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ORIGINAL PAPER

Conversion of Norway spruce forests in the face of climate change: a case study in Central Europe

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Abstract Steadily increasing damage to Norway spruce forests in Europe has caused researchers and managers to consider whether these forests can be converted to more stable ecosystems. In a central European mountain region, we investigated whether management systems (MSs) specified by regional stakeholders provide sound alternatives to the currently applied management. We used the forest model Sibyla to explore whether the tested MSs differ in their sensitivity to climate change in terms of altered biomass production, stand structure, forest damage, and financial outcome. The tested MSs were no-management (NM), currently applied management (BAU), and management based on the preferences of forest managers (FM) or on the preferences of other stakeholders (OSH). With NM, spruce remained dominant during the simulation period 2010–2100, and the rate of damage significantly increased. Spruce also remained dominant with FM, while the abundance of non-spruce species significantly increased with BAU and OSH. The rate of salvage logging converged at 50% of the total harvest for all MSs up to 2050. Climate change reduced biomass production (−15%) with all MSs but had a negligible effect on biodiversity indicators. The average initial value of the simulated stands was 20,000 € ha^{-1} and the nominal value in 2100 was between 1900 and 10,900 € ha^{-1}. The Net Present Value calculated with the 2% interest rate was negative during the whole simulation period (−5600 to −18,500 € ha^{-1} in 2100). Effect of climate change on all financial indicators was negative. Our findings indicate that secondary spruce forests are highly vulnerable and that the systems proposed by both forest managers and other regional stakeholders failed to significantly reduce forest damage and stabilize forest production.

Keywords Biomass production · Forest diversity · Net present value · Forest modelling · Alternative forest management

Introduction

Man-made Norway spruce (*Picea abies* L. Karst) forests occupy a substantial part of the forest land in Central Europe (Spiecker et al. 2004). Although these forests are considered indispensable because they generate softwood timber, maintain biodiversity, moderate gravitation hazards, and support recreation, damage to these forests has been steadily increasing in recent decades (Hlásny and Sitková 2010; Briner et al. 2013), and their long-term sustainability has been repeatedly questioned (e.g. Grodzki 2010; Hlásny and Turčáni 2013; Hlásny et al. 2015a). Conversion of spruce stands growing on unsuitable sites to broadleaved or mixed stands has been one of the central

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issues of European silviculture in recent decades (Spiecker et al. 2004; Löf et al. 2010). The alternative management strategies usually include avoidance of clearcutting and of intense site-preparation techniques, fostering stand structure and tree species diversity, or promotion of natural regeneration (Puettmann et al. 2015). Such practices might reduce stand susceptibility to damage and increase tree survival (Griess et al. 2012; Jönsson et al. 2012; Neuner et al. 2014), which is particularly important in regions with intensifying disturbance regimes. Interest in these alternatives has increased also because of the growing recognition of forest multifunctionality (Gustafsson et al. 2012; Briner et al. 2013; Gamfeldt et al. 2013) and the increasing desire of diverse stakeholder groups to participate in forest management decisions (Phalen 2009; O'Brien et al. 2013; Shackelford et al. 2013; Sarvašová et al. 2014; Marzano et al. 2015). Stakeholder participation (e.g. via municipality or nature conservation representatives) in forestry decision-making, which has been uncommon in traditional forestry, might be particularly beneficial because it would provide regional expertise and would increase the public acceptance of the final outcome (Beckley et al. 2005).

Alternative management strategies that would address the ecological and environmental limits of forest exploitation as well as related financial risks have been increasingly scrutinized (e.g. Seidl et al. 2007, 2014; Lindner et al. 2010; Jactel et al. 2012; Ray et al. 2014; Schelhaas et al. 2015). The transition from traditional production-oriented forest management to alternative practices has received substantial attention in developed countries (Puettmann et al. 2015) but less attention in countries with transitional economies in Central and Eastern Europe. Particularly challenging is the transition from "even-aged, single-species-oriented management" towards "close-to-nature management"; the time required for such a transition is likely to exceed the time span of a rotation period (Roessiger et al. 2011). In the case of Norway spruce forests, which are the focus of this study, the current high market demand for softwood and the uncertain options for marketing the surrogate, mostly hardwood species are additional constraints to such forest conversion.

The modification of long-term management practices is particularly challenging in the face of climate change, which might both support and counteract the conversion efforts (Hlásny et al. 2014; Zlatanov et al. 2015). In Central Europe, the complex interactions between climatic stressors and the climate-sensitive dynamics of diverse pests represent the main climate change-related threat to regional forests (Thomas et al. 2002; Lakatos and Molnár 2009; Hlásny and Turčáni 2013; Klapwijk et al. 2013), and such interactions are likely to be amplified by climate change (Seidl and Rammer 2016). Climate change is expected to induce significant financial losses related to changes in species distribution, a deterioration of growing conditions, and an increase in damage (Kirilenko and Segjo 2007). For example, a shift towards less profitable species could reduce the economic value of current forest land by 28% by the year 2100 (Hanewinkel et al. 2012). Such effects can, however, differ along environmental gradients and management types. For example, Härtl et al. (2016) found that climate change slightly increased the economic return in an Austrian mountain region but slightly decreased the economic return in a Slovak mountain region.

In the current study, we used a modelling approach to analyse alternatives to traditional forest management in a central European mountain region. This region is part of a set of study areas in European mountains that are being investigated in a unified design in the 7th EU Framework Programme Project "Advanced Multifunctional Management in European Mountain Ranges" (ARANGE). The objectives of the current study were to determine whether management systems (MSs) specified by the regional stakeholders provide sound alternatives to the currently applied management. The reasons for searching for such alternatives are mainly related to (1) the increasing damage to the regional forests, the moderation of which apparently exceeds the capacity of current management; (2) the interest in increasing the financial profits generated by the forests; and (3) the interest in strengthening forest multifunctionality. The latter reason mainly concerns support of biodiversity, which has not been adequately supported by the current timber-oriented management even though biodiversity is positively associated with the quality of many ecosystem services and reduces susceptibility to damage (e.g. Díaz et al. 2006; Griess et al. 2012). We used a regionally adopted forest model to test whether the tested MSs differ in their sensitivity to climate change in terms of altered biomass production, forest damage rate, and stand structure. A particularly important objective was to evaluate the financial outcome generated by the MSs and how such financial outcomes will be affected by climate change. The research strives to extend our understanding of the development of secondary Norway spruce forests in Central Europe under climate change and to evaluate the risks and opportunities related to management strategies preferred by the regional stakeholders.

In support of these objectives, we tested the following hypotheses: (1) in spite of a high damage rate of spruce and in spite of climate change-mediated improvement of growing conditions for non-spruce species, spruce will remain dominant in the regional species composition; (2) climate change will reduce regional biomass production; (3) MSs that reduce the rotation period and increase species diversity will experience reduced damage because of the reduced stand susceptibility to most damaging hazards (Jactel et al. 2012; Griess et al. 2012; Morin et al. 2014);

and (4) the financial outcome generated by all MSs will decrease in the future because of ongoing damage and the adverse effects of climate change on forest productivity.

Data and methods

Study region and experimental design

The research was conducted in the Goat Backs mountain region of Slovakia, which has an area of 8226 ha and an elevation range of 650–1554 m a.s.l. The region has 62% forest cover dominated by Norway spruce (84%) (Fig. 1). Air temperature during the growing season (April–September) ranges from 12 to 15 °C, and growing season precipitation ranges from 380 to 510 mm. Cambisols and Podsols prevail, while Rendzinas occur on the calcareous bedrock at the highest elevations. Timber production constitutes ca. 95% of the regional forestry economy. Game hunting and recreation are other ecosystem services, which are actively utilized by the regional communities. More information about the region can be found in Hlásny et al. (2015a).

Forest development simulations for the period 2010 to 2100 were run for 25 forest stands, which were representative of stand and site conditions in the study area based on elevation, soil type, and tree species composition (Table 1; Fig. 1). The stands were selected based on forest management plans, forest soil maps, and a digital elevation model archived in the National Forest Centre, Slovakia.

In the forest dynamics model that was used (see the next section), the initial state of all stands was defined using diameter distribution functions and height curves for each tree species. Height and diameter data were collected in a field survey conducted in 2011 in each of the preselected forest stands. Specifically, three 0.045-ha plots per ha were

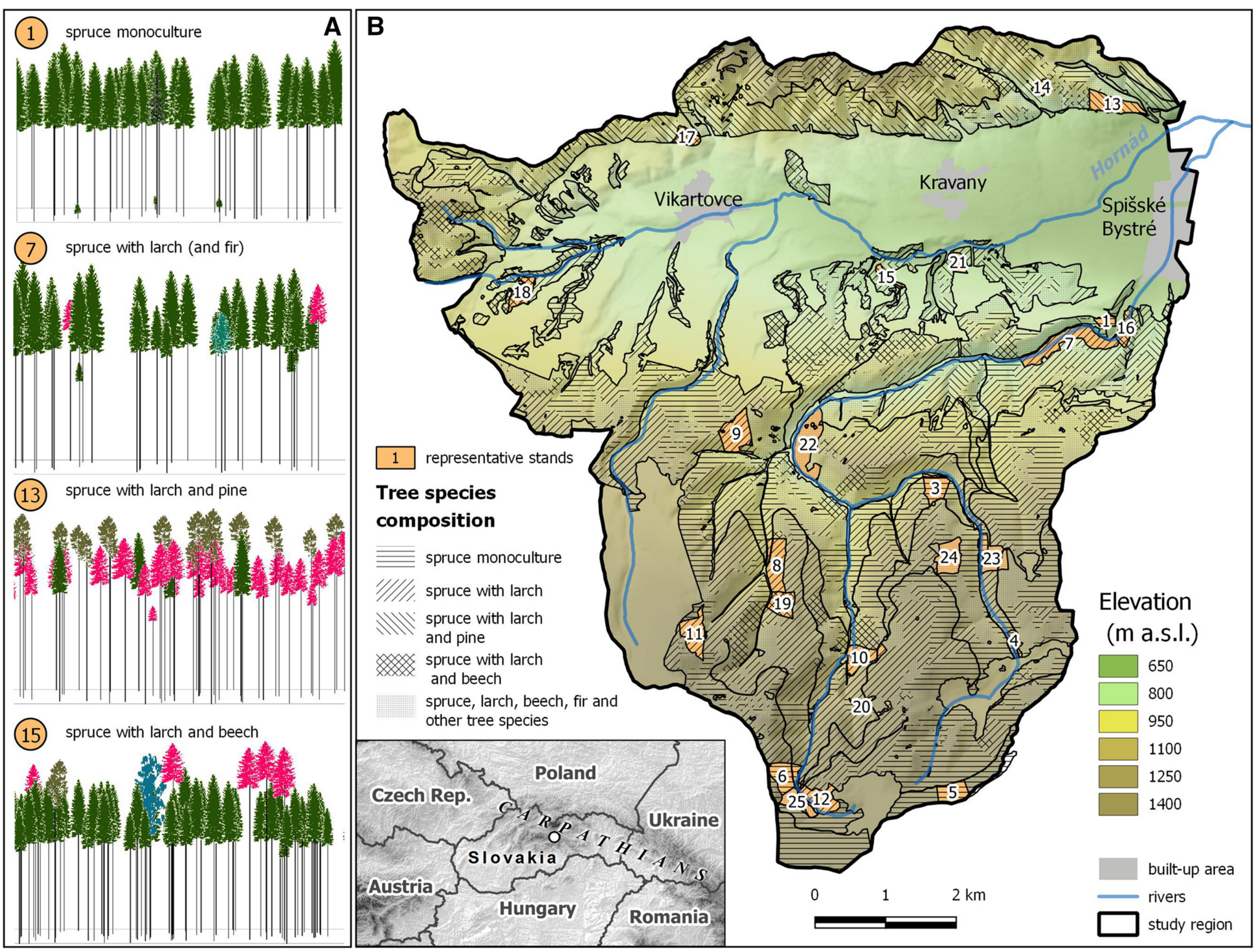


Fig. 1 Goat Backs Mts. model region. Tree species composition, elevation, and location of the 25 stands used for the simulation are indicated (stand codes are as in Table 1) (**a**). The initial state of four selected representative stands arranged on an elevational gradient is indicated as well (**b**)

Table 1 Criteria used to select the 25 representative stands for forest development simulations. Each combination of site and stand criteria is represented by a single stand

Species composition	Elevation (m a.s.l.)	Soil type	Stand ID
Spruce	600–800	Cambisol	1
	801–1100	Cambisol	2
		Podsol	3
	1101–1500	Cambisol	4
		Podsol	5
		Rendzina	6
Spruce with larch	600–800	Cambisol	7
	801–1100	Cambisol	8
		Podsol	9
	1101–1500	Cambisol	10
		Podsol	11
		Rendzina	12
Spruce with larch and pine	600–800	Cambisol	13
	801–1100	Cambisol	14
		Podsol	15
Spruce with larch, beech, and fir	600–800	Cambisol	16
	801–1100	Cambisol	17
		Podsol	18
	1101–1500	Cambisol	19
		Rendzina	20
Mixture of spruce, larch, pine, fir, beech, and maple	600–800	Cambisol	21
	801–1100	Cambisol	22
	1101–1500	Cambisol	23
		Podsol	24
		Rendzina	25

established in each stand to facilitate the calculation of the above parameters. Finally, the collected data were used to designate the virtual plots with size 200 × 12.5 m (0.25 ha). This rectangular shape was required to facilitate the simulation of harvesting operations, which are performed in parallel strips. Because information on tree positions was not available, the initial state of each plot was generated 15 times with different tree distributions while preserving the characteristic species mixture patterns observed in the stands. The results of the simulations were averaged and recalculated to per hectare units. To obtain regional estimates, the evaluated forest development indicators were weighted by the area of each of the 25 stand–site combinations occurring in the region (Table 1; Fig. 1).

The model

The simulations of forest development under four MSs and five climate scenarios were performed using the empirical forest dynamics model Sibyla (Fabrika and Ďurský 2005), the core components of which are based on the Silva model (Pretzsch et al. 2002). Sibyla is a tree-based model with a 1-year simulation step. The model has been repeatedly used in climate change impact studies in central European temperate forests by Hlásny et al. (2011, 2014, 2015a) and Härtl et al. (2016).

The model is suitable for studying the effects of climate change because several processes in the model are sensitive to climate. Tree growth is controlled by several ecological parameters, including six climate variables, which affect height and diameter increment through the modified site index (Table 2, Kahn 1994). Because species differ in their sensitivity to climate, climate change might modify competitive interactions between species and thus induce change in tree species composition [as was demonstrated by Hlásny et al. (2015a)].

Inherent tree mortality is indirectly sensitive to climate, i.e. changing the site index alters tree increment, which along with stand density and tree height (Ďurský 1997) determines the probability of tree death. If site conditions substantially worsen, inherent tree mortality might significantly affect the stand development. Contrary to inherent tree mortality, disturbance-related mortality was assumed to be insensitive to climate in the model. The mortality setting used was specific to tree species and age class, and was parameterized based on the forest disturbance records

Table 2 Projected changes in forest development drivers used in the forest dynamics model Sibyla

Driver	1980–2010	(2071–2100)–(1980–2010)	Mean and range of climate scenarios (2071–2100)
Atmospheric NO_x (ppb)	307.9	+18.9	326.8
Atmospheric CO_2 (ppm)	355.1	+69.7	424.8
Soil nutrients (0–1)[a]	0.35	–	–
Soil water content ($m^3\ m^{-3}$)[b]	0.29 (0.25–0.32)[c]	−0.08	0.21 ± 0.05
Number of days > 10° C (days)	138.0	+34.0	172.0 ± 56.0
Annual temperature amplitude (°C)	21.7	−0.8	20.9 ± 2.9
Mean temperature during veget. period IV–IX (°C)	12.2	+2.3	14.52 ± 3.8
Precipitation totals during veget. period IV–IX (mm)	518.0	−70.0	447.0 ± 364.0
de Martone aridity index (mm $°C^{-1}$)[d]	23.3	−4.9	18.4 ± 13.4

Mean and range of five climate change scenarios are given for each climate variable. The data are for 950 m a.s.l. elevation, but the variability of projected changes along the elevation gradient is low

[a] Soil nutrients were derived from soil characteristics and were not subject to any changes during the simulation period

[b] Soil moisture development was simulated individually for each stand using daily climate data and soil properties using the hydrological model ISSOP (Hlásny et al. 2015b)

[c] Range of the soil water content for the simulated stands

[d] (Precipitation totals during vegetation season)/(mean temperature during vegetation season + 10)

for the period 1998–2009 using the data from the whole of Slovakia (Hlásny et al. 2015a). The used disturbance mortality rates consider an aggregate effect of all disturbances in the region. The model uses the mortality rates derived from the national statistics to determine the mortality rate for each single tree in a stand based on species and age, and evaluates stochastically whether or not a tree dies. This approach can induce bias in mortality estimates, because damage rates derived from records on stand damage (whole or partial) are applied at the scale of a tree. This bias, however, should be reduced by averaging the simulation outputs over a number of stands and simulations, as was done here.

Regeneration is another process indirectly sensitive to climate. Regeneration density is driven by several constant tree species-specific parameters (seed production, germination rate, etc.) and by other parameters that change during stand development (stand density and site conditions, which also include climate). Climate effects on species-specific regeneration density were calculated based on the Slovak National Forest Inventory data (Merganič and Fabrika 2011). Climate change might also affect regeneration based on climate-mediated changes in the growth and inherent mortality of the mother stand. Both natural and artificial (i.e. planting) regeneration modules were activated during the simulation; MS-specific artificial regeneration rates are indicated in Supplementary Material C.

Management is implemented in the model in terms of stand age, diameter classes, and proportions of stocking volume to be extracted in individual years of the simulation period. A number of thinning and harvesting techniques, which are commonly applied in Central Europe, can be activated.

Forest development drivers

Outputs of five regional climate models (which were driven by the emission scenario A1B) and a statistically generated stable climate (corresponding to the period 1980–2010 and referred to as the baseline climate) were used to drive the forest development simulations. For details of the downscaling approach, see Bugmann et al. (2017) and Supplementary Material A. The scenarios are referred to as c1–c5 and are ordered according to the magnitude of the projected air temperature increase. The IPCC SRES A1B scenario (Nakicenovic and Swart 2000) was used to describe the future development of CO_2 emissions. The evolution of nitrogen deposition was based on Dentener (2006). The changes in forest development drivers used in the model between the period 2071–2100 and 1980–2010 are indicated in Table 2.

Evaluated management systems

Four MSs were investigated. Two of the MSs (FM and OSH) represent alternatives to the currently applied MS (BAU) and were designed based on communication with the Regional Stakeholder Panel (RSP), which was established in the 7FP EU Project ARANGE. The RSP consisted of eight persons representing public authorities, non-governmental organizations, and community members, and six

Table 3 Basic characteristics of the investigated forest management systems

Management system	Management type	Target age structure	Rotation period (years)	Final cut	Final cut area (ha)	Planting	Target tree species
NM	–	Uneven	–	–	–	–	–
FM	Rotation	Even	90	Shelterwood	<3.0	No	Spruce
BAU	Rotation	Even	100	Shelterwood	<3.0	Non-spruce species	Spruce, larch, fir, beech
OSH	Continuous cover	Uneven	>120	Shelterwood to selection	<0.02	Non-spruce species	Spruce, fir, beech

BAU currently applied management, *FM* management system based on the preferences of forest managers, *OSH* management system based on the preferences of other stakeholders, *NM* no-management system

persons representing forest owners and managers (see Supplementary Material B for details). The MSs were designed based on a questionnaire that identified stakeholder opinions on the forces driving management decisions in the region and on the prospects of the regional forests (Supplementary Material C). Researchers then transformed the questionnaire responses into MS descriptions, which were then implemented in the forest dynamics model.

BAU is an even-aged, uniform, shelterwood system that emphasizes timber production and an ongoing reduction in the share of spruce. The FM system is based on the preferences of regional forest managers and has a strong focus on financial indicators and on the maintenance of a high share of spruce. The OSH system is based on the preferences of other stakeholders, such as the municipality representatives and nature conservation organizations, and promotes close-to-nature management with emphasis on forest diversity, extended rotation and regeneration periods, and small-scale interventions that do not substantially open the canopy. In addition, unmanaged forest development (NM) was evaluated (Table 3). Detailed information about the four MSs and the stakeholders participating in MS definition is provided in Supplementary Material B and C.

Evaluated indicators

The ability of the tested MSs to modify the current tree species composition was evaluated based on the changes in species-specific aboveground biomass volume in standing trees (V). The total production of aboveground biomass (TVP) was evaluated as cumulative volume of aboveground biomass (stem, branches, trunk, foliage) regardless of whether it was harvested or not. The volume of tree compartments was calculated based on the equations proposed by Petráš et al. (1985) and Petráš and Pajtík (1991). The cumulative volume of dead wood generated by both disturbance and inherent tree mortality (i.e. the dead wood component of the TVP, which is referred to as TVP-D) was evaluated as an indicator of MS capacity to moderate forest damage. The ratio of salvage and total (salvage plus planned) harvested volume was used as an additional variable indicative of the effect of MS on damage rate.

Tree species diversity was evaluated using the true diversity index D (Jost 2006), which is the exponential form of Shannon's diversity index S:

$$\begin{cases} S = -\sum_{i=1}^{N} p_i \ln(p_i) \\ p_i = \dfrac{d_i}{G} \end{cases}$$

where N is the number of species, d_i is the basal area of species i (m^2; in this study calculated using the basal area of tree species with diameter $\geq$5 cm), and

$$G = \sum_{j=1}^{S} g_j \left(\mathrm{m}^2\right)$$

Hence, the true diversity index D is defined as:

$$D = \exp(S)$$

Tree size diversity index H was evaluated based on the post hoc index presented by Staudhammer and LeMay (2001):

$$\begin{cases} H_{\text{size}} = \dfrac{H_{\text{DBH}} + H_{\text{H}}}{2} \\ H_{\text{DBH}} = -\sum_{i=1}^{N_{\text{DBH}}} \dfrac{g_i}{G} \ln\left(\dfrac{g_i}{G}\right) \\ H_{\text{H}} = -\sum_{i=1}^{N_{\text{H}}} \dfrac{g_i}{G} \ln\left(\dfrac{g_i}{G}\right) \end{cases}.$$

where N_{DBH} and N_{H} are the number of DBH and height classes present in the stand, g_i is the basal area (m^2) of DBH or height class i, and G is the basal area of the stand (m^2). We used 5-cm classes for DBH and 2-m classes for height (Cordonnier et al. 2013).

The dead wood volume, which is commonly used as an indicator of biodiversity (e.g. Lassauce et al. 2011), was not evaluated because all managed systems strive to remove the dead wood instantly in order to avoid supporting bark beetle populations.

The financial viability of the MSs was evaluated using the net present value (NPV) generated during the simulation period, which is defined as the sum of all discounted net revenue flows. We used a modified NPV definition by Klemperer (1996), which allowed us to compare NPV generated by the MSs during the simulation period, which differs from the rotation cycle (Table 3) (e.g. Roessiger et al. 2016). Instead of calculating only a single NPV for the entire simulation period (2010–2100), we calculated NPV as a sequence of periods starting from year 0 of the simulation period (2010) and incrementally increasing in 5-year steps to 2100 (i.e. the periods 2010–2010, 2010–2015, 2010–2020…, 2010–2100). Each NPV calculation included the discounted net revenue flows incurred during the respective period and the discounted difference between the financial stand value at the end of each period (i.e. years 2010, 2015, 2020, etc.) and the initial stand value in year 2010.

For the sake of simplicity, we also use the term NPV for the financial outcome generated by NM, where only the difference between the discounted financial stand value at the end of the simulation period and the initial stand value was considered (i.e. without any revenue flows). The applied interest rate was 2%.

To facilitate this analysis, we collected timber prices and harvesting costs, silviculture costs, and afforestation costs for all management operations for each MS (Supplementary Material D). To calculate the NPV, we summed the discounted net cash flows of each silvicultural and harvest operation conducted during the simulation period (2010–2100) and weighted them by stand area.

Results

Change in tree species composition

Simulations driven by the baseline climate data (i.e. not affected by climate change) showed that the proportion of non-spruce species, in terms of species-specific biomass volume, slightly increased with NM (Fig. 2a). While most species preserved their initial percentage of representation during the simulation, silver fir (*Abies alba* Mill.) increased substantially (from ca. 2 to 8%). At the end of the simulation period, the spruce proportion decreased from the initial 87 to 78%.

The FM system also preserved a high percentage of spruce; the difference between the initial and final percentage was only −5%. The share of fir increased in the FM system, and the increase was greater (from 2 to 13%) than in the NM system.

Even though the underlying management principles were different, the BAU and OSH systems showed surprisingly similar development of species percentages. The percentage of spruce decreased from an initial value of ca. 87 to 47% with BAU and to 41% with OSH. The percentages of beech (*Fagus sylvatica* L.), silver fir, and maple (*Acer pseudoplatanus* L.) increased with both BAU and OSH.

Climate change accelerated the decrease in the percentage of spruce with all MSs and generally accelerated the change in tree species composition (Fig. 2b). Climate change caused an additional decrease in the percentage of spruce by up to 3% with NM and by up to ca 10% with BAU, FM, and OSH.

Total volume production

The TVP simulated under the baseline climate differed only slightly among the MSs by the end of simulation period (Fig. 3). The TVP was smallest with FM (638 m^3 ha^{-1}) and was highest with NM (685 m^3 ha^{-1}). The per-species TVP corresponded to the changes in species composition described earlier: by the end of simulation period, the non-spruce species accounted for 10% of the TVP with FM but for 15–24% of the TVP with the other MSs.

The response of TVP to climate change differed greatly among climate change scenarios with all MSs. While the moderate scenarios, c1–c3, induced TVP changes ranging from +2 to −7% relative to the baseline climate simulations, the two severe scenarios, c4 and c5, caused decreases in the TVP from −12 to −18%. The decrease was greatest with FM.

Tree species and size diversity

The evaluated diversity indices responded differently to the four MSs (Fig. 4). With NM, tree species diversity D remained unchanged, but tree size diversity H increased steadily throughout the simulation period. FM caused only a minor increase in D, while OSH and BAU caused a substantial increase in D. OSH and FM caused a minor increase in H, while BAU caused a substantial increase in H. Thus, the most positive effect on both indices was obtained with BAU management.

Climate change affected D and H only marginally. While the divergence in simulations driven by the five climate change scenarios was low with NM, both positive and negative responses were simulated with the other MSs. The most distinct response was the climate change-induced increase in D with FM.

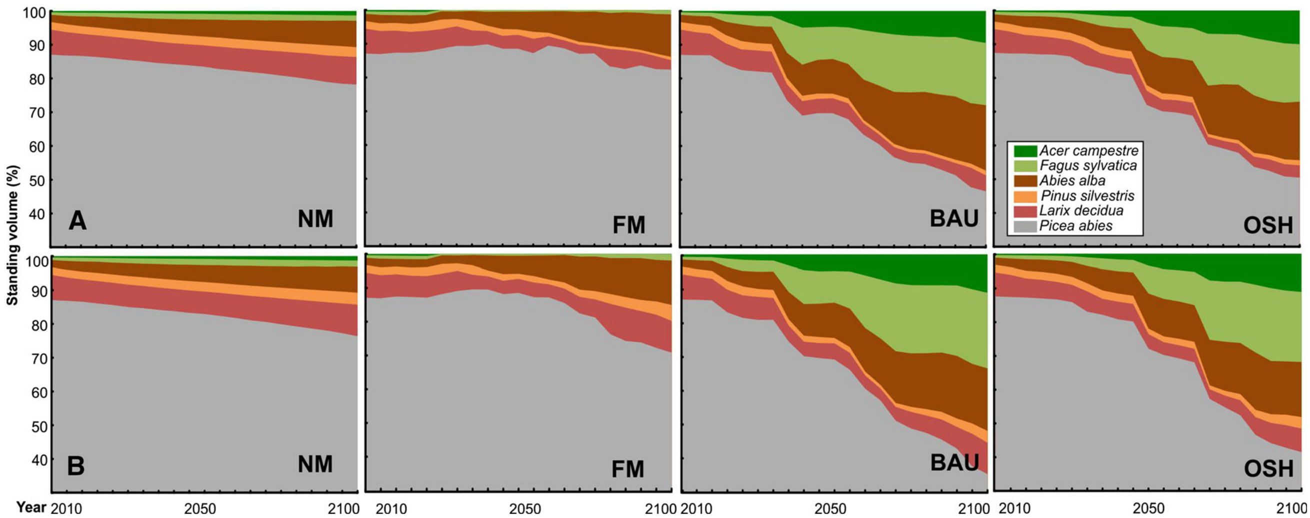


Fig. 2 Development of tree species composition in terms of the relative standing volume of main forest tree species (% of the total standing volume) under four management systems. Row A shows simulations driven by the stable climate referring to the period 1981–2010, and row B shows average of simulations driven by five climate change scenarios. Management codes: *NM* no-management, *FM* management based on the preferences of forest managers, *BAU* currently applied management, *OSH* management based on preferences of other regional stakeholders

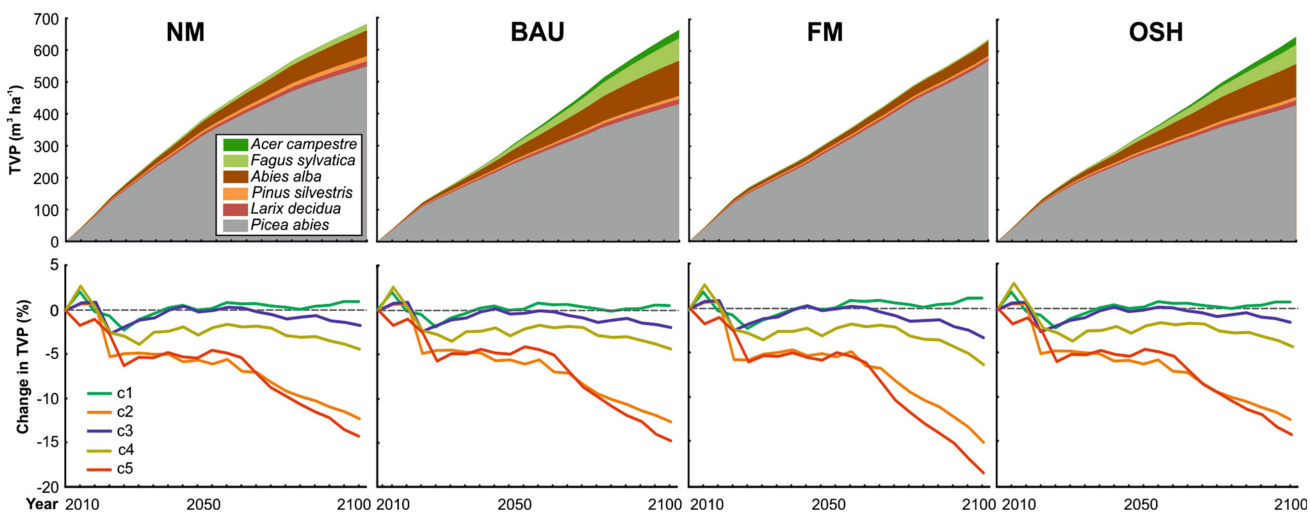


Fig. 3 Total volume production (TVP) per tree species simulated under a baseline climate (*upper row*), and change in TVP averaged for all tree species simulated under five climate change scenarios (c1–c5) (*lower row*) for four management systems (management codes are described in Fig. 2)

Forest damage

Because we assumed that the parameters of species- and age class-specific disturbance rates were stable during the simulation period, and because the relative effect of inherent mortality on the total amount of damaged wood was marginal under all climate change scenarios, only simulations driven by the baseline climate are presented (Fig. 5).

TVP-D was almost two times higher with NM than with the managed systems (871 vs. 454–500 m^3 ha^{-1} during the simulation period); differences among the managed systems were negligible (±5%). In addition, TVP-D accounted for 72% of the TVP with NM but only for 38–43% of TVP with managed systems (Fig. 5a). TVP-D began to diverge between the NM and the managed systems after ca. 2020.

Despite differences in the underlying principles of the MSs, the ratio of salvage and total harvested volume generated by the managed systems converged at 50% in 2070 and remained stable during the rest of the simulation period (Fig. 5b). The differences between MSs were

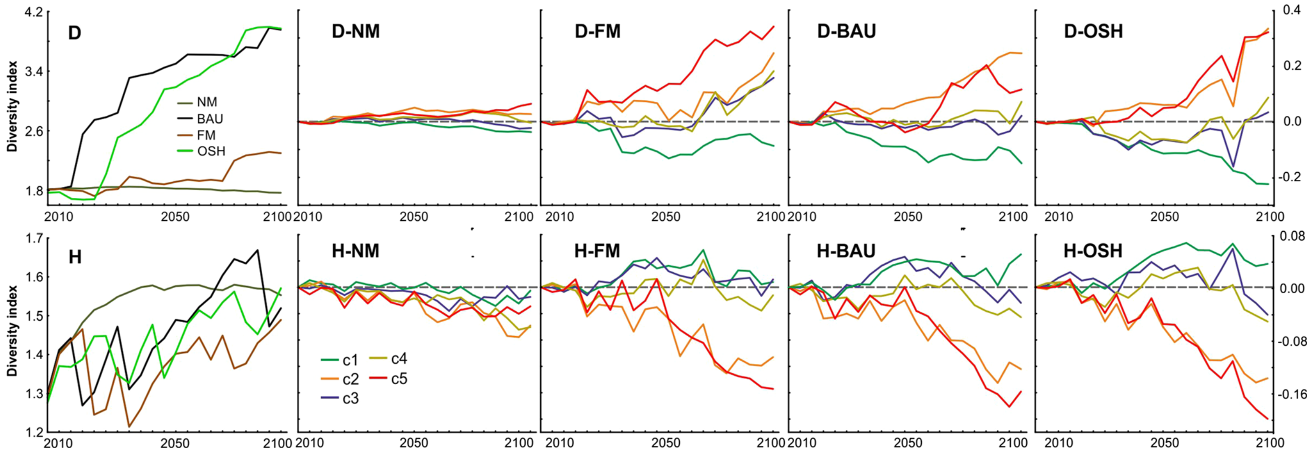
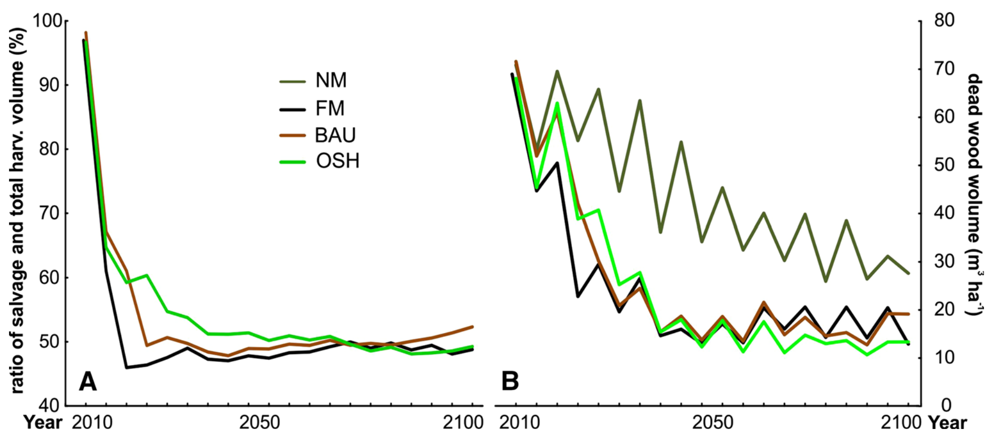


Fig. 4 Tree species diversity index D and tree size diversity index H simulated during the period 2010–2100 with four management systems. Panes D and H indicate the development of the indices under the baseline climate. The remaining panes show the difference between simulations driven by climate change scenarios (c1–c5) and the baseline climate. Management codes are described in Fig. 2

Fig. 5 Ratio of salvaged and total harvested volume simulated under three management systems (**a**), and dead wood volume simulated under four management systems (**b**). The simulations were driven by the baseline climate corresponding to the period 1981–2010. Management codes are described in Fig. 2



greatest mainly during the period 2015–2060; with BAU, for example, the ratio had already reached ca. 45% by 2020.

Net present value

The mean initial value of all simulated stands was 20,000 € ha^{-1}. The simulations show that NPV was negative during the entire simulation period with all MSs (Fig. 6), which indicates that the discounted costs of management operations exceeded the revenues; the NPV ranged from −5600 to −18500 € ha^{-1} in year 2100, depending on MS. The largest drop in value was with NM, where dead wood decomposed without generating revenue. Hence, the value −18,500 € ha^{-1} reflects a high rate of damage typical of NM (Fig. 5) and indicates that the stands were rather disintegrated in 2100.

To better compare NM with the managed systems, we proposed a modified NM ($NM_{salvage}$) that considered the dead wood generated during the simulation as the revenue; NPVs for both NM and $NM_{salvage}$ are displayed in Fig. 6. The analysis showed that the NPV in 2100 was −8400 € ha^{-1} with $NM_{salvage}$, but the NPV had converged to −5860 € ha^{-1} ± 2% with all of the managed systems.

A detailed analysis of processes behind the negative NPV showed that most harvests occurred at the beginning of the simulation period, because 43% of the forest stands were initially 100–110 years old. Specifically, the average planned harvest rate was 4.9 m^3 ha^{-1} year^{-1} up to 2050, and then decreased to 1.5 m^3 ha^{-1} year^{-1}. Obviously, the harvest of mature trees contained a high share of salvage logging. For the managed systems the mean annual volume of salvage logging was 5.6 m^3 ha^{-1} to 2050 and then

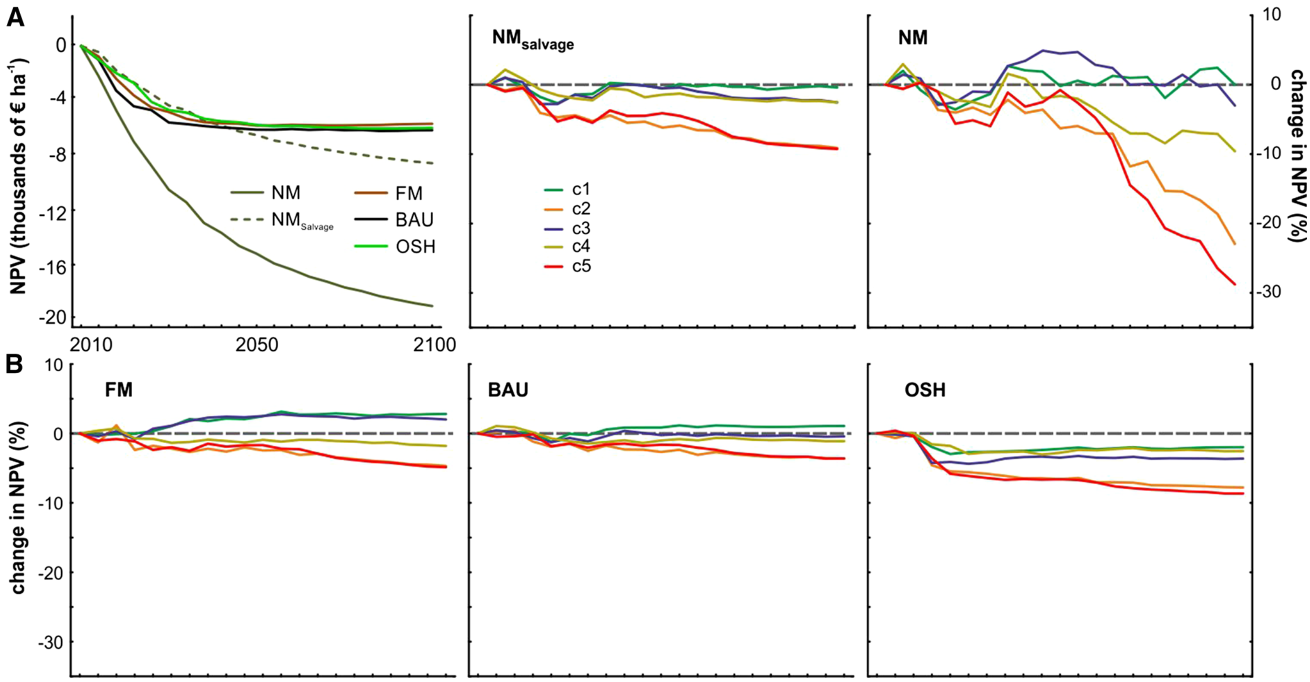


Fig. 6 Net present value (NPV) calculated for a sequence of time periods over the simulation period (from 2010–2010 to 2010–2100) generated by four management systems (**a**) and the relative response of NPV to five climate change scenarios. For an improved comparison of NPV generated with managed and unmanaged systems, a hypothetical example is also presented for the NM in which the dead wood generates revenues ($NM_{salvage}$)

1.6 m^3 ha^{-1} in the remaining period, i.e. it exceeded the planned harvests.

A theoretical exercise showed that if the salvage logging is assumed to have the parameters of regular harvests, the NPV in 2100 would have been −400 € ha^{-1} with BAU; 2200 € ha^{-1} with FM; 700 € ha^{-1} with OSH; −18,500 € ha^{-1} with NM; and 8800 € ha^{-1} with $NM_{salvage}$.

With the three managed systems and $NM_{salvage}$, NPV was reduced by climate change by up to 5% (average of five climate change scenarios) (Fig. 6). The decrease was greatest with OSH and $NM_{salvage}$ (−5%) and was ca—1.5% with FM and BAU. While the decrease in NPV was only slightly greater with $NM_{salvage}$ than with the managed alternatives, the decrease with NM was larger (on average 13%). This, however, can be considered an artefact because the NPV calculation for NM uses only the final stand value in year 2100, which is greatly influenced by climate change. In contrast, NPV for the remaining MSs consists of a number of single cash flows that occurred during the whole simulation period.

As the complementary information, we calculated the nominal value of the remaining stand in 2100, which had an initial average value of 20,000 € ha^{-1} in 2010. In 2100, the average of baseline climate simulations produced the value of 10,864 € ha^{-1} for NM, while it was only 2962 € ha^{-1} for FM, 2019 € ha^{-1} for BAU and 1908 € ha^{-1} for OSH. Extremely low values for the managed systems were mainly related to the decline of overmature stands during the first half of the simulation period, continuous removal of dead trees, which resulted in a low standing volume and an increased share of less productive species. Effect of climate change on the value of the remaining stand was more pronounced than the effect on NPV. The reduction was 13% for NM, 32% for FM, 22% for BAU and 20% for OSH.

Discussion

In this study, we investigated the development of a central European mountain forest as affected by several management alternatives and climate change. The research was motivated by the growing concern about the future of commercial forests dominated by Norway spruce and by the growing desire to convert such forests to ones that are better adapted to environmental change (Spiecker et al. 2004; Löf et al. 2010; Hlásny et al. 2015a). Although management obviously plays a key role in mitigating the effects of climate change on such forests (Bravo et al. 2008), the possible development trajectories remain largely unexplored (Bugmann 2014). We investigated forest development indicators of ecological and commercial importance and the responses of such indicators to management and climate change. In this section, we discuss the ability of the investigated MSs to help forests adapt to climate change, and we also discuss the limitations of our approach.

Unmanaged forest development

Most forest stands investigated in the current study were secondary, i.e. they were established by foresters and in many cases were not well suited to site conditions. The poor prospects for this kind of forest raises the question as to whether unmanaged development might enable natural processes to restore the original species composition of the forests as was suggested, for example, by Drever et al. (2006). Our simulations indicated that this was not the case for the study region, at least not in the time span of the simulation period. With NM, spruce remained dominant, and damage remained high. Our simulations showed that the frequent disturbances, which open the canopy, supported the regeneration of spruce and that deciduous species appeared only slowly, which was consistent with observations of Drobyshev (2001). The slow rate of change agrees with Schelhaas et al. (2015), who suggested that European forests are very inert and that altering their species composition requires a long time. In the study area, this inertia is apparently related to the regeneration success of spruce, even though climate change increasingly favours the non-spruce species (Hanewinkel et al. 2012; Hlásny et al. 2011, 2015a; Schelhaas et al. 2015). The simulated regeneration success of spruce agrees with observations in the study region and in other regions (e.g. Ulbrichová et al. 2006).

Tree species diversity remained low during the whole simulation period, while the diversity of tree size significantly increased, providing potential benefits to biodiversity through the increased diversity of habitat (Staudhammer and LeMay 2001). Diversity indices were only marginally affected by climate change. Interestingly, the inter-climate model variability of *D* and *H* was significantly lower with NM than with the managed systems, which implies that the predictability of stand structure indicators is better with unmanaged development than with managed development

The increase in the abundance of dead wood in NM, which is periodically removed in the managed systems, can support biodiversity (Lassauce et al. 2011). In spruce forests, however, increased abundance of dying and newly dead trees supports bark beetle populations (Økland and Berryman 2004; Hlásny and Turčáni 2013), which have been severely damaging the regional forests in recent decades (Hlásny and Sitková 2010; Kunca et al. 2015). Therefore, the coupled effect of dead wood accumulation and climate change-induced acceleration of bark beetle development (Jönsson et al. 2007; Berec et al. 2013) might be devastating for the regional forests. In fact, substantial forest deterioration, which was manifested by the simulated drop in the value of the remaining stand in 2100, occurred even though the applied disturbance mortality rates were derived from the period 1998–2009 and therefore without the amplifying effect of future climate change (e.g. Seidl and Rammer 2016).

For the above reasons, we argue that one cannot expect that application of a no-management regime will stabilize secondary forests or significantly increase conservation values within the period of ca. 100 years; instead, the unmanaged development is likely to accelerate the ongoing forest decline.

Managed forest development

The currently used management system, BAU, does to some degree support forest conversion by including the planting and protecting of non-spruce species. BAU performed very well in our simulations with regard to tree species composition (i.e. spruce proportion decreased from 87 to 47%) and stand diversity as indicated by *D* and *H* indices. Still, the rate of salvage felling remained greater than 50% of the total harvests, which represents a significant obstacle to forest management and highlights the need for more radical interventions.

FM reflected the desire of forest managers to produce softwood timber and to reduce planting costs. FM also reflected the expectation of forest managers that a reduction in the rotation period might help sustain spruce-oriented management (Schelhaas et al. 2010; Lagergren and Jönsson 2010; Sedjo 2010). It is interesting that in spite of the good performance of BAU in our simulations, forest managers tended to favour an increased emphasis on spruce and a reduced effort to introduce non-spruce species, i.e. the forest managers tended to favour practices that promised to maintain short-term profitability. However, our analyses of the financial outcome failed to indicate that NPV would be greater with FM than with the other MSs. The reduction from a 100-year rotation period for BAU to a 90-year rotation period for FM is unlikely to substantially increase forest stabilization, particularly given the age-dependent disturbance-related mortality rates used in our study (Hlásny et al. 2015a). These rates indicate that there is about 70% probability for a spruce stand to be destroyed up to the age of 90 years. Given that bark beetles have been responsible for ca. 40% of the total forest damage during the recent decade (Kunca et al. 2015) and that bark beetles typically attack trees older than 70 years (Wermelinger 2004), a 70-year rotation period could be considered. Such a rotation is applied, for example, in Croatia (Matić et al. 2010) and in some Baltic and Nordic countries (Rytter et al. 2013). This reduction in the length of the rotation period, although well justified, might not be acceptable because of a strong persistence of traditional practices as well as a temporary surplus of harvested timber that could exceed the wood-processing capacities (Lindner et al. 2008).

OSH was mainly driven by the interest in recovering natural processes. OSH attempts to support biodiversity (by planting admixed tree species), forest aesthetic value and promotes long rotation periods and small-scale interventions. This approach was effective in changing the forest species composition because spruce percentage decreased to only 41% in the last year of the simulation. On the other hand, the longer rotation periods, which increased the abundance of vulnerable overmature trees, caused a high rate of salvage logging to persist longer than with BAU and FM (Schelhaas et al. 2010). Such development might imply a positive effect of disturbances on changes in species composition and biodiversity (Müller et al. 2008; Thom and Seidl 2015), i.e. the disturbances might provide opportunities for more climatically adapted species to establish (Buma and Wessman 2013). In the long run, the small-scale interventions of OSH might better support asynchronous forest dynamics than the other systems and thus support the forest's inherent adaptive mechanisms (Morin et al. 2014).

In all MSs, climate change- and disturbance-mediated support to biodiversity should be considered as an opportunity in forest adaptation efforts in spruce dominated stands. Indeed, adverse effects such as the presented decrease in productivity by up to 15% or support to bark beetle outbreaks (Jönsson et al. 2007; Fleischer et al. 2016) must not be marginalized.

Forest financial value

The complexity of our approach was increased by considering the effect of management and climate change on the financial outcome. Studies on the anticipated effect climate change on forest NPV are scarce (e.g. Hanewinkel et al. 2012; Borys et al. 2015), and this limits our ability to compare our findings with those of other studies. However, information on financial losses and gains can be particularly supportive of management decisions under climate change (Bravo et al. 2008; Eliasch 2008) and can be effective in convincing forest managers to invest in forest adaptation.

Minor differences in NPV development between the tested MSs showed that NPV was not very sensitive to the intensity and timing of the management operations. It is noteworthy that while future forest production was predicted to be greatly affected by climate change, this did not directly translate to changes in NPV because of a strong discounting effect in the distant future; for c4 and c5 climate change scenarios, the TVP decreased by up to 18% while NPV decreased by 0–8% for all MSs. This implies that NPV was mainly affected by a high rate of salvage operations rather than by differences in MSs or by climate change-induced decrease in forest productivity.

A substantial part of NPV in all MSs was derived from the income generated from damaged spruce wood. We found that the NPV could have been ca. 30–40% higher if timber prices had not decreased because of the damage and if the harvesting costs had not increased due to salvage operations. This suggests that reducing the damage by both decreasing inherent forest vulnerability and increasing the efficiency of forest protection is important in the region and might even compensate for losses in biomass production induced by climate change. As Thom et al. (2013) suggested, however, the response of the forest disturbance regime to management can be rather delayed, which underscores the difficulties in stabilizing the regional forests.

The adopted discount rate of 2% should be considered as an assumption; such a value was used, for example, by Roessiger et al. (2011, 2013), while both lower and higher values were used by other authors (e.g. Brunette et al. 2014). The discount rate significantly affected the estimates of NPV in the current study; for example, a discount rate of 0% caused NPV to increase monotonously during the simulation period for all MSs except for NM (data not shown), which is opposite to the presented results. Therefore, the presented decrease in NPV should be considered as strictly specific to the applied discount rate.

Methodological advances and limitations

The complexity of the employed modelling approach generates concerns about the reliability of the estimates for use in forestry decision-making (Yousefpour et al. 2012; Lindner et al. 2014). Among the whole "cascade of uncertainty" (Lindner et al. 2014), the only uncertainties we considered involved climate change scenarios and stochasticity in stand initialization (i.e. 15 replicates); stand initialization, however, only marginally affects the uncertainty of final estimates (Hlásny et al. 2014). It is possible that we significantly underestimated uncertainty of final estimates by adopting only one assumption regarding disturbance-related mortality and by using only one forest model; these factors were found to substantially affect the variability of modelling output (Hlásny et al. 2014; Horemans et al. 2016). Such an underestimation of uncertainty might induce overconfidence in the presented results and potentially lead to poor decisions, such as an inadequate consideration of reversible and flexible management options (Hallegatte 2009; Seppälä et al. 2009). On the other hand, the credibility of our results is supported by our use of highly realistic definitions of applied silviculture and harvesting operations and by previous findings that management greatly affected the variability of modelling outputs (e.g. Hlásny et al. 2014; Horemans et al. 2016).

The model that we used has been repeatedly tested against observed data (e.g. Schmid et al. 2005; Hlásny et al. 2011, 2014; Bošel'a et al. 2013; Horemans et al. 2016), and tree growth performed well in most of the distribution of investigated tree species. However, the model has not been tested at extreme sites, and its performance under poor growing conditions (e.g. those induced by the c5 scenario at water-limited sites) is unknown. This is, however, a limitation of most models (Anderegg et al. 2015) and must be considered in the interpretation.

We assumed that parameters of species- and age class-specific disturbance-related mortality were stable during the entire simulation period (i.e. specific to the period 1998–2009) despite the proven increase in climate-induced tree mortality (van Mantgem and Stephenson 2007), which is expected to be further amplified by climate change (Seidl et al. 2015; Seidl and Rammer 2016). This assumption was adopted because the empirical approach to forest modelling that we used is unable to simulate the complexity of future disturbance regimes and thus cannot provide reliable estimates of the future damage to forests. The damage rates that were used characterize the disturbance regime during damage culmination phase of 1998–2009 (Hlásny et al. 2015a), and were thus considered to represent a reasonable and plausible estimate of future development. Still, the underestimation of variability in simulation output related to this assumption, and the respective consequences for forestry decisions, needs to be considered as discussed above.

The presented changes in forest development indicators induced by climate change represent a mixture of positive and negative responses, which usually occur along an elevation–climatic gradient and which have been reported from diverse environments (e.g. growth and productivity decline in the dry–warm part of the gradient and increase in the cool–moist part of the gradient) (Hlásny et al. 2011, 2015a; Mina et al. 2015; Zlatanov et al. 2015; Pardos et al. 2016). We, however, presented an aggregate response that can be relevant for regional management planning, particularly for assessing future biomass production and financial outcome on a scale of a larger management unit.

Because the applied management operations were differentiated based on stand and site conditions (including elevation), and because simulation outputs were weighted by the total area associated with each of the 25 simulation stands (i.e. area of each combination of stand and site, Fig. 1; Table 1), the presented aggregate information provides a reliable regional estimate of the evaluated indicators and their response to climate change. However, that we represented the forest development of the entire study area by simulating 25 stands unrealistically synchronized forest development. This, however, should not detract from our inferences, which mostly relied on comparison between managements and/or climate change scenarios.

Conclusions

Our study evaluated the combined effect of climate change and four forest management systems on various aspects of forest development in a mountainous region in Central Europe. Although two management systems were proposed by the regional stakeholders as alternatives to the currently applied management, our simulations indicate that forest management based on the stakeholder views will fail to substantially improve future forest production, damage, or financial value relative to the current management. At the same time, it is obvious that a reduction in the current high damage rate and adaptation to climate change will require substantial modifications of the current system. Our findings indicate that the transition from the current even-aged spruce-oriented management to management that promotes more diverse stands, shorter rotations, and smaller-scale interventions might be a viable alternative that would support sustainable, multipurpose forests, and their adaptation to climate change.

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