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Interdependence of adaptive forest management and ecosystem service provision

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Anhang I

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Abstract

The cumulative thesis at hand contributes to investigating the comprehensive theme complex of the interdependence between forest management and ecosystem service provision. I suggest how to regulate mixed species stands in order to achieve a desired set of ecosystem services. Within that scope, I developed an algorithm for assessing the important ecosystem service groundwater recharge in dependence of forest management. In order to exemplify and evaluate the developed approaches, I finally derived ecosystem service trade-offs for adaptive forest management within two distinct case study areas. This research hence contributes to clarifying the interdependence between the investigation objects: in one direction how forest management affects ecosystem service provision and in turn how to achieve desired ecosystem services. Forest attributes are crucial for these clarifications. Thus, under consideration of forest attributes I am applying the results of my publications for answering issues of assessing ecosystem services, characterizing forest management orientations, aiming at ecosystem service portfolios and realizing them through adaptive forest management.

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1. Comprehensive theme complex of the cumulative thesis

1.1. Scientific problem

Ecosystem services (ESS) are benefits which an ecosystem like the forest provides to the human society and which directly or indirectly contribute to human wellbeing (Costanza et al., 2017). There are different terms used within the scope of ecosystem service research and their application in science is not uniform (Costanza et al., 2017). Beside ecosystem service, also ecosystem good, function, and process are important terms in this context. To distinguish these words it is worth understanding that these benefits society receives from ecosystem services may comprise ecosystem goods, functions and processes. However according to the definition by Costanza et al. (2017) ecosystem processes and functions can be ecosystem services, but they do not necessarily need to: this depends on whether they contribute to a human related benefit. Hence, these terms cannot be considered as synonyms. Ecosystem functions result from groups of biophysical structures and processes. Ecosystem processes in turn, are complex interactions between biotic and abiotic elements of ecosystems, comprising material and energy cycles (Hermann et al., 2011). Ecosystem goods are materials measurable in units produced by the ecosystem which serve for ecosystem service enjoyment (La Notte et al., 2017). Costanza et al. (1997) consider ecosystem goods and services together as ecosystem services.

In contrast to the distinction of the related terms as disambiguation, there is also a content related definition and classification within the whole set of ecosystem services. In the case of forestry, that classification has developed through political milestones in forest management like Sylvicultura Oeconomica (Carlowitz, 1713) and Helsinki criteria (MCPFE, 1993) that consider claims of society to forests. Nowadays, these claims are recognizable in policy of many countries by aiming at provision of social, economic, and ecological sustainability. An important step forward from the scientific point of view was a publication in Nature (Costanza et al., 1997) that pointed to more specific ecosystem service definitions, classifications, and descriptions. Today, there are three additional ecosystem service classification systems used worldwide: Millennium Ecosystem Assessment (MEA) (Millenium Ecosystem Assessment, 2005), The Economics of Ecosystem and Biodiversity (TEEB) (TEEB, 2010), and the Common International Classification of Ecosystem Services (CICES) (Haines-Young and Potschin, 2018). These classification systems differ slightly, but in summary, the most substantial key categories of ecosystem services are: provisioning services, regulating and maintenance services, and cultural services (La Notte et al., 2017). Ecosystems regulate for instance water flows, climate, and erosion and thus supply regulating services to humans. Furthermore, ecosystems provide water and raw materials and thus supply provisioning services. In addition, ecosystems are culturally used in terms of recreation, tourism, education and thus supply cultural services.

Of particular importance within the thesis at hand are the ecosystem services groundwater recharge, wood production, and carbon sequestration. Biodiversity is therefore a prominent example for a variable affected or fostered by forest management and hence influences ecosystem services (Sing et al., 2018). Biodiversity is the foundation of ecosystem functions (Millenium Ecosystem Assessment, 2005). Ecosystem functions, like biodiversity are the basis of services and thus they are critical for the provision of all other services (Costanza et al., 2017). This definition clarifies the differentiation between the biodiversity and ecosystem services. Furthermore, this statement underpins the benefit biodiversity provides to humans. However, in the face of future challenges and societal demands a variety of ecosystem service and biodiversity indicators require consideration by forestry. In this dissertation at hand, tree species and structural forest stand diversity are used as a biodiversity indicator. The tradeoffs between groundwater recharge, tree species and structural diversity, wood production, and carbon sequestration are crucial. Tradeoffs between these ecosystem services are crucial for the future of human wellbeing. In terms of climate change drought incidents will likely become more frequent (Spinoni et al., 2018; Turral et al., 2011). The level of groundwater recharge might then become a limiting factor for public water supply. Wood production provides renewable raw materials, and preserves jobs. Carbon sequestration is essential for climate change mitigation. Diversity in tree species and structure is an important criterion of economic and ecological risk mitigation not only in terms of climate change (Neuner and Knoke, 2017).

Beyond forest management at the stand level, forest management on landscape scale level has become a common concern within European countries (Blattert et al., 2018; Michanek et al., 2018). Scientific investigations in terms of landscape consider the pattern of various ecosystems and land cover types (Burkhard et al., 2010; Frank et al., 2012; Groot et al., 2010b). In this simulation study at hand, we focus on forest ecosystems on large areas of the landscape and consider only forest land use. Thus, this dissertation if landscape is mentioned deals with large aggregated forest areas on the landscape scale level but does not deal the whole landscape like it is defined in Groot et al. (2010b).

In order to achieve the desired societal goals through both management planning and responsible allocation of resources, management and policy actors require information about ecosystem service provision and tradeoffs as dependent on forest management. The title of this dissertation: "Interdependence of adaptive forest management and ecosystem service provision" means that forest management determines prevalent ecosystem service provision and in turn, desired ecosystem service provision necessarily determines adaptive forest management. Against that background, three distinct objects of investigation can be identified: forest management, forest ecosystem, and ecosystem service provision. Considering the mutual dependency of these objects, several challenges within the scope of ecosystem service science become obvious: for answering how a particular composition of ESS may be achieved through dedicated management decisions, one has to clarify how management in turn affects ESS provision. The forest ecosystem is the investigation object in-between that links both others. A profound understanding of the crucial correlations among the three objects being considered is pivotal for conceiving the whole complex. This is underpinned by Felipe-Lucia et al. (2018), who stated that "in order to understand how forest management affects multiple ecosystem services, we need to understand how particular forest attributes, which can be altered by management practices, affect different ecosystem services".

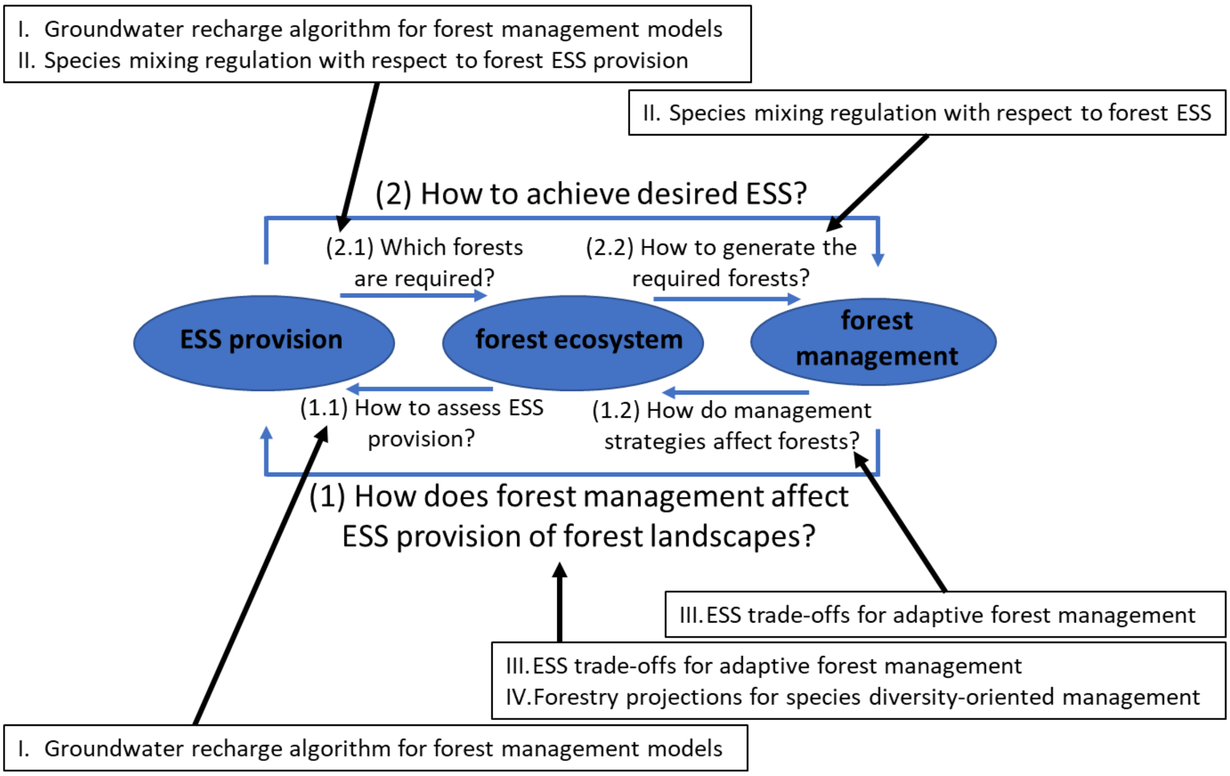

Figure 1: Theme complex of the dissertation thesis at hand. Considering the mutual dependency of the three investigation objects on each other, major challenges within the scope of ecosystem service (ESS) science become obvious. All four publications of the dissertation at hand contribute to answering the superior questions (1 and 2) and their partial problems: 1.1: Publication I; 1.2: Publication III; 1: Publication III and IV; 2.1: Publication I and II; 2.2: Publication II.

Within that scope, the dissertation thesis at hand approaches two key challenges. The first and fundamental one is to analyze the effect of forest management on ESS provision (Figure 1 (1)). The second and superordinate one is to explain how to generate and how to manage forest that is feasible for providing desired ESS (Figure 1 (2)). In detail, the thesis has the following objectives:

(1) clarifying how forest management affects ecosystem service provision by forests on landscape scale level
  (1.1) through providing methods for the assessment of ESS provision by different forest ecosystems
  (1.2) through investigating how distinct management strategies affect forest ecosystems and their ESS provision.
(2) clarifying how to achieve desired ESS
  (2.1) through investigating which forest ecosystem attributes are required for a desired ESS provision
  (2.2) through developing methods that generate forest types with according attributes.

The research questions of this dissertation thesis have so far been rarely worked on in science. It is not well known yet how a forest ecosystem needs to be quantitatively designed in order to obtain the desired provision of ESS (Felipe-Lucia et al., 2018). Moreover, there is an additional challenge in evaluating and quantifying ESS, as they are often difficult to measure (Boerema et al., 2017; Mavrommati et al., 2017). Furthermore, there are no silvicultural alternatives and management concepts for explicitly influencing ESS (Blattert et al., 2017; Langner et al., 2017; Suda and Pukall, 2014). This is particularly the case with regard to mixed species stands (Pretzsch and Zenner, 2017) which play an increasingly important role in the regulation of ESS (Liu et al., 2018).

Therefore, on the one hand methods for the evaluation of ESS and forest regulation concepts are to be developed. On the other, the correlations between silviculture, forest ecosystem attributes and ESS provision need to be investigated. All publications of this dissertation contribute to fixing these scientific gaps of knowledge. Each publication contributes to at least one particular research question of the theme complex presented by Figure 1. Accordingly, they will be presented in the following order:

Publication I (Schwaiger et al., 2018b):

Schwaiger, F., Poschenrieder, W., Rötzer, T., Biber, P., Pretzsch, H., 2018. Groundwater recharge algorithm for forest management models. Ecological Modelling 385, 154–164. 10.1016/j.ecolmodel.2018.07.006.

Publication II (Schwaiger et al., 2018a):

Schwaiger, F., Poschenrieder, W., Biber, P., Pretzsch, H., 2018. Species Mixing Regulation with Respect to Forest Ecosystem Service Provision. Forests 9 (10), 632. 10.3390/f9100632.

Publication III (Schwaiger et al., 2019):

Schwaiger, F., Poschenrieder, W., Biber, P., Pretzsch, H., 2019. Ecosystem service trade-offs for adaptive forest management. Ecosystem Services 39, 100993. 10.1016/j.ecoser.2019.100993.

Publication IV (Toraño Caicoya et al., 2018):

Toraño Caicoya, A., Biber, P., Poschenrieder, W., Schwaiger, F., Pretzsch, H., 2018. Forestry projections for species diversity-oriented management: an example from Central Europe. Ecol Process 7 (1), 357. 10.1186/s13717-018-0135-7.

Publication I and II contribute to answering which forest ecosystem attributes are required for the provision of groundwater recharge, diversity in species and structure, and productivity (2.1). Publication I contributes moreover to the assessment of ecosystem service provision, i.e. groundwater recharge (1.1). Publication III and IV contribute to directly answering how forest management affects ESS provision (1). In particular, publication III states how prevalent forest management strategies in Germany affect forest attributes that are markedly relevant for ESS provision (1.2). Publication II suggests a novel approach for generating desired mixed species forests (2.2).

1.2. Solution approaches / material and methods

1.2.1. Case study area specific ecosystem service provision from inventory based scenario simulation on an aggregated forest landscape scale level

Scenario simulation with SILVA on an aggregated forest landscape scale level

The simulation approach used by Schwaiger et al. (2019) (Publication III) transforms inventory based individual tree data from a forest landscape to data of forest ecosystem service provision. The development of the simulation output is thereby sensitive to forest management. Three stakeholder-specific forest management scenarios were simulated on landscape scale level with the forest management model SILVA (Pretzsch et al., 2002). Multifunctionally oriented forest management was characterized by continuous cover strategies with facilitation of broadleaved tree species. Wood production oriented forest management aims at high shares of coniferous tree species and harvest at maximum yield. The third scenario refrained from any forest management intervention.

The data basis of the approach were sample plots that constitute the primary inventory unit in grid-based forest inventories. Therefore, prior to any simulation, an initial data preparation process classified all inventory plots within the landscape being considered by their stand type. The data preparation then formed one representative virtual stand per each stand type from the structural properties of the type's inventory plots. The whole set of the landscape's virtual stands was used to define a common initial state for all simulation scenarios that referred to the landscape. Within each individual scenario, the forest's development was forward projected through simulation per virtual stand. Each virtual stand was assigned a representative area size that resulted from the representative areas of all plots therein. Per each scenario, all stand specific simulation results could thus be weighted by their representative area and totalled for representing the whole landscape's forest development. In order to clearly distinguish all management scenarios by their specific effect, exclusively one of them was applied to the total forest landscape per each simulation run (Figure 2). Based on each scenario's forest attribute development, the management specific ESS provision of the forest was then evaluated.

SILVA (Pretzsch et al., 2002) is a distance-dependent individual tree model that applies the potential modifier method (Pretzsch, 2009). While that forest management unit-oriented tool refrains from representation of tree-intrinsic processes, its theory of growth is still sensitive to climate and soil conditions. SILVA's growth algorithm works on time steps of five years each. In order to represent the management-related effect of stand treatment on competition and growth SILVA has a top height-driven silvicultural module that implements rule based interventions.

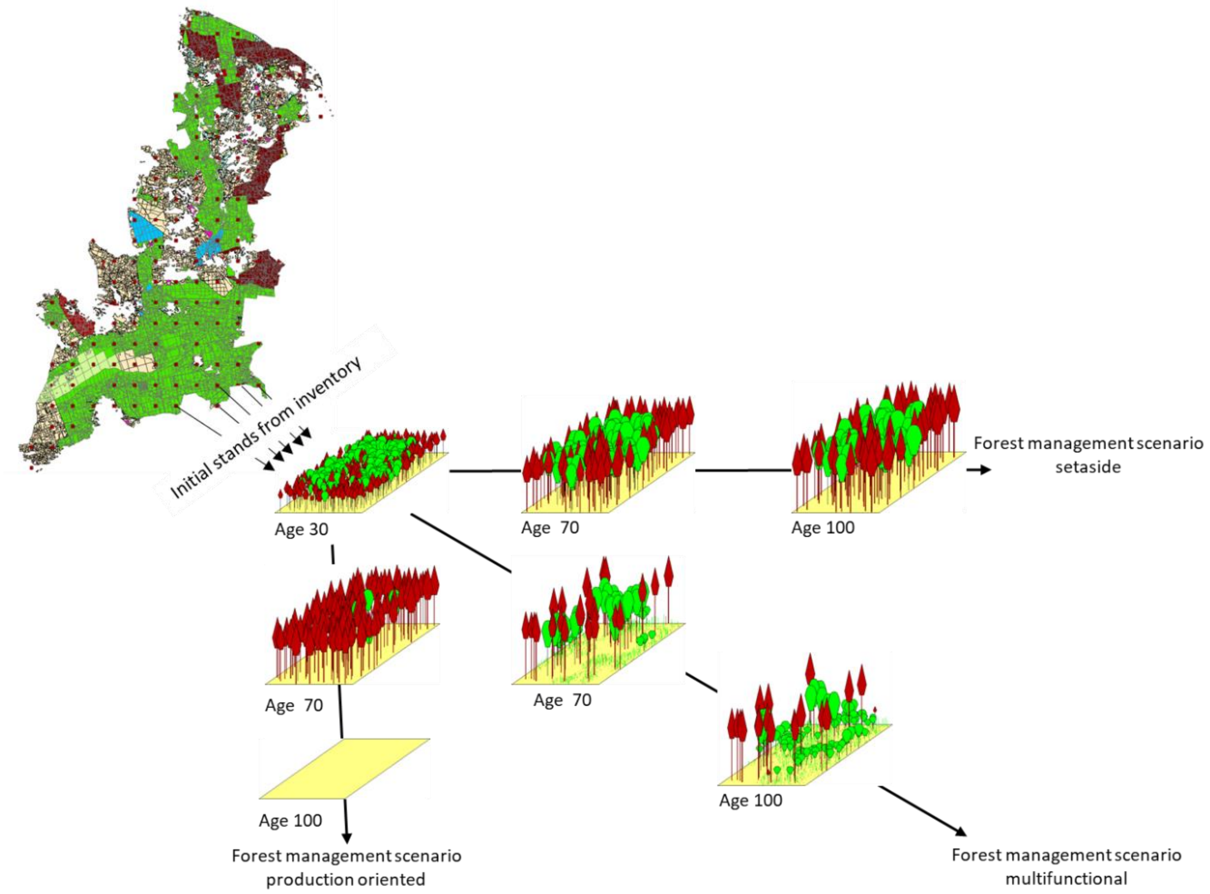

Figure 2: Schematic representation of the inventory based scenario simulation with SILVA. Forward projection of initial virtual stands derived from forest inventories represents forest growth of the whole landscape area. Each of three different forest management scenarios was applied to the whole forest landscape (setaside, multifunctional, production oriented) (Schwaiger et al., 2019).

Ecosystem service simulation

All simulated ecosystem services (Table 1) are based on SILVA output of forest stand structure. For structural diversity, the Species-Profile-Index (Pretzsch, 1996) was used; for productivity, the volume increment was used; for groundwater recharge a specifically developed simulation algorithm (explanation in section 1.3.2) was applied (Schwaiger et al., 2018b); for carbon sequestration a new model of Biber and Black (2019) (explanation in Publication III) was used.

Table 1: Ecosystem service and Biodiversity estimation based on indicators available from SILVA stand structure simulation output.

Ecosystem service	Indicator / Estimate
Groundwater recharge	groundwater recharge (Schwaiger et al., 2018b)
Productivity	volume increment
Structural diversity	Species-Profile-Index (Pretzsch, 1996)
Carbon sequestration	emission savings (substitution), carbon balance (Biber and Black, 2019)

Case study areas and their inventory data base

Forest growth was simulated in two different case study areas in Germany - a less and a more productive one. The forest area of the case study area Lieberose Schlaubetal Neuzelle (LSN) in North East Germany, Brandenburg has a forest area size of 78 000 ha. Within that landscape there are volume per species proportions of about 83 % Scots pine (*Pinus sylvestris* L.) and 8 % sessile oak (*Quercus petraea* (Mattuschka) Liebl.). The forest area within the case study area Augsburg Western Forests (AWF) in South Germany, Bavaria has a size of 53 000 ha. The species shares in AWF are about 75 % of Norway spruce (*Picea abies* (L.) H. Karst) and 7 % of European beech (*Fagus sylvatica* L.). Hence, in case study area AWF shade tolerant tree species are dominant while in LSN primarily light demanding tree species are prevalent.

The database we used for our simulation study comprised several forest inventories of distinct spatial resolution. Within AWF, data from two inventories of different type were available. One was the Bavarian State Forest inventory (BSFI) that has exclusively been conducted on state-forest areas in Bavaria and has a grid width of 200 m. A further one is the German National Forest Inventory (NFI) that also covers private and corporate forest areas. In AWF, the grid width of that national inventory data set (Thünen Institute, 2012) was at 2.8 km. Within the second case study area, LSN, the inventory of the federal state of Brandenburg (MLUL, 2015) was available on forest areas of any tenure type. It has a common grid width of 2 km × 2 km.

1.2.2. Combination of two different forest growth models for groundwater recharge simulation

The modelling approach introduced by Schwaiger et al. (2018b)enables estimating the quantitative sensitivity of the essential ecosystem service groundwater recharge to contrasting options of forest management. Therefore, a practice-oriented forest growth model was calibrated, using a process-based forest growth model and thus substituting empiricism. To this end we first extended the observation based management model SILVA (Pretzsch et al., 2002; Pretzsch and Kahn, 1998) with multilinear functions that describe groundwater recharge as dependent on forest stand structure attributes (Figure 3). In order to calibrate these statistical models, we used the process-based eco-physiological growth model BALANCE (Grote and Pretzsch, 2002; Rötzer et al., 2017). We then applied each species-specific multilinear model in order to calculate groundwater recharge based on simulated stand structure development as we had obtained it from the SILVA management model.

In order to obtain a feasible data set for calibrating each multilinear model to site and species, first of all the stand structure range within each case study area was analyzed. Virtual pure stands were then generated based on that range for representing the area's stand structure variation per most relevant tree species. In a further step, groundwater recharge was quantified for each structure type using the BALANCE model. For evaluating the plausibility of the multilinear model, it was finally applied to simulate groundwater recharge of one representative virtual stand within the case study area AWF (see section 1.2.1) based on the stand structure development that had been forward projected by the forest management model.

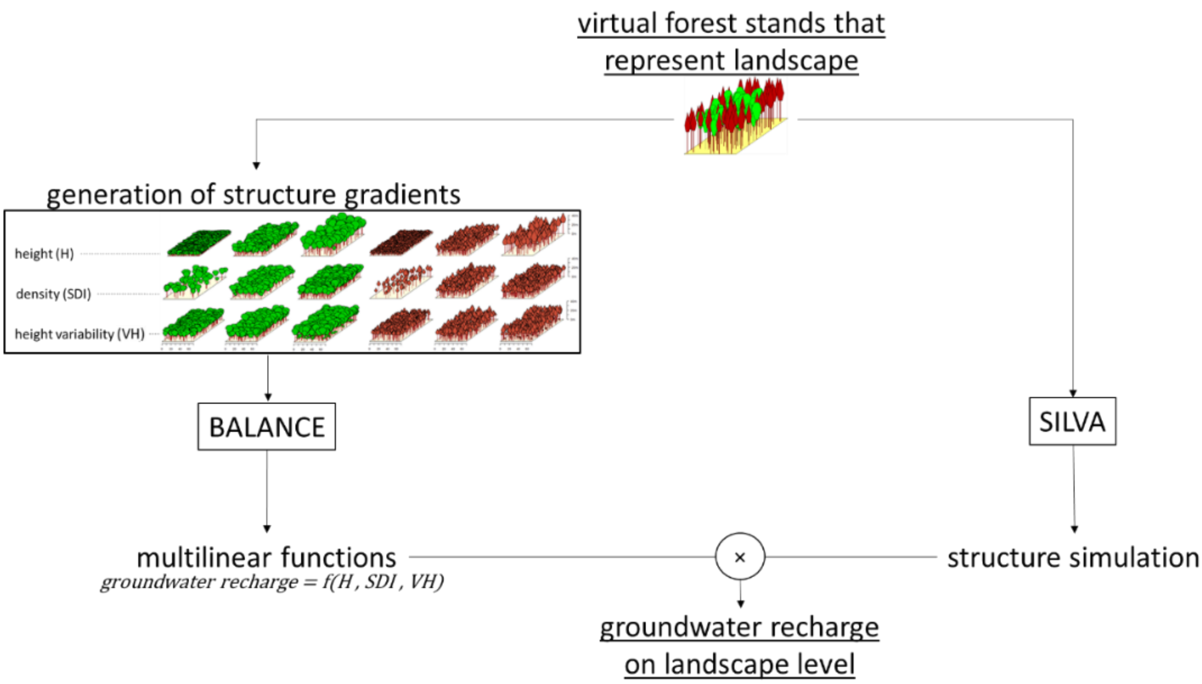

Figure 3: Conceptual diagram of the approach for simulating groundwater recharge on the landscape scale level by means of an observation-based forest management model (SILVA) and a process-based forest growth model (BALANCE). Structure gradients that cover the range inside the landscape were generated. BALANCE was used to derive multilinear functions for estimating groundwater recharge as dependent on structure variables. These functions were applied to stand structures obtained from simulations using SILVA (Schwaiger et al., 2018b).

1.2.3. Transfer of scientific knowledge into adaptive management practice of mixed species forests

This approach implements recent scientific knowledge into a novel forest management algorithm. To this end, time-dependent relations between desired basal-area share, required growing space share and an adequate stem density are derived that enable achieving a particular species mixing proportion. Such a species mixture in turn may be dedicated to a particular ecosystem service-portfolio. The approach includes mixing effects on stand density as a result of recent research that strongly contributes to quantitative mixed stand management (Bayer et al., 2013; Pretzsch and Biber, 2016). However, it is open for refinements that likely will result from ongoing work on the effects of mixture on stand development. We implemented the algorithm in the forest management model SILVA. This enabled us to exemplarily simulate the development of mixed-species stands with different, constant, desired species shares. Hence, we could investigate the differential effect of these basal area shares on groundwater recharge, diversity, and wood production.

The algorithm provides species-specific tree number guide curves in a mixed forest stand of two tree species (Figure 4). For both tree species, a stem number guide curve over time (from mean diameter over time) is derived. This curve guarantees a desired basal area shares. The desired species shares are expressed as basal area shares for practical reasons. The biological key to mixing regulation, however, are the growing space shares. Therefore, both have to be connected. Thus, the basic idea is the consideration of the species-specific individual tree growing space requirement over time that changes along mean diameter growth. This method determines the number of trees that are required to produce a desired basal area composition at optimum efficiency of per-tree growing space allocation. Accounting for that relation, the stand area is optimally utilized by providing each individual tree, on average, with its biologically-based growing space requirement as it is defined by the tree’s crown projection area.

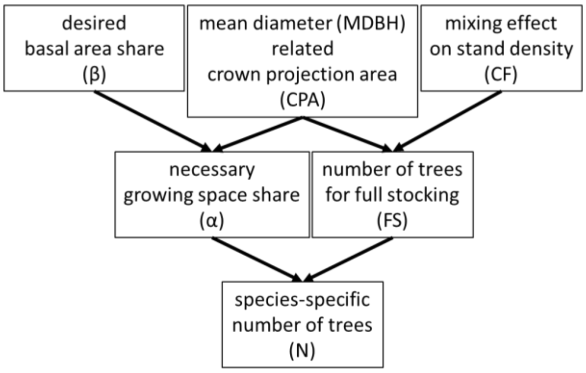

Figure 4: Conceptual diagram of the algorithm for the regulation of mixture proportion that aims at the computation of feasible tree number guide curves. For obtaining the number of trees (*N*), the necessary growing space share (α) gets full stocked for achieving the desired basal area share (β). The number of trees per hectare at full stocking (*FS*) results from the species-specific crown projection area (*CPA*) and a correction factor (*CF*) for suitability in mixed species stands. The necessary growing space share results from the desired basal area share and the stand age dependent diameter (*MDBH*) related crown projection area (Schwaiger et al., 2018a).

1.3. Results and discussion

1.3.1. How to achieve desired ESS

Which forests are required

Which forest attributes are required first and foremost depends on the ecosystem service affinity of the decision maker. The aimed ecosystem service portfolio can be promoted through fostering adequate forest attributes (Felipe-Lucia et al., 2018). As well in literature as in two publications of this dissertation thesis there are findings about correlations between forest attributes and ecosystem services. Thus, both these two publications contribute to answering how forests have to be designed for providing the desired ESS.

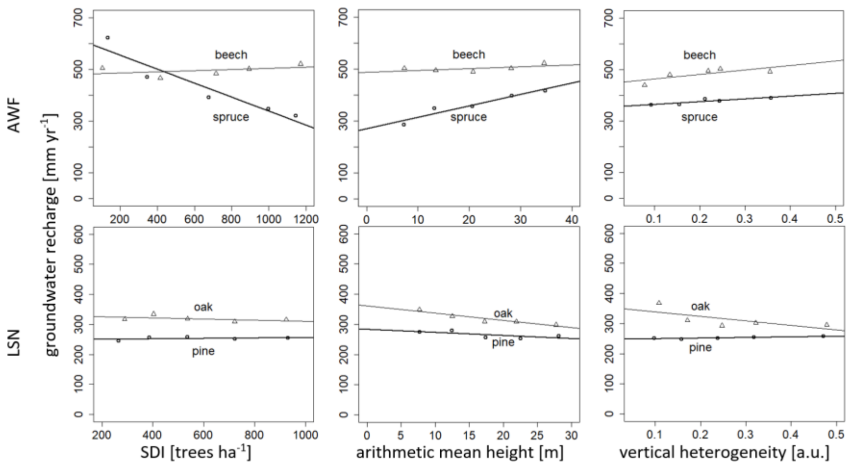

Figure 5: Groundwater recharge over forest stand structure attributes per case study area Augsburg Western Forests (AWF), Lieberose Schlaubetal Neuzelle (LSN) (Schwaiger et al., 2018b).

Publication I (Schwaiger et al., 2018b) reveals species-specific correlations between groundwater recharge quantity and forest stand structure variables like mean height, vertical heterogeneity and stand density. Main results of this publication comprise that groundwater recharge in virtual pure stands of beech (case study area AWF, see section 1.2.1), pine, and oak (LSN) is just slightly sensitive to forest stand structure attributes (Figure 5 and Table 2). Only density in spruce stands considerably influence groundwater recharge. Tree species instead is crucial and thus, as a conclusion, for maximization of groundwater recharge growing space proportion of broadleaves should be increased. Furthermore, a large mean height and a low stand density support the groundwater recharge contribution of spruce (AWF). An additional result of Schwaiger et al. (2018b) was that the effect of forest attributes on groundwater recharge was notably site-specific. Consequently, the effects were of different importance depending on site and precipitation. Felipe-Lucia et al. (2018) also describe that the influence of forest attributes is often related to site properties.

Table 2: Parametrization of multiple linear regressions explaining groundwater rechargedepending on forest stand structure attributes . General form of the model: GWR=y-intercept+ a·MH+b·VH+c·SDI; MH=arithmetic mean height; VH=vertical heterogeneity; SDI=stand density index; GWR=groundwater recharge; a, b, c=slopes; Beta coefficients enable to compare the influence of the structure variables (Schwaiger et al., 2018b).

			slopes for groundwater functions			beta coefficients for slopes		
case study area	tree species	y-intercept	MH	VH	SDI	MH	VH	SDI
AWF	European beech	463.71		154.61			0.19	
	Norway spruce	418.72	5.61	234.46	−0.28	0.38	0.14	−0.74
LSN	sessile oak	386.19	−2.01	−132.54		−0.30	−0.35	
	Scots pine	279.91	−1.17			−0.17		

Furthermore, Publication II (Schwaiger et al., 2018a) exemplifies how forest attributes might be used in order to exert influence on ecosystem services. This publication deals with the influence of mixing proportions and its effect on diversity, groundwater recharge quantity and productivity. It reveals that the forest attribute tree species mixing proportion is crucial for the achievement of the desired set of ecosystem service provision. In particular, it is essential for the optimization of that set. High basal area shares of spruce in mixture with beech increased productivity but decreased groundwater recharge in the case study areas. Highest diversity could be achieved at 50 % basal area share (Figure 6). The best mixing proportion for multifunctional purposes was a beech basal area share of 20 to 50 % that corresponds to a beech growing space share of approximately 50 to 80 %.

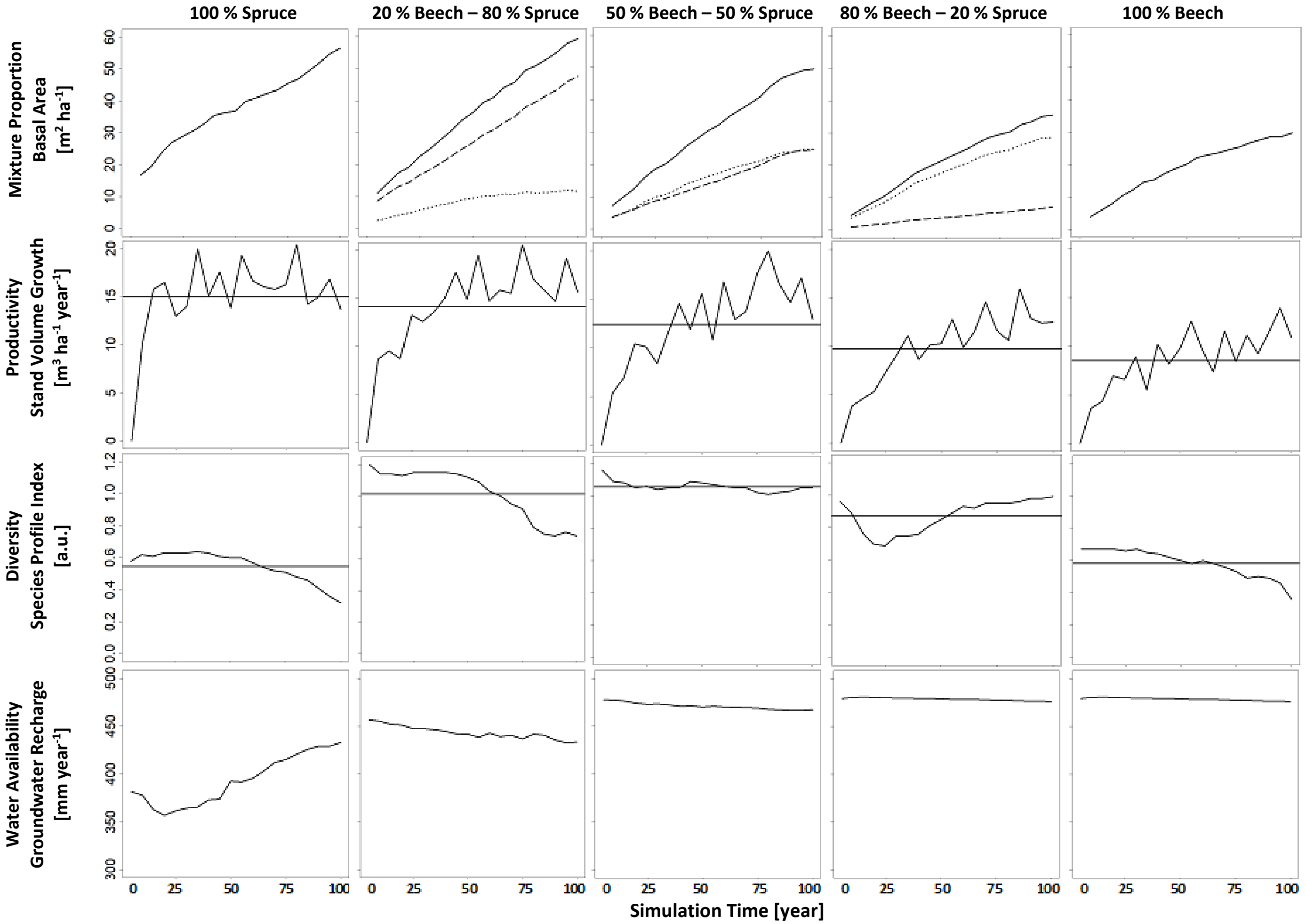

Figure 6: Mixing proportion (basal area composition) is a crucial forest attribute for the ability to provide different ecosystem services (water availability, diversity and productivity). These results are based on simulation runs within an mixed stand of European beech and Norway spruce; dotted line-beech, dashed line-spruce, horizontal line-average, solid line-total stand (Schwaiger et al., 2018a).

How to generate the required forests

On the one hand it must be known what characteristics of forest attributes are necessary for the provision of certain ESS. On the other hand, moreover, the actors in forest management need practice-oriented tools to achieve the required forest attributes accurately. According to Gadow and Füldner (1995) the crucial requirements for achieving sustainability can exclusively be met if criteria of objective control are created and subjected to a more detailed description and quantification. Nowadays, in terms of forest stand regulation there is particular requirement for treatment prescriptions that focus on mixed species stands. Therefore, the scientific focus today is notably on the development of novel silvicultural guidelines for mixed species forests (Bauhus et al., 2017b; Coll et al., 2018; Pretzsch and Zenner, 2017). Quantitative silvicultural guidelines are largely limited to even-aged, homogeneous and monospecific stand types (Bauhus et al., 2017a; Bauhus et al., 2017c). Most current guidelines for the management of mixing proportions do apply steering principles. However they do not yet enable the quantitative control of desired future stand properties (Ammann, 2005; Ammer, 2008; Hansen and Nagel, 2016; Oliver and Larson, 1996; Pretzsch and Zenner, 2017; Schröpfer et al., 2009; Utschig et al., 2011). However, mixed stands, in addition to common wood production, are notably important for the provision and regulation of the entire modern ecosystem service portfolio. Therefore, the dissertation thesis at hand introduces a novel regulation method for mixed species stands. Thus, it contributes to the development of novel tools for generating forests of a composition and structure that supports the provision of desired ecosystem services.

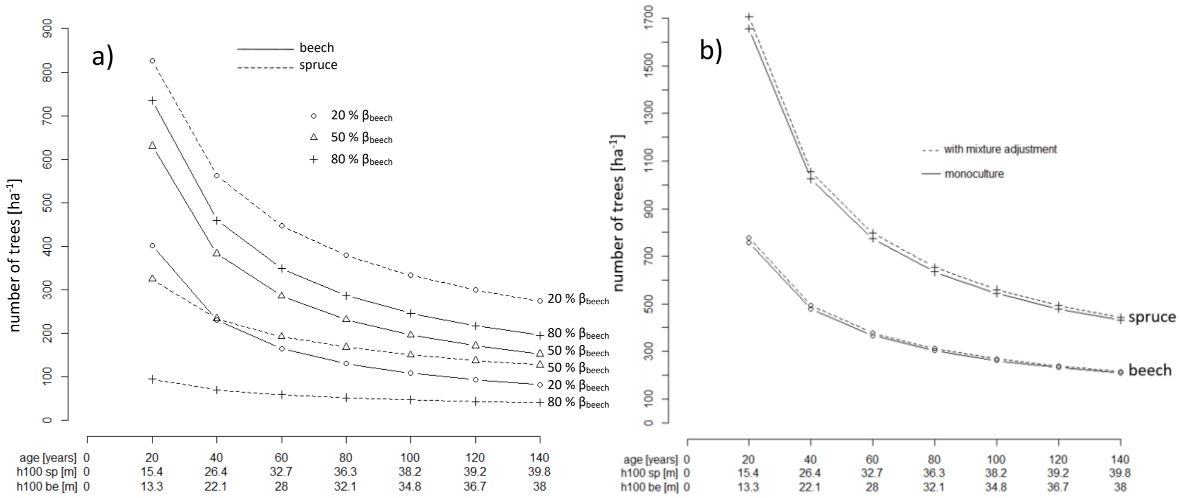

Figure 7: a) Tree number guide curves for three different basal area mixing proportions (β) in an even-aged mixed species stand of beech and spruce. b) Number of trees per hectare in spruce and beech monocultures, and with adjustment for mixed species stands (Schwaiger et al., 2018a).

In Schwaiger et al. (2018a) this dissertation work suggests how to adjust a desired basal-area mixing proportion at optimum growing space use in an even-aged mixed species stand of beech and spruce on a fertile site at any given point in time. To this end the thesis designs and applies tree number guide curves that consider latest scientific knowledge about mixed species forests. The curve's calculation procedure implements an adaption of the monoculture-specific tree numbers to the growth-allometry in mixed species stands. A species-specific relation of per-tree growing space to per-tree basal area enables defining an equivalence ratio that quantifies how many trees of one species would correspond to a given tree number of the other species presuming crown closure without crown overlap in a pure stand at given average tree basal area (nameded tree number ratio in the following). Considering such equivalence aspects can help practitioners to achieve the desired mixing relations (Pretzsch and Zenner, 2017). The equivalence ratio is a fundamental result of the study and as well as the mixing effect (Schwaiger et al., 2018a) contributes to the resulting prescription of tree number per species over time.

Within the stand type being considered, for a 50 % basal area mixture, the tree number ratio shifts with age (Figure 7 a): at the age of 20 years beech requires twice the tree number of spruce, i.e. two beech trees are equivalent to one spruce tree for equivalent basal area shares. This ratio approaches one to one with increasing stand age. In contrast, one beech requires twice the growing space as one spruce, independent of time (Figure 7 b). Accordingly, striving for a 50 % growing space mixture, the tree number ratio remains almost constant over time at two spruces per one beech (Figure 7 b). In the case of the 20 years old stand with equal basal area shares this means in effect a four times higher growing space of beech compared to spruce. Quantifying the species-specific relations of growing space and basal area contributes to the practical realization of the suggested mixing regulation approach. Further results of Schwaiger et al. (2018a) help to realize quantitative mixing regulation in practice: constant basal area share of beech requires decreasing growing space share with stand age (Figure 8). However, growing space regarding beech is always larger than the share of basal area at the same age.

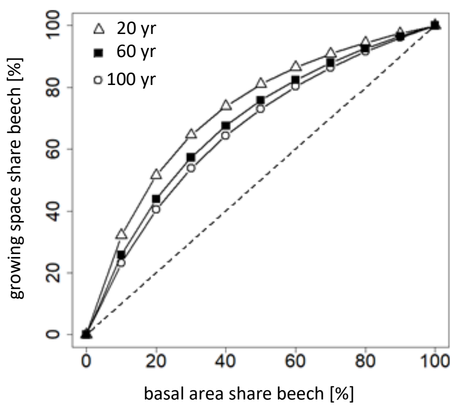

Figure 8: Conversion between growing space share and basal area share in a mixed stand of European beech and Norway spruce. Each line presents one stand age: triangle = 20 years, square = 60 years, circle = 100 years (Schwaiger et al., 2018a).

1.3.2. How forest management affects the ESS provision by forest landscapes

Several studies have revealed that a multifunctional forest management orientation with a low intensity of management and a stand treatment that aims to establish continuous cover forestry are most suitable for the provision of high or moderately high levels of distinct services at the same time (Pang et al., 2017; Pukkala, 2016; Sing et al., 2018; Triviño et al., 2017). Multifunctional management provides more ecosystem services than single-objective management that aims e.g. at the maximization of economy or wood production (Pukkala, 2016). In turn, higher intensity management and production oriented management when they maximize one single service have a negative impact on biodiversity, health and recreation, and water supply services (Sing et al., 2018). However, Pang et al. (2017) reveal that productive forest management achieves a 30% higher wood production than multifunctional forest management. Accordingly, there is a trade-off between maximization of one ecosystem service and maintaining high levels of several services at the same time on stand level.

Triviño et al. (2017) show that no forest management orientation alone is able to maximize a number of ecosystem services like wood production, carbon storage and biodiversity at the same time on landscape level. The absolute maximization of single ecosystem services needs a single service-oriented management. Hence, a combination of different management orientations is needed to resolve the conflict among maintaining both the highest range and the highest level of multiple ecosystem service provision on landscape level (Sing et al., 2018).

Within the dissertation thesis at hand two publications directly state how forest management may affect ecosystem service provision (Publication III and IV, see Section 1.3). Schwaiger et al. (2019) reveal significant influence of the forest management pathway on ecosystem service provision within both case study areas being considered. While the results of the study strongly depend on the site and on the initial situation inside each location, there are general trade-offs that pertain to both regions. In both case study areas the production oriented forest management pays for productivity with structural diversity (Figure 9). In contrast, multifunctionally oriented forest management pays for groundwater recharge with productivity losses. The setaside buys current carbon sequestration what means increasing forest carbon stocks and pays for it with sustainable carbon sequestration due to lack of emission savings. The study reveals increasing diversity and groundwater recharge at the expense of productivity in both case study areas. In addition, a synergy between productivity (as volume increment) and carbon sequestration is demonstrated and it is revealed that the strategy of increasing volume stocks is highly efficient for short-term carbon sequestration. However, it is clearly demonstrated that only forest use makes the forest a sustainable carbon sink. Furthermore, Schwaiger et al. (2019) quantify the difference in the magnitude of ecosystem service provision between both case study areas and sites, and consequently reveal that forest management can just partly balance that site-related variation.

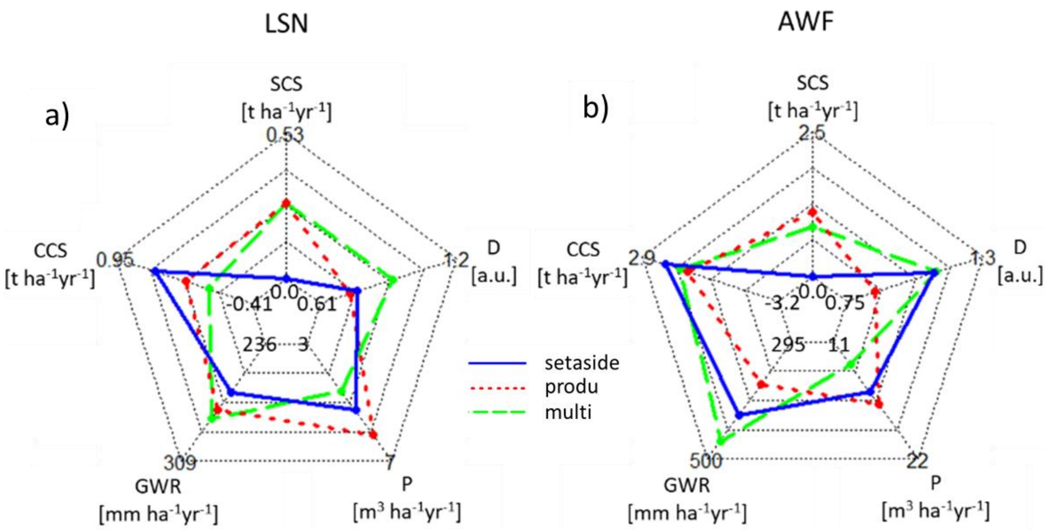

Figure 9: Average provision of ecosystem services over the whole simulation period per case study area and simulation scenario; these services are current carbon sequestration (*CCS;* proxy = overall balance), sustainable carbon sequestration (*SCS*; proxy = emission savings), diversity (*D*), groundwater recharge (*GWR*) and

productivity (P); the middle of each radar chart corresponds to the minimum value per service indicator observed within any scenario or time step, while the outermost value close to the axis label corresponds to the service indicator's maximum value; LSN and AWF designate the case study area; Multi = multifunctional forestry scenario, Produ = wood production scenario and setaside scenario = ecological process protection (Schwaiger et al., 2019).

Toraño Caicoya et al. (2018) (Publication IV, where the candidate of the dissertation at hand is coauthor, see Section 1.1) demonstrate the relevance of scale for the interpretation of management effects on stand structural and tree species diversity. Two different diversity indicators were used in this publication. The species profile index (Pretzsch, 1996) focuses on vertical and horizontal information and the species intermingling (Füldner, 1996) focuses mainly on horizontal structures.

Under multifunctional forest management (denoted as Multi in the following) the average of forest stand-specific diversity remains constant over time on landscape scale. In addition, Multi leads to increasing homogeneity of the forest stand-specific diversity at the landscape level (exemplarily in Figure 10). This development is due to increasing structure in poorly structured stands and from decreasing structure in highly structured stands (e.g. group mixture in mixture by single trees). Under production oriented forest management (denoted as Produ in the following) the average of stand-specific diversity decreases over time on landscape scale. Structure decreases in almost all stands, both in already poorly structured stands and in highly structured stands. These results of Toraño Caicoya et al. (2018) clearly show that a greater variety of heterogeneous and homogeneous stand-specific structures can be achieved by mixing different forest management orientations at the landscape level.

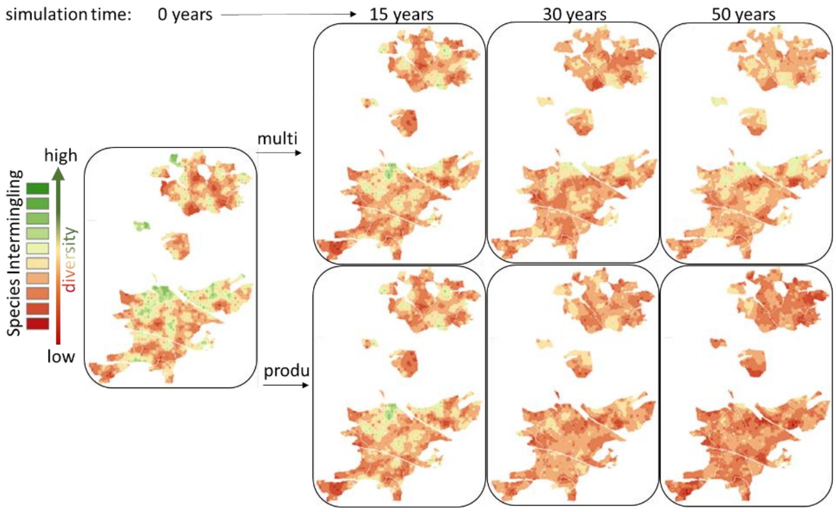

Figure 10: Development of Species Intermingling respectively horizontal stand diversity depending on two different forest management scenarios: Multi = multifunctional, Produ = wood production oriented (Toraño Caicoya et al., 2018).

How to assess ESS provision

Public discourse and awareness of the forest's ecosystem service provision can increase through the announcement of its existence, but only by valuating the services well-founded decisions can be made (Costanza et al., 2017). Just by this, trade-offs can be weighed and considered in decisions. In every decision we value more or less consciously. Thus, as long as we are forced to make decisions, there will be an evaluation process. Valuation of ecosystem service provision can help society in many cases, where tradeoffs exist, to make better decisions (Braat and Groot, 2012; Groot et al., 2010a). Thus, assessment of ecosystem services has become a prominent scientific topic (Zhang et al., 2015). Valuation methods are the key tools to locate ecosystem service provision on value scales of sustainability, fair distribution, and efficiency (Costanza et al., 2017). To be adequate for this purpose in policy and decision making, a comprehensive assessment needs an individual combination of specific methods (Pandeya et al., 2016). According to Pandeya et al. (2016), one single valuation approach alone cannot cover the whole context ecosystem services are embedded in, because contextuality is individual in locally very diverse systems. Different assessment approaches may have specific and complementary roles in the evaluation of ecosystem services (Pandeya et al., 2016). Choosing a method for a particular application may depend on many factors, external context, the ecosystem services, limitations of the methods, data availability, resources, and expertise (Harrison et al., 2018).

The value of an ecosystem service or good is indicated quantitatively or qualitatively (Pandeya et al., 2016). A rising number of ecosystem service valuation methods has been developed, and there are publications that have collected, compared, and categorized them (Groot et al., 2002; Harrison et al., 2018; Pandeya et al., 2016). There are monetary resp. economical valuation methods, biophysical resp. ecological valuation methods, and socio-cultural ones. Biophysical methods predominant at mapping of ecosystem services, often use modelling approaches (Baró et al., 2015; Burkhard et al., 2012; Kopperoinen et al., 2014; Zulian et al., 2018) (Harrison et al., 2018). Socio-cultural methods that are predominant at understanding preferences or social values, often use deliberative valuation (interactive valuation method, different actors form value judgements) approaches (Kelemen et al., 2013), preference ranking approaches (Calvet-Mir et al., 2012), multi-criteria analysis approaches (Langemeyer et al., 2016), and photo-elicitation surveys (García-Llorente et al., 2012a) (Harrison et al., 2018). Monetary methods predominant at economic valuation, often use stated preference (survey-based technique for establishing valuations) (Bateman et al., 2002; García-Llorente et al., 2012b) and revealed preference (technique for establishing valuations based actual decisions people make, in contrast what they would state) (Gibbons et al., 2014; Langemeyer et al., 2015) pricing tactics (Harrison et al., 2018).

Although the process of creating concepts for ecosystem services in science and politics has had an obvious advance, the quantitative assessment of ecosystem services is still a challenge. This is particularly true when local decision makers require local information. For that purpose, simulation techniques play an increasingly important role (Nelson et al., 2009). In particular, water balance and landscape development are predestined for model evaluation because they depend on the interaction of many boundary conditions and are difficult to depict in empirical results (Pandeya et al., 2016).

Table 3: Equations for groundwater recharge calculation in mixed species stands according to Schwaiger et al. (2018b). AWF = Augsburg Western Forests, LSN = Lieberose Schlaubetal Neuzelle, GWR = groundwater recharge, VH = vertical heterogeneity, MH = arithmetic mean height, SDI = Stand Density Index (Reineke, 1933), GSS = growing space share (Schwaiger et al., 2018b).

case study area	tree species	equation
AWW	beech	$GWR = 463.71 + 154.61\ \mathrm{VH}$
	spruce	$GWR = 418.72 + 5.61\ \mathrm{MH} + 234.46\ \mathrm{VH} - 0.28\ \mathrm{SDI}$
LSN	oak	$GWR = 386.19 - 2.01\ \mathrm{MH} - 132.54\ \mathrm{VH}$
	pine	$GWR = 279.91 - 1.17\ \mathrm{MH}$
All	total in mixed species stand	$GWR = GSS_{species1}\ GWR_{species1} + GSS_{species2}\ GWR_{species2}$

This thesis at hand contributes to the development of ecosystem service valuation. Within the scope of Publication I, Schwaiger et al. (2018b) therefore exemplify the simulation of groundwater recharge as dependent on forest management (see section 1.2.2). The study's modeling approach enables a quantitative assessment of that essential service's variation by contrasting silvicultural options. Thus, it is suitable to practice forest management planning with trade-off estimations among ecosystem services on landscape level. The approach abstracts quantification of groundwater recharge to key variables (tree species, and stand height, density, heterogeneity) essential for consideration of the pure influence of forest management on landscape level (Table 3). In both case study areas tree species was the most important predictor and forest stand structure was less important. That result underpins Felipe-Lucia et al.

(2018), who revealed that forest attributes are the strongest predictors of most forest ecosystem services.

How management strategies affect forest ecosystems

In Europe, there are three major basic forest management strategies. Each of them is specific to stakeholder groups (Blattert et al., 2018). Multifunctional forestry is often preferred by state forest organizations. Production oriented forestry is often preferred by private forest owners. In order to restrict active forestry, nature conservation organizations habitually call for larger proportions of setaside areas. Furthermore, strategies for conservation of diversity and simultaneous use of wood are prevalent, segregated (combination of productive and setaside areas) versus the integrated (multifunctional) approach. Forest management strategies differently affect forest stand and landscape characteristics. Therefore, in Europe, these management strategies need to be the subject of forest ecosystem service research. Felipe-Lucia et al. (2018) state that increasing structural heterogeneity, maintaining large trees, and canopy gaps should be accelerated by multifunctional forest management for achieving its purpose. This reveals the potential of forest management to influence ecosystem services by its effect on forest ecosystem attributes. Thus, it is important to investigate the development of forest ecosystems under different forest management strategies.

Within the dissertation at hand (Publication III (Schwaiger et al., 2019)), a rule based silvicultural setting was simulated according to each of all three major basic forest management orientations. That way, this thesis contributes to the investigation how modern forest management strategies affect forest ecosystem attributes (Figure 11) with respect to ecosystem service provision. Setaside had the highest amounts of deadwood, although no harvest residues could accumulate in that scenario. In the case of scenarios Multi and Produ, the amount of deadwood was about the same due to a similar volume of residues from harvest. In both case study areas (LSN, AWF) the volume stock of Multi was constant at about 50% of the maximum value in the setaside. The average stocks of the production-oriented management are about the same in both case study areas, but stocks in LSN are more stable. Multi reduces the coniferous wood shares and thus ensures a forest conversion in favor of broadleaves. In both case study areas the Species-Profile-Index reveals that the production-oriented landscape consisted of poorly structured stands. Larger trees (of higher diameter) are to be found especially in the setaside-scenario. Still, the multifunctional scenario constantly over time maintains a high amount of large trees as compared to the production oriented one.

The thesis at hand underpins the work of Bösch et al. (2018) who stated that provision of forest ecosystem services is site-specific, but also depends on the forest owners' objectives. Accordingly, all landscape stock characteristics are higher in AWF if compared to LSN. Thus, the absolute effect of forest management on ecosystem attributes and corresponding ecosystem services is notably related to each site's potential.

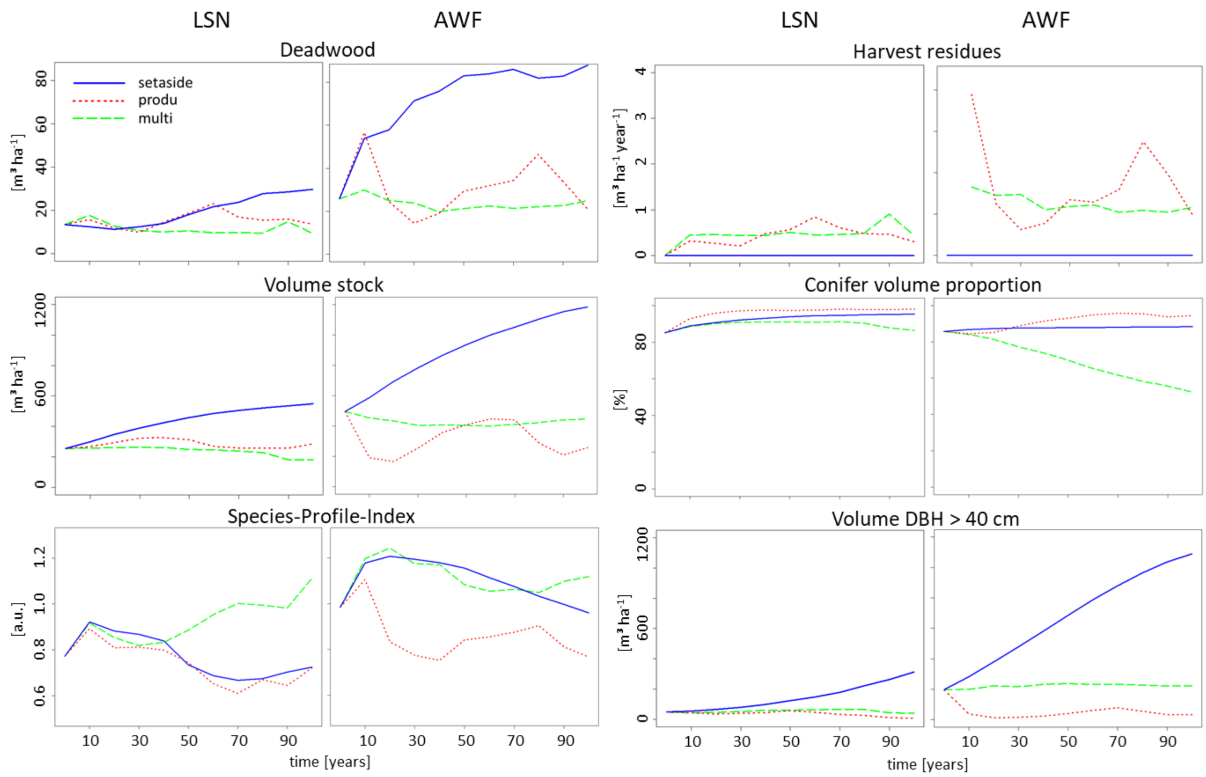

Figure 11: Case study specific development of forest ecosystems (in LSN = Lieberose Schlaubetal Neuzelle and AWF = Augsburg Western Forests) on landscape level

depending on forest management strategies (Multi = multifunctional, Produ = production oriented, Setaside = no interventions). Volume stock and Species-Profile-Index from Publication III (Schwaiger et al., 2019) the other variables are part of to Schwaiger et al. (2019) but were not pre-published.

As there is a clear link between climate change and extreme weather events (Eckstein et al., 2018), the lack of windthrow and insect calamities within the forest growth model used (SILVA, see section 1.2.1.) requires to assess the possible biasing effect of disturbances on the scenario results. The setaside scenario in particular requires to estimate the range of calamity effects, because absence of interventions markedly increases the risk of calamities. For example, beetles, windthrow and fire may lead to calamity-induced dynamics and an according change of the forest landscape. Thus, a change to a more structural and tree species rich forest landscape than presented by the setaside scenario is possible within the simulation period. While setasides will however be marked by clear dominance of conifers in the absence of calamities within the next 100 years, multifunctional management will maintain an increase of broadleaved species shares. Primeval forest dynamics however are known to take place within the order of centuries.

1.4. Conclusions and future research

The thesis at hand contributes to answering questions that arise from the interdependence between adaptive forest management and ecosystem service provision, comprising the way from ecosystem service provision over the ecosystem itself to forest management and vice versa. In the one of both directions being considered, the study analyses how desired ecosystem services depend on ecosystem properties and management, and thus in the other direction, how forest management through its effect on ecosystem traits maintains ecosystem service provision. As species mixture today is strongly emphasized by German forestry guidelines, I suggest how to regulate mixed species stands in order to achieve a desired set of ecosystem services. Within that scope, I have developed an algorithm in order to assess the important ecosystem service groundwater recharge in dependence of forest management. In order to exemplify and evaluate the whole set of developed approaches, I finally derived ecosystem service trade-offs for adaptive forest management within two markedly distinct case study areas.

The main results of this thesis comprise that groundwater recharge in the case study areas is just slightly sensitive to forest stand structure attributes (Figure 5 and Table 2). The tree species composition is however crucial and thus, as a conclusion, for maximization of groundwater recharge the growing space proportion of broadleaves should be increased. Moreover, findings underpin that the forest attribute tree species mixing proportion is crucial for the achievement and optimization of the desired ecosystem services portfolio. This thesis demonstrates that greatest variety of heterogeneous and homogeneous stand-specific structures on landscape level can just be met by a combination of different forest management orientations. The thesis also quantifies the difference in the magnitude of ecosystem service provision between two case study areas and sites, and consequently reveals that forest management can just partly balance that site-related variation. That work moreover points to an increasing diversity and groundwater recharge at the expense of productivity in both case study areas. In contrast to that tradeoff, a synergy between productivity (as volume increment) and carbon sequestration is demonstrated and it is revealed that the strategy of increasing volume stocks is highly efficient for short-term carbon sequestration. However, it is clearly demonstrated that only forest use makes the forest a sustainable carbon sink.

The thesis at hand points thus to a trade-off between groundwater recharge and carbon sequestration which is crucial, because it would mean that forest management is faced with a notable dilemma. Reducing climate change by carbon sinks within a segregated forest management system that involves both production oriented stands and setasides would aggravate the groundwater recharge issue. Multifunctional forest management seems to be the most appropriate way to gain high carbon sequestration and groundwater recharge at the same time. The results of the thesis clearly indicate that multifunctional forest management can maintain essential public interest objectives in both case study areas over the next 100 years.

Knowledge about the relation between forest attributes and ecosystem services still holds a high innovation potential. Studies by Pretzsch et al. (2016) investigate mixing effects on forest stand productivity as dependent on tree species shares. However, further studies that investigate other ecosystem services as related to mixing ratios are rare. Furthermore, biodiversity assessment could be improved through findings about correlations between mixture, structure and ecological key species (Hilmers et al., 2018; Maltamo et al., 2014). Knowledge about the relation between ecosystem services and forest attributes is essential for the development of service-oriented silvicultural procedures and forest management concepts.

Within that scope, mixture regulation is a crucial instrument for controlling the provision of ecosystem services. Quantitative silvicultural guidelines are however largely limited to even-aged, both homogeneous and monospecific stand types (Bauhus et al., 2017a; Bauhus et al., 2017c). Given the increasing relevance of mixed species stands, the step from qualitative silvicultural descriptions towards quantitative rules of species-specific regulation is now overdue (Pretzsch et al., 2017). The thesis at hand therefore introduces an approach for implementing scientific knowledge about mixed species stands within an algorithm for quantitative control of species proportions at optimal utilization of growing space. This novel method is open for further refinements that will likely result from ongoing research on the effects of mixture on stand development. In particular, there are clues that the aforementioned mixing effects are not constant but dependent on site quality, mixing proportions, and stand age (Zhang et al., 2012). A well-reasoned candidate for implementing that relation could be a multiple regression model that explains a correction factor to stand density (see section 1.2.3) by site quality, mixing proportion and stand age.

2. Abstracts of the scientific publications

2.1. Contributions of the candidate

The author of this cumulative thesis at hand was responsible for the data sampling by execution of the simulations in Publication I, II, and III. Furthermore he implemented the forest management settings in all four Publications. Development work like conception and implementation of software solutions concerning SILVA and BALANCE was done in cooperation with and under leadership of Werner Poschenrieder. The author conducted the literature research, analyzed the data and was responsible for the preparation in Publication I, II, and III. The articles were processed together with coauthors. Astor Toraño Caicoya was responsible for Publication IV where the author of this thesis at hand was coauthor.

2.2. Lead authorship

2.2.1. Publication I: Groundwater recharge algorithm for forest management models

Schwaiger, F., Poschenrieder, W., Rötzer, T., Biber, P., Pretzsch, H., 2018. Groundwater recharge algorithm for forest management models. Ecological Modelling 385, 154–164. 10.1016/j.ecolmodel.2018.07.006.

Multifunctionality is a critical objective in forest management planning. Water related ecosystem services are only sparsely implemented in Forest Management Models (FMM) although water scarcity is highly relevant. This study proposes an approach to integrate groundwater recharge into an FMM. The approach is based on knowledge transfer between two different forest growth models. For site-specific simulations on the landscape level, observation-based models require functions that describe groundwater recharge in a non-

mechanistic way. However, groundwater recharge is difficult to measure and strongly depends on environmental conditions. Thus, we calibrated the observation-based FMM as dependent on site-conditions within two different case study areas, using a process-based forest growth model for substituting empiricism. Relations between forest structure and groundwater recharge were derived with multiple linear regressions and included in an FMM. The approach simulates groundwater recharge plausibly and as related to site conditions and stand management on landscape level. The amount of that recharge was remarkably influenced by tree species and stand structure at both sites. Groundwater recharge was between 30–50% of the occurring precipitation and higher within broadleaved stands. Exemplary simulation of a European beech - Norway spruce mixed forest stand reveals a trade-off between groundwater recharge and stand volume growth depending on forest management.

2.2.2. Publication II: Species mixing regulation with respect to forest ecosystem services provision

Schwaiger, F., Poschenrieder, W., Biber, P., Pretzsch, H., 2018. Species Mixing Regulation with Respect to Forest Ecosystem Service Provision. Forests 9 (10), 632. 10.3390/f9100632.

The control and maintenance of species composition inside mixed stands is a highly relevant objective of forest management in order to provide multifunctionality and climatic resilience. In contrast to this requirement there is, however, an evident lack of quantitative methods for mixture regulation. In this context, we propose an approach for the regulation of mixture proportions that has been implemented in a forest management model. The approach considers species-specific growth characteristics and takes into account the mixing effect on stand density. We present five exemplary simulations which apply that regulation method. Each simulation maintains one of five desired species compositions. In these simulations, we consider the species European beech and Norway spruce under good site conditions, thus representing the most prominent mixed stands in Central Europe. Based on this model experiment, we analyze the potential benefit of controlled mixing regulation for achieving desired levels and combinations of ecosystem service provision, in particular productivity, diversity, and groundwater recharge. We found that a constant 50% basal area share of beech (equivalent growing space share of 80% to 70% depending on stand age) provided the most balanced supply of ecosystem services. Groundwater recharge considerably decreased when beech basal area shares were held below 50%. We discuss the ecological and practical implications of the regulation approach considering various mixing proportions.

2.2.3. Publication III: Ecosystem service trade-offs for adaptive forest management

Schwaiger, F., Poschenrieder, W., Biber, P., Pretzsch, H., 2019. Ecosystem service trade-offs for adaptive forest management. Ecosystem Services 39, 100993. 10.1016/j.ecoser.2019.100993.

Quantifying ecosystem services as dependent on forest management and analyzing tradeoffs between them can help to make decisions on management more effective, efficient, sustainable, and stable. We use a forest management model (SILVA) to predict changes in ecosystem service provisions. Three stakeholder specific forest management scenarios (multifunctional, wood production, set-aside) for each of two different case study areas in Germany (a more and a less productive one) were simulated. We want to therewith answer how ecosystem service and biodiversity indicators (groundwater recharge, carbon sequestration, wood production, structural diversity of forest stands) depend on forest management and site. Forest management had significant influence on ecosystem service provisions in both case study areas. However, the results strongly depend on the site and on the initial situation in each location. In both case study areas, the production oriented forest management pays for productivity with structural diversity. In contrast, multifunctional oriented forest management pays for groundwater recharge with productivity losses. In the set-aside scenario, current carbon sequestration is high due to increasing forest carbon stocks, however sustainable carbon sequestration is low due to the lack of emission savings.

2.3. Co-authorship

2.3.1. Publication IV: Forestry projections for species diversity-oriented management: an example from Central Europe

Toraño Caicoya, A., Biber, P., Poschenrieder, W., Schwaiger, F., Pretzsch, H., 2018. Forestry projections for species diversity-oriented management: an example from Central Europe. Ecol Process 7 (1), 357. 10.1186/s13717-018-0135-7.

I**ntroduction**: Changes in socio-economy and climate are affecting the demand of wood products globally. At the same time, society requires that forest supporting structures like biodiversity are maintained and preserved while the demand for wood products is also covered. Management support systems, like forest simulation models, that are able to analyze connections as well as quantify trade-offs between forest structure management and biodiversity indicators are highly sought. However, such models are generally developed for the local plot or stand scale only and ecosystem-scale analyses are missing. In this study, we analyzed ways to interpret results from the single-tree forest simulator SILVA from the local to the ecosystem scale. We also analyzed the impacts of forest management on biodiversity using two species diversity indicators, the species profile index and the species intermingling, for

scenarios adapted from the GLOBIOM model in the case study “Augsburg Western Forests”, a high productive region in South-Germany. In order to evaluate diversity tendencies across the ecosystem, we applied a moving window methodology.

Results: The relevance of scale for the interpretation of management effects on species diversity was shown and clear differences between scenarios revealed. The differences between scenarios were particularly visible when comparing the two diversity indicators, especially because the species profile index focuses on vertical and horizontal information and the species intermingling focuses mainly on horizontal structures. Under a multifunctional scenario, biodiversity values could be preserved at all scales in the vertical dimension. However, under a bio-energy-oriented scenario diversity at the local scale was reduced, although at the ecosystem level, and only in the horizontal dimension, diversity remained at relatively high values.

Conclusions: With this work, we can conclude that integrative modeling, with multiple scenarios, is highly needed to support forestry decision making and towards the evolution of forest management to consider the ecosystem scale, especially when the optimization of diversity is a management priority.

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Groundwater recharge algorithm for forest management models

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ABSTRACT

Multifunctionality is a critical objective in forest management planning. Water related ecosystem services are only sparsely implemented in Forest Management Models (FMM) although water scarcity is highly relevant. This study proposes an approach to integrate groundwater recharge into a FMM. The approach is based on knowledge transfer between two different forest growth models. For site-specific simulations on the landscape level, observation-based models require functions that describe groundwater recharge in a non-mechanistic way. However, groundwater recharge is difficult to measure and strongly depends on environmental conditions. Thus, we calibrated the observation-based FMM site-specific for two different case study areas, using a process-based forest growth model and substitute empiricism. Relations between forest structure and groundwater recharge were derived with multiple linear regressions and included in a FMM. The groundwater recharge was remarkably influenced by tree species and stand structure at both sites. The approach simulates groundwater recharge plausibly depending on site conditions and stand management on landscape level. Groundwater recharge was between 30–50% of the occurring precipitation and higher within broadleaved stands. Exemplary simulation of a European beech - Norway spruce mixed forest stand reveals a trade-off between groundwater recharge and stand volume growth depending on forest management.

1. Introduction

In the face of climate change, forest development must be aligned to meet a broader range of tasks as in the past. Accordingly, modern forest management must consider a wide spectrum of ecosystem services. The Helsinki Criteria (MCPFE, 1993) implicates changes in forests that are managed with a focus on wood production and the maximization of financial gain toward more multifunctional forest ecosystems. Because of the strong paradigm shift within the past decades, it has become increasingly important to have an understanding of the manner in which sensitive ecosystem service provisions react to forest management and the possible conflicts and compatibility of various services (Biber et al., 2015). Practicing sustainable development remains a challenge today (Pandeya et al., 2016). In Germany, for example, there are efforts by several political parties to pay forest owners for the provisioning of modern ecosystem services and supportive forest management practices (DFWR, 2017).

Water is an increasingly critical economic factor in decision-making in industries such as mining, power, and tourism (WWAP, 2012). Today's global water withdrawal consists of one third of groundwater (Kundzewicz and Döll, 2009). By 2025, half of the world's population will be living in water-stressed areas (WHO, 2017). When considering the total water requirements of society and ecosystems, even more humid areas such as substantial parts of Europe, North America, South-West Australia, and South America are prone to water scarcity (Rijsberman, 2006). Water scarcity, on the one hand, is a result of increasing consumption and decreasing availability of water.

The human population and consumption of water per person are increasing (FAO, 2011). Industries, agriculture, and municipalities are the biggest consumers of water. Agriculture is the biggest consumer worldwide. Within Europe, industries have the largest demand for water (FAO, 2011). According to the Food and Agriculture Organization (FAO), freshwater is not being used efficiently. In addition to increasing water withdrawal, the second reason for water scarcity is

Abbreviations: AWF, Augsburg Western Forests; CCF, Continuous Cover Forestry; et_a, actual evapotranspiration; fc, field capacity; FMM, Forest Management Model; GSS, species-specific growing space share; GWR, groundwater recharge; h, tree height; Lnr, soil layer number; LSN, Lieberose Schlaubetal Neuzelle; MH, arithmetic mean height; n, number of trees; NFI, National Forest Inventory; PAWC, plant available water content; per, percolation; prec, precipitation; PWP, permanent wilting point; qmd, quadratic mean diameter; SDI, Stand Density Index; swc, soil water content; VH, vertical heterogeneity

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decreasing water availability. According to Natkhin et al. (2012), a decline in groundwater levels is recognizable across several regions in northeast Germany. Turral et al. (2011) stressed on the ways in which climate change influences water availability. They underpin that climate change will significantly reduce the recharge of groundwater in dry regions like South America and Africa. In Europe, climate change brings higher probability of droughts. Groundwater serves as the primary buffer for decreasing water supply. Therefore, it is highly important to be aware of the ways in which land use and landscape structures govern the availability of water.

Forests normally consume more water than cropland (Müller, 2011). During droughts, however, forests become more efficient and consume much less water (Zimmermann et al., 2008). Intensive agricultural land use is challenged with groundwater pollution. Therefore, about half of the water protection area in Germany is within forests (LfU, 2015). An upcoming issue that is often discussed is the development of a water cycle that is dependent on climate change and land cover (Peel, 2009; Oudin et al., 2008). However, only few investigations consider forest stand structure (Natkhin et al., 2012). Beside soil and climate conditions, the composition of tree species and stand structure influence groundwater recharge (Müller, 2013). Typical variables for describing the forest stand characteristics are tree species, stand density, tree height, and vertical heterogeneity. Findings in the literature concerning the magnitude of groundwater recharge differ strongly from each other. This is because of the differences between precipitation quantity and other site and stand conditions. As a benchmark, a range between 20–50% of the precipitation is a plausible quantity of groundwater recharge (Anders et al., 2004; Rust, 2009; Müller, 2011; Sutmöller and Meesenburg, 2012; Müller, 2013).

Groundwater recharge has been proved to depend on tree species. Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) recharges less groundwater than Norway spruce (*Picea abies* (L.) H.Karst.) and Scots pine (*Pinus sylvestris* L.), and these two tree species recharge less groundwater than European beech (*Fagus sylvatica* L.) (Prietzel and Bachmann, 2011). Sutmöller and Meesenburg (2012) reported a significant difference between Norway spruce and European beech in their groundwater recharge. Norway spruce stands form less groundwater than European beech stands. Groundwater recharge primarily occurs outside the vegetation period. During winter, deciduous species almost consume no water, whereas conifers transpire and consume water (Hölting and Coldewey, 2013; Rötzer et al., 2017). This is the main reason for which conifers form less groundwater than deciduous trees. Previous work have also reported that species with low growth provide higher groundwater recharge (Pöhler et al., 2013). Ilstedt et al. (2016) and Sutmöller and Meesenburg (2012) show that groundwater recharge was maximized at intermediate tree densities. Müller (2011) demonstrated that stands with small trees form more groundwater than stands with big trees, but the influence of the tree species is higher than the influence of the tree height. Delzon and Loustau (2005) show an age-related decline in stand water use. Therefore, previous work connote contrary findings about the impact of stand height on groundwater recharge.

Although the process of creating concepts for ecosystem services in science and politics has an obvious advantage, the quantitative assessment of ecosystem services is still a challenge. This is particularly true when local decision makers require local information. For that purpose, simulation techniques play an increasingly important role (Nelson et al., 2009). Projects that investigate the long-term effect of forest management on the provision of ecosystem services, like ALTERFOR (Alternative models for future forest management) (ALTERFOR, 2017), are fully based on landscape scale simulation scenarios. Therefore, management models must now provide a broad range of ecosystem service results. The simulations of silvicultural treatment, which are part of forest management models, enable the investigation of forest management with respect to all relevant ecosystem services. Simulation models that intrinsically represent the interaction between various driving forces and management are an essential tool for estimating the long-term management effects on ecosystem services. In particular, water balance and landscape development are predestined for model evaluation because they depend on the interaction of many boundary conditions and are difficult to be depicted in empirical results (Pandeya et al., 2016).

The objectives of this study, which is based on two case study areas are:

(1) to present an algorithm that integrates quantitative groundwater recharge into forest management models
(2) to analyze whether there are species-specific relations between forest stand structure and groundwater recharge
(3) to examine whether there is a trade-off between productivity and groundwater recharge

2. Material and methods

2.1. Procedure

Our approach combines a process-based forest growth model with an observation-based one (Fig. 1). Therefore, we derive multilinear

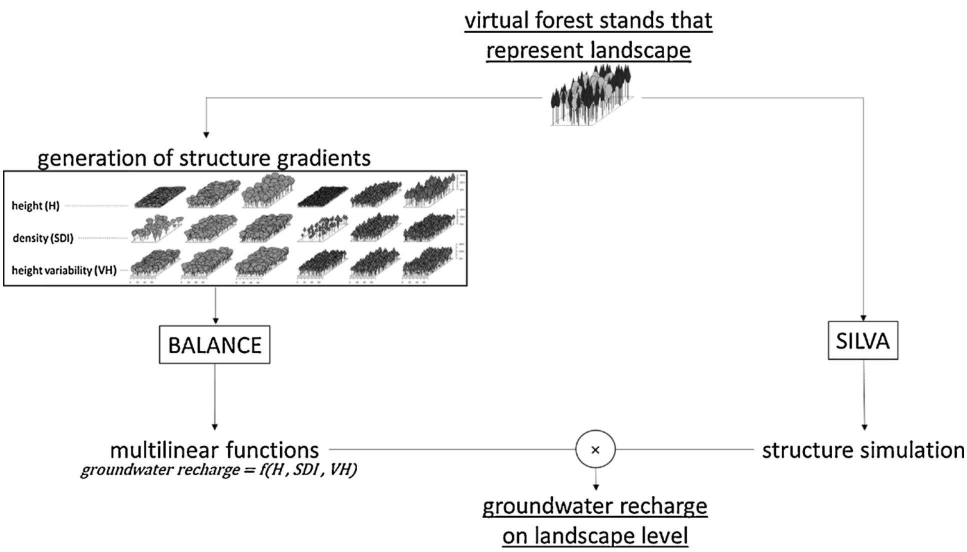

Fig. 1. Conceptual diagram of the approach for simulating groundwater recharge on the landscape level by means of an observation-based forest management model (SILVA) and a process-based forest growth model (BALANCE). Structure gradients that cover the range inside the landscape were generated. BALANCE was used to derive multilinear functions for estimating groundwater recharge as dependent on structure variables. These functions are applied to stand structures obtained from simulations using SILVA.

functions that describe groundwater recharge as dependent on forest stand structure attributes. To that end, we use the process-based eco-physiological growth model BALANCE (Grote and Pretzsch, 2002; Rötzer et al., 2017). We then use those functions to calculate groundwater recharge based on simulated stand structure development obtained from the practice-oriented management model SILVA (Pretzsch and Kahn, 1998; Pretzsch et al., 2002).

The stand structure range within the case study areas was initially analyzed. Virtual stands were then generated for obtaining stand structure gradients. Subsequently, groundwater recharge was quantified for each structure type. Finally, the productivity and groundwater recharge of one stand within the case study area was exemplarily simulated with the forest management model.

2.2. Software tools (BALANCE, SILVA, STRUGEN)

BALANCE is a detailed and complex process-based forest growth model (Grote and Pretzsch, 2002; Pretzsch et al., 2015; Rötzer et al., 2017). Growth response is simulated at the tree level by considering the influences of competition, stand structure, species mixture, and management impacts. BALANCE simulates tree growth on the base of environmental conditions and their changes over time. Dendrometrical characteristics of trees change with interactions between multiple physiological processes that depend on physical and chemical site conditions. The altered structure of trees, such as asymmetric crown shapes, generate spatially explicit patterns within the stand. The processes are calculated for several crown and root layers. Therefore, the growth within the tree can be calculated spatially explicit for each layer based on energy supply and resource availability.

The most fundamental processes of BALANCE are the carbon, water, and nutrient flows. Daily inputs of temperature, radiation, precipitation, humidity, and wind velocity form the basis of calculating the micro-climate and water balance of each layer. The physiological processes assimilation, respiration, nutrient up-take, growth, senescence, and allocation affect the dimensional tree growth that is calculated once a year. In this manner, tree growth and the development of site conditions like water balance and percolation can be simulated but this model approach requires detailed input data concerning weather and soil conditions. Eq. (1) describes the water balance calculation within BALANCE (Rötzer et al., 2017):

$$\Delta \mathrm{swc} = \mathrm{prec} - \mathrm{et_a} - \mathrm{per} \tag{1}$$

swc = soil water content, Δswc = change in swc, prec = precipitation, $\mathrm{et_a}$ = actual evapotranspiration, per = percolation

The actual evapotranspiration ($\mathrm{et_a}$) is the sum of transpiration, interception, and soil evapotranspiration and is based on the approach of Penman-Monteith (e.g., Allen et al., 1998; DVWK, 1996). For this study, we assume that groundwater recharge (GWR) can be considered as the sum of percolation (per) from the lowest soil layer. Percolation is the reduction of soil water content to field capacity (fc), and arises if soil water content is above field capacity.

$$\text{if:} \quad \mathrm{swc} > \mathrm{fc} \quad \text{than:} \quad \mathrm{per} = \mathrm{swc} - \mathrm{fc} \tag{2}$$

In BALANCE all parameters of the water balance (e.g. Eqs. (1) and (2)) are calculated continuously in daily time steps.

SILVA (Pretzsch and Kahn, 1998; Pretzsch et al., 2002, 2015; Pretzsch et al., 2017; Pretzsch, 2002) is an empirical forest management model based on individual tree growth. The simulation results comprise estimates of growth and yield, including ecological and socio-economic indicators. This growth model breaks stands down into a mosaic of individual trees and simulates the interactions in a space-time system. This approach follows a combination of process and observation-based modeling, and uses only a few initial stand parameters, rough site variables, and silvicultural prescriptions, and stand dynamics are simulated in five-year cycles. This time interval is also the standard time interval for measuring on the trial plots that were used for model evaluation. Each simulation cycle consists of four steps. The first step is the quantification of spatial growth conditions of a tree by calculating a competition index. The second step is the identification of trees that will be removed according to the rules of a user-defined thinning concept. Following which, dimension changes are calculated for all trees inside the stand. In the fourth step, a mortality model uses the previously calculated competition index to determine which trees did not survive.

If a detailed inventory of a stand is not available, the missing data for simulation can be generated with **STRUGEN** (Pretzsch, 1993, 1997; Pretzsch, 2010). STRUGEN is a software tool that consists of an algorithm for stand structure generation. Based on a tree list or stem number-diameter distribution, a realistic stand can be established. For the position of a tree, the stand area is covered with random, uniformly distributed x- and y- coordinates. To manufacture desired structures of a stand, the generated dot coordinates are accepted with different levels of probability. In essence, the points must pass through some filters that regulate macro structures. For example, only those positions are accepted that have an appointed distance to neighboring trees that are generated. This process is repeated until the desired diameter distribution of the tree species is achieved. Subsequently, further processes are performed to introduce additional tree species.

This completion of required values and the creation of first structure states provides realistic initialization data for simulation. Agreement between the real and created structures does not imply that each tree in the stand must have the same position, but the characteristics of the structure can correspond.

2.3. Stand structure within the case study areas

The size of the case study area Lieberose Schlaubetal Neuzelle (LSN) in North East Germany, Brandenburg is 90.000 ha. The tree species comprise about 65% Scots pine and 11% sessile oak (*Quercus petraea* Mattuschka) Liebl.). The size of the case study area Augsburg Western Forests (AWF) in South Germany, Bavaria is 60.000 ha and the tree species comprise about 62% Norway spruce and 11% European beech.

Based on data from the third German National Forest Inventory (NFI) (Thünen-Institut, 2017), we created virtual stands by using STRUGEN (Pretzsch, 1997). Those virtual stands represent the landscape of the case study areas. The virtual stands were then analyzed according to the stand structure. Three structure indices suitable for characterizing stand structure were selected and calculated for each stand. The first index is the stand density index (SDI) according to Reineke (1933).

$$SDI = n\left(\frac{qmd}{25}\right)^{1{,}605} \tag{3}$$

qmd = quadratic mean diameter in cm, n = number of trees per hectare

The Second index is the arithmetic mean height (MH) of a stand.

$$MH = \frac{1}{n}\sum_{i=1}^{n} h_i \tag{4}$$

h = tree height

The third indicator is the variations coefficient of the tree height and includes information concerning the vertical heterogeneity (VH).

$$VH = \frac{1}{MH}\sqrt{\frac{1}{n-1}\sum_{i=1}^{n}(h_i - MH_i)^2} \tag{5}$$

variables as in Eq. 3 and 4

Table 1 presents the characteristics of the case study areas. Comparing both areas, a higher range and average of all structures in AWF is apparent, excluding the vertical heterogeneity that has a slightly lower average than in LSN.

Table 1
Structure values within case study areas AWF and LSN. MH = arithmetic mean height, VH = vertical heterogeneity, SDI = stand density index.

	AWF			LSN		
	min	max	mean	min	max	mean
SDI [trees ha^{-1}]	125	1256	774	240	910	582
MH [m]	6.1	35.5	22.7	8.0	28.0	18.8
VH [a.u.]	0.05	0.44	0.20	0.05	0.40	0.21

Table 2
Soil characteristics of the two sites in the case study areas AWF and LSN. LNr = layer number, up = upper border, down = lower border, fc = field capacity, PWP = permanent welting point, PAWC = plant available water content.

case study area	LNr	up [cm]	down [cm]	fc [mm/dm]	PWP [mm/dm]	PAWC [mm/dm]
AWF	1	8	0	38.4	13.4	25
	2	0	−3	50.4	28.8	21.7
	3	−3	−7	44.2	15.9	28.3
	4	−7	−30	39	13	26
	5	−30	−50	40.3	20.2	20.2
	6	−50	−75	39.4	25.1	14.4
	7	−75	−90	36.9	25.2	11.7
	8	−90	−125	36.3	21.1	15.2
LSN	1	5	0	38.4	13.4	25.0
	2	0	−5	28.4	3.2	25.2
	3	−5	−10	25.1	2.1	23.0
	4	−10	−30	20.2	1.6	18.6
	5	−30	−60	19.0	1.5	17.4
	6	−60	−90	19.3	2.3	17.0

2.4. Site conditions: soils and climate

The prevalent soil type in LSN (Table 2) consists of sand-dominated base-poor unconsolidated rocks with reduced nutrient supply. In contrast, soils in AWF (Table 2) are generally lowland soils consisting of loess clay with high nutrient supply (Grüneberg et al., 2016; Andreae et al., 2016). For the case study area in Bavaria (AWF), moisture data can be summarized with plant available water content of 257 mm in a profile with a depth of 125 cm. In Brandenburg (LSN) plant available soil water content of 177 mm characterizes the 90 cm deep profile. The soil characteristics are taken from Level II permanent observation plots, which form an international program (ICP-Forests, 2011). The level II plots are uniformly established in Europe and were selected representing the conditions of the landscapes (Forst Brandenburg, 2017). The recordings of the Level II plots were used to calculate the plant available water content with the pedotransferfunction Hypres (Wösten et al., 1999) inside the hydrologic balance model LWF-Brook90 (Hammel and Kennel, 2001).

The climate conditions were provided from the International Institute for Applied Systems Analysis (IIASA) within the ALTERFOR project and are based on the climate model HadGEM2-ES (Collins et al., 2011). The used developments depict climatic conditions considering climate change over the last 50 years. Within the considered time range, temperature increased by 1 °C and the precipitation quantities remained mostly constant (Fig. 2 given by decade). Table 3 presents the mean values for the whole simulation time. The precipitation average in LSN is slightly less than two thirds of the one in AWF. The average of the annual mean temperature and the wind speed are higher in LSN, whereas the average of the radiation and the relative humidity are higher in AWF.

2.5. Generating stands for obtaining structure gradients

To parameterize groundwater recharge functions, we generated five

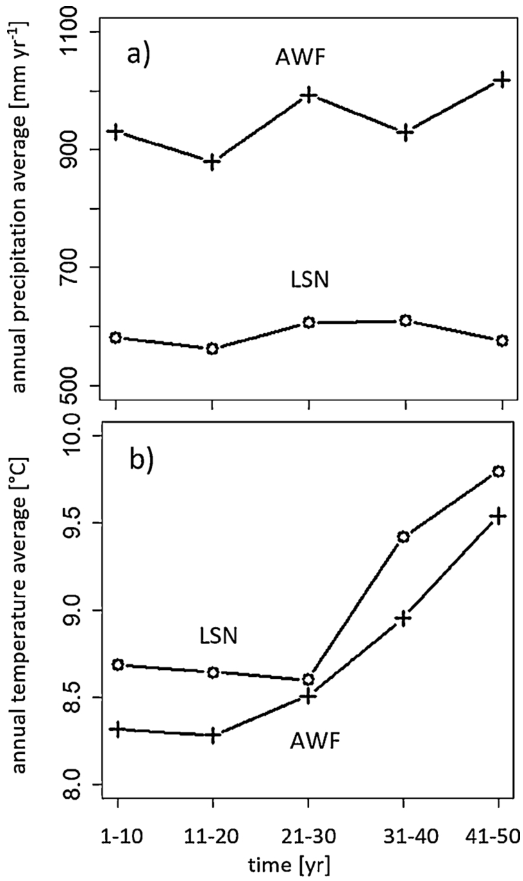

Fig. 2. Description of the climate conditions, which consider climate change over the last 50 years. Case study areas: LSN, AWF; development of the annual precipitation (a) and temperature average (b) is shown in decadal steps.

Table 3
Description of the climate conditions per case study area. Climate data average over 50 yr.

case study area	precipitation [mm/yr]	annual mean temperature [°C]	radiation [J/cm²/d]	relative humidity [%]	wind speed [m/s]
AWF	937.8	8.8	892.2	80.8	2.3
LSN	587.2	9.0	819.8	75.8	2.8

representative stand structures for each species and structure index i.e., model predictor. To that end, we considered two species with the highest share per case study area. (Bavaria: Norway spruce, European beech; Brandenburg: Scots pine, sessile oak). For each species and each of the three predictors, we derived a set of five monospecific stands (Fig. 3). Within each of these predictor-related sets, the stands varied in the value of the related predictor, while the remaining predictors were held at their landscape specific average. In this manner, a total of 60 stands was generated, i.e., 15 stands per tree species. In Fig. 3, this setup is visualized for spruce. The set of virtual stands for each predictor covers the predictor's entire value range within the case study area considered (Table 1). For example, the middle row in Fig. 3 shows five stands that represent the landscape-specific spectrum of the stand density. Similarly, each stand represents the average of the predictors in the top and bottom row. Therefore, in the middle of Fig. 3, all stands have approximately the same VH and arithmetic MH but vary in stand density.

To generate the desired stands, corresponding data for STRUGEN suitable for achieving the desired structure values during the generation process is required. STRUGEN requires the number of trees as well as

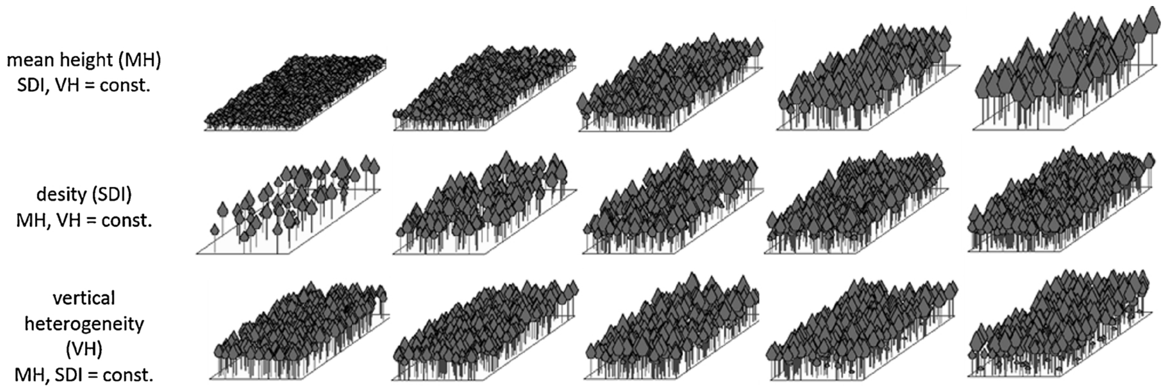

Fig. 3. Exemplary visualization of Norway spruce stands, which were generated along each of three structure gradients (SDI = stand density index, MH = arithmetic mean height, VH = vertical heterogeneity). Those stands were generated for Norway spruce, European beech, sessile oak, and Scots pine and are BALANCE input data. The value range per structure variable demonstrates the situation within the case study area. While raising one variable, both remaining structure variables were set to their mean value within the case study area. Structure values per generated stands are presented in Table 4.

the basal area per height class. To generate stands with a desired VH, one set of heights was sampled from an empirical height distribution within the case study area considered. Each height value from that set was associated to a corresponding tree diameter, which is defined as the mean diameter of all trees in the case study area of the corresponding height and tree species. To achieve the desired densities, the total number of trees was determined using the quadratic mean diameter and the SDI (Eq. (3)).

BALANCE simulations require buffer trees that surround the selected stand and serve to counteract edge-effects. Those trees are needed for creating the same conditions at the edge of a stand. To obtain a stand size of 0.25 ha, the side length of the generated stand with STRUGEN was 70 m. Trees with a distance to the edge of smaller than 10 m count as buffer trees.

Table 4 presents the structure values of each generated stand. The values according to the stands in the middle of Fig. 3 are signed in the middle of Table 4 with “SDI1-5_sp” on the right-hand side.

Table 4
Structure values of the generated stands along gradients in both case study areas AWF and LSN. VH = vertical heterogeneity, SDI = stand density index, MH = arithmetic mean height, pi = Scots pine, sp = Norway spruce, oa = sessile oak sp., be = European beech.

LSN				AWF			
stand	SDI [trees ha^{-1}]	VH [a.u.]	MH [m]	stand	SDI [trees ha^{-1}]	VH [a.u.]	MH [m]
VH1_pi	698	0.10	22.1	VH1_sp	658	0.09	23.0
VH2_pi	698	0.16	22.0	VH2_sp	753	0.15	22.8
VH3_pi	707	0.24	21.7	VH3_sp	683	0.21	22.6
VH4_pi	794	0.32	22.2	VH4_sp	890	0.24	24.5
VH5_pi	892	0.47	21.4	VH5_sp	798	0.36	23.3
VH1_oa	684	0.11	21.8	VH1_be	703	0.08	23.0
VH2_oa	696	0.17	21.8	VH2_be	707	0.13	22.8
VH3_oa	698	0.25	21.4	VH3_be	687	0.22	22.8
VH4_oa	807	0.32	22.0	VH4_be	829	0.25	24.2
VH5_oa	897	0.48	21.1	VH5_be	772	0.36	22.9
SDI1_pi	266	0.21	18.4	SDI1_sp	133	0.21	22.6
SDI2_pi	385	0.21	17.9	SDI2_sp	345	0.20	22.3
SDI3_pi	536	0.20	18.1	SDI3_sp	677	0.19	22.9
SDI4_pi	722	0.22	18.4	SDI4_sp	998	0.19	22.9
SDI5_pi	930	0.20	18.5	SDI5_sp	1145	0.19	23.0
SDI1_oa	289	0.21	18.6	SDI1_be	105	0.18	21.9
SDI2_oa	403	0.22	17.8	SDI2_be	416	0.18	23.2
SDI3_oa	537	0.21	18.0	SDI3_be	717	0.19	22.8
SDI4_oa	722	0.22	18.2	SDI4_be	894	0.19	22.5
SDI5_oa	925	0.21	18.4	SDI5_be	1171	0.18	22.9
MH1_pi	557	0.22	7.7	MH1_sp	793	0.21	7.2
MH2_pi	525	0.22	12.4	MH2_sp	648	0.19	13.1
MH3_pi	573	0.22	17.4	MH3_sp	779	0.20	20.6
MH4_pi	604	0.20	22.5	MH4_sp	767	0.18	28.2
MH5_pi	628	0.19	28.1	MH5_sp	841	0.20	34.8
MH1_oa	555	0.22	7.7	MH1_be	789	0.17	7.3
MH2_oa	545	0.21	12.5	MH2_be	693	0.17	13.4
MH3_oa	586	0.22	17.3	MH3_be	732	0.17	20.7
MH4_oa	586	0.21	21.9	MH4_be	830	0.20	28.2
MH5_oa	626	0.20	27.8	MH5_be	662	0.21	34.6

2.6. Simulation of groundwater recharge per stand of structure gradient

For each of the two case study areas, exactly one climate development was used. Both developments were fifty years long. Each climate development was divided into five consecutive ten-year periods. Each generated stand of the structure gradient was simulated with BALANCE for each of the five development periods. In this manner, we maintained consistency in the structures used for groundwater recharge function calibration and the structures employed during dynamic simulation. For each stand, the results of all corresponding simulation runs were averaged. Therefore, each stand with a known structure combination could be matched with one value of groundwater recharge.

The AIC (Akaike, 1974) procedure, provided by Venables and Ripley (2002), served for species-specific, stepwise model selection. This procedure only accepts predictors that sufficiently contribute for estimation, otherwise they are not suggested for being part of the model. Thus, four statistical linear models were created. Those models are species-specific groundwater recharge functions that have the general form of Eq. (6).

$$GWR = \text{y-intercept} + a \cdot MH + b \cdot VH + c \cdot SDI \tag{6}$$

GWR = groundwater recharge, MH = arithmetic mean height, VH = vertical heterogeneity, SDI = stand density index.

In addition, the beta coefficients of each statistical model were calculated to enable a comparison of the slope and the influence among the model's predictors. Each beta coefficient results from a regression that uses standardized predictors, each with a variance of one. Each predictor was divided by its variance. For clarity and validation, the groundwater recharge average of all five climate time steps was plotted over the structure gradients. Wherever applicable, a line of linear regression was added to the plots.

2.7. Exemplary test application with the forest management model SILVA

To demonstrate the application of the results, a mixed European beech and Norway spruce stand was simulated by the forest management model SILVA. Two management pathways were simulated. One

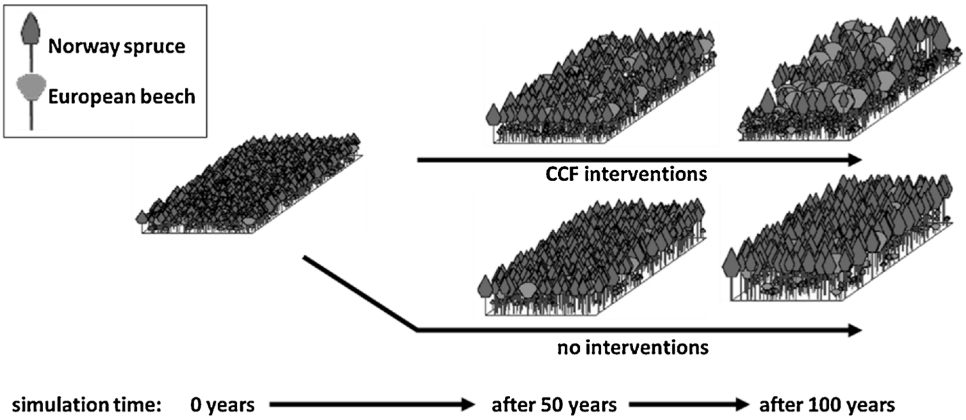

Fig. 4. Illustration of the exemplary test simulation (100 years). An initial Norway spruce and European beech stand (left) was simulated with two different silvicultural activities. The upper area (CCF interventions) shows the development of continuous cover forestry (CCF). CCF reduces the stock to enable vertical structure and fosters broadleaved trees. In contrast, the pathway of no interventions (below) represents a set aside.

assumed that no interventions took place. The other one, however, applied thinning and harvest interventions to establish Continuous Cover Forestry (CCF). The structure development of the initial stand from period zero up to 100 years notably depends on the treatment scenario (Fig. 4): the CCF forest management is characterized by future crop tree thinnings combined with target diameter harvests. One objective of forest management is to maintain stock at a level that provides a continuous cover. The other objective is the rise of the broadleaved species proportion.

Within each of the two management pathways, the statistical groundwater model was applied to the resulting stand structure on a per-time-step-basis. To that end, the model's predictors (see section 2.3, Table 1) were calculated for each simulation period. Then, for each time step, the species-specific groundwater recharge was calculated (Eq. (6)). The unweighted species-specific contribution ($GWR_{species}$) was then weighted by mixing the proportion ($GSS_{species}$) of each tree species (Eq. (7)):

$$GWR_{total} = GSS_{species1}\ GWR_{species1} + GSS_{species2}\ GWR_{species2} \tag{7}$$

GWR_{total} = groundwater recharge of mixed stand, $GWR_{species}$ = species-specific groundwater recharge contribution, $GSS_{species}$ = species-specific growing space share

The mixing proportion for each species was calculated as the species-specific share of crown projection area, i.e., growing space.

3. Results

3.1. Structure dependent groundwater functions

The relation of groundwater recharge to structure was species-specific (Fig. 5). Furthermore, it is discernible that there was a higher level of groundwater recharge in AWF than in LSN. Norway spruce was sensitive to all structure variables. The VH of Norway spruce demonstrated less influence than its arithmetic MH. Within the simulated Norway spruce stands, density was the most important predictor. Whereas, the lines of the linear regressions for European beech have low slopes. At most, VH caused a notable drift in groundwater recharge. Sessile oak has a horizontal regression line at the SDI, but the groundwater recharge was decreasing with increasing arithmetic MH and vertical structure. Scots pine was only reacting on arithmetic MH. Obviously, in LSN as well as in AWF, within the virtual coniferous species stands percolation was lower than in the broadleaved species stands.

The groundwater recharge of European beech and Scots pine was only dependent on one structure variable (Table 5). In contrast, the recharge values of sessile oak were dependent on arithmetic MH and VH. Moreover, the groundwater recharge of the virtual Norway spruce stands was clearly influenced by all structure variables, MH, VH, and SDI. It is necessary to consider that it is not possible to check all species in this study against each other. Only the species of one case study area are comparable among each other because they rely on the same environmental conditions. Each species was solely investigated within one case study area. Comparing Norway spruce and European beech (AWF), the broadleaved species has a higher y-intercept than the coniferous. The same is true for Scots pine and sessile oak (LSN), where the same relationship is discernible.

The standardized beta coefficients of Norway spruce in AWF underpin a stronger influence of the arithmetic MH on groundwater recharge than of the VH. Nevertheless, the arithmetic MH of a Norway spruce stand in AWF seems to have less influence than the SDI. In such a Norway spruce stand, however, VH had less influence than in a European beech stand. Within sessile oak stands, the arithmetic MH had a larger effect than in a Scots pine stands (LSN).

3.2. Exemplary test simulation

The exemplary simulation of over 100 years that assumed silvicultural treatment, compared to the untreated one, leads to higher values of the Norway spruce proportion and, furthermore, of the stocking volume (Fig. 6a and b). The Norway spruce share in the untreated stand is approximately constant whereas the curve of the treatment scenario develops toward higher shares of European beech. Furthermore, the untreated stock increases throughout the entire simulation time, but the stock in the treatment scenario is constant between 400 and 500 m^3. Therefore, within that scenario, a CCF system has been established.

Density in the treatment scenario first decreases and then oscillates within a sustained level. In contrast, the density in the simulation without treatment is higher and approximately constant. Initially the vertical structure slightly falls in the treated and in the untreated stand. The development without treatments is then by continuous increase. The treated stand's VH increases rapidly within a short initial period and then remains constant. The arithmetic MH is lower in the treated stand than in the untreated stand.

Norway spruce inside the exemplary mixed stand contributed between 300 and up to 400 mm yr^{-1} to groundwater recharge (unweighted contribution, see Section 2.7, Fig. 7a). Within that scope, the treated stand had considerably higher recharge values than the untreated one. The unweighted contribution of European beech (Fig. 7b) remained almost constant and was nearly independent of the treatment scenario. European beech has higher unweighted contribution of groundwater recharge than spruce. As a coherent outcome, the mixed species stand groundwater recharge obtained as the weighted sum of the contributions of both species (see Section 2.7) shows an increasing influence of European beech. It overrides a development that is initially governed by the Norway spruce curve.

The stand volume growth decreases during simulation time (Fig. 7c). This trend, is visible in both management pathways, but is

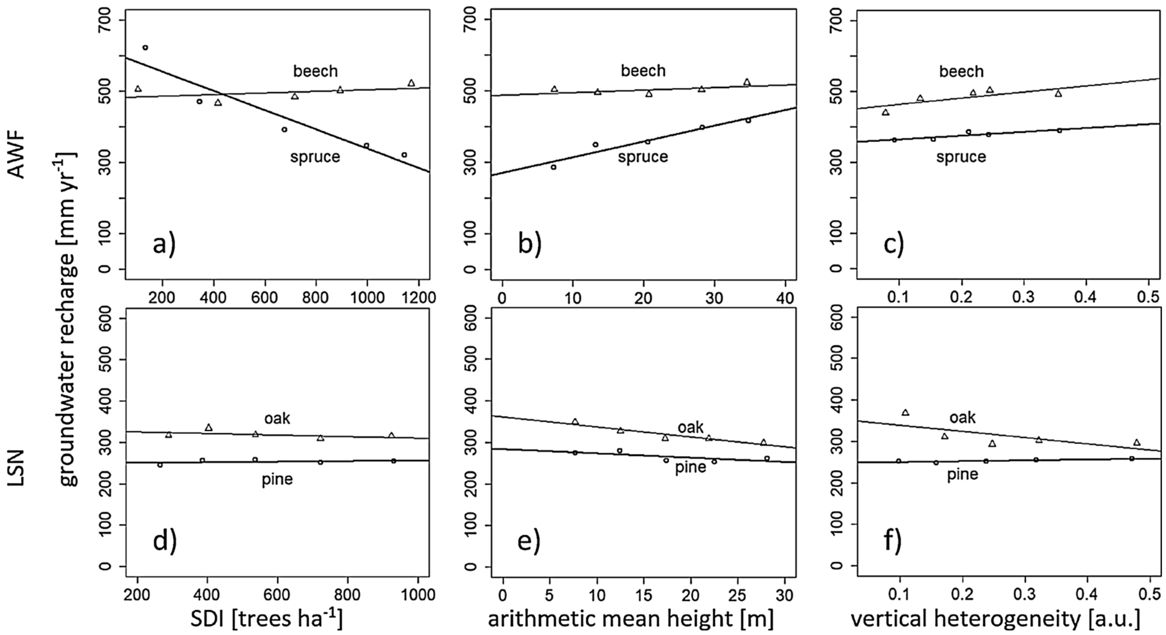

Fig. 5. Groundwater recharge over each structure index per case study area AWF (a, b, c), LSN (d, e, f,).

stronger in the treatment scenario.

There was a clear trade-off between stand volume growth and groundwater recharge (Fig. 8). The treated as well as the untreated scenario demonstrate this trend, but the scenario of the treated version is steeper. Higher groundwater recharge, thus, was paid for with lower stand volume growth.

4. Discussion

4.1. Relevance of forest management for ecosystem services provision

To improve sustainability, current forest management planning must consider a multitude of ecosystem services on the background of various planning alternatives (Lasch et al., 2005). Water scarcity steadily becomes more critical (WWAP, 2012). A relevant process on that background is the effect of forest management activities on the groundwater system. That interaction, however, has rarely been investigated (Smerdon et al., 2009).

This study shows that forest management can raise groundwater recharge, reduce stand productivity, change tree species composition, and thus influences ecosystem services. Consequently, it is important to be aware of the ways in which management affects ecosystem service provisioning. Possibly, trade-offs between ecosystem services are particularly related to certain management systems.

Therefore, our study presents a method for revealing possible conflicts between groundwater recharge and production on the landscape scale level. To that end, we developed a novel approach and applied it to test its suitability for trade-off analyses within two treatment scenarios. As also underpinned by previous work (Pöhler et al., 2013), there was a clear trade-off between groundwater recharge and productivity. Both treatment variants that we exemplified exhibit a trade-off in the same direction, but of different slope. Considering the CCF variant, that result may be explained by a decrease in wood production and a concomitant increase in groundwater recharge. Within that variant, stand productivity declined because of a decreasing stand density and a drop in the proportion of spruce. Groundwater recharge, conversely, became larger because of an increase in the proportion of beech and a concomitant decrease in stand density. Within the non-thinned variant, only a decline in productivity caused the trade-off, and groundwater recharge was almost constant. That decline in productivity was likely because of raising mortality resulting from increased stock and tree density.

4.2. Validity of the results

4.2.1. Implementation procedure

Simulation results are always as exact and valid as the modeling approach allows. This approach stripped-down the modeling of the most important predictors for groundwater estimation to reduce the aimed prediction to essential variables strongly correlating with forest management. In particular, this approach underpins the practicability of a method that integrates relations from a mechanistic approach into

Table 5
Results of multiple linear regressions with stepwise AIC procedures (Venables and Ripley, 2002). Used general form of the model: GWR = y-intercept + a·MH + b·VH + c·SDI; This function enables the calculation of groundwater recharge dependent on structure variables. MH = arithmetic mean height; VH = vertical heterogeneity; SDI = stand density index; GWR = groundwater recharge; a, b, c = slops; Beta coefficients enable to compare the influence of the structure variables.

case study area	tree species		slopes for groundwater functions			beta coefficients for slopes		
		y-intercept	MH	VH	SDI	MH	VH	SDI
AWF	European beech	463.71		154.61			0.19	
	Norway spruce	418.72	5.61	234.46	−0.28	0.38	0.14	−0.74
LSN	sessile oak	386.19	−2.01	−132.54		−0.30	−0.35	
	Scots pine	279.91	−1.17			−0.17		

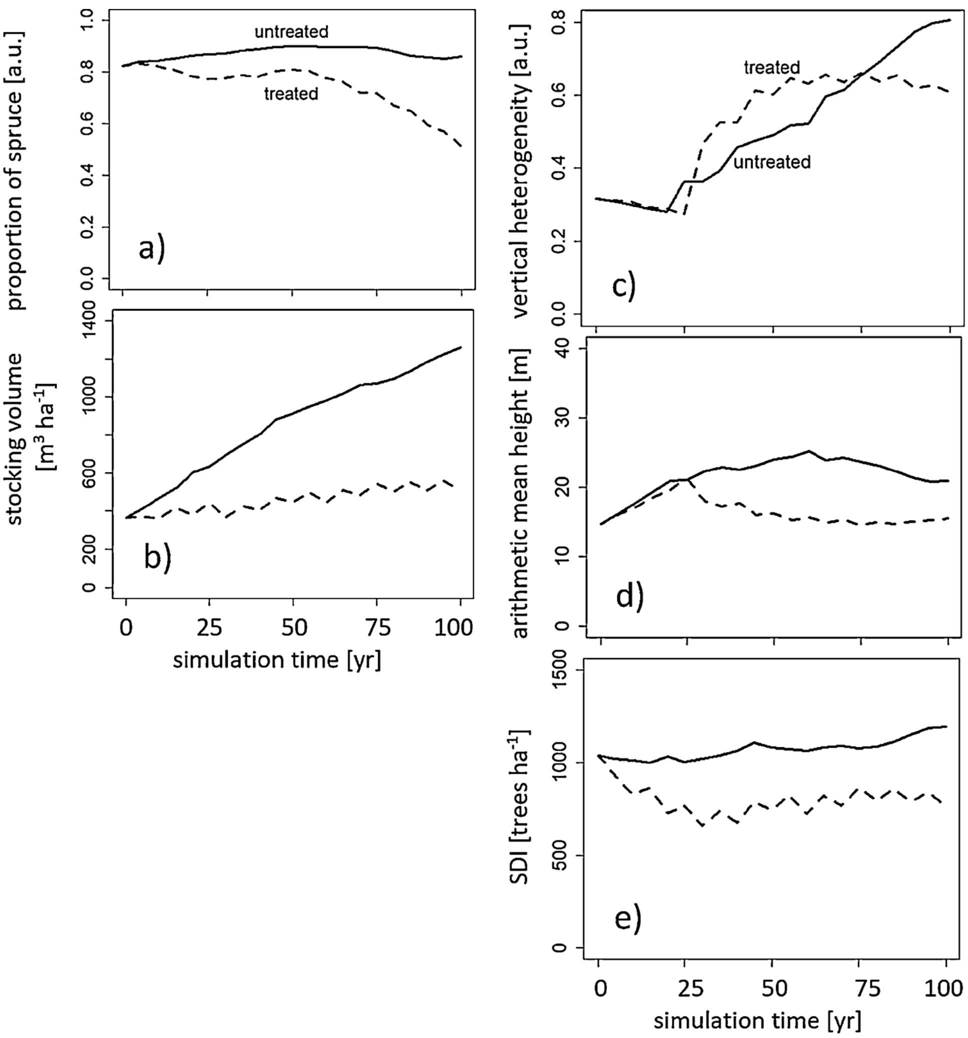

Fig. 6. Stand development within the exemplary simulation scenarios of ecosystem service provision including groundwater recharge (see Section 2.6). On the left (a, b), stocking volume and tree species proportion are governed by the management path within the European beech Norway spruce mixed species stand (dotted line: continuous cover forestry, CCF, solid line: no treatment). CCF is characterized by lower stocking volume and proportion of spruce. The diagrams on the right illustrate the structure development (c, d, e).

a model of higher abstraction level. Observation-based models are commonly parameterized with functions resulting from field trails. Modeling with a detailed representation, including a high resolution of the system, is quite suitable for investigating ecological relations based on a causal explanation of process interactions. Therefore, process-based models are suitable for deriving functions for observation-based models. According to Pandeya et al. (2016), water balance and locally required results for landscapes are very suitable for simulation.

But this higher level of abstraction leads to limitations in the validity of the results. This approach aims to estimate averages of magnitudes as well as strength and direction of changes with respect to forest management. This means more smooth results as probably prevalent in reality. The sensitivity of deflection is buffered to the average of changes in the direction.

4.2.1.1. Balance. BALANCE which was used for the percolation estimations is a process based growth model which simulates the three dimensional development of trees and forest stands and estimates consequences of environmental conditions. The carbon-, water- and nutrient flows of individual trees of different tree species form the fundamental processes for the growth simulations. A first version of the growth model BALANCE was published in 2002 (Grote and Pretzsch, 2002), the development of the model is still ongoing (Rötzer et al., 2013a,b; Rötzer et al., 2017). Extensive validations of the model BALANCE were published for growth processes and stand development (Grote and Pretzsch, 2002, Rötzer et al., 2005; Rötzer et al., 2010; Rötzer et al., 2013a,b), micro climate (Rötzer et al., 2010), phenology (Rötzer et al., 2004; Rötzer et al., 2005; Rötzer et al., 2010) and water balance (Rötzer et al., 2005; Rötzer et al., 2010; Rötzer et al., 2017). This way, the process based growth model BALANCE is able to realistically simulate growth and water balance (including percolation) for Central European forests.

4.2.1.2. Parameter selection. Neither parameters for environmental conditions nor the fundamental process parameters for driving groundwater recharge were used directly in the predictive models (groundwater functions). However, both are taken into account in the modeling approach. The used process model reflects the important processes and is sensitive to the environmental conditions that were explicitly set for the two case study areas. Therefore, we performed no sensitivity analysis in advance to find out which are the most important factors influencing groundwater recharge. Because, rather than identifying the most important predictors for groundwater recharge, the prediction should be limited to management-related parameters in order to reduce the predictive model to the impact of forest management. The predictive models are thus only valid in the case

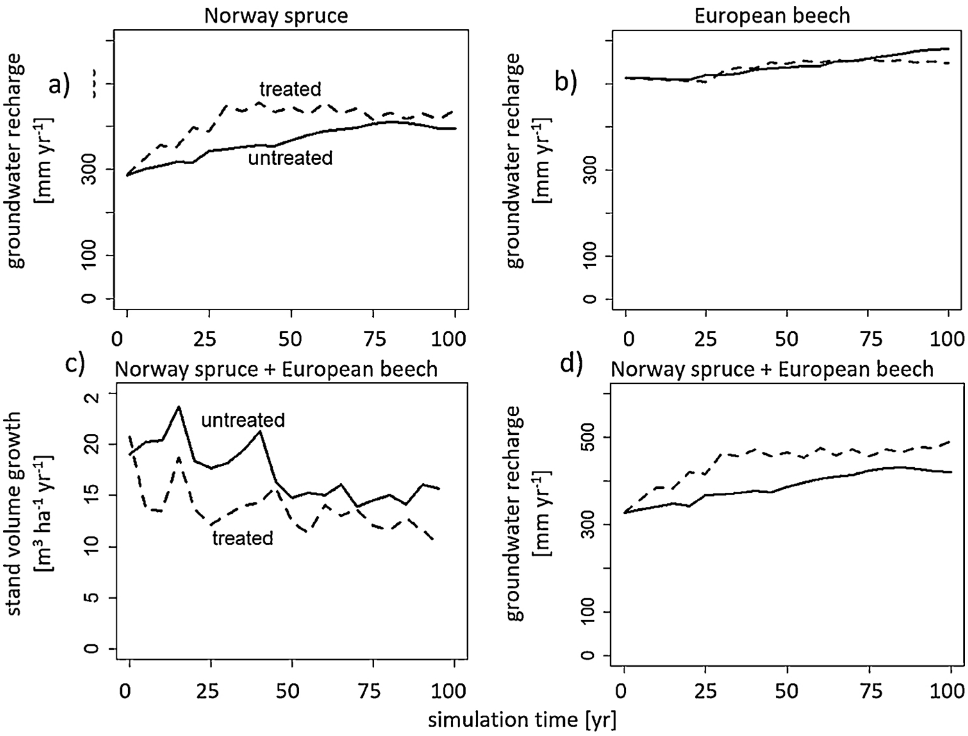

Fig. 7. Top graphics describe the species-specific development (a, b) of the unweighted groundwater recharge contribution within the exemplary test simulation of one mixed species forest stand. The bottom graphics demonstrate the development of volume growth and groundwater recharge in total (c, d).

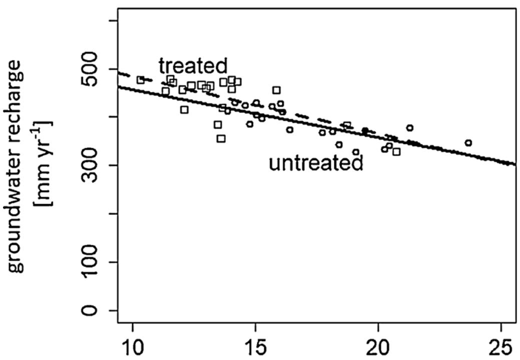

Fig. 8. Trade-off between groundwater recharge and stand productivity within the exemplary test simulation of a European beech Norway spruce mixed stand.

study areas with the set environmental conditions. As a consequence of not implementing the fundamental predictors for groundwater recharge directly in the prediction model this approach has the mentioned limitations and this can be regarded as weakness, but at the same time thereby some benefit arises. This approach abstracts quantification of groundwater recharge to the essential variables regarding the pure influence of forest management on landscape level.

For rough quantitative estimation of groundwater recharge in dependence on forest management on landscape level, the results show, that species and structure are convenient predictors, under assumed environmental conditions. The overriding target of this article was to develop a procedure resp., which is an algorithm to statistically simulate groundwater recharge for forest management purposes. Our proposed procedure, its prediction models and parameters seem to be the appropriate way. The process leads to recognizable differentiations and changes in the groundwater recharge simulation of the forest management model, and therefore the algorithm and its parameters fulfill its purpose.

A further finding in this study is that only a subset of the predictors considered had a pivotal influence on groundwater recharge. The consideration of structure variables as predictors makes the procedure more complicated and they had a small influence as compared to the influence of the consideration of the tree species. This can be deduced from the differences between the y-intercepts (Table 5) compared with the slopes. In this study and this CSAs this knowledge would have been possible to be applied to simplify both model calibration and application. For an even more straightforward estimation of groundwater recharge for the purposes regarded in this paper, the prediction could have been stripped-down to the tree species as the only parameter of the prediction model.

4.2.2. Relation between groundwater recharge and stand structure

Because climate and soil conditions were set in BALANCE, they are the basis where the derived relation between groundwater recharge and forest stand structure pertain. For improved comparability, the results of groundwater recharge can be considered as a share of precipitation. That generalization reveals that the results have a plausible magnitude. In this study, the groundwater recharge values account for approximately 30–50% of the occurring precipitation in the case study areas. Such values are conform with those from the literature (Anders et al., 2004; Rust, 2009; Müller, 2011; Sutmöller and Meesenburg, 2012; Müller, 2013). The magnitude of the values, however, is at the top of the range reported.

Our modeling approach aims to represent groundwater recharge within any mixed species stand considered. Therefore, our approach aggregates various independent estimations. Each estimation is related to the whole stand's structure but is specific to a monoculture of one of the stand's tree species. The estimations' fundamental data were obtained through simulation of the recharge-process within constructed monocultural stands, each of which represented a structure type from forest landscape analysis. To meet the overall structure of any mixed stand, these virtual stands, irrespective of their tree-species, had to represent the entire range of structures, including mixed species stands from inventory analysis. Thus, the virtual stands exhibited structures that were not necessarily typical of the tree-species monocultures therein. However, as a crucial property, they represent spatial relationships that may occur within mixed stands. Such relationships influence throughfall, interception, and transpiration through flux resistances and competition within the stand canopy (Jarvis and McNaughton, 1986). Our approach neglects the possible dependence of these intrinsic variables on the direct interaction among trees of different species. However, the approximation presented is fast and feasible for estimating long-term trends of groundwater recharge within assumed climate conditions and forest management scenarios on landscape scale level.

In the literature intermediate stand density is recommended for forest management aiming high groundwater recharge (Sutmöller and Meesenburg, 2012; Ilstedt et al., 2016). The reasons are that sparse stand density facilitates high discharge rates and ground cover vegetation. High stand density causes high interception and transpiration and thus reduces groundwater recharge. Groundwater recharge is not very sensitive to stand density within this study except the virtual spruce stands. At first, intermediate density mean another SDI and number of trees for each species. As already mentioned the study at hand investigated a limited range of structures that were typical for mixed species stands within the case study areas considered and thus not for each tree species separately. For example same SDI-value resulting from the mixed species stands in a CSA could imply typically high density for beech and almost low density for spruce in monoculture. Environmental and site specific conditions as well as the range of simulated SDI-values for each species similar could be reasons for the initially unexpected relation between stand density and groundwater recharge in this study.

Finally, it should not be assumed that the relations are necessarily typical of the tree species in general. Much more, the results are a snapshot, which results in a sensible calibration for the FMM as functional likeness of species mixtures on the entire case study area and environmental conditions.

5. Conclusions

The modeling approach presented leads to plausible results of groundwater recharge. It enables the estimation of the quantitative changes of the essential ecosystem service to contrasting options of forest management. Therefore, it is suitable to practice forest management planning with trade-off estimations among ecosystem services on the landscape level. Within an exemplary scenario, the approach underpinned a trade-off between groundwater recharge and productivity. This important relation was amplified through a change in tree species composition within CCF management. The influence of the model's predictors on groundwater recharge notably differs in strength. The dominating tree species has a higher influence than stand structure. However, both stand structure and tree species have a significant effect that is case study area-specific.

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Disclaimer

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Declarations of interest

None.

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Species Mixing Regulation with Respect to Forest Ecosystem Service Provision

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Abstract: The control and maintenance of species composition of mixed stands is a highly relevant objective of forest management in order to provide multifunctionality and climatic resilience. In contrast to this requirement there is, however, an evident lack of quantitative methods for mixture regulation. In this context, we propose an approach for the regulation of mixture proportions that has been implemented in a forest management model. The approach considers species-specific growth characteristics and takes into account the mixing effect on stand density. We present five exemplary simulations that apply the regulation. Each simulation maintains one of five desired species compositions. In these simulations, we consider the species European beech and Norway spruce under good site conditions, thus representing the most prominent mixed stands in Central Europe. Based on this model experiment, we analyze the potential benefit of controlled mixing regulation for achieving desired levels and combinations of ecosystem service provision, in particular productivity, diversity, and groundwater recharge. We found that a constant 50% basal area share of beech (equivalent growing space share of 80% to 70% depending on stand age) provided the most balanced supply of ecosystem services. Prominently, groundwater recharge considerably decreased when beech basal area shares were held below 50%. We discuss the ecological and practical implications of the regulation approach and different mixing shares.

Keywords: mixed forest stands; regulation of mixture proportions; ecosystem services; sustainable forest management; simulation of mixture proportions; forest management model

1. Introduction

In the course of recent decades, it has become a prominent goal of forest policy worldwide that future forests provide a broad spectrum of ecosystem services. Coincidentally, private forest stakeholder groups call for targeted usage of subsidies to pay forest owners for the provision of ecosystem services other than the traditional one of wood production [1].

Mixed species stands are widely accepted to provide a broad range of benefits. They can minimize the risk of calamities [2]. They can be more stable and more economical than monospecific stands [3]. Thus, in many cases, they are better suited for the multifunctional provision of ecosystem services than monocultures [4]. However, this depends on the identity of species and on the ecological context in which the mixed species stands are embedded [5].

Pretzsch and Forrester [6] underpin that the ecosystem service productivity of mixed stands depends on the shares of the contributing species. Mixture regulation, thus, is an obvious instrument for controlling the provision of ecosystem services. Within that scope, the tradeoff between groundwater recharge, tree species and structural diversity, and wood production is crucial. Because drought incidents will likely become more frequent in the future [7,8], groundwater recharge might then become a limiting process for public water supply. Diversity is a criterion of risk mitigation towards climate change as well as a forest management goal itself. Wood production provides renewable raw materials and preserves jobs. However, the specific effect of mixing proportions on these prominent ecosystem services has, to our knowledge, not been studied so far.

Due to the rapid paradigm shift towards mixed stands within Central Europe, there is a strong requirement for novel methods that control the development of mixed species forest stands [9]. Under unmanaged conditions, mixture obviously converges towards a natural state that depends on the species-specific site suitability [10]. According to Gadow and Füldner [11], the crucial requirements for achieving sustainability can exclusively be met if criteria of objective control are created and subjected to a more detailed description and quantification. Given the increasing relevance of mixed species stands, the step from qualitative descriptions towards species-specific quantitative regulations is overdue [12,13]. Coll et al. [14] reveal key questions about mixed forest management through a survey conducted among forest managers. They point out a knowledge gap concerning the quantitative regulation of mixed species stands.

Quantitative silvicultural guidelines are largely limited to even-aged, homogeneous monospecific stand types [15,16], except for very few existing approaches for density management, for example [17,18]. However, most existing concepts, if any, typically emphasize the qualitative steering of the species composition [9,19,20]. Management planning, in contrast, requires regulation, i.e., the maintenance of quantitative target values of mixing proportions [9]. Most current guidelines for the management of mixing proportions apply steering principles [21–24]. Abetz and Ohnemus [18] define the number of trees as dependent on the production target, time and risk. Thus, the resulting growing space per tree does not consider the species-specific growing requirements like a typical development of the crown projection area. Rather, it considers only the requirements of the silvicultural actor and assumes a dynamic adjustment of the species' growth requirements. A crucial aspect for mixing regulation is the ongoing availability of growing space per tree, considering species-specific typical stand and individual tree growth [16]. Typically, thinning actions, however, do not take into account varying conditions of individual tree growth at onward time scales [24]. Recent scientific advances increased the understanding of mixed species forests, but the extent to which this information is already suitable for consideration in practice is questionable [14]. Pretzsch and Zenner [9] stated that mixed species forest management guidelines should consider five aspects. When establishing mixed species stands, the temporal or spatial association of the species has to be designed: (i) a species combination with appropriate complementarity in mineral nutrients and water exploitation, as well as in space filling and light use, has to be chosen; (ii) according to mixing proportions; (iii) according to stand density; (iv) these have to be regulated during stand life; and (v) the final aspect in a quantitative management guideline for mixed species forests is the goal-oriented initiation of regeneration by volume reduction in the overstory.

Therefore, first and foremost, in the study at hand we present an approach for the quantitative regulation of mixture proportions. In order to evaluate our approach, we implemented it in the forest management model SILVA [25]. This enabled us to exemplarily simulate the development of mixed-species stands with different desired species shares. To this end, we chose the species Norway spruce (*Picea abies* (L.) H. Karst.) and European beech (*Fagus sylvatica* L.) on a fertile site which represented a very typical Central European setting so we could investigate the differential effect of the basal area shares on groundwater recharge, diversity, and wood production.

The study's key objectives were:

To propose a quantitative, growing space-based approach for regulation of mixture proportions in mixed stands;

To demonstrate the efficiency of the approach by means of scenario simulations for a highly prominent tree species mixture in Central Europe;

To assess the effect of mixture regulation on the provision of the wood production, diversity, and groundwater recharge ecosystem services.

2. Material and Methods

2.1. Approach for the Regulation of Mixture Proportion

Our approach aims to provide species-specific tree number guide curves in a mixed forest stand of two tree species. For both tree species, a mean diameter stem number guide curve is derived, which guarantees the desired basal area shares. The desired species shares are expressed as basal area shares for practical reasons. The biological key to mixing regulation, however, are the growing space shares. That is why both have to be connected. Thus, the basic idea is the consideration of the species-specific individual tree growing space requirement, including its change along mean diameter growth. This method determines the number of trees that are needed to produce a desired basal area composition. In the course of this, the stand area is optimally utilized by providing each individual tree, on average, with its biologically-based growing space requirement oriented by crown projection area.

The growing space share (α) and the species-specific number of trees for full stocking (*FS*) are both needed to calculate the species-specific number of trees (*N*) (Figure 1). We get α from the crown projection area (*CPA*) and the desired basal area share (β). We get *FS* from the *CPA* and the mixing effect on stand density (*CF*). The *CPA* is related to the species-specific mean diameter at breast height (*MDBH*) of the stand at each stand age.

CF is available from the literature and β can be arbitrarily chosen. Thus, at first, (i) we derived the relation between *MDBH* and *CPA*, then (ii) we showed how to calculate *FS* and (iii) α for finally (iv) getting *N*.

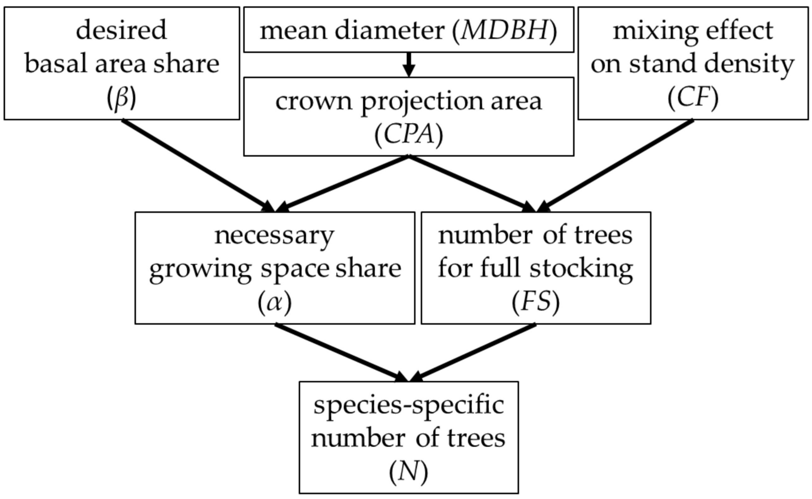

Figure 1. Conceptual diagram of the approach for the regulation of mixture proportion and of the computation of tree number guide curves. To obtain the number of trees (*N*), we filled the necessary growing space share (α) required for the desired basal area share (β) until full stocking. The number of trees per hectare for full stocking (*FS*) results from the species-specific crown projection area (*CPA*) and a correction factor (*CF*) for suitability in mixed species stands. The necessary growing space share results from the desired basal area share and the stand age dependent diameter (*MDBH*) related crown projection area.

2.1.1. Mean Diameter at Breast Height-Related Crown Projection Area (*CPA*)

In order to adapt the regulation approach to species-specific growth characteristics, we provide a species-typical value of required growing space to an average tree of the stand with *MDBH*. Therefore, we use the crown projection area. The following allometry describes the growth of the crown projection area (*CPA* (m^2)) as dependent on the mean diameter at breast height (*MDBH* (cm)).

$$CPA = c\, MDBH^{d} \quad (1)$$

where c and d are species-specific parameters. In order to obtain parameters for Equation (1), the equation was linearized and fitted to data from the network of long-term observation plots in Bavaria [26]. The database from the network of long-term observation contains 28,802 data sets with species-specific single tree information about crown projection area and diameter at breast height. Parameters were obtained that way for the European beech (*Fagus sylvatica* L.) and Norway spruce (*Picea abies* (L.) H. Karst, Scots pine (*Pinus sylvestris* L.), and sessile oak (*Quercus petraea* (Mattuschka) Liebl.).

2.1.2. Number of Trees for Full Stocking (*FS*)

Out of the *CPA* related to the *MDBH*, we can calculate the stem density at crown closure in a monoculture. However, recent research has shown that facilitation effects between tree species increase the maximum density within a mixed-species stand [27–29]. Correction factors for this density increase have been proposed by Pretzsch et al. [30] based on long-term observation data of the four main Central European tree species and each combination of them (Table 1).

Table 1. Correction factors for species combinations of four tree species (values resulting from evaluations in the context of [30], see Appendix A). European beech, Norway spruce, Scots pine and sessile oak.

Species Combination	Correction Factor (*CF*)
spruce/pine	1.44
spruce/beech	1.03
pine/beech	1.40
oak/beech	1.25

That effect is likely due to a more efficient sharing of the canopy space among species. It results in a stem density higher than the one at total crown closure with negligible crown overlap [31,32] (Equation (2)). This information allows us to estimate the stem number per ha at full stocking *FS* for each species in a mixed stand more realistically:

$$FS = \frac{10,000}{CPA} CF \tag{2}$$

with *CPA* being the crown projection area according to Equation (1).

2.1.3. Necessary Growing Space Share (α)

The necessary growing space share that corresponds to a desired basal area share results from the species-specific and stand age-dependent relation between crown projection area and basal area on tree and stand level. The tree's investment of growing-space-into basal area (*IT*) describes the average basal area a tree has on one unit of its growing space (Equation (3)):

$$IT = \frac{MBA}{CPA} \tag{3}$$

where *MBA* (m^2) is the mean basal area of a tree and *CPA* (m^2) is the corresponding crown projection area. The stand's investment of growing space into basal-area (*IS*) describes the basal area a tree species in a stand has on one unit of its growing space (Equation (4)):

$$IS = \frac{BAS}{A} \tag{4}$$

where *BAS* (m^2) is the basal area of one species on stand scale and A (m^2) is the sum of the crown projection areas of one species on stand scale. We assume that the relation between the growing-space investments (Equations (3) and (4)) of the two species (1 and 2) is independent of whether we consider just two individual average trees or whole stands (Equation (5)):

$$\frac{IT_1}{IT_2} = \frac{IS_1}{IS_2} \tag{5}$$

where each index value refers to exactly one species and *IT* and *IS* are defined as in Equations (3) and (4). Inserting Equation (3) for *IT* and Equation (4) for *IS* into Equation (5), we thus obtain:

$$\frac{A_2}{A_1} = \frac{BAS_2}{BAS_1}\,\frac{MBA_1}{CPA_1}\,\frac{CPA_2}{MBA_2} \tag{6}$$

The mean basal area *MBA* depends on the mean diameter at breast height (*MDBH*):

$$MBA = \frac{\pi}{4}\,MDBH^2 \tag{7}$$

Inserting Equation (1) for *CPA* and Equation 7 for *MBA* in Equation (6), we obtain Equation (8):

$$\frac{A_2}{A_1} = \frac{BAS_2}{BAS_1}\,\frac{c_2\ \ MDBH_2^{d_2\,-\,2}}{c_1\ \ MDBH_1^{d_1\,-\,2}}. \tag{8}$$

Equation (8) describes the ratio of the absolute growing space values of basal area and growing space. Equations (9) and (10) describe the relative values respectively, the percentages of basal area share and growing space share of species 1:

$$\alpha_1 = \frac{A_1}{A_1 + A_2} \tag{9}$$

$$\beta_1 = \frac{BAS_1}{BAS_1 + BAS_2} \tag{10}$$

Defining α_2 and β_2 accordingly, we may write Equation (8) using relative shares of growing space and basal area instead of absolute ones. Hence, we obtain the relative growing space share (α) of one species, as dependent on its relative basal area share (Equation (11)):

$$\alpha_1 = 1/\left(\left(\left(\frac{1}{\beta_1} - 1\right)\ \frac{c_2\,MDBH_2^{d_2\,-\,2}}{c_1\,MDBH_1^{d_1\,-\,2}}\right) + 1\right) \tag{11}$$

This equation depends on the mean diameter at breast height (*MDBH*) of both species considered, namely species 1 and 2.

2.1.4. Species-Specific Stem Number (*N*) Guide Curves

For obtaining the species-specific number of trees (*N*), we fill the necessary growing space share (α) required for the desired basal area share (β) until the number of trees for full stocking (*FS*) is prevalent:

$$N = \alpha \times FS \tag{12}$$

We replace the variables α and *FS*, using Equation (2) (*FS*), Equation (1) (*CPA*) and Equation (11) (α). Furthermore, we consider the mean diameter development as depending on time *t* (where the *MDBH* of both species is given). Thus, we obtain the stem number (per ha) guide curve equation, using the example of species 1 (Equation (13)):

$$N_{1(t)} = (10{,}000\ CF)\ /\ \left(\left(\left(\left(\frac{1}{\beta_1} - 1\right)\ \frac{c_2\,MDBH_{2(t)}^{d_2\,-\,2}}{c_1\,MDBH_{1(t)}^{d_1\,-\,2}}\right) + 1\right)c_1\ \ MDBH_{1(t)}^{d_1}\right) \tag{13}$$

For both species in a mixed stand, this equation allows us to calculate an age- or diameter-dependent stem number per ha that guarantees the desired basal area share while keeping the stand fully stocked.

2.2. Example Calibration of the Species Mixing Regulation

We applied the method for developing stem number guide curves for a mixed stand of Norway spruce and European beech. In order to reach a desired basal area share, we calibrated Equation (13) according to the fitted values for c and d from Section 2.1.1 (relation between *MDBH* and *CPA*).

Additionally, in order to calibrate guide curves for spruce and beech, a species-specific development of *MDBH* is required. For this purpose, we assume a stand with even-aged tree species. This approach could also be calibrated with any other diameter development of two tree species, such as for stands with delay in the age of one species. Additionally, two single values of *MDBH* could be used, for example, to get the number of trees needed in an existing stand for a wanted basal area mixture. Equation (14) describes the species-specific relation between stand age and *MDBH*:

$$MDBH_{(t)} = e^{v + p \ \log(t)} \tag{14}$$

where we consequently consider time t as species-specific stand age. For comparing the species-specific growth potential, we used sites of best yield class in Germany for both tree species, taken from the German National Forest Inventory (NFI) [33]. This dataset was used to fit the species-specific parameters v and p of Equation (14).

In order to implement the curves into the single tree-based forest simulation model SILVA [25,34], additionally we derived the species-specific relation of top height (i.e., average height of the 100 highest trees per hectare) to stand age:

$$h_{100(t)} = w\left(1 - e^{-u\ t}\right)^k \tag{15}$$

Therefore, we fitted the species-specific parameters w, u and k according to the best yield class from the German NFI.

2.3. Simulation Study with Exemplary Calibrated Mixing Regulation

2.3.1. Intention of the Simulation Study

In order to demonstrate an application of the mixing regulation approach in the context of multifunctional forestry, we conducted a simulation study about the provision of selected ecosystem services in relation to different species mixture proportions. This simulation study expands on the exemplary guide curve calibration (Section 2.2) for the Norway spruce/European beech mixture. Five simulation runs were executed. Therefore, the applied desired basal area shares (β) of both species were 0%, 20%, 50%, 80% and 100%, which considered both species' monospecific stands as extreme combinations. Each simulation run covers a time span of 100 years. The simulation outcomes were evaluated with respect to the ecosystem services of (i) wood production (using stand volume growth as indicator variable); (ii) diversity (with the species profile index [34] as proxy variable); and (iii) groundwater recharge (quantified with a new approach by Schwaiger et al. [35]).

2.3.2. Forest Management Model Settings

For the simulation study, we used the single tree-based forest management model SILVA [25,34]. The site conditions of the model were set to the *MDBH* and h_{100} development of beech and spruce assumed and calculated in Section 2.2. All simulation runs used the thinning kind of selective thinning and therefore the stand density was regulated according to the guide curves from the regulation approach of the study at hand and calibrated in Section 2.2.

2.3.3. Calculation of the Ecosystem Services: Diversity, Productivity and Groundwater Recharge

The species profile index (K) [36] is a combined measure of a stand's richness in both species and vertical structure. Basically, it is an extension of the concept of the Shannon Species Diversity Index [37]. In a single layered monoculture, its value is 0, while its maximum value for a two-species mixture is $\ln(6) \approx 1.79$, which would indicate both species being equally represented in and among the stand's upper, middle, and understory.

Stand volume growth was used as a proxy for the ecosystem service productivity. The stand volume was calculated by summing up the single tree volumes. Volume increment was defined as the difference between the stand volume in a simulation step and the volume in the timestep before.

The ecosystem service groundwater recharge (GWR) was calculated according to a novel approach of Schwaiger et al. [35] already implemented in SILVA. Based on extensive simulation studies with an ecophysiological forest simulation model, Schwaiger et al. [35] propose a linear function for estimating GWR (in mm/year) based on indicators of stand structure (Equation (16)).

$$GWR = a + r\,MH + o \cdot VH + m \cdot SDI \tag{16}$$

where GWR, within a first step, is the estimated groundwater recharge of a monoculture, MH is the stand's arithmetic mean height, VH is the variation coefficient of tree height and SDI [38] is the stand density index. The variables a, r, o and m are species- and site-specific parameters. The approach of Schwaiger et al. [35] applies to the site of best yield class Augsburg Western Forests in Bavaria. The site conditions set for the simulation study of this article at hand are assumed to represent one average site of best yield class in Germany. Thus, the site assumptions of both studies are comparable and therefore the groundwater approach of Schwaiger et al. [35] can be used for this simulation study. Schwaiger et al. [35] suggest the parameter values $a = 418.72$, $r = 5.61$, $o = 234.46$, $m = -0.28$ for Norway spruce, and $a = 463.71$, $r = 0$, $o = 154.61$, and $m = 0$ for European beech.

Each of these two species-specific contributions, in a further step, was weighted by the mixing proportion as growing space share GSS in order to estimate the whole stand's groundwater recharge GWR_{total} (Equation (17)):

$$GWR_{total} = GSS_{spruce}\,GWR_{spruce} + GSS_{beech}\,GWR_{beech} \tag{17}$$

3. Results

3.1. Exemplary Guide Curve Calibration

3.1.1. Assumed Diameter and Top Height over Stand Age

Assuming or knowing the relation of $MDBH$ and h_{100} to stand age is a basis for calibration of the presented regulation approach. We exemplarily assume values for beech and spruce (calculation in Section 2.2) to calibrate the approach for the simulation study.

Growth of top height and diameter at breast height is higher in the case of spruce compared to beech (Figure 2). At age 100, the prevalent top height for beech is 35 m and 38 m for spruce. Moreover, in a 100-year-old stand, a mean beech diameter at breast height of 30 cm and a mean spruce diameter at breast height of 37 cm can be assumed.

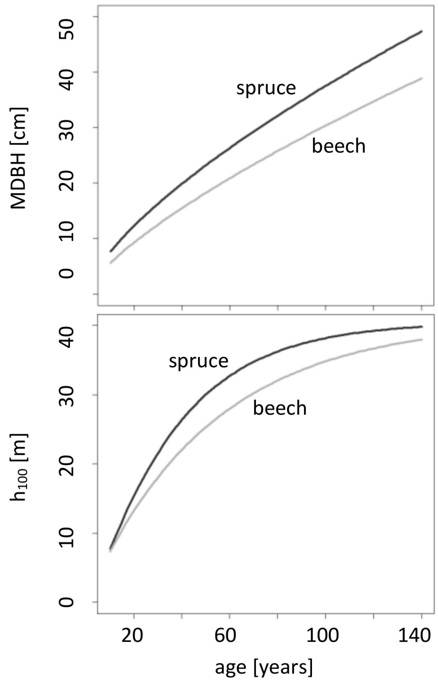

Figure 2. Assumed development of top height and mean diameter at breast height (*MDBH*) for the parametrization of the regulation approach within the exemplary simulation; parameters in Tables 2 and 3 (more detailed database information, see Appendices B.2 and B.3).

Table 2. Estimates of the *MDBH* functions for European beech, Norway spruce (Equation (14); Section 2.2; more detailed database information, see Appendix B.2).

Tree Species	v		p		n
	Estimate	*Std. Error*	*Estimate*	*Std. Error*	
Beech	0.04223	0.07	0.73227	0.02	54,512
Spruce	0.4462	0.04	0.6904	0.01	122,743

The fitted values (Tables 2 and 3) describe the species-specific curves of *MDBH* and h_{100} in Figure 2 and thus reveal the exact difference between the assumed growth mean diameter at breast height and top height of spruce and beech. To sum up, we can say that the assumed growth potential of spruce regarding h_{100} and *MDBH* is higher compared to beech.

Table 3. Estimates of the h_{100}-functions for European beech, Norway spruce (Equation (15); Section 2.2; more detailed database information, see Appendix B.3).

Tree Species	w		u		k		n
	Estimate	*Std. Error*	*Estimate*	*Std. Error*	*Estimate*	*Std. Error*	
Beech	40.72	0.46	0.01916	0.00	0.9808	0.03	54,512
Spruce	40.45	0.20	0.03106	0.00	1.2549	0.03	122,743

3.1.2. Diameter Related Crown Projection Area

The diameter related crown projection area (Figure 3) is a basis for the presented regulation approach. Those values are suitable for the parametrization of the mixing regulation concerning the four most important tree species in Germany. Thus, they generalize our approach beside the exemplary simulation.

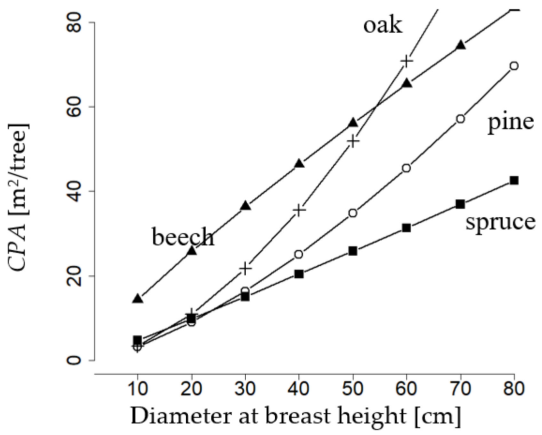

Figure 3. Crown projection area (*CPA*) over diameter at breast height for sessile oak (cross), European beech (triangle), Scots pine (circle), Norway spruce (square), as expected with the fitted model after Equation (1), parameters in Table 4 (more detailed database information, see Appendix B.1).

Norway spruce, European beech, sessile oak, and Scots pine, notably differ in their parameters (Table 4) obtained through fitting of the crown projection area function (Section 2.1.1, Equation (1)). That of European beech starts with a large value of about 15 m^2 at diameter at breast height of 10 cm, while the one of the remainder species is at only 5 m^2. The crown projection area of European beech, starting from low values of diameter at breast height and throughout the whole diameter at breast height range, is markedly larger than that of Scots pine and Norway spruce. Up to a diameter at breast height of 50 cm, it also surpasses that of sessile oak. However, the slope of the crown projection area over diameter at breast height of beech decreases with diameter at breast height. Conversely, that of oak strongly increases. Thus, at a diameter at breast height of more than 50 cm, oak outruns the crown projection area of all other species. Pine, which like oak, is a light-demanding species, has a similar course of crown projection area over diameter at breast height as oak and approximates the values of beech at a diameter at breast height of 80 cm. Spruce has the lowest crown projection area over the whole range of diameter at breast height and one that constantly increases with diameter at breast height. To sum up, we can say that the species-specific relations between diameter and crown projection area are very different, even intersections are visible. Consequently, this relationship is of fundamental importance for the mixture regulation approach of this study.

Table 4. Estimates of the crown projection area functions for European beech, Norway spruce, Scots pine and sessile oak (Equation (1); Section 2.1.1; more detailed database information, see Appendix B.1).

Tree Species	***ln(c)***		***d***		
	Estimate	***Std. Error***	***Estimate***	***Std. Error***	***n***
Beech	0.712	0.03	0.85	0.01	10,348
Spruce	−0.8921	0.03	1.06	0.01	9997
Pine	−2.21	0.05	1.48	0.02	4520
Oak	−2.66	0.05	1.70	0.01	3937

3.1.3. Exemplary Guide Curve Calculation

The difference between the number of trees for full stocking in monoculture and mixture is one essential intermediate result at the calculation of the guide curve. The exemplary calibrated curves we calculated for a mixture of spruce and beech consider the following difference. Full stocking at crown closure without overlap in monoculture (Section 2.1.2, Equation (2)) has just slightly lower values as the one under assumption of overlapping crowns resp. with mixture adjustment (Figure 4). This is true for both tree species we focus on. However, beech stands have a stem density that is about 50%

of that of spruce stands, a direct consequence of the *MDBH-CPA* relations shown in Figure 3. That proportion continues over the whole stand age.

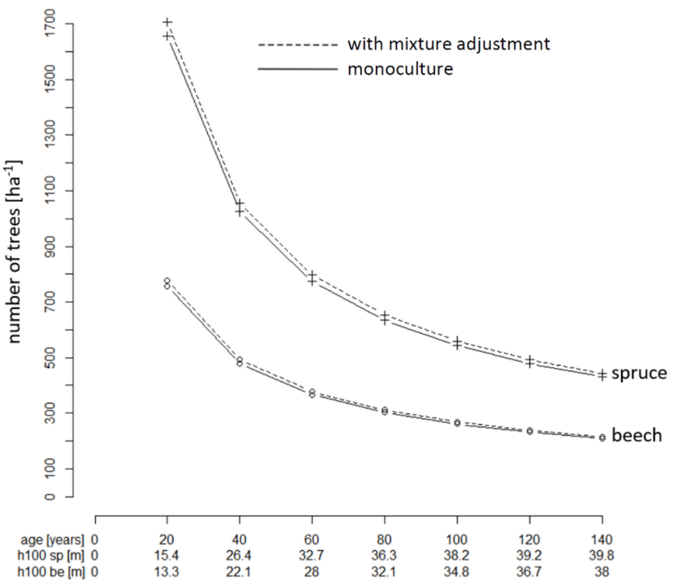

Figure 4. Number of trees at full stocking per ha (*FS*) over stand age (*t*) based on Equation (2), given for Norway spruce (*sp*) and European beech (*be*); given in addition to over top height h_{100}. Curves shown with a solid line refer to monospecific stands and therefore assume a mixture adjustment *CF* of 1.0 in Equation (2) (Section 2.1.2); curves shown with a dotted line assume that crowns overlap according to a mixture adjustment *CF* of 1.03 in Equation (2) (Section 2.1.2, Table 1). The mean diameter at breast height in Equation (2) was taken according to the stand age (Figure 2); crown projection parameters (*c*, *d*) from Table 4.

The growing space share of beech mixed with spruce is larger than its basal area share at any mixture proportion of both species being considered (Figure 5). For example, 50% of basal area share requires a beech growing space share of 80% in a 20-year-old stand. This ratio depends on the age of the stand. In an approximately 120 year-old stand, a 50% basal area mixture requires only 70% of the growing space.

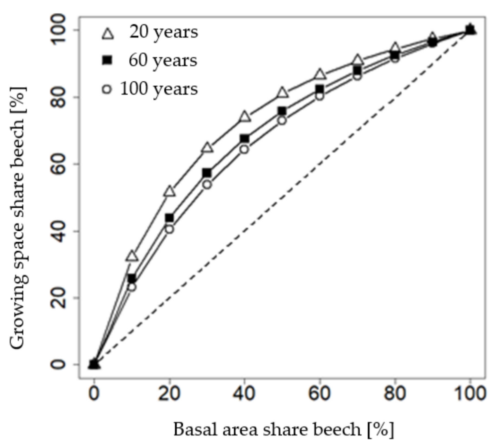

Figure 5. Conversion between growing space share (α) and basal area share (β) in a mixed stand of European beech (*be*) and Norway spruce (*sp*). Each line presents one stand age (*t*): triangle = 20 years, square = 60 years, circle = 100 years. α was calculated as α_1 from Equation (11) and as dependent on the basal area share given as β_1 in Equation 11. The diameter at breast height in Equation (2) was taken according to the stand age (Figure 2); parameters c_1 and c_2 in Equation (11) were taken from Table 4.

The resulting tree number guide curve (Equation (13)) for European beech and a 50% basal area mixture are marked by a strong decrease of the tree number per ha from age 20 to 140 (Figure 6), which is the typical behavior of stem number curves in even-aged stands. While the beech growing space decreases overtime, the number of trees accordingly drops from about 630 to about 160 (beech) and 320 to about 150 (spruce), respectively. Thus, for equal basal area shares, twice as many beech trees as spruces are required in the beginning, while in an old stand with equal basal area shares, the tree numbers are about balanced. This results from the shifting relation between basal area and growing space shares, as shown above.

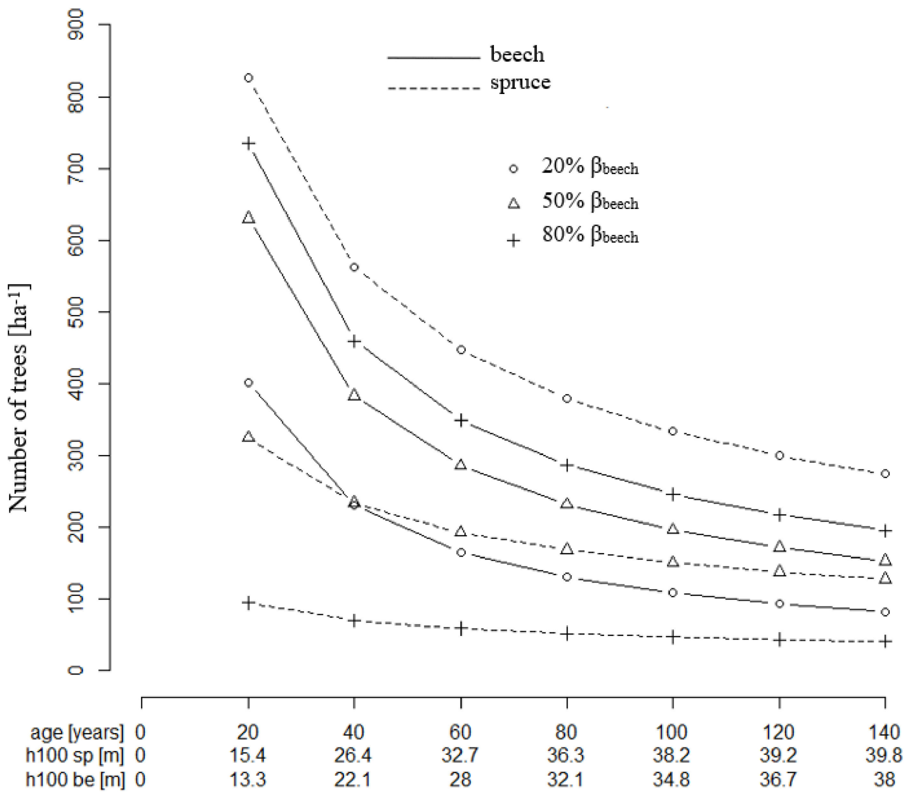

Figure 6. Guide curves of tree number per ha over age calculated with Equation (13) for Norway spruce (*sp*) and European beech (*be*) and each of three different basal area shares ($\beta_{species}$ = 20%, 50% and 80%); The diameter at breast height was taken according to the stand age (Figure 2); parameters c_1 and c_2 in Equation (13) were taken from Table 3.

3.2. Simulation Study Quantifying Ecosystem Services Provision Depending on Species Shares

The results of our simulation runs show that the indicators for the ecosystem services we focus on, namely water availability (groundwater recharge), diversity (species profile index), and productivity (stand volume growth), notably depended on the mixture proportions (Figure 7). The latter were defined as the desired basal area shares of beech and spruce, controlled in the simulations using the method developed above. Over the whole simulation period, the simulated beech basal area shares (Figure 7, first row, dotted lines) had an average of 23%, 52%, and 82%, when the tree number guide curves had been adapted to a basal area share of 20%, 50% respectively 80% (Table 5) which we deem reasonably close to the desired values. The resulting simulated growing space shares and basal area shares can be compared with the default desired basal area shares in Table 5.

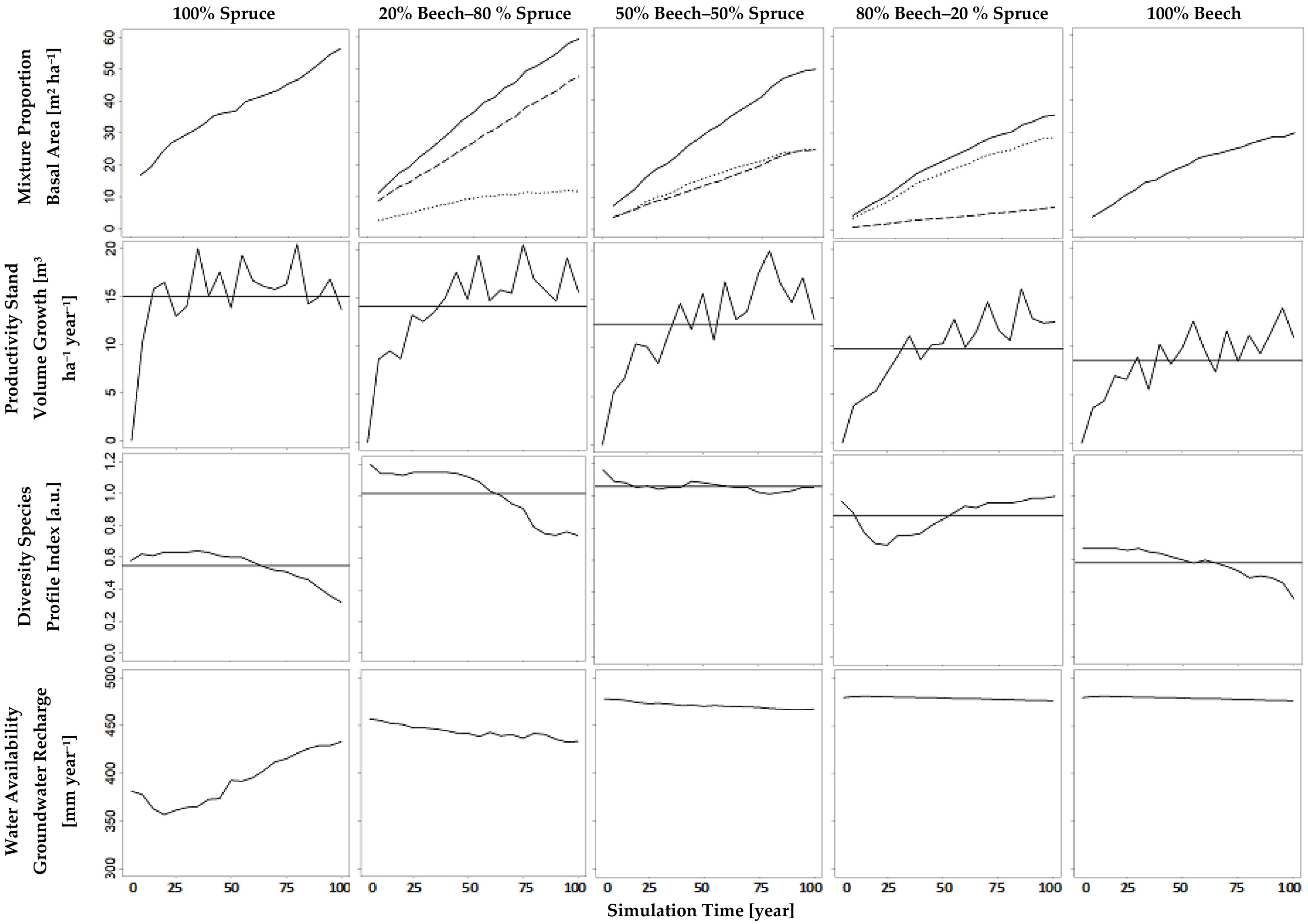

Figure 7. Influence of basal area composition on water availability, diversity and productivity based on simulation runs within a mixed stand of European beech and Norway spruce (Section 2.3); each column refers to exactly one run; that run presumed a relation of beech vs. spruce basal area as given by the column header; dotted line—beech, dashed line—spruce, horizontal line—average, solid line—total stand.

An increasing basal area share of beech reduced the stand's total basal area (Figure 7, first row). Accordingly, it caused a reduction of stand volume growth, i.e., productivity. Conversely, an increasing basal area share of that deciduous species markedly fostered groundwater recharge, i.e., water availability; but a beech share higher than 50% did not increase groundwater recharge anymore (Figure 7, bottom row). The results show that groundwater recharge, within that study, is not very sensitive above a beech basal area share over 50%. This is due to the fact that even small basal area shares of beech result in large growing space shares (Table 5). At a basal area share beech of 50%, the indicator of diversity, species profile index was highest.

Table 5. Average values over simulation time of 100 years with default desired basal area shares.

Desired Beech Basal Area Share (%)	Average of Simulated Beech Basal Area Share (%)	Average of Simulated Beech Growing Space Share (%)
0	0	0
20	23	52
50	52	79
80	82	94
100	100	100

The curves in Figure 7 illustrate a trade-off between productivity and water availability with changing mixture proportions. Water availability may be aggravated in the future through the choice of a tree species that aims at increasing forest productivity; 100% spruce maximizes productivity and minimizes groundwater recharge; 100% beech maximizes groundwater recharge and minimizes productivity. The optimized provision of both ecosystem services can be reached with a beech share between 20% and 50% in a mixed stand with spruce.

Hence, the results reveal an advantage of monocultures regarding the maximization of single ecosystem services. However, they also point out disadvantages of a monocultural stand. In the simulation study of this paper, even small shares of a secondary tree species considerably increase two ecosystem services, whereas coincidentally only one ecosystem service slightly decreases. Small basal area shares of spruce in beech stands increase productivity and diversity. That increase is being paid for by only a minute decrease of groundwater recharge. In turn, small basal area shares of beech in spruce stands increase groundwater recharge and diversity. This is being paid for by an only slightly decreasing productivity.

Comparing diversity with productivity, we see a trade-off in spruce-dominated stands (100% to 50% spruce) and a synergy in beech-dominated (50% to 100% beech) stands. Furthermore, a change from a synergy to a trade-off is visible comparing diversity with groundwater recharge. In spruce-dominated stands, there is a synergy between diversity and groundwater recharge and in beech-dominated stands, there is a trade-off.

In a real decision making situation, such results based on controlled mixture proportions could be presented to stakeholders. Clearly, it would depend on the stakeholders' value judgements which mixture proportions they prefer. Productivity-oriented stakeholders like large private forest owners e.g., would typically prefer the spruce monoculture or a 20% beech share at most, because the latter still maintains a high level of wood increment while already profiting from the benefits coming along with having a small share of beech. Multifunctional- and diversity-oriented stakeholders like state forest managers would prefer the 50% mixture because the provision of all ecosystem services considered is the most balanced one. A stakeholder who is responsible for guaranteeing water supply, e.g., a forest owning municipality with own wells, would possibly also prefer the composition with 50% beech shares due to a sparsely decreasing groundwater recharge with higher shares of beech. Coincidentally, the other ecosystem services would decrease with higher beech shares than 50% basal area.

4. Discussion

4.1. The Approach Contributes to Develop Quantitative Guidelines for Mixed Species Forests

According to Pretzsch and Zenner [9], the proposed regulation approach of this study contributes to the development of quantitative guidelines for mixed species forests. Therefore, the approach helps to bridge the gap from science to practice. The approach of this article considers two of five aspects required for quantitative guidelines (temporal or spatial association of the species has to be designed (i); species combination with appropriate complementarity in mineral nutrients and water exploitation, space filling and light use has to be chosen (ii); mixing proportions (iii) and stand density (iv) have to be regulated during stand life; goal-oriented initiation of regeneration by volume reduction in the overstory (v)). The tree number guide curves are suitable for regulation of mixture proportion (aspect iii) and stand density (aspect iv) in thinning interventions. The exemplary simulations show that the method was effective in achieving the desired mixture proportions throughout the whole simulation time of 100 years. The tree number guide curves constructed with our method are based on the stand age and species-specific growing space requirements, which explicitly include species-mixing effects.

4.2. Mixing Proportions Are Crucial for Managing the Ecosystem Services Provision

By steering species mixture proportions, forest management influences the provision of ecosystem services. Multifunctional forestry is highly relevant, especially in European state forests. Nevertheless, forest stakeholders often focus on a small set or only one ecosystem service like wood production without the effects on other services being considered [39–41]. Trade-offs between ecosystem services are often caused by different tree species that provide different ecosystem services [4,42]. There are results that reveal that tree diversity influences the delivery of ecosystem services [41,43]. However, from the perspective of operational forest management, there is still uncertainty about the extent and quantity. Hence, it is important to better evaluate the effect of tree species mixing, i.e., tree species diversity, on ecosystem service provision [5,44]. Two recently published review papers call for more research concerning this topic to enable substantiated consulting for forest management and policy to broaden the consideration of ecosystem service provision in practice [44,45]. Combined with proxies for ecosystem services, the approach results in added value for practice, as the present simulation study has shown.

As an attempt towards attaining unambiguous species share regulation in practice and regarding modelling tools, we propose our approach for regulating the basal area share per species within mixed species stands. The results of applying it in the framework of an exemplary simulation study underpin that the approach presented is suitable for identifying trade-offs between various ecosystem services at a mixing proportion being considered. We see an important advantage in being able to directly and closely control species shares for such purposes. Investigations by Pretzsch et al. [30] pertaining to the mixing effects on forest stand productivity consider tree species shares. Further studies that investigate other ecosystem services as related to mixing ratios are rare.

From a stakeholder's or decision maker's point of view, such an approach allows to choose species compositions more rationally depending on the envisaged ecosystem services. Forest management influences ecosystem service trade-offs and thus influences the multifunctionality of forests [46]. We obtain the tradeoffs between ecosystem services by varying species shares in our plausible exemplary simulation studies and mostly as expected in a qualitative sense. In a quantitative sense, however, they allow us to check where exactly a desired optimum, which is a considerable support for planning, can be found. When a certain range of mixture proportions turns out to be of special interest, finer subdivisions of species shares can be applied and investigated for their effects on ecosystem service provision.

4.3. Important Considerations within the Regulation Approach

The suggested calculation of the tree number guide curves in this study is based on site- and species-specific development of height, diameter, and crown projection area. Generally, known ecological strategies of the tree species are recognizable in the crown-diameter allometry results, as shown in Figure 3. Vieilledent et al. [47] found higher *CPA* for spruce compared to the results of this study. But additionally, it is known that crown-allometry is quite variable. Thus, we think that an approximation to species-specific mean values is conceivable and should be tried. By allowing the implementation of such allometries straightforwardly from whatever species, our approach directly takes into account the consequences for silvicultural mixture regulation arising from the contributing species' ecology.

The age-dependent growing space proportion that a species requires for the desired basal area share can be calculated by the aforementioned allometric developments. Pretzsch and Schütze [48] stated that beech uses its growing space not as efficiently as spruce for biomass production. On the one hand, at the same diameter at breast height, a beech tree occupies more growing space than a spruce. On the other hand, beech stems are thinner than spruce stems at the same age. This combination causes spruce to require just 20% growing space share in a spruce-beech mixed forest in order to have a basal area share of 50%. This illustrates how little growing space has to be available for spruce to be not only the secondary but co-dominating tree species.

In a 50% basal area mixture, the stem number ratio between the species approximates from two beeches to one spruce to almost one beech to one spruce. The tree number ratio in a stand with 50% growing space mixture is almost two spruces to one beech all the time. Those relationships are the most important ones to enable the quantitative regulation of the desired mixture proportions.

4.4. Weaknesses, Limitations, Further Development

Mori et al. [45] stated that it is not trivial to bridge gaps between science and practice. However, forest research is expected to support managing, conserving, and restoring mixed species forest ecosystems [45]. We conclude that the approach presented in this article represents such a bridge between practice and research. Because it allows a clear, unambiguous control of the mixture proportions and takes into account new scientific findings such as the mixing effect on stand density. However, there are also some weaknesses and potential to include more recent scientific advances and knowledge about mixed species forests. The usage of recent scientific advance in this article is expandable concerning three issues: implementation of knowledge about mixed species forests (i) and uneven-aged forests (ii) into the regulation approach, and ecosystem service assessment (iii).

The study at hand introduces an approach for implementing scientific knowledge about mixed species stands within an algorithm for quantitative control of species proportions at optimal utilization of growing space. Within that scope it exemplifies a set of variables that is pivotal and, thereby, sensitizes for the level of detail that deserves consideration in practice. The approach includes mixing effects on stand density as a result of recent research that strongly contributes to quantitative mixed stand management. However, it is open for refinements that likely will result from ongoing work on the effects of mixture on stand development. In particular, there are clues that the aforementioned mixing effects are not constant but dependent on site quality (*SQ*), mixing proportions (β), and stand age (t) [49]. The improvement of highest priority will thus be an extended definition of the correction factors (*CF*) through functions that use the three influencing variables as predictors. A well-reasoned candidate for implementing that relation is a multiple non linear model CF (*SQ*, β, t). Furthermore, research might refine the estimates of species specific allometric coefficients. Crown allometrics are different in mixed species stands than in monocultures, as investigations with modern technology like terrestrial laser scanning have revealed [31]. Therefore, additional correction factors could be implemented for adjustment of the correlation between mean diameter at breast height (*MDBH*) and crown projection area (*CPA*). However, the authors point out that management approaches should not become overly complex in practice. When transferring knowledge into practice, the principle of "as complex as necessary and as simple as possible" should be followed. This means the sensitivity of the output of the approach regarding additional detail and knowledge from further research has to be in an order of magnitude which is relevant for decision making in practice.

In addition to knowledge about mixed species stands, knowledge about structure should also be implemented. It was beyond the scope of this study to take into account uneven-aged stands but we see the potential to calibrate the approach for any temporal or spatial association of the species according to the claims of Pretzsch and Zenner [9] (Section 4.1 (i)). So, further development of the approach could concentrate on the application and modification not only for *MDBH* over the same stand age for both species. Additionally, diameter distributions instead of one single *MDBH* would be an opportunity for further improvement with respect to enlargement.

Since mixture regulation itself is at the forefront of this article, the assessments of the ecosystem services are rather subordinate. Therefore, the effort to assess the ecosystem services was kept practicable. Existing variables should be used. Estimation of the productivity of a system was assessed by the use of volume increment. The species profile index results from the tree number per height class and species. Based on the concept of entropy that is a well-established proxy of ecological diversity, that index identifies the highest level if all tree species are distributed equally across the three vertical stand strata. It may indicate diversity in scenarios which produce identical basal-area proportions per species on stand level. However, that effect is not mechanically determined, because on the one hand the species profile index considers vertical structure, and on the other hand our approach regulates basal area shares which is not the same as the tree number proportions that the index is based on. We are aware that the species profile index is only one of many possible indicators for biodiversity, but as it covers two important aspects of diversity (species and structural richness) in one number, we deemed it especially applicable for the study at hand. However, a practical extension of that method will certainly comprise a complementary set of biodiversity indicators, such as a deadwood metric or a metric based on very large trees [50]. The predictors of groundwater recharge are overall structural attributes of the mixed stand being considered. That way, the method takes into account the collective spatial stand structure that results from occupation of the stand's canopy per species. Thus, it presumes a horizontally homogeneous crown distribution pattern among all tree species per canopy layer, as a differentiation by attributes of spatial clustering is likely impracticable. In order to account for species-specifity of the relation between structure and groundwater recharge, groundwater recharge prediction applies the structural attributes to statistical models (Equation (16)) that had been fitted on a per-species basis in pure stands, and weights the outcome by species proportion. These models were derived from extensive systematic simulation runs with an ecophyisological forest growth model. However, it should be noted anyway that this approach [35] is a method that is designed for enabling a forest management simulation model roughly to estimate groundwater recharge which was—to the best of our knowledge—not possible before.

5. Conclusions

The novel thinning approach of this study enables a quantitative and precise regulation of mixture proportions in real and simulated mixed species stands. The study underpins the benefit of maintaining a desired mixture proportion for a controlled provision of ecosystem services. The proposed approach is not limited to the species chosen as an example in this study. It is efficient for conducting simulation studies for forest practice, especially in the context of the very modern question of multiple ecosystem service provision. An exemplary simulation study suggests that monocultures are ideal for the provision of one major ecosystem service. However, even small basal area proportions of admixed species may notably increase the provision of other services, while just sparsely decreasing the provision of the stand's major service.

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Appendix A. Stand Density Correction Factors

The stand density correction factors shown in Table 1 result from evaluations made in the context of the publication [30] written in German.

Table A1 below shows the full set of values calculated by the authors of [30] who kindly permitted us to publish this table and corresponding results, which we briefly explain.

Table A1. Number of trees per ha (*N*) in mixed-species stands in relation to the neighboring monocultures calculated separately for five selected species assemblages (as resulting from evaluations in the context of [30] and kindly provided by the authors). Ratios mixed/mono above/below 1.00 indicate a superiority/inferiority of the species' performance in mixed-species stands versus monocultures. Ratios in bold numbers indicate significant differences ($p < 0.05$) between mixed-species stands and monocultures.

Variable	Species Combination	*n*	Species 1 Mixed/Mono (±SE)	Species 2 Mixed/Mono (±SE)	Total Stand Mixed/Mono (±SE)
Number of trees *N* (trees ha^{-1})					
	spruce/pine	7	1.78 (± 0.38)	1.06 (± 0.12)	1.44 (± 0.25)
	spruce/larch	10	2.72 (± 1.62)	1.07 (± 0.20)	1.57 (± 0.54)
	spruce/beech	52	0.90 (± 0.05)	1.20 (± 0.06)	1.03 (± 0.06)
	pine/beech	17	1.22 (± 0.10)	1.59 (± 0.13)	1.40 (± 0.09)
	oak/beech	24	1.23 (± 0.08)	1.27 (± 0.13)	1.25 (± 0.10)

The investigation focused on single-layered mixed-species stands consisting of two tree species. It considered combinations of datasets from mixed and monospecific stands of the corresponding species at equal site conditions. Therefore, that previous work included a total of nine species combinations. The study exclusively included stands representing maximum densities which were not or only weakly treated. The datasets come from an extensive network of long-term observation plots [26] complemented with temporary investigation plots. In total, the data comprise 141 neighboring combinations of mixed and monospecific stands. The included combinations of monospecific and mixed stands are predominantly situated in Germany, but also represent other regions in Central Europe.

For comparing the number of trees per ha regarding both species in total (Table A1), the measured values of the mixed stands were set in relation to the values from the adjacent monospecific stands weighted to the mixing proportions prevalent in the mixed stand being compared. Therefore, the resulting weighted value formed the reference for comparison with the corresponding observed value of the adjacent mixed stand. For that comparison on the level of the tree species (Species 1 and Species 2 in Table A1), the contribution per tree species to the mixed stand was scaled up to one hectare with the mixing proportion and then compared with the corresponding neighboring monospecific stand.

To compare the mean values, the corresponding values of the mixed stand were divided by those of the monospecific stand of the same species [30]. The mean quotient over all combinations then serves to check whether the values in the mixed stands are greater than those in the monospecific stands. If 1.0 was beyond the confidence intervals of the average quotients, the differences between mixed and monospecific stands were significant at the level $p \leq 0.05$. If the confidence interval is above 1.0, the mixed stand is superior to the monospecific stand(s); if the confidence interval is below 1.0, the mixed stand is inferior to the monospecific stand.

Appendix B

Appendix B.1. Diameter-Related Crown Projection Area

The relation between mean stand diameter and the corresponding crown projection area as expressed in Equation (1) is crucial for the mixing regulation approach proposed in this study. We present the parameter values and their standard errors in Table 4. The data we used for estimating these parameter values come from the network of long-term observation plots in Bavaria [26], which comprises about 300 trials where the oldest date back to the 1870ies.

Equation (1) may be straightforwardly linearized by taking the equation's logarithm to estimate the parameters c ($\ln(c)$) and d from intercept and slope. We calibrated Equation (1) based on individual tree data. The parameters of that crown projection area function were obtained by fitting the linearized Equation (1) with ordinary least squares (*OLS*) regression separately for the tree species European beech, Norway spruce, Scots pine, and sessile oak. Additional characteristics of the diameter-dataset provides Table A2. The dataset contains a dbh range from 0.7 cm to 131.9 cm and a cpa range from 0.20 m^2 to 431.7 m^2 and thus the magnitudes necessary for application purposes in this study are covered. The sample size for spruce and beech were highs.

Table A2. Characteristics of the crown projection area-dataset used for fitting Equation (1).

Tree Species		Min	Median	Max	*n*
spruce	cpa (m^2)	0.22	11.80	251.95	9.997
	dbh (cm)	0.7	24.3	109.7	
pine	cpa (m^2)	0.26	10.05	151.36	4.520
	dbh (cm)	5.1	21.5	85.5	
beech	cpa (m^2)	0.29	26.04	431.70	10.348
	dbh (cm)	3.1	18.5	127.6	
oak	cpa (m^2)	0.20	19.39	348.03	3.937
	dbh (cm)	3.6	29.6	131.9	

cpa = crown projection area, dbh = diameter at breast height.

Appendix B.2. Diameter over Stand Age

We required two typical mean diameter developments over age. This was necessary for exemplarily inferring crown projection area development from diameter development within the exemplary calibration and demonstration of the approach.

In order to obtain a generic relation of mean tree diameter to age that covers most common conditions of site and thinning, we analyzed the data of the third NFI [33]. We formed one subset of the NFI data per tree species and age in years. Per subset, we formed the weighted average diameter. In order to obtain the diameter-to-age relation, we fitted Equation (14) to the resulting data set. Therefore we used OLS-regression Additional characteristics of the diameter-dataset provides Table A3. The dataset contains a dbh range from 7 cm to 51.7 cm and an age range from 9 to 140 years and thus the magnitudes necessary for application purposes in this study are covered. The sample size for spruce is more than twice as high as for beech.

Table A3. Characteristics of diameter-dataset used for fitting Equation (14).

Tree Species		Min	Median	Max	*n*
beech	age (year)	9	74.50	140	54,512
	dbh (cm)	7.0	23.3	46.5	
spruce	age (year)	9	74.50	140	122,743
	dbh (cm)	8.4	29.7	51.7	

dbh = diameter at breast height.

Appendix B.3. Top Height over Stand Age

The relation of top height over stand age (Equation (15)) was exclusively used for controlling the time per thinning intervention in SILVA that has to be defined per top height within the model preferences. In order to obtain a species-specific data set of top height to age we first formed the top height as the 95% height quantile per plot. In beech-dominated plots, we considered that top height as the one of beech. In spruce-dominated plots we defined it as the one of spruce. In order to gain early and frequent interventions during simulation, we calibrated Equation (15) to the 90% quantile of top height over age that results in a relatively young age per given top height. Therefore, we used a non-linear regression. Additional characteristics of the height dataset are provided in Table A4. The dataset contains a top height range from 3.1 m to 46.6 m and an age range from 9 to 140 years, and thus the magnitudes necessary for application purposes in this study are covered. The sample size for spruce is more than twice as high as for beech.

Table A4. Characteristics of height dataset used for fitting Equation (15).

Tree Species		Min	Median	Max	*n*
beech	age (year)	10	83.00	140	54,512
	top height (m)	3.1	26.4	46.1	
spruce	age (year)	9	60.00	140	122,743
	top height (m)	3.6	27.0	46.6	

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Ecosystem service trade-offs for adaptive forest management

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Carbon sequestration

A B S T R A C T

Quantifying ecosystem services as dependent on forest management and analyzing tradeoffs between them can help to make decisions on management more effective, efficient, sustainable, and stable. We use a forest management model (SILVA) to predict changes in ecosystem service provisions. Three stakeholder specific forest management scenarios (multifunctional, wood production, set-aside) for each of two different case study areas in Germany (a more and a less productive one) were simulated. We want to therewith answer how ecosystem service and biodiversity indicators (groundwater recharge, carbon sequestration, wood production, structural diversity of forest stands) depend on forest management and site. Forest management had significant influence on ecosystem service provisions in both case study areas. However, the results strongly depend on the site and on the initial situation in each location. In both case study areas, the production oriented forest management pays for productivity with structural diversity. In contrast, multifunctional oriented forest management pays for groundwater recharge with productivity losses. In the set-aside scenario, current carbon sequestration is high due to increasing forest carbon stocks, however sustainable carbon sequestration is low due to the lack of emission savings.

1. Introduction

In addition to the traditional, distinctly production-oriented stand management, two more recent management paradigms characterize forestry in Europe today. As a strong antithesis to the absolute priority of wood production, the social demand to create set-asides, i.e. areas completely exempt from forestry usage, has increased within recent decades. According to Borrass et al. (2017), multifunctional forest management is an approach that considers several additional aspects in addition to the aspect of wood production and implements them in forest management practice. Thus, a mid-position is taken by multifunctional forestry between the set-aside concept and a purely production-oriented forest management. That type of forestry is typical for municipal and state forest enterprises in Germany (Borrass et al., 2017; Ministerium für Ländliche Entwicklung, Umwelt und Landwirtschaft des Landes Brandenburg, 2006; Bayerische Staatsforsten, 2008; Niedersächsisches Ministerium für Ernährung, Landwirtschaft und Verbraucherschutz, 2017). Such forest owners aim to be economical and at the same time they also consider a variety of additional social concerns (Borrass et al., 2017; Boncina, 2011).

Beyond forest management at the stand level, forest management on a landscape scale level has become a common concern in European countries (Michanek et al., 2018; Blattert et al., 2018). Scientific landscape investigations consider the pattern of various ecosystems and land cover types (Burkhard et al., 2010; De Groot et al., 2010; Frank et al., 2012). In this simulation study, we focus on forest ecosystems on large areas of the landscape and consider only forest land use. This study therefore deals with large aggregated forest areas on the landscape scale level but does not deal with the entire landscape as defined in De Groot et al. (2010). Forest management on such an aggregated forest landscape scale level is discussed against the background of biodiversity conservation (Michanek et al., 2018; Blattert et al., 2018). Two notably distinct strategies have been established in Europe for bringing together both production and maintenance of biodiversity. One strategy is to combine areas for wood production with set-aside areas (Michanek et al., 2018; Boncina, 2011). That way, wood production and biodiversity are both provided within one spatially distinct forest area (Michanek et al., 2018). Wood production and biodiversity are therefore spatially segregated from each other. Contrary to the segregated approach, the integrated approach applies multifunctional oriented forest management in order to integrate all aspects in one forest stand (Boncina, 2011; Michanek et al., 2018).

The idea of ecosystem services has led to methods that reasonably assess the benefits that ecosystems provide to humans (Raum, 2018).

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The anthropogenic focus of the ecosystem service concept also requires a detailed understanding of the stakeholders' interests in the goods and other services that ecosystems provide. The linkage of ecosystem services to stakeholders is crucial for sustainable ecosystem management (Raum, 2018). In famous political milestones in forest management such as the Sylvicultura Oeconomica (Carlowitz, 1713) and Helsinki criteria (MCPFE, 1993), the early and the more recent claims of society in forests have been written down. These claims are now recognizable in the forest policies of many countries that aim to provide social, economic, and ecological sustainability in state forests (U.S. Department of Agriculture Forest Service, 2012; Niedersächsisches Ministerium für Ernährung, Landwirtschaft und Verbraucherschutz, 2017; Bayerisches Staatsministerium für Ernährung, Landwirtschaft und Forsten, 2005; Caboun et al., 2014; Brukas et al., 2013). In order to achieve these goals through forest management planning and resource allocation on the political level, forest managers and policy actors require knowledge about ecosystem services and their tradeoffs and synergies (Daily et al., 2009). Detailed scientific knowledge therefore has to be represented on an abstraction level that is typical for decision makers and stakeholders (Armatas et al., 2018; Deal et al., 2017; Kline et al., 2013; European Commission, 2011; IPBES, 2018). Biodiversity is therefore a prominent example for a variable affected or fostered by forest management, and hence influences ecosystem services (Sing et al., 2018). Biodiversity is the foundation of ecosystem functions (Millennium Ecosystem Assessment, 2005). Ecosystem functions like biodiversity are the basis of services and they are therefore critical for the provision of all other services (Costanza et al., 2017). This definition clarifies the differentiation between biodiversity and ecosystem services. Furthermore, this statement underpins the benefit biodiversity provides to humans. However, in the face of future challenges and societal demands, a variety of ecosystem service and biodiversity indicators require consideration by forestry management. The tradeoffs between groundwater recharge, tree species and structural diversity, wood production and carbon sequestration are crucial. Since the frequency of drought incidents will increase in the future (Turral et al., 2011; Spinoni et al., 2018), groundwater recharge will likely become a limiting process for public water supply. Biodiversity and its tree species richness and structural diversity indicators are criteria of risk mitigation towards climate change as well as a forest management goal itself (Knoke et al., 2008; Brockerhoff et al., 2017). Wood production provides renewable raw materials that are in high demand and wood production jobs are maintained therewith (Sikkema et al., 2011; Hetsch, 2008). Carbon sequestration contributes to the mitigation of climate change and can be fostered through both afforestation and forest management (Naudts et al., 2016). However, that target might be missed unless it is generally acknowledged that not every type of forestry is suitable for the mitigation of climate change. Although the process of creating concepts for ecosystem services in science and politics has an obvious advantage, in practice the quantitative assessment of ecosystem services is still a challenge (Hamel and Bryant, 2017).

Previous work has underpinned the influence of forest management on ecosystem service and biodiversity indicators (Felipe-Lucia et al., 2018). Pang et al. (2017) have found substantial trade-offs between provisioning services and other services. Furthermore, Sing et al. (2018) showed that low intensity management is unsuitable for high biomass production, yet it provides high or moderately high levels of other services. Higher intensity management affects biodiversity, health and recreation, as well as water supply services negatively. As a consequence of higher water demand from fast growing species, there is a trade-off between productivity and water availability (Ellison et al., 2012; Schwaiger et al., 2018; Nisbet et al., 2011). Species choice seems to be a key variable for water supply and higher interception losses of conifers compared to broadleaved species are especially crucial (Keenan and van Kijk, 2010; Fürst et al., 2006).

Scientific knowledge about various aspects of forest carbon sequestration has been augmented. Yet, the operational implementation of alternative forest management strategies requires transfer of these findings on an operational scale where actual management decisions are taken (Seidl et al., 2007). Toraño Caicoya et al. (2018) pointed out the relevance of the investigated forest area scale for interpretation of management effects. Results show the potential of simulation methods for the provision of information about trade-offs between ecosystem services (Marques et al., 2017). In particular, water balance and development on a landscape level scale are predestined to be investigated quantitatively with simulation models (Pandeya et al., 2016), because development of water balance and variables related to large forest areas depend on the interaction of many boundary conditions. It is therefore difficult to investigate the effect of forest management on those variables empirically.

The study at hand evaluates simulation scenarios of future forest development for two German aggregated forest areas on a landscape level scale, a pine-dominated one characterized by dry and poor conditions and a spruce-dominated one characterized by moist and fertile conditions. One case study area is therefore representative of less productive Central European sites, and the second case study area is representative of highly productive Central European sites. Accordingly, the simulated management scenarios individualize common management orientations specifically for both of these generalized sites. The study applies every management orientation separately per simulation run (simulation of 100 years) to the entire forest area. These forest management orientations (stakeholder group aiming at specific output) focus on (1) wood production, (2) no management (set-aside), and (3) multifunctional forestry. The study thus compares six simulation runs of markedly diverse site and management conditions in order to evaluate their effect on growing stock and harvest and – beyond that – on groundwater recharge, tree species composition, structural diversity, wood production and carbon sequestration. Our study aims to

(1) Reveal the leeway for influencing a portfolio of forest ecosystem service and biodiversity indicators.
(2) Quantify different carbon stocks in scenarios with and without forest management.
(3) Reveal indications for trade-offs between ecosystem service and biodiversity indicators.

2. Material and methods

2.1. Case study areas

The Lieberose Schlaubetal Neuzelle (LSN) case study area in northeastern Germany, Brandenburg has a forest area size of 78,000 ha. The German Federal State of Brandenburg owns 54% of the area. Within that landscape, there are volumes per species proportions of about 83% Scots pine (*Pinus sylvestris* L.) and 8% sessile oak (*Quercus petraea* (Mattuschka) Liebl). This case study area was chosen to represent a typical, less productive Central European site. In contrast, we selected a highly productive Central European site. The forest area of the second case study area is 53,000 ha and its name is Augsburg Western Forests (AWF). The German Federal State of Bavaria owns 25% of this area. This area is located in southern Germany, Bavaria and has species compositions of about 75% Norway spruce (*Picea abies* (L.) H. Karst) and 7% European beech (*Fagus sylvatica* L.). Hence, shade tolerant tree species are dominant in AWF and primarily light demanding tree species are prevalent in LSN.

Both case study areas have in common that age and hence diameter classes are not distributed equally within the forest area (Fig. 1). Most stands growing on the aggregated forest area on a landscape level scale have a mean diameter of less than 30 cm. Notwithstanding, the extent of areas having a small mean diameter differs between both case study areas. In LSN, 75% of the area is stocked with stands having a mean diameter of less than 30 cm, whereas in AWF, there are more than 55%

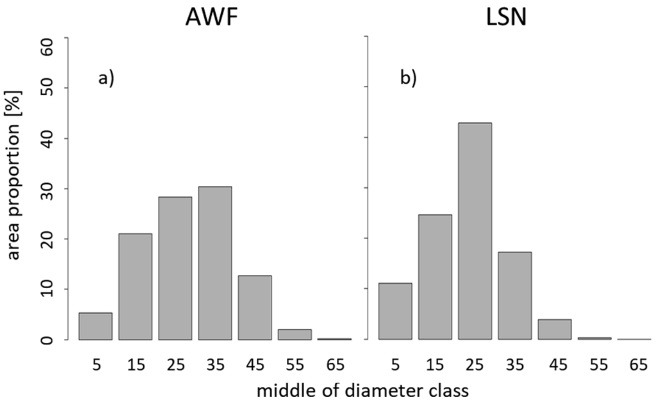

Fig. 1. Forest area proportions of mean diameter classes within both case study areas. AWF has higher shares of larger diameters than LSN.

of stands with a mean diameter of less than 30 cm (Fig. 1 a and b).

The sites in LSN are marked by sand-dominated, base-poor unconsolidated rocks of weak nutrient supply. Their plant available soil water content is typically 177 mm and characterizes an average soil profile of 90 cm depth (Table A.4). In contrast, the site-conditions in AWF are dominated by lowland soils consisting of loess (a clay-silt deposit) with high nutrient supply (Andreae et al., 2016; Grüneberg et al., 2016). Accordingly, the plant-available water content is 257 mm with a profile of 125 cm depth (Table A.4). LSN, in addition to its less favorable soil conditions, has a comparatively low precipitation of 590 mm/yr^{-1}. Thus, the precipitation average in LSN is less than two thirds of the one in AWF, which is as much as 940 mm/yr^{-1}.

2.2. Forest growth simulation with SILVA

SILVA (Pretzsch et al., 2002) is a distance-dependent, individual tree growth model. We used it to conduct scenario simulations based on forest inventory data (Fig. 2). Thereby, we predict forest management-dependent forest growth of 100 years. This simulator applies the potential modifier method (Pretzsch, 2009). This method refrains from representation of individual tree-internal ecophysiological processes, but modifies individual tree growth potentials in dependence of its individual situation of competition. The potential growth is sensitive to climate and soil conditions (Kahn and Dursky, 1999) and is based on one age-dependent function of potential height growth and an additional one of potential diameter growth. SILVA's growth algorithm works with time steps of five years and represents the effect of specially-related management interventions on competition and growth. Therefore, SILVA's growth module within each step computes potential stem size growth per each individual tree based on the tree's current height, diameter, and a concise set of climate and soil quality indicators. In order to obtain the tree's effective dimension growth within a considered time step, the model reduces the tree's potential height and diameter increase by competition factors that are based on dominance relations within the tree's vicinity. Thus, this forest growth simulator is appropriate for the representation of the effect of forest management on forest growth on an aggregated forest landscape scale with a high level of detail. SILVA has a top height-driven silvicultural module that implements comprehensive rule-based intervention settings.

2.3. Input data preparation

Simulations on an aggregated forest landscape scale level with SILVA are predominantly based on sample plots that constitute the primary inventory unit in the widely-used, grid-based type of forest inventory. The density of such inventories typically limits the resolution of scenarios to one that is notably coarser than the stand level. However, knowledge of the grid-width enables association of each sample plot to a representative area for representation of the aggregated forest landscape average of simulation results. Therefore, prior

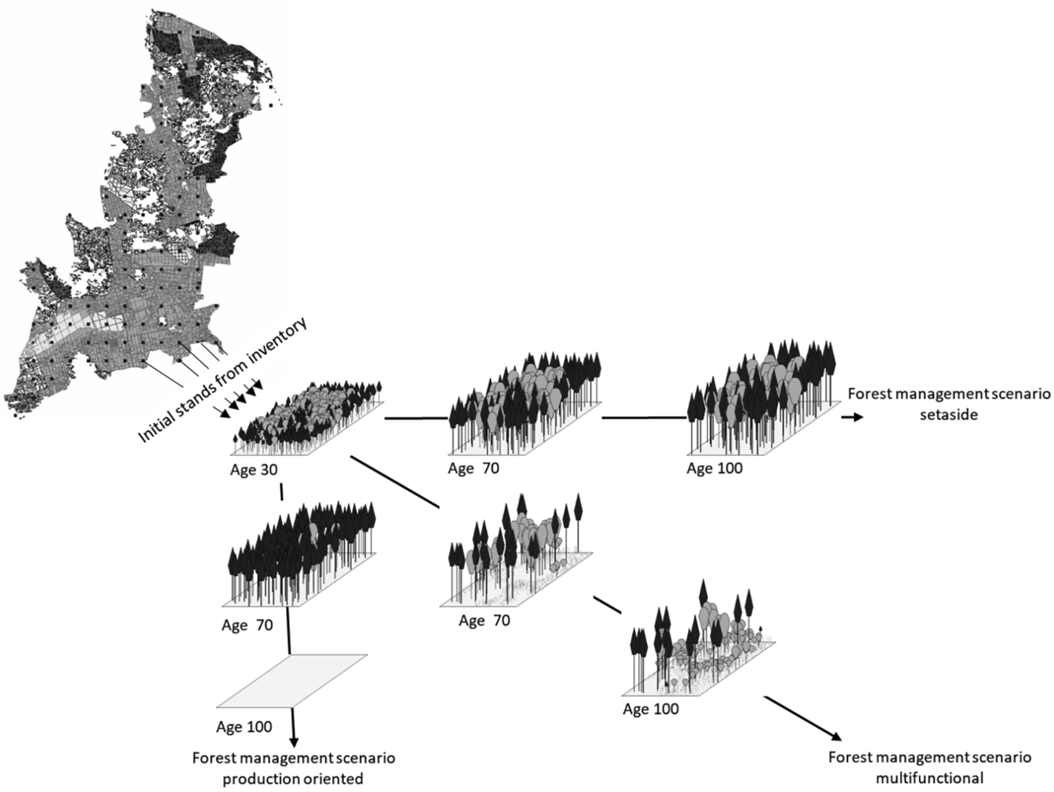

Fig. 2. Schematic representation of inventory-based scenario simulation with SILVA. Forest growth of initial virtual stands derived from forest inventories were simulated with three different forest management scenarios (set-aside, multifunctional, production oriented).

Table 1
Forest management scenarios for AWF.

Where and when				What kind and strength of intervention	
Multifunctional Augsburg Western Forests	Coniferous stands	top height	12–25 m	Positive selective thinning	
		frequency	5 yr	Number of positively selected trees	100 trees ha^{-1}
				Competitors to remove per positive selected tree	2 trees ha^{-1}
				Maximum harvest volume	55 m^3 ha^{-1}
		top height	25–32 m	Positive selective thinning	
		frequency	5 yr	Number of positively selected trees	200 trees ha^{-1}
				Competitors to remove per positive selected tree	1 tree ha^{-1}
				Maximum harvest volume	65 m^3 ha^{-1}
		top height	> 32 m	Positive selective thinning	
		frequency	10 yr	Number of positively selected trees	200 trees ha^{-1}
				Competitors to remove per positive selected tree	1 tree ha^{-1}
				Target diameter felling	Coniferous 45 cm
					Broadleaves 60 cm
				Maximum harvest volume	80 m^3 ha^{-1}
		top height	> 32 m	Planting	
		frequency	once	Number of planted trees	5500 beeches ha^{-1}
					500 firs ha^{-1}
	Broadleaved stands	top height	12–17 m	Positive selective thinning	
		frequency	10 yr	Number of positively selected trees	100 trees ha^{-1}
				Competitors to remove per positive selected tree	2 trees ha^{-1}
				Maximum harvest volume	40 m^3 ha^{-1}
		top height	17–30 m	Positive selective thinning	
		frequency	10 yr	Number of positively selected trees	75 trees ha^{-1}
				Competitors to remove per positive selected tree	1 tree ha^{-1}
				Maximum harvest volume	70 m^3 ha^{-1}
		top height	> 30 m	Positive selective thinning	
		frequency	10 yr	Number of positively selected trees	75 trees ha^{-1}
				Competitors to remove per positive selected tree	1 tree ha^{-1}
				Target diameter felling	Coniferous 45 cm
					Broadleaves 60 cm
				Maximum harvest volume	80 m^3 ha^{-1}
Wood production Augsburg Western Forests	Coniferous stands	top height	12–16 m	Thinning from below	
		frequency	10 yr	Maximum harvest volume	35 m^3 ha^{-1}
		top height	16–28 m	Thinning from below	
		frequency	10 yr	Maximum harvest volume	100 m^3 ha^{-1}
		top height	> 28 m	Shelterwood cutting	
		frequency	10 yr	Maximum harvest volume	500 m^3 ha^{-1}
		top height	> 30 m	Planting	
		frequency	once	Number of planted trees	4000 spruces ha^{-1}
	Broadleaved stands	top height	12–17 m	Positive selective thinning	
		frequency	10 yr	Number of positively selected trees	75 trees ha^{-1}
				Competitors to remove per positive selected tree	1 tree ha^{-1}
				Maximum harvest volume	40 m^3 ha^{-1}
		top height	17–30 m	Positive selective thinning	
		frequency	10 yr	Number of positively selected trees	75 trees ha^{-1}
				Competitors to remove per positive selected tree	1 tree ha^{-1}
				Maximum harvest volume per intervention	70 m^3 ha^{-1}
		top height	> 30 m	Shelterwood cutting	
		frequency	10 yr	Maximum harvest volume	500 m^3 ha^{-1}
		top height	> 30 m	Planting	
		frequency	once	Number of planted trees	4000 spruces ha^{-1}

to simulation, an initial data preparation process classified all inventory plots in the case study areas by their stand type. The variables of the stratification were quadratic mean diameter class, main tree species, secondary tree species, and owner type. Thereby, 277 stand types (see Table A.1) were identified in AWF and 66 stand types (see Table A.2) were identified in LSN. The data preparation subsequently formed one representative stand per each stand type from the structural properties of the type's inventory plots. Such a virtual stand constitutes an efficient spatial unit of simulation, since it has a size of only a few ha. Concomitantly, each virtual stand is associated to exactly one stand type and thus represents a particular area size that results from the type's inventory plots. Several distinct inventory types contributed to stratification and construction of virtual stands, since the availability per inventory data set was dependent on the case study area.

Two inventory types were available within AWF. One had an exceptionally dense grid width of 200 m. It could be obtained exclusively from state forest areas. We used 3455 plots from this inventory. In non-state forest areas in AWF, the only inventory that was available was the German National Forest Inventory (Polley, 2011). The grid width of that national inventory data set (Thünen Institute, 2012) is notably coarser than that of the Bavarian State forest (2.8 km). However, in the National Forest Inventory each grid point represents a north-to-south oriented square with an edge length of 150 m. There is exactly one inventory plot at the corners of each square. We used 692 plots from this inventory. The second case study area, LSN, is covered by the inventory of the German Federal State of Brandenburg (Ministerium für Ländliche Entwicklung, Umwelt und Landwirtschaft des Landes Brandenburg, 2015). This inventory comprises the National Forest Inventory plots that have a grid width of 4 km in that region. However, the German Federal State's inventory complements this grid with additional points and therefore has a denser grid with a grid width of 2 km. We used 1672 plots from this inventory.

2.4. Forest management scenarios

The settings of the forest management scenarios in the study at hand were designed per case study area and in close cooperation with stakeholders from each forest management orientation group (Juerges et al., 2017). Additionally, we used official guidelines that were available in both case study areas (Bayerische Staatsforsten, 2009, 2011; Ministerium für Ländliche Entwicklung, Umwelt und Landwirtschaft des Landes Brandenburg, 2006). Municipal and state forest organizations represented the multifunctional forestry oriented stakeholder group within each case study area. In contrast, the typical objectives of large private forest owners who are financially dependent on wood production and therefore apply conventional stand treatment contributed to the management settings in the production scenario (Juerges et al., 2017). In the production-oriented scenario, rotation period and target diameter were adjusted to (1) the site-specific point in time that corresponded to the culmination of the mean annual increment (Schober, 1995) and (2) furthermore to the harvest costs per cubic meter of wood (law of piece volume) (Möhring, 2010). Stakeholders of nature conservation called for larger proportions of set-aside areas. From the stakeholder-specific information of both case study areas, we hence created the following accentuated forest management settings consequently idealized for one single forest management orientation. This way, we implemented forest management settings that frame the most extreme options stakeholders in the study areas would consider. This reveals the leeway for forest management options.

2.4.1. Multifunctional oriented forest management

In AWF, the multifunctional forest management scenario primarily aims at moderate volume stocks that are a prerequisite for structured stands and continuous cover. In LSN, a somewhat less heterogeneity among individual trees was aimed for, since the share of oak shall be increased while maintaining the quality of sawn timber obtained from that species: to that end overstory gaps have to be opened to maintain even-aged oak regeneration at high local stem density. Multifunctional oriented forest management in the study at hand is characterized by positive selective thinning that is complemented from the onset of harvest with target diameter felling. In positive selective thinning, trees are removed to increase the volume increment of the best trees. In LSN, target diameter felling is combined with group cutting in order to initiate oak regeneration (Tables 1 and 2). Therefore, the aimed increase of broadleaved species' shares shall be achieved by consequent facilitation in positive selective thinning interventions and group-wise planting of broadleaves inside of the coniferous monocultures. Considering both case study areas, AWF and LSN, the multifunctional management scenario aims at comprehensive advanced artificial beech reproduction under spruce canopy and oak planting inside of the gaps in the pine stands.

2.4.2. Wood production oriented forest management

The adjustments of the wood production oriented forest management scenario are dominated by thinning from below if coniferous stands are considered with shorter rotations compared to the multifunctional oriented forest management scenario. At the end of the rotation period, final fellings are executed fast (removal of crop trees for a fast change to the next forest stock generation) through large harvest amounts as part of shelterwood cutting systems (Tables 1 and 2). Short rotations aim at the maximization of mean annual increment. Low removal quantities in pre-commercial thinning interventions serve minimization of costs and maximization of standing volume at final felling age. In broadleaved stands, positive selective thinning is continued until the volume increment has culminated and diameters are large enough to reduce harvest costs per cubic meter and to dimensions suitable for sawable wood. Within the production-oriented scenario, such stands are to be replaced by coniferous stands. In general, the share of coniferous species is increased through positive selection interventions and planting of spruce in stands that are still dominated by broadleaved species.

2.4.3. Process protection oriented set-aside

The set-aside scenario does not involve any silvicultural interventions. That scenario of ecological process protection represents the typical development of a forest landscape in the total absence of any further management intervention.

2.5. Assessment of ecosystem services and biodiversity

For ecosystem service and biodiversity assessment, we simulated the development of ecosystem service and biodiversity indicators (Table 3). Biodiversity was assessed by the structural diversity of a forest and thus the Species-Profile-Index (Pretzsch, 1996) was used. For productivity, the volume increment was used. For groundwater recharge, a new and suitable simulation algorithm was applied (Schwaiger et al., 2018). For carbon sequestration, a new model of Biber et al. (2018) was applied.

2.5.1. Carbon sequestration

For assessing carbon sequestration, we used an approach that has been tailored for post-hoc-application on the output of modern forest growth simulators in the context of the EU-project ALTERFOR (Biber et al., 2018). This method traces the most important C-stocks in the forest, in wood products (as related to a given forest area), and estimates C-emission savings due to using wood products instead of such made from other raw materials. The forest based stocks are above ground coarse wood, coarse roots, fine above ground material (branches, twigs, leaves), and fine roots. All of these stocks relate to living trees. They are estimated based on the available wood volumes, applying classic conversion factors as provided e.g. by Pretzsch (2009). Above and below ground deadwood in the forest is only traced by its coarse wood fraction. Their initial amounts have to be provided by the user as fractions of the living above ground volume (Table A.3). Inflows to these stocks come from mortal trees and coarse wood components remaining in the forest after harvest. Their outflows are calculated by means of exponential decay with typical half-life times provided by the user (Table A.3). Carbon stocks in the soil are not accounted for so far, since changes in soil carbon over time are usually small compared to the above mentioned ones. We consider the effort required for users to quantify their initial values and their in-and outflows disproportional compared to the added value they would bring for applications such as in this study.

If wood is harvested, the resulting C-flows are split into saw logs, pulpwood, logs that remain in the forest, stumps, and other harvest residues. Stumps and remaining logs increase the deadwood C-stocks. A user-defined share of the other harvest residues is assumed to be used for energy provision; the remainder is also accounted for as deadwood (Table A.3).

For the saw logs, the user can define the percentage of losses due to industry processing (sawdust, etc.). These losses are assumed to be used for energy provision. The remaining material is used either for producing sawn wood, wood-based products (typically chipboards) or for generating energy. The user is required to specify the relative shares of these different utilizations. In a very similar way, harvested pulpwood, after subtracting process losses, can be used for producing paper or wood-based products or for energy provision.

The C-amounts stored in the produced sawn wood, wood-based products and paper are inflows to the three C-stocks of products made from these three raw product categories. The outflows of these stocks are calculated as exponential decay considering typically different half-life times of these product categories. The product stocks comprise only the amounts of products that come from the forest area of interest, i.e. imported products are not included. Their initial values can be estimated based on available statistics (Pilli et al., 2015; IPCC, 2006).

As a final stage, the model allows calculation of the net C-emission savings resulting from using wood for energy provision (process losses, and optionally parts of the harvest residues, saw logs and pulpwood) and products (made from sawn wood, wood-based products, paper)

Table 2
Forest management scenarios for LSN.

Where and when				What kind and strength of intervention	
Multifunctional Lieberose Schlaubetal Neuzelle	Coniferous stands	top height	12–20 m	Positive selective thinning	
		frequency	10 yr	Number of positively selected trees	150 trees ha^{-1}
				Competitors to remove per positive selected tree	2 trees
				Maximum harvest volume	40 m^3 ha^{-1}
		top height	20–28 m	Positive selective thinning	
		frequency	10 yr	Number of positively selected trees	150 trees ha^{-1}
				Competitors to remove per positive selected tree	1 tree
				Maximum harvest volume	50 m^3 ha^{-1}
		top height	> 28 m	Positive selective thinning	
		frequency	10 yr	Number of positively selected trees	150 trees ha^{-1}
				Competitors to remove per positive selected tree	1 tree
				Target diameter felling (combined with group cutting)	Coniferous 40 cm Broadleaved 60 cm
				Maximum harvest volume	130 m^3 ha^{-1}
		top height	> 28 m	Planting	
		frequency	once	Number of planted trees	7000 oaks ha^{-1}
	Broadleaved stands	top height	12–22 m	Positive selective thinning	
		frequency	10 yr	Number of positively selected trees	100 trees ha^{-1}
				Competitors to remove per positive selected tree	2 trees
				Maximum harvest volume	30 m^3 ha^{-1}
		top height	22–30 m	Positive selective thinning	
		frequency	10 yr	Number of positively selected trees	80 trees ha^{-1}
				Competitors to remove per positive selected tree	1 tree
				Maximum harvest volume	50 m^3
		top height	> 30 m	Positive selective thinning	
		frequency	10 yr	Number of positively selected trees	60 trees ha^{-1}
				Competitors to remove per positive selected tree	1 tree
				Target diameter felling (combined with group cutting)	Coniferous 40 cm Broadleaved 60 cm
				Maximum harvest volume	100 m^3 ha^{-1}
Wood production Lieberose Schlaubetal Neuzelle	Coniferous stands	top height	12–18 m	Thinning from below	
		frequency	10 yr	Maximum harvest volume	30 m^3 ha^{-1}
		top height	18–27 m	Thinning from below	
		frequency	10 yr	Maximum harvest volume	40 m^3 ha^{-1}
		top height	> 27 m	Shelterwood cutting	
		frequency	10 yr	Maximum harvest volume	150 m^3 ha^{-1}
		top height	> 27 m	Planting	
		frequency	once	Number of planted trees	5000 pines ha^{-1}
	Broadleaved stands	top height	12–22 m	Positive selective thinning	
		frequency	10 yr	Number of positively selected trees	100 trees ha^{-1}
				Competitors to remove per positive selected tree	2 trees
				Maximum harvest volume	40 m^3
		top height	22–30 m	Positive selective thinning	
		frequency	10 yr	Number of positively selected trees	80 trees ha^{-1}
				Competitors to remove per positive selected tree	1 tree
				Maximum harvest volume	50 m^3 ha^{-1}
		top height	> 30 m	Positive selective thinning	
		frequency	10yr	Number of positively selected trees	60 trees ha^{-1}
				Competitors to remove per positive selected tree	1 tree
				Target diameter felling	Coniferous 40 cm Broadleaved 60 cm
				Maximum harvest volume	150 m^3 ha^{-1}
		top height	> 30 m	Planting	
		frequency	once	Number of planted trees	5000 pines ha^{-1}

Table 3
Ecosystem service and Biodiversity estimation based on indicators available from SILVA stand structure simulation output.

Target dimension	Indicator
Groundwater recharge	Groundwater recharge (Schwaiger et al., 2018)
Productivity	Volume increment
Biodiversity	Species-Profile-Index (Pretzsch, 1996)
Carbon sequestration	emission savings, carbon balance (Biber et al., 2018)

instead of other raw materials (displacement factor estimates based on (Smyth et al., 2017; Sathre and O'Connor, 2010; Oliver et al., 2014).

This approach allows the calculation of periodic carbon balances, optionally including the forest C-stocks only, extending the scope by also considering product C-stocks, and finally also with the inclusion of C-emission savings.

For indication of carbon-sequestration related ecosystem services, we distinguish between the overall balance (current carbon sequestration) including wood product, forest and emission saving C-stocks and the sequestration resulting exclusively from emission savings (sustainable carbon sequestration). A positive overall carbon balance is the annual change of carbon storage that reveals a desirable net uptake of carbon. Negative balances reveal a net emission. Carbon-sequestration resulting from emission savings is sustainable because the stock of accumulated emission savings cannot decay. In contrast, forest and wood product stocks are balanced within a long time range because input and output become equal. The only remaining and thus sustainable carbon-sequestration regarding a long period results from emission savings.

2.5.2. Biodiversity

We assessed biodiversity by its structural diversity of a forest stand indicator and the Species-Profile-Index (Pretzsch, 1996) was therefore used. The species profile index (*K*) (Pretzsch, 1996) is a combined measure of a stand's richness in both tree species and vertical structure. Basically, it is an extension of the concept of the Shannon (1948) Species Diversity Index. In a single layered monoculture, its value is 0, while its maximum value for a two-species mixture is $\ln(6) \approx 1.79$, which would indicate both species being equally represented in and among the stand's upper, middle, and understory.

2.5.3. Groundwater recharge

The ecosystem service groundwater recharge (*GWR*) was calculated according to a novel approach of Schwaiger et al. (2018) that had already been implemented in SILVA. Based on extensive simulation studies with an ecophysiological forest simulation model, Schwaiger et al. (2018) propose an algorithm for estimating *GWR* (in mm a^{-1}) in mixed species stands with indicators of stand structure.

$$GWR = a_0 + a_1 MH + a_2 \cdot VH + a_3 \cdot SDI \quad (1)$$

In a first step (Eq. (1)), *GWR* is estimated species-specific, *MH* is the mixed species stand's arithmetic mean height measured in meters, *VH* is the variation coefficient of tree height, *SDI* (Reineke, 1933) is the stand density index. The variables a_0, a_1, a_2, and a_3 are species and site-specific parameters (Table 4). Schwaiger et al. (2018) parameterized Eq. (1) for beech and spruce in AWF and for oak and pine in LSN. In AWF, we used beech parametrization as a proxy for all broadleaved species and spruce parametrization as a proxy for all conifers. In LSN we used oak parametrization as a proxy for all broadleaved species and pine parametrization as a proxy for all conifers.

In a further step, groundwater recharge contribution of both tree species in a mixed species stand were weighted by the mixing proportion as growing space share (*GSS*) (Eq. (2)).

$$GWR_{total} = GSS_{species1} \cdot GWR_{species1} + GSS_{species2} \cdot GWR_{species2} \quad (2)$$

2.5.4. Productivity

In this paper, productivity of an ecosystem is assessed by the annual forest volume increment. Productive forest ecosystems have high volume increments and the decision what to do with the grown wood depends on preferences of the forest owners. One opportunity is to harvest and produce sawn timber; another option is to leave the grown wood in the forest for the benefits of standing volume or dead wood. The variable annual volume increment is therefore a good measure of the amount of potentially usable wood produced.

2.6. Result data preparation

All result data preparation was carried out with R 3.4.1 (R Core Team, 2013). In order to reveal indications for trade-offs between ecosystem service and biodiversity indicators, we first formed each indicator's average over the whole simulation time (100 years). The averages were compared specific to each management scenario and case study area. We thereby attempted to form pairs of ecosystem service indicators from one notably low and one notably high value. Within the study at hand, such a pair is defined as a tradeoff.

Table 4
Parameters for the stand structure dependent calculation of the species-specific groundwater recharge contribution in mixed species stands.

Case study area	tree species	a_0	a_1	a_2	a_3
AWF	European beech	463.71		154.61	
	Norway spruce	418.72	5.61	234.46	−0.28
LSN	Sessile oak	386.19	−2.01	−132.54	
	Scots pine	279.91	−1.17		

Furthermore, we show absolute and relative development of the ecosystem service and biodiversity indicators. From this approach, we expect to reveal the management-specific influence on the indicators, while the influence of the site is excluded. The simulation output had therefore been standardized and thus been given as a percent of the indicators' maximum per scenario simulation run. We consider this maximum to be the case study area-specific potential.

3. Results

3.1. Development of forest volume, carbon stock and broadleaf proportion

3.1.1. Volume stock

The time course of the forest volume stock was notably dependent on both case study area and management scenario (Fig. 3 a, b). Inside LSN, the volume stock in the multifunctional scenario was constant in the first 80 years and then markedly dropped to approximately 180 m^3 ha^{-1} (Fig. 3 b). In AWF, the forest stock of the multifunctional scenario remained almost constant on a level of about 400 to 450 m^3 ha^{-1} (Fig. 3 a). In contrast, the forest stock of the wood-production scenario fluctuated between 250 m^3 ha^{-1} and 300 m^3 ha^{-1} in LSN (Fig. 3 b) and between 200 m^3 ha^{-1} and 450 m^3 ha^{-1} in AWF (Fig. 3 a). The set-aside scenario is characterized by a permanently increasing forest stock in both case study areas (Fig. 3 a and b). However, forest stock increased slower, the more advanced the simulation time steps were.

Comparing both case study areas, it was obvious that LSN had lower forest stocks than AWF (Fig. 3 a and b). However, both landscapes became more similar in the simulated stock per scenario if the volume stock values had been standardized and therefore given as percentage of the stock's maximum per case study area (Fig. 3 c and d). Comparing these relative stocks per multifunctional option and per set-aside option, there was an obvious similarity of stock development between both case studies. If the site effect had been taken into account that way, the volume stock of the set-aside scenario in both landscapes would have started at 40%. At the end of the simulation period, it attained its highest value after an increase of 60%. The multifunctional scenario in contrast had a constant value of about 40%. However, the wood-production scenario differed notably between both case study areas even in the development of standardized stock. The average of the relative stock that had been standardized by the maximum stock per site was only at 30% in AWF and even at 50% in LSN.

3.1.2. Broadleaf proportion

Within the LSN investigation area, the broadleaf growing space proportion (share of forest stand area covered by a tree species' canopy) of the multifunctional scenario remained constant at its initial value of about 13% for the first 50 years (Fig. 3 f). Within the ensuing 40 years, the share of broadleaved species increased only slightly. Subsequently after 90 simulation years, it moved strongly upward to approximately 25%. In AWF, the growing space proportion of broadleaved species, if the multifunctional scenario is considered, increased permanently from 25% to more than 60% (Fig. 3 e). The wood production scenario that focused on coniferous species induced a continued decrease of the broadleaved species' share in both case study areas to about 5%. The set-aside scenario in both case study areas was characterized by just slightly, but permanently decreasing broadleaf proportions. Comparing both case study areas, it is apparent that LSN had lower growing space proportions of broadleaves than AWF (Fig. 4 a and b). Considering forest conversion towards higher broadleaf shares, the influence of forest management was obviously smaller in LSN than in AWF.

3.1.3. Total carbon stock

Regarding the whole simulation time, all scenarios have different carbon stocks of forest, wood products and cumulative emission savings in both case study areas. In LSN, the carbon stock including cumulative emission savings (Fig. 4 b) ranged from 160 to 250 t ha^{-1}. In the more

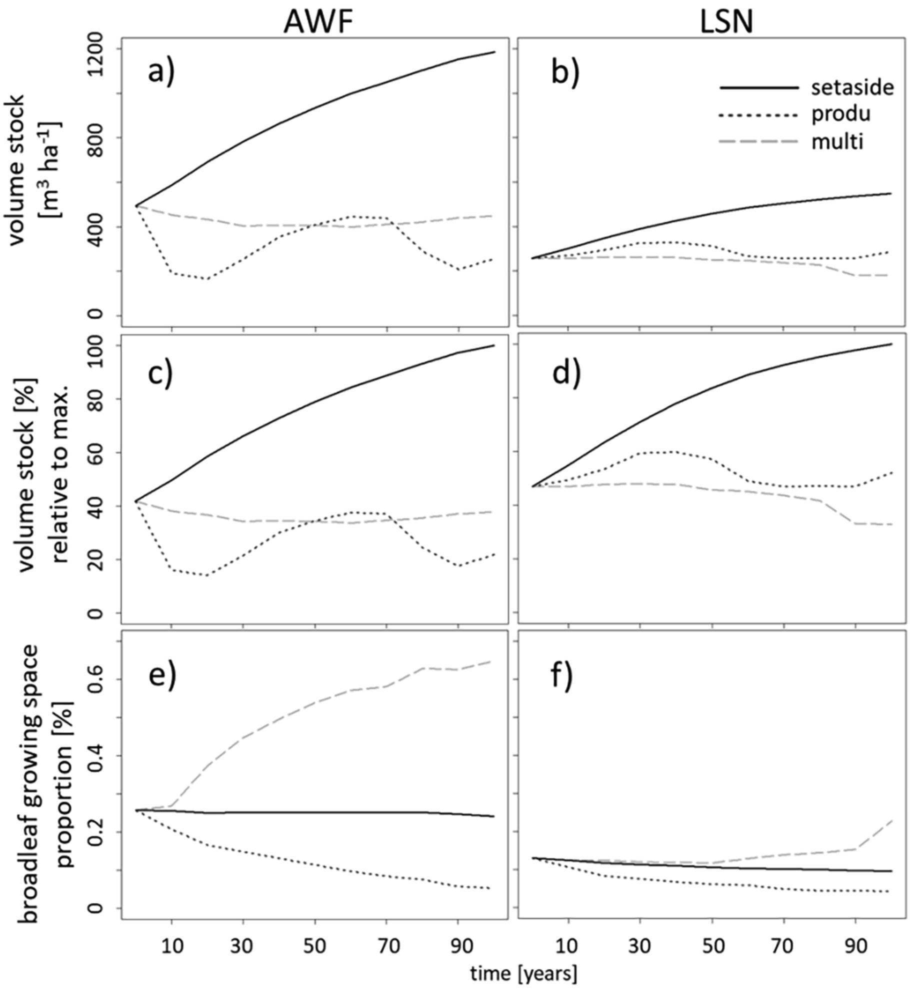

Fig. 3. Development of forest volume stock in $m^3\ ha^{-1}$ (a, b) and in percentage as proportion of occurred simulated maximum (approximation to the case study area-related potentially possible volume stock) (c, d); development of the broadleaf proportion given as the share of growing space/stand area covered by a tree species' canopy (e, f); all figures show the development in dependence of the forest management scenario (see Section 2.3): multifunctional (multi), wood-production (produ), or set-aside; LSN and AWF designate the case study area (see Section 2.1).

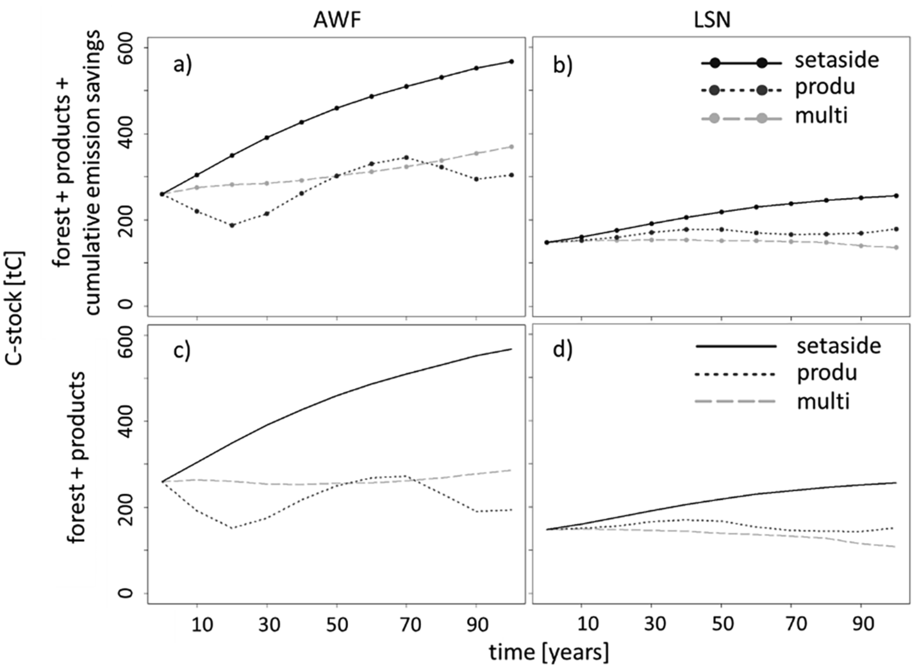

Fig. 4. Carbon stock over simulation time given in tons per ha (forest, products and cumulative emission savings in a, b; forest and products in c, d), with all figures showing the development as dependent on the forest management scenario (see Section 2.3): multifunctional (multi), wood production (produ), or set-aside; LSN and AWF designate the case study area (see Section 2.1).

productive AWF case study area, the scenarios ranged from 180 to 300 t ha^{-1} (Fig. 4 a). Substitution of non-wood materials exclusively resulted from management scenarios with harvest. Thus, cumulative emission savings in the set-aside scenario are zero (same set-aside curves in Fig. 4 a and c, b and d). In both case study areas, the overall carbon stock including emission savings (Fig. 4 a, b) had a marked positive trend in both the multifunctional and the wood production scenario. If the substitution effect had however been neglected, the wood production scenario and the multifunctional scenario would have had almost constant values over the simulation time (Fig. 4 c, d). Although it had no substitution effect, the set-aside scenario was marked by a strong initial increase of carbon stock over time. In both case study areas, that scenario achieved the highest carbon stock of about 250 t C ha^{-1} in LSN and of almost 600 t C ha^{-1} in AWF. However, the carbon stock increase in both case studies became notably slower towards the end of the simulation period and finally ceased in LSN.

If the carbon stock had been standardized through division by its maximum per case study area, it would have revealed a higher impact of forest management in AWF than in LSN. Forest management caused the carbon stock (Fig. 4 a and b) to vary by only 10% in LSN vs. 30% in AWF, relating the values to the occurring simulation maximum (set-aside marks the highest value of bound carbon possible within the simulated scenarios).

3.2. Development of ecosystem services and biodiversity indicators

3.2.1. Sustainable carbon sequestration

The scenarios differed in their sustainable carbon sequestration, as became obvious from their emission savings (Fig. 5c and d). In the LSN investigation area, the sustainable carbon sequestration of the multifunctional scenario remained almost constant at its initial state of about 0.25 tC yr^{-1} ha^{-1} (Fig. 5 d). At the end of the simulation time, when volume stock decreased, multifunctional emission savings conversely increased to about 0.5 tC yr^{-1} ha^{-1}. In AWF, considering the multifunctional scenario, emission savings remained at almost 1 tC yr^{-1} ha^{-1} over the whole simulation period. A more fluctuating development can be seen in the wood production scenario. In LSN, emission savings are between 0.2 and 0.6 tC yr^{-1} ha^{-1} and in AWF between 0.4 and 2.75 tC yr^{-1} ha^{-1} (Fig. 5 c and d). The sum of emission savings per scenario (which is also shown in Fig. 4 as cumulative emission savings) over the whole simulation time is equal with harvest in both scenarios in each distinct case study area. The set-aside scenario that presumed absence of forestry interventions and harvest was characterized by zero emission savings. Thus there was no substitution effect in either case study area (Fig. 5 c and d). Comparing the case study areas, it is apparent that LSN clearly has lower sustainable carbon sequestration than AWF.

3.2.2. Current carbon sequestration

The indicator for current carbon sequestration (carbon balance) was strongly dependent on the management scenario (Fig. 5 a and b). Current carbon sequestration intermittently had negative values in the wood production and multifunctional scenarios. These negative balances correlate with periods of volume stock loss (Fig. 3). Carbon bound in the volume stock decreases at the moment of harvest and shifts mostly to the wood products storage. However, from this point in time, carbon is emitted for example by rotting of harvest residuals or burning. Thus, negative balances are created with periods of volume stock loss. In both case studies and particularly in AWF, the balance of the production scenario fluctuated due to the intermittent strong incident of harvest. At this, the multifunctional and the wood production scenarios had a positive average carbon balance (Fig. 5 a) and b). Furthermore, the set-aside balance fell below the average of the multifunctional and production scenario in both case study areas towards the end of the simulation time. While the set-aside scenario generally had higher current carbon sequestration than both of the other forest management scenarios, it did not produce any wood products. Thus, the release of carbon from deadwood and wood products in that scenario was only counterbalanced by forest growth. As the forest stock approached its saturated carbon storage, the carbon balance at ecological process protection continuously decreased towards a zero balance and carbon footprint.

3.2.3. Groundwater recharge

Comparing both case study areas by groundwater recharge (Fig. 6 a and b), it is apparent that LSN has less groundwater recharge than AWF, which is marked by higher annual average precipitation. In the considered scenarios, forestry interventions in AWF caused the recharge values to vary between 300 mm yr^{-1} and even 500 mm yr^{-1}, while in LSN they induced a variation range of only 250 mm yr^{-1} to 260 mm yr^{-1}. Even the lowest value of groundwater recharge in AWF is much higher than the highest value in LSN. In AWF, groundwater recharge in the long term increased in the set-aside and in the

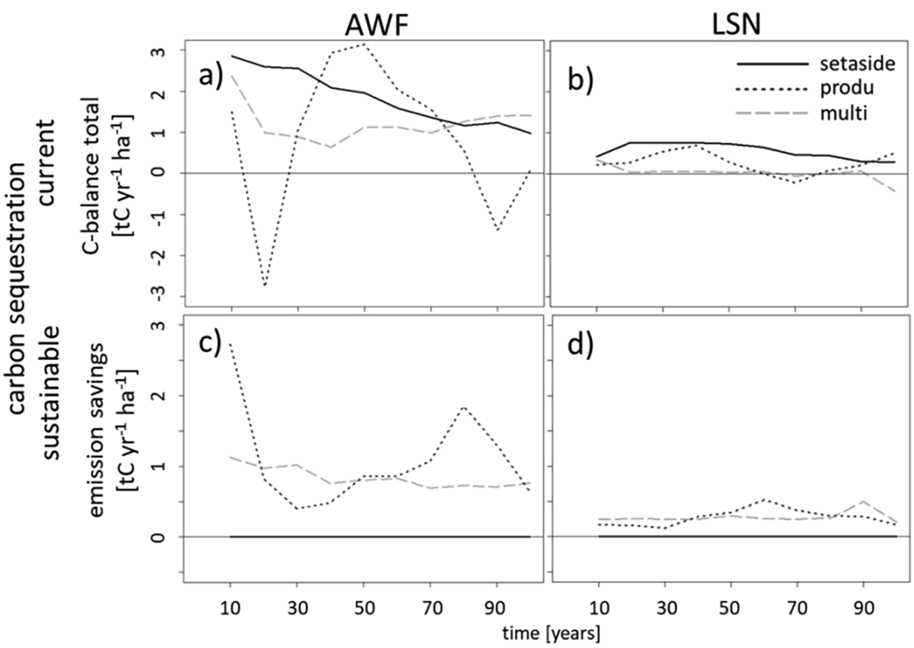

Fig. 5. Development of C-balance total (see Section 2.4) over the simulation time given in tons per ha (a, b), and carbon emission savings (see Section 2.4) over the simulation time given in tons per ha (c, d); all figures show the development to be dependent on the forest management scenario (see Section 2.3): multifunctional (multi), wood production (produ), or set-aside; LSN and AWF designate the case study area (see Section 2.1).

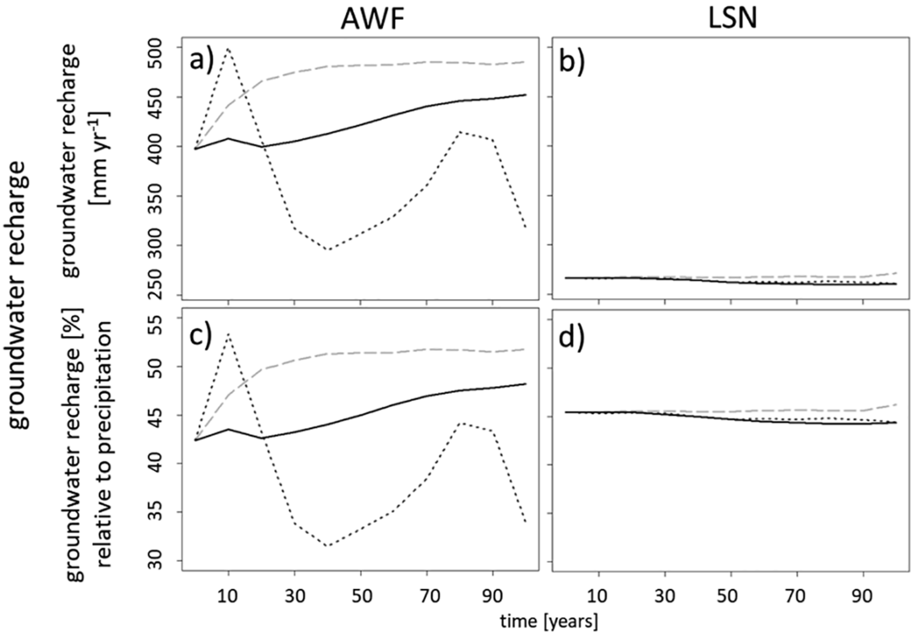

Fig. 6. Development of groundwater recharge (see Section 2.4) over the simulation time given in mm yr^{-1} (a, b), and groundwater recharge over the simulation time standardized through division by the average of annual precipitation per case study area (c, d); all figures show the development to be dependent on the forest management scenario (see Section 2.3): multifunctional (multi), wood production (produ), or set-aside; LSN and AWF designate the case study area (see Section 2.1).

multifunctional scenario. In contrast, the wood production scenario had a negative trend. In LSN, groundwater recharge of the set-aside and the wood production scenario were slightly decreasing. Groundwater recharge of the multifunctional scenario was however constant over 90 years and increased during the last 10 years of simulation.

Even if the groundwater recharge had been standardized – in that particular case – through division by the average annual precipitation per case study area, it had a notably larger range of variation in AWF. Obviously, management in AWF had a stronger influence on groundwater recharge than in LSN in the simulation time period being considered (Fig. 6 c and d). However, the magnitude of mean standardized groundwater recharge is equal when comparing both case study areas.

3.2.4. Productivity

Regarding the average of volume increment within 100 simulation years, the wood production scenario had the highest, the multifunctional one the lowest, and the set-aside scenario an intermediate productivity (Fig. 6 a and b) in both case study areas.

Comparing the case study areas, it is apparent that AWF had a two to three times higher volume increment per scenario than LSN. Indeed, if productivity had been standardized through division by its maximum per case study, its relative range would have been equivalent in LSN and AWF (Fig. 6 c and d). In relative terms, the lowest value that occurred during the simulation time was about 50% of the case study-related maximum within each case study area.

3.2.5. Biodiversity

In LSN, all three scenarios differed just slightly within the first 40 years (Fig. 7 b). Subsequently however, the structural diversity of forest stands biodiversity indicator decreased in the production and set-aside scenarios. On the other hand, the structural diversity of forest stands in the multifunctional scenario nearly doubled in the second half of the simulation time. In AWF, the structural diversity of forest stands within the multifunctional scenario fluctuates between 1 and 1.2 (Fig. 7 a). However, the development of biodiversity in the set-aside scenario was quite similar to the one in the multifunctional scenario in the first 70 years. Biodiversity in the set-aside scenario decreased subsequently. In the wood production scenario, the stands had lower biodiversity than in the other scenarios throughout the entire simulation time. Comparing the case study areas, it is apparent that AWF had a little higher biodiversity than LSN.

3.3. Tradeoffs and synergies

According to the approach mentioned in the "Result data preparation" section, we derive the following indications for trade-offs between the ecosystem service and biodiversity indicators. The production scenario in AWF has the lowest diversity and groundwater recharge as well as the highest sustainable carbon sequestration and productivity compared to both other scenarios (Fig. 8 a). The multifunctional scenario has the highest diversity and groundwater recharge as well as the lowest productivity. In LSN, under multifunctional management, biodiversity and groundwater recharge are highest and current carbon sequestration is lowest (Fig. 8 b). The production scenario has the highest productivity but the lowest diversity. The set-aside scenario has the lowest sustainable carbon sequestration and the highest current carbon sequestration as well as the lowest groundwater recharge on average over the 100 year simulation time in both case study areas (Fig. 9).

Thus in both case study areas, the simulations reveal a trade-off between productivity and biodiversity and between current carbon sequestration and sustainable carbon sequestration. Furthermore, the results reveal a synergy between groundwater recharge and biodiversity and there are indications for a synergy between productivity and carbon sequestration (independent of whether the carbon is stored in a stock of cumulative emission savings or in forest and product stocks).

4. Discussion

4.1. Relevance for adaptive forest management

Implementation of ecosystem services in landscape-related forest management and certification is a major concern (Savilaakso and Guariguata, 2017) due to their assessment on the basis of forest management actions. One challenge therefore is to define forest ecosystem characteristics that enable evaluation of forest ecosystem service output. Within that scope, Felipe-Lucia et al. (2018) have underpinned the importance of multiple forest attributes for revealing the drivers of ecosystem service provision. A second challenge is to locally implement management actions that promote the development of a desirable forest ecosystem. Our study reveals tradeoffs and synergies between key ecosystem services in order to support adaptive modern forest management that aims at comprehensive ecosystem service provision. To

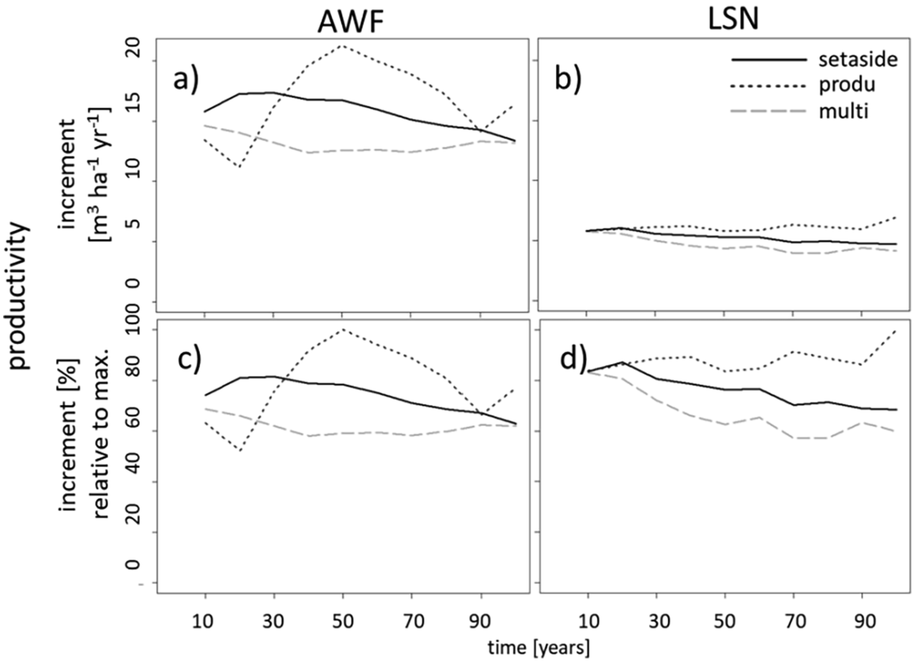

Fig. 7. Productivity as volume increment (see Section 2.4) in $m^3\ ha^{-1}\ yr^{-1}$ over simulation time (a, b), and productivity after it was standardized through division by the productivity's simulated maximum per case study area (c, d); all figures show the development to be dependent on the forest management scenario (see Section 2.3): multi-functional (multi), wood production (produ), or set-aside; LSN and AWF designate the case study area (see Section 2.1).

that end, we simulated three scenarios that range from set-aside through multifunctional forest management to maximization of wood production. These realistic scenarios have revealed considerable flexibility of adaptive forest management in the control of forest development and its related ecosystem service provision within a range of extreme adjustment options.

Forest owners can locate their own management principle within that scenario range. Based on the scenarios' comprehensive outcomes, they might re-evaluate their achievement of objectives. Politicians and decision makers who are in practice responsible for forest ecosystem service provisions on the level of the landscape may therefore establish a discussion with forest owners for implementation of modified management pathways. The results show the development of the indicators on the assumption that the whole forest area is managed by the same management. Furthermore, the silvicultural settings were adjusted according to the extremes of the politically relevant options. As a consequence, we reveal strong contrasts in the results. Political actors can use this information to implement their desired provision of ecosystem services and biodiversity. Such a participatory adaptation based on scientific knowledge might on the one hand achieve this by modification of the extreme silvicultural settings used in the scenarios. On the other hand, they can use the results to influence the stakeholders in the implementation of a mixture of the scenarios on forest area shares that lead to the desired provision of ecosystem services and biodiversity on the aggregated forest landscape level.

Another approach for adaptive ecosystem service management is a modification of the simulated management pathways that aim at tradeoff reduction. For example, a modified scenario might differ from the original one in the tree species being selected for forest conversion. Considering not only broadleaved tree species for converting forests in the multifunctional scenario, but also fostering currently rare conifers like Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) and silver fir (*Abies alba* L.) will probably diminish the synergy between groundwater recharge and diversity. However, this alternative selection of species will likely reduce the tradeoff between productivity and diversity. More productive broadleaved tree species like red oak (*Quercus rubra* L.) and black locust (*Robinia pseudoacacia* L.) could additionally retain groundwater recharge quantity.

For political justification of management scenarios and furthermore for their implementation, knowledge about the specific advantages of each management pathway is indispensable. For example, the wood production scenario reveals a particularly high potential of maximum increment in both case study areas. Sustainable forest management with the goal of maximum wood production can generate around 20% higher production than multifunctional forest management. Accordingly, private forest owners who implement multifunctional management accept elimination of potential additional income. In the case of the state forest, the voluntary waiver of financial interest might even provoke public criticism. Hessenmöller et al. (2018) also found reduced volume increment in uneven-aged beech stands in Germany resulting from continuous cover forestry. They found 30% lower volume growth compared to age class forests (Hessenmöller et al., 2018). Public forests would therefore have to justify why there is no

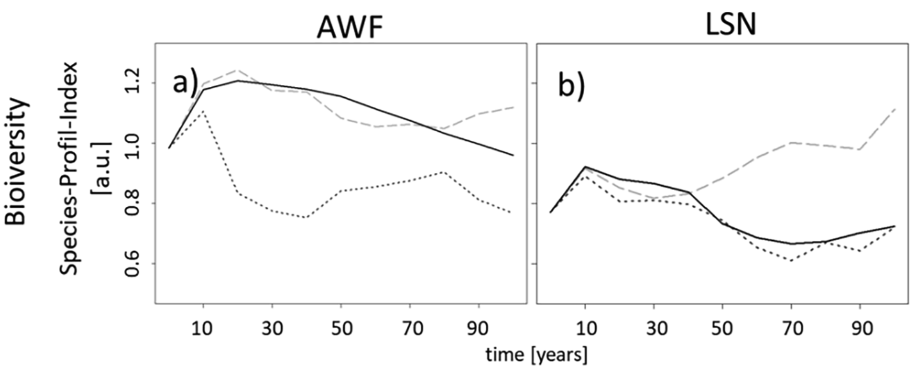

Fig. 8. Development of the Species-Profile-Index biodiversity indicator (see Section 2.4) over the simulation time (a, b); all figures show the development as dependent on the forest management scenario (see Section 2.3): multifunctional (multi), wood production (produ), or set-aside; LSN and AWF designate the case study area (see Section 2.1).

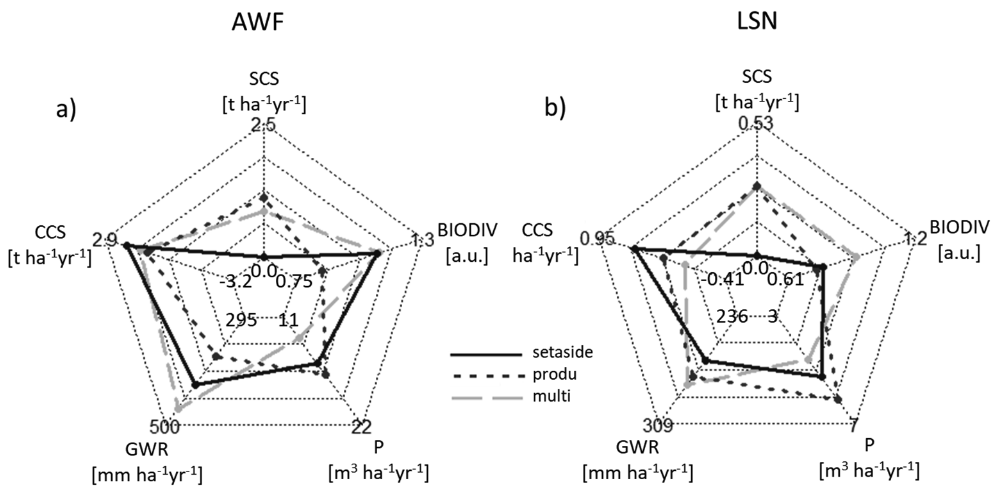

Fig. 9. Average provision of ecosystem services over the whole simulation period per case study area (see Section 2.4) and simulation scenario; these services are current carbon sequestration (*CCS*), sustainable carbon sequestration (*SCS*), biodiversity (*BIODIV*), groundwater recharge (*GWR*) and productivity (*P*); the middle of each radar chart corresponds to the minimum value per service indicator observed within any scenario or time step, while the outermost value close to the axis label corresponds to the service indicator's maximum value; LSN and AWF designate the case study area (see Section 2.1); multifunctional (multi), wood production (produ), or set-aside scenario (see Section 2.3).

maximization of wood production. However, the results of this study clearly indicate that multifunctional forest management can maintain essential public interest objectives in both case study areas over the next 100 years. The conversion of the forest leads to higher shares of broadleaves. Current literature underpins a higher level of multiple ecosystem services in forests with more tree species (Gamfeldt et al., 2013). However, multifunctional management considering biodiversity maximization on the stand level scale only can reduce biodiversity (Heinrichs et al., 2019). According to Hinrichs, multifunctional management that mixes tree species on an aggregated forest landscape scale facilitates biodiversity instead. Private forest managers may thus justify their management with an aim on wood production, because they can essentially contribute to the raw material supply when others neglect this aspect. Furthermore, state forestry may prove its contribution to public welfare through non-commercial services.

4.2. Differences between sites in ecosystem service provisioning

The study at hand underpins the work of Bösch et al. (2018) who stated that provision of forest ecosystem services is site-specific, but also depends on the forest owners' objectives. Accordingly, all landscape stock characteristics and ecosystem service provisions are higher in AWF when compared to LSN. However, the absolute effect of forest management on ecosystem services is notably related to each site's potential for ecosystem service provisions. On the one hand, there is a basic difference in the magnitude of ecosystem service provisions between both sites. On the other hand, the absolute influence of management depends on the site and forest management can only partially balance the site-related difference of ecosystem service provision. Still, the effect of management on a service being considered has to be related to the site's potential for provision of that service in order to assess the local benefit of management decisions. For example, groundwater recharge is fundamentally different in both case study areas. In LSN, it is markedly lower due to site differences, which are in particular precipitation, soil and current forest stock characteristics. In that region of less favorable recharge conditions, a change in groundwater recharge is much more valuable compared to a change of the same value in AWF. Moreover, due to the dependence of that fundamental ecosystem service on structure and species composition, management might in the long term support groundwater recharge through an increase of the share of broadleaved species.

There are also differences between both locations in regard to the volume stock development in both case study areas and when comparing each counterpart of the management scenarios. The development of the forest attribute volume stock is certainly a significant driver for ecosystem service development. In AWF, multifunctional forestry can afford a much higher equilibrium of forest stock than in LSN. This is due to the respective growth potential of the site and the main tree species. In absolute terms, the higher equilibrium of volume stock can be afforded on the site with higher growth potential and shade tolerant tree species. If compared to the maximum volume stock potential, higher equilibrium volume stock can be afforded on the site with lower growth potential and light demanding tree species. However, the average stock in the wood production scenario is very similar in both case study areas. This is probably due to the fact that the volume stock having the maximum growth at both locations does not depend on the site-specific maximum volume stock potential.

4.3. Development depends on initial forest landscape attributes

Beyond each case study area's specific site conditions, initial state of forest structure and species composition also had an essential influence on long term ecosystem service provision. Forests as relatively inert systems depended on their type of establishment and earlier development even 100 years ago. The tree species proportions and especially the area distribution of diameter classes had significant influence on respective development. This is evident from the following results in both case studies.

A region-specific diameter class distribution in LSN that was generally marked by a high frequency of small diameters caused periodic variations of volume stock in the production-oriented management pathway. The increase of volume stock in the first 50 years of the production-oriented scenario was due to the small proportion of mature stocks being cut. In turn, after 50 to 60 simulation years, large area proportions became mature for harvest. They gave rise to a notably high harvest intensity that rapidly reduced the volume stock to its initial value. The significant delay of harvest and volume stock reduction in the multifunctional management scenario was due to the notably later onset of cutting in that scenario.

In AWF, significant area proportions of harvestable diameter classes led to a notable volume stock reduction already at the start of simulation. That reduction was particularly rapid in the production-oriented scenario and led to a strong long-term fluctuation of volume stock at wide amplitude. The volume stock reduction of the multifunctional scenario in AWF, when compared to that in LSN, is only small and its effect on the long-term fluctuation of volume stock is moderate. This is likely due to the region-specific forest conversion towards higher shares of European beech that – unlike in LSN, where conversion towards higher shares of oak occurred – does not require large gaps in the canopy.

The management influence on groundwater recharge seems to be low in LSN. However, this is due to the progress of forest conversion. Due to the area distribution of diameter classes, the conversion of the forest is only possible in the later simulation process, because harvesting premature stands would mean monetary losses. The broadleaf's share therefore rises later and only then does groundwater recharge increase. In the next 100 years, the influence of management is therefore low, assuming that monetary losses are to be avoided.

4.4. Simulation without calamity incidences

Since there is a clear link between climate change and extreme weather events (Eckstein et al., 2018), the lack of windthrow and insect calamities in the model requires an assessment of its possible biasing effect on the scenario results. The set-aside scenario in particular requires an estimation of the range of calamity effects, because absence of interventions is suspected to increase the risk of calamities (Päätalo, 2000; Stephens and Moghaddas, 2005; Hanewinkel et al., 2011). The set-asides in both AWF and LSN accumulated high amounts of dead wood (fuel for wildfire (Donato et al., 2016)). They were marked by large diameters of spruce resp. pine trees, and lower structural diversity (higher risk for bark beetle and windthrow (Knoke et al., 2008)). Tree species diversity also decreased due to the prevailing dominance of old conifer trees within ageing and increasingly monolayered mature stands. Calamity events in contrast would temporarily and drastically lower volume stocks and thereby create structural diversity. For example, beetles, windthrow and fire may lead to calamity-induced dynamics and accordingly a change of the forest landscape. A possible change to a more structural and tree species rich forest landscape is therefore possible in the simulation period. Primeval forest dynamics are however known to have taken place during the course of centuries. While set-asides will thus be marked by clear dominance of conifers in the absence of calamities in the next 100 years, multifunctional management will maintain an increase of broadleaved species' shares.

Modifying diversity, broadleaved species' shares and volume stock through simulation of extreme events would also affect other ecosystem services particularly in the set-aside scenarios. For example, a higher share of deciduous tree species will foster groundwater recharge (Schwaiger et al., 2018) and lower C-sequestration (Höllerl and Bork, 2013). The simulation of damage events is however subject to fluctuations and strong uncertainty. Neglecting calamities, in contrast, provides more reliable results in case of a well-defined and transparent presumption. These outcomes underpin the potentials of forest management and of the case study areas. They provide a stable knowledge base for a discussion of the possible deflection of each scenario result through a yet unknown rate of future calamities.

4.5. Necessity of the overall carbon balance consideration

For evaluation of the functionality of carbon sequestration as related to the management scenario, it is very important that the applied carbon storage simulation model considers all carbon flows and stocks within the forest ecosystem. The forest ecosystem can provide a carbon sink in two different ways. On the one hand, it can simply store carbon in standing stock biomass and wood products. On the other hand, it can substitute non-wood products that are regularly being produced under high emissions of carbon. Storing carbon and wood products in the forest biomass is only effective until storage is saturated. In contrast, due to emission savings, a constant annual positive balance can be achieved. This is certainly only true if substitution happens in reality. Higher availability of wood products in addition to the use of fossil material must therefore not lead to an enlargement of energy sectors and the construction industry by the additional use of wood. Furthermore, this is only true if we assume availability and use of fossil materials and thus the existing possibility of substitution. Most crucial in terms of substitution is indeed reduction of the use of concrete and steel for instance in building houses, or of gas and coal in thermal and electric energy production (Gustavsson et al., 2006; Smyth et al., 2017).

In the set-aside scenario, carbon is fixed by increasing the storages. In both harvest scenarios, carbon is primarily fixed by substitution. In the study at hand, more carbon was bound over 100 years in the set-aside scenarios due to an increase in stock volume than in scenarios of active management due to substitution. Thus, for sequestration of as much carbon as possible over 100 years, increasing volume stocks are best. Increasing storages is therefore efficient, but limited in time and associated with decreased ecological stability and increased economic risk through calamities, which would consequently prevent high standing forest carbon stocks. In contrast, lower volume stocks resulting from harvests, with additional subsequent substitution of fossils, make the forest a sustainable carbon sink.

5. Conclusion

Our simulation study points out differences in ecosystem service and biodiversity indicators in two different sites in central Europe. The study shows the development of ecosystem service and biodiversity indicators in dependence of the initial state of forest stand attributes, and furthermore on forest management. The assessment of quantitative changes of the indicators over the next 100 years shows how different forest management practices affect the indicators in the longer term. Results from the LSN case study area are representative for less productive Central European sites. Results from AWF are representative for highly productive Central European sites. Accordingly, the simulated management scenarios individualize common management orientations specifically for both very different sites. The leeway for designing management pathways and for steering ecosystem service provision over the long term becomes clearer, if exactly one of each management pathways is applied to the whole landscape using accentuated forest management settings consequently idealized for one single forest management orientation. The study quantifies the difference in the magnitude of ecosystem service provision between both case study areas and sites, and consequently reveals that forest management can only partially balance that site-related variation within forest management options conceivable for stakeholders in the case study area. It points to increasing diversity and groundwater recharge at the expense of productivity in both case study areas. In addition, a synergy between productivity (as volume increment) and carbon sequestration is demonstrated and it is revealed that the strategy of increasing volume stocks is highly efficient for short-term carbon sequestration. However, it has been clearly demonstrated that only forest use makes the forest a sustainable carbon sink. The study's results are thus particularly useful for adaptive forest management to mitigate trade-offs or politically justify compensation for monetary losses through multifunctional management.

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Disclaimer

Responsibility for the information and views set out in this article/publication lies entirely with the authors.

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Appendix

See Tables A.1–A.4

Table A.1
Characteristics of the stand types in Lieberose-Schlaubetal-Neuzelle (LSN).

Owner type	Main tree species	Secondary tree species	Diameter class
Brandenburg			21–30 cm
Brandenburg			31–40 cm
Brandenburg			41–50 cm
Brandenburg			51–60 cm
Brandenburg			11–20 cm
Brandenburg	spruce		21–30 cm
Brandenburg	pine		21–30 cm
Brandenburg	pine		31–40 cm
Brandenburg	pine		41–50 cm
Brandenburg	pine	beech	51–60 cm
Brandenburg	pine	oak	31–40 cm
Brandenburg	pine	other broadleaved	11–20 cm
Brandenburg	pine	other broadleaved	21–30 cm
Brandenburg	pine	other broadleaved	31–40 cm
Brandenburg	beech	pine	41–50 cm
Brandenburg	beech	oak	51–60 cm
Brandenburg	oak	spruce	51–60 cm
Brandenburg	oak	pine	41–50 cm
Brandenburg	valuable hardwoods	pine	11–20 cm
Brandenburg	valuable hardwoods	pine	21–30 cm
Brandenburg			0–10 cm
Germany	pine	spruce	31–40 cm
Germany	other broadleaved	pine	31–40 cm
Germany			31–40 cm
Germany			41–50 cm
Municipal			41–50 cm
Municipal	pine		31–40 cm
Municipal	pine	oak	41–50 cm
Municipal	pine	other broadleaved	31–40 cm
Municipal	pine	other broadleaved	41–50 cm
Municipal	oak		41–50 cm
Municipal	oak	beech	31–40 cm
Municipal	other broadleaved	oak	31–40 cm
Municipal			11–20 cm
Municipal			21–30 cm
Municipal			31–40 cm
Private small			41–50 cm
Private small	pine		31–40 cm
Private small	pine		41–50 cm
Private small	pine	oak	41–50 cm
Private small	pine	other broadleaved	41–50 cm
Private small	oak		41–50 cm
Private small	other broadleaved	pine	31–40 cm
Private small	valuable hardwoods	pine	41–50 cm
Private small			0–10 cm
Private small			11–20 cm
Private small			21–30 cm
Private small			31–40 cm
Private small			51–60 cm
Private large			41–50 cm
Private large			> 60 cm
Private large			51–60 cm
Private large	spruce		21–30 cm
Private large	pine		21–30 cm
Private large	pine		31–40 cm
Private large	pine	spruce	31–40 cm
Private large	pine	spruce	41–50 cm
Private large	pine	oak	31–40 cm
Private large	pine	other broadleaved	21–30 cm
Private large	oak		51–60 cm
Private large	other broadleaved	pine	11–20 cm
Private large	other broadleaved	pine	31–40 cm
Private large			0–10 cm
Private large			11–20 cm
Private large			21–30 cm
Private large			31–40 cm

Table A.2
Characteristics of the stand types in Augsburg Western Forests (AWF).

Owner type	Main tree species	Secondary tree species	Diameter class
Municipal			11–20 cm
Municipal			21–30 cm
Municipal			31–40 cm
Municipal			> 60 cm
Municipal	spruce		21–30 cm
Municipal	spruce		31–40 cm
Municipal	spruce		41–50 cm
Municipal	spruce	beech	51–60 cm
Municipal	valuable hardwoods	spruce	51–60 cm
Municipal			0–10 cm
Municipal			41–50 cm
Municipal			51–60 cm
Bavaria_Ott			0–10 cm
Bavaria_Ott			11–20 cm
Bavaria_Ott			21–30 cm
Bavaria_Ott			31–40 cm
Bavaria_Ott			41–50 cm
Bavaria_Ott			51–60 cm
Bavaria_Ott			> 60 cm
Bavaria_Ott	spruce		0–10 cm
Bavaria_Ott	spruce		11–20 cm
Bavaria_Ott	spruce		21–30 cm
Bavaria_Ott	spruce		31–40 cm
Bavaria_Ott	spruce		41–50 cm
Bavaria_Ott	spruce		51–60 cm
Bavaria_Ott	spruce		> 60 cm
Bavaria_Ott	spruce	pine	31–40 cm
Bavaria_Ott	spruce	pine	0–10 cm
Bavaria_Ott	spruce	pine	11–20 cm
Bavaria_Ott	spruce	pine	21–30 cm
Bavaria_Ott	spruce	pine	41–50 cm
Bavaria_Ott	spruce	pine	51–60 cm
Bavaria_Ott	spruce	pine	> 60 cm
Bavaria_Ott	spruce	beech	0–10 cm
Bavaria_Ott	spruce	beech	11–20 cm
Bavaria_Ott	spruce	beech	21–30 cm
Bavaria_Ott	spruce	beech	31–40 cm
Bavaria_Ott	spruce	beech	41–50 cm
Bavaria_Ott	spruce	beech	51–60 cm
Bavaria_Ott	spruce	beech	> 60 cm
Bavaria_Ott	spruce	oak	11–20 cm
Bavaria_Ott	spruce	oak	21–30 cm
Bavaria_Ott	spruce	oak	31–40 cm
Bavaria_Ott	spruce	oak	41–50 cm
Bavaria_Ott	spruce	oak	> 60 cm
Bavaria_Ott	spruce	other broadleaved	0–10 cm
Bavaria_Ott	spruce	other broadleaved	11–20 cm
Bavaria_Ott	spruce	other broadleaved	21–30 cm
Bavaria_Ott	spruce	other broadleaved	31–40 cm
Bavaria_Ott	spruce	other broadleaved	41–50 cm
Bavaria_Ott	spruce	other broadleaved	51–60 cm
Bavaria_Ott	spruce	valuable hardwoods	21–30 cm
Bavaria_Ott	spruce	valuable hardwoods	31–40 cm
Bavaria_Ott	spruce	valuable hardwoods	41–50 cm
Bavaria_Ott	spruce	valuable hardwoods	51–60 cm
Bavaria_Ott	spruce	other conifers	21–30 cm
Bavaria_Ott	spruce	other conifers	31–40 cm
Bavaria_Ott	pine		31–40 cm
Bavaria_Ott	pine		41–50 cm
Bavaria_Ott	pine		51–60 cm
Bavaria_Ott	pine		> 60 cm
Bavaria_Ott	pine	spruce	21–30 cm
Bavaria_Ott	pine	spruce	31–40 cm
Bavaria_Ott	pine	spruce	41–50 cm
Bavaria_Ott	pine	spruce	51–60 cm
Bavaria_Ott	pine	spruce	> 60 cm
Bavaria_Ott	pine	beech	21–30 cm
Bavaria_Ott	pine	beech	31–40 cm
Bavaria_Ott	pine	beech	41–50 cm
Bavaria_Ott	pine	beech	51–60 cm
Bavaria_Ott	pine	oak	0–10 cm
Bavaria_Ott	pine	oak	31–40 cm
Bavaria_Ott	pine	oak	41–50 cm
Bavaria_Ott	pine	valuable hardwoods	41–50 cm

(*continued on next page*)

Table A.2 (*continued*)

Owner type	Main tree species	Secondary tree species	Diameter class
Bavaria_Ott	pine	valuable hardwoods	> 60 cm
Bavaria_Ott	beech		11–20 cm
Bavaria_Ott	beech		31–40 cm
Bavaria_Ott	beech		41–50 cm
Bavaria_Ott	beech		51–60 cm
Bavaria_Ott	beech		0–10 cm
Bavaria_Ott	beech		21–30 cm
Bavaria_Ott	beech	spruce	0–10 cm
Bavaria_Ott	beech	spruce	11–20 cm
Bavaria_Ott	beech	spruce	21–30 cm
Bavaria_Ott	beech	spruce	31–40 cm
Bavaria_Ott	beech	spruce	41–50 cm
Bavaria_Ott	beech	spruce	51–60 cm
Bavaria_Ott	beech	spruce	> 60 cm
Bavaria_Ott	beech	pine	0–10 cm
Bavaria_Ott	beech	pine	21–30 cm
Bavaria_Ott	beech	pine	31–40 cm
Bavaria_Ott	beech	pine	41–50 cm
Bavaria_Ott	beech	pine	51–60 cm
Bavaria_Ott	beech	pine	> 60 cm
Bavaria_Ott	beech	oak	21–30 cm
Bavaria_Ott	beech	oak	31–40 cm
Bavaria_Ott	beech	oak	41–50 cm
Bavaria_Ott	beech	oak	> 60 cm
Bavaria_Ott	beech	other broadleaved	0–10 cm
Bavaria_Ott	beech	other broadleaved	11–20 cm
Bavaria_Ott	beech	other broadleaved	21–30 cm
Bavaria_Ott	beech	other broadleaved	31–40 cm
Bavaria_Ott	beech	valuable hardwoods	0–10 cm
Bavaria_Ott	beech	valuable hardwoods	11–20 cm
Bavaria_Ott	beech	valuable hardwoods	21–30 cm
Bavaria_Ott	beech	valuable hardwoods	31–40 cm
Bavaria_Ott	beech	valuable hardwoods	41–50 cm
Bavaria_Ott	beech	valuable hardwoods	51–60 cm
Bavaria_Ott	oak		11–20 cm
Bavaria_Ott	oak		41–50 cm
Bavaria_Ott	oak		51–60 cm
Bavaria_Ott	oak		0–10 cm
Bavaria_Ott	oak		21–30 cm
Bavaria_Ott	oak		31–40 cm
Bavaria_Ott	oak	spruce	21–30 cm
Bavaria_Ott	oak	spruce	31–40 cm
Bavaria_Ott	oak	pine	31–40 cm
Bavaria_Ott	oak	beech	0–10 cm
Bavaria_Ott	oak	beech	11–20 cm
Bavaria_Ott	oak	beech	21–30 cm
Bavaria_Ott	oak	beech	31–40 cm
Bavaria_Ott	oak	beech	41–50 cm
Bavaria_Ott	oak	beech	51–60 cm
Bavaria_Ott	oak	beech	> 60 cm
Bavaria_Ott	oak	other broadleaved	11–20 cm
Bavaria_Ott	oak	other broadleaved	21–30 cm
Bavaria_Ott	oak	valuable hardwoods	21–30 cm
Bavaria_Ott	oak	valuable hardwoods	31–40 cm
Bavaria_Ott	oak	valuable hardwoods	> 60 cm
Bavaria_Ott	other broadleaved		11–20 cm
Bavaria_Ott	other broadleaved	spruce	21–30 cm
Bavaria_Ott	other broadleaved	spruce	31–40 cm
Bavaria_Ott	other broadleaved	spruce	41–50 cm
Bavaria_Ott	other broadleaved	pine	11–20 cm
Bavaria_Ott	other broadleaved	pine	31–40 cm
Bavaria_Ott	other broadleaved	beech	11–20 cm
Bavaria_Ott	other broadleaved	beech	21–30 cm
Bavaria_Ott	other broadleaved	beech	31–40 cm
Bavaria_Ott	other broadleaved	oak	21–30 cm
Bavaria_Ott	other broadleaved	valuable hardwoods	11–20 cm
Bavaria_Ott	valuable hardwoods		31–40 cm
Bavaria_Ott	valuable hardwoods		41–50 cm
Bavaria_Ott	valuable hardwoods		51–60 cm
Bavaria_Ott	valuable hardwoods		0–10 cm
Bavaria_Ott	valuable hardwoods	spruce	0–10 cm
Bavaria_Ott	valuable hardwoods	spruce	31–40 cm
Bavaria_Ott	valuable hardwoods	spruce	41–50 cm
Bavaria_Ott	valuable hardwoods	pine	21–30 cm
Bavaria_Ott	valuable hardwoods	pine	31–40 cm

(*continued on next page*)

Table A.2 (*continued*)

Owner type	Main tree species	Secondary tree species	Diameter class
Bavaria_Ott	valuable hardwoods	beech	11–20 cm
Bavaria_Ott	valuable hardwoods	beech	21–30 cm
Bavaria_Ott	valuable hardwoods	beech	31–40 cm
Bavaria_Ott	valuable hardwoods	beech	41–50 cm
Bavaria_Ott	valuable hardwoods	oak	0–10 cm
Bavaria_Ott	valuable hardwoods	oak	31–40 cm
Bavaria_Ott	valuable hardwoods	other broadleaved	0–10 cm
Bavaria_Ott	other conifers	spruce	31–40 cm
Bavaria_Ott	other conifers	beech	21–30 cm
Bavaria_Ott	other conifers	beech	41–50 cm
Bavaria_Zus			0–10 cm
Bavaria_Zus			21–30 cm
Bavaria_Zus			31–40 cm
Bavaria_Zus			41–50 cm
Bavaria_Zus			51–60 cm
Bavaria_Zus			> 60 cm
Bavaria_Zus	spruce		21–30 cm
Bavaria_Zus	spruce		31–40 cm
Bavaria_Zus	spruce		41–50 cm
Bavaria_Zus	spruce		51–60 cm
Bavaria_Zus	spruce		> 60 cm
Bavaria_Zus	spruce	pine	31–40 cm
Bavaria_Zus	spruce	pine	41–50 cm
Bavaria_Zus	spruce	beech	0–10 cm
Bavaria_Zus	spruce	beech	11–20 cm
Bavaria_Zus	spruce	beech	21–30 cm
Bavaria_Zus	spruce	beech	31–40 cm
Bavaria_Zus	spruce	beech	41–50 cm
Bavaria_Zus	spruce	beech	51–60 cm
Bavaria_Zus	spruce	beech	> 60 cm
Bavaria_Zus	spruce	oak	21–30 cm
Bavaria_Zus	spruce	oak	41–50 cm
Bavaria_Zus	spruce	other broadleaved	0–10 cm
Bavaria_Zus	spruce	valuable hardwoods	21–30 cm
Bavaria_Zus	spruce	valuable hardwoods	31–40 cm
Bavaria_Zus	spruce	valuable hardwoods	41–50 cm
Bavaria_Zus	pine		31–40 cm
Bavaria_Zus	pine	spruce	31–40 cm
Bavaria_Zus	pine	spruce	41–50 cm
Bavaria_Zus	pine	beech	31–40 cm
Bavaria_Zus	pine	beech	51–60 cm
Bavaria_Zus	pine	valuable hardwoods	41–50 cm
Bavaria_Zus	beech		21–30 cm
Bavaria_Zus	beech		31–40 cm
Bavaria_Zus	beech		41–50 cm
Bavaria_Zus	beech		51–60 cm
Bavaria_Zus	beech		11–20 cm
Bavaria_Zus	beech	spruce	31–40 cm
Bavaria_Zus	beech	spruce	41–50 cm
Bavaria_Zus	beech	spruce	51–60 cm
Bavaria_Zus	beech	spruce	> 60 cm
Bavaria_Zus	beech	pine	51–60 cm
Bavaria_Zus	beech	pine	> 60 cm
Bavaria_Zus	beech	oak	21–30 cm
Bavaria_Zus	beech	oak	41–50 cm
Bavaria_Zus	beech	oak	> 60 cm
Bavaria_Zus	beech	other broadleaved	21–30 cm
Bavaria_Zus	beech	other broadleaved	41–50 cm
Bavaria_Zus	beech	other broadleaved	51–60 cm
Bavaria_Zus	beech	valuable hardwoods	11–20 cm
Bavaria_Zus	beech	valuable hardwoods	51–60 cm
Bavaria_Zus	oak		11–20 cm
Bavaria_Zus	oak		51–60 cm
Bavaria_Zus	oak	other broadleaved	0–10 cm
Bavaria_Zus	other broadleaved		11–20 cm
Bavaria_Zus	other broadleaved		21–30 cm
Bavaria_Zus	other broadleaved		31–40 cm
Bavaria_Zus	other broadleaved		51–60 cm
Bavaria_Zus	other broadleaved	spruce	21–30 cm
Bavaria_Zus	other broadleaved	spruce	31–40 cm
Bavaria_Zus	other broadleaved	oak	11–20 cm
Bavaria_Zus	other broadleaved	valuable hardwoods	0–10 cm
Bavaria_Zus	valuable hardwoods		21–30 cm
Bavaria_Zus	valuable hardwoods		51–60 cm
Bavaria_Zus	valuable hardwoods		> 60 cm

(*continued on next page*)

Table A.2 (*continued*)

Owner type	Main tree species	Secondary tree species	Diameter class
Bavaria_Zus	valuable hardwoods	spruce	41–50 cm
Bavaria_Zus	valuable hardwoods	beech	31–40 cm
Bavaria_Zus	valuable hardwoods	other broadleaved	21–30 cm
Bavaria_Zus			11–20 cm
Private small			41–50 cm
Private small			51–60 cm
Private small			> 60 cm
Private small			31–40 cm
Private small	spruce		31–40 cm
Private small	spruce		41–50 cm
Private small	spruce		> 60 cm
Private small	spruce	pine	31–40 cm
Private small	spruce	pine	41–50 cm
Private small	spruce	pine	51–60 cm
Private small	spruce	oak	0–10 cm
Private small	spruce	other broadleaved	11–20 cm
Private small	spruce	valuable hardwoods	21–30 cm
Private small	pine	spruce	31–40 cm
Private small	pine	spruce	41–50 cm
Private small	beech		11–20 cm
Private small	oak		41–50 cm
Private small	valuable hardwoods		41–50 cm
Private small	valuable hardwoods	spruce	11–20 cm
Private small	valuable hardwoods	spruce	41–50 cm
Private small	valuable hardwoods	beech	21–30 cm
Private small	valuable hardwoods	beech	41–50 cm
Private small			0–10 cm
Private small			11–20 cm
Private small			21–30 cm
Private large			11–20 cm
Private large	spruce		21–30 cm
Private large	spruce		31–40 cm
Private large	spruce		41–50 cm
Private large	spruce		51–60 cm
Private large	spruce	pine	21–30 cm
Private large	spruce	pine	31–40 cm
Private large	spruce	pine	41–50 cm
Private large	spruce	beech	41–50 cm
Private large	spruce	oak	41–50 cm
Private large	spruce	other broadleaved	11–20 cm
Private large	pine	spruce	41–50 cm
Private large	pine	spruce	51–60 cm
Private large	beech	spruce	31–40 cm
Private large	oak	spruce	31–40 cm
Private large	oak	beech	31–40 cm
Private large	oak	beech	51–60 cm
Private large	other broadleaved	spruce	11–20 cm
Private large			0–10 cm
Private large			21–30 cm
Private large			31–40 cm
Private large			41–50 cm
Private large			51–60 cm
Private large			> 60 cm

Table A.3
Used values for parametrization of the carbon evaluation model (Section 2.4); agb = above ground biomass component, bgb = below ground biomass component; WBP = wood based products.

Parameter	Value	Explanation
PbiomassToCarbon	0.50	Conversion factor of one mass unit biomass to one mass unit Carbon
PagbMerchWoodToAgBiomassBroadleaves	0.80	Conversion factor from above ground merchantable broadleaf wood to total above ground biomass (from m^3 into tons)
PagbMerchWoodToAgBiomassConifers	0.70	Same as above (for conifers)
PwoodDensityBroadleaves	0.58	General wood density in t/m^3
PwoodDensityConifers	0.41	General wood density in t/m^3
ProotBiomassShareBroadleaves	0.24	Share of root biomass in total broadleaved tree biomass
ProotBiomassShareConifers	0.20	Share of root biomass in total coniferous tree biomass
PstumpShareHR	0.50	Share of stumps in harvest residual volume
PstumpShareMort	0.10	Share of stumps in mortal tree volume
PhalfLifeTimeDeadRootsStumps	17.50	Half-life time (years) of deadwood in roots and stumps
PhalfLifeTimeDeadLogs	12.50	Half-life time (years) of above ground deadwood (logs)
PdeadWoodFragmentationLossFac	0.15	Relative share of additional annual deadwood loss from both pools due to fragmentation

(*continued on next page*)

Table A.3 (*continued*)

Parameter	Value	Explanation
PinitDeadWoodShare	0.06	Relative deadwood amount (related to remaining growing stock) for initialization of deadwood stocks
PprocessingLossFactor	0.50	Share of the harvested biomass (volume) getting lost during processing
PsawlogsToWBP	0.40	Share of sawn logs that is used for producing WBP (wood based products)
PsawlogsToEnergy	0.02	Share of sawn logs that is taken for energetic use
PpulpWoodToWBP	0.30	Share of pulpwood that is used for producing WBP
PpulpWoodToEnergy	0.10	Share of pulpwood that is taken for energetic use
PhresidToEnergy	0.00	Share of harvest residues (except stumps) which is taken for energetic use
PhalfLifeTimePaper	2.00	Half-life time of paper (and pulp) in years
PhalfLifeTimeSawnWood	35.00	Half-life time of sawn wood in years
PhalfLifeTimeWoodBasedProd	25.00	Half-life time of wood based products (panels)
PDisplacementEnergy	0.27	Displacement factor for energetic wood use
PDisplacementProductsWBP	0.47	Product displacement factor for wood based products
PDisplacementProductsSawnWood	0.54	Product displacement factor for sawn wood

Table A.4
Soil characteristics of the two sites in the case study areas AWF and LSN. The soil profiles are representative for each case study area. LNr = layer number, up = upper border, down = lower border, fc = field capacity, PWP = permanent welting point, PAWC = plant available water content.

Case study area	LNr	Up [cm]	Down [cm]	fc [mm/dm]	PWP [mm/dm]	PAWC [mm/dm]
AWF	1	8	0	38.4	13.4	25
	2	0	−3	50.4	28.8	21.7
	3	−3	−7	44.2	15.9	28.3
	4	−7	−30	39	13	26
	5	−30	−50	40.3	20.2	20.2
	6	−50	−75	39.4	25.1	14.4
	7	−75	−90	36.9	25.2	11.7
	8	−90	−125	36.3	21.1	15.2
LSN	1	5	0	38.4	13.4	25.0
	2	0	−5	28.4	3.2	25.2
	3	−5	−10	25.1	2.1	23.0
	4	−10	−30	20.2	1.6	18.6
	5	−30	−60	19.0	1.5	17.4
	6	−60	−90	19.3	2.3	17.0

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RESEARCH

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Forestry projections for species diversity-oriented management: an example from Central Europe

Astor Toraño Caicoya*, Peter Biber, Werner Poschenrieder, Fabian Schwaiger and Hans Pretzsch

Abstract

Introduction: Changes in socio-economy and climate are affecting the demand of wood products globally. At the same time, society requires that forest supporting structures like biodiversity are maintained and preserved while the demand for wood products is also covered. Management support systems, like forest simulation models, that are able to analyze connections as well as quantify trade-offs between forest structure management and biodiversity indicators are highly sought. However, such models are generally developed for the local plot or stand scale only and ecosystem-scale analyses are missing. In this study, we analyzed ways to interpret results from the single-tree forest simulator SILVA from the local to the ecosystem scale. We also analyzed the impacts of forest management on biodiversity using two species diversity indicators, the species profile index and the species intermingling, for scenarios adapted from the GLOBIOM model in the case study "Augsburg Western Forests", a high productive region in South-Germany. In order to evaluate diversity tendencies across the ecosystem, we applied a moving window methodology.

Results: The relevance of scale for the interpretation of management effects on species diversity was shown and clear differences between scenarios revealed. The differences between scenarios were particularly visible when comparing the two diversity indicators, especially because the species profile index focuses on vertical and horizontal information and the species intermingling focuses mainly on horizontal structures. Under a multifunctional scenario, biodiversity values could be preserved at all scales in the vertical dimension. However, under a bio-energy-oriented scenario diversity at the local scale was reduced, although at the ecosystem level, and only in the horizontal dimension, diversity remained at relatively high values.

Conclusions: With this work, we can conclude that integrative modeling, with multiple scenarios, is highly needed to support forestry decision making and towards the evolution of forest management to consider the ecosystem scale, especially when the optimization of diversity is a management priority.

Keywords: Forest model, Multifunctional forestry, Growing window, GLOBIOM

Introduction

There is an increasing consensus among ecologists and resource managers that landscape management needs to become more sustainable (Fischer et al. 2017), and despite many ongoing debates, multifunctional management needs to be improved and its ecological value promoted (Jactel et al. 2017). However, there is a growing demand of wood products making wood production, still a pivotal provisioning ecosystem service, which has a great economic importance and is the main service that allows forest enterprises to remain profitable (Hurmekoski et al. 2015; O'Brien and Bringezu 2018). Thus, the prioritization of wood production and the intensification of productive ecosystems, which often threatens to harm species diversity, need to be carefully analyzed (Fischer et al. 2017). Although not all ecosystems need to be multifunctional, it is essential to modify current forestry planning to secure both timber yield and biodiversity (Dieler et al. 2017).

Whereas in other parts of the world plantations for intensive wood production are separated from forests for conservation of biodiversity or recreation, European

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forests aim at integrating a multitude of different functions in the one area (Pretzsch 2009). In the last century, many European regions have intensified forestry and agriculture practices prioritizing the short-term economic benefits of the landowners. However, the importance of maintaining multiple ecosystem services is being increasingly recognized (Food and Agriculture Organization of the United Nations 2003), and studies and discussions about it have emerged to become one of the most pressing and relevant subjects in environmental management (Bengtsson et al. 2000).

Sustainable forest management (SFM) is nowadays widely accepted as the main goal for forest policy and practice (Barbati et al. 2014), and due to their ecological and socio-economic importance, management practices often integrate the sustainable supply of given ecosystem services (Laginha Pinto Correia et al. 2017; Paquette and Messier 2011). Management creates forest structures that at the same time influence species diversity. However, species diversity also influences multiple ecosystem services (Pretzsch et al. 2008; Pretzsch and Schütze 2009), although these intrinsically associated interactions are very complex as species diversity plays an important role at many levels of the ecosystem service production (Triviño et al. 2017). Even if research on the influence that ecosystem management has on species diversity has addressed this integration for two decades, a full integration that makes predictions relevant to practical problems is still lacking (Grimm et al. 2017). Investigating the relationship between species diversity and ecosystem, multifunctionality should enable more efficient trade-offs between forest exploitation, ecosystem functioning, and environmental conservation (Laginha Pinto Correia et al. 2017). Thus, much remains to be learned about the relationships between biodiversity and ecosystem functionality (Mori et al. 2017), and the overall importance of biodiversity for the integrated functioning of ecosystems remains unclear (Lefcheck et al. 2015).

There is a growing number of studies which quantify and study biodiversity and connect biodiversity with forest parameters. For example, Gamfeldt et al. (2013) shows that, using results from boreal and temperate production forests, the relationships between tree species richness and multiple ecosystem services are positive to all ecosystem services. Moreover, increasing biodiversity has been put forward as an important factor for risk reduction and adaptation strategies in the face of climate change (Forrester et al. 2016). Elmqvist et al. (2003) and after him Cavin et al. (2013) also claim that the maintenance of species diversity means to promote ecosystem resilience in the face of environmental change (Folke et al. 2004).

One of the main recognized indicators for biodiversity is forest age structure. The structure created by trees becomes increasingly more relevant with age, supporting additional structures created by associated individuals, thus support greater species richness, abundance, and functional species diversity (Díaz et al. 2012). Nevertheless, other variables than stand age, like tree species composition, tree size, and vertical and horizontal structures, have been documented to have a significant impact on biodiversity (Laginha Pinto Correia et al. 2017).

Meanwhile, management needs to be more flexible and use novel measures, like predictions from forest models, to face large uncertainties (Mori et al. 2017). Examples like Lämås and Eriksson (2003) and Triviño et al. (2017)) show how it is possible to increase non-timber objectives. Therefore, in theory, it is possible to optimize the trade-offs between different objectives by applying diversified forest management planning at the ecosystem level, although, maintaining simultaneously high levels of several non-timber and timber objectives.

Further complexity adds on, as proper implementation of sustainable forest management will depend on an acceptable balance between the economic, ecological, and the social ecosystem services (Corrigan and Nieuwenhuis 2016). Considering the substantial contributions of forest ecosystem services to global society (Thompson et al. 2011) and the wide biodiversity that forests support, forest sectors including stakeholders, i.e., practitioners and scientists, have significant responsibility for the integrity and sustainability of future societies (Mori et al. 2017). Identifying their preferences and perceptions in the forest sector is also very important for understanding possible sources of conflict in the context of a changing management strategy (Grilli et al. 2016) and key to fulfill the growing need to develop methods for a more integrated and adaptive governance (Mooney et al. 2005 and Palacios-Agundez et al. 2014).

However, it is yet a challenge to use, develop, and/or adapt such support tools that are able to model and account for multiple complex interactions (Triviño et al. 2017). Forest growth simulators, which are applied as standard tools for forest productivity planning, arise as essential tools in practical forestry and forest research. Yet, although they can integrate some ecosystem services beyond wood production in their systems, they just begin to be used also for multifunctional planning (Biber et al. 2015; Fahlvik et al. 2014). Thus, simulators and the software tools that investigate how multi-objective simulations under different forestry strategies affect species diversity will serve managers to plan and direct their management strategies being able to consider for the provision and maintenance of biodiversity.

Forest ecosystem management relevance and implications

Patterns and relationship analysis with respect to biodiversity may have very different interpretations depending on the considered scale. For example, biodiversity effects at large spatial scales in ecosystems are demonstrated in Oehri et al. (2017) to be at least as large as the ones reported from small-scale experimental systems. Moreover, forest biodiversity conservation can be achieved, not only considering approaches from the establishment of large ecological reserves at large scales but also through the maintenance of individual forest structures at the smallest spatial scale (Lindenmayer et al. 2006).

Johst et al. (2011) states the importance of considering both spatial and temporal landscape attributes when designing conservation measures in dynamic ecosystems. On the one hand, all services may not be maximized similarly within the landscapes across the whole sample region (Gamfeldt et al. 2013). For example, for temperate forests, SFM may need to analyze whether within-stand habitat heterogeneity may enhance biodiversity, which would help them to decide whether to apply fine-grained uneven-aged management over more traditional coarse-grained even-aged management (Schall et al. 2017). On the other hand, many ecosystems are characterized by continuous changes in habitat structure affecting spatial and temporal habitat heterogeneity, which produces habitat fragmentation and destruction (Laginha Pinto Correia et al. 2017, Snäll et al. 2015).

The biggest overall change to model across ecological levels is, according to Grimm et al. (2017), to think how ecosystem characteristics emerge from characteristics of the individuals and then aggregate them into the ecosystem level. This is highly relevant in this study, in which differences between species diversity may have very different interpretation from the local (inventory plot level) to the ecosystem.

Objectives

This work has been developed within the frame of the GreenFutureForest project (BiodivERsA: GreenFutureForest 2016) which strives for new insights for forest planning by upscaling these methods to the landscape level. Specifically, this project is a first step towards this goal. Thus, its main objective is the simulation of management scenarios that integrate objectives from different ecosystem services, typical from Central European forests under high urban pressure, and, as a consequence, study the impacts that these have on forest diversity supporting structures (Haines-Young and Potschin 2018).

Precisely, with this work we introduce the simulation of forest structures from central European forest types for assessing potential biodiversity: (1) promising species diversity indicator calculations, like the species profile index (spi) and the species intermingling, will be performed for a large case study region, Augsburg Western Forests (AWF) in Southern Germany. 2) Two realistic management scenarios, adapted from the GLOBIOM model, will be simulated: a multifunctional scenario, which focus on multiple ecosystem services and bio-energy scenario, which focus on timber production for energy use. 3) Conclusions for management will be drawn depending on the structural parameters achieved for each scenario, with special look at implications at the forest ecosystem scale.

Methods

Data

Augsburg Western Forests

The Augsburg Western Forests (AWF) case study region is located in the federal state of Bavaria, in Southern Germany, to the west of the city of Augsburg (Fig. 1). The case study region is a so called "Nature Park" (German: "Naturpark"), which is a legally defined region, where a permanent environmentally friendly land use is strived for and where recreation and/or tourism are important, especially due to the proximity to the city of Augsburg. This is a strong argument for maintaining high biodiversity. At the same time, the region is among the most productive forest regions of Germany which made artificially established Norway spruce (*Picea abies*, L.) stands the predominant forest type, while deciduous stands would be dominating without human interference. Insofar, this region is a classic example for potential conflicts between ecosystem services like timber production and supporting structures like biodiversity.

The region is characterized by a gentle hilly landscape and divided by the streams Schmutter, Neufnach, and Zusam into gently undulating plateaus and flat interfluves. It is part of the tertiary hills between the Danube and the Bavarian Alpine Foreland. Covering approximately 120,000 ha, 43% is covered by forests and 55% is a protected landscape area (see more details in Tables 1 and 2).

With a mainly oceanic climate with hints of continental in the river valleys, the average temperature during growing season (May–September) is 14–15 °C, although it presents a warmer climate at the bottom of the river valleys (Lech/Danube). The average precipitation minimum is found in the Danube valley (650 mm), while the maximum precipitation is found in the southern side of the site (900 mm). As it is common in Bavaria, the main wind direction comes from the west.

Inventory data

Two sets of inventory data were available for this study: inventory data from the Bavarian State Forest Enterprise (in German BaySF) and the Federal Forest Inventory (in German BWI) (BWI 2017; Polley 2014).

BaySF data are acquired in 2010 on a dense grid (ca. 100 × 100 m), on the forest enterprises of Zusmarshausen und Ottobeuren. Additionally, plots from

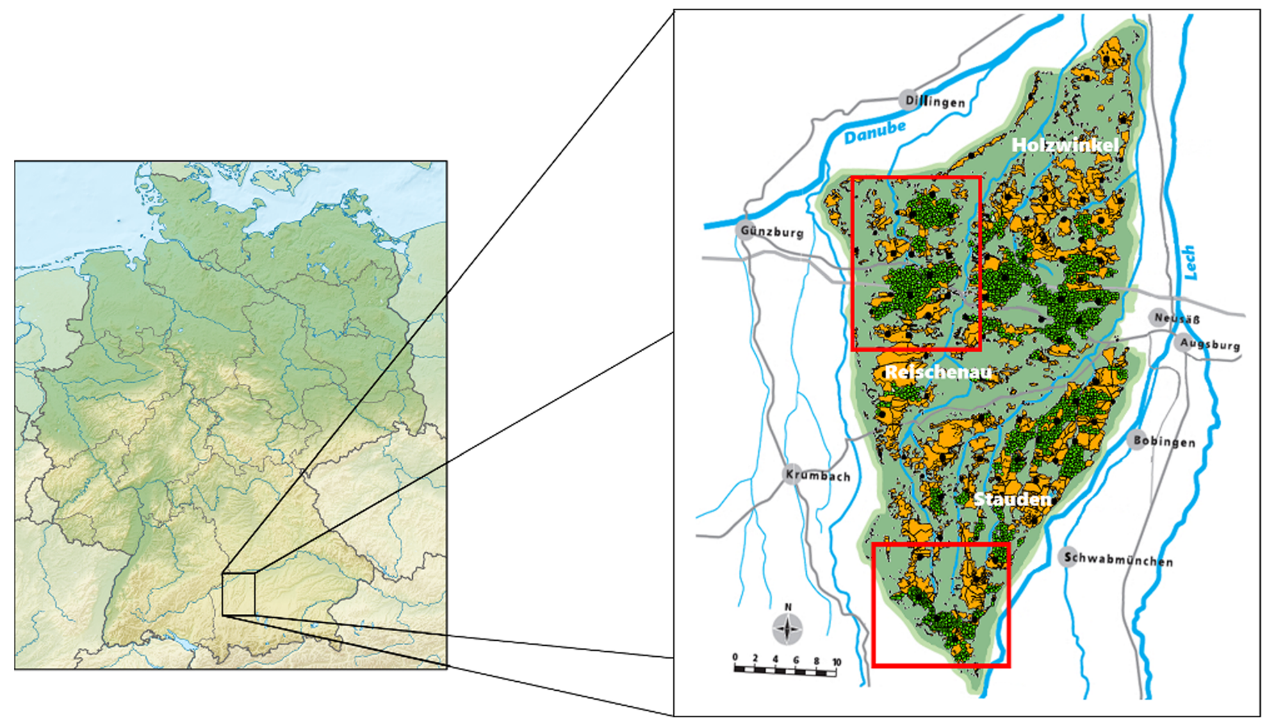

Fig. 1 Test site Augsburg Western Forests. The red squares highlight to the subset areas used for the display of results. The square on the top corresponds to Zusmarshausen and the square on the bottom to Ottobeuren. Bavarian State Forests (BaySF) inventory plots are displayed in green and Federal Forest Inventory (BWI) plots in black

the federal forest inventory (2×2 km grid) are used in order to complete information on private forest. Tree species, individual tree location, diameter at breast height (dbh), and height information are available for both data sets.

In Fig. 1 BaySF inventory plots (green) and BWI (black) are displayed. Note that the location of the BWI is due to data protection only approximate. For a detailed analysis of the results, we have selected two interesting and differentiated subsets, displayed by red squares in Fig. 1. The northern area corresponds to Zusmarshausen, which is characterized by permanent cover forestry with enhancement of species mixtures, and the southern to Ottobeuren, which is characterized by a rather conservative management with a strong focus on monocultures and timber production.

Table 1 Forest tenure distribution in the test site Augsburg Western Forests

	State forest	All tenure types
Forest area	13,100 ha	46,207 ha
Standing volume	4,800,000 m^3 (366 m^3 ha^{-1})	16,900,000 m^3 (366 m^3 ha^{-1})
Growth	10.6 m^3 ha^{-1}	10.6 m^3 ha^{-1}

Scenario simulation

Forest growth simulator SILVA

SILVA has been developed since 1989 at the Chair of Forest Growth and Yield in the Technical University of Munich. The main purpose of the simulator is to offer practitioners support in the sustainable management forests. SILVA is a single-tree-based model. It is distance-dependent (tree positions matter) and age-independent. Simulating time scale is from 5 years up to a rotation period.

SILVA (Pretzsch et al. 2008, 2002) is used as a standard planning tool on the 800,000 ha forest area owned by the federal German state of Bavaria. It is valid for the most important tree species in central Europe in pure and mixed stands. With local adjustment, e.g., based on inventory data, it can be tuned for about 80% of the central European forests. The simulator has been developed based on a unique dataset of long-term experimental plots with 150 years of history. It simulates growth of single trees in forested conditions. As the competition among trees is evaluated in a spatially explicit way, the model can cover a broad range of existing and also novel silvicultural concepts. SILVA can simulate for even-aged as well as for uneven-aged mixed and monospecific forests. A schematic representation of different SILVA simulations is displayed in Fig. 2. The model can be applied stand-wise, but also on ecosystem level where grid-based forest inventory data are available.

Table 2 Tree species distribution in the test site Augsburg Western Forests

Type	V [m^3 ha^{-1}]	Growth [m^3 ha^{-1}]	% of area
Pure conifer	428	12	55
Main conifer and > 15% deciduous	348	11	20
Pure deciduous	225	5	12
Main deciduous and > 15% conifer	278	7	9
50% conifer, 50% deciduous	341	9	3

For this study, the initial stand structure was based on inventory plot data. To limit the computational effort per simulation run to the required level, we refrained from simulation on a per-plot basis. Instead, we aggregated inventory plots of similar characteristics into virtual stands of only a few hectares to efficiently represent much larger ecosystem subunits. To that end, we formed one plot set per virtual stand. Each plot set was defined by species composition, tenure type (private vs. state or community owned forest), and a tree size class. Species composition was exclusively given as the dominating species, if that species had a stand basal area proportion of at least 90%. If, otherwise, one species had a basal area share of at least 55%, that species was defined as the dominating one. Under that precondition, if at least one from the remainder species had a basal area share of at least 20%, that species was defined as the admixed species. If, however, no such species existed, an admixed species group (coniferous, deciduous) was defined. That group was given as the one of largest basal area share if both the coniferous and the deciduous group were formed from the non-dominating species. Inventory plots that had no tree species with a basal area share of at least 55% were assigned to a dominating species group only, given as the one of larger basal area share. If both the coniferous and the broadleaved group had identical share, the broadleaved one was selected. In order to classify plots by their developmental state, we defined the tree size class as class of the average diameter weighted by species and tree height (Table 3).

Later, the single-tree results coming for each stratum were re-assigned back to the inventory plots and differentiation indexes calculated. Ecosystem results were obtained by interpolation between inventory plots.

Frame scenarios from the GLOBIOM model

The scenarios used in this study were based on Global Biosphere Management Model GLOBIOM (Forsell et al. 2016) designed by the International Institute for Applied Systems Analysis (IIASA). GLOBIOM is a global recursive dynamic partial equilibrium model of the forest and agricultural sectors, where economic optimization is based on the spatial equilibrium modeling approach (Havlík et al. 2014) and is used to analyze the competition for land use between agriculture, forestry, and bio-energy, which are the main land-based production sectors. As such, the model can provide scientists and policymakers with the means to assess, on a global basis, the rational production of food, forest fiber, and bio-energy (Nordström et al., 2016) and can be used to explore the various trade-offs and synergies around land use and ecosystem services (Forsell et al. 2016).

The scenarios proposed in GLOBIOM must be adapted to the scale of this work. For this reason, two management scenarios that follow the GLOBIOM principles were defined and adapted for the test site AWF, integrating contrasting objectives in terms of ecosystem services: (a) bio-energy and (b) multifunctional. The bio-energy scenario focuses on the production of timber. It has been defined considering a rapid development of the European bio-energy sector; forest harvests are both driven by the increasing demand for bio-energy and the foreseen increasing demand for woody materials (this increases the demand for both timber and pulpwood). The

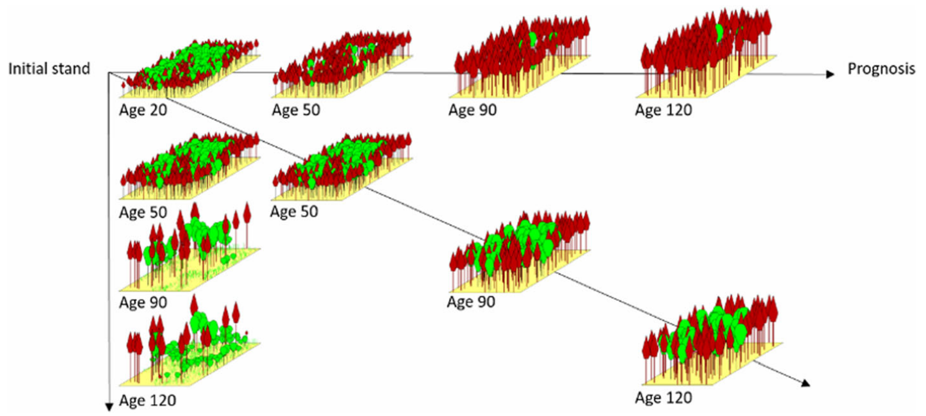

Fig. 2 Potential forest structure simulations based on management scenarios

Table 3 Classification levels of the average diameter used for stratification

Class	Diameter [cm]	% of area
4	0 to 8	3
12	8 to 15	6
20	15 to 25	15
30	25 to 35	28
40	35 to 45	28
50	45 to 55	15
60	55 and more	5

multifunctional scenario considers a harmonization of forest services, maintaining permanent forest cover and higher proportion of mixtures, especially broadleaves and richer forest structures.

We have simulated 50 years (in ten periods) for each scenario. The scenarios vary the forestry in terms of tree species composition (e.g., purely coniferous, various mixed forests, broadleaved forests) and management (e.g., even-aged forestry with clearcutting, continuous-cover forestry with selective cuttings). Thus, two distinct management regimes have been applied. For the bio-energy scenario, we applied a thinning from below starting at 12 m height and extracting max 30–60 m^3/ha in conifer stands and 35 m^3/ha in broadleaved in the pre-commercial phase, and a shelterwood concept starting at 28 m height and removing max. 500 m^3/ha in the harvesting phase. All operations are applied every 5 years. For the multifunctional scenario, we applied selective clearing starting at 12 m height, removing max 55 m^3/ha for conifers and 70 m^3/ha for broadleaves, and a target diameter felling and tending in the harvest phase starting at 30 m height, removing max 140 m^3/ha for conifers and 70–80 m^3/ha for broadleaves. All operations were also applied every 5 years. Exact details about the specific thinning and harvesting operations can be found in Tables 4 and 5.

Biodiversity: forest species diversity indicators

Spatial structure, which is the horizontal and vertical spatial arrangement of individual trees and other plants at a given point in time, determines the integrity and stability of a forest to a large extent (Pretzsch 2009). In comparison with direct quantitative measurements of biodiversity, stability, or sustainability, the use of structural parameters is advantageous as the data can be collected rapidly or already exists in forest inventory data. Moreover, current knowledge indicates that the diversity of the plant and animal species present increases with increasing structural differentiation (Pretzsch 2009). In this work, we have tested two differentiation indexes: species profile index and species intermingling index by Füldner.

Species profile index (Pretzsch 2009)

Species profile index A for species profiles (Pretzsch 1995), outlined below, is based on the Shannon and Weaver (1948) diversity index. In addition to the proportion of the species within a stand, index A takes into account the presence of these species in different height zones (Pretzsch 2009):

$$A = -\sum_{i=1}^{S}\sum_{j=1}^{Z} p_{ij} \cdot \ln\left(p_{ij}\right), \tag{1}$$

where S represents the number of species present, Z the number of height zones (three in this example), N the total number of individuals, n_{ij} the number of individuals of the species i in zone j, and p_{ij} the proportion of a species in the height zone $p_{ij} = n_{ij}/N$. The number of individuals of species i in zone j is counted. By calculating the sum of the products of the proportion of a species and the logarithmic proportion of that species for $i = 1 - S$, and for the height zones $j = 1 - Z$, one obtains an index that quantifies the overall species diversity and the vertical spatial occupancy of the species present in the forest stand.

In Fig. 3, a schematic representation for three forest conditions is shown, with potential values for a mono-layered stand of Norway spruce, a two-layered stand of Norway spruce and European beech, and multi-layered, typical mountain mix stand of Norway spruce, European beech, and silver fir.

Species intermingling index by Füldner (1996) (Pretzsch 2009)

The species intermingling index M_i by Füldner (1996) describes the spatial structure of the species mixture in a stand. Index M_i is defined as the proportion of neighbors of another species:

$$M_i = \frac{1}{n} \times \sum_{j=1}^{n} v_{ij}, \tag{2}$$

where i is the center tree, j refers to the neighboring trees j, $j = 1, \ldots, n$, and n represents the number of neighbors included in the analysis. The parameter v_{ij}:

$$v_{ij} = \begin{cases} 0, & \text{if neighbor belongs to the same species as central tree } i \\ 1, & \text{if neighbor belongs to a species differenct from central tree } i \end{cases} \tag{3}$$

is a dual discrete variable that takes the value 0 when the neighbor considered j belongs to the same species as the center tree i.

Diversity from inventory plot to ecosystem level: growing window analysis

We have estimated mean parameters for the species profile index and the species intermingling using a growing window, which is always initiated over forested area. The

Table 4 Thinning specifications, bio-energy scenario. Con. stands for coniferous and dec. for decideous

Stand type	Phase	Starts at top height [m]	Species	Operation	Frequency	Number of future crop trees	Number of competitors	Target diameter	% of targeted removed per intervention	Maximum volume removed per intervention [Vfm ha^{-1}]
Coniferous	Precommercial	12	Coniferous	Thinning from below	5					30
			Broadleaved	Thinning from below	5					30
		16	Coniferous	Thinning from below	5					60
			Broadleaved	Thinning from below	5					60
	Harvest	28	Coniferous	Shelterwood cutting over	5 5					500
				35 years						
			Broadleaved	Shelterwood cutting over	5					500
Broadleaved	Precommercial	0	Coniferous	Removal of broadleaved[a]	5	1000	All within			25
			Broadleaved	Removal of broadleaved[a]	5					25
		17	Coniferous	Thinning from below	5					35
			Broadleaved	Thinning from below	5					35
	Harvest	30	Coniferous	Shelterwood cutting over	5					500
			Broadleaved	Shelterwood cutting over	5					500
Oak	Precommercial	12	Coniferous	Thinning from above	5					25
			Broadleaved	Thinning from above	5					25
		17	Coniferous	Thinning from above	5					50
			Broadleaved	Thinning from above	5					50
	Harvest	30	Coniferous	Target diameter felling	5			30	100	500
			Broadleaved	Target diameter felling	5			30	100	500

[a]Through selection of 1000 coniferous future crop trees per hectares and removal of all competitors within 10 m radius

window has an initial size of 100 m^2, and it grows in steps of 10 m^2 until it reaches the maximum side of the ecosystem unit. In each step, we calculated the mean, the maximum, and minimum value for each resolution (scale) unit. We have repeated the procedure with six different random initial locations.

We performed this analysis in two very different areas or the test site: Zusmarshausen, in the north part, with 3447 ha, as an example of a diverse unit and Ottobeuren, in the south, with 1283 ha and under management that enhances more intensive thinning and single species (coniferous), and therefore, a priori, not as diverse as the rest of the enterprises in the test site.

We use this analysis as a proxy to the estimation of diversity in the scales equivalent to alpha (< 100 ha), betta (> 100 ha), and gamma (comparison between ecosystem units) diversity. We also show the different interpretations that species profile index and species intermingling can offer and how they can complement each other for diversity management at the ecosystem scale.

Table 5 Thinning specifications, multifunctional scenario. Con. stands for coniferous and dec. for decideous

Stand type	Phase	Starts at top height [m]	Species	Operation	Frequency	Number of future crop trees	Number of competitors	Target diameter	% of targeted removed per intervention	Maximum Volume removed per intervention [Vfm ha^{-1}]
Coniferous	Precommercial	0	Coniferous	Removal of coniferous[a]	5					25
			Broadleaved	Removal of coniferous[a]	5	1000	All within			25
							10 m radius			
		12	Coniferous	Selective clearing	5	100	1 around con			55
						150[b]	2 around dec			
			Broadleaved	Selective clearing	5	100	1 around con			55
							2 around dec			
		25	Coniferous	Selective clearing	5	150	1 around con			140
						200[b]	2 around dec			
			Broadleaved	Selective clearing	5	100	1 around con			140
							2 around dec			
	Harvest	32	Coniferous	Target diameter felling	5	100 con	1	45 con	30	80
						50 dec		65 dec		
			Broadleaved	Target diameter felling	5	100 con	1	45 con	30	80
						50 dec		65 dec		
Broadleaved	Precommercial	12	Coniferous	Selective clearing	5	100	3 around con			30
						150[b]	2 around dec			
			Broadleaved	Selective clearing	5	150	3 around con			30
							2 around dec			
		17	Coniferous	Selective clearing	5	200	1			70
						150[b]				
			Broadleaved	Selective clearing	5	100	1			70
	Harvest	30	Coniferous	Target diameter felling	5	100		45	20	80
			Broadleaved	Target diameter felling	5	50		65	20	80
						60[c]				
Oak	Precommercial	12	Coniferous	Thinning from above	5					30, 70[d]
			Broadleaved	Thinning from above	5					30, 70[d]
	Harvest	30	Coniferous	Target diameter felling	5			30	100	500
			Broadleaved	Target diameter felling	5			30	100	500
			Oak	Target diameter felling	5	60	All within	65	100	500

[a]Through selection of 1000 broadleaved future crop trees per hectares and removal of all competitors within 10 m radius
[b]Pine, larch
[c]Oak
[d]Starting from 17 m

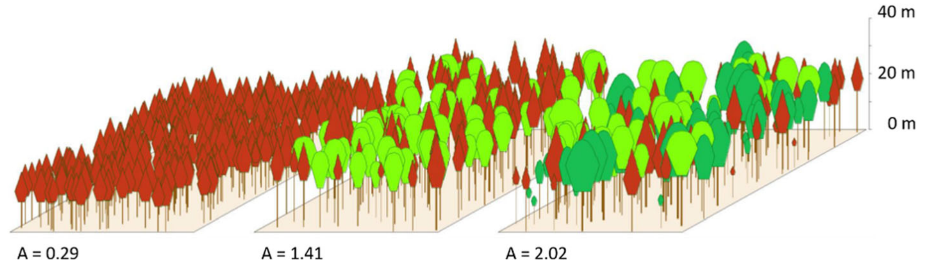

Fig. 3 Species profile index *A* for, from left to right, a mono-layerd stand of Norway spruce, a two-layered stand of Norway spruce and European beech, and multi-layered stand of Norway Spruce, European beech, and silver fir

Results

Structure differentiation indicators and scenarios

After a simulation period of 50 years and with 394 strata defined, the mean values for both species profile index and species intermingling showed distinctive trends. The mean values are shown in Fig. 4 for the species profile index and Fig. 5 for the species intermingling. In the case of species profile index, we observed how the multifunctional scenario, in blue, which during the first 15 years maintained stable levels achieving its maximum between 15 and 30 years, later became very stable again. In contrast, the bio-energy scenario, in black, showed an abrupt decrease at the beginning of the management simulation to later continuing decreasing at lower rates. In the case of species intermingling index, both scenarios decreased in comparing to the original state, being very similar for about 15 years, when for the multifunctional starts increasing. At the end of the simulation, the multifunctional scenario pointed towards a recovering trend, while bio-energy seemed to achieve a steady behavior. The different behavior between species profile index and species intermingling indicates that in the bio-energy scenario, the vertical distribution of species was affected before the horizontal.

In Figs. 4 and 5, the average trends for the entire site are shown. Depending on the scenario, distinctive effects were observed along the territory. In the following, we show these effects for a subset of the site in the area of Zusmarshausen (see subset in Fig. 3).

In Fig. 6, four maps are shown for the bio-energy scenario. During the first 15 years (three periods), species profile index values remained rather stable. However, afterwards, we observed a rapid loss of diversity. Nevertheless, local differences could be observed, as some areas remain more resilient, already visible after 15 years. This effect produced islands of diversity, and interconnectivity was reduced over the simulation periods.

In Fig. 7, analogue maps for the multifunctional scenario are also shown. In this case, the rapid decrease of the bio-energy case was not observed. The mean values over the study area remained very stable, and a tendency increasing the area with the highest values was observed. Thus, connectivity between islands of diversity was improved, especially noticeable after 30 years (Fig. 7—middle-right). The range of values from the index *A* moved from areas with values that represent monocultures and homogenous stands to values characteristic from highly diverse forest (typical from mountain-mix-forests).

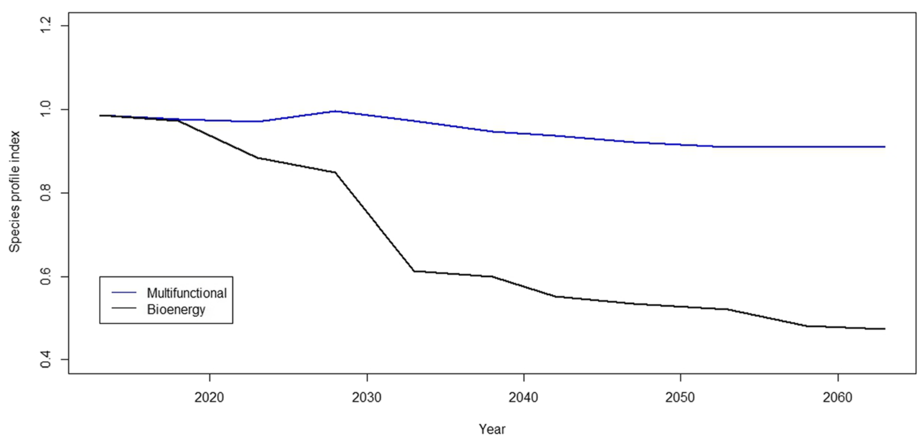

Fig. 4 Mean species profile index for the test site Augsburg Western Forests for two scenarios: multifunctional (blue) and bio-energy (black)

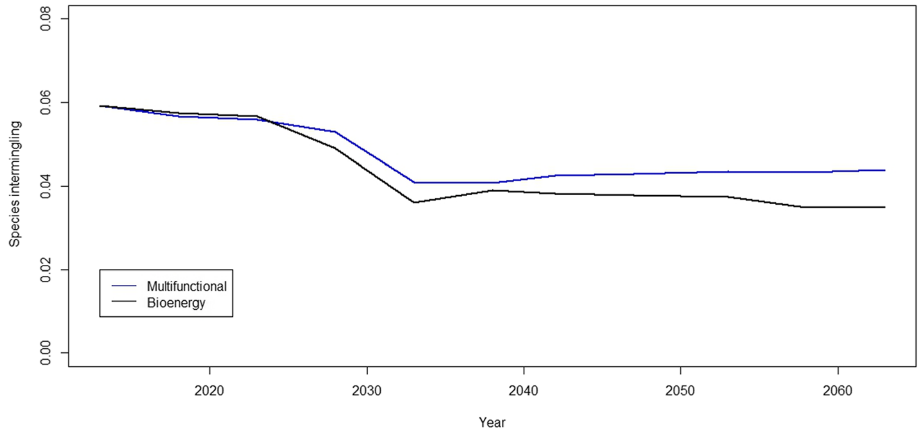

Fig. 5 Mean species intermingling for the test site Augsburg Western Forests for two scenarios: multifunctional (blue) and bio-energy (black)

At 30 years' time, it is especially noticeable that high average values (light green) were dominant. However, after this point until 50 years, average values became dominant and generally lower than at the maximum achieved at 15–30 years. This was translated into medium-high diversity values that remained stable and distributed homogenously over the site.

Results for differentiation using the species intermingling index are displayed in Figs. 8 and 9. We have observed similar patterns, as in the case of the species profile index, although with a constant decrease as expected from the mean values shown in Fig. 5.

Species intermingling index characterizes only the diversity along the horizontal dimension. Therefore, and especially in the bio-energy case, where thinnings from below are performed with higher volumes and at lower dbh, more drastic impacts in the results of this index were observed. We have also observed the same island behavior where high values remain but isolated from the rest of the area. However, this pattern was also present in the multifunctional scenario, indicating that the horizontal mixture of tree species does not change as fast as in the mixture in the vertical dimension. Particularly different is the decreasing trend present in the species intermingling in comparison with the species profile index in the multifunctional scenario. After 15 years, some areas decreased rapidly to later increase up to average values after 30 years, especially noticeable in the lower-right corner. We could also observe a general homogenization trend across the area towards average values, being slightly higher and more homogenous in the multifunctional scenario than in the bio energy scenario.

Ecosystem analysis

In Fig. 10 for Zusmarshausen and Fig. 11 for Ottobeuren, we show the influence of scale in the estimation of species differentiation parameters, for six different and randomly chosen locations/growing paths. In both figures, the variation among paths on the first steps of aggregation (small windows) was high and it was reduced rapidly. However,

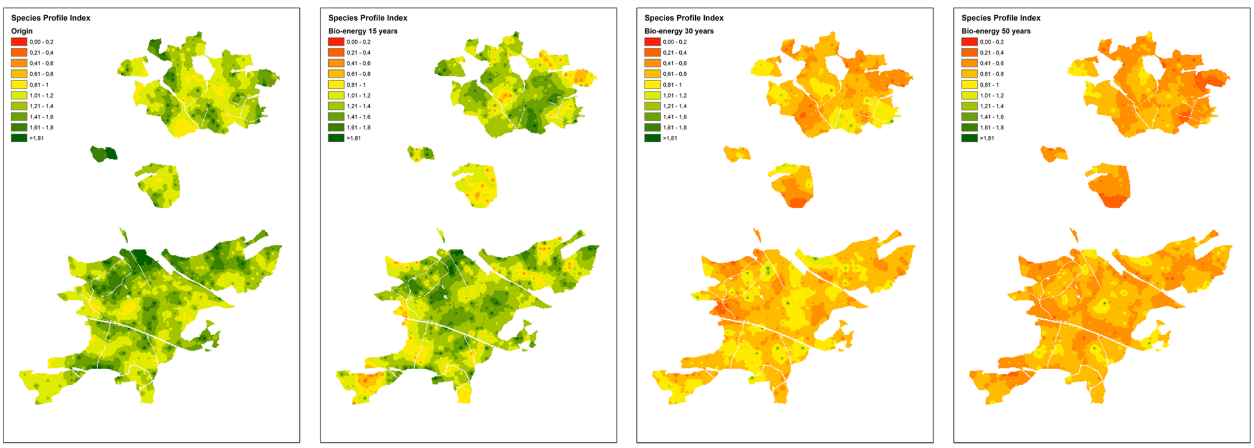

Fig. 6 Species profile index, bio-energy scenario. From left to right: present time, 15, 30, and 50 years' simulation steps

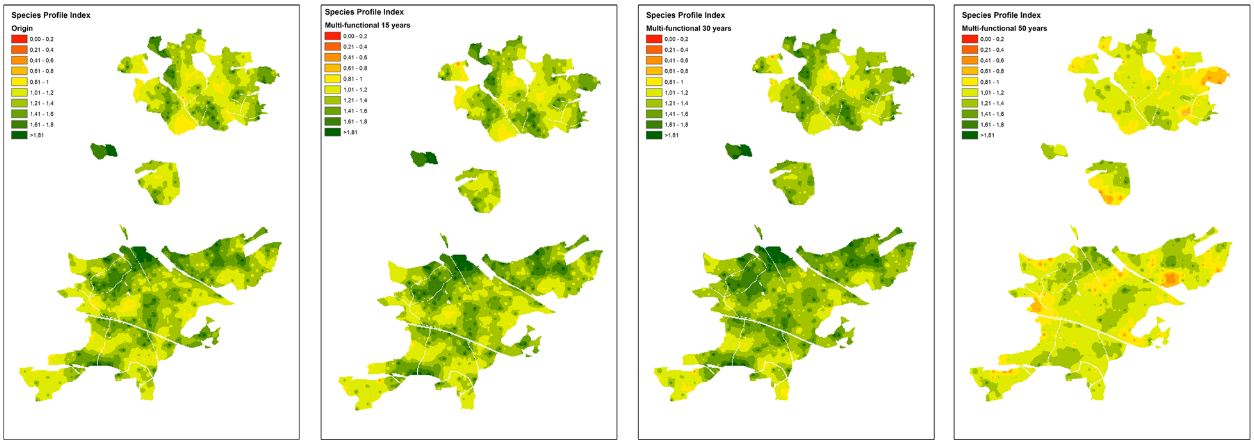

Fig. 7 Species profile index, multifunctional scenario. From left to right: present time, 15, 30, and 50 years' simulation steps

in Zusmarshausen, it remained oscillating and slowly approaching the mean at approx. 2500 ha. In Ottobeuren right after ~ 100 ha, all paths converged very close to mean. We could also observe that the final mean, as well as the maximum and minimum values, was higher in Zusmarshausen than in Ottobeuren, confirming that a higher proportion of diverse forests is present in the first site with respect to the second.

In Fig. 12, we display the results of the growing window test for the species profile index, for the multifunctional and bio-energy scenarios, after 15 and 30 years. With the first results, we could observe that, on the one hand, after three simulation periods, that is 15 years, in the multifunctional case, the mean species profile index remained constant. On the other hand, the fluctuations at mid-scales were reduced, even if the maximum values at the finest scale tended to grow. This indicated a general homogenization at this scale, with higher values at the alpha-diversity scale, but general reduction in diversity at the beta-diversity dimension. This trend was even more evident after 30 years, when the mean value even lightly increased.

In Figs. 13 and 14, we have estimated the mean and 95% confidence intervals, absolute and normalized to the mean, respectively, for the ten random paths displayed in Figs. 10 and 11. The amplitude of the confidence interval, defined by the standard error, varies depending on the widow size. For all cases, the amplitude was maximum at the smallest scale and approaches 0 at the maximum, where all paths converge at the mean of the site. However, we could observe differences along the path, which are site and variable (species profile index, or intermingling) dependent. Main differences were observed in the converging speed, which was already pointed out for Figs. 10 and 11.

In Zusmarshausen in the case of the species profile index, the amplitude of the confidence interval remained constant with just a local minimum at 450 ha, while in Ottobeuren, it decreased rapidly, already converging at 250 ha, and remaining narrow from this point on. For

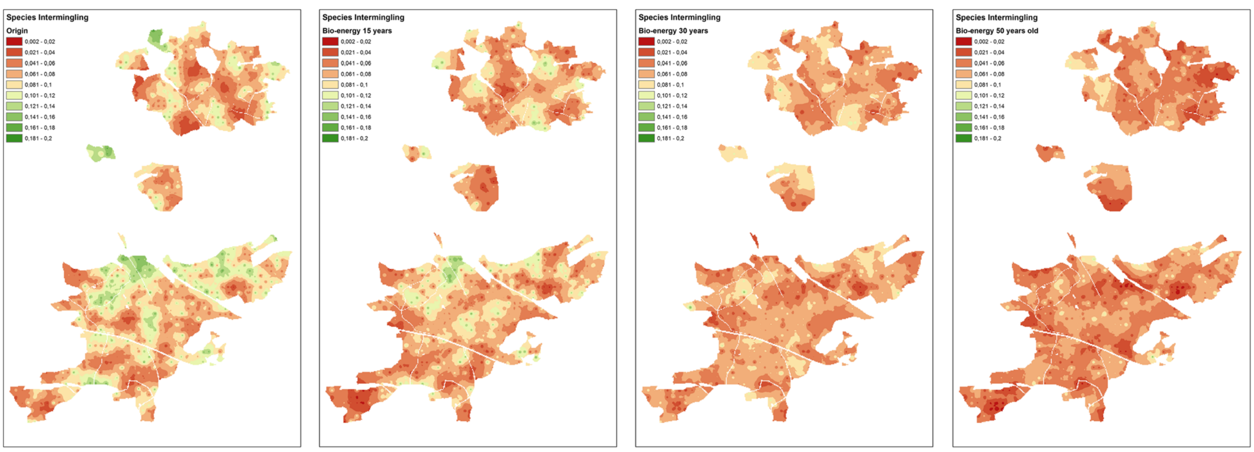

Fig. 8 Species intermingling, bio-energy scenario. From left to right: present time, 15, 30, and 50 years' simulation steps

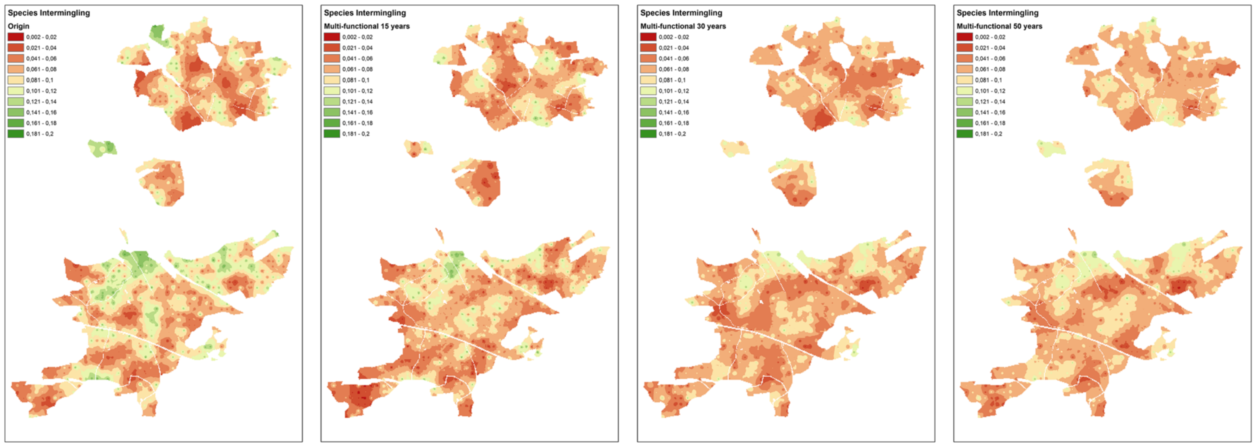

Fig. 9 Species intermingling, multifunctional scenario. From left to right: present time, 15, 30, and 50 years' simulation steps

the species intermingling, tendencies were similar but with some clearer differences. In Zusmarshausen, the amplitude was wider and decreases monotonically until it converged at the maximum site's area, while in Ottobeuren also a minimum at ~ 250 ha was achieved converging at 1000 ~ha, similarly to the species profile index.

Discussion

Single-tree model structure sensitivity

Being SILVA a single-tree simulator where the location of all trees is known, structural indicators, which are closely connected with biodiversity, could be estimated directly from the tree lists that the simulator provides at each simulation period, in contrast to simulators based on stand variables, which would have strong limitations analyzing structures that arise from the distribution of individual trees. Moreover, as the connection to biodiversity is based on the distribution at tree level of certain species or their intermingling, this issue arises to be of capital importance for works like this, and for their future development (Crookston and Dixon 2005; Fahlvik et al. 2014; Pretzsch et al. 2017, 2002).

The results we obtained showed the high sensitivity that simulations show across the entire ecosystem unit. Even if up to now each inventory point is not simulated independently but in strata, the evolution of each region on the ecosystem scale remained very realistic, as we will describe in detail in the following. Moreover, the management practices which we chose for each scenario followed the expected trends. It is especially significant to observe the very different diversity scenarios after

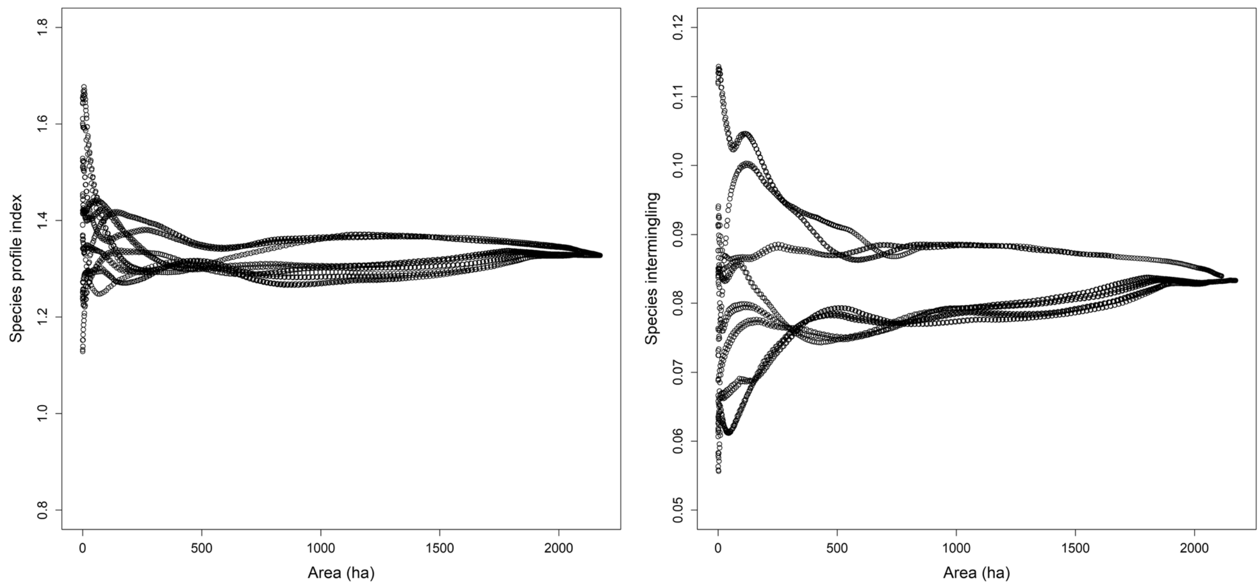

Fig. 10 Growing window scale experiment for Zusmarshausen. On the left: species profile index. On the right: species intermingling

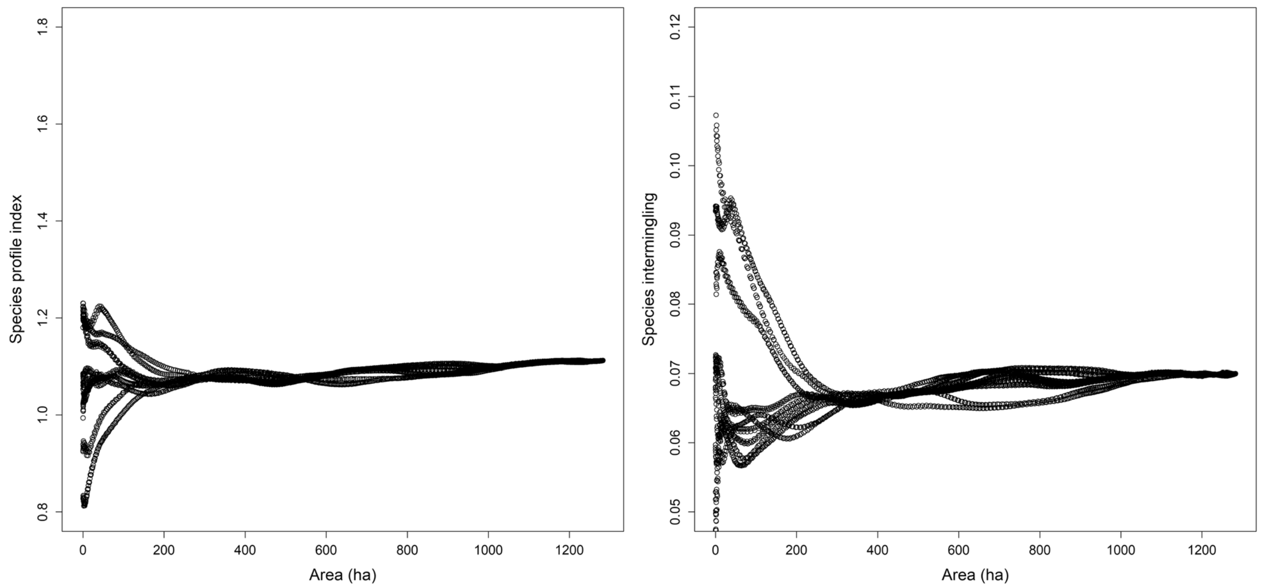

Fig. 11 Growing window scale experiment for Ottobeuren. On the left: species profile index. On the right: species intermingling

50 years, with two different management strategies showed results that point towards completely different types of forest ecosystems depending on management decisions. Implications of forest management, which depend on bio-socio-economic and climate trends, on biodiversity could be distinguished based of forest growth simulations.

Impact of treatments in the output diversity indicators

Management operations, i.e., thinning types, intensity, and final felling, were selected in order to achieve specific timber demands for each scenario. Thus, diversity outputs were a consequence and not the management goals. This means that our intention was to evaluate the impacts that management decisions within the society/economic environment have on biodiversity, seen as an ecosystem service. Our results showed the tendencies for a first simulation period of 50 years, even if the intention of this study is to show long-term simulations, which was enough to allow an assessment of management effects on habitat biodiversity (Griesser and Lagerberg 2012).

The consequences derived from the selected management scenario were clearly appreciated. On the one hand, the selection of conservative management principles, summarized in the multifunctional scenario, clearly maintained sustainable high levels of diversity, which generally improved in the entire site, increasing connectivity areas with the highest levels of species profile index. This scenario favored especially late rotations with future tree thinning strategies, promoting species mixture and multi-layering while preserving constant standing volumes. On the other hand, management strategies that change the forest ecosystem into a bio-energy-dominated scenario, the diversity represented by the species profile index, dropped drastically. The management guidelines (see the "Frame scenarios from the GLOBIOM model" section) stablished in this scenario generated a quick timber extraction during the first periods, accelerating the decrease in diversity. Forest structure became more homogenous as big diameters were not needed and conifers, which grow faster and are more suitable for the industry, were promoted.

The combination of added information from the two differentiation indices was complementary and useful in the interpretation of two effects: the increase of species mixture along the vertical dimension, which is mainly contained in the species profile index, together with the clustering of species within stands or local management units, which is explained by the species intermingling index. We observed that even if clear differences in species profile index were observed between management types, both had similar effects in the species intermingling, i.e., reducing the diversity on the horizontal dimensions. Thus, we concluded that management had a homogenizing effect in the mid-term, but probably experimenting higher changes at the rotation period.

Diversity across the ecosystem, management implications

The ecosystem dimension showed the importance of scale for diversity estimations. Being AWF a known, a priori, diverse forest ecosystem, we could test this fact, even when comparing the areas of Zusmarshausen and Ottobeuren, two differently managed enterprises in the AWF region.

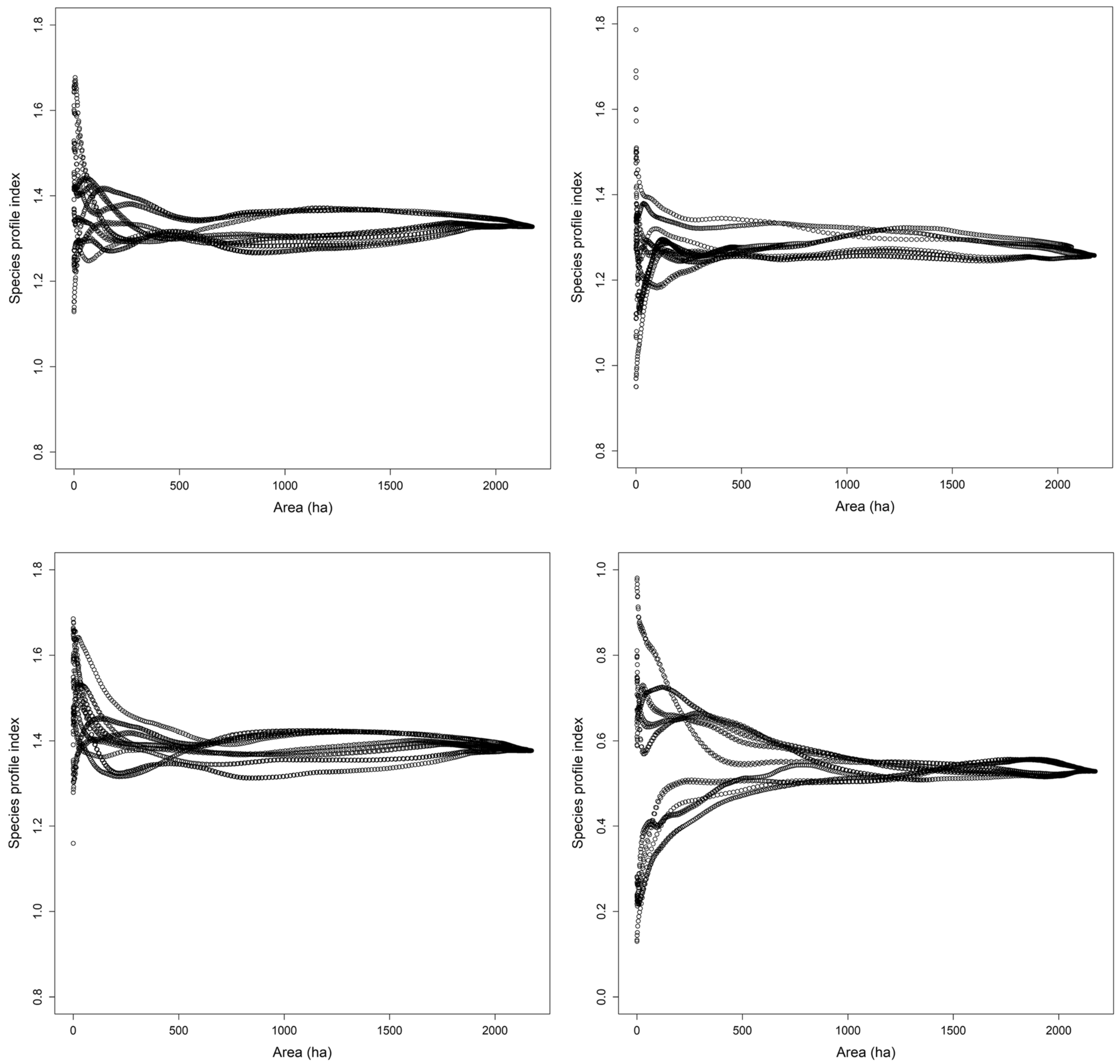

Fig. 12 Growing window plots for species profile index in Zusmarshausen for the multifunctional scenario on the left panel and bio-energy on the right. On the top after 15 years and on the bottom after 30 years. Note that for a better display, the *y*-axis for the bio-energy scenario after 30 years is set from 0 to 1

The growing window test offered information depending on how fast the window tended to meet the mean value of the test site. Thus, depending on the diversity patterns, this fluctuated across different scales until it reached the mean. Using values from the species profile index, in the very diverse experiment on Zusmarshausen, the window needed to reach almost 2000 ha to meet the mean, and the difference between the finer and middle-coarse scales doubled the mean, indicating not only the presence of highly diverse stands but also a high within-stand heterogeneity. A direct comparison with the test performed in Ottobeuren showed, in fact, the opposite diverse conclusions, as it was expected from the management objectives. In this case, the average in the window approached quickly the mean for all the paths, indicating a very low diverseness at all considered scales, that is a lower mean and homogenous conditions all over the site.

Taking into account the different information contained in the two differentiation indices, we could also study the horizontal and vertical structure influence in the distribution of diversity across scales, until the ecosystem level. The clearest difference was identified when comparing the starting situation for Zusmarshausen and

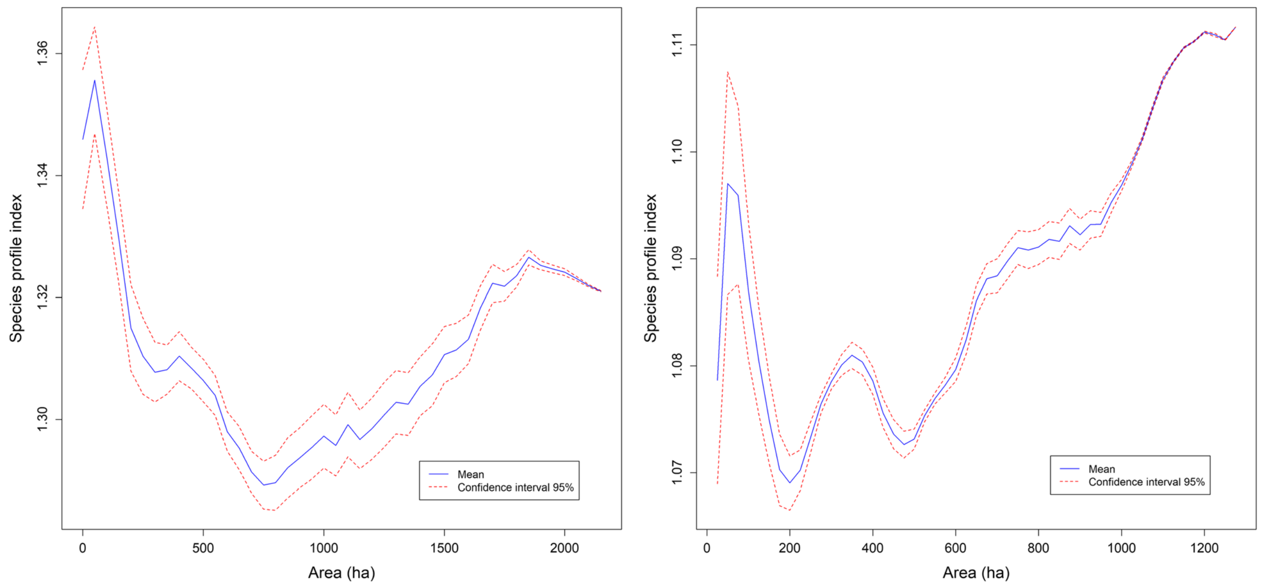

Fig. 13 Confidence interval plots for Zusmarshausen (on the left) and Ottobeuren (on the right). The blue line represents the mean value for each of the growing window paths (see Fig. 12) and the dashed lines the 95% confidence interval boundaries

Ottobeuren. As expected, the species profile index converged much faster in Ottobeuren than in Zusmarshausen, indicating that the vertical distribution of species is relatively homogenous in the entire site. However, in the case of species intermingling, Ottobeuren remained more diverse in middle scales (400–1000 ha) indicating a patchy distribution of stands with different degrees of intermingling. This also leads, as Gamfeldt et al. (2013) points out, to a potential reduction of services in areas like Ottobeuren, because different species correlate with different services, indicating that monoculture practices will lead to reduced provision of at least some of the services.

Management that enhances uneven aged structures like in the multifunctional scenario showed that high levels of medium to high local (alpha) diversity could be maintained steadily over time, and therefore keeping structures with great potential for high local floristic diversity (Bagnaresi et al. 2002). We also observed that in the multifunctional scenario, the management units at a broader scale became more homogenous due to uneven-aged management (Schall et al. 2017). This happened at the same time that the overall indices increase, especially the species profile index, which indicated an increase of diversity in the local (alpha) scale. Thus, it means that all species were distributed and mixed rather equally over the territory. Several studies (Brokaw and Busing 2000; Griesser and Lagerberg 2012; Schall et al. 2017; Whittaker et al. 2001) suggest that some species require heterogeneous niches for survival, so such kind of management applied to the entire territory, even if improving the average diversity, may harm the distribution of such species. Therefore, forest management needs to evolve into considering the ecosystem level, when the optimization of both local and regional diversities is a priority.

Accordingly, with the results obtained by the simulations carried out during this work, we can suggest that forest management strategies that take place at the local level need to be adapted after relatively short periods of time, in order to improve heterogeneity at the ecosystem scale. Thus, not only paying attention to average levels but also to spatial features that are only visible at these scales, this analysis also showed that we could potentially derive a proxy to alpha and beta diversity, at least relatively between sites and bidirectional (micro to macro scale and vice versa) (Whittaker et al. 2001). The variation in the mean species profile index offered an accurate description of the sites heterogeneity. Moreover, conclusions regarding gamma diversity could be accessed by comparing the two test sites, if these are considered as different communities.

Conclusions

Single-tree forest simulators like SILVA can simulate spatially explicit forest structures. The simulation of scenarios based on realistic management strategies, which depend on the socioeconomic conditions, is a great tool to analyze in an objective manner potential effects and future trends in species diversity. Future models may directly connect the diversity-oriented forest structure parameters, like the ones shown in this work, and variables generated from (meta)population models. At this respect, we demonstrated that differentiation indexes

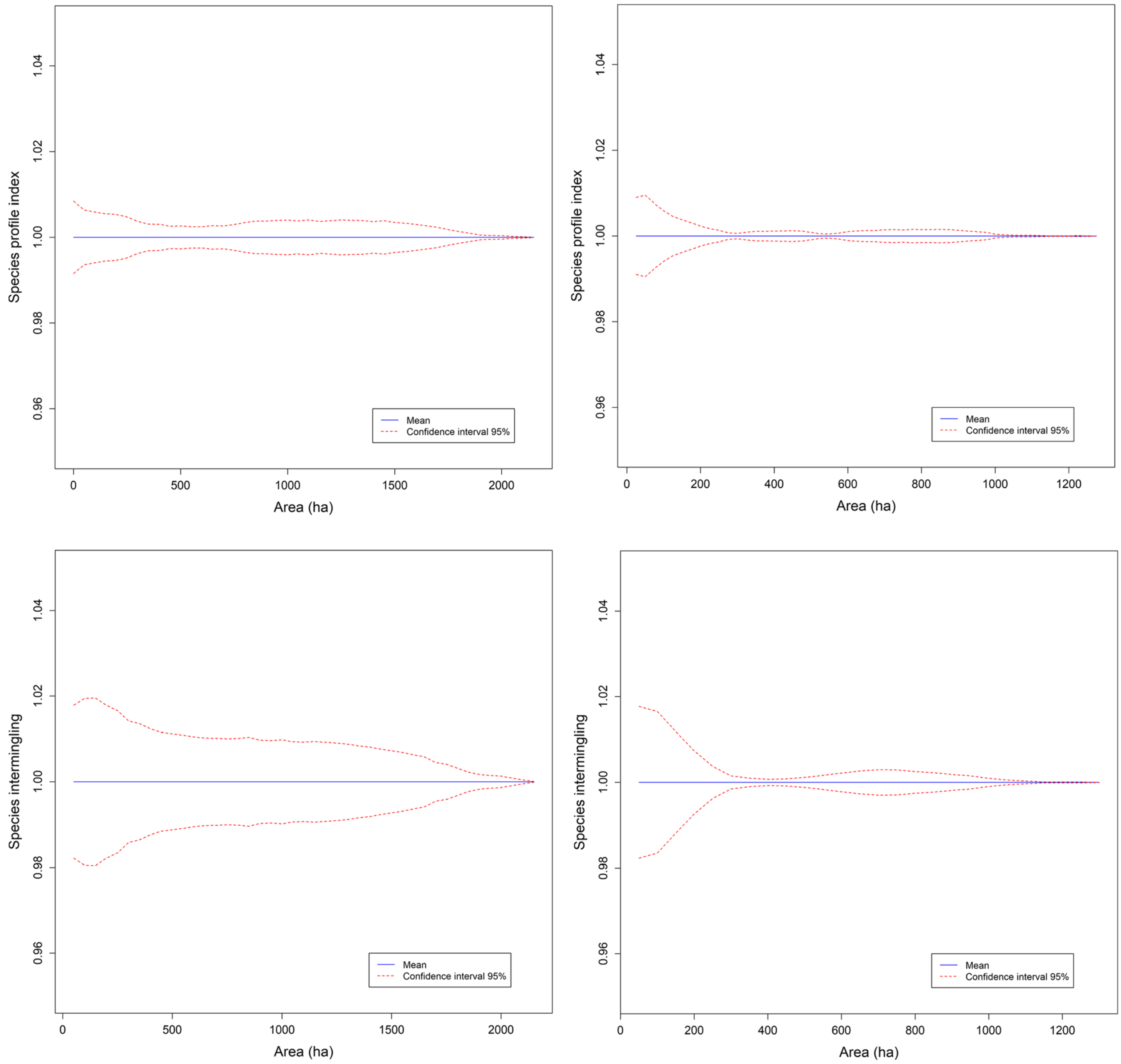

Fig. 14 Confidence interval plots for Zusmarshausen (on the left) and Ottobeuren (on the right), normalized to the mean, for the species profile index on the top panel and species intermingling on the bottom panel. The blue line represents the mean value for each of the growing window paths (see Fig. 12) and the dashed lines the 95% confidence interval boundaries

can be effectively modeled and have the potential to complement traditionally used forest structure parameters (basal area, dominant height, volume, etc.).

In very productive regions under high urban pressure like AWF, where multiple ecosystem services are sought, management scenarios had a high impact on the structure and biodiversity and, therefore, silvicultural practices can strongly influence the overall diversity distribution, not only for the local scale, but also for the whole forest ecosystem. The scale implications were translated into ecosystem patterns that can be not only a consequence but also an objective of management. Therefore, analyses like the ones shown in this study can be integrated in the implementation of management plans with diversity of objectives at the ecosystem scale and which can respond to different interests within the stakeholder community.

Conclusions and recommendations, based on Central European forest data, towards sustainable ecosystem management can be drawn at each simulation step (for any time horizon), and, therefore, management strategies can be adapted to achieve sustainable and profitable exploitation of forests and contribute to the study of diversity patterns within the forest component of the landscape. Furthermore, an understanding of future changes in forest structures resulting from defined management objectives can also contribute to satisfy changing societal needs, which demand multiple use of forests while sustaining species diversity.

Abbreviations

AWF: Augsburg Western Forests; BaySF: Bavarian State Forests (in German: Bayerische Staatsforsten); BWI: German National Forest Inventory (in German: Bundeswaldinventur); Con: Coniferous; Dbh: Diameter at breast height; Dec: Deciduous; SFM: Sustainable forest management

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Availability of data and materials

Please contact the author for data requests.

Authors' contributions

ATC developed the concept presented in this manuscript. He performed the simulations and calculated the differentiation indices, the generation of the 2D maps, as well as the multi-scale analysis. He was responsible for writing this manuscript. WP supported the algorithm development and the simulation and the adaptation of the management scenarios from GLOBIOM. FS implemented the management guidelines for the adapted scenarios. PB and HP contributed to writing the manuscript and the interpretation of the results. All authors read and approved the final manuscript.

Ethics approval and consent to participate

Not applicable.

Competing interests

The authors declare that they have no competing interests.

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