distribution and size of erosion plots on the farm.

the volume of accumulated sediments. The precipitation vessels are weighed and then put in an oven at 105 °C for 24 hours. Once all the water in the precipitation vessel has evaporated, the vessel is reweighed to calculate the volume of sediments (g/m^2). The runoff coefficient is measured taking into account the total rainfall of the event and the amount of water collected in the carafe.

2.3. Data analysis

For the statistical analysis, the mean and standard deviation were used in each of the variables studied to compare the results. For the study of the evolution of the bare soil, tree cover and quantification of the soil losses, the means with error plots graphs were used. In these graphs, appear the whiskers (± 1 Standard Deviation), Kruskal-Wallis test and F test, p (ANOVA). For the correlation between soil losses and total rainfall per event, the scatterplot graphs were used, where the correlation coefficient (r) and the Spearman correlation (p) appear. Statistical analysis was carried out by using Statistica 6.0 software.

3. Results

3.1. Rainfall characterization

Figure 5 shows the climate graph for the period studied. The total accumulated rainfall per month is shown on the bar chart and the average temperature in the same period appears on the line chart. In the climate graph, a distinction is made between the two periods studied so far. The 2018/2019 period (one full year) and the 2019/2020 period (4 months). The total accumulated rainfall in the first period (2018/2019) is 645 mm, when the average

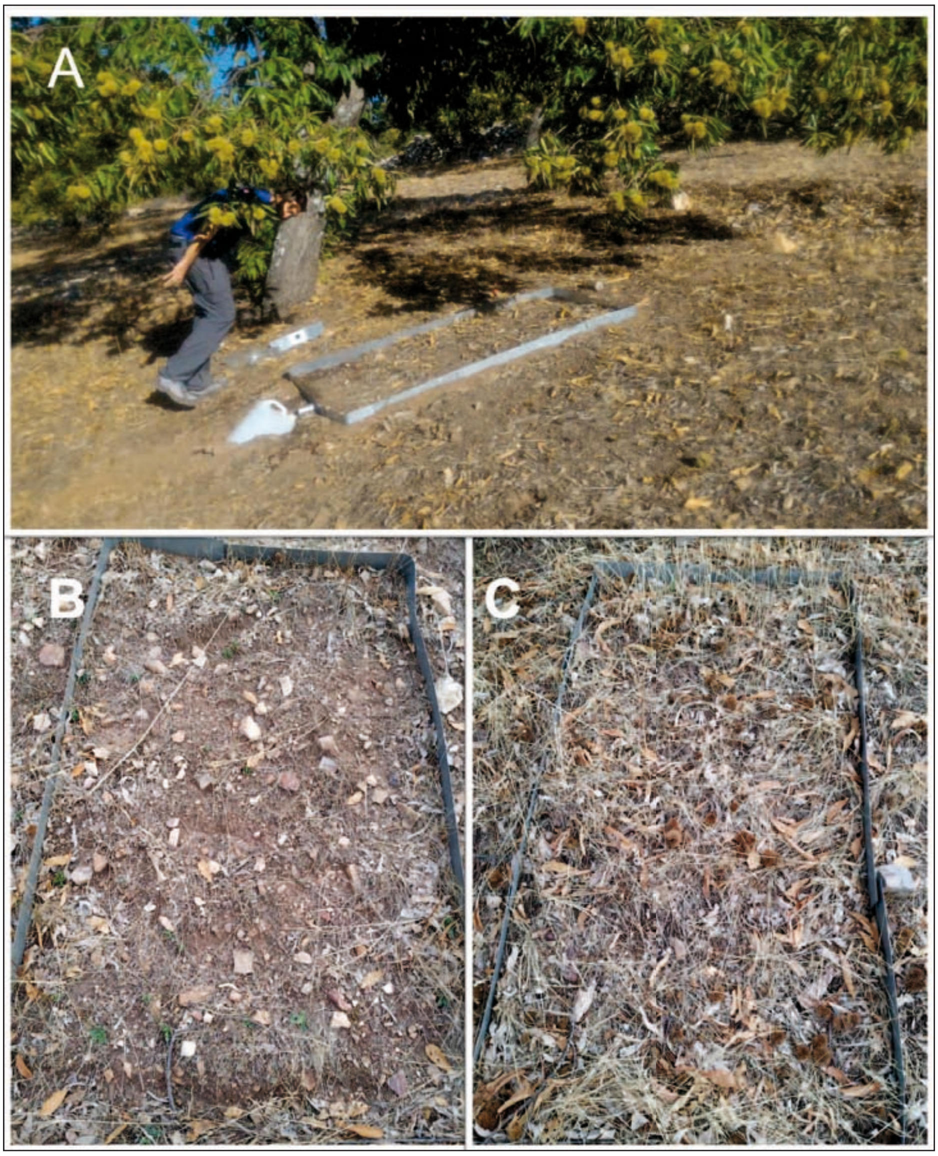

Fig. 4. Some plots during installation (A), open area (B) and under tree (C).

annual rainfall for a series of 12-year data (data available at the Guadalupe meteorological station) is 905 mm, a difference 260 mm.

According to the climatic seasons, in the 2018/2019 period the greatest amount of rainfall is concentrated in the autumn months (288 mm) while in the data series, the rainiest months are those of winter. The average rainfall in these months is 346 mm, 176 mm more than in the winter months recorded in the first period (170 mm). In general, in all months of the period rainfall values are recorded below those of the series, except for the months of October and November (124 and 163 mm, respectively), 11 and 68 mm more than in the data series in the same months.

In terms of temperature, the difference between the first period and the data series is 0.4 °C, higher in the study year 2108/2019 than in the data series (16.1 °C and 15.7 °C, respectively). In the second period (2019/2020), precipitation data, in general, is below the average of the series data in the same months, with a difference in the autumn season that reaches 55 mm.

Table 1 shows the total rainfall accumulated in Guadalupe station per each events and the accumulated rainy days. The greatest rainfall is mainly concentrated in the autumn and winter months. In general, rain events are reduced during spring and summer although some important event appears during the month of April and May (events 15 – 16) that exceed 50 mm. On the other hand, the most important rain events (5 – 11 – 25 events) have been recorded during the autumn and winter, registering a total rain of 80.2, 101.1 and 220.5 mm respectively.

3.2. Bare soil

Fig. 6 shows the average percentage of bare soil measured in the plots in each of the events. In general, bare soil follows a pattern related to seasonality depending on the time of year. In this case, the time of the highest percentage of bare soil in the chestnut soils coincides with the end of summer and, practically, all of autumn (see Table 1). During the period between late summer and early autumn, the soil surface is cleaned of leaves and branches in order to condition it to the chestnut harvest. Hence, during all this time there is a percentage of bare soil above 50%. However, as of November the percentage of bare soil begins to reduce considerably, reaching a percentage less than more than 25% (58.2% event 2 and 31.4% event 16). This stage initially coincides with the leaf fall of the trees that causes the soil to cover almost entirely of fallen leaves and some broken branches due to the natural pruning of the tree. And later, with the beginning of spring and the growth of all the grass marshmallows that grow during this time.

3.2. Tree cover

Fig. 7 shows the soil surface (%) covered by tree canopies. There are two different stages in the evolution of the percentage of tree cover. On one hand, we have found percentage values that are above 70 – 80%. These data coincide with the period of greater foliation of the chestnut, just before or during the time of the harvest (autumn months). However, we find other values quite lower than those aforementioned. These values that barely exceed 40% in some close to 20% corresponds to the winter stage. During this time, the tree loses the leaf in its entirety and it is not until the spring months when it begins again to reach higher foliation values. In addition to these two extreme moments, we find two transition stages that correspond, at first with the end of autumn and start of winter, and in a second moment, with the start of spring and the end of summer.

3.3. Soil losses

Figure 8 shows the sum of the average in g m^{-2} of all the plots distributed under tree and in open areas in plantations of different ages for the two study periods. There are 4 plots for each of the variables represented, except in the under tree plots of 1990s plantation where there are a total of 6 plots (see Fig. 2). We can see that there is an obvious difference between the two periods studied, due to the difference in the events recorded in one year and another. However, there are differences between soil losses recorded under trees and open areas and between the two plantations. For the first year of study, soil losses in plots in open areas of the 1960s plantation reach 51.3 g m^{-2} while in the 1990s plantation they reach 86.5 g m^{-2}, a difference of more than 30 g m^{-2}.

This situation is repeated in the second year of study with values for the plantation of 1960 and 1990 that reach

Table 1. Characterization of the rain events measured in mm.

Event	Date (*)	Rainfall	Event	Date	Rainfall	Event	Date	Rainfall
1	2018–09–17 (2)	3.94	9	2018–12–23 (6)	16.0	17	2019–05–18 (2)	7.3
2	2018–10–19 (7)	66.2	10	2018–01–20 (3)	25.2	18	2019–09–22 (11)	16.5
3	2018–10–28 (5)	19.5	11	2019–02–02 (5)	101.1	19	2019–10–19 (4)	17.0
4	2018–10–29 (2)	38.2	12	2019–02–14 (2)	3.6	20	2019–10–26 (2)	19.9
5	2018–11–10 (7)	80.2	13	2019–02–23 (0)	—	21	2019–11–09 (9)	23.4
6	2018–11–23 (8)	35.5	14	2019–03–09 (4)	26.6	22	2019–11–16 (3)	11.4
7	2018–11–25 (2)	37.6	15	2019–04–12 (8)	56.9	23	2019–11–23 (4)	54.3
8	2018–12–13 (7)	33.7	16	2019–05–01 (8)	77.4	24	2019–12–06 (8)	37.1
						25	2019–12–24 (11)	220.5

* Rainy days.

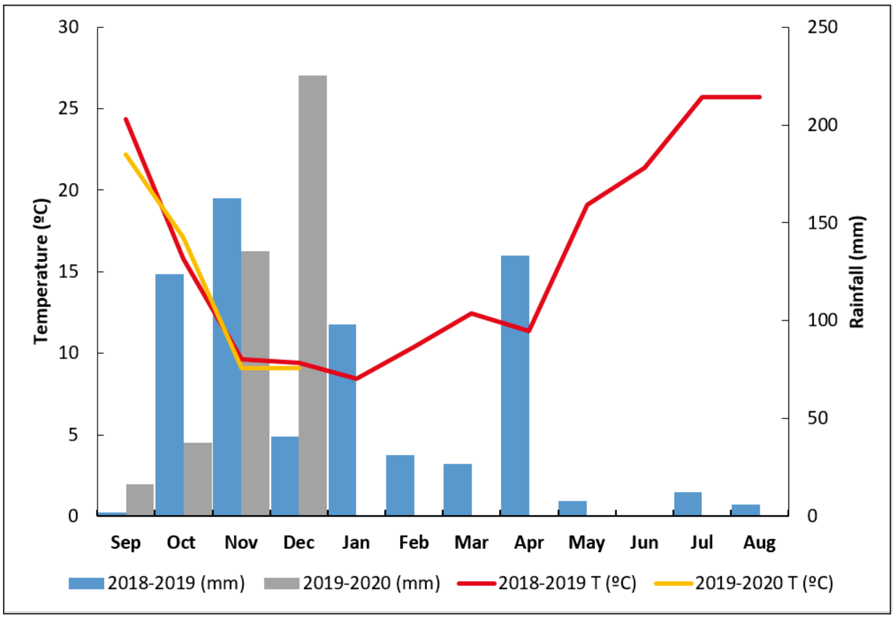

Fig. 5. Climate graph for the period studied. On the left axis the temperature is represented and on the right axis the rain. The blue bars and the red line represent the 2018/2019 data period and the gray bars and the yellow line the 2019/2020 period.

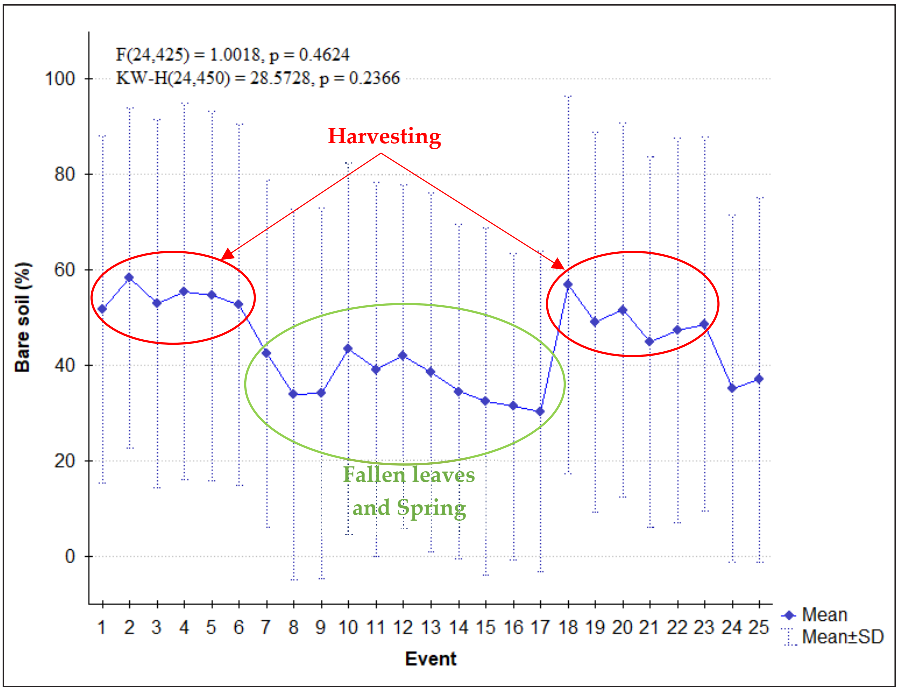

Fig. 6. Evolution of percent of bare soil per events. The solid line indicates the mean and the dashed lines (whiskers) indicate the mean ± 1 standard deviation.

soil losses of 13.1 g m^{-2} and 34.6 g m^{-2}, respectively. Regarding soil losses registered under tree soil erosion is reduced considerably, as if we compare between plantations with different ages. In the first year of the study, in the plots under tree of the plantation of 1960 losses of soil of 7.3 g m^{-2} were registered while in the plots of the plantation of 1990 they reached 13.5 g m^{-2}. In the second year of study, the same situation was recorded as in the first year although the volume of soil loss was lower (4.8 g m^{-2} and 2.8 g m^{-2}, respectively). The instability of the soil in the youngest plantation and the greater tree cover of the oldest plantation play a fundamental role in the loss of soil.

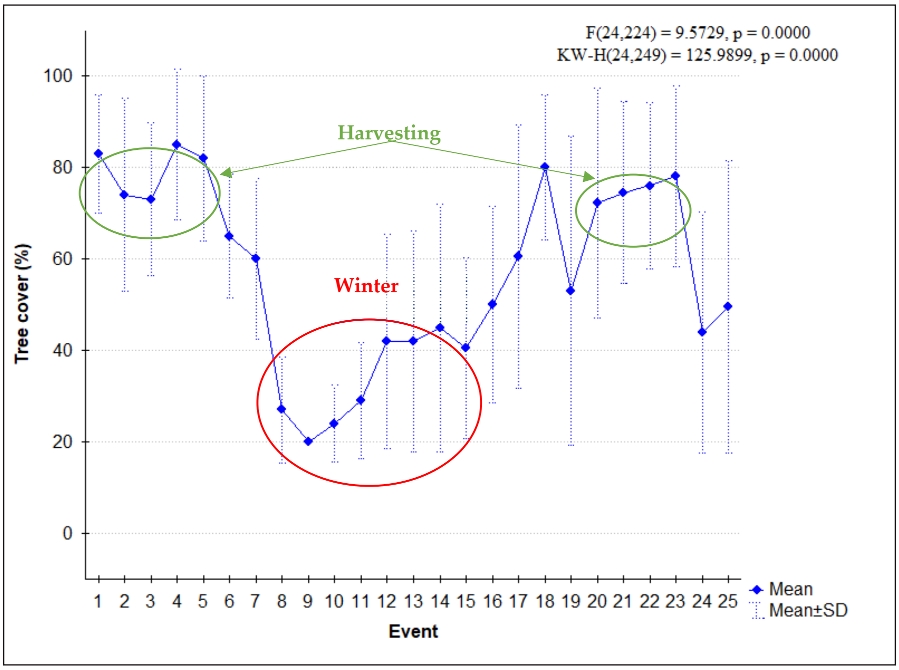

Fig. 7. Evolution of percent of tree cover per events. The solid line indicates the mean and the dashed lines (whiskers) indicate the mean ± 1 standard deviation.

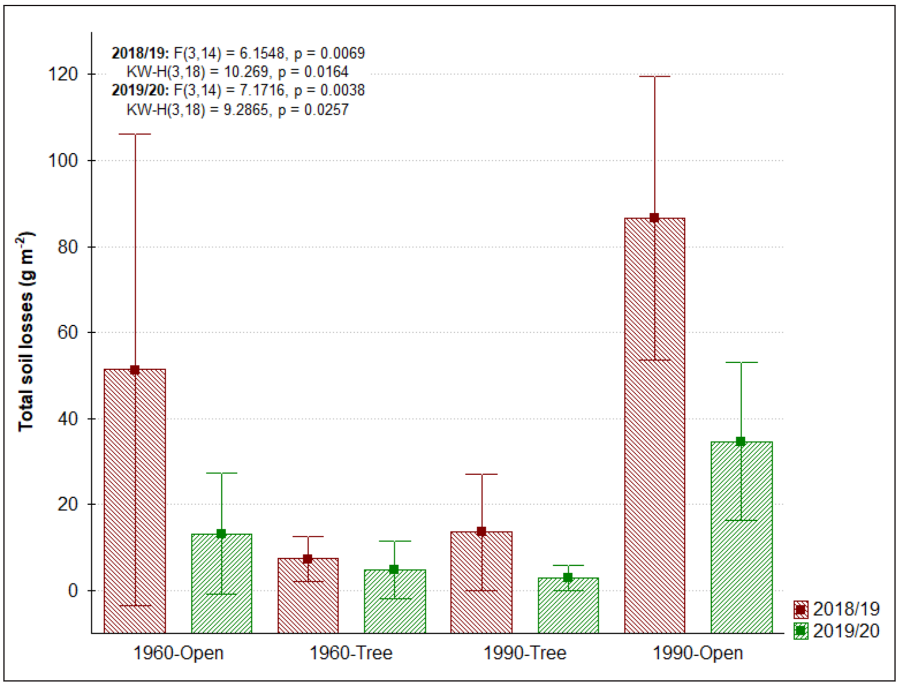

Fig. 8. Total of soil losses in plots under tree and in open areas in plantations of different ages in the period studied (2018 – 2019 / 2019 – 2020).

3.4. Correlation analysis

Table 5 shows the correlation coefficient between the variables studied. The bare soil shows a high negative correlation with the tree cover ($r = -0.567, p < 0.001$). However, bare soil show a high positive correlation with soil losses ($r = 0.531, p < 0.001$). It also shows a significant positive correlation with runoff and sediment concentration ($r = 0.475$ and $r = 0.310, p < 0.001$, respectively). Interestingly, soil losses also show a negative correlation with tree cover ($r = -0.276, p < 0.001$). In turn, soil losses have a high correlation with runoff and sediment concentration ($r = 0.813$ and $r = 0.745, p < 0.001$, respectively).

Table 2. Spearman's correlation coefficients between soil erosion parameters, rainfall, bare soil and tree cover percentage and sediment concentration.

	Rainfall	Bare soil	Tree cover	Runoff	Sediment concentration
Bare soil	−0.010				
Tree cover	−0.002	−0.567***			
Runoff	0.190***	0.475***	−0.344***		
Sediment concentration	0.140**	0.310***	−0.096	0.403***	
Soil losses	0.245***	0.531***	−0.276***	0.813***	0.745***

Significant correlations have been highlighted in bold. *, **, *** mean significant at $p < 0.05$, < 0.01, and < 0.001, respectively.

Fig. 9 shows the scatterplot between soil losses and volume of rainfall recorded by events. The correlation that exists between soil losses and the amount of rainfall is not very high ($r = 0.149$, $p = 0.002$). Soil losses in general show values below 20 g m^{-2} in events ranging between 30 and 40 mm. We can see that the same amount of rain can produce more or less erosion.

Figure 10 shows the scatter plot between the values of soil losses and the percentage of bare soil. Soil losses are highly correlated with the percentage of bare soil

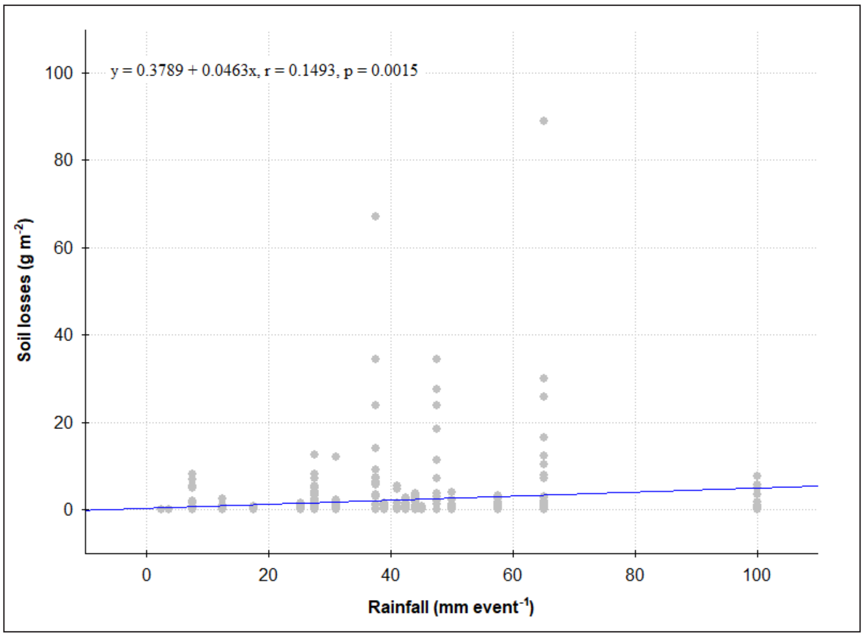

Fig. 9. Correlation between soil losses (g m^{-2}) and the amount of rainfall (mm event^{-1}) per events.

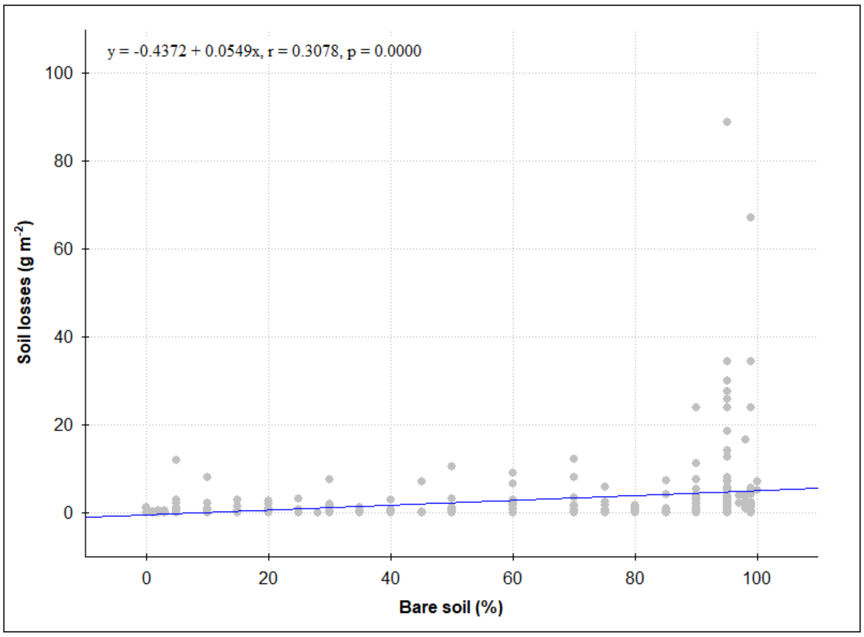

Fig. 10. Correlation between soil losses (g m^{-2}) and percentage of soil loses.

(r = 0.308, p = 0.000). As the percentage of bare soil increases, soil losses increase. The values of soil losses rise significantly from percentages of more than 85% of bare soil.

4. Discussion

The results obtained in erosion plots installed in the same farm but with different plantation ages (1960s and 1990s) were compared. In addition, the comparison includes the resulting values of the plots installed in open areas and under tree. Bare soil values show a regular trend along something. In chestnuts the highest values of bare soil (more than 50%) are concentrated at the end of the summer and during the autumn months. Interestingly, in the case of chestnut trees, the stage with the largest area of bare soil under a tree coincides with the moment of greatest foliation of chestnut tree. The main reason for this situation is mainly due to the cleaning of the soil in late summer and early autumn to facilitate the collection of chestnuts. The cleaning of the soil is carried out in different ways, by grazing, using herbicides or eliminating all herbaceous cover under tree with rakes.

Tree cover evolution values follow a similar trend to those of bare soil. It is in the harvest months where the highest tree cover values were recorded, being the lowest during the winter. The action exerted by the tree as an interceptor of raindrops in these cases is essential, reducing the volume of precipitation that reaches the ground as demonstrated by Mateos & Schnabel (2013). Since in this case the largest volume of bare soil coincides with the maximum tree cover a clear balance between parts appear. Thus, cleaning the soil under a tree does not seem to be an unsustainable practice of soil erosion management, in terms of splash erosion effects.

The greatest losses of soil were recorded in open areas with values that reach 86.5 g m^{-2}, values that differ many of those found by Guzmán et al. (2013) in an olive grove crop between rows (1.4 kg m^{-2}). In addition, the rates of soil loss are higher in the younger plantation than in the oldest one demonstrated by Rodrigo Comino et al. (2017) in a vineyard farm. The lowest soil erosion values were measured in plots under a tree, reaching a maximum soil loss of 13 g m^{-2} in the 1990s plantation. As Cerdà et al. (2019) showed, vegetation plays an essential role in soil erosion, exponentially reducing soil losses throughout the year. In other soil erosion studies, rates are higher than those found in chestnut trees, as Comino et al. (2016) found in vineyards in Spain and Germany, as a result of increased tillage and the lack of herbaceous cover due to the application of herbicides. However, these soil erosion rates are considerably reduced in abandoned agricultural areas, as Cerdà et al. (2018) demonstrated in mountainous areas of the Mediterranean, where the disappearance of tillage and the appearance of herbaceous cover were the key factors. Something similar can occur in our study area where the advance of *cistus ladanifer* occupying a large part of the plot could reduce the effects of water erosion.

The losses of soil and the volume of rains by events did not show a high correlation (r = 0.149). In our case, the same amount of rainfall produces the same erosion rates. However, in other studies it has been shown that the increase in the volume of rains causes an increase in soil losses as demonstrated by Arnaez et al. (2007) or Mohamadi & Kavian (2015) comparing different rainfall intensities. This situation leads us to consider that soil losses with similar behavior are determined by other factors.

The percentage of bare soil and soil losses show a greater correlation (r = 0.307) than previous variables. In this study, loss of soil may be greater than other studies, e.g. Antoneli et al. (2018) because the percentage of bare soil is high. Antoneli et al. (2018) found soil losses in different pastures type that ranged between 10 g m^{-2} and 20 g m^{-2}, with a percentage of bare soil between from 9 to 24%, soil losses increased when percentage of bare soil exceeds 30%. Nevertheless, in our study, below 85% of bare soil, the soil losses did not exceed of 20 g m^{-2}, while above 85% of bare soil, soil losses exceed of 30 g m^{-2}. A high soil exposure accelerates degradation processes causing the loss of other vital functions of the soil such as nutrient recycling.

5. Conclusions

In the southern chestnut trees located in Extremadura, the greatest losses of soil were recorded in open areas and in the youngest chestnut plantation. The tree protects the soil by acting as a protective shield against the impact of raindrops, providing the soil with a layer of leaf litter that helps curb the effects of water erosion. The larger size of the tree canopy and better soil stability in older plantations helps reduce soil losses. The highest percentage of bare soil in southern chestnut trees coincides with the moment of greatest tree cover, creating a balance between parts. However, the results indicate that soil losses increase as the percentage of bare soil increases. Given this situation, the cleaning of the soil under the trees should be avoided during the harvest as suggestion, trying to have the soil covered with vegetation through the year. Nonetheless, further research (installation of Gerlach troughs and a gauging station) and a longer series of data are still needed in order to draw definitive conclusions.

Acknowledgments

We would like to thank Lope Pulido (owner) for allowing us to install all the necessary material to carry out this research and future ones. Also thank the Regional Government of Extremadura for funding the Jesus PhD scholarship (PD 18016). This *work has been financially supported by the research Project IB16052 "Desarrollo de un Sistema de evaluación integral espacialmente distribuido para explotaciones de ganadería extensiva" founded by Junta de Extremadura and European Found of Regional Development (European Union).*

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ORIGINAL PAPER

Northern red oak (*Quercus rubra* L.) as a species suitable for the upcoming seasons with frequent dry periods

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Abstract

Northern red oak (*Quercus rubra* L.) has higher tolerance to dry periods and to lower soil nutrient content. It seems to be more resistant to tracheomycosis and therefore it should be considered as a promising species on selected forest sites. The research plots are situated in the Louny region with significant rainfall deficit; good model area for future expected climatic changes. New plantation of northern red oak and sessile oak on together 10 subplots with 500 plants were measured between 2015 and 2019. Results proved significant level ($\alpha = 0.05$) of better height growth of northern red oak as well as higher resistance to dry periods (lower mortality). Lang's Rain Factor proved to be a good indicator of difficult climatic condition during growing seasons and therefore could be used for realistic evaluation of possible success or failure of artificial regeneration.

Key words: climatic changes; regeneration; height growth; Lang's Rain Factor

Editor: Igor Štefančík

1. Introduction

Northern red oak is characterized by higher tolerance to dry sites, even the climatic conditions at the original sites of northern red oak include annual precipitation totals between 700 – 2030 mm. Annual average temperature of this area is between 4 and 16 °C (Sander 1990). In Europe, this species usually grows at altitudes of 250 – 800 m s.l. (Nagel 2015) with annual rainfall of only 500 – 550 mm per year (Nagel 2015; Burkardt 2018). The species shows higher resistance to low rainfall than sessile oak and pedonculate oak (Durand et al. 1983; Timbal & Dreyer 1994; Dressel & Jäger 2002). The physiological explanation for this tolerance is that northern red oak leaves have a higher tolerance to water scarcity, mainly because they have a higher stomata density on the leaf than domestic oaks (Timbal & Dreyer 1994) and species can thus respond more efficiently and quickly to water deficits. The species is also significantly more resistant to very low temperatures, with temperatures up to −30 °C or even −40 °C (Brus 2011). It is also resistant to late spring frosts as it sprouts later than domestic oaks (Jacamon 1987). On the other hand, this tree often tends to form forks, which is a consequence of the higher number of terminal buds and therefore it is necessary to take into account this problem especially in young stands and possibly to pruning and form a crown of young oaks in time. Also in autumn, especially in the case of young trees, there is a risk of the crown breaking, as the dry leaves persist for long periods on the branches and can cause considerable damage if larger amounts of wet snow occur.

Northern red oak is reported in the literature as a species, which is suitable for arid and semiarid areas, as it copes well with the lack of precipitation. According to these characteristics, northern red oak is suitable for expected climatic changes bringing a different distribution of precipitation with probably less total annual amount. Its resistance to air pollution and higher resistance to frost is also known (Miltner & Kupka 2016). According to a survey of the health status of oak stands, it appears that northern red oak is a species that is minimally damaged due to tracheomycosis (Štefančík 2011a). Northern red oak is considered a tree species with relatively easy natural regeneration, but requires frequent silvicultural interventions (Štefančík 2011b). At the same time, it is suitable at sites of high risk of soil erosion and soil degradation (Miltner & Kupka 2016). Northern red oak has a high wood production (Kupka et al. 2018). However, its drought tolerance is more important nowadays, and

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**Corresponding author. Ivo Kupka, e-mail: kupka@fld.czu.cz* hence better regeneration (both natural and artificial) in areas with more frequent dry periods.

Many authors report lower northern red oak requirements for nutrient content in the soil (Bauer 1953; Miltner et al. 2016). Northern red oak plantations are therefore more productive, especially at poorer sites. Northern red oak tolerates acidic sandy soils, but also performs well on heavier gley soils. It does not prefer calcium rich soils, although it seems that with age its tolerance to higher calcium content increases. Sites with high levels of groundwater are also unsuitable. In general, deeper soils slightly acidic with a minimum calcium content are suitable for northern red oak.

The root system of northern red oak is similar to that of domestic oaks. The young tree has a well-developed tap root, which in five years can reach a length of up to 70 cm, later forming a cordial root system (Bauer 1953; Nagel 2015). Later, significant lateral roots are formed, which extend to a considerable distance from the trunk, creating a solid root system that resists wind damage (Timbal & Kremer 1994; Réh & Réh 1997). Root system is growing throughout the whole growing season without interruption during summer period (Dickson 1994; Hubert 1994).

Further work has also shown the effect of light periodicity on the growth of seedlings (Guo et al. 2001). This study has shown that after the first year, when the plant needs more protection from full sunlight, from the second year its development is significantly better if it has sufficient light (Guo et al. 2001). This is important to keep in mind when planning the regeneration of this species, whether natural or artificial.

Appropriate density of mature stand is an important issue when applying management methods of shelterwood systems. Spetich et al. (2002) studied the influence of parent stand density on the growth of oak seedlings. The experiment was carried out with the plantation of two-year bare-root oak seedlings. At a basal area stocking of 0.8, at a collar root diameter of 6 mm, only one in 144 seedlings was fully vital after 11 years. Using thicker and older seedlings (collar root diameter of 22 mm), this ratio was one in five at the same stocking. A similarly beneficial effect on the growth of undergrowth or underplanting is the opening of the canopy by removing the secondary stand (Craig et al. 2014). Similar results were obtained by Hartman et al. (2005), which recommends reducing the crown canopy to 25% of the full canopy. After that he achieved the highest seedlings density of natural regeneration. Also, Dey (2014) emphasizes that in order to preserve the oak in the stands, care must be taken to maintain it at a level so that it can be naturally regenerated. Major et al. (2013) when assessing the natural renewal of northern red oak in the NW Germany found that the crown canopy is not the main limiting factor, but it is also an undergrowth with an average height of about 2 m, which influences the northern red oak regeneration. It is therefore recommended to remove this undergrowth component after the seed year (Major et al. 2013). Also Hart et al. (2012) concluded that strong competition of red maple (*Acer rubrum*) makes it impossible to restore northern red oak and without maple reduction this species gradually disappears from stands in the eastern US. These findings are confirmed by another study that looked at the development of the species composition 30 years after final cutting (Gould et al. 2005). This study confirmed that oak seedlings have significantly higher aboveground biomass and greater leaf area index LAI than seedlings growing under the shade under the upper layer. Schweitzer & Dey (2011) also noted in their study that the manipulation of the crown canopy significantly influenced the species composition of the regeneration. Active intervention is required to keep the northern red oak in the new regeneration.

Another option for regenerating northern red oak is the clear-cutting. Meadows & Stanturf (1997) or Fan et al. (2015) state that the most successful natural regeneration of northern red oak can be achieved after clear-cutting. Shelterwood cutting or leaving seed trees will not ensure uniform regeneration over the entire area. Selective cutting is not suitable for this species. Also Aldrich et al. (2005) confirmed the long-term observation (since 1926) of the species composition of mixed oak stands that its gradual reduction takes place if no clearcut is performed in the stands. Collins & Battaglia (2002), who used a small scale, group like clearcuts with a radius of 7, 10, 14, 20, 29 and 40 m on which randomly spaced acorns sown under the hoe, also investigated the appropriate cutting size. The survival rate was 21 – 37% in the first year and only 8 – 26% in the second year. The size of germinated seedlings and the size of their assimilation apparatus in the first year after germination had a more significant effect on mortality than the size of the gap cutting. In terms of micro-relief, the survival rate was slightly higher for acorns planted on higher sites than on depressions. Here, too, the importance of light for the survival of oak seedlings is confirmed.

Also Elliott & Knoepp (2005) assessed the effect of different regeneration cuttings in oak stands this time on the physical condition of the soil and nutrient content. Their work also recommends the use of (small) gap cutting, rather than the shelterwood cutting, both in terms of the physical condition of the soil after the many travels of harvestors, and in terms of accessibility of the main soil nutrients.

Northern red oak is often cited as an example of an invasive species. Available literature suggests that its capacity for natural seed regeneration is site dependent (Healy et al. 1999). On poor sites, it becomes abundant and may have an invasive character, while on richer soils it is significantly less invasive (Henin & Vandekerkhove 2016).

The aim of this study is to assess the regeneration of northern red oak and its growth in the first years after planting under given climatic conditions of recent years in an area characterized by longer dry periods. It also assesses the value of the Lang´s rain factor to evaluate local climatic conditions.

2. Material and methods

The Louny region of northern Bohemia is made up mostly of areas lying in the rain shadow of the Ore Mountains with a significant rainfall deficit. It is therefore a model area in terms of expected climate change. Established research areas consist of 5 single-sub-areas for each species in a random arrangement in one site with a total area of 20 × 50 m. The surrounding edge of research area planted by the same species varies between 1 to 2 m to avoid side effect. Research plots are established near Holedeč and basic information are given in Table 1.

Table 1. The basic data on research plot.

Species	Forest site	Altitude [m a.s.l.]	Average size of plant [cm]
Red oak	1C5*	220	30
Sessile oak	1C5*	220	32

Note: * Carpineto-Quercetum.

Two-year seedlings of bare-rooted (1+1) were planted in spring 2015. Spacing of the seedlings were 1.3 m × 1.3 m. The area is fenced to eliminate the negative effects of game. The differences between the height of red oak and sessile oak seedlings were statistically insignificant. Established plots with planting of red oak seedlings were monitored and measured in autumn in the years 2015 – 2019. The total height and actual height increment with the precision of 1 cm were measured. Mortality and appearance vitality were evaluated. The average height of sessile oak and red oak were 44 cm and 45 cm respectively at the end of the year 2015.

The data was processes by statistical package STATISTICA. Non-parametric two-sample Kolmogorov-Smirnov test were applied on data sets to evaluate differences on significant level $\alpha = 0.05$ for $n_1 = n_2 = 250$.

The official meteorological data for the Ústí nad Labem region, where these areas are located, show the following annual aggregate data (Table 2).

3. Results

The first step in data processing was to evaluate climatic conditions which are definitively one of the key factor for survival rate and growth of new plantations. The graphs in Fig. 1 and 2 show the course of monthly temperatures and precipitation in individual years.

It is clear from the graphs that while the temperatures were very similar in these years, the volume of precipitation varied considerably from one year to the next and in particular 2018 and 2019 were very unfavourable in terms of precipitation. Although the red oak plantation were in good condition at that time and 3 – 4 years after planting, this unfavourable rainfall balance was reflected in them. Although precipitation in 2018 is higher than in 2016, temperatures are also higher, especially in spring (April May) and summer (July and August). In terms of precipitation, 2018 was therefore more favourable than 2016. However, the results on experimental plots with oak do not confirm this. It is therefore necessary to use

Table 2. Meteorological data in the Ústí nad Labem area.

Year	Average year temperature	Annual precipitation	Precipitation in large season	Temperature in large season	Precipitation in main season	Temperature in main season
	[°C]	[mm]		[°C]	[mm]	[°C]
2015	9.5	591	467	13.0	281	16.3
2016	8.9	655	467	14.0	373	16.5
2017	8.9	667	500	13.1	414	15.2
2018	9.7	438	260	15.9	231	16.9
2019[1]	10.9	384	384	13.5	275	16.6

Note 1: data for 2019 include data for 1 – 10 only months

Note 2: in accordance with bioclimatology, the following periods are considered here:

- large growing season with monthly temperatures higher than 5 °C
- main growing season with monthly temperatures higher than 10 °C.

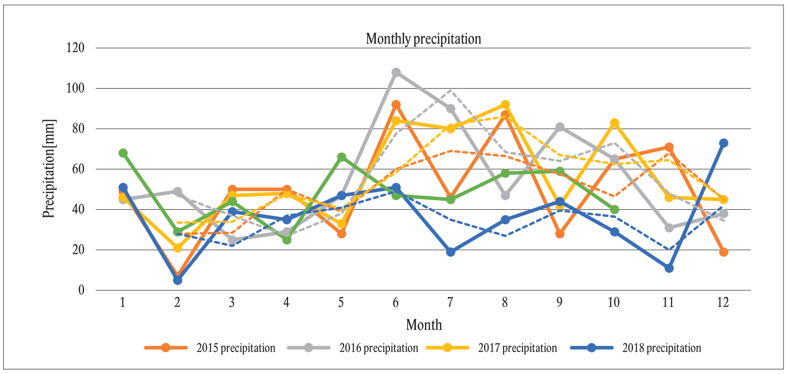

Fig. 1. Monthly precipitation in the research area in the period 2015 – 2019.

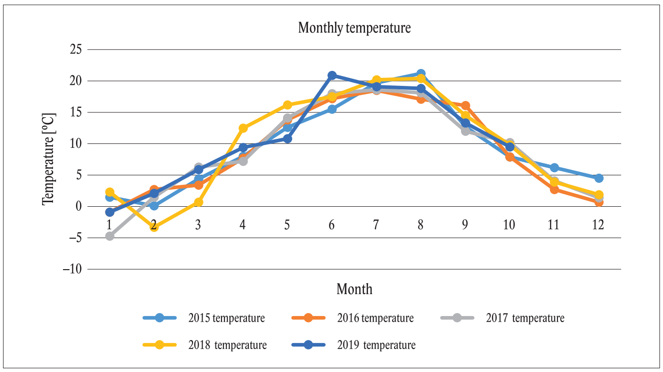

Fig. 2. Average month temperature in the research area in the period 2015 – 2019.

a different climatic characteristic, namely Lang's Rain Factor (LRF).

It takes into account the rainfall to temperature. Higher temperatures mean higher evapotranspiration and hence a higher need for precipitation to maintain the same vegetation conditions. The LRF value for the whole year is 73.60 for 2016, but only 45.15 for 2018, which is only 61.3% of the 2016 value. Even greater differences in these years are noticeable, considering only the growing season. In the large growing season, the values are 33.26 (2016) and 16.38 (2018) which represents only 49.2% of 2016. The same situation is for the main growing season where the LRF is 22.55 (2016) and 13.68 (2018) which is again only 60.7%. The figures are given more clearly in Table 3.

Table 3. Lang's Rain Factors for period of the years 2016 and 2018 for research area.

Year	Year LRF	Large growing LRF[1]	Main growing LRF[2]
2016	73.60	33.16	22.55
2018	45.15	16.38	13.68

Note:
[1] – large growing (season) is the period with average day temperature higher than 5 °C;
[2] – main growing (season) is the period with average day temperature higher than 10 °C.

These unfavourable conditions were indeed reflected in the research plots, as shown in the following graphs. Even the relative mortality rate is lower for red oak in the first three years after planting (2015 – 2017), these differences were not pronounced distinctly (Fig. 3). In 2018, however, there is a significant break (due to unfavourable climatic conditions), when the relative mortality of sessile oak reached almost 29%, while in red oak plantation it was only 11%, i.e. just over a third of the mortality of domestic oak.

Similar differences are also evident in the height development of plantations (see Fig. 4).

While the seedlings were nearly identical in height at the time of planting (2015), their development in the first two years after planting was not different on significant (level $\alpha = 0.05$; $D_2 = 0.089 < D_{2max} = 0.121$). Year 2018 markedly slowed down the growth of sessile oak due to unfavourable climatic conditions, while red oak growth decreased only slightly. From that year differences between these two species and this differences were statistically significant level ($\alpha = 0.05$; $D_2 = 0.132 > D_{2max} = 0.121$). The same picture could be recognised for height increment where the differences are significant even for level $\alpha = 0.01$ ($D_2 = 0.150 > D_{2max} = 0.146$). These differences persist till the last year (2019).

4. Discussion

Planting is in a semi-arid area at an altitude of 220 m. a.s.l. This is an area suitable for growing oak. Planting mortality in our plots was significantly lower in the first year (see Fig. 3), than reported by Jacobs et al. (2005),

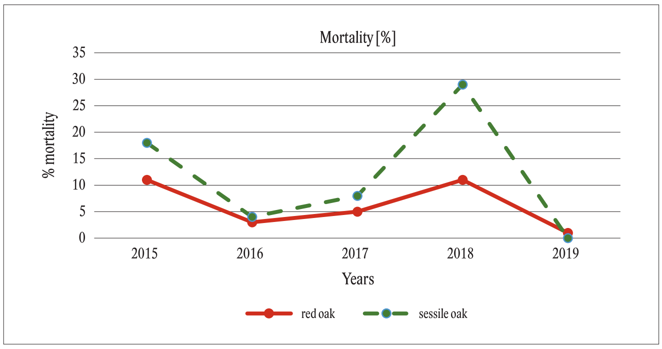

Fig. 3. Mortality in new plantation of red oak and sessile oak.

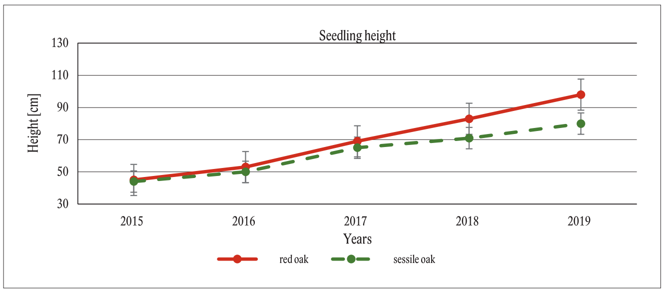

Fig 4a. Height growth of red oak and sessile oak in the period of 2015 – 2019.

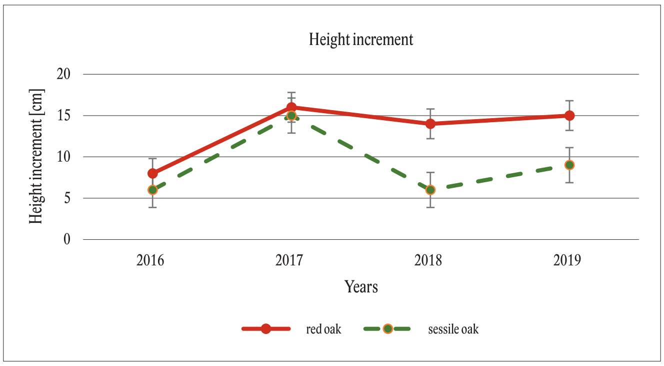

Fig 4b. Height increment of red oak and sessile oak.

where the mortality rate of unfertilised seedlings is up to 40%, while in our areas the mortality rate was 18% for sessile oak and 11% for red oak. This is in line with the recommendation of Oswalt et al. (2006) – sort out poor-quality seedlings before planting – which was also done in this case. This will save the cost of planting seedlings that do not have a high chance of survival.

No herbicide was used to reduce the weed, but only mechanical reduction was applied. The aim was to obtain data that are not affected by the application of chemicals, as some studies have pointed out (Demchik & Sharpe 2001; Schreffler & Sharpe 2003; Schuler & Robinson 2006).

The height growth of oak seedlings in our plots was very similar to the findings published so far (e.g. Spetich et al. 2002; Hartman et al. 2005; Craig et al. 2014) although some of them give data for the development of seedlings under mature stand and not in the open area, as in this case (see Fig. 4).

It is also clear from the graph in Fig. 4 that red oak grew better than sessile oak and from 2017 (2 years after planting) it started to reach statistically significantly higher average height than domestic sessile oak. This is even more evident in the adjacent graph of the same figure where the height increments are expressed. Since 2017, they are significantly higher than those of domestic oak.

The results proved the advantages of red oak plantations in terms of height growth and lower mortality in semi-arid and arid climatic conditions. They indicate the shorter time for safe reforestation which could be important advantages in near future.

Climatic characteristics (precipitation and temperature) co-determine the success of afforestation. The LRF can serve as a suitable criterion for assessing local climatic conditions and their impact on the field performance of seedlings and hence for a more objective assessment of the quality of planting.

5. Conclusions

The results of surveys on research plots with sessile oak and red oak plantations confirmed that in areas with higher temperatures and lower precipitation as expressed by Lang's rain factor, red oak can be successfully grown, showing better resistance to dry climatic conditions than domestic oaks in both growth and field performance first years after planting. With regard to the current development of basic climatic characteristics, it is necessary to consider adaptation to these conditions also with regard to species composition changes in our forests.

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ORIGINAL PAPER

Does shelterwood regeneration on natural Scots pine sites under changing environmental conditions represent a viable alternative to traditional clear-cut management?

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Abstract

In the context of climate change, the crucial question is how silvicultural treatments should be modified, in order to reach favourable conditions for initiating natural regeneration of forest stands. The aim of the study was to evaluate the influence of clear-cutting, basal area reduction (0.4, 0.6 and 0.8) and soil preparation (milling cutter, forestry mulcher, brush rake and control variant without soil preparation) on the regeneration of Scots pine (*Pinus sylvestris* L.) in the conditions of natural pine sites (Northern Bohemia). Seedling numbers, heights and crown widths were recorded on transects representing all combinations of stand densities treatments and soil preparations variants three years after the silvicultural operations. Seed traps were installed to determine the numbers of seeds from 2016 to 2018. Soil water potential (Delmhorst Instrument Company) and soil temperature (sensor Pt1000 A–class sensors, EMS Brno) were measured. All variants of soil preparation had higher numbers of seedlings compared to control variant. When comparing stand densities, total numbers of seedlings were the highest in stand density 0.4 (cutter 32,402 ± 34,208 S.D. ind.ha^{-1}; mulcher 26,832 ± 24,088 S.D. ind.ha^{-1}; rake 24,496 ± 22,913 S.D. ind.ha^{-1}). This stand density was also beneficial with respect to seed numbers, seedling characteristics and soil moisture and temperature characteristics. We conclude that shelterwood regeneration on natural Scots pine sites is promising silvicultural approach and may become an important tool in mitigating negative effects of climate change in the future.

Key words: stand density, soil preparation, fructification, natural regeneration, soil moisture, soil temperature

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1. Introduction

Scots pine (*Pinus sylvestris* L.) is the main tree species in half of Europe's forests (Sharma et al. 2017; Lundqvist et al. 2019). Its natural occurrence is associated with dry and/or poor sites with limited competition of climax tree species (Linder 1997; Musil & Hamerník 2007; Vacek et al. 2010, 2019). Although pine is mainly considered resistant to limited water availability, pine stands all across Europe have been massively damaged by repeated drought events in recent years (Merlin et al. 2015; Vacek et al. 2016, 2017; Gao et al. 2017; Buras et al. 2018). Drought is probably the primary factor of damage that triggers the infestation with bark beetles and fungal diseases (Hódar & Zamora 2004; Dobbertin et al. 2007; Wermelinger et al. 2008) or mistletoe (*Viscum album* L.) (Rigling et al. 2010; Zweifel et al. 2012; Mutlu et al. 2016). Mature stands with homogenous structure and often of unsuitable ecotype are mostly affected (Bottero & Vacchiano 2015, van Halder et al. 2019).

Basic tool for increasing the adaptability of forest stands to changing environmental conditions is their spatial structuring (Schütz 2002; del Río Gaztelurrutia et al. 2017; Ameztegui et al. 2017) and species mixing (Czerepko 2004; Pretzsch et al. 2013; Zeller et al. 2017; Vacek et al. 2019b). It is expected that such stands have greater resistance to biotic and abiotic disturbances, thus these silvicultural approaches can also become an effective tool to mitigate climate change due to the increased carbon storage (Ercanli 2018).

However, on natural pine sites the option of species mixing is very limited and the main approach remains spatial structuring (Montero et al. 2001; Gaudio et al. 2011a; Kojola et al. 2012; Spathelf & Ammer 2015). Besides different thinning regimes in young stands that mostly create structural heterogeneity within one cohort of trees, in mature stands shelterwood regeneration creates temporal stand structures that rest on two partially

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**Corresponding author. Jakub Brichta, e-mail: brichtaj@fld.czu.cz* overlapping tree generations. In addition to the common clear-cuts with or without retained seed-trees, different forms of shelterwood management in different parts of Scots pine natural range are in the scope of silvicultural research (Montero et al. 2001; Gaudio et al. 2011a; Kojola et al. 2012; Spathelf & Ammer 2015; Bílek et al. 2016; Tullus et al. 2018; Lundqvist et al. 2019). These alternative methods include uniform large-scale shelterwood as well as small-scale management approaches that result into multi-layered Scots pine stands.

Shelterwood regeneration is a common management practice in Scandinavian countries (Hyppönen et al. 2013; Lundqvist et al. 2019), Germany (Spathelf et al. 2015; Drössler et al. 2017), Poland (Bielak et al. 2014; Aleksandrowicz-Trzcińska et al. 2017), and also in some parts of the Czech Republic (Bílek et al. 2017; 2018). It is often stated that shelterwood regeneration is more favourable with respect to microsite characteristics (Montero et al. 2001; Matías & Jump 2012; Aleksandrowicz-Trzcińska et al. 2017; Vítámvás et al. 2019) and compared to clear-cut area, lower light intensity can lead to higher quality of pine regeneration (Pardos 2017; Schönfelder et al. 2017, 2018; Lundqvist et al. 2019). In the context of climate change, the crucial question is how temperature and moisture characteristics of the microsite can be improved with silvicultural approaches and which treatment is leading to the success of natural regeneration.

Although sporadic studies referring to ecological plasticity of pine suggest very strong competitiveness of pine regeneration (Picon-Cochard et al. 2006), it is distinctly more sensitive to root competition compared to shade tolerant species (Erefur et al. 2008) and pine seedlings usually do not succeed in dense herbal cover of blueberry (*Vaccinium myrtillus* L.) or heather (*Calluna vulgaris* [L.] Hull.) (Hyppönen et al. 2013). Thus, on natural pine sites it is not only necessary to decrease the stand density, but also to disturb the soil surface in order to promote natural regeneration (Varmola et al. 2004; Saursaunet & Matrisen 2018).

In natural conditions the competition of ground vegetation if often temporarily supressed by fires (Engelmark et al. 1994; Corace 2009). In the absence of these natural processes that would stimulate pulses of natural regeneration, in managed stands soil preparation using different techniques is necessary (Ackzell 1993; Barbeito et al. 2011; Aleksandrowicz-Trzcińska et al. 2014; Saursaunet & Matrisen 2018). They should lead to higher density of natural regeneration and higher vitality of individual seedlings (Karlsson & Örlander 2000). However, also some risks must be considered. High soil temperature without shading of herb cover (Bedford & Sutton 2000; Oleskog & Sahlen 2000), loss of nutrients from exposed soil (Piirainen et al. 2007) or freezing of root systems of regeneration individuals (Chantal et al. 2003) are among the most important. Ploughing and soil milling are the most common techniques of forest soil preparation. However, as these techniques are expensive, it is also common to use disc harrow or brush rakes instead (Posmetyev et al. 2016).

The aim of the study was to evaluate the influence of clear-cutting, basal area reduction and soil preparation on the regeneration of Scots pine in the conditions of natural pine sites three years after the silvicultural operations. To assess the feasibility of shelterwood regeneration in given conditions we hypothesized that:

1) Number of seeds after seed fall increases with stand density and is the lowest in clear-cut.
2) Seedling numbers are influenced by parent stand density and soil preparation positively.
3) Seedling parameters (height and crown width) are influenced by parent stand density negatively, while the effect of soil preparation is positive.
4) Higher soil moisture and lower temperature are associated with higher stand density, the least favourable treatment with respect to moisture and temperature is the clear-cut.

2. Material and methods

2.1. Study site and experimental design

The area of interest is located near to the municipality Doksy. It is characterized by a large area of natural Scots pine sites. Forest stands in study are managed by Military Forests and Farms of the Czech Republic, state enterprise. The entire study locality (N 50°33.77548’, E 14°43.49143’) is in a flat terrain, at 300 m a.s.l. Parent rock is sandstone, with dominant soil type Arenic Podzol (Soil map 2020). Mean annual air temperature is 7.3 °C and average maximum temperature is 31.5 °C. Mean annual precipitation is 635 mm (Tolasz 2007).

The experimental plots were established for monitoring of natural regeneration success of Scots pine in different stand densities and soil preparation techniques. For this purpose, the parent stand was divided into 4 sections of different harvest intensities (treatments) with resulting four stand densities: clear-cut, 0.4, 0.6 and 0.8 (Table 1). Harvests were conducted in February 2016. Methodology of parent stand inventory is given in Abdollahnejad et al. (2019).

Each of the 4 treatments had a rectangle design of approx. 60 × 250 m. In each of these treatments of stand density, four variants (31.25 m × 60 m) of soil preparation techniques were performed in two repetitions. Variants of soil preparations were following: 1) Milling cutter KSH 700 with treatment in stripes of approx. 1 m distance (Milling cutter); 2) Forestry mulcher Meri Crusher 1.8 ST with full-area character of preparation (Forestry mulcher); 3) Brush rake SH 01 (Brush rake); 4) Control variant without soil preparation (Control) (Fig. 1).

Table 1. Summary stand characteristics for individual treatments of basal area reduction derived from field inventory.

Stand density after harvest	Stand density before harvest	Mean G±S.D. [$m^2\ ha^{-1}$]	Mean V±S.D. [$m^3\ ha^{-1}$]	Mean N±S.D. [$ind.ha^{-1}$]	Harvest intensity [%]
Clear-cut	0.8	0.0 ± 0.0	0.0 ± 0.0	0 ± 0	100
0.4	0.8	10.8 ± 4.0	123.5 ± 58.5	121 ± 34	55
0.6	0.8	17.8 ± 4.2	195.2 ± 34.1	234 ± 59	30
0.8	0.9	24.4 ± 3.3	258.3 ± 47.3	323 ± 45	20

Note: Stand density – calculated as share of measured stand volume per hectare (before and after the harvest) and the theoretical full stand volume per hectare derived from yield tables; G – basal area ($m^2\ ha^{-1}$) after harvest; G – basal areal standard deviation ($m^2\ ha^{-1}$); V – wood volume ($m^3\ ha^{-1}$) after harvest; V – wood volume after harvest standard deviation ($m^2\ ha^{-1}$); N ($ind.ha^{-1}$) – number of trees per 1 ha after harvest; Harvest intensity (%) – calculated as percentage of harvest on the measured stand volume before harvest; S.D. – standard deviation. Wood volumes were calculated according to Petráš & Pajtík (1991).

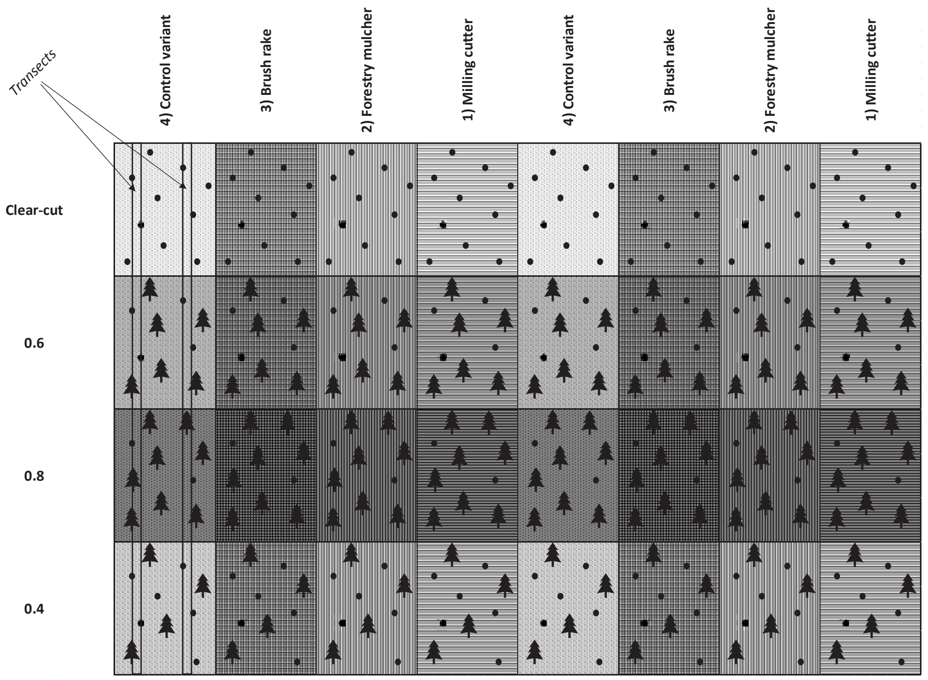

Fig. 1. Design of the experimental plot: different treatments of stand density (clear-cut; 0.4; 0.6; 0.8) and different variants of soil preparation: 1) Milling cutter; 2) Forestry mulcher; 3) Brush rake; 4) Control).

2.2. Data collection

For each treatments of stand density and variant of soil preparation technique, two transects covering 2 × 60 m^2 each were established (n = 64) (Fig. 1). The transect was divided into squares of 2 × 2 m^2, for each square plot all seedlings according to species were registered. For seedlings higher than 10 cm, total height and crown width were recorded (accuracy 1 cm). All measurements were done in autumn 2018 three years after the harvest operation and soil preparation treatment.

After the harvest in 2016 in each treatment of stand density 7 seed traps (size 1 × 1 m^2) were placed (n = 28). Traps were installed permanently from 2016 to 2018, each trap was placed at a boundary of two variants of soil preparations always in the middle section of each stand density treatment. Seeds were counted always in November after the end of the vegetation period.

Water potential measurement was carried out using absorption GB–2 gypsum sensor blocks (Delmhorst Instrument Company) connected to data loggers MicroLog SP3 (EMS Brno). Temperature sensor (Pt1000 A-class sensors, EMS Brno) was connected to the datalogger. The gypsum sensor blocks were placed 7 – 9 cm under the soil surface; for one stand density always 4 sensors were installed in the central part of the treatment to avoid the effect of stand edges (n = 16). The soil temperature (depth 7 – 9 cm) was measured together with the soil water potential in the same position. Interval of measurements was 1 hour.

2.3. Data analysis

The differences in seed numbers between selected stand densities in 2016 – 2018, as well as the differences in regeneration density, seedling height and crown width between selected soil preparation techniques and stand densities were analysed by Kruskal-Wallis test, as normality of data (tested by Shapiro-Wilk normality test) was violated in all cases. Subsequent multiple comparisons were conducted according to Siegel and Castellan (1988). The soil water potential and soil temperature were plotted separately for each stand density treatments, in case of soil temperature, one-week moving average curves are also shown.

All computations were performed using R (R Core Team 2019), plots were made by "ggplot2" (Wickham, 2016) and "ggpubr" (Kassambara, 2020) packages. All statistical tests were conducted at significance level $\alpha = 0.05$.

3. Results

3.1. Seed numbers

The highest mean seed numbers per 1 m^2 were registered for stand density 0.8 (58 ± 16 S.D. in 2016, respectively 37 ± 11 S.D. in 2017, respectively 41 ± 7 S.D. in 2018). Conversely, during the whole observation period the lowest mean seed numbers per 1 m^2 were consistently registered on clear-cut area (stand density 0.0) (4 ± 3 S.D. in 2016, respectively 4 ± 1 S.D. in 2017, respectively: 2 ± 2 S.D. in 2018). Stand densities 0.4 and 0.6 were intermediate in seed numbers, nevertheless statistically significant differences were registered mostly for stand density 0.0 (Fig. 2).

3.2. Regeneration density

Regeneration density for different stand densities and soil preparation techniques 3 years after the harvest is displayed in Fig. 3C. Generally, all variants of soil preparation had higher numbers of seedlings compared to control variant. Total numbers of seedlings were the highest in stand density 0.4. The highest mean total number of seedlings was registered for following combinations of stand density treatment and soil preparation variant: 0.4 – cutter (32,402 ± 34,208 S.D. ind. ha^{-1}); 0.4 – mulcher (26,832 ± 24,088 S.D. ind.ha^{-1}) and 0.4 – rake (24,496 ± 22,913 S.D. ind.ha^{-1}). Conversely, the lowest mean total number of seedlings was registered for the variant 0.5 – control (5,972 ± 12,897 S.D. ind.ha^{-1}) and 0.7 – control (5,960 ± 9,047 S.D. ind.ha^{-1}).

Number of seedlings lower than 10 cm (Fig. 3A) had the highest density in combinations 0.4 – cutter (19,824 ± 29,340 S.D. ind.ha^{-1}) followed by 0.8 – cutter (15,090 ± 18,847 S.D. ind.ha^{-1}) and 0.4 – rake (13,876 ± 17,230 S.D. ind.ha^{-1}). The worst with respect to seedlings numbers were the control variants in combination with stand density treatment 0.6 (4,037 ± 10,097 S.D.) and 0.8 (3,990 ± 7,240 S.D. ind.ha^{-1}).

Number of seedlings higher than 10 cm (Fig. 3B) was the highest in combination clear-cut – cutter (14,979 ± 16,654 S.D. ind.ha^{-1}) and 0.4 – cutter (12,578 ± 14,894 S.D. ind.ha^{-1}). The control variant showed the lowest values in all cases of stand density treatments (0.0: 4,980 ± 4,656 S.D. ind.ha^{-1}; 0.3: 3,780 ± 5,036 S.D. ind.ha^{-1}; 0.6: 1,898 ± 3,758 S.D. ind.ha^{-1}; 0.8: 1,970 ± 3,682 S.D. ind.ha^{-1}).

3.3. Seedlings characteristics

Superior mean heights of seedlings were observed in stand density treatments clear-cut (stand density 0.0) (20.0 – 21.9 cm) and 0.4 (18.0 – 21.3 cm). In all soil preparation variants, there were significantly lower seedlings in stand densities 0.6 (13.7 – 18.3 cm) and 0.8 (13.5 – 15.7 cm) (Fig. 4). The stand density treatment 0.0 was also superior with respect to crown widths of individual seedlings (19.6 – 21.3 cm) (Fig. 5).

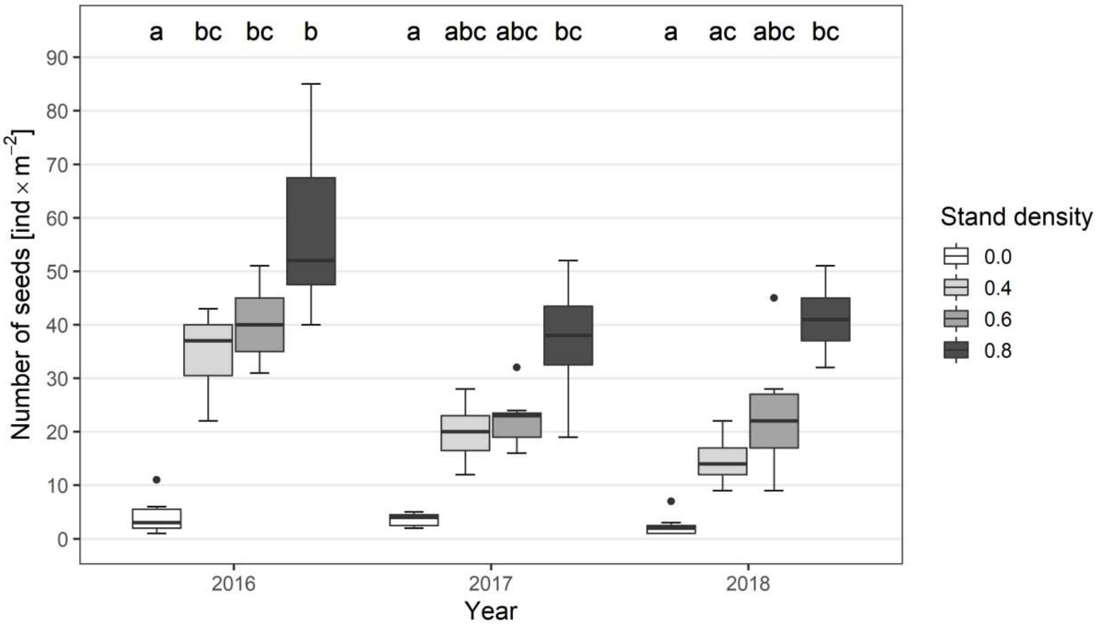

Fig. 2. Number of seeds per 1 m^2 in different stand densities. Different letters indicate significant differences among groups ($p < 0.05$); values represent medians of number of seeds, upper and lower quartile.

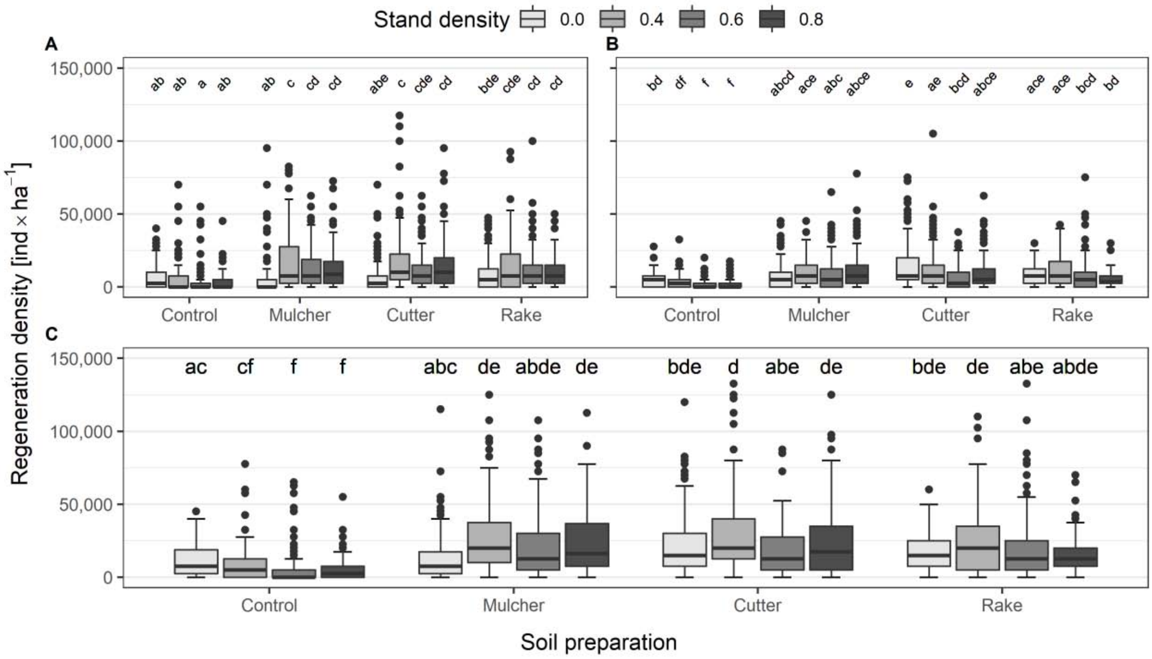

Fig. 3. Numbers of seedlings per 1 ha for different stand densities and soil preparation techniques: Number of seedlings lower than 10 cm (A); Number of seedlings higher than 10 cm (B); and total number of seedlings (C). Different letters indicate significant differences between groups ($p < 0.05$); values represent median, upper and lower quartile.

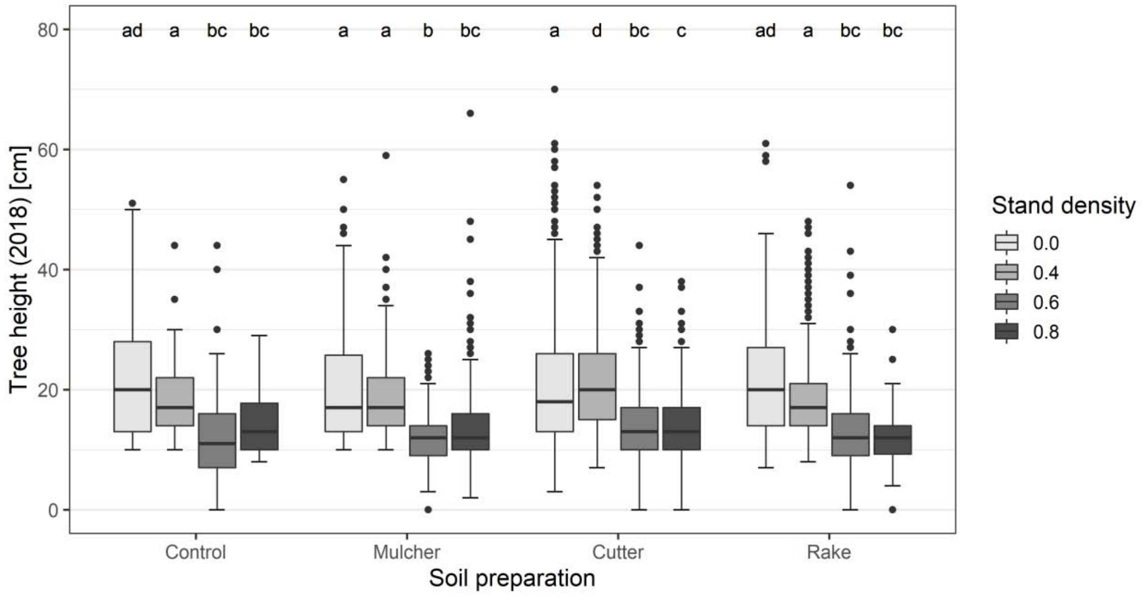

Fig. 4. Height of seedlings in different variants of soil preparation and stand density (individuals higher than 10 cm; year 2018). Different letters indicate significant differences between groups ($p < 0.05$); values represent median, upper and lower quartile.

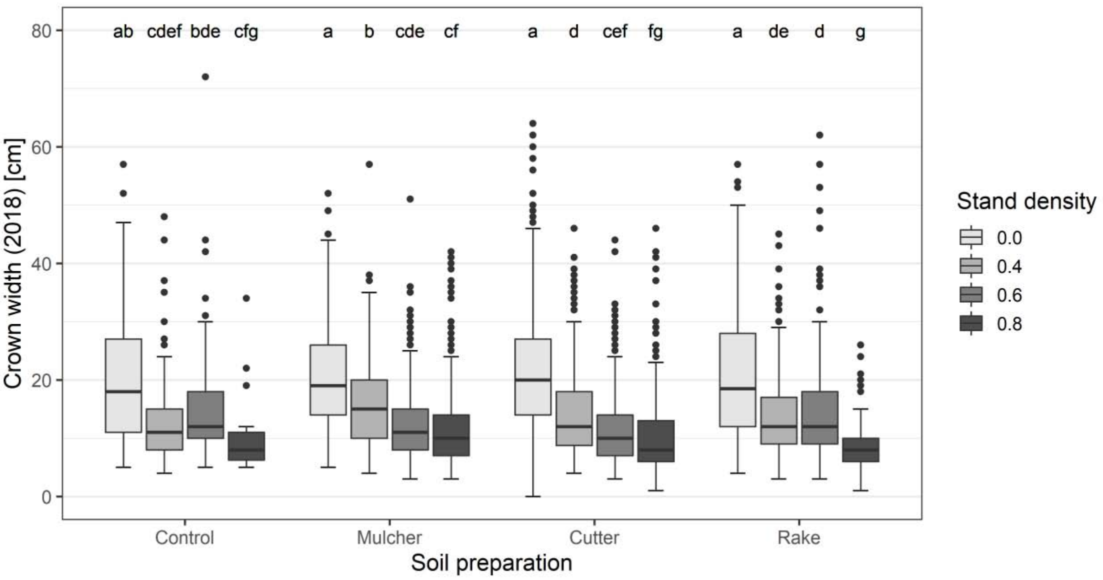

Fig. 5. Crown width of seedlings in different variants of soil preparation and stand density (individuals higher than 10 cm; year 2018); different letters indicate significant differences between groups ($p < 0.05$); values represent median, upper and lower quartile.

3.4. Soil water potential and soil temperature

Early increase of soil water potential was observed in stand densities 0.8 and 0.6, whereas on clear-cut (density 0.0) and in stand density 0.4 the soil water potential increased with a delay and the drought period was less pronounced (Fig. 6). For example, the period with values of soil water potential above 5 bar lasted in stand density 0.8 from late June to the end of October, whereas in stand density 0.4 it lasted from early August to September.

Figure 7 shows temperatures 7 – 9 cm under the soil surface from June 2018 to November 2019. The high-

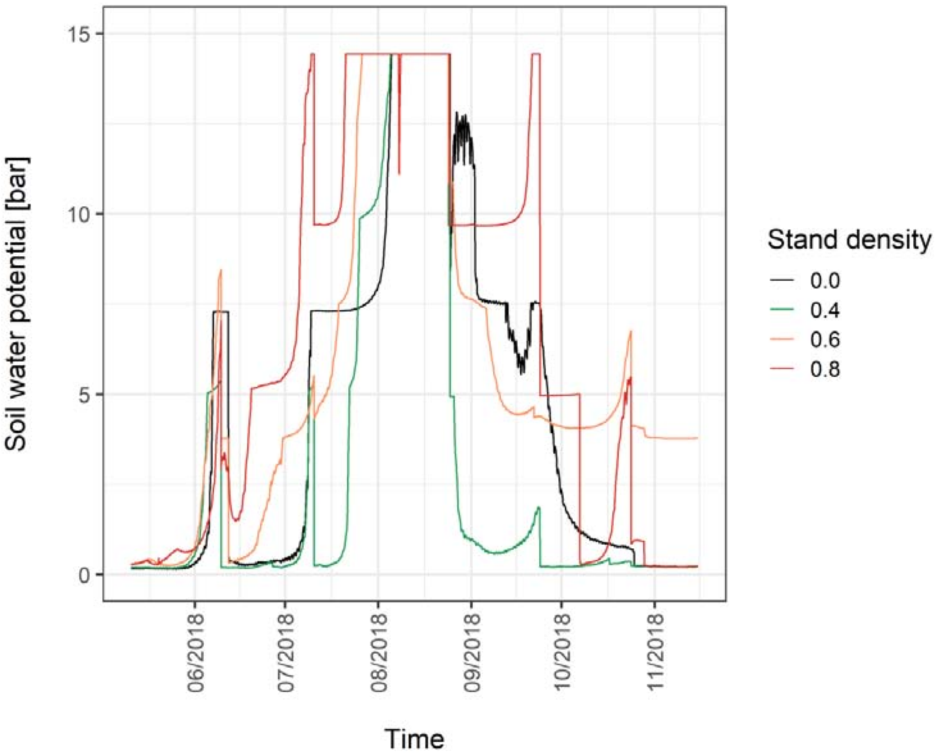

Fig. 6. Soil water potential in vegetation period 2018 for different stand densities.

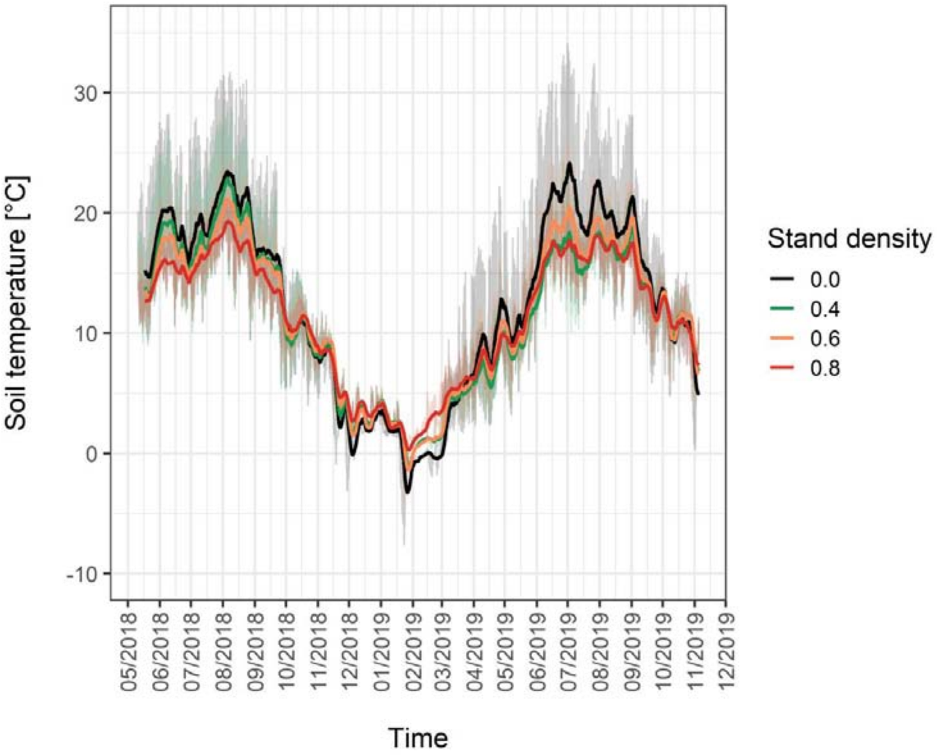

Fig. 7. Soil temperature in individual months in 2018/2019 for different stand density plots. The original values (in background) are for better readability represented by one-week moving average curves.

est temperatures in summer as well as the lowest temperatures in winter were measured on clear-cut (stand density 0.0). To the contrary, the lowest amplitude of soil temperatures was observed in the case of highest stand density 0.8. These values of stand densities 0.4 and 0.6 were between the two mentioned treatments. Generally, the absolute differences in mean temperatures between clear-cut and stand density 0.8 were the highest in July and August, intermediate in February, March and low in the remaining months.

4. Discussion

4.1. Seed numbers

In accordance with our first hypotheses, the highest seed numbers were in all years (2016 – 2018) recorded in the highest stand density, whereas on the clear-cut area the seed fall was the lowest. Differences between the particular stand densities from 0.4 to 0.8 were apparent, yet not statistically significant. Since most seeds are generally in close proximity to seed trees (Kuuluvainen & Pukkala 1989), the high density of these mature individuals guarantees high seed density over the entire area. It is important to note that seed traps were placed always in the middle section of each stand density, and particularly in the case of clear-cut, higher seed numbers can be expected with decreasing distance to adjacent mature stands.

As released trees increase the diameter increment (Zdors & Donis 2017; Brichta et al. 2019) and crown volume (Spathelf & Ammer 2015), they probably also increase masting ability (Mukassabi etl al. 2012). As these processes are gradual and depend on the responsiveness of individual trees, with increasing time since seed harvest smaller differences in seed numbers between lower and higher stand densities can be expected. Similar results were reported by Parker et al. (2013), who stated that total seed production expressed on a per hectare and unit pine basal area basis did not differ by harvest treatment. The same authors conclude that the seed cut of the uniform shelterwood system applied to the second growth of white pine stands is unlikely to adversely affect white pine (*Pinus strobus* L.) seed production, seed quality, or potential for natural regeneration during mast seeding events.

4.2. Regeneration density

We partly reject the second hypotheses, as the most favourable conditions for seedling establishment and resulting seedling numbers 3 three years after the seed cut were reached in stand density 0.4. Parent stand density itself cannot be simply regarded as a factor with only positive effect on regeneration density. Mature trees represent competition for resources among themselves but they also have adverse effect on regeneration individuals in the understory (Wiedemann 1925; Valkonen et al. 2002; Sánchez-Salguero et al. 2015). This negative effect may not be apparent immediately after the seed fall, but will increase with the growth of juvenile trees (Belend et al. 2000). Similar conclusions have been found for example by Pitt et al. (2011) in white pine stands. To the contrary, lower regeneration density on the clear-cut results probably from lower initial seed numbers as described in the first part.

Beland et al. (2000) used a similar experiment design with soil preparation by disc cutter and division of the stand into several treatments of stand density (200 stems ha^{-1}; 160 stems ha^{-1}; clear-cut area of approx. 2.5 ha). While in our study the highest seedling numbers (32,402 ha^{-1}) were recorded in the stand density 0.4 (approx. 121 stems 1 ha^{-1}), in their study for low stand density (160 stems ha^{-1}) 53,000 seedlings ha^{-1} were reported. Even higher numbers of regeneration were then found in the stand with the highest density (200 stems ha^{-1}; 90,000 seedlings ha^{-1}). Their study also found a very low number of seedlings on clear-cut area (3,700 seedlings ha^{-1}) with cutter soil preparation, specifically this value for total number of seedlings is significantly lower than our clear-cut area values on all soil preparation variants (cutter: 21,960; mulcher: 13,550; brush rake: 17,110; control: 11,500 seedlings ha^{-1}). However, our results may be affected by higher temperatures and distinct moisture deficit in 2018 (ČHMÚ 2019) resulting probably in higher seedling mortality during the vegetation period. Although pine is mainly considered resistant to limited water availability, in recent years pine stands all across Europe have been massively damaged by repeated drought events (Merlin et al. 2015; Vacek et al. 2016, 2017; Gao et al. 2017; Buras et al. 2018). Therefore, we assume that even in the local natural pine habitats, the seedlings were damaged by drought. Relatively lower stand densities are reported also from Turkey (Kara & Topaçoğlu 2018), where 5 years after reducing the parent stand density to 50%, average seedling numbers amounted to 7,000 ha^{-1}. Aleksandrowicz-Trzcińska et al. (2014) reported 9,190 seedlings ha^{-1} on plots after the scarification by forest plough on clear-cut of 2.5 ha.

In accordance with the second hypotheses, a positive effect of soil preparation on the seedlings numbers in all treatments of stand densities was confirmed. Similar findings were reported by many authors (Örlander et al. 1990; Karlsson & Örlander 2000; Hille & Ouden 2004; Barbeito et al. 2011). Besides immediate regeneration success, mainly deeper soil preparation also significantly lowers the C/N ratio, increases the amount of P, Mg and S (Örlander et al.1996) and supresses the competition of ground vegetation (Hille & Ouden 2004; Nilsson et al. 2006; Gaudio et al. 2011b; Hyppönen et al. 2013). Deeper soil preparation is also recommended by Posmetyev et al. (2016), who adds that not only does it achieve good results in terms of natural regeneration density, but is also economically effective. Despite this, there are large differences in preferences and value of soil preparation according to a particular habitat (Ackzell 1993; Ammer et al. 2011; Aleksandrowicz-Trzcińska et al. 2014; Saursaunet & Matrisen 2018).

4.3. Seedling characteristics

Based on our results, we confirm the hypotheses that seedling height and crown width are influenced by parent stand density negatively, nevertheless the effect of soil preparation on these regeneration parameters was not apparent.

Superior mean heights of seedlings in clear-cut and stand density 0.4 are mainly related to higher resources availability, as was confirmed also in numerous controlled experiments. Vítámvás et al. (2019) reported for 1–year-old seedlings that root length, shoot dry mass and root dry mass were the highest in full light conditions, however the differences in shoot length were not so pronounced. Riikonen et al. (2016) documented high ability of pine seedlings to respond to release cut by primarily increasing the crown width. In older literature, there are many proofs about the superior growth of young plants growing in larger distances to seed trees (eg. Wiedemann 1925).

We were not able to confirm positive effect of soil preparation on seedling growth, nevertheless Nilsson et al. (2006) and Gaudio et al. (2011b) showed superior height growth of regeneration individuals after soil scarification. The reason is better soil chemical and mechanical properties and lower competition of herbaceous vegetation (Karlsson & Örlander 2000; Varmola et al. 2004; Saursaunet & Matrisen 2018). To the contrary, Pierik & De Wit (2014) related high competition of herbaceous vegetation to superior height growth, but at the expense of crown width and chlorophyll content in assimilation organs.

4.4. Soil water potential and soil temperature

Based on our results, we reject the hypotheses that more favourable soil moisture and temperature characteristics are associated with higher stand density, and that the least favourable treatment with respect to moisture and temperature is the clear-cut. Although the lowest amplitude of soil temperatures was observed in the case of highest stand density 0.8, and the highest temperatures in summer as well as the lowest temperatures in winter were observed on the clear-cut, higher stand densities (0.6 and 0.8) were related to lower soil moisture and possibly higher risk of drought stress for seedlings. Similar results presented several authors (Limousin et al. 2008; del Campo et al. 2014, Gebhardt et al. 2014) who stated that the release cut had a positive effect on the amount of water in the transplantation streams of mature trees, but also the soil water potential itself. The reason for the increase in the amount of water in the stand is mainly the lower rate of mature trees crown interception (Limousin et al. 2008), but also their lesser competition and evapotranspiration (Kellomäki et al. 1997; Giuggiolia et al. 2013). Del Campo et al. (2014) also stated that a strong stand density reduction increased the amount of water in the soil by up to 20 – 40%.

5. Conclusion

Based on our research, we conclude that the best outcome of natural regeneration was achieved after seed cut decreasing the stand density to 0.4. This was manifested by sufficient seed numbers, highest regeneration density, superior seedling characteristics and positive effect on soil moisture and temperature characteristics. The effect of soil preparation of any type was crucial for the regeneration success, but the best results were documented for treatment with milling cutter (3–fold increase in total mean numbers of seedlings compared to control variant versus 2.6–fold increase for forestry mulcher and 2.4–fold increase for brush rake). We conclude that shelterwood regeneration with initial stand density reduction to 0.4 on natural Scots pine sites represents justified alternative to clear-cutting and may in the future become an important tool in mitigating negative effects of climate change.

Acknowledgment

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ORIGINAL PAPER

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; Bošeľa 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 & Hejcman 2012; Bulušek et al. 2016; Šimůnek et al. 2019). Bošeľa 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 al. 2012; Bošeľa 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). Bošeľa 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 protection 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 (subregion 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 (*Popolus 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-

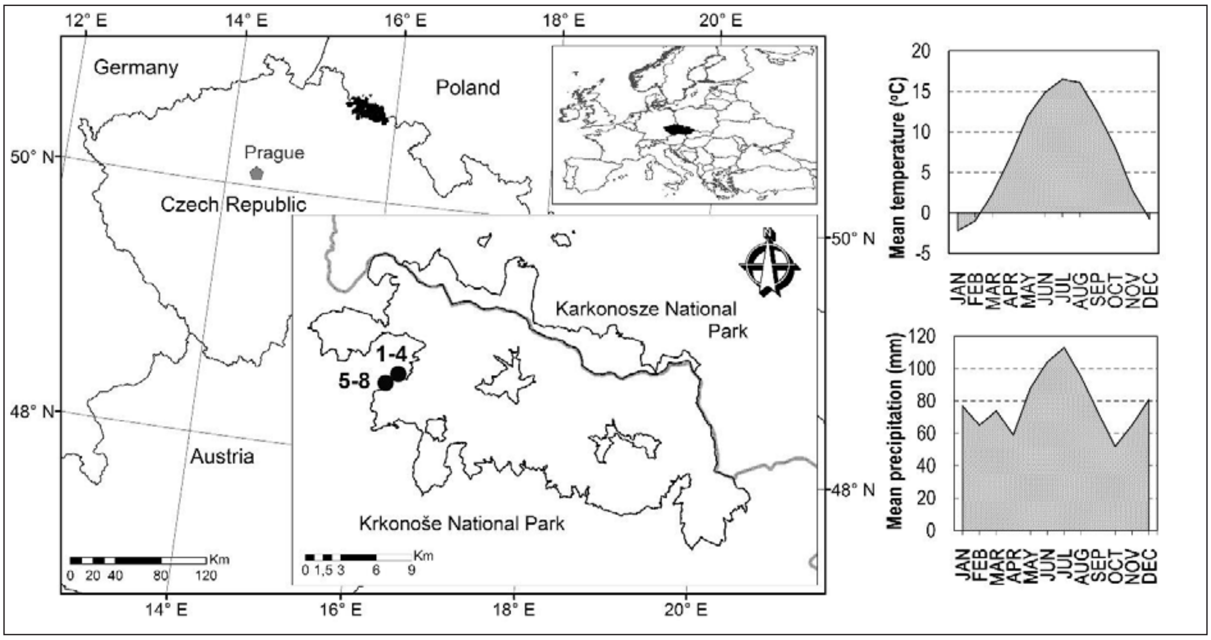

Fig. 1. Localization of mixed stands on experimental research plots Končiny 1 – 8 in Krkonoše Mountains National Park (protection zone) and mean monthly climatic values (1960 – 2018); map was made in software ArcGIS 10 (Esri).

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 created 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.

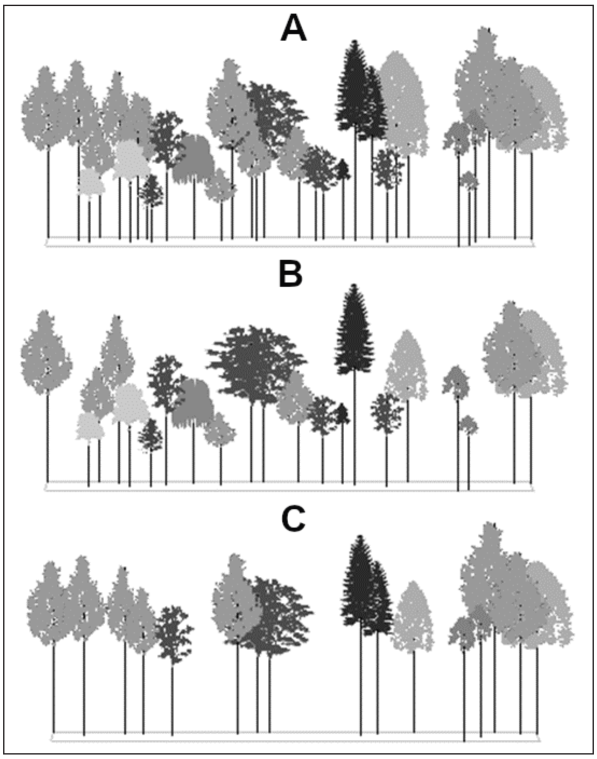

Fig. 2. Vertical schema of mixed forest stand A) before thinning, B) after application of the climate change adaptation thinning and C) thinning from below; figure was made in software SIBYLA Triquetra 10 (Fabrika & Pretzsch, TUZVO).

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 (Füldner 1995), crown differentiation and total stand diversity (Jaehne & Dohrenbusch 1997; Table 2). Stand volume was calculated according to Petráš & Pajtík (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)	Pretzsch (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 m^2 ha^{-1} and stand volume reached 127 – 377 m^3 ha^{-1}. After thinning interventions, stand volume decreased from 185 to 159 m^3 ha^{-1} in variant B (− 14%) and to 175 m^3 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 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).

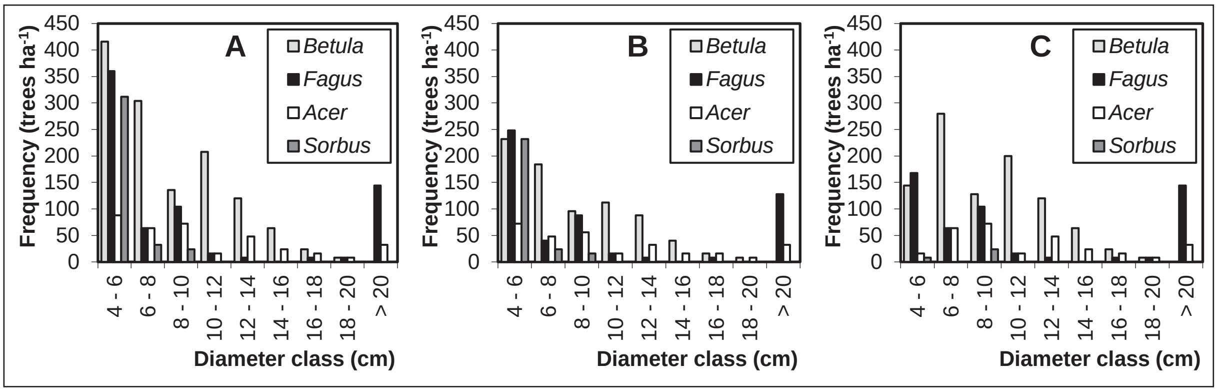

Fig. 3. Histogram of tree layer diameter structure according to main tree species in A) control variant (before thinning), B) after applied climate change adaptation thinning and C) after simulative thinning from below.

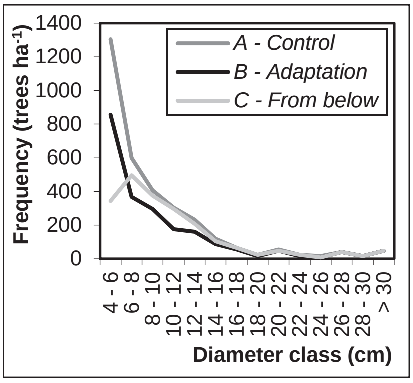

Fig. 4. Summary histogram of complete tree layer diameter structure in A) control variant (before thinning), B) after applied climate change adaptation thinning and C) after simulative thinning from below.

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

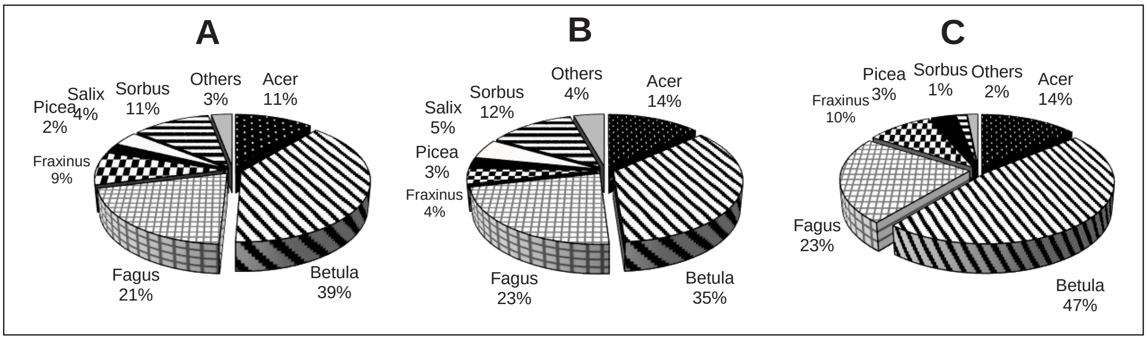

Fig. 5. Tree species composition in A) control variant (before thinning), B) after applied climate change adaptation thinning and C) after simulative thinning from below; others tree species include: *Prunus avium*, *Abies alba* and *Populus tremula*.

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

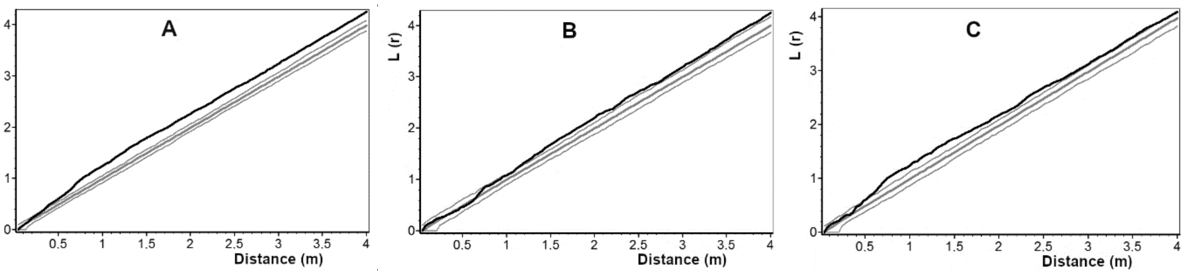

Fig. 6. Spatial pattern of tree layer on experimental research plots 1 – 8 expressed by *L*–function in A) control variant (before thinning), B) after applied climate change adaptation thinning and C) after simulative thinning from below; the black line represents the *L*–function for real distances 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.

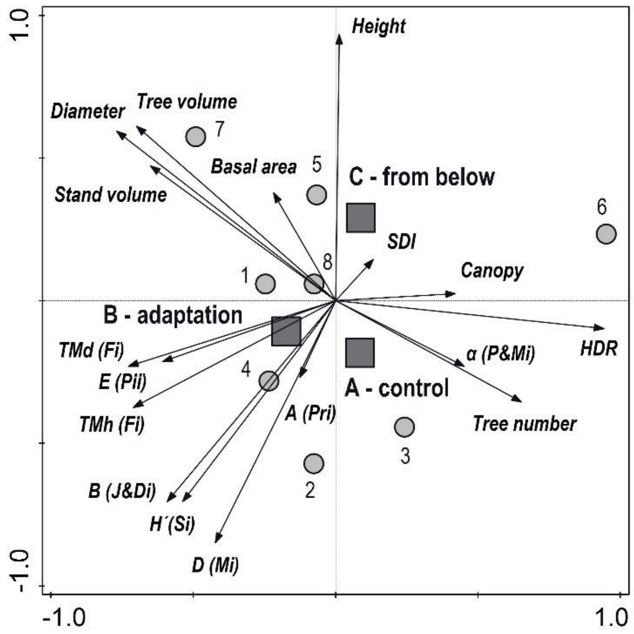

Fig. 7. Ordination diagram showing results of the PCA of relationships between variants of thinning (A – control before thinning, B – adaptation thinning, C – thinning from below), stand characteristics (Stand volume, Tree volume, Basal area, DBH, HDR, Height, Canopy, Tree number) and structural diversity (*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 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; Mikulenka 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 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 m^3 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; Hédl 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, Bošeľa 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 an 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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