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# MULTIFUNCTIONAL MOUNTAIN FOREST MANAGEMENT - EVALUATING ADAPTATION SCENARIOS UNDER CLIMATE CHANGE IN A FOREST LANDSCAPE IN THE EASTERN ALPS

Dissertation
to obtain the doctoral degree (Dr. nat. techn.) at the University of Natural Resources and Life Sciences, Vienna

Submitted by
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*"I like to think of the tree itself: first the close dry sensation of being wood; then the grinding of the storm; then the slow, delicious ooze of sap. I like to think of it, too, on winter's nights standing in the empty field with all leaves close-furled, nothing tender exposed to the iron bullets of the moon, a naked mast upon an earth that goes tumbling, tumbling, all night long. The song of birds must sound very loud and strange in June; and how cold the feet of insects must feel upon it, as they make laborious progresses up the creases of the bark, or sun themselves upon the thin green awning of the leaves, and look straight in front of them with diamond-cut red eyes..... (...) One by one the fibres snap beneath the immense cold pressure of the earth, then the last storm comes and, falling, the highest branches drive deep into the ground again. Even so, life isn't done with; there are a million patient, watchful lives still for a tree, all over the world, in bedrooms, in ships, on the pavement, lining rooms, where men and women sit after tea, smoking cigarettes. It is full of peaceful thoughts, happy thoughts, this tree."*

Virginia Woolf, The Mark on the Wall

## Preface

This thesis synthesises three separate journal articles as a cumulative dissertation. A detailed description of all analysis and results can be found in the original articles, provided in the Appendix 8.1 to 8.3. Differences in the formatting of the articles are due to the requirements of the scientific journals.

Paper 1: Irauschek F, Barka I, Bugmann H, Courbaud B, Elkin C, Hlásny T, Klopcic T, Mina M, Rammer W, Lexer MJ (2020) Evaluating five forest models using multi-decadal inventory data from mountain forests. Ecological Modelling (accepted Manuscript)

Paper 2: Irauschek F, Rammer W, Lexer MJ (2017) Can current management maintain forest landscape multifunctionality in the Eastern Alps in Austria under climate change? Regional Environmental Change 17:33–48. http://doi.org/10.1007/s10113-015-0908-9

Paper 3: Irauschek F, Rammer W, Lexer MJ (2017) Evaluating multifunctionality and adaptive capacity of mountain forest management alternatives under climate change in the Eastern Alps. European Journal of Forest Research 136:1051–1069. http://doi.org/10.1007/s10342-017-1051-6

I finalized this thesis in a time of lockdown due to the COVID-19 pandemic. A crisis that made people think about the unity and interconnectedness of people living in this world and the fragility of our existence. Nobody can tell yet how long this crisis will last. However, I am confident that it will eventually be overcome by motivated scientists, who are currently conceptualizing solutions around the world and will finally succeed. I think one of the strengths of humanity is to develop new ideas by using our imagination. We can sit on a couch, staring at the wall and travel through the life of a tree, just as Virginia Woolf wrote in her inspiring and dreamy short story "A mark on the wall".

## ACKNOWLEDGEMENTS

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To my girlfriend, Mariana, for her love and support, during all these years I was working on this thesis.

## Eidesstattliche Erklärung / Statutory Declaration

Deutsch:

Ich erkläre eidesstattlich, dass ich die Arbeit selbständig angefertigt habe. Es wurden keine anderen als die angegebenen Hilfsmittel benutzt. Die aus fremden Quellen direkt oder indirekt übernommenen Formulierungen und Gedanken sind als solche kenntlich gemacht. Diese schriftliche Arbeit wurde noch an keiner Stelle vorgelegt.

English:

I hereby declare that I am the sole author of this work. No assistance other than that which is permitted has been used. Ideas and quotes taken directly or indirectly from other sources are identified as such. This written work has not yet been submitted in any part.

Wien, am 10.3.2021

(Florian Irauschek)

# ABSTRACT

Mountain forests provide essential goods and services for our society. However, climate change impacts and changing demands question the currently practiced management strategies. Adaptive forest management aims for actions to avoid negative consequences from expected climatic conditions and take advantage of the anticipated changes in the ecosystem. The goal of this study was to assess the effects of mountain forest management strategies under climate change, with particular focus on detailed, spatially explicit management plans and analyses of effects on ecosystem services at different spatial scales.

The main methodological tool was the forest ecosystem model PICUS v1.5. To assess the validity of the model outputs, it was tested against inventory data and compared against four other ecosystem models in a case study in the Dinaric Mountains, in Slovenia. PICUS was then applied in a second case study in the Eastern Alps, Austria to evaluate current management practices and eight adaptive management alternatives. The simulations included different cutting patterns (patch-, slit- and strip-cuts), harvest intensities, artificial regeneration and sanitary management under six climate scenarios. Outcomes for the catchment were analyzed for the ecosystem services timber production, carbon sequestration, nature conservation and protection against gravitational hazards (rockfall, erosion and snow avalanche release).

Results for the model evaluation demonstrated a good performance of PICUS and other individual tree-based models in simulating complex mountain forest ecosystems under management. Results from the studies analyzing currently applied and adaptive management in the application study showed that none of the alternatives was best regarding all ecosystem services. Patch-cut regimes at low intensity level appeared as a well-suited strategy to maintain landscape multifunctionality. Disturbances by the spruce bark beetle pose a major threat to the stability of the spruce-dominated forests in the future. To strengthen the resilience of the forests, increased forest management intensities accompanied by game management activities are required to foster the establishment of other tree species. The occurring trade-offs between ecosystem services demonstrated the potential for targeted planning processes, especially for protection against gravitational hazards and nature conservation areas. A common understanding of ecological processes and the possibilities of their modification through management is vital to adapt mountain forests for the challenges from climate change and complex demands from society.

## KURZFASSUNG

Bergwälder sind naturnahe Ökosysteme die unsere Gesellschaft mit wichtigen Gütern versorgen und essentielle Funktionen erfüllen. Die derzeitigen Bewirtschaftungskonzepte werden jedoch durch den Klimawandel und sich ändernde gesellschaftliche Interessen in Frage gestellt. Die adaptive Waldbewirtschaftung zielt darauf ab Maßnahmen zu setzen, um negative Auswirkungen des Klimawandels zu verhindern und positive Effekte im Ökosystem zu nutzen.

Ziel dieser Dissertation war, die Auswirkungen von Bewirtschaftungskonzepten in Bergwäldern unter Berücksichtigung des Klimawandels zu analysieren. Dabei wurde der Fokus auf eine räumlich explizite Nutzungsplanung gesetzt und die Veränderung von Ökosystemleistungen auf verschiedenen räumlichen Skalenebenen analysiert. Methodisch basierte die Studie auf der Anwendung des Waldökosystemmodells PICUS v1.5. Um die Qualität der Simulationsergebnisse zu testen, wurde das Model in einer Studie zusammen mit vier anderen Simulationsmodellen mit historischen Waldinventurdaten in Slowenien evaluiert. In einer zweiten Studie wurde PICUS in den Ostalpen, in Österreich, im Montafon angewendet, um das aktuelle Bewirtschaftungskonzept und acht alternative Konzepte unter fünf Klimawandelszenarien zu vergleichen. Die getesteten Simulationsszenarien umfassten verschiedene Hiebsformen (Schlitzhieb, Streifenhieb und buchtig ausgeformter Lochhieb), Nutzungsintensitäten und Kunstverjüngung. Die Resultate für das Revier wurden mithilfe von Indikatoren für Holzproduktion, Kohlenstoffspeicherung, Naturschutz und Schutz vor Naturgefahren (Steinschlag, Erosion und Lawinenanbruch) verglichen.

Die Ergebnisse der Evaluierungsstudie demonstrierten, das PICUS, wie die anderen getesteten einzelbaum-basierten Modelle, konzeptionell soweit ausgereift ist, dass eine Simulation von Bewirtschaftung und Klimawandelauswirkungen in komplexen Bergwaldökosystemen mit hoher Güte möglich ist. Die Ergebnisse der Simulationsstudie zeigten, dass keine der Bewirtschaftungsalternativen alle Ökosystemfunktionen gleichzeitig gut erfüllen konnte. Die buchtige Hiebsform in niedriger Nutzungsintensität war am besten geeignet, um die derzeitige Multifunktionalität zu erhalten. In allen Simulationsszenarien wurde ein starker Anstieg von Borkenkäferkalamitäten prognostiziert, wodurch die Stabilität der fichten-dominierten Wälder bedroht wird. Um die Resilienz der Wälder zu stärken, ist eine Intensivierung der Bewirtschaftung kombiniert mit einer Reduktion der Wilddichten notwendig, um eine ausreichende Verjüngung von Mischbaumarten zu erreichen. Die teils starken Wechselwirkungen zwischen den Ökosystemleistungen zeigen die Notwendigkeit einer örtlichen Priorisierung der Zielsetzungen in der Bewirtschaftung auf, vor allem hinsichtlich der Schutzfunktion und in ökologisch sensitiven Bereichen. Es ist essentiell, dass alle beteiligten Akteure ein gemeinsames Verständnis für die in Bergwäldern ablaufenden Prozesse gewinnen, um die Wälder durch gezielte Bewirtschaftung an die Herausforderungen durch den Klimawandel und die komplexen gesellschaftlichen Anforderungen anzupassen.

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# 1 INTRODUCTION

Management of forest ecosystems located in mountainous landscapes poses many challenges for decision-makers. The most obvious feature of mountain forests is the complex topography, which causes sharp gradients of climate parameters, variable soil conditions and increased direct water runoff and erosion (Beniston 2003). Forests in European mountain ranges are usually characterized by low growth dynamics, limited accessibility, long timber extraction distance and steep terrain. Nonetheless, they provide essential goods and services for society, such as clean drinking water, protection against gravitational hazards and high-value timber (EEA 2011). European mountain forests hold a high degree of ecosystem naturalness, serve as habitat for endangered species (Nagy et al. 2003) and provide aesthetic values for local inhabitants and tourism (EEA 2011). Mountain forest landscapes are dynamic socio-ecological systems, evolving as a result of slow and fast drivers of natural processes (Thom et al. 2013) and social processes (Holling and Gunderson 2002) (Figure 1). The ecosystem service concept (MEA 2005) is well suited to study these complex systems. It includes service supply by the ecosystem on one side and demand by humankind on the other side while holding humans as an integrative part of the ecosystem. The importance of different ecosystem services can be quantified and compared in a multidimensional way, in the categories of ecological, socio-cultural and economic value (de Groot et al. 2010).

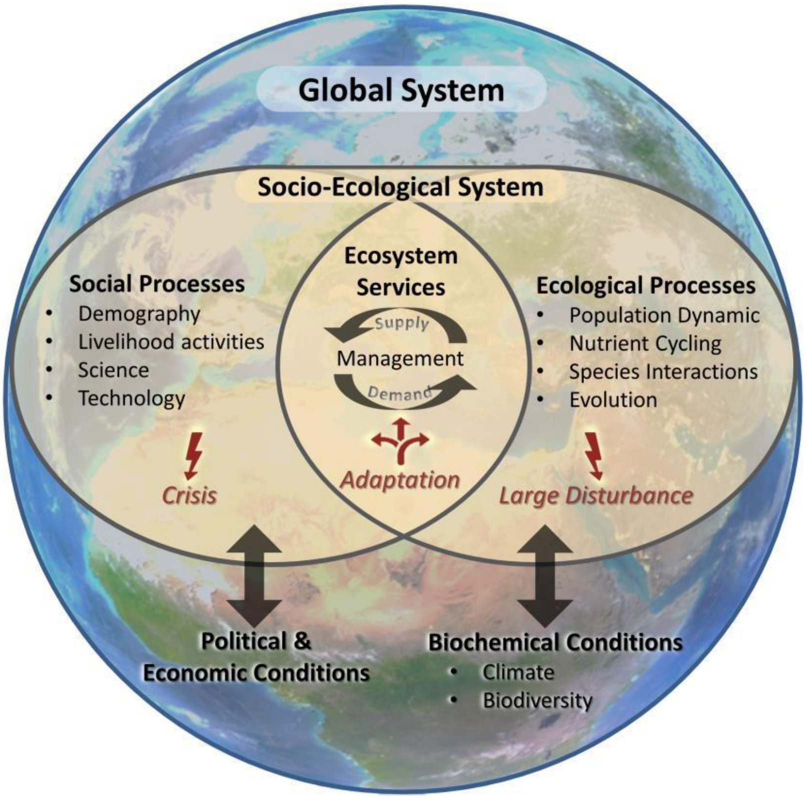


**Figure 1.** The context of the ecosystem service concept in a dynamic socio-ecological and global system (adapted from Virapongse et al. 2016).

Sustainable management of ecosystem services is defined as the human interaction with ecological processes to satisfy demands from a long-term perspective (MEA 2005). Therefore, it is essential to create a resilient socio-ecological system, capable of withstanding severe disruptions both from the ecological (disturbances) and the sociological (crisis) realms (Virapongse et al. 2016) (Figure 1). Following this dynamic system understanding, sustainability is a process and requires regular evaluation to adapt management practices (Berkes et al. 2003). Hence, sustainable forest management is based on system knowledge concerning ecological and social processes, sound knowledge on ecosystem service demand and supply, as well as tools to plan and evaluate management interventions.

European mountain forests have been managed for hundreds of years. Evolving societal demands have caused substantial changes in management objectives and restrictions, which subsequently resulted in changing forest properties over time. Forests always served as the primary source of energy and raw material for humanity, but the high demands from early modern age industries caused regional shortages of wood and large clear-cut areas. Due to the low growth rates and longevity of trees in mountain forests, these legacies are still noticeable today and contribute to the relatively uniform and overaged protective forests in the Alps (Niese 2011). Another important factor contributing to the current status of mountain forest landscapes is the locally deeply rooted tradition of intensive game management (Nussbaumer 2000; Milner et al. 2006). Subsequently, high deer densities for trophy hunting are favoring disturbance sensitive mono-species Norway spruce (*Picea abies* (L.) Karst) stands and prolong critical regeneration phases (Schodterer 2011). Many parts of Central European mountain ranges are permanently populated by humans and energy production and storage through water turbines and dams have been sharply increasing in the last century (EEA 2007). Tourism has a long history in the European Alps; starting at the end of the 18$^{th}$ century (Lauterbach 2010), it has grown into a major source of income for many regions today, with individual visitors searching for authentic alpine landscapes as well as equipped and safe accommodation facilities all year long (Wehrli et al. 2007). The societal demands for uninterrupted transportation, energy and communication infrastructure require forests in mountainous areas which provide a comprehensive high-level protective functionality against gravitational hazards such as rockfall, avalanches, soil erosion and water runoff peaks.

A global factor challenging socio-ecological systems is climate change. Studies show that warming rates are more pronounced in mountain regions (Wang et al. 2014; Pepin et al. 2015). In the Alps, mean annual temperatures have already increased by more than one degree and, in the next decades, a further increase is expected, accompanied by changes in the precipitation regime and decreasing snow cover durations (Gobiet et al. 2014). As a result, growth, mortality and regeneration of trees are affected and disturbance regimes are expected to increase in frequency and/or magnitude (Kromp-Kolb et al. 2014). Forests have adapted historically to changing environmental conditions. Still, the impacts and the speed of climate change may be beyond their biological adaptive capacity and result in severe disruptive consequences on protective and economic services. With increasing awareness and evidence of possible impacts, scientists started to propose strategies to deal with the effects of climate change (e.g., Linder 2000; Spittlehouse 2005). Adaptive forest management was introduced, as a process of "monitoring and anticipating change and undertaking actions to avoid the negative consequences or take advantage of potential benefits of those changes" (Keenan 2015).

Central European forests have a long history of multifunctional management to ensure the provisioning of manifold services and the livelihoods of people living within the wider region of mountain ranges. Recently, forest managers have started to recognize that the narrow view of meeting the strategic and operative company goals solely within traditional microeconomics is not sufficient. Within this frame, forest management is much more than extracting marketable timber and game management for favoring trophy hunting. Management activities are all sorts of possible interventions in forest structures and natural processes. Moreover, it is important to consider the explicit decision not to intervene, because natural development is evaluated as beneficial and the expenses can be spent more cost effectively elsewhere. To avoid conflicts and to secure future economic potentials, forest managers have to consider various stakeholder groups in planning and decision making processes (Vacik and Lexer 2014). While legislation, subsidies and counseling by public administration provide only a rough frame for adaptation on the local scale, managers are facing the problem of setting up operative harvesting plans and management decisions that will have long-term implications. One widely acknowledged overall goal concerning climate change impacts is to design management strategies in order to prevent large scale disturbances, which cause long-lasting disruptions of ecosystem services and high restoration costs. Homogeneous stand structures in Norway spruce-dominated forests, occurring in large areas of the montane and lower subalpine belts in the Alps, are seen as especially susceptible to large scale disturbances by storms, snow load and bark beetles (Brang et al. 2006). Stands with clustered spatial tree distribution, higher tree species and size diversity and long internal edges are seen as beneficial for various reasons: Heterogeneous structures provide breaks for disturbances, enhance stability of trees by decreasing tree height to diameter ratios and offer more suitable niches for the establishment of regeneration (Motta and Haudemand 2000). Even though these overall goals are acknowledged, forest managers have to decide where, when and how much intervention is needed to foster a horizontal and vertical structure of the forests to guarantee the required forest services (Dorren et al. 2004). Ecosystem services depend on the forest state (i.e., state variables) and the change of state over time (i.e., flow variables or ecosystem output) and emerge on different spatial scale levels. Hence, impacts caused by management have to be evaluated at several temporal and spatial levels. For example, gravitational processes operate on the slope to landscape scale (e.g., rock trajectories along the slope), while, on the other hand, silvicultural interventions to initiate regeneration need a fine-grained management resolution at the patch or even tree level (Maroschek et al. 2015). Processes with similar complexity have to be considered for many goods and services provided by mountain forest ecosystems (e.g., provision of drinking water, wildlife habitat management).

The multi-scale nature and related potential trade-offs between ecosystem services are enormous challenges for forest ecosystem management, especially under the anticipated influence of climate change (Seidl et al. 2013). Appropriate concepts and tools for decision support are thus needed to fulfill the societal demands in a long term perspective (Muys et al. 2011; Vacik and Lexer 2014). Various forest simulation models have been developed based on different theoretical concepts along with the rising computing capacity starting from the 1960s (see Fabrika and Pretzsch 2013; Shifley et al. 2017). However, until now, most models remained in the science domain and are applied by their original authors and a small group of technical experts (Vanclay 2003). Those models which are available for end-users are typically growth and yield focused models, that do not consider climate change effects and rarely include disturbance factors (e.g., BWINpro (Hansen and Nagel 2014), MOSES (Hasenauer 1994), MOTTI (Salminen and Hynynen 2001)). In applied research projects, models are inevitable tools to explore mid- to long-term implications of changing environmental conditions (Lexer et al. 2002; Hanewinkel et al. 2012; Elkin et al. 2013), different management approaches (Söderbergh and Ledermann 2003; Seidl et al. 2008b; Schelhaas et al. 2014) and disturbance regimes (Seidl et al. 2011a; Temperli et al. 2013) on forest development. They can deliver results for different scales of the ecosystem and are a prerequisite for transparent decision support systems (Mäkelä et al. 2012). The successful application of simulation models is based on proficient model development and requires detailed knowledge of initial forest states and ecosystem processes. Over the last twenty years, the rapidly growing availability of data, originating from large research networks and remote sensing sources, has stimulated many advanced modeling approaches (Fournier et al. 2000; Seidl 2017). Yet, no supermodel has appeared on the science catwalk, which covers all aspects of forest ecosystem processes realistically and offers reliable proxies for ecosystem services across all temporal and spatial scales. On the other side, for a specific case study setting, only a distinct number of services is relevant for decision-makers. Therefore it is essential to carefully choose and evaluate existing models with regard to simulated sub-processes and output parameters (Monserud 2003). Another important step for a successful forest landscape assessment is to ensure acceptance of the decision support tool amongst the diverse stakeholders, including parties outside the traditional forest sector. Hence, the successful application goes hand in hand with an intelligible description of all underlying model assumptions and by objective validation of outputs, including estimates of uncertainty (Muys et al. 2011). Model intercomparison studies are recommended (e.g., Huber et al. 2013; Warszawski et al. 2014), where, within a harmonized framework, a set of models is compared with each other and independent observational data. Model intercomparisons show great potential to objectively evaluate the strengths and deficiencies of models and foster scientific exchange between working groups. Ultimately, the establishment of transparent processes to ensure model quality throughout all stages of model development, evaluation and application, is a prerequisite for a deeper understanding of ecological systems and scientific progress (Augusiak et al. 2014).

Forest ecosystem models, applied in a proficient and responsible manner, can give insight into the dynamics of complex ecological systems and the interacting effects of biological processes and management under climate change. Subsequently derived proxies for ecosystem services are fundamental assets of decision support systems in conflicting stakeholder settings.

# 2 OBJECTIVES

The overall aim of this thesis was to assess the effects of mountain forest management strategies on ecosystem service provisioning under the influence of climate change.

Specifically, the assessment included the following tasks:

(1) Evaluation of the forest ecosystem model PICUS version 1.5 by comparing projected stand development under management against inventory data and other forest models.
(2) Design of realistic, spatially explicit management plans depicting the currently practiced mountain forest management regime and potential adaptive variants in long-term scenarios for the forest ecosystem model PICUS version 1.5.
(3) Application of the simulation and assessment framework to evaluate multifunctionality and adaptive capacity of alternative mountain forest management plans under climate change.
(4) Analysis of the effects of spatial scales in the quantification of ecosystem services for mountain forest ecosystems.

The objectives of this thesis were pursued in three scientific publications. A brief overview is presented below in Figure 2.

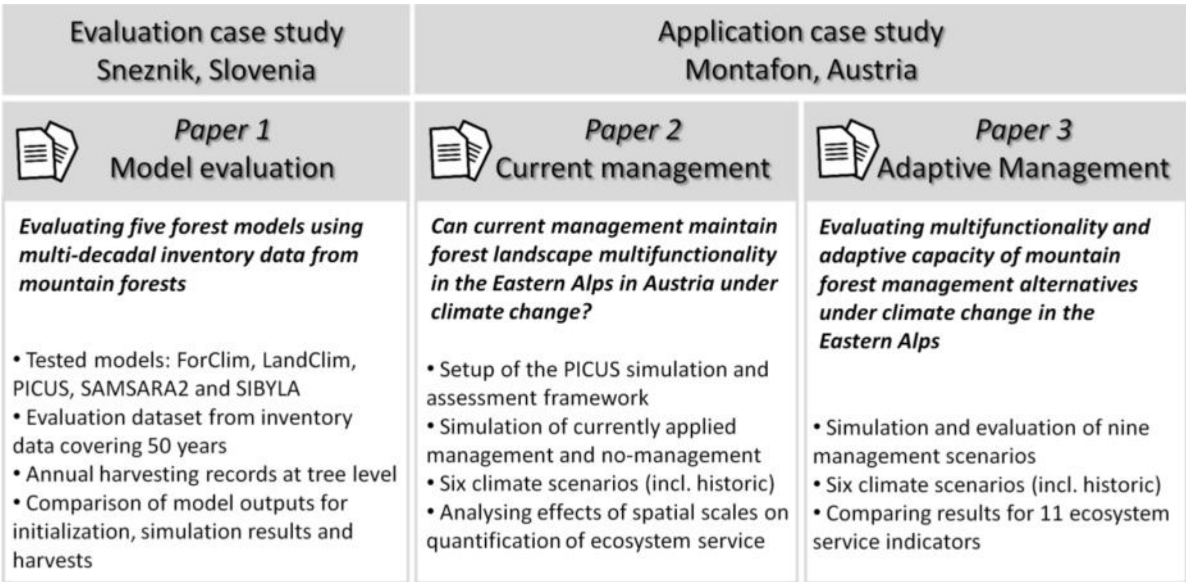


**Figure 2.** Content and case studies used in the individual papers.

# 3 Materials and Methods

## 3.1 The PICUS simulation and assessment framework

For this study, the hybrid forest ecosystem model PICUS version 1.5 was employed. The hybrid model PICUS 1.3 (Seidl et al. 2005) was created by combining the classical gap model PICUS 1.2 (Lexer and Hönninger 2001) and algorithms from the 3-PG stand-level production model (Landsberg and Waring 1997). In succeeding PICUS versions 1.3 to 1.5, core model components remained unchanged as described in detail in Seidl et al. (2005). In the latest PICUS version 1.5 the functionality was extended for simulating and analyzing contiguous landscapes by implementing irregular stand shapes, a raster-based management module and single tree model outputs for analysis in a landscape assessment tool (Maroschek et al. 2015). An overview of publications dealing with conceptual model extensions, applications and evaluations is provided in Figure 3. Below, a brief summary of PICUS 1.5 properties is given, with particular focus on spatial aspects of simulation processes and outputs.

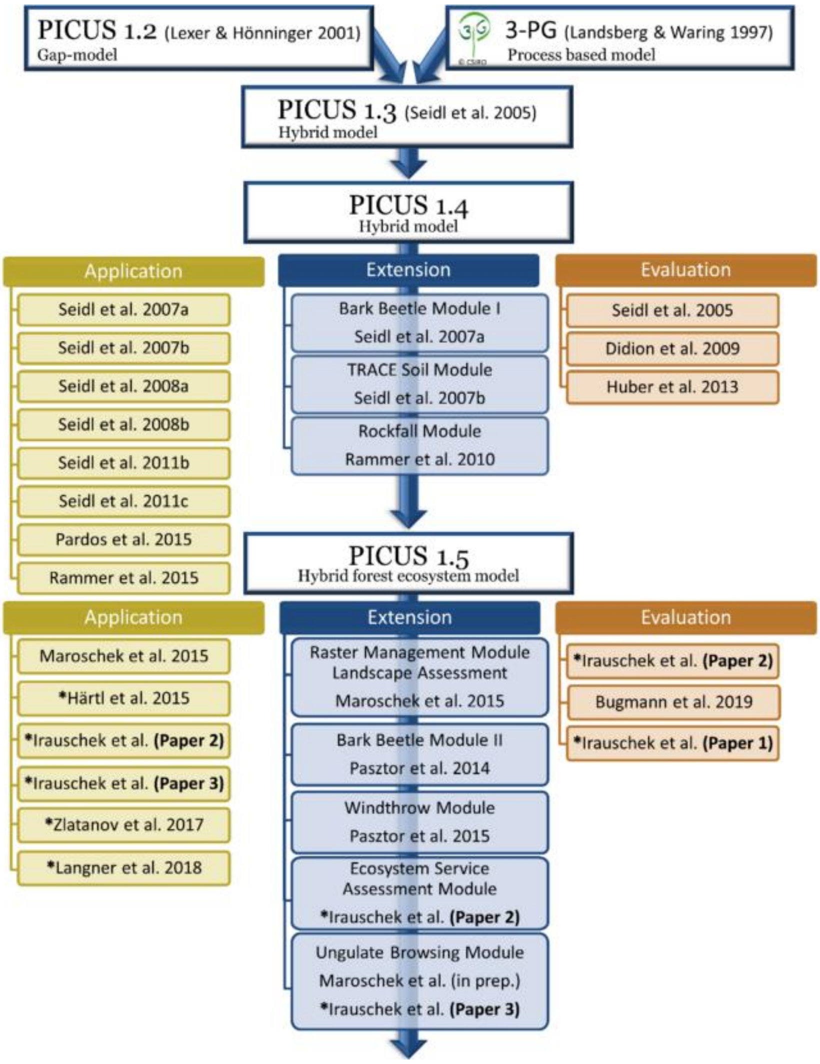


**Figure 3.** Model development towards PICUS version 1.5 including a chronological publication history starting from version 1.4. *) Publications with personal contribution of the thesis author

PICUS version 1.5, henceforward referred to as PICUS, operates on individual trees positioned on a grid of 10 x 10 m patches, whereupon the leaf biomass of trees is arranged in crown cells of 5 m height. Employing 3-PG algorithms (Landsberg and Waring 1997), stand-level net primary production is estimated by radiation interception and light use efficiency, which depends on temperature, precipitation, solar radiation, vapor pressure deficit, soil water and nutrient supply. Redistribution of assimilates to individual trees is accomplished according to the relative competitive success of individuals within the gap model environment (Lexer and Hönninger 2001). The temporal resolution of the simulation is monthly with annual integration of tree population dynamic processes. As climate input, monthly values of temperature, precipitation, solar radiation and vapor pressure deficit are required. Tree regeneration dynamics are simulated within five height classes (Woltjer et al. 2008), and a browsing module can simulate species-specific reduction of height growth for regeneration caused by ungulates (Paper 2). PICUS includes a module to account for detailed Carbon cycling processes in the soil and litter layer (Seidl et al. 2007b). A process-based module for European spruce bark beetle (*Ips typographus* L.) estimates annual infestation risk, damage intensity and mortality pattern for spruce trees, based on temperature and tree dimensions (Lexer and Hönninger 1998; Seidl et al. 2007a). PICUS offers a comprehensive management module, including single tree selection harvests based on a scripting language, tree selection on 2 x 2 m resolution raster grids, as well as planting operations at the level of the 10 x 10 m patches. PICUS operates on simulation units up to 25 ha. One simulation entity consists of spatially explicit patches with uniform soil attributes (pH, water storage and plant available Nitrogen). If tree positions are not provided as input data, trees are randomly distributed, and tree maps of different simulation entities can be exported and loaded into a landscape assessment tool for combined analysis and visualization (Maroschek et al. 2015).

## 3.2 THE EVALUATION CASE STUDY SNĚŽNIK

The case study Sněžnik was used for evaluating the PICUS model within the frame of a forest model intercomparison study (Paper 1).

### 3.2.1 STUDY AREA DESCRIPTION

The Sněžnik Mountain is a karst limestone plateau in the northern part of the Dinaric Mountains, in Slovenia, Europe. The forest soils are mainly chromic Cambisols and rendzic Leptosols. The study area extends from 800 m to 1300 m a.s.l. (meters above sea level). The local climate has Mediterranean influences with high summer and low winter temperatures. Temperatures range between 6.8 °C mean annual temperature (MAT) at 800 m and 3.1 °C MAT at 1300 m a.s.l., with precipitation increasing with elevation from 1670 mm to 1930 mm mean annual precipitation (MAP). Mountainous silver fir (*Abies alba* Mill.) - European beech (*Fagus sylvatica* L.) - Norway spruce forests are the prevalent natural forest type. Forests are currently managed in a combination of single stem and small-scale irregular shelter-wood system entitled “free style silviculture” (Mlinsek 1968; Boncina 2011).

### 3.2.2 FOREST MODEL INTERCOMPARISON

In the project ARANGE (Advanced Multifunctional Forest Management in European Mountain Ranges), forest ecosystem models originating from different European regions were applied as integrative planning and decision support tools. This framework was utilized to compare PICUS (see section 3.1), the gap model ForClim (Bugmann 1996), the landscape model LandClim (Schumacher et al. 2004) and the spatially explicit empirical models SAMSARA2 (Courbaud et al. 2015) and SIBYLA (Fabrika 2005). The models were tested against inventory data from nine compartments in the Sněžnik case study, covering a 50-year period and including annual harvest records. To account for uncertainty in drivers of forest development, simulation scenarios for the initial state of small trees, browsing rates by ungulates and sanitary harvests were defined. The models were compared regarding the accuracy of (i) initialization using historical inventory data, (ii) implementation of management from harvest records and (iii) simulated compartment states. The statistics Nash-Sutcliffe Efficiency (NSE) (Nash and Sutcliffe 1970), the mean error (ME) and the root-mean-square error (RMSE) were calculated to compare basal area of the model outputs. The Diameter Distribution Error (DDE), described as "total variation distance index" in Levin et al. (2009) was calculated to compare forest structures. Periodic volume growth was calculated in a standardized approach by using local height curves and tree volume functions (see Paper 1).

## 3.3 The application case study Montafon

As the main study area for this thesis, the case study Montafon was used in Paper 2 and 3.

### 3.3.1 Study area description

This study area is situated in the Austrian province Vorarlberg in the Montafon valley. The forests are managed by the Stand Montafon Forstfonds, an association owned by local municipalities, holding about 6500 ha of forest land in the region. Land use has shaped the landscape for at least 500 years (Bußjäger 2007). While the valley bottoms are currently largely occupied by settlements, infrastructure and agricultural use, forests are situated on the steep hillsides up to the tree line, limited by alpine pastures and ski slopes on suitable sites. Norway spruce is the dominating tree species with a share of 96%, followed by Silver fir with 3% and minor shares of European beech and other broadleaved trees (Maier 2007). The most relevant forest ecosystem service is protection against multiple natural hazards (rockfall, avalanche release and erosion). Other essential forest services are the production of valuable timber, game management and nature conservation (in Natura 2000 and forest nature reserves) (Malin and Maier 2007).

A catchment in a side valley named Rellstal has been chosen for detailed analysis. Here forest stands are situated on steep north- and south-facing slopes and stretch between 1060 m and 1800 m a.s.l.. Depending on bedrock, soil types are Leptosols (Rendzinas and Rankers), Podzols and rich Cambisols. The historical climate (period 1961-1990) was characterized by low temperatures and ample precipitation (6.2 °C MAT and 1486 mm MAP at 1300 m a.s.l.). Five climate change scenarios for the simulation period 2000 to 2100 were employed based on simulations from the ENSEMBLES project (Hewitt and Griggs 2004). Downscaling to the local values was described in Bugmann et al. (2017). The climate change scenarios covered a wide range of possible transient future conditions (+2.6 up to +6 °C MAT increase between the historical climate and the period 2071-2100) (for details see Paper 2). Detailed forest structures and soil properties in the case study catchment were estimated using terrestrial data (stratified raster sampling of forest and soil attributes) and airborne remote sensing data (normalized crown model and volume map (Hollaus et al. 2006; Hollaus et al. 2007). Utilizing both datasets, an algorithm generated tree-maps containing size, species and location of individuals for distinct stand polygons (see Maroschek et al. 2015 and Paper 2). Together with the non-forest area (mainly steep ditches and forest roads), in total, 53 forest stands added up to 270 ha catchment area used in the simulation experiment. Based on inventory data, the current annual ungulate browsing probability for tree regeneration was estimated for the landscape. Probability for browsing (annual fraction of simulation patches experiencing browsing) was 0.31 for spruce, 0.68 for fir, 0.41 for maple and 0.38 for beech.

### 3.3.2 Local model evaluation

For evaluating the accuracy of PICUS in reproducing tree growth and stand structure development in the application case study, a dataset based on tree ring analysis originating from an adjacent valley was utilized (Neumann 1993). For details, see Paper 2.

### 3.3.3 Forest management strategies

For steep terrain conditions, as observed in the case study Montafon, any extraction of harvested timber is based on cable yarding with temporary skyline systems. Therefore, limitations regarding hauling distances and clearance of tracks had to be considered for the design of spatially explicit harvesting alternatives. Details of current and possible alternative management regimes were based on forest management records, analysis of recent harvests on orthophotos and forest stakeholder interviews conducted in the ARANGE project and for Maroschek et al. (2015). For simulating the development of the case study catchment in PICUS, the 53 forest stands were merged to 18 harvesting units. Thus, the effects of management on structural dynamics of the forest ecosystem could be simulated in a detailed and contiguous way. Another argument was that current strategies aim for uneven-aged forests, which will result in gradually diminishing differences between forest stands. Hence, the harvesting units were defined based only on the topography and road infrastructure, which determine the setup of skyline harvesting systems (see Paper 2).

For this study, a rigid harvest pattern design was conceptualized to allow for a contrasting and comprehensible alteration of management interventions. At first, possible skyline tracks were delineated across the case study landscape, depending on terrain attributes and suitable landing sites on forest roads. Then harvests were timed based on tree maturity and uniformity of stand structure along the skyline tracks. The schedule was fixed for all active management alternatives. To study the effects of cutting pattern along the skyline tracks, small slit-cuts (300 m²), currently practiced irregularly shaped patch-cuts (1500-2000 m²) and strip-cuts (5000 m²) were compared (Figure 4). Area turnover rates were simulated as currently observed (250 years, low-intensity) and in a high-intensity management alternative (150 years) by decreasing the track return intervals and including additional tracks. Cutting pattern and turnover rates were simulated in a full-factorial design. In addition, two management variants were simulated incorporating artificial regeneration: (i) For strip-cuts at 150 years rotation length, a mix of spruce, larch and maple was planted after felling and (ii) in a sanitary management alternative, trees infested by bark beetles were cut down on site and resulting gaps were planted with sycamore maple. For comparison, a management regime without interventions was simulated. For further details on skyline track management (Paper 2) and management alternatives (Paper 3), compare the original publications in the appendix of this thesis.

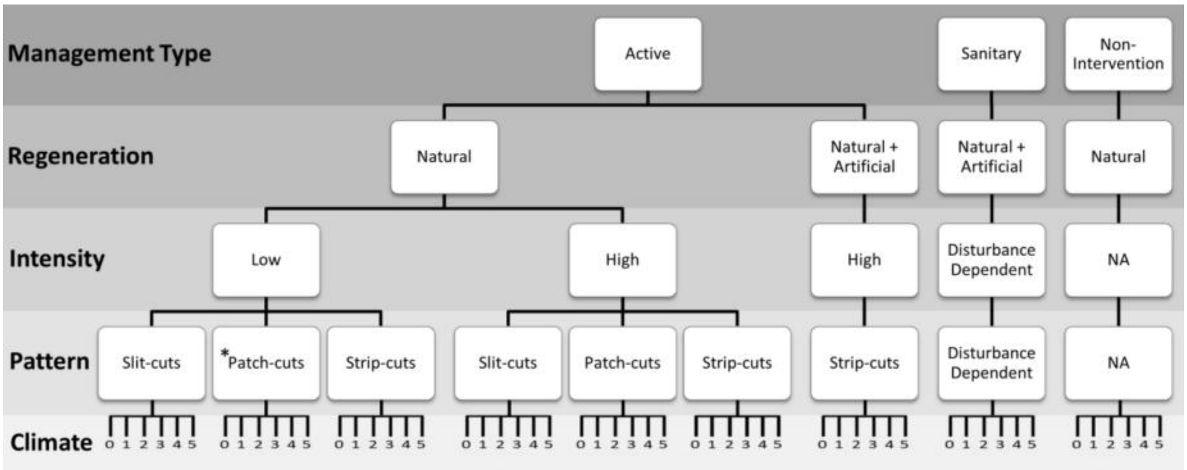


**Figure 4.** Simulation scenarios as applied in paper 3. 0 = historical climate, 1 to 5 = climate change scenarios. NA = not applicable. *) Currently practiced management (Patch-cut pattern x Low Intensity x Natural Regeneration).

### 3.3.4 ECOSYSTEM SERVICE INDICATORS

The assessment of ecosystem services from model outputs was based on the extensive set of ecosystem service indicators developed and compiled in the ARANGE project (Cordonnier et al. 2013; Bugmann et al. 2017). For the case study Montafon, a set of 11 indicators was selected, representing timber production, protection against gravitational hazards, Carbon storage and nature conservation, complemented by an additional indicator for tree regeneration status (Table 1). The simulation output was available at annual resolution. After loading it into the landscape assessment tool, grain sizes of 1, 5 and 10 ha grid cells were used to calculate indicator values (see Paper 2).

Spatial indicator aggregation at catchment level was done as presented in Table 1. Temporally, results were aggregated in three 33-year periods. Indicators for timber production and Carbon sequestration were scaled up to the landscape level by area-weighted average. Other ecosystem services, which require a smaller and rigid spatial context, were calculated for 1-ha cells. For scaling up to the landscape scale, area percentiles (10$^{th}$, 50$^{th}$ and 90$^{th}$) were used for interval scaled indicators, whereas area percentages were calculated for categorical indicators (see Paper 3).

**Table 1.** Indicators for ecosystem service assessment in the Montafon case study catchment for comparing simulation results for current management practices and eight adaptive management alternatives (Paper 3). Temporal aggregation in three 33-year periods. Percentiles = $10^{th}$, $50^{th}$ and $90^{th}$ area percentile. % area = area percentage of three categories.

| Ecosystem Service | Indicator | Unit / Index Interval | Temporal aggregation | Spatial grain size | Spatial aggregation |
|---|---|---|---|---|---|
| Timber production | Timber harvested | $m^3\ ha^{-1}\ yr^{-1}$ | mean | harvest unit | mean |
| | Volume increment | $m^3\ ha^{-1}\ yr^{-1}$ | mean | harvest unit | mean |
| | Bark beetle damage | $m^3\ ha^{-1}\ yr^{-1}$ | mean | harvest unit | mean |
| | Standing timber | $m^3\ ha^{-1}$ | mean | harvest unit | mean |
| Carbon storage | Carbon in Trees, Soil & Woody Debris | $t\ ha^{-1}$ | mean | harvest unit | mean |
| Nature Conservation | Bird Habitat Quality | [good, med, poor] | mode | 1-ha cell | % area |
| | Tree Species Diversity | [-] | mean | 1-ha cell | percentiles |
| | Tree Size Diversity | [-] | mean | 1-ha cell | percentiles |
| Protection against Gravitational Hazards | Snow Avalanches | [0, 1] | minimum | 1-ha cell | percentiles |
| | Landslides | [good, med, poor] | lowest | 1-ha cell | % area |
| | Rockfall | [0, 0.99] | minimum | 1-ha cell | percentiles |
| Tree regeneration | Availability of Regeneration | [true, false] | mean | 1-ha cell | % area |

# 4 RESULTS

## 4.1 MODEL INTERCOMPARISON RESULTS

At first, the models' initializations for 1963 were compared with the initial inventory data. PICUS, SAMSARA2 and SIBYLA showed a very accurate initialization. The root mean squared errors (RMSE) of basal area were below 0.7 $m^2ha^{-1}$ and the mean diameter distribution error (DDE) below 2 %. ForClim had a higher RMSE of 1.4 $m^2ha^{-1}$ due to lower initialized basal area in two compartments. LandClim had relatively high deviations from the observed inventory values (RMSE 6.9 $m^2ha^{-1}$, mean DDE 6.0 %), always initializing lower basal area densities.

Starting with the initial compartment characteristics, the models simulated the development according to the scenarios for initialization of small trees, browsing by ungulates and sanitary harvests. Resulting basal area was compared with the observations in 1983 and 2013. SAMSARA2, SIBYLA and PICUS showed good model performance (NSE > 0.5; mean RMSE < 5 $m^2$ $ha^{-1}$) with PICUS being the only model performing well in both observation periods.

Compared with the local estimate for total volume growth (approximately 9 $m^3ha^{-1}yr^{-1}$), ForClim, SAMSARA2 and SIBYLA simulated lower growth (5.6, 7.0 and 7.1 $m^3ha^{-1}yr^{-1}$), LandClim was clearly overestimating (15.7 $m^3ha^{-1}yr^{-1}$), and PICUS was fairly close (8.4 $m^3ha^{-1}yr^{-1}$). The simulated compartments were managed regularly in intervals of 10 to 20 years and, in some cases, sanitary logging occurred almost every year. The models SAMSARA2 and SIBYLA reproduced the recorded harvest volumes very accurate with deviations below 0.5 $m^3ha^{-1}yr^{-1}$ (basal area sum over all compartments). PICUS (-0.8 $m^3ha^{-1}yr^{-1}$) and ForClim (-1.4 $m^3ha^{-1}yr^{-1}$) also showed good agreement with the harvest records. Simulated harvests differed more for LandClim (+2.2 $m^3ha^{-1}yr^{-1}$), whereat particularly the diameter structure of harvests deviated considerably from the records.

The uncertainties in data were addressed by simulation of the twelve scenarios combinations. The range of results depicted the predicted model uncertainty, and furthermore showed the general sensitivity of the models towards scenario assumptions. The variability, as triggered by scenarios for initialization and browsing, was smallest for LandClim (2% of the mean result) and largest for SAMSARA2 (8%). Adding the scenarios for sanitary harvesting increased the variation of the mean output to 24% (SAMSARA2), 11% (PICUS) and 6% (SYBILA).

Overall, the error statistics indicated that simulated basal area development was most accurate for the individual tree-based models PICUS, SAMSARA2 and SIBYLA. ForClim, a non spatial gap model, was also performing well, although performance ratings were slightly lower due to higher divergence in one of the simulated compartments. On the other hand, ForClim was performing best regarding diameter structure conformity. LandClim, considering its coarser spatial and temporal simulation and management resolution, delivered reasonable results regarding basal area statistics. Nevertheless, the performance of LandClim regarding diameter distributions was poor (see Paper 1).

## 4.2 APPLICATION OF THE PICUS SIMULATION AND ASSESSMENT FRAMEWORK

### 4.2.1 EVALUATION OF THE ECOSYSTEM SERVICES ASSESSMENT APPROACH

As a prerequisite to interpret differences in ecosystem service indicators simulated by PICUS, the model-induced stochastic variation was evaluated, by comparing the results of 10 identical model runs. The main source for variation was tree mortality. Damage by bark beetle showed the highest variability, indicated by a coefficient of variation (CV) of 9 - 17% (historical climate, range due to managements). The variation decreased with increasing climate change effects as damages occurred more frequently (CV 3 - 6% in most severe climate). Another contributing source was the spatial aggregation of indicators across the landscape. Indicators aggregated as mean values showed very low variation (CV < 3%), while nature conservation and gravitational hazard indicators represented by critical sub areas and rare structures showed moderately higher variability (CV < 8%) (see Paper 2).

The utilized ecosystem services indicators were designed to provide estimates at stand scale. Yet, the classical silvicultural stand definition (i.e., a forest subarea of variable size with similar tree and site attributes) is not precise in terms of spatial context. Therefore, the effect of different grain sizes (1, 5 and 10 ha cells) on indicator values was analyzed. The results showed significant differences only for the bird habitat indicator and for protection against landslides. Most sensitive was the bird habitat indicator, depending on rare structural elements such as veteran trees and standing deadwood with large dimension. Here the increase of cell size resulted in more "average" habitat conditions (i.e., decreasing area rated as "bad" for all simulated scenarios and in a decrease of "good" area) in most cases. For further analysis it was concluded, that indicators for nature conservation and protection against gravitational hazards, which require a rigid spatial context, should be assessed using grain sizes of 1 ha. For indicators of timber production and Carbon sequestration linear scaling of the simulation units to the landscape level was sufficient.

Analysis showed that grain size was also affecting the landscape level results regarding multifunctionality (i.e., the area shares with simultaneous provisioning of multiple ecosystem services). Multifunctionality decreased with increasing number of considered services and increased with larger grain size. For example, considering a set of four ecosystem services and a requirement of at least 2 fulfilled services for multifunctionality, the respective area increased from 76% at 1 ha grain size to 100% at 10 ha grain size (non-intervention, aggregate for third simulation period). Moderate climate change increased the multifunctional area share, while under severe climate change this trend was reversed (see Paper 2).

### 4.2.2 EVALUATION OF THE CURRENT MANAGEMENT REGIME

Under historical climate (i.e., no climate change) and the current management regime (patch-cuts with low intensity) the forest stocks in the catchment increased from 436.5 to 490.1 $m^3\ ha^{-1}$ at the end of the simulated period, while on average 1.9 $m^3\ ha^{-1}$ of timber was extracted per year. Closely correlated with volume were in situ carbon pools, which increased slightly from 220 to 223 t ha$^{-1}$. The protective indicators improved (e.g. Landslide protection area rated "good" increased from 48% to 71%). Similarly, the bird habitat conditions generally improved from 49% to 64% suitable area (combined "moderate" and "good" rating), but the area with "good" rating decreased from 10% to 0% in the last simulation period. Tree species diversity, which was already low in the beginning (diversity index of 1.33), further decreased to 1.23 (see Paper 2).

Effects of the five climate change scenarios on forest growth rates were positive or at least neutral. Only in the most severe climate scenario (+5.9 °C MAT) a negative impact in the last simulation period was observed. However, in parallel a strong intensification of the bark beetle disturbance regime and related damages in the spruce dominated landscape was simulated (from 0.5 m³ ha$^{-1}$ yr$^{-1}$ under historical climate up to 2.9 m³ ha$^{-1}$ yr$^{-1}$). Disturbances reduced the number of veteran trees and increased deadwood volume and, with exception of the mildest scenario (+2.6 °C MAT and +20% MAP), reduced standing tree volume and Carbon stocks. Climate change favored the bird habitat quality in the catchment, because of more standing deadwood of large dimensions and canopy openings due to bark beetle damages (21% area "good" rating in the most severe scenario). Tree diversity indicators showed a moderate increase in the last simulation period, but could not reverse the generally decreasing temporal trends (e.g., decline of silver fir basal area shares from 6.2 to 4.7% in the most severe climate scenario) (see Paper 2).

### 4.2.3 Comparison of management alternatives

In general, the observed direct effects of the climate scenarios on forest dynamics were less distinctive than effects of management. On the other hand, the management intensities had considerably larger effects than the cutting patterns. Where not relevant, the intensity effects were compared by averaging the results of different cutting pattern. Presented summary results focus on the third simulation period (2067-2100), when the effects of management and climate were more pronounced.

With higher management intensity the harvested timber in the case study catchment increased from 2.5 m³ ha$^{-1}$ yr$^{-1}$ to 3.3 m³ ha$^{-1}$ yr$^{-1}$ (historical climate). This intensification resulted in increased mean periodic increments and the standing timber decreased only slightly until 2100. There was a trend of higher bark beetle damages in the no intervention and sanitary management scenarios (3.1 m³ ha$^{-1}$ yr$^{-1}$, severe climate change), followed by low intensity alternatives (2.7 m³ ha$^{-1}$ yr$^{-1}$), while high intensity management variants tended to have lower damages (2.2 m³ ha$^{-1}$ yr$^{-1}$). However, increased management intensities resulted in adverse effects on protective functionality. Within the entire simulation period of 100 years, protection against gravitational hazards was best in the non-intervention alternative. On the other hand, this scenario showed low area shares with regeneration, indicating a low resilience against disturbances and longterm development of ecosystem service provisioning.

Comparing the different cutting patterns, the strip-cut pattern was clearly the least favorable regarding protective services. Patch and slit-cut pattern showed very similar results, whereat the fine-grained slit-cut pattern showed significant improvement of protective functionality only in high intensity management. Also bird habitat suitability was observed as being highly sensitive to management. Current management (patch-cut pattern at low intensity) delivered the best results under most climate scenarios.

Slit- and patch-cut pattern were slightly better in promoting tree species diversity in the regeneration compared to the strip-cuts, but a relevant change in species composition was only observed in the artificial planting scenarios, because it was assumed that planted trees were protected from browsing effects. Under climate change, artificial regeneration became more effective. Furthermore, results indicated a slight increase in sensitivity to cutting pattern under the climate change scenarios, but in general the relative performance of the different management options remained similar, independent of the climate scenario.

Given these diverse results for the different management strategies, no alternative was identified as beneficial regarding all ecosystem service indicators (Figure 5). For the majority of ecosystem services (protection, Carbon storage and partly nature conservation) high intensity management strategies were not favorable. With non-intervention and sanitary management current high Carbon stocks were maintained and protection against gravitational hazards further improved. However, sanitary management was highly adverse for bird habitat quality, due to the cutting of dead standing trees. Low intensity management with patch- or slit-cuts provided good habitat quality for birds and showed a good performance as an integrative scenario for all considered ecosystem services (see Paper 3).

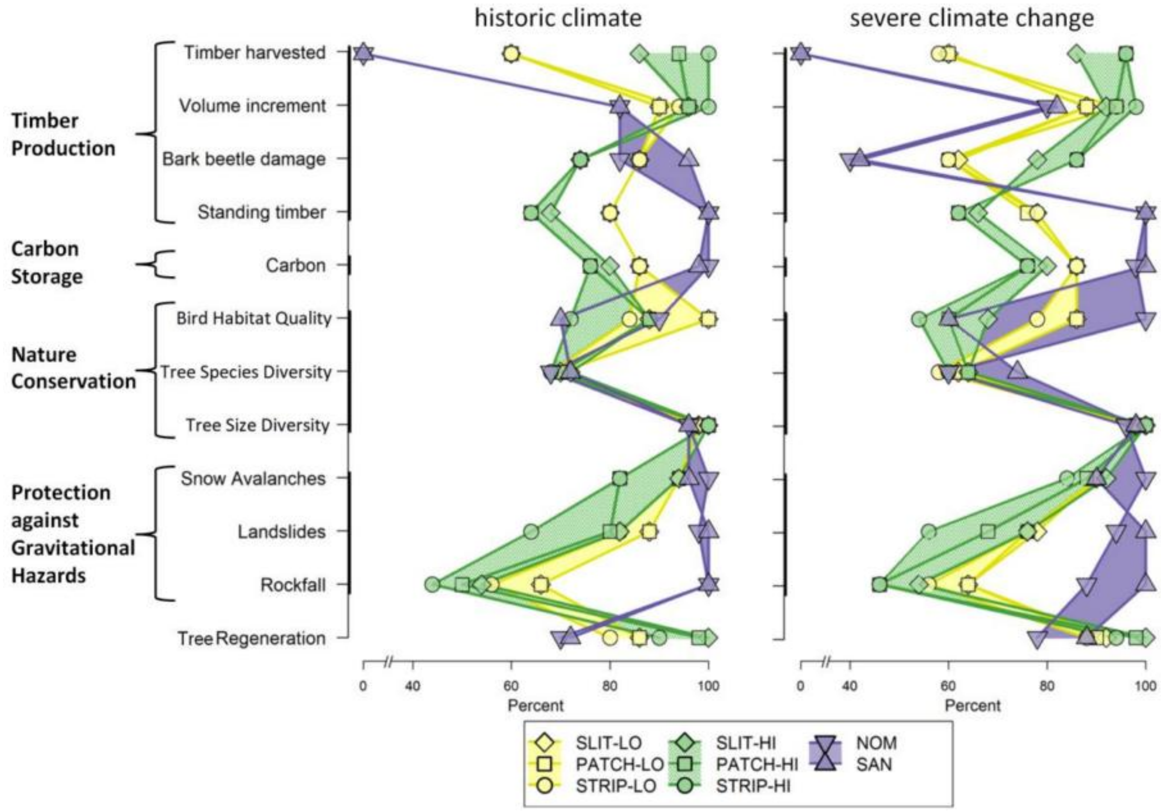


**Figure 5.** Ecosystem service indicators in period 2067-2100 relative to best management scenario according to preference direction. Colored band depicts range for management intensities: low intensity (LO) = *yellow*, high intensity (HI) = *green* (pattern), no intervention (NOM) and sanitary (SAN) = *purple*. Tree species diversity and Tree size diversity are represented by 50th percentile, Protection against snow avalanches and landslide by 10th percentile, Protection against landslides and bird habitat quality as share of area rated as “good”.

# 5 DISCUSSION

For planning harvesting activities in mountain forest ecosystems many critical questions have to be considered. Silvicultural techniques in mountain forests stands are known for the wider Alpine region (e.g., Burschel and Huss 2003) and in practice for well-defined initial states and management goals (cf Hirsiger et al. 2013; Veneziani et al. 2014). However, if many ecosystem services are considered and the analyzed forest area gets larger, sustainable management gets complex and the planning approach applied is often perceived as "kind of trial and error" (Dorren et al. 2004). Application of the PICUS simulation and landscape assessment framework offered the possibility for a comprehensive long-term test of current management regimes but also of more extreme interventions up to the implications of doing nothing. The simulated management alternatives caused distinct pattern across the forest landscape, superimposed by interacting disturbances and tree population dynamics influenced by climate change. The application case study results demonstrated the important role of disturbances caused by the European spruce bark beetle in Norway spruce dominated mountain forests, which are highly sensitive to climatic changes. Expected damages set off the warming-induced gain in volume increment, simulated for the cool-wet conditions in the spruce dominated case study in most climate scenarios and furthermore impede protective services.

Main outcome of the comparison between the management alternatives was that the best integrative results, considering all ecosystem services, could be achieved with low intensity patch-cut management characterized by a rotation length of 250 years and cutting areas of 1500 to 2000 m² along the cable yarding tracks. This management approach furthermore indicated a good potential to initiate and facilitate regeneration processes for long term resilience of the mountain forest ecosystem.

Brang et al. (2017) published findings of a field study assessing the effects of a management similar to the simulated strip-cut pattern. Results in Brang et al. (2017) showed that no substantial delay of regeneration is expected in the relatively large openings because of harsher climatic conditions or mechanical effects such as snow gliding. This underpins the simulation results for the growth of regeneration in the simulated strip-cut management. However, browsing by ungulates is another crucial factor for the successful establishment of regeneration because high browsing rates prolong critical regeneration phases and oppress an increase of tree species diversity as a means to adapt against climate change in the long term. The fundamental influence of ungulates has been confirmed by field research (Hirsiger et al. 2013; Veneziani et al. 2014) and other simulation studies (Didion et al. 2011; Klopcic et al. 2017). Results from the tested management scenarios including artificial regeneration (high intensity strip-cuts and sanitary management) demonstrated the potential for facilitation of alternative tree species for climate change adaptation in the spruce dominated case study. However, protecting planted trees from browsing would be very costly and not always practically feasible in steep mountain forest conditions. Yet, for interpreting simulation results methodological constraints have to be considered. Usually, as also for this study, in simulation models tree regeneration is affected at species level by browsing rates, which are assumed constant across space and time. Browsing rates were determined by an inventory for the case study catchment, but they do not necessarily reflect local deer densities because their impact depends on many factors such as inter-annual migration, disturbances by human leisure activities and hunting and the availability of herbaceous forage (Kuijper et al. 2010). Game-management decisions made at landscape scale have to aim for ungulate densities in balance with the ecosystem and have to be in accordance with climate change adaptation goals. Therefore, a reduction of deer densities may be a first step. Additional efforts have to include altering of spatial and temporal habitat use by selective hunting activities (Royo et al. 2017). However, the relationship between deer density and browsing impact may not be linear (Kuijper et al. 2010) and the abundance of tree regeneration and herbaceous vegetation across the landscape affects the general carrying capacity of the habitat (Reimoser and Gossow 1996). Therefore, regular monitoring of tree regeneration browsing rates, their influence on vegetation development and evaluation of game-management decisions are necessary for successful adaptive forest management.

There are many ways this study could be extended by including processes relevant in mountain forest ecosystems in a more detailed way. For instance, interaction effects between gravitational hazards and vegetation (Zurbriggen et al. 2014; Rammer et al. 2015), interaction effects of forest structure on the impact of ungulate browsing (Royo et al. 2017), simulation of deer populations and distributions (Millington et al. 2013), microhabitat models for biodiversity assessment (Courbaud et al. 2017) or the explicit planning of harvesting equipment (Bont et al. 2014). However, additional sub-models and higher process details usually come at the price of increasing requirements for model input data and model evaluation datasets. Growing model complexity does not necessarily increase the quality of produced scientific output (Grimm et al. 2005). Another way for extending the knowledge base for decision support can be the parallel application of multiple models operating on different spatial levels (e.g., Lam et al. 2004; Zlatanov et al. 2017). However, this requires careful evaluation of model conformity across the case study. The presented evaluation study (paper 1) demonstrated a reasonably well performance of the landscape model LandClim at the stand scale, qualifying it for such a complementary application for analysis at the landscape scale.

Regardless of the choice of model detail and scale, in complex and comprehensive application studies there is always the danger that model developers partly neglect or rush through stages of reliable model development and evaluation (Augusiak et al. 2014) or application results in the end do not reach practical decision-makers for implementation of adaptive management in the real world (Mattsson et al. 2018). To increase acceptance of simulation tools for decision support, one mayor challenge is the objective validation and verification of outputs (Muys et al. 2011). Model intercomparison exercises bear the potential for increasing the credibility of ecosystem modeling. These studies offer great opportunities to compare detailed model outputs and point out differences, deficiencies and best practice examples. Presented results, for example, showed differences in mortality estimates for the tested forest models and demonstrated the efficiency of the applied model harvesting routines, which are substantially contributing to the accuracy of simulations in managed forest ecosystems.

The novelty of the simulation approach applied in this thesis lies in the combination of detailed implementation of harvesting patterns and ecosystem processes on a contiguous forest catchment case study. Classical stand level forest models usually simulate generically shaped stands of some hectare size, which may be scaled up with respective represented area in the landscape. These models potentially offer high resolution management routines, but cannot depict the harvesting pattern along the slope caused by cable yarding in mountainous terrain. Forest landscape models on the other hand simulate pattern-process interactions such as seed dispersal, disturbance propagation and hydrologic flow across contiguous landscapes starting at extends usually larger than 10 $km^2$ (Keane et al. 2015). The usual drawback is a necessary lower detail in spatial resolution and management. The characteristics of the applied PICUS version 1.5 lie between these two model categories. PICUS can simulate forest dynamics, including disturbances, for spatially explicit harvesting units up to 25 hectares and offers continuous indicator assessments by utilizing the landscape assessment tool. The model offers detailed single-tree management prescriptions, spatially explicit initialization, algorithms for sanitary management and sub-modules such as the ungulate browsing and the bark beetle module. For evaluating the resilience of mountain forest ecosystems, a broad view on all regionally relevant ecological processes is necessary (Dorren et al. 2004). Strategies for adaptive forest management therefore include ecosystem service indicators based on complex forest features such as diameter structure, species composition and spatial structure of the tree population. Realistic integration of spatial details and mortality and regeneration processes are therefore highly relevant for applied simulation models. Tree growth, which is a predominant criterion in production forests and often the main focus of attention in model evaluation and performance ratings, is only one factor contributing to management decisions in mountain forestry.

Recently many publications showed the importance and practical relevance of multi-dimensional and integrative landscape studies to address complex management and policy issues (O'Farrell and Anderson 2010; Villa et al. 2014; Shifley et al. 2017; Luo et al. 2018). Contrary to Scandinavian countries, where landscape level planning has a longer tradition (Fries et al. 1998; Andersson et al. 2006), it is rarely applied in forestry in Central Europe so far. Here the consideration of protective services against gravitational hazards requires single tree resolutions and fine grained gap analysis of indicator outputs, provided at large spatial scales (see Heinimann 2010; Seidl et al. 2013). This study provides an application example in a mountain forest catchment of 2.7 $km^2$, emphasizing tree level processes while keeping track of effects and conceptual limitations at larger scales for the time scope of a whole century. The general finding, that none of the analyzed management alternatives was best for all considered ecosystem services indicated that the social context of a decision-making situation determines which services are prioritized and subsequently which management approach suits best. This knowledge finally can be used to distinguish zones for specific ecosystem service priorities and provide targeted management recommendations.

# 6 CONCLUSIONS

The results of the assessment of management scenarios demonstrated that in the case study catchment, the currently practiced patch-cut regime at low intensity level appears as a well-suited strategy to maintain landscape multifunctionality. However, predicted disturbances by spruce bark beetle pose a major threat to the stability of the forest structures and hence call for adaptation actions. Indicators of stability and resilience against disturbances include (i) a diverse tree species composition, (ii) sufficient natural regeneration and (iii) diverse vertical and horizontal forest structures (Motta and Haudemand 2000). If adaptation by increasing tree species diversity in the currently spruce dominated landscape was the main goal, increased forest management intensities accompanied by ambitious game management would be required to foster the fast establishment of species such as fir, sycamore maple and beech. As a drawback, this may locally negatively impact on several ecosystem services. Therefore, careful evaluation of harvesting activities is necessary in zones with, e.g., high demand for protection against gravitational hazards or in sensitive bird habitats.

This study demonstrated the feasibility of model-assisted decision support in complex mixed mountain forest ecosystems. However, decision making about adaptation to climate change and sustainable management of ecosystem services is an evolving process and not a one-time application. For successful adaptive forest management, evaluation of the applied management strategies is necessary. It is important to regularly update projected forest dynamics by considering improved model routines, climate projections and refined management strategies.

Multidimensional and interdisciplinary approaches are crucial to increase our knowledge for adapting the forests to the challenges resulting from climate change and manifold demands from society. For creating an impact, this knowledge has to be disseminated amongst forest stakeholders and local and transnational communities depending on intact mountain forest ecosystems.

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# 8 APPENDIX

## 8.1 PAPER 1

Irauschek F, Barka I, Bugmann H, Courbaud B, Elkin C, Hlásny T, Klopcic T, Mina M, Rammer W, Lexer MJ (2020) Evaluating five forest models using multi-decadal inventory data from mountain forests. Ecological Modelling (accepted Manuscript)

# Evaluating five forest models using multi-decadal inventory data from mountain forests

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# Highlights

- Five forest models are tested against independent longterm forest inventory data
- Model sensitivity to uncertainty in framing forest conditions is tested by simulation scenarios
- Overall performance of empirical and process-based models did not differ
- This study prepares the ground for multi-model applications in decision support

# Abstract

Forest ecosystem models, being widespread science tools and used for forest management decision support are usually evaluated individually against field data sets, while model intercomparison and joint evaluation studies are rare. We tested five forest models according to a harmonized protocol against data from nine forest compartments in the Sněžnik region, in Slovenia. The suite of models included stand- and landscape-scale, empirical- and process-based models used across Europe. The test dataset originated from inventory data covering 50 years (tree measurements 1963, 1983 and 2013) and included annual harvesting records at tree level. Uncertainties in data and forest conditions were considered by defining 12 scenarios varying initial regeneration, browsing pressure and harvest modalities. We evaluated the models` ability to initialize forest conditions accurately, whether management interventions could be implemented based on harvest records, and how well basal area and diameter structure could be predicted.

Simulation results for basal area development showed good to satisfactory performance for all models, at which SAMSARA2, SIBYLA and PICUS showed the best agreement. Comparison of simulated and observed diameter distributions showed good performance of ForClim, PICUS, SAMSARA2 and SIBYLA. Model output variability was between 6% and 24%, indicating the relevance to consider uncertainties that can be attributed to specific sources. There was no clear hierarchy between more empirical or more process-based models regarding accuracy of stand development projections. The cohort-based landscape model LandClim showed the lowest stand-level accuracy and scenario sensitivity, but results nevertheless qualified it for complementary application at landscape scale. Within individual-based models, spatially explicit models seemed to be more suitable for heterogeneous mixed mountain forests. The findings demonstrated the usefulness of inventory datasets for model testing and intercomparison.

**Keywords:** model intercomparison, tree growth, tree mortality, forest management, ungulate browsing, forest inventory data

# 1 Introduction

Forest simulation models are powerful tools for testing and evaluating the mid- to long-term implications of different management strategies on future forest development and related ecosystem service provisioning (Söderbergh and Ledermann 2003; Schelhaas et al. 2014). Changing environmental conditions and intensifying disturbance regimes (e.g., Seidl et al. 2011; Temperli et al. 2013) have increased the complexity in forest resource planning and management, and consequently the role of model-based decision support has drawn a lot of attention recently (Muys et al. 2011; Linkevičius et al. 2019).

One response to this growing challenge has been an increased reliance on forest simulation models, that are potentially climate sensitive and allow for alternative and novel forest management strategies, to be evaluated. Concomitantly, there has also been an increase in testing the benefits, limits and credibility of forest models (Courbaud et al. 2015; Foster et al. 2017). Model evaluation studies are required to build trust and confidence in model outputs and are thus a prerequisite for any model application in practical decision support. In view of growing interest in complex forest structures, multi-species mixtures, the provisioning of various ecosystem services beyond timber production and the need to consider the effects of a changing climate, the demands being placed on forest models have grown considerably over the recent years. Silvicultural regimes that have been proposed to adapt forests to climate change often focus on small-scale silvicultural measures and the creation of heterogeneous stand conditions to foster forest resilience (Christensen 1997; Puettmann et al. 2009). Adequately representing these forest structures, and the underlying ecosystem processes that generate them, requires individual-based, climate-sensitive forest modeling approaches that allow the simulation of complex silvicultural tree selection and cutting patterns (Söderbergh and Ledermann 2003; Grimm et al. 2005; Mina et al. 2017).

From the perspective of a potential model user, it can be difficult to decide what the appropriate model for a specific task and location should be based on its original theoretical concept. The reason is that many models, being scientific tools under permanent development rather than ready products, are continuously refined, extended or hybridized with other models (Larocque et al. 2011). Therefore, to identify strengths and weaknesses of different models, intercomparison studies are recommended (Huber et al. 2013), where multiple models are tested within a harmonized framework against independent observational data. While model comparison studies are frequently published for carbon and water flux models (e.g., Ryan et al. 1996; Hanson et al. 2004; Jin et al. 2016; Thurner et al. 2017) multi-model evaluation studies for forest ecosystem models are rare. For instance, Badeck et al. (2001) tested six gap models against observed structure and species composition of a virgin forest. Other studies comparing models originating from the same region have been published by Härkönen et al. (2010), Huber et al. (2013) and McCullagh et al. (2017). Recently, Bugmann et al. (2019) compared the behavior of mortality algorithms implemented in several forest models. However, studies comparing forest models that originate from different countries and different ecological and management contexts against long-term observational data sets from managed forests are rare (but see e.g., Mäkelä et al. 2000; Lindner et al. 2005).

Consistent data sets from managed multi-species forests that extend over several decades are rare, particularly when tree-level data are required. Usually, data from silvicultural experiments are utilized for this purpose (Mäkelä et al. 2000; Yaussy 2000; Lindner et al. 2005; Seidl et al. 2005).

When long-term observational data are used for a model evaluation study, the issue of information quality arises (Gadow 2000). Historical data from decades ago may be subject to uncertainty with regard to accuracy of measurements, and there may be gaps with regard to tree species-specific information, and unknown calipering thresholds. Moreover, usually no information about tree positions, forest structure or spatial species mixture types is available. The timing of harvests as well as the composition of harvested volume (species, dead and alive trees) may also not be known exactly. Given the relevance of legacies for future forest development, erroneous initial forest conditions may propagate over time and increase uncertainty, which limits the power of model evaluation studies.

In the FP7 project ARANGE (http://www.arange-project.eu), several forest models, originally developed for different European forest types and representing different conceptual modeling approaches, were employed to explore management alternatives for mountain forests in major European mountain ranges (Bugmann et al. 2017). In addition, multi-decade forest inventory data from the Dinaric Mountains in Slovenia were available within the project consortium. This setting provided the opportunity to compare five established models in a model intercomparison study.

Specific questions of the study were:

(1) How well can forest models be initialized with historical inventory data?

(2) How well can forest models implement historical management schemes derived from harvest records?

(3) How well do observed stand trajectories and model simulations match with regard to volume, basal area and diameter structure?

# 2 Material and methods

## 2.1 Study area

The observational time series data comes from an area near the Sněžnik Mountain (1796 m a.s.l.), in the northern part of the Dinaric Mountains, Slovenia, Europe. The Sněžnik area is a karst limestone plateau, transformed in the last glacial period. The soils are mainly chromic Cambisols and rendzic Leptosols. The climate in the northern Dinaric Mountains has Mediterranean influences, with warm summer temperatures (long-term mean from July to August is 18.3 °C at 800 m a.s.l., and 14.9 °C at 1300 m a.s.l.) and low winter temperatures (mean January temperatures -0.6 °C to -4.1 °C). Mean annual temperature ranges from 6.8 °C at 800 m to 3.1 °C at 1300 m, with annual precipitation between 1670 mm and 1930 mm, respectively. Mean summer precipitation (May to September) ranges from 650 to 740 mm. The upper timberline is located at approximately 1550 m.

Mountainous silver fir (*Abies alba Mill.*) - European beech (*Fagus sylvatica L.*) - Norway spruce (*Picea abies Karst.*) forests are the prevailing natural forest type, with frequent occurrence of sycamore maple (*Acer pseudoplatanus L.*) and wych elm (*Ulmus glabra Huds.*), while small-leaved lime (*Tilia cordata Mill.*), rowan (*Sorbus aucuparia L.*), common whitebeam (*Sorbus aria (L.) Crantz*), yew (*Taxus baccata L.*) and some other species can also be found sporadically. The first major regular utilization of these forests started in the second half of the 19$^{th}$ century when silver fir was promoted, while in line with the economic principles of that time, beech was weeded out and used for charcoal and potash production and wood distillation (Perko 2002). At the beginning of the 20$^{th}$ century, an uneven-aged single stem selection system (i.e., plenter system) was introduced (Schollmayer 1906). Due to a noticeable decrease in fir vitality and its insufficient regeneration and recruitment (Klopcic et al. 2010), a combination of single stem and small-scale irregular shelterwood system was introduced in the 1960s. Afterwards it was adapted to a more flexible, site and stand specific continuous cover system labeled “free style silviculture”, combining elements of single stem selection, irregular shelterwood and shelterwood approaches, which has been applied since then (Mlinsek 1968; Boncina 2011).

Within the Sneznik area a set of nine compartments with a total area of 60.0 ha was chosen for the model evaluation study. The sites are located apart from each other on elevations between 800 m to 1300 m a.s.l..

## 2.2 Forest inventory dataset

Along with the introduction of uneven-aged forest management at the beginning of the 20$^{th}$ century, a permanent division of forests into compartments was established and since 1912 eight forest inventories have been conducted. Before 1973 inventories were implemented by fully callipering the compartments. In 1973 and 1983, inventories were executed as full callipering of a sample of compartments, while in 1993 permanent sample plots were established and used since then. The full-callipering data were available from inventories in 1963 and a follow-up measurement in 1973 (compartment 17A) or 1983 (all other compartments). Due to low sampling densities in the 1993 and 2003 inventories, this data could not be utilized to calculate reliable values for individual compartments. Thus, for the current study, an angle-count sampling inventory (Bitterlich 1952) was conducted in 2013 on a 50 x 50 m grid to gather data compatible with the historical surveys in 1963, 1973 and 1983. Thus, compartment polygons did not change throughout the observation period starting in 1963. From all inventories, stem numbers per hectare were available per tree species in 5 cm DBH (diameter at breast height)-classes, starting at 10 cm DBH. Only live trees were recorded in the inventories. For details about the nine study compartments see Table SM1.

Depending on the site index, two sets of height functions relating tree height to DBH for individual species were assigned to the compartments to calculate initial tree height in 1963 (Table SM2).

Starting in 1963, a historical register of annual harvests per compartment was available, documenting the harvests in 5 cm DBH-classes. Table SM1 presents selected information about the harvesting activities in the forest compartments.

## 2.3 Climate data

The forest models require daily or monthly climate data to drive the simulations (see Table SM3). For each of the nine forest compartments a daily time series of climate data covering the period 1963-2013 was prepared based on the nearest grid cell (Lat. 45.625, Long. 14.375) of the E-OBS data set (van den Besselaar et al. 2011). The MT-CLIM routines (Running et al. 1987; Thornton and Running 1999) were used to adjust the E-OBS climate record for elevation, slope and aspect of the nine sites and to estimate incoming global radiation of the daylight period and vapor pressure deficit (see Thornton et al. 2000).

## 2.4 Forest models

The five models were the gap model ForClim (Bugmann 1996), the landscape model LandClim (Schumacher et al. 2004), the hybrid 3D patch model PICUS (Hönninger and Lexer 2001, Seidl et al. 2005) and the spatially explicit empirical models SAMSARA2 (Courbaud et al. 2015) and SIBYLA (Fabrika 2005). The models are briefly introduced and their key features summarized in Table SM3. For detailed descriptions, we refer to the original sources.

### 2.4.1 ForClim

ForClim is a climate-sensitive forest succession (gap) model that has been developed to simulate forest dynamics over a wide range of environmental conditions (Bugmann 1994). The model simulates establishment, growth and mortality of individual trees on small independent patches, using a minimum of ecological assumptions to capture the influence of climate and ecological processes on forest dynamics (Bugmann 1996; Didion et al. 2009b). ForClim is structured into four sub-models: weather, water, plant, and management. The PLANT sub-model is the core of ForClim, where establishment and growth of tree cohorts (i.e., trees of the same species and age) are simulated based on light availability, soil nutrients, browsing intensity and bioclimatic indices calculated within the sub-models WEATHER and WATER. Tree mortality is modeled as a combination of constant "background" mortality and a stress-induced component. The MANAGEMENT sub-model enables the simulation of a wide range of silvicultural treatments such as clearcutting, shelterwood, thinning, planting, and others. In this study, we applied ForClim version 3.0 (Rasche et al. 2011), complemented by an empirical harvesting algorithm for simulating removals of an exact number of stems for every tree species by diameter class (single stem removal; see description in Mina et al. (2017)). ForClim is currently parameterized for 31 European tree species and has been tested for the representation of natural forest dynamics of temperate forests of the Northern Hemisphere (e.g., Didion et al. 2009a).

### 2.4.2 LandClim

LandClim is a process-based forest landscape model (Schumacher et al. 2004; Schumacher et al. 2006) designed to simulate forest dynamics and disturbances at large spatial scales ($10^3$ to $10^6$ ha) over long periods of time (hundreds to thousands of years). In LandClim, landscapes are represented as a 25 x 25 m grid with specific topographic and climatic input variables for each cell. Within each cell, a simplified forest gap model (Bugmann 2001) simulates establishment, growth, competition and mortality of trees on an annual time step. Similar to ForClim, trees are simulated using a cohort approach (i.e., a computational simplification where one representative individual is simulated for all trees of the same species and age within a cell (Bugmann 1996)). Tree growth is simulated using a logistic growth equation, where species-specific maximum growth rate and size are reduced by light availability, degree-day sum and a drought index (Schumacher et al. 2004). Establishment and mortality are stochastic processes. Each year, the potential for tree establishment is determined as a function of environmental filters (i.e., available light at the forest floor, minimum winter temperature, growing degree-day sum, drought index, and browsing). Mortality probability is determined as a combination of stress, density-dependent and intrinsic mortality. LandClim can simulate management in 10-year intervals on defined management areas by selecting and removing a percentage of trees fulfilling specified DBH constraints. The model has been tested and adapted to the European Alps (Briner et al. 2013; Elkin et al. 2013; Temperli et al. 2013), North American Rocky Mountains (Schumacher et al. 2006; Schwörer et al. 2016), and Mediterranean forests (Henne et al. 2015). For this study, the individual compartments were simulated in LANDCLIM as independent entities without landscape level interactions among them to produce results comparable to the stand-level models.

### 2.4.3 PICUS

The forest model PICUS version 1.5 (Lexer and Hönninger 2001; Seidl et al. 2005; Irauschek et al. 2017a), henceforth referred to as PICUS, is a hybrid of classical gap model components and process-based stand-level NPP algorithms (Landsberg and Waring 1997). The spatial core structure of PICUS is an array of 10 x 10 m patches with vertical crown cells of 5 m in height. Interactions between patches are considered via a three-dimensional light model and spatially explicit seed dispersal. Stand-level NPP is estimated with a model of light use efficiency (Landsberg and Waring 1997), which depends on intercepted radiation, temperature, precipitation, vapor pressure deficit, soil water and nutrient supply. Distribution of assimilates to individual trees is based on the relative competitive success of the individual trees. Tree mortality depends on age and stress conditions. Natural tree regeneration considers seed production and distribution, germination and establishment. Up to the height of 130 cm seedlings are simulated with a height class approach. Beyond that threshold, they are considered as individuals in the tree population (Irauschek et al. 2017b). PICUS includes a flexible management module enabling the implementation of silvicultural treatments at tree level depending on tree attributes and patch location. The model includes 17 parameterized tree species and has been validated (Seidl et al. 2005; Didion et al. 2009a) and applied in numerous studies all over Europe (Lexer et al. 2002; Maroschek et al. 2015; Pardos et al. 2015; Zlatanov et al. 2017).

### 2.4.4 SAMSARA2

SAMSARA2 is an individual-based and spatially explicit model to simulate regeneration, growth and mortality of individual trees in mixed and uneven-aged mountain forest stands (Courbaud et al. 2015). The model builds on the theory that light interception by tree crowns is a key driver in uneven-aged stands, because they present a strong vertical heterogeneity favoring asymmetric competition between trees and between the canopy and seedlings. In SAMSARA2, competition for light within a stand is calculated based on light ray interception by tree crowns. SAMSARA2 has been calibrated empirically for silver fir and Norway spruce stands within the montane elevation belt of the Alps in France. Ecological factors other than light, such as climate and site conditions are not directly taken into account. Fertility of the site is taken into account indirectly through the value of the different demographic parameters. According to Courbaud et al. (2015) the model should be recalibrated if applied under different site conditions. Annual diameter increment of individual trees depends on their size and the amount of light intercepted by their crown during the growing season. Natural mortality depends on tree diameter and a competition index defined as the basal area of larger trees within a radius of 15 m. Seeds are produced by adult trees and germination, growth and survival of seedlings depend on the light reaching the ground calculated in the center of 25 $m^2$ cells. Trees participate in light interception above DBH of 7.5 cm. Specific management algorithms allow the simulation of detailed silvicultural strategies, varying both the characteristics of harvested trees and their spatial arrangement within a stand (Lafond et al. 2012; Lafond et al. 2014).

### 2.4.5 SIBYLA

The SIBYLA model (Fabrika 2005) is based on the SILVA simulator (Pretzsch et al. 2002). It is an empirical, distance-dependent ecological niche-based model that simulates the growth of individual trees. The expected height increment is estimated from the potential height increment of the tree and a multiplier, which characterizes the effects of competition, soil and climatic conditions (see Pretzsch and Kahn 1998). SIBYLA was parameterized and validated using forest inventory data from Germany, Switzerland and Slovakia. The model is parameterized for five main European forest tree species – Norway spruce, silver fir, Scots pine, European beech and oak (*Quercus sp.*); other species can be simulated on the basis of their ecological and morphological similarity with the aforementioned species and using calibration functions. The model consists of sub-models for mortality, competition, growth, regeneration and thinning and a stand structure generator. Growth responses to environmental drivers (growing degree-days, annual temperature amplitude (°C), mean air temperature (°C) and precipitation in the growing season (mm), De Martonne (1925) index of aridity, soil moisture and site nutrient status) were formalized according to Kahn (1994). The model can simulate several cutting and thinning techniques typically applied in Central Europe (Fabrika and Ďurský 2005).

## 2.5 Simulation scenarios

**Table 1.** Simulation scenario overview. N = total number of factorial scenario combinations. Scenarios implemented by a model are marked by a tick.

| Factor | Code | Scenario | ForClim | LandClim | PICUS | SAMSARA2 | SIBYLA |
|---|---|---|---|---|---|---|---|
| | | | N = 6 | N = 6 | N = 12 | N = 12 | N = 12 |
| Initialization of small trees | A1 | no small trees | ✓ | ✓ | ✓ | ✓ | ✓ |
| | A2 | small trees initialized | ✓ | ✓ | ✓ | ✓ | ✓ |
| Browsing by ungulates | B1 | no browsing | ✓ | ✓ | ✓ | ✓ | ✓ |
| | B2 | current browsing | ✓ | ✓ | ✓ | ✓ | ✓ |
| | B3 | 1/3 browsing | ✓ | ✓ | ✓ | ✓ | ✓ |
| Harvesting | C1 | regular logging (only living trees) | × | × | ✓ | ✓ | ✓ |
| | C2 | sanitary and regular logging | ✓ | ✓ | ✓ | ✓ | ✓ |

### 2.5.1 Small tree initialization

For trees smaller than 10 cm DBH no information was available. However, most forest models consider trees of smaller sizes (in LandClim from DBH 5 cm, ForClim and SIBYLA from height 1.3 m and in PICUS from 10 cm seedling height). To allow a harmonized initialization of those models, two scenarios were defined for small tree initialization: (i) no small trees (A1, Table 1); (ii) small trees initialized using the threshold of 130 cm seedling height (A2). For the latter, the regeneration sub-model of SIBYLA (Fabrika 2005) was employed to estimate the initial number of trees in 1963 for diameter classes not covered by the historical inventory data (DBH 0-5 cm and 5-10 cm; see details of the process in the Supplementary material) for use in all models.

### 2.5.2 Browsing by ungulates

Over the observation period (1963-2013), there was a considerable influence on regeneration by ungulate browsing (Klopcic et al. 2010). Because consistent browsing data was not available for each measurement period, browsing intensities per species over the entire simulation period were estimated based on detailed browsing inventories carried out between 1992 and 2000, expert knowledge and historical deer census data. To consider the uncertainties in browsing parameters, a total of three browsing scenarios were defined (B1, B2 and B3; see Table 1). Further details are given in Table SM4.

### 2.5.3 Harvesting

The historical register of harvests separately accounted for sanitary and regular logging. This means that harvests included, at least partly, dead or “near dead” trees (i.e. sanitary fellings). According to internal reports from the Sněžnik region, there is the tendency that in compartments with longer extraction distances less sanitary harvests are carried out due to economic reasons. This poses several problems for the specification of harvested trees in the simulations. To consider the potential effects of different modes of selecting trees for extraction, two harvesting scenarios were implemented: (i) only living trees harvested (C1); (ii) in a sanitary management scenario trees that had died in the simulations in the two years preceding a planned harvest were preferably selected for harvest (C2).

## 2.6 Analysis approach

We compared observed basal area in 1963 with the initial state of compartments in the model simulations (i.e., initial state), 1983 (except for compartment 17A, that was first remeasured in 1973; for simplicity we refer to the first remeasurement date in the results as “1983” only) and 2013 with simulated basal area using the root-mean-square error (RMSE) and Nash-Sutcliffe Efficiency (NSE).

$$RMSE = \sqrt{\frac{\sum_{i=1}^{9}(observed_i - simulated_i)^2}{9}} \qquad (1)$$

The root-mean-square error represents the square root of the quadratic mean of the differences between predicted (*simulated*) and observed basal areas for the 9 forest compartments (*i*) (Eq. 1). The RMSE is always non-negative, where a value of 0 indicates a perfect fit to the data. RMSE can be related to the standard deviation of the measured values to provide a standardized index where values less than half the standard deviation may be considered low (Singh et al. 2004).

$$ME = \frac{\sum_{i=1}^{9}(observed_i - simulated_i)}{9} \qquad (2)$$

As an additional bias metric, we show the mean error (ME) (Eq. 2).

$$\mathrm{NSE} = 1 - \frac{\sum_{i=1}^{9}(observed_i - simulated_i)^2}{\sum_{i=1}^{9}(observed_i - \bar{x}_{observed})^2} \qquad (3)$$

The Nash-Sutcliffe Efficiency is a normalized statistic that determines the relative magnitude of the residual variance compared to the variance in the measured data (Nash and Sutcliffe 1970) (Eq. 3). NSE takes on values from -∞ to 1. Values close to 1 correspond to a perfect match of data and simulations. NSE of 0 indicates that the model predictions are as accurate as the mean of the observed data and values below 0 indicate that the observed mean is a better predictor than the model. The denominator in Eq. (3) determines to some extent the calculated NSE values. Because in the current study forest compartments were managed with an uneven-aged continuous cover regime, the variation of the observed basal area values was small and consequently to achieve high NSE values difficult. The R-package “hydroGOF” (Zambrano-Bigiarini 2017) was applied to calculate NSE.

$$\mathrm{DDE} = \frac{1}{2}\sum_{j=1}^{14}\left|\frac{n\ observed_j}{N\ observed} - \frac{n\ simulated_j}{N\ simulated}\right| \qquad (4)$$

To compare observed and simulated DBH distributions, we used the Diameter Distribution Error (DDE), described as “total variation distance index” in Levin et al. (2009) (Eq. 4), where *j are* 14 diameter classes of 5 cm width (starting at 10 cm), n is the stem number per ha per diameter class, and N is the total sum of trees per ha (see also Saad et al. 2015). The sum of the differences of the relative frequency in the diameter classes is multiplied by ½ to scale the error between 0 and 100%, with optimum DDE at 0%. DDE was chosen because it is a distribution-free measure that calculates the diameter distribution error independent of the total number of trees. The diameter classes were the same as those in the inventory records.

For each compartment we calculated tree volume for the three measurement years, for all trees that had died during the simulations as well as for the harvested trees in a standardized approach by using local height curves (Table SM2) and individual tree stem volume functions (Pollanschütz 1974) to compare periodic volume growth and mortality, while avoiding possible biases due to biomass and volume estimation methods used by individual forest models only (Thurnher et al. 2013).

$$Total\ Growth_p = T\ Stock_{p(end)} - T\ Stock_{p(begin)} + \sum Harvest_p + \sum Mortality_p \tag{5}$$

The total periodic production of timber (Total Growth) is calculated with (Eq. 5). For a specific period (*p*), timber stock *(T Stock)* corresponds to the standing tree volume, *Harvest* is the standing tree volume of trees harvested in period *(p)* and *Mortality* is the standing tree volume of trees that had died during period *(p)*.

## 2.7 Simulation protocol

The modeling groups jointly defined the scenario settings (cf. Table 1) and received the same data consisting of (i) tree data to initialize simulation stands representing the forest compartments in 1963, (ii) site and soil attributes, (iii) daily weather data for the period 1963-2013, and (iv) historical harvest data for the period 1963-2013. Each group performed the simulation runs (see Table 1) and delivered the model output data in a harmonized format to the lead author, who aggregated and analyzed the data. Tables SM1 and SM2 show site attributes available to the modeling groups and aggregated information on tree and harvesting data.

# 3 Results

## 3.1 Forest model initialization performance

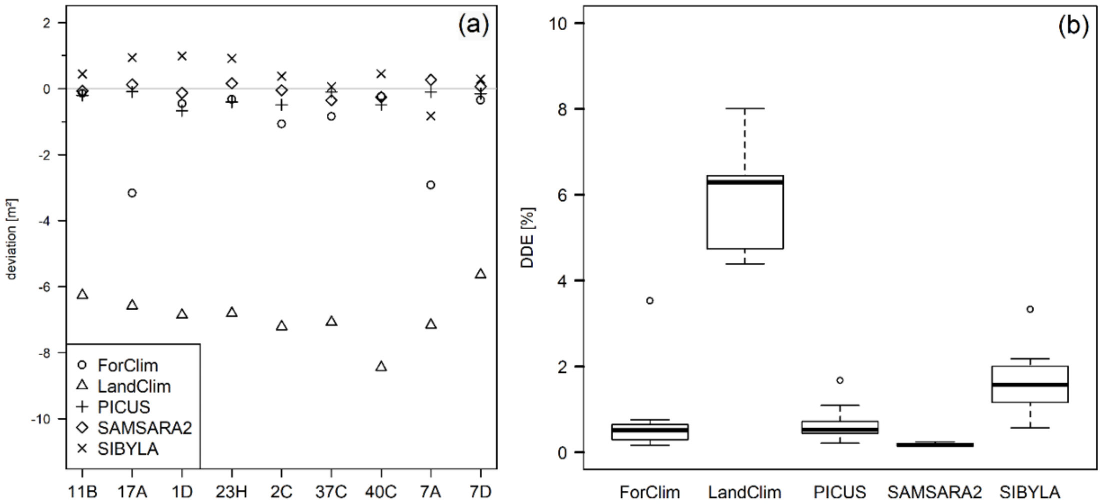


Figure 1. Statistics for initialization of nine compartments (scenario A2; small trees initialized). Left (a): Deviation of observed and initialized basal area per hectare (trees > 10cm DBH). Right (b): Diameter Distribution Error (DDE), maximum whiskers range is 1.5 times the interquartile range.

Total basal area of trees with DBH above 10 cm in 1963, as initialized in PICUS, SAMSARA2 and SIBYLA, matched the observations very well (Figure 1a). RMSE was 0.3 $m^2 ha^{-1}$ for PICUS, 0.2 $m^2 ha^{-1}$ for SAMSARA2, and 0.7 $m^2 ha^{-1}$ for SIBYLA (Table SM8). Initial basal area in LandClim showed the highest absolute deviations (RMSE 6.9 $m^2 ha^{-1}$), with lower basal area in all compartments compared with the inventory in 1963.

A comparison of initial DBH distributions in the models and the inventory in 1963 revealed a similar picture (Figure 1b). The diameter distribution errors (DDE) for SAMSARA2 were below 1% and for PICUS and ForClim below 2%, with one department as exception for ForClim with DDE over 4%. DDE values for SIBYLA were between 1.5% and 3.5% indicating good agreement with the 1963 inventory. LandClim had moderate DDE values from 4 to 9%. Inspection of LandClim diameter distributions revealed that most of the mismatch in diameter structure occurred in the high DBH classes.

## 3.2 Simulation of compartment development

Starting with the initial compartment characteristics (see 3.1 above), the models simulated forest development according to the scenarios in Table 1. In Figure 2 the simulated basal area development for each model is shown per compartment and compared to the inventory records. Most models slightly underestimated mean compartment basal area after 50 simulation years in 2013: ForClim at 27.0 $m^3$ $ha^{-1}$, SAMSARA2 26.3 $m^3$ $ha^{-1}$, PICUS 26.1 $m^3$ $ha^{-1}$ and SIBYLA at 25.5 $m^3$ $ha^{-1}$ versus the mean inventory value of 28.4 $m^3$ $ha^{-1}$. On average LandClim slightly overestimated basal area at the end of the simulation period in 2013 (29.3 $m^3$ $ha^{-1}$). The mean error (ME) over all scenarios and compartments for the models was -3.0 $m^3$ $ha^{-1}$ (ForClim), -3.2 $m^3$ $ha^{-1}$ (PICUS), -1.6 $m^3$ $ha^{-1}$ (SAMSARA2), -2.5 $m^3$ $ha^{-1}$ (SIBYLA) and +3.2 $m^3$ $ha^{-1}$ for LandClim. More details on simulated basal area are shown in Figure 3.

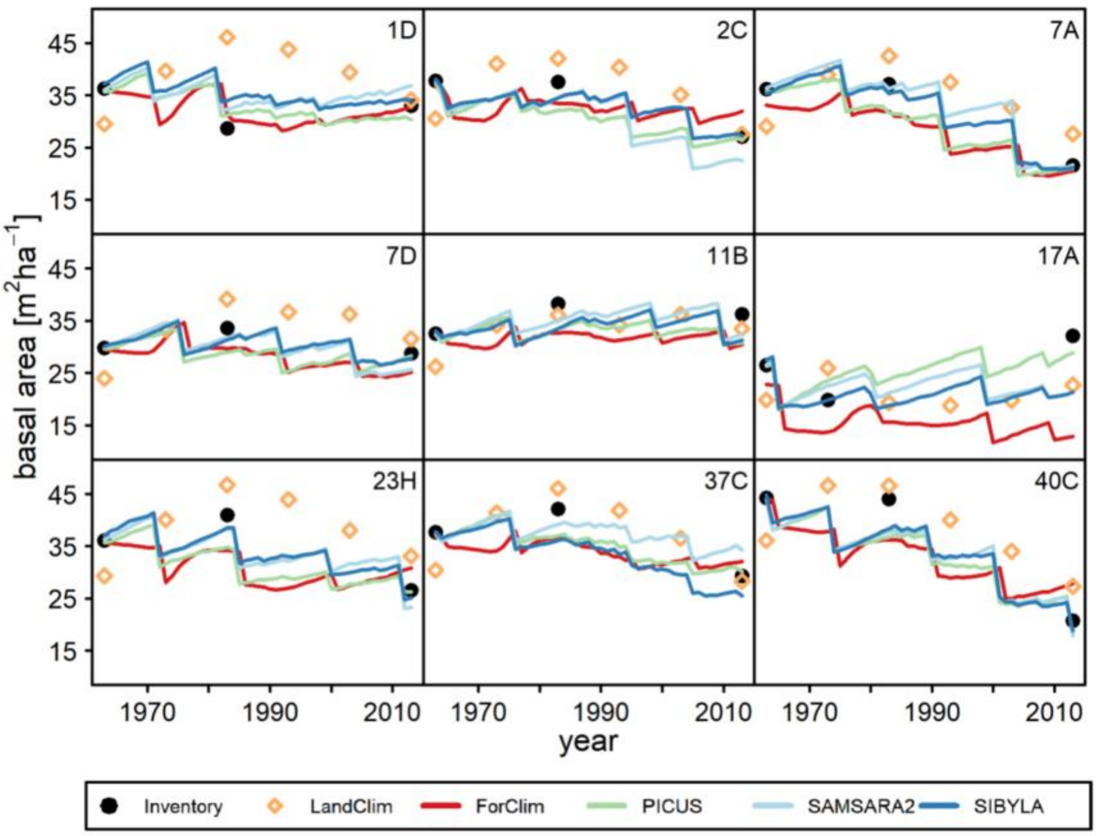


**Figure 2.** Trajectories of simulated basal area development from 1963 to 2013 showing the scenario with the lowest error (NSE) in 2013 for each model. ForClim: A1 B1 C2, LandClim: A1 B1 C2, PICUS: A2 B3 C2, SAMSARA2: A2 B2 C1, SIBYLA: A1 B3 C1. Black dots denote inventory records.

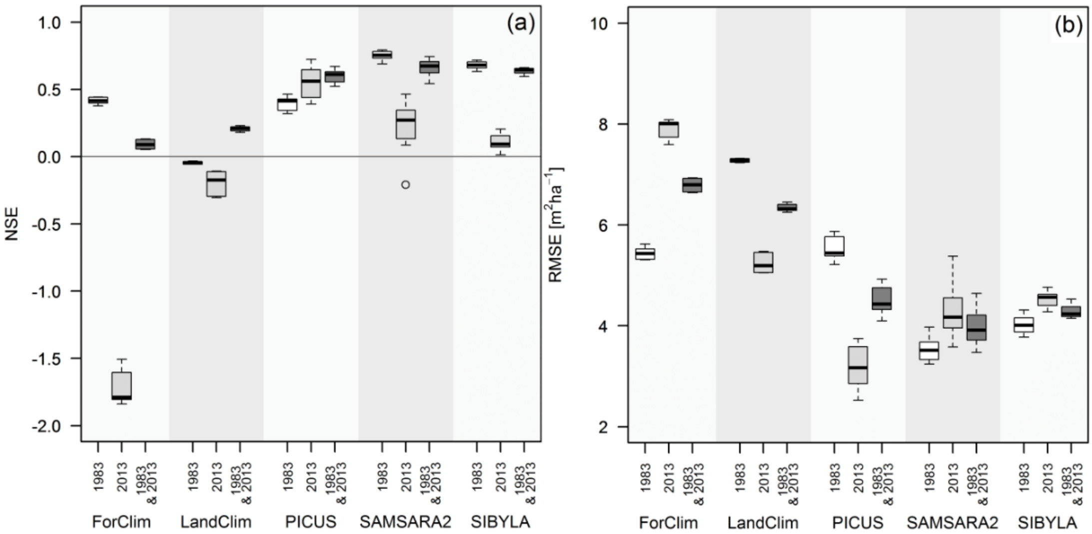


**Figure 3.** Error indicators for basal area per model in 1983, 2013, and cumulated for both periods (1983 & 2013) over all compartments and scenarios. **Left (a)**: Nash-Sutcliffe Efficiency (NSE). **Right (b)**: Root Mean Squared Error (RMSE). N = 6 (scenarios).

Overall, SAMSARA2, SIBYLA and PICUS showed good model performance (NSE > 0.5; mean RMSE < 5 m$^2$ ha$^{-1}$) with PICUS being the only model performing well in both observation periods. LandClim had negative NSE values in both periods, meaning that the mean of the observations was a better predictor than the model. ForClim had the worst of all observed NSE values in 2013 (ranging from - 1.5 to -1.8), but NSE was highly influenced by poor performance in compartment 17A. Interestingly, this was the only compartment dominated by beech, while in the other compartments silver fir was dominating. Also other models (SIBYLA and SAMSARA2) showed the largest deviations from the observed values in compartment 17A for the observations in 2013.

It is interesting to look at the sensitivity of the models to the scenario assumptions. Overall, the variation in model output, as triggered by scenario A (initialization) and B (browsing), was smallest for LandClim (2% of the mean result) and largest for SAMSARA2 (8%); relative range in output for ForClim, PICUS and SYBILA was approx. 4 to 6% (Figure 4). Adding scenarios C (harvesting mode) increased the variation to 24% (SAMSARA2), 11% (PICUS) and 6% (SYBILA) of the mean output (see also Figure SM1 and Table SM8).

Comparing simulated and observed diameter distributions at different time points is a rigorous test of the ability of the models to project forest structure over the 50-year period. Both in 1983 and 2013, LandClim showed the largest mean DDE value (23.6 %, 69.8 %) and ForClim the lowest (13.7 %, 29.8 %), with SYBILA, SAMSARA2 and PICUS having similar DDE values as ForClim. For all models DDE increased from 1983 to 2013 (Figure 5).

What was the scenario that produced the best outcome with each of the five models? To answer this question the mean NSE and RMSE related to basal area over the entire simulation period were calculated for each simulation scenario and for each model the scenario with the maximum NSE was selected as "best" scenario" (Table SM6). In a similar procedure, mean DDE statistics were calculated for the entire simulation period and the scenario with the lowest DDE per model was selected (Table SM6). Regarding the simulated diameter distributions, scenario A2, which included small trees in the initialization, resulted in more accurate simulated diameter distributions for all models. The scenario assumptions that produced the best overall match with regard to basal area differed among the five models. While for ForClim and LandClim “no browsing” (B1) combined with “sanitary management” (C2) produced better results, for PICUS, SAMSARA2 and SIBYLA the browsing pressure as observed in 2013 (B2) in combination with sanitary management (C2) resulted in more accurate results. For all models the two error statistics NSE and RMSE consistently ranked the same scenario at the top.

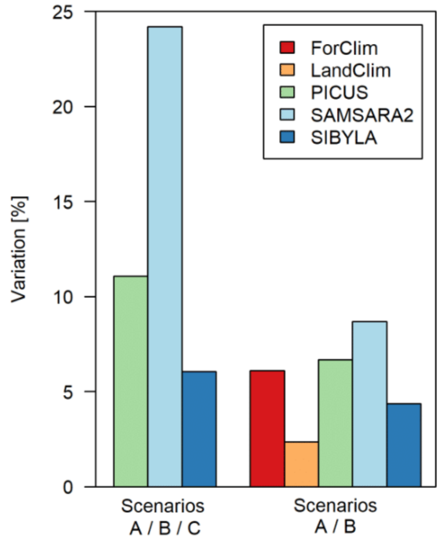


**Figure 4.** Variation in basal area in the year 2013 as simulated in the scenarios from Table 1. Mean range of scenario results normalized by the mean scenario result for 2013. Scenario codes: A = Initialization of small trees, B = Browsing by ungulates, C = Harvesting.

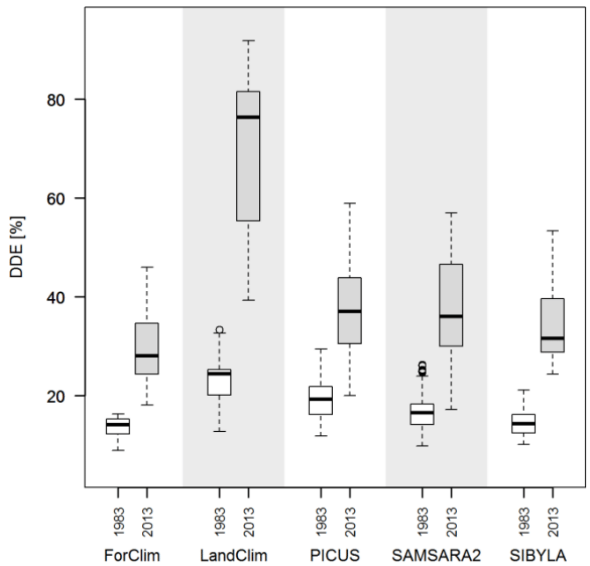


**Figure 5.** Diameter distribution error (DDE) per model in 1983 and 2013 for all simulated scenarios and compartments.

## 3.3 Realization of harvests

According to the harvest records, the compartments were treated regularly in an interval of 10 to 20 years. Moreover, in some compartments sanitation logging occurred almost every year (e.g., several operations in compartment 40C between 1985 and 1990 or in compartment 2C in the years 1980 to 1995) (see Figure 2). When comparing the diameter structure of simulated harvests with the records, all models except LandClim performed well (not shown) with SYBILA showing the best match of simulated and observed diameter distribution of harvested trees. For all models, deviations occurred mainly in the smaller diameter classes below 20 cm DBH.

## 3.4 Total volume production

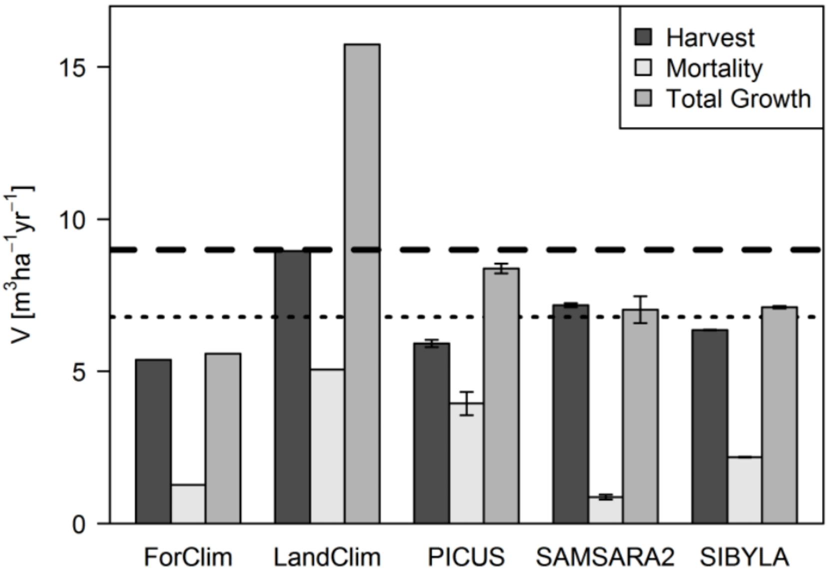


**Figure 6.** Simulated yearly *Harvests*, *Mortality* and calculated *Total Growth* for the period 1963-2013 (mean over 9 compartments). *Dashed line* shows regional Total Growth estimate, *dotted line* shows historical harvests (mean over 9 compartments). For ForClim and LandClim bars show the results for scenario A2-B2-C2 (small tree initialized, current browsing, sanitary and regular logging). For PICUS, SAMSARA2 and SIBYLA bars show mean values for scenario A2-B2-C1 (regular logging) and A2-B2-C2 (sanitary and regular logging), the whiskers indicate results for C1 and C2.

Utilizing the harvest records, the periodic harvest volumes *(Harvest)* can be calculated for the historical period, while tree *Mortality* which has not been extracted and thus is not included in *Harvest* is not known. Consequently, no observed *Total Growth* could be calculated for the nine study compartments. However, for comparison we could retrieve the local total mean annual volume growth from forest management plans (about 9 $m^3$ $ha^{-1}$ $yr^{-1}$). This general estimate can be compared with *Total Growth* according to Equation 4 for each model simulation. From Figure 6 it is evident that there are differences among the models. Compared with the local estimate, ForClim, SAMSARA2 and SIBYLA simulated somewhat lower *Total Growth* (5.6, 7.0, 7.1 $m^3ha^{-1}yr^{-1}$), LandClim is clearly overestimating (15.7 $m^3ha^{-1}yr^{-1}$), and PICUS is closest to the reference value of 9 $m^3ha^{-1}yr^{-1}$ (8.4 $m^3ha^{-1}yr^{-1}$). SAMSARA2 and SIBYLA reproduced the recorded harvest volumes accurately. PICUS and ForClim show only minor underestimates, whereas LandClim clearly overestimated *Harvests*. Large differences between the models are visible regarding *Mortality*. Here, ForClim, SAMSARA2 and SIBYLA clearly simulated much lower tree mortality (ForClim 0.9 $m^3ha^{-1}yr^{-1}$, SAMSARA2 1.3 $m^3ha^{-1}yr^{-1}$ and SIBLYA 2.2 $m^3ha^{-1}yr^{-1}$) than LandClim (5.1 $m^3ha^{-1}yr^{-1}$) and PICUS (3.9 $m^3ha^{-1}yr^{-1}$). The effect of the sanitary management scheme, causing a shift of volume from *Mortality* to *Harvest*, is small. Details for both inventory periods are shown in Table SM7.

# 4 Discussion

### Assessment approach

One major challenge for increasing the acceptance of simulation tools for decision support is the objective validation of models, to build trust in the validity of model output (Muys et al. 2011). Multi-model evaluation exercises offer a great opportunity to compare detailed model outputs and point out differences, strengths and deficiencies. Recently studies have started to quantify parametric uncertainty (Hartig et al. 2012), and some results exist on the contributions of model sub-processes to overall parametric uncertainty (Augustynczik et al. 2017), but the quantification of structural uncertainty is less advanced.

A prerequisite for multi-model evaluation studies is that the participating model groups are provided with identical site, stand and climate information and follow a harmonized protocol that avoids specific fitting of model settings to local conditions. In a comparative analysis of 15 forest models with regard to model sensitivity to different tree mortality algorithms, Bugmann et al. (2019) refrained from having all participating models run under identical conditions. However, this approach adds another source of uncertainty and makes it even harder to track those model sub-processes that contribute most to the variation in model output and to respond to the question of why a specific model does better or worse in a specific situation.

The historical long-term compartment-based dataset, in combination with management records as used in this study, offered great potential to test models in species-rich forest ecosystems in realistic management contexts. Inventories carried out by forest enterprises usually do not fulfill the criteria for long term model evaluation exercises. Here the focus usually lies on cost efficiency and less on continuous long-term datasets. Methodologies usually switched from full calipering of trees over wide forest areas, to statistically more advanced and less cost intensive sampling methods starting from the 1950s onwards. As a result, continuous datasets consisting of tree data with sufficient sampling densities over a long time period are very scarce. The tree and harvest data from the presented case study cover a remarkably long period of 50 years. They represent entire compartments and are thus better representatives of real forests compared to small plots of large-scale forest inventories (see Huber et al. 2013). However, to fully understand forest development in the nine compartments, information on tree mortality, stand structural information and tree regeneration would be essential. In an attempt to frame sources of uncertainty in the evaluation data, 12 scenarios were defined to specify explicit assumptions on the initial state of regeneration, browsing pressure by ungulates over the monitoring period and the inclusion of tree mortality in harvesting decisions (Table 1). What could be considered a weakness of the evaluation data was turned into a strength of the assessment approach because it added rigor to the comparison of model output and observations. It also facilitated analysis of the sensitivity of the models to assumptions that have to be made quite often, when data and model input from decades ago are used.

### Model initialization

A first crucial step in practical model applications is to initialize stand structures from available information. To our knowledge, the ability of models to create realistic initial tree populations has not been considered before. All models involved in this study include special routines for initialization to avoid stand structures that are not compatible with internal processes dealing with competition for resources and space and may therefore result in undesired behavior of simulated tree populations, especially in the first simulation years (see Supplementary material). Generally, these routines are most important for complex forest stands with high tree density, vertical tree layers, diverse species mixtures, many size classes and patchy spatial structure. In the model SIBYLA, for example, the models internal consistency of the virtual tree population is evaluated by estimating the suitability of the tree positions according to the nearest and second nearest neighbor trees using a probabilistic approach (Pretzsch 1993). For rejected trees, new coordinates are generated until microstructural requirements are fulfilled. The PICUS model operates in a similar way within its 10 x 10 m resolution to avoid the initialization of non-viable tree neighborhoods. The gap model ForClim uses an algorithm to optimize the leaf area of virtual trees to avoid excessive shading. SAMSARA2 avoids initialization problems by using a uniform distribution for estimating coordinates, which may visually result in very regular stand structures. In LandClim, compartments are initialized spatially explicit at the level of the simulation cells (25 by 25 m). The effort that has been put into generating realistic initial forest states shows that, although so far not much attention has been paid to this issue, (i) creating realistic tree population structures may have crucial implications for simulated short- to midterm population behavior, and (ii) indicates the relevance of structural information and tree coordinates (see Table SM3). For instance, PICUS, SAMSARA2 and SIBYLA could have directly utilized tree coordinates for initializing forest stands. All models except LandClim were able to generate realistic forests regarding basal area density and diameter structure (compare Figure 1 and Table SM8 for an overview). The landscape model LandClim, which had not been developed to simulate fine-grained tree populations at population level, initialized forests with substantially lower basal area compared to observations and also did not match well the initial diameter structure of the validation data set.

## Forest simulation results

In simulating managed mixed mountain forests over five decades, beyond growth, regeneration and mortality of trees, also harvests and the response of tree populations to these human disturbances need to be mimicked realistically. The results showed that spatially explicit individual tree-based models performed better in simulating basal area development than the non-spatial gap model ForClim, and, not surprisingly, the landscape model LandClim. Overall, the best performing models in our study yielded RMSE values for basal area over a simulation period of 50 years, which are about 10 to 12% of the compartment basal area. This simulation result is comparable to the range of basal area mean error values from large-scale forest inventories (e.g., Næsset 2007). The mean error (ME) values over all scenarios indicated that all models on average underestimated the observed basal area values by about 10% of the mean observed values, except the LandClim model which yielded a slightly larger positive bias.

An even greater challenge for forest models is to capture the development of the size structure of the tree population. With the exception of LandClim, all models performed reasonably well. Furthermore, the study did not confirm two general beliefs: First, and interestingly, empirical models (SYBILA, SAMSARA2) did not perform significantly better regarding the accurate prediction of DBH structure than the process-based models ForClim and PICUS (compare Guisan and Zimmermann 2000; Fontes et al. 2011). A similar finding for the PICUS model had already been reported in Seidl et al. (2005). Second, none of the models had been developed for Dinaric mountain forests. As the model evaluation protocol avoided the calibration of the tested models to local tree growth data, the expectations towards accurate model results in simulating 50 years of tree growth were low. However, the stand-level models performed reasonable (compare Figure 7). The spatially explicit models PICUS, SAMSARA2 and SYBILA performed overall better than the non-spatial gap model ForClim and showed very concordant results. On the other hand, ForClim showed the best performance regarding the diameter distributions.

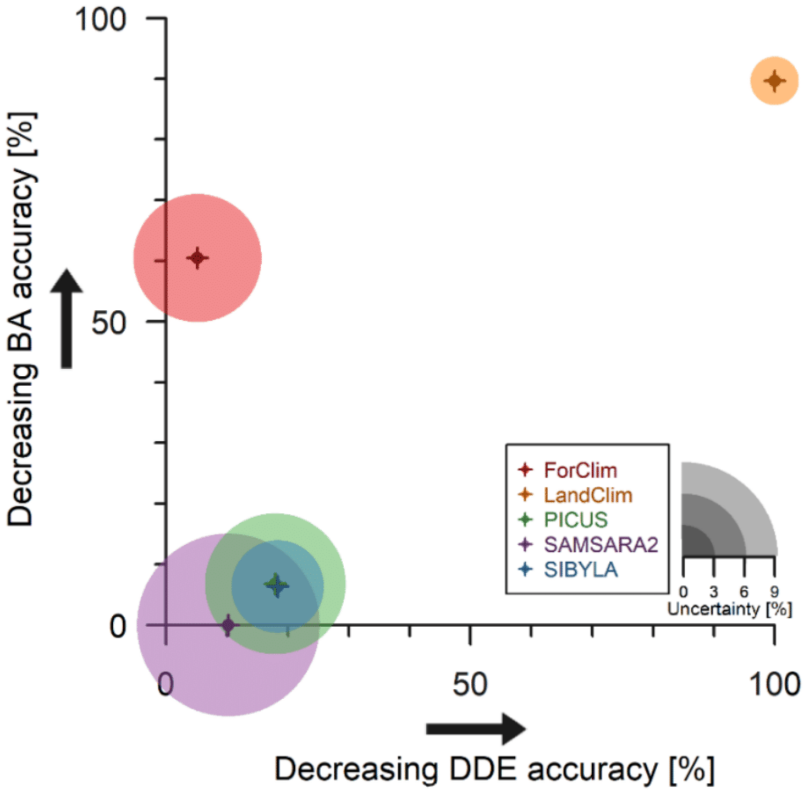


**Figure 7.** Relative model performance scaled to the respective maximum and minimum values. Maximum BA accuracy and DDE accuracy in origin (0/0). BA accuracy = sum of relative RMSE for initialization and Nash-Sutcliffe Efficiency (NSE) for simulation results (pooled 1983 and 2013). DDE accuracy = sum of relative diameter distribution error (DDE) for initialization and simulation results (mean 1983 and 2013). The circle diameters correspond to the relative variation resulting from scenario assumptions (results 2013, scenarios A (Initialization of small trees) and B (Browsing by ungulates). Data in Table SM8.

The current study clearly showed that a model should perform well in a series of aspects beyond tree growth that drive forest development. Over the 50-year simulation period, the interacting processes tree growth, regeneration, mortality and tree selection and harvesting determine the standing stock as well as the DBH distribution at a specific point in time. Hülsmann et al. (2018) point out that mortality subroutines in forest models are typically rather fundamental and lack empirical basis. In a recent study Bugmann et al. (2019) concluded that the sensitivity of forest models to changes in mortality algorithms is larger than the sensitivity to climate change signals. Unfortunately, the testing data used for our study did not include tree mortality. The extent to which harvests included a share of natural tree mortality was also not known. However, the scenarios C1 (harvests include only living trees) and C2 (sanitary management, harvests also include dead trees) allowed to estimate the effect of the underlying assumptions. Our results showed that a seemingly simple process such as harvesting trees might produce quite different results among the models.

All models except LandClim could handle harvest prescriptions on an annual basis. However, harvest algorithms within the models differ in the way they handle tree removals in specific DBH-classes. Some algorithms search for specific trees (size and species) in the simulated forest. As differences between observed and simulated DBH-structures increase with increasing simulation length, more flexible algorithms (e.g., trees may be taken from neighboring DBH-classes) may be beneficial, especially for very detailed harvest operations such as single-tree or sanitary management records (Mina et al. 2017). Our results indicated that harvest algorithms incorporating multiple criteria, such as basal area, tree diameter and diameter distributions, can improve overall modeling accuracy (Lafond et al. 2012). The importance of consistently considering natural tree mortality in harvest algorithms will even increase with increasing relevance of disturbances in future climates (Seidl and Rammer 2017).

Framing the uncertainty in the evaluation data (initialization of small trees, browsing by ungulates, tree selection for harvesting) by means of 12 scenarios proved to be an adequate approach to (i) consider data uncertainty, and (ii) test the sensitivity of the models towards the scenario assumptions (see Figure SM1 and Tables SM5 and SM6). It was interesting to note that the browsing scenarios (B1, B2, B3) had a strong influence on the predictions, and were predominantly affecting diameter structures, whereas basal area was less affected. SAMSARA2 seemed to be particularly sensitive to the harvesting assumptions.

What is a useful threshold for model sensitivity? This question cannot be answered easily. As a general rule, the sensitivity of a model to a driving gradient must allow to reproduce observable data along the gradient (e.g., Lexer and Hönninger 2004). Interestingly, the average variation, caused by the full set of scenarios (as simulated by SAMSARA2, PICUS and SIBYLA), is approximately in the same order of magnitude as the error in basal area or volume estimates of large-scale forest inventories.

LandClim, the only landscape model included in the comparison set, delivered satisfactory estimates of basal area development per compartment, though it operates on coarser resolutions regarding initialization, management and growth simulation, and was designed to simulate landscape level disturbances and processes that were not the focus of this study. The advantage of landscape models is that they can be used to complement stand-level models to explore forest development at multiple spatial scales from stand to landscape scale, which is gaining importance with growing confidence that disturbance regimes will intensify under a warming climate. Interacting disturbance agents operate at multiple spatial scales, which is beyond the reach of stand-level models (e.g., Elkin et al. 2013; Hlásny et al. 2019).

# 5 Conclusions

Several conclusions can be drawn from this study:

First, a strong effort should be made by research institutes and forest owners, to access and store historical inventory datasets and harmonize contemporary inventories with historical data to be able to capitalize on the benefits of a long consistent time series. In particular, detailed harvest records are crucial to understand time series data of stocks, which are the usual monitoring focus. Knowing harvested trees by tree species and by status (live, dead) would be a huge leap forward for further development and testing of forest ecosystem models.

Second, there is no clear hierarchy between more empirical or more process-based models regarding the accuracy of stand development projections.

Third, individual-based models are more precise for stand-level predictions than the tested landscape model LandClim, but the latter performed reasonably well at the stand scale, qualifying it for a complementary application at the landscape scale. This serves as a great example for a finding from a model intercomparison study, where the ultimate goal is not that every contestant shows excellent performance regarding standard outputs. More important is a critical comparison between models to differentiate why certain models are better under specific conditions observed in the case study for specific outputs. Following a detailed description of underlying model assumptions and objective evaluation of outputs, recommendations can then be given regarding the specific applicability of models and their limitations. However, model performance demonstrated in a specific ecological and management setting should not be generalized carelessly to diverging conditions. Repeated successful model evaluation experiments in different settings will build trust in model performance.

Fourth, in model evaluation studies a comprehensive set of model output variables should be assessed. There is a fast-growing demand by decision makers for reliable prediction of stand structural features beyond volume or basal area, because for forest management planning ecosystem service provisioning is based on indicators on species composition and structure of the tree population (e.g., Maroschek et al. 2015). The current study has shown that there are forest models available that qualify for forest management decision support.

Fifth, as a very relevant mission joint model intercomparison and evaluation studies compare currently available ecosystem models by challenging them with applications in novel case study settings and foster scientific collaboration among the modeling groups. This prepares the ground for multi-model applications in decision support, a prerequisite to quantify and explain uncertainty from model assumptions.

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## 7 Credit Author Statement

**Florian Irauschek:** Writing - original draft, Methodology, Formal analysis, Visualization, Simulation. **Ivan Barka:** Simulation, Writing - Review & Editing. **Harald Bugmann:** Conceptualization, Writing - Review & Editing. **Benoit Courbaud:** Simulation, Writing - Review & Editing. **Che Elkin:** Simulation, Writing - Review & Editing. **Tomás Hlásny:** Simulation, Writing - Review & Editing. **Matija Klopcic:** Data Curation, Writing - Review & Editing. **Marco Mina:** Simulation, Writing - Review & Editing. **Werner Rammer:** Simulation, Visualization, Writing - Review & Editing. **Manfred J Lexer:** Conceptualization, Methodology, Writing - Review & Editing.

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# 10 Supplementary Material

## Details on model initialization

### SIBYLA

For stand initialization in SIBYLA the module "Generator" (Fabrika 2005) was used, which is a modified version of the STRUGEN module from the model SILVA (Pretzsch 1993). First, measured DBH-classes were transformed into 4 cm DBH-classes required by the model. A uniform distribution was used to estimate DBH of the single trees. Tree heights were calculated using the height curves provided by model operators. Tree crown diameter and height to crown base were generated according to Pretzsch (2001). Tree position maps were generated in two steps: First, trees were randomly distributed at the plot. Then, this initial distribution was evaluated by estimating the plausibility of the positions according to the nearest and second nearest neighboring tree using an approach proposed by (Pretzsch 1993). For discarded trees, new coordinates were generated and the previous procedure was applied iteratively. Each compartment was simulated in the given total area, but due to computational limitations of SIBYLA, each compartment was split into several plots with an area of one hectare. Harvesting operations were performed using external database operations, removing a specified number of trees within the prescribed DBH-classes.

### PICUS

PICUS simulated an area of 3 ha for each compartment (20 x 15 patches with 100 m$^2$ each). Initial stem numbers and harvesting records in DBH-classes where converted. Fractional tree numbers (e.g., 0.3 trees for a DBH-class) were considered using a random number to avoid biases due to rounding. Single trees were generated by assuming uniform DBH distribution within the 5 cm classes and subsequent estimation of tree height with the provided DBH-height-functions. Initially, trees are located randomly and tree crown lengths are estimated according to the PICUS 3D-light environment (Lexer and Hönninger 2001). For generating optimized tree positions, an algorithm was used, where the light modifier of PICUS is utilized to check whether a tree can survive within its neighborhood. In an iterative process, trees of the largest diameter classes are relocated and died trees are “refilled” at random positions to find suitable stand structures.

### ForClim

In FORCLIM the stands were initialized using DBH and height formulas, allocating each tree randomly to the simulated number of patches obtained by dividing site area by the default patch size (i.e., 800 m$^2$). In the simulation runs 200 patch replicates were simulated per compartment (Didion et al. 2009a), adding up to a total simulated area of 16 ha per compartment (Mina et al. 2017).

### SAMSARA2

The size of a SAMSARA2 simulation unit is 4 ha. For each compartment, a list of trees corresponding to the species and DBH distribution was created. Tree DBH was drawn from a uniform distribution for each DBH class. Individual tree coordinates were drawn from uniform distributions.

**LandClim**

In LandClim a contiguous landscape of 60 ha was simulated, representing the forest compartments in fixed 25 by 25 m cells. The state of each grid cell was represented by the number and biomass of trees in cohorts (individuals in 10-year age classes for each species). To generate the initial state per cell, the inventory data was converted deterministically to stem count per cell. Then a DBH was drawn from DBH-distributions developed for each species per compartment. In the end, DBH was converted to biomass and age was assigned according to provided functions.

**Table SM1.** Site information, initial inventory information in 1963 and harvesting records for the period 1963 to 2013 for the simulated forest compartments sorted by elevation. Site Index = height at age 100 for silver fir, Elev = mean elevation above sea level, Exp = Exposition in 4 cardinal and 4 intercardinal directions, Depth = Soil depth, Coarse = Volume fraction of coarse stones, Nav = plant available Nitrogen, WHC = water holding capacity, ΣHarvest = sum of tree volume harvested from 1963 to 2013, Harvest = number of recorded harvest activities 1963 to 2013, MinHarvest = minimal volume harvested, MaxHarvest = maximal volume harvested. pH of mineral soil was 6.5 for all compartments.

| | Site data | | | | | | | | | Inventory 1963 | | | Harvest Records 1963-2013 | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Compart-ment | Area [ha] | Site Index [m] | Elev [m] | Slope [°] | Exp | Depth [cm] | Coarse [%] | Nav [kg $ha^{-1} yr^{-1}$] | WHC [mm] | Basal Area [$m^2$ $ha^{-1}$] | Volume [$m^3$ $ha^{-1}$] | Stems [N $ha^{-1}$] | ΣHarvest [$m^3$ $ha^{-1}$] | Harvest [N] | Min Harvest [$m^3$ $ha^{-1}$] | Max Harvest [$m^3$ $ha^{-1}$] |
| 40C | 6.87 | 33 | 815 | 5 | S | 30 | 25 | 68 | 98 | 44.3 | 495 | 555 | 605.4 | 20 | 0.2 | 147.2 |
| 2C | 7.81 | 33 | 825 | 20 | N | 30 | 25 | 68 | 98 | 37.8 | 414 | 555 | 351.4 | 23 | 0.3 | 83.7 |
| 37C | 7.06 | 33 | 860 | 10 | S | 30 | 25 | 68 | 98 | 37.7 | 410 | 556 | 322.0 | 26 | 0.3 | 71.0 |
| 23H | 5.46 | 33 | 935 | 5 | SW | 30 | 25 | 68 | 98 | 36.1 | 383 | 616 | 385.4 | 19 | 0.0 | 97.4 |
| 7A | 5.17 | 33 | 965 | 25 | NW | 25 | 30 | 66 | 85 | 36.1 | 425 | 414 | 344.2 | 14 | 0.8 | 109.4 |
| 1D | 9.00 | 33 | 968 | 20 | E | 30 | 25 | 68 | 98 | 36.2 | 376 | 642 | 263.6 | 21 | 0.3 | 75.0 |
| 7D | 5.99 | 33 | 1045 | 20 | NE | 25 | 30 | 66 | 85 | 29.8 | 332 | 460 | 237.5 | 10 | 0.3 | 76.8 |
| 11B | 6.93 | 33 | 1205 | 5 | N | 25 | 30 | 66 | 85 | 32.6 | 344 | 571 | 262.6 | 13 | 0.6 | 91.2 |
| 17A | 5.71 | 22 | 1290 | 20 | NE | 25 | 30 | 66 | 85 | 26.5 | 262 | 889 | 278.4 | 5 | 13.1 | 113.0 |

**Table SM2.** Height functions dependent on diameter at breast-height for tree species, as employed in model initialization, depending on site index of the compartments (see Table SM1). h = tree height [m], DBH = diameter at breast-height [cm].

| Site Index | Species | Height Function |
|---|---|---|
| 33 | silver fir<br>Norway spruce<br>other conifers | $\boldsymbol{h = 1.3 + \frac{(DBH + 0.5)^2}{22.4816 + 0.0430 \times (DBH + 0.5) + 0.0253 \times (DBH + 0.5)^2}}$ |
| 33 | European beech<br>acer pseudoplatanus<br>Wych elm<br>other broadleaved | $\boldsymbol{h = 1.3 + \frac{(DBH + 0.5)^2}{9.6300 + 0.0812 \times (DBH + 0.5) + 0.0313 \times (DBH + 0.5)^2}}$ |
| 22 | silver fir | $\boldsymbol{h = 12.1508 \times ln\,(DBH) + 0.0002135 \times DBH^2 - 19.794}$ |
| 22 | Norway spruce<br>other conifers | $\boldsymbol{h = 12.7618 \times ln\,(DBH) + 0.0004455 \times DBH^2 - 22.117}$ |
| 22 | European beech | $\boldsymbol{h = 9.2540 \times ln\,(DBH) + 0.0005268 \times DBH^2 - 9.863}$ |
| 22 | acer pseudoplatanus<br>wych elm<br>other broadleaved | $\boldsymbol{h = e^{0.8525 + 0.7076 \times ln\,(DBH) - 0.0001525 \times DBH^2}}$ |

**Table SM3.** Summarized features of the compared forest ecosystem models. rep.= Simulation repetitions, Temp = Temperature, Precip = Precipitation, VPD = Vapor Pressure Deficit, Rad = Radiation, NA = Species not initialized, BL = broad-leafed

| | **ForClim** | **LandClim** | **PICUS** | **SAMSARA2** | **SIBYLA** |
|---|---|---|---|---|---|
| **model type** | gap model | gap model | hybrid<br>process based x gap model | process based model | empirical model |
| **spatial resolution** | single trees<br>0.08 ha patches | tree cohorts<br>25x25 m grid-cells | single trees<br>10x10 m patches | single trees<br>5x5 m grid-cells | single trees<br>distance dependent |
| **simulation setup** | independent patches<br>9 x (0.08 ha x 200 rep.) | contiguous landscape<br>60 ha | contiguous stands<br>9 x 3 ha | contiguous stands<br>9 x (4 ha x 5 rep.) | contiguous stands<br>9 x variable (original) ha |
| **input data** | soil data<br>monthly climate:<br>Temp, Precip | soil data<br>monthly climate:<br>Temp, Precip | soil data<br>monthly climate:<br>Temp, Precip, VPD, Rad | monthly climate:<br>Rad | soil data<br>yearly climate:<br>Temp, Precip |
| **harvesting process** | per stand:<br>single tree selection<br>from 5cm DBH-class | per grid cell:<br>criteria: stem number<br>and size constraints | per stand:<br>single tree selection<br>from 5cm DBH-class or<br>neighboring DBH-class | per stand:<br>criteria: stem number,<br>volume and DBH-<br>distribution | per stand:<br>single tree selection<br>from 4cm DBH-class |
| **regeneration process** | from tree size<br>ingrowth at DBH 0 cm | from tree size<br>ingrowth at DBH 7.5 cm | from seed<br>ingrowth at DBH 0.5 cm | from seed<br>ingrowth at DBH 7.5 cm | from tree size<br>ingrowth at DBH 0 cm |
| **Used calibrated tree species** | silver fir<br>Norway spruce<br>European beech<br>sycamore maple | silver fir<br>Norway spruce<br>European beech<br>sycamore maple<br>wych elm | silver fir<br>Norway spruce<br>European beech<br>sycamore maple | silver fir<br>Norway spruce<br>European beech | silver fir<br>Norway spruce<br>European beech |
| **Replaced tree species** | wych elm = NA<br>other BL trees = lime<br>other conifers = yew | other BL trees = beech<br>other conifers = spruce | wych elm = maple<br>other BL trees = maple<br>other conifers = yew | sycamore maple = beech<br>wych elm = beech<br>other BL trees = beech<br>other conifers = silver fir | sycamore maple = beech<br>wych elm = beech<br>other BL trees = beech<br>other conifers = silver fir |

**Table SM4.** Yearly browsing probabilities per species for trees from seedling until height of 1.5 m.

| Species | > 0 cm height | > 15 cm height | > 30 cm height |
|---|---|---|---|
| silver fir | 0.09 | 0.46 | 0.42 |
| Norway spruce | 0.07 | 0.12 | 0.10 |
| European beech | 0.26 | 0.28 | 0.34 |
| sycamore maple | 0.54 | 0.77 | 0.85 |
| wych elm | 0.64 | 0.68 | 0.78 |
| other broadleaves | 0.51 | 0.74 | 0.80 |
| all species | 0.35 | 0.45 | 0.44 |

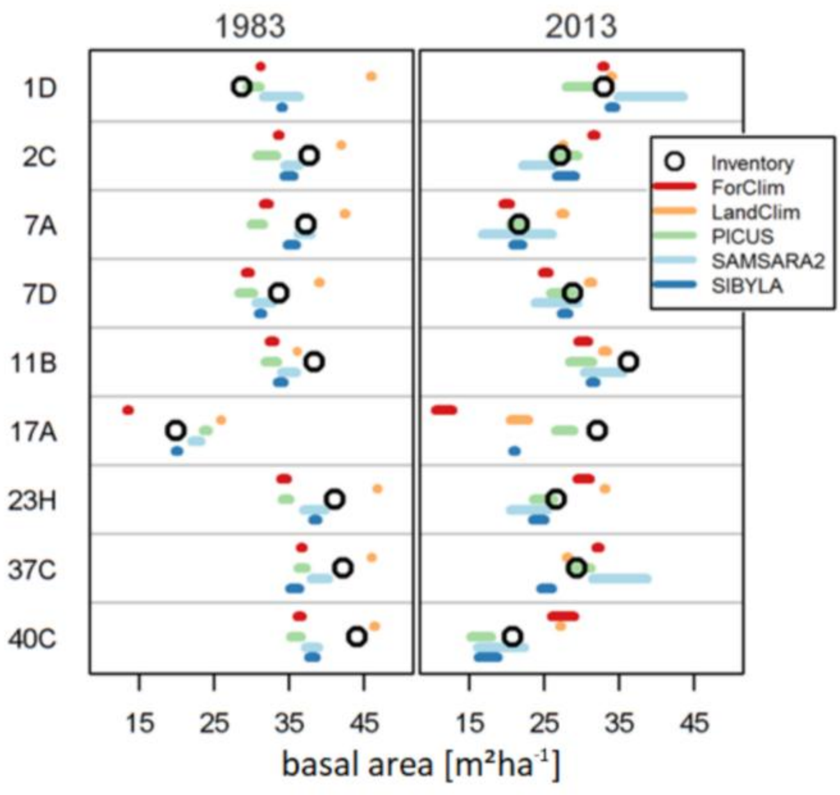


**Figure SM1.** Basal area as simulated in the scenarios. Range (min, max) of basal area per compartment for 1983 and 2013.

**Table SM5.** Basal area error index statistics per model for simulated scenarios in 1983, 2013, and for the pooled results 1983 and 2013. "mean", "min", "max" are the respective mean, minimum and maximum values per model. Q25%, Q50% and Q75% are the respective quantiles.

| | | NSE | | | | | | RMSE | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | mean | min | max | Q25% | Q50% | Q75% | mean | min | max | Q25% | Q50% | Q75% |
| ForClim | 1983 | 0.42 | 0.38 | 0.44 | 0.40 | 0.42 | 0.44 | 5.44 | 5.31 | 5.62 | 5.33 | 5.44 | 5.52 |
| | 2013 | -1.72 | -1.84 | -1.51 | -1.80 | -1.79 | -1.65 | 7.91 | 7.59 | 8.08 | 7.81 | 8.01 | 8.03 |
| | 1983 & 2013 | 0.09 | 0.05 | 0.13 | 0.06 | 0.09 | 0.12 | 6.79 | 6.64 | 6.93 | 6.69 | 6.80 | 6.89 |
| LandClim | 1983 | -0.04 | -0.06 | -0.03 | -0.05 | -0.04 | -0.04 | 7.28 | 7.23 | 7.32 | 7.25 | 7.27 | 7.31 |
| | 2013 | -0.19 | -0.30 | -0.11 | -0.26 | -0.17 | -0.13 | 5.24 | 5.05 | 5.48 | 5.09 | 5.19 | 5.39 |
| | 1983 & 2013 | 0.21 | 0.18 | 0.23 | 0.20 | 0.21 | 0.22 | 6.34 | 6.25 | 6.45 | 6.29 | 6.32 | 6.39 |
| PICUS | 1983 | 0.40 | 0.32 | 0.46 | 0.34 | 0.41 | 0.43 | 5.53 | 5.21 | 5.87 | 5.39 | 5.45 | 5.76 |
| | 2013 | 0.55 | 0.39 | 0.72 | 0.46 | 0.56 | 0.64 | 3.19 | 2.52 | 3.74 | 2.88 | 3.17 | 3.53 |
| | 1983 & 2013 | 0.60 | 0.52 | 0.67 | 0.56 | 0.61 | 0.63 | 4.52 | 4.09 | 4.92 | 4.34 | 4.43 | 4.75 |
| SAMSARA2 | 1983 | 0.75 | 0.69 | 0.79 | 0.74 | 0.76 | 0.78 | 3.53 | 3.24 | 3.97 | 3.34 | 3.52 | 3.66 |
| | 2013 | 0.23 | -0.21 | 0.46 | 0.14 | 0.27 | 0.34 | 4.25 | 3.58 | 5.38 | 3.97 | 4.17 | 4.54 |
| | 1983 & 2013 | 0.67 | 0.54 | 0.74 | 0.63 | 0.67 | 0.70 | 3.95 | 3.47 | 4.64 | 3.75 | 3.92 | 4.19 |
| SIBYLA | 1983 | 0.68 | 0.63 | 0.72 | 0.67 | 0.68 | 0.70 | 4.02 | 3.78 | 4.31 | 3.92 | 4.01 | 4.10 |
| | 2013 | 0.11 | 0.01 | 0.20 | 0.08 | 0.09 | 0.14 | 4.52 | 4.28 | 4.77 | 4.45 | 4.57 | 4.61 |
| | 1983 & 2013 | 0.64 | 0.60 | 0.66 | 0.63 | 0.65 | 0.65 | 4.28 | 4.15 | 4.53 | 4.18 | 4.24 | 4.33 |

**Table SM6.** Scenarios with best NSE and RMSE vs. DDE results. NSE, RMSE and DDE are mean values over all compartments in 1983 and 2013.

| | **Basal Area Test** | | | **Diameter Distribution Test** | |
|---|---|---|---|---|---|
| **Model** | **Best Scenario** | **$NSE_{max}$** | **$RMSE_{min}$** | **Best Scenario** | **$DDE_{min}$** |
| ForClim | A1-B1-C2 | 0.13 | 6.64 | A2-B1-C2 | 21% |
| LandClim | A2-B1-C2 | 0.23 | 6.25 | A2-B1-C2 | 46% |
| PICUS | A2-B3-C2 | 0.67 | 4.09 | A2-B2-C2 | 28% |
| SAMSARA2 | A2-B3-C1 | 0.74 | 3.47 | A2-B2-C2 | 24% |
| SIBYLA | A1-B2-C1 | 0.66 | 4.15 | A2-B2-C2 | 24% |

**Table SM7.** Details for both inventory periods are shown in. All simulated scenarios with small tree initialization (A2) are shown as mean values for the simulated compartments including standard deviation (SD). V = standing volume [$m^3\ ha^{-1}$], H = harvested volume [$m^3\ ha^{-1} yr^{-1}$], M = volume of trees died [$m^3\ ha^{-1} yr^{-1}$], $\Delta V_{total}$ = Total growth [$m^3\ ha^{-1} yr^{-1}$] (see Equation 3).

| | | 1963 | | 1983 | | 1963-1983 | | 1963-1983 | | 1963-1983 | | 2013 | | 1983-2013 | | 1983-2013 | | 1983-2013 | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Scenario** | **Model** | **V** | ***SD*** | **V** | ***SD*** | **H** | ***SD*** | **M** | ***SD*** | **$\Delta V_{total}$** | **SD** | **V** | **SD** | **H** | **SD** | **M** | **SD** | **$\Delta V_{total}$** | **SD** |
| | Inventory | 382.2 | *±66.1* | 390.9 | *±159.2* | 5.9 | *±1.8* | | | | | 327.3 | ±50.7 | 7.3 | *±3.4* | | | | |
| A2-B1-C2 | ForClim | 370.1 | *±69.4* | 355.0 | *±83.3* | 5.7 | *±1.6* | 1.5 | *±0.1* | 6.3 | ±0.9 | 324.3 | ±82.8 | 5.2 | *±1.6* | 1.1 | *±0.2* | 5.3 | *±1.2* |
| A2-B2-C2 | ForClim | 370.1 | *±69.4* | 355.6 | *±81.5* | 5.7 | *±1.6* | 1.4 | *±0.1* | 6.3 | ±0.9 | 318.8 | ±84.7 | 5.2 | *±1.6* | 1.1 | *±0.2* | 5.1 | *±1.3* |
| A2-B1-C2 | LandClim | 294.4 | *±52.9* | 498.0 | *±122.3* | 8.8 | *±2.0* | 4.2 | *±1.1* | 23.2 | ±5.9 | 382.1 | ±69.3 | 9.1 | *±1.7* | 5.7 | *±1.4* | 10.9 | *±1.7* |
| A2-B2-C2 | LandClim | 294.4 | *±52.9* | 497.7 | *±122.4* | 8.8 | *±2.0* | 4.2 | *±1.1* | 23.1 | ±5.9 | 381.4 | ±71.0 | 9.0 | *±1.7* | 5.7 | *±1.4* | 10.8 | *±1.7* |
| A2-B1-C1 | PICUS | 379.4 | *±65.0* | 365.6 | *±54.8* | 5.7 | *±1.6* | 4.0 | *±1.1* | 8.9 | ±0.5 | 304.2 | ±56.4 | 5.8 | *±2.6* | 4.6 | *±0.8* | 8.4 | *±0.5* |
| A2-B1-C2 | PICUS | 379.4 | *±65.1* | 366.7 | *±58.3* | 5.9 | *±1.6* | 3.6 | *±0.9* | 8.7 | ±0.4 | 314.7 | ±58.3 | 6.2 | *±2.8* | 3.5 | *±0.6* | 7.9 | *±0.4* |
| A2-B2-C1 | PICUS | 379.5 | *±64.8* | 367.6 | *±59.3* | 5.7 | *±1.6* | 3.8 | *±1.1* | 8.8 | ±0.5 | 301.9 | ±58.7 | 5.8 | *±2.6* | 4.7 | *±1.0* | 8.3 | *±0.6* |
| A2-B2-C2 | PICUS | 380.1 | *±65.1* | 367.3 | *±59.1* | 5.9 | *±1.6* | 3.5 | *±0.8* | 8.8 | ±0.6 | 311.7 | ±57.5 | 6.1 | *±2.6* | 3.6 | *±0.8* | 7.9 | *±0.4* |
| A2-B1-C1 | SAMSARA2 | 381.9 | *±65.1* | 398.2 | *±71.9* | 5.8 | *±1.9* | 1.0 | *±0.2* | 7.6 | ±0.8 | 336.6 | ±92.7 | 7.4 | *±3.5* | 0.8 | *±0.2* | 6.0 | *±0.9* |
| A2-B1-C2 | SAMSARA2 | 381.9 | *±65.1* | 411.1 | *±69.8* | 5.8 | *±1.9* | 1.0 | *±0.1* | 8.2 | ±1.1 | 371.9 | ±94.5 | 8.0 | *±3.0* | 0.8 | *±0.3* | 6.8 | *±1.0* |
| A2-B2-C1 | SAMSARA2 | 381.9 | *±65.1* | 396.6 | *±72.6* | 5.8 | *±1.9* | 1.0 | *±0.2* | 7.5 | ±0.9 | 332.8 | ±82.2 | 8.2 | *±2.9* | 0.7 | *±0.2* | 5.9 | *±0.7* |
| A2-B2-C2 | SAMSARA2 | 381.9 | *±65.1* | 407.9 | *±70.5* | 5.8 | *±1.9* | 1.1 | *±0.1* | 8.1 | ±0.9 | 372.0 | ±81.5 | 8.0 | *±3.0* | 0.9 | *±0.2* | 6.9 | *±0.9* |
| A2-B1-C1 | SIBYLA | 387.5 | *±64.0* | 390.3 | *±78.7* | 5.8 | *±1.7* | 2.3 | *±0.7* | 8.3 | ±1.1 | 315.4 | ±69.9 | 6.7 | *±2.9* | 2.1 | *±0.5* | 6.3 | *±1.0* |
| A2-B1-C2 | SIBYLA | 387.5 | *±64.0* | 392.8 | *±80.3* | 5.8 | *±1.7* | 2.3 | *±0.7* | 8.4 | ±1.2 | 317.2 | ±68.2 | 6.7 | *±2.9* | 2.1 | *±0.5* | 6.3 | *±1.0* |
| A2-B2-C1 | SIBYLA | 387.5 | *±64.0* | 390.3 | *±78.9* | 5.8 | *±1.7* | 2.3 | *±0.7* | 8.3 | ±1.1 | 315.3 | ±70.4 | 6.7 | *±2.9* | 2.1 | *±0.5* | 6.3 | *±1.0* |
| A2-B2-C2 | SIBYLA | 387.5 | *±64.0* | 393.7 | *±80.2* | 5.8 | *±1.7* | 2.3 | *±0.7* | 8.4 | ±1.1 | 316.2 | ±67.1 | 6.7 | *±2.9* | 2.1 | *±0.5* | 6.3 | *±1.0* |

**Table SM8.** Aggregated model outputs for initialization, simulation (pooled results 1983 and 2013) and scenario variation after 50 simulation years (2013). Variation = mean range of scenario results normalized by the mean scenario result for 2013. Scenario codes: A= Initialization of small trees, B = Browsing by ungulates, C = Harvesting.

| | | ForClim | LandClim | PICUS | SAMSARA2 | SIBYLA |
|---|---|---|---|---|---|---|
| **Initialization** | RMSE [$m^2$ $ha^{-1}$] | 1.6 | 6.9 | 0.3 | 0.2 | 0.7 |
| | DDE [%] | 0.8 | 6.0 | 0.7 | 0.2 | 1.7 |
| **Simulation** | NSE [ ] | 0.09 | 0.21 | 0.60 | 0.67 | 0.64 |
| | mean DDE [%] | 21.7 | 46.6 | 28.5 | 26.8 | 24.4 |
| **Variation** | Scenario A/B/C [%] | NA | NA | 11.1 | 24.2 | 6.1 |
| | Scenario A/B [%] | 6.1 | 2.3 | 6.7 | 8.7 | 4.4 |

## 8.2 PAPER 2

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ORIGINAL ARTICLE

# Can current management maintain forest landscape multifunctionality in the Eastern Alps in Austria under climate change?

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**Abstract** In Central Europe, management of forests for multiple ecosystem services (ES) has a long tradition and is currently drawing much attention due to increasing interest in non-timber services. In face of a changing climate and diverse ES portfolios, a key issue for forest managers is to assess vulnerability of ES provisioning. In a case study catchment of 250 ha in the Eastern Alps, the currently practiced uneven-aged management regime (BAU; business as usual) which is based on irregularly shaped patch cuts along skyline corridors was analysed under historic climate (represented by the period 1961–1990) and five transient climate change scenarios (period 2010–2110) and compared to an unmanaged scenario (NOM). The study addressed (1) the future provisioning of timber, carbon sequestration, protection against gravitational hazards, and nature conservation values under BAU management, (2) the effect of spatial scale (1, 5, 10 ha grain size) in mapping ES indicators and (3) how the spatial scale of ES assessment affects the simultaneous provision of several ES (i.e. multifunctionality). The analysis employed the PICUS forest simulation model in combination with novel landscape assessment tools. In BAU management, timber harvests were smaller than periodic increments. The resulting increase in standing stock benefitted carbon sequestration. In four out of five climate change scenarios, volume increment was increasing. With the exception of the mildest climate change scenario (+2.6 °C, no change in precipitation), all other analysed climate change scenarios reduced standing tree volume, carbon pools and number of large old trees, and increased standing deadwood volume due to an intensifying bark beetle disturbance regime. However, increases in deadwood and patchy canopy openings benefitted bird habitat quality. Under historic climate, the NOM regime showed better performance in all non-timber ES. Under climate change conditions, the damages from bark beetle disturbances increased more in NOM compared with BAU. Despite favourable temperature conditions in climate change scenarios, the share of admixed broadleaved species was not increasing in BAU management, mainly due to the heavy browsing pressure by ungulates. In NOM, it even decreased and mean tree age increased. Thus, in the long run NOM may enter a phase of lower resilience compared with BAU. Most ES indicators were fairly insensitive to the spatial scale of indicator mapping. ES indicators that were based on sparse tree and stand attributes such as rare admixed tree species, large snags and live trees achieved better results when mapped at larger scales. The share of landscape area with simultaneous provisioning of ES at reasonable performance levels (i.e. multifunctionality) decreased with increasing number of considered ES, while it increased with increasing spatial scale of the assessment. In the case study, landscape between 53 and 100 % was classified as multifunctional, depending on number and combinations of ES.

**Keywords** Mountain forests · Ecosystem services · Scale · PICUS · Climate change · Forest management

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## Introduction

Mountain regions provide a diverse range of ecosystem services (ES). In the Eastern Alps in Central Europe, mountain forests serve as a source of timber to support the needs of industry as well as of fuel wood for subsistence use. Forests have to protect slopes from landslides and soil erosion and protect settlements and infrastructure against gravitational natural hazards like snow avalanches and rockfall (Malin and Maier 2007; Dorren et al. 2004). In Austria, for instance, 31 % of forest area has been assigned such a protective role as top priority in forest spatial planning (Niese 2011). Regionally, this percentage may even be as high as 66 %. Due to close to nature status of many mountain forests, the share of nature protection areas, such as those under the EU Natura 2000 regulations, is particularly high in mountain areas. Recently, provisioning of drinking water and the dampening of run-off peaks for hydropower production as well as carbon sequestration have also been recognized as key ecosystem services. The importance of these ES in general and for specific stakeholder groups in particular may vary strongly from region to region (European Environment Agency 2010). This multitude of vital ES demands has led to the paradigm of multifunctional forestry where forests have to provide many ES simultaneously at relatively small spatial scale (Nijnik et al. 2010).

Forests are multifunctional by nature (Kaljonen et al. 2007) and when ecosystem services are complementary or neutral integration may be a feasible approach (Raudsepp-Hearne et al. 2010). In case of conflicting ES, trade-offs must be considered. Inefficient solutions may be the consequence (e.g. Jacobsen et al. 2013) if management concepts enforce integration at small scales (i.e. stand level, a few hectares). In such situations, zoning is then a useful approach to disentangle ES conflicts (Côté et al. 2010). While local practical solutions to balance actual ES demands have been established ever since by managers and stakeholders, the paradigm of multifunctionality in general has been rarely touched in policy making and governance (Suda and Pukall 2014).

In recent years, the paradigm of landscape-level planning in forest management has evolved in forest sciences (e.g. Fries et al. 1998). While in Scandinavian countries there is already ample experience in landscape-level planning (e.g. Lämås and Eriksson 2003), in Central European forestry it has been seldomly implemented in practice so far. The issue of scale in ES provisioning has only recently attracted more attention in landscape ecology and land use planning (e.g. Grêt-Regamey et al. 2014; Wu et al. 2002; Raudsepp-Hearne et al. 2010). Landscape-level planning implies that (1) multiscale processes such as disturbance regimes are considered in forest management, (2) different ES may require different spatial scales for quantification and monitoring and (3) different ES or portfolios of ES may be prioritized in different parts of the landscape. A prerequisite for such approaches to manage for portfolios of ES is sound knowledge about the interrelatedness of different ES and how this may depend on forest management regimes and other drivers such as climate change. Planning across ownerships imposes particular challenges (Nijnik et al. 2010). Thus, in landscapes with small-scale ownership structure integration of ES provisioning may be the only practical solution. Future climate change may impact ES differently (Seidl et al. 2007; Lindner et al. 2010; Hanewinkel et al. 2012), thus adding complexity to forest management decision making with the need to find new balances in ES provisioning. In forestry, the issue at which spatial scale the provision of specific ES portfolios is feasible and economically efficient is still a matter of debate and calls for focused research.

Here, we set out to assess a currently practiced uneven-aged forest management regime in a catchment in the Eastern Alps in Austria under climate change conditions and evaluate impacts on ES (timber production, carbon sequestration, nature conservation values and protection against snow avalanches, landslides and erosion). We used the forest ecosystem model PICUS version 1.5 in combination with a recently developed landscape assessment tool (Maroschek et al. 2015). We were furthermore particularly interested in (a) the effect of spatial scale on mapped ES indicators and (b) how the spatial scale of ES assessment (i.e. grain size) affects the simultaneous provision of several ES (i.e. multifunctionality) in a mountain forest landscape.

## Materials and methods

### Study area

The study area is located in the Province of Vorarlberg in Austria, close to the Swiss border in the Rellstal valley (47.08° N, 9.82° E). Landowner is the Stand Montafon Forstfonds (SMF), which owns about 6.500 ha forest land in total. Depending on bedrock, the soils are rendzinas, rankers, podzols and rich cambisols. The terrain is steep, with slope angles from 30° to 45°, which makes forest management difficult and underlines the protective function against gravitational natural hazards (snow avalanches, rockfall, landslides and erosion). The case study is a catchment of 250 ha total area (234 ha forest area) in the upper part of the valley at altitudes between 1060 and 1800 m a.s.l. The timber line that potentially may be as high as 2000 m a.s.l. has been strongly shaped by human activities such as livestock grazing and alpine pasturing. During the last decades, those activities have been widely regulated, and since then grazing has been abandoned in the study area (Malin and Maier 2007). As a consequence successional dynamics are moving the timberline upward. Forest management has been practiced since more than 500 years (Bußjäger 2007). The current management objectives of the owner are income generation from timber production and securing sustainable protection against snow avalanches and landslides (Malin and Lerch 2007). In addition, major shares of the forest area are under Natura 2000 regulations with a focus on bird habitat protection for black woodpecker (*Dryocopus maritimus*) and three-toed woodpecker (*Picoides tridactylus*) (Grabherr 2000).

### Forest

The forests in the case study area are dominated by Norway spruce (*Picea abies*, 96 % of growing stock) with minor shares of Silver fir (*Abies alba*, 3 %), European beech (*Fagus sylvatica*, 1.6 %) and other broadleaved species (e.g. *Acer pseudoplatanus*, *Fraxinus excelsior*, 1 %). Historic forest management has led to mostly uneven-aged patchy stand structures with a considerable share of large old trees (Malin and Lerch 2007; Malin and Maier 2007). Game management has favoured high densities of ungulates, and consequently, the browsing pressure on Silver fir and broadleaves is high. According to internal records of the owner, productivity ranges from 3.5 to 12 $m^3$ $ha^{-1}$ $year^{-1}$ depending on site and stand composition and structure (Malin and Maier 2007). The current mean standing stock in the case study area is 455 $m^3$ $ha^{-1}$. The largest part of the forest is located on steep slopes which are not accessible by forest roads but require skyline-based harvesting systems for timber extraction.

### Climate data

A baseline climate represented by the historic climate of the period 1961–1990 (c0) and five transient climate change scenarios (c1–c5), each consisting of a 100-year time series covering the period 2010–2110 of daily temperature, precipitation, radiation and vapour pressure deficit, were prepared for the model simulations. The baseline climate was generated from available daily instrumental data of the historic period 1961–1990 from the meteorological station Feldkirch (9.6° long, 47.27° lat) and adjusted for representative site types within the case study area regarding altitude, slope and aspect using the algorithms in Thornton and Running (1999). The five climate change scenarios were based on regional climate simulations from the ENSEMBLES project (Hewitt and Griggs 2004; www.ensembleseu.org) which had been downscaled from the grid scale of the regional climate model simulations to local sites. For details on the downscaling approach see Bugmann et al. (this volume). Mean historic climate at 1000 m a.s.l. is characterized by 6.2 °C mean annual temperature and 1150 mm annual precipitation with 840 mm during summer season from May to September. In all climate change scenarios, temperature increased (+2.6 °C in c1, +3.0 °C in c2, +3.5 °C in c3, +4.3 °C in c4, +6.0 °C in c5). In all climate change scenarios except c1, there was a relative shift of precipitation from summer (May–September) to winter with a reduction in summer by −7 % in c2, −32 % in c3, −19 % in c4 and −14 % in c5. Climate anomalies were all related to the baseline climate and the period 2081–2110 of the climate change scenarios.

### The PICUS forest ecosystem model

#### *General*

The model used for this study is the hybrid (sensu Peng 2000) forest ecosystem model PICUS version 1.5. The model is a hybrid of classical gap model components (PICUS v1.2, Lexer and Hönninger 2001) and process-based stand-level NPP algorithms (3PG, Landsberg and Waring 1997). A detailed description of the model is provided in Seidl et al. (2005). Here, just a brief overview on the core model concept is given.

PICUS simulates growth, regeneration and mortality of individual trees on a grid of 10 × 10 $m^2$ patches. Tree biomass is arranged in cells with a vertical depth of 5 m. A three-dimensional light model, allowing for the explicit consideration of direct and diffuse radiation within the canopy, is used to estimate absorbed radiation for each tree. Stand-level productivity is estimated with a simplified model of light use efficiency (Landsberg and Waring 1997) which depends on temperature, radiation, vapour pressure deficit, soil water and nutrient supply. Redistribution of assimilates to individual trees, assuming fixed respiration rates (Landsberg and Waring 1997), is accomplished according to the relative competitive success (i.e. biomass increment in the preceding year) of the individuals (see Lexer and Hönninger 2001). The development of seedlings and saplings is modelled in a size class approach within five height classes (Woltjer et al. 2008). The PICUS model furthermore includes a descendent of the TRACE soil model as described in Seidl et al. (2008) which in the current study was used to simulate the decomposition of C from litter and deadwood as a prerequisite to monitor effects of management and climate on the soil C pools. Dead trees, if not removed through forest management, are transferred stochastically from snags to wood detrital pools on the forest floor. The PICUS model includes also a bark beetle disturbance module which (1) computes the stochastic infestation risk for simulated forest stands, (2) estimates the damage intensity if an infestation occurs and (3) distributes the resulting tree mortality within the simulated stand (Seidl et al. 2007). PICUS contains a flexible management module based on a scripting language allowing for spatially explicit harvesting interventions as well as planting operations at the level of the 100 $m^2$ patches. The basic time step of the simulation is monthly with annual integration of the tree population dynamics processes. The model requires information about the soil water storage capacity, the pH value of the mineral soil as well as plant-available nitrogen as a proxy for nutrient supply as well as a number of parameters for the soil submodel. The PICUS model in the current study was driven by monthly values of temperature, precipitation, solar radiation and vapour pressure deficit of the atmosphere. With the current model version, stands of up to 25 ha can be simulated. PICUS has been tested intensively (e.g. Huber et al. 2013; Didion et al. 2009; Seidl et al. 2005).

A PICUS simulation can start from bare ground or with any defined stand structure. The initial state of a simulated stand (trees with diameter at breast height (DBH) > 2 cm) can be provided as a tree list and a related map with tree positions containing species, DBH and height for each individual or as a species-specific DBH distribution and a height-diameter model. If no tree coordinates are available individuals can be distributed randomly or based on qualitative information about the mixture form (i.e. small groups and patches). Please note that the population dynamics model and the NPP module do not distinguish the position of individual trees below the 100 $m^2$ patch resolution. Regeneration as species-specific density (n $ha^{-1}$) in 5 height classes can be initiated as patchy pattern (100 $m^2$ resolution) or as a homogeneous regeneration layer throughout the simulated stand. For the calculation of spatial stand structural indices, tree maps can be exported and loaded into a landscape assessment tool (LAT). The LAT can visualize and analyse multiple single-tree maps on a digital terrain model. Tree and standing dead wood attributes (coordinates, species, and dimension) can be analysed in freely selectable subareas or with moving window approaches (see Maroschek et al. 2015) and exported for further analysis or enhanced visualization purposes.

*Model calibration*

PICUS has been developed as a generalized model of tree population dynamics and forest development aiming at a generic species parameterization (see Seidl et al. 2005). In the current study, the same species parameterization for European temperate forest ecosystems was used as established in Seidl et al. (2010, 2011) and later applied in Huber et al. (2013) and Maroschek et al. (2015). There was one exception related to the tree regeneration module. In trial runs, it became obvious that tree regeneration, particularly for pioneer species (*Alnus* sp., *Betula pendula*, *Populus tremula*), developed too fast, and density as well as height development on patch cut areas could not be matched with observations from the field. The main reason for this mismatch was competition by grass and herb species which were not considered in the model. To adjust establishment rate in the smallest height class (<10 cm) and early growth, the germination rate of seeds and height growth potential in the five height classes were reduced (see Table SM-1 in Supplementary Material).

*Model evaluation*

In order to evaluate the ability of PICUS to reproduce tree growth and development of stand structure in the case study area time-series data as collected by Neumann (1993) were used. Neumann (1993) reconstructed a 30-year time series of growth in three Norway spruce stands at altitudes of 950, 1230 and 1690 m a.s.l. (see Supplementary Material) in the adjacent valley close to the study site from a tree and stump inventory in 1991 and tree ring and stem analysis for selected trees. The stands at 1230 and 1690 m were uneven-aged with heterogeneous canopies and the stand at 950 m a.s.l even-aged. For the model evaluation, the stands were initialized according to the stand characteristics in 1961 and a random thinning was conducted in every 10-year period to mimic the stem numbers given in Neumann (1993). Historic climate to drive the simulations and soil parameters were provided as described in "Climate data" and "Forest initialization" sections. Simulated dominant height, average height, average DBH and basal area were compared with the data in Neumann (1993). Simultaneous $F$ tests for regression models of observed versus predicted values for mean height and DBH and basal area (states in 1971, 1981 and 1991 as well as periodic increments) indicated good fit of the simulations. Simulated dominant height showed larger deviations, particularly at the high altitude site. For details, see Supplementary Material.

## Forest initialization

Based on 53 polygons that had been derived manually from aerial images, a terrestrial inventory was carried out for the purpose of this study. At least eight inventory plots per polygon on a base raster of 50 × 50 m were measured using angle-count sampling to gather information about basal area shares of tree species, diameter distributions of tree species, a height-diameter regression model, a description of tree regeneration (density by species and height class, mixture form) and soil attributes. From Hollaus et al. (2006, 2007), a normalized crown model and a volume map derived from high-resolution LiDAR data were available. Based on these data, tree maps (size, species and location of individual trees) were generated for each of the polygons. For details of the approach, see Maroschek et al. (2015). All polygons had a spatially explicit position, and the 53 tree maps were subsequently mapped into the 250 ha landscape (Fig. 2).

The landscape was then structured into 18 harvesting units (HU). These 18 HUs were used as basic simulation entity (4–20 ha in size, Fig. 1) in the PICUS model. The main rationale for the delineation of the HUs has been topography which determines the efficient location of skyline tracks for timber harvesting. In prior work, based on interviews with local forest management staff and supported by GIS a total of 131 skylines had been located in the catchment area. The parameter values for the site types in the polygons and HU, respectively, were taken from the soil data base for Austrian forests as described in Seidl et al. (2009).

### Forest management

The currently practiced management regime (BAU; business as usual) is aiming at uneven-aged, structurally diverse forests. Due to steep terrain, timber harvesting is bound to motor-manual felling, delimbing and cutting the stems to length. The logs are extracted to forest roads at the base of the slopes by cable yarding with skyline systems. Skyline tracks typically extend diagonally across the slope to avoid vertical corridors which may favour avalanches and rockfall. The mean skyline length in BAU management over the 100 year analysis period is 534 m (minimum 110 m, maximum 955 m). Current management features patch cuts along the skyline track. Size and shape of the patches is variable with a typical maximum width of 50 m (i.e. maximum lateral skidding distance) and a mean length of 40-50 m along the skyline (compare Fig. 1). Spacing and timing of the skylines depend on the maturity of forests on one hand, and on the avoidance of negative visual impact by the implementation of too many locally clustered skylines and related intensive timber harvesting activities on the other hand. Current management relies fully on natural regeneration. No tending and thinning operations are carried out in the rejuvenated patches. The general silvicultural aim is to maintain and further develop the heterogeneous uneven-aged forest structure. According to management records and as implemented in the BAU regime in this analysis, each year approximately 0.4 % of the forest area is subject to felling operations (i.e. 0–2 skylines per year which corresponds to an average of 0.83 ha patch cut area per year). Overall, the implemented BAU management results in a complete area turnover of the case study catchment of 250 years. Based on Maroschek (unpublished data), the annual browsing probability in the model runs for *A. alba* seedlings was set to 0.78, for *F. excelsior* 1.0, *A. pseudoplatanus* 0.51 and *F. sylvatica* 0.70.

For comparison, a no-management regime (NOM) without any active silvicultural intervention has also been simulated. Browsing intensity in NOM was as in the BAU scenario.

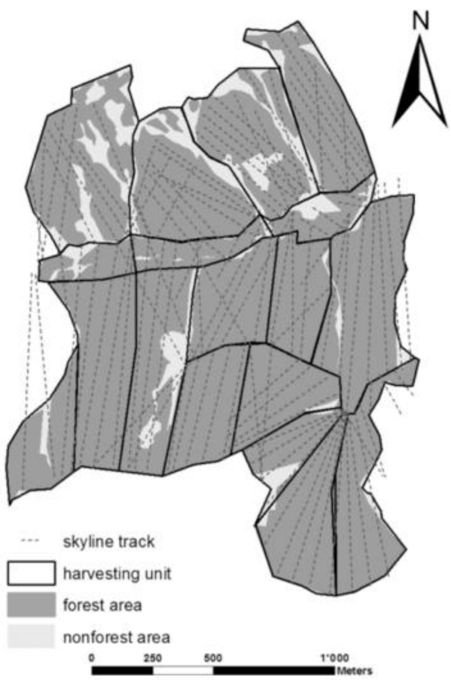


**Fig. 1** Study landscape structured into 18 harvesting units as simulation entities and 131 skyline tracks to implement BAU management 2010–2110

### Analysis

#### *Ecosystem service indicators*

A set of indicators is used to characterize the level of ES provisioning (Table 1). Timber production is represented by standing volume of life trees (*V*), the harvested stemwood volume (THV), the periodic mean increment (VI) and the timber volume killed by bark beetle infestations (BBD). CS includes carbon in tree biomass, standing deadwood, coarse woody debris and soil carbon. Biodiversity indicators are species diversity (*D*; Eqs. 1a, 1b) (Jost 2007), tree size diversity (*H*; Eqs. 2a, 2b), the volume in standing deadwood (SDVW; DBH > 20 cm) and the number of large living trees (LLTN; DBH > 50 cm).

**Table 1** Ecosystem service indicators for current "business-as-usual" management (BAU) and the no-management regime (NOM) under historic climate (c0) for three assessment periods: P1 = 2010–2043, P2 = 2044–2077, P3 = 2078–2110

| Acronym | Explanation | Unit | BAU | | | NOM | | |
|---|---|---|---|---|---|---|---|---|
| | | | P1 | P2 | P3 | P1 | P2 | P3 |
| **Timber production** | | | | | | | | |
| *V* | Standing volume living trees | $m^3\ ha^{-1}$ | 436.5 | 450.4 | 490.1 | 466.4 | 525.6 | 599.6 |
| THV | Annual volume harvested | $m^3\ ha^{-1}\ year^{-1}$ | 1.9 | 1.6 | 2.8 | – | – | – |
| VI | Annual net volume increment | $m^3\ ha^{-1}\ year^{-1}$ | 4.9 | 5.6 | 6.5 | 5.0 | 5.6 | 5.9 |
| BBD | Volume killed by bark beetles | $m^3\ ha^{-1}\ year^{-1}$ | 0.6 | 0.6 | 0.5 | 0.6 | 0.6 | 0.5 |
| **Carbon sequestration** | | | | | | | | |
| CS | Carbon (trees, standing deadwood, coarse woody debris and soil carbon) | $t\ ha^{-1}$ | 220.2 | 218.2 | 222.9 | 237.6 | 244.4 | 253.1 |
| **Biodiversity** | | | | | | | | |
| *D* | Tree species diversity | – | 1.33 | 1.32 | 1.23 | 1.32 | 1.29 | 1.22 |
| *H* | Tree size diversity (mean Shannon diversity of DBH and height) | – | 5.75 | 5.41 | 5.60 | 5.72 | 5.21 | 4.96 |
| AA | Basal area share of *Abies alba* | % | 6.2 | 4.6 | 3.4 | 6.1 | 4.6 | 3.6 |
| BL | Basal area share of broadleaves | % | 3.4 | 3.3 | 2.6 | 3.3 | 2.9 | 2.0 |
| SDWV | Standing deadwood volume (DBH < 20 cm) | $m^3\ ha^{-1}$ | 22.9 | 20.5 | 23.8 | 24.4 | 23.5 | 28.8 |
| LLTN | Large living trees (DBH > 50 cm) | $n\ ha^{-1}$ | 56.1 | 54.5 | 49.7 | 60.4 | 65.0 | 66.3 |
| BHQ | Bird habitat quality | % | [51/39/10] | [40/58/01] | [36/64/00] | [46/44/10] | [36/60/05] | [23/75/02] |
| **Protection against gravitational hazards** | | | | | | | | |
| API | Avalanche protection index | 0–1 | 0.91 | 0.96 | 0.98 | 0.93 | 0.98 | 0.98 |
| LPI | Landslide and erosion protection | % | [07/45/48] | [02/26/71] | [01/27/71] | [04/42/55] | [00/25/75] | [00/26/74] |

Categories for BHQ and LPI: 1 = bad, 2 = moderate, 3 = good, provided in percentage of area for categories [1/2/3]. Indicator values for BHQ, API and LPI are based on 1 ha samples and all other indicators based on HU simulation entities

$$D = \exp(H) \tag{1a}$$

$$H = -\sum_{i=1}^{S} p_i \ln(p_i) \tag{1b}$$

where $S$ is the number of tree species and $p_i$ is the relative basal area share of species ($i$).

$$H_{\text{size}} = \frac{H_{\text{DBH}} + H_H}{2} \tag{2a}$$

$$H_{\text{DBH}} = -\sum_{m=1}^{N_{\text{DBH}}} p_m \ln(p_m) \tag{2b}$$

$$H_H = -\sum_{n=1}^{N_H} p_n \ln(p_n) \tag{2c}$$

where $N_{\text{DBH}}$ is the number of 5 cm DBH classes for trees >5 cm DBH and $N_{\text{H}}$ is the number of 2 m height classes for trees taller than 4 m. $p_m$ is the relative basal area within a DBH class and $p_n$ is the relative basal area within a height class.

Furthermore, a bird habitat quality (BHQ) index on ordinal scale (BHQ1 = poor habitat quality, BHQ2 = moderate, BHQ3 = good habitat quality) characterizes habitat quality for black woodpecker (*D. maritimus*) and three-toed woodpecker (*P. tridactylus*) as key bird species in the study region. BHQ is a composite indicator based on structural attributes of the forest (standing deadwood with DBH > 30 cm, large living trees with DBH > 50 cm, canopy cover) and time since previous management activities as an indicator for anthropogenic disturbance (for details, see Bugmann et al. this issue).

The avalanche protection index (API) indicates protection against snow avalanche release. The index is calculated from mean slope, basal area and average diameter (Eq. 3).

$$\text{API} = \min\left[\frac{G}{(0.2901 * \text{mDBH} + 1.494) \times (0.1333 * \text{slope} - 3)}; 1\right] \tag{3}$$

where $G$ is stand basal area ($m^2\ ha^{-1}$), mDBH is mean DBH (cm) and slope is related to the respective stand (°).

The indicator for landslide and erosion protection (LPI) builds on crown cover defined by projected crown area for trees with DBH > 5 cm (compare also Frehner et al. 2005). LPI is ordinally scaled with three categories (LPI1 < 30 % canopy cover, ≥30 % LPI2 < 60 %, LPI3 ≥ 60 %).

For details on indicator definition, see Bugmann et al. (this volume).

*Assessment approach*

BAU and NOM scenarios were each simulated under historic climate and five climate change scenarios, and model output of the simulated HUs (tree level) was mapped into a digital elevation model. Four spatial aggregation levels were defined for the calculation of the ES state indicators: (1) harvesting units (HU), (2) 1 ha, (3) 5 ha and (4) 10 ha grid cells (i.e. grain size; see Fig. 2).

The flow indicators TVH [total harvested volume (THV)], VI (mean periodic volume increment) and BBD (bark beetle damage) were only provided for the harvesting units and then aggregated at landscape level.

First, ES provisioning at landscape scale under historic climate and climate change conditions is presented. For calculation of ES indicators, the model output for the 18 HU was used. The indicators BHQ, API and LPI required a local spatial context and were therefore calculated as mean value from 1 ha samples (see Fig. 2).

Second, we tested the effect of grain size in estimating ES provisioning the landscape. The different grain sizes were also used to test for effects of analysis period and climate scenarios. ANOVA and Tukey tests were employed for continuous indicators. Shapiro–Wilks and Levene tests were used to test normality and heteroscedasticity of indicators. For ordinal indicators (BHQ, LPI), nonparametric Friedman and Wilcoxon tests were employed. To provide identical sample sizes for the Friedman test, 100 samples of 15 grid cells each were randomly drawn from the 1 ha and 5 ha pixels and compared to the 15 10 ha cells.

Third, the effect of grain size on ES indicator calculation was analysed to explore the joint provisioning of ES at different spatial scales (i.e. multifunctionality).

## Results

### Ecosystem service provisioning under historic climate and climate change scenarios

Under historic climate (c0) and BAU management, standing stock increased from 436.5 to 490.1 $m^3$ $ha^{-1}$ at the end of the simulation period (periodic mean in period P3) because (1) harvests remained clearly below the periodic increment [mean TVH of 1.9 $m^3$ $ha^{-1}$ $year^{-1}$ over the entire analysis period versus a mean increment (VI) of 5.7 $m^3$ $ha^{-1}$ $year^{-1}$] and (2) bark beetle induced tree mortality remained at relatively low level (0.5–0.65 $m^3$ $ha^{-1}$ $year^{-1}$). In the NOM scenario without harvests, volume increased to 599.6 $m^3$ $ha^{-1}$ in P3 (Table 1). When forests are managed according to BAU, the increment increased under climate change scenarios c1, c3 and c4, while in c5 productivity decreased (Fig. 3) depending on the interplay of precipitation and temperature. In warmer climates, damages from bark beetle disturbances increased in BAU (448 % under c5; see Fig. 3) and in NOM (522 % under c5; not shown). Consequently, in combination with harvests and other natural tree mortality, standing stock decreased under all climate change scenarios (−15 % in P3 under BAU management in climate c5) except under climate scenario c1, which resulted in slight increases in P3 under both BAU and NOM (Fig. 3; Table 2).

Closely correlated with volume are in situ carbon pools which increased in BAU under historic climate (c0). In the final assessment period P3, the NOM regime holds an additional amount of 30.2 t $ha^{-1}$ compared with BAU (Table 1). Carbon storage was fairly insensitive to climate change scenarios c1–c4 under both management regimes (BAU, NOM). However, under the severe climate change scenario c5 the carbon pools decreased by −7.5 % in BAU as well as in NOM (Table 2).

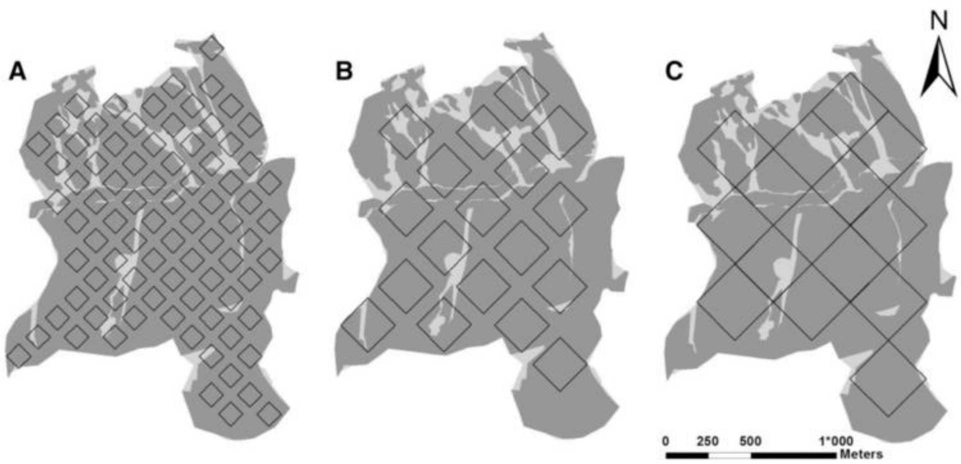


**Fig. 2** Three grain sizes for the assessment of ES indicators in the study landscape (indicated as *black boxes* for **a** 1 ha, $n = 84$; **b** 5 ha, $n = 18$; **c** 10 ha, $n = 15$). *Dark grey area* shows forested area and *light grey* non-forest area

Under historic climate, the shares of Silver fir and admixed broadleaved species were decreasing over time under BAU as well as NOM, whereas under all climate change scenarios the opposite trend occurred. In relative terms, Silver fir and broadleaves could benefit from more favourable growing conditions under the warming scenarios. As a consequence, while tree species diversity (*D*) for the entire landscape was declining in BAU and NOM under historic climate, it was increasing slightly under all climate change scenarios, strongest under scenarios c4 and c5 (Table 1, response under BAU management in Fig. 3). Under historic climate, tree size diversity was declining under BAU as well as NOM management regime (Table 1). Under climate change conditions, tree size diversity increased under BAU management due to newly regenerated trees in the patch cuts (see Fig. 3), while the effect of bark beetle damages alone was not sufficient to increase size diversity in the NOM scenario (Table 2).

Under historic climate, the bird habitat quality (BHQ) index improved and showed a shift towards categories BHQ2 ("moderate") and BHQ3 ("good") under BAU as well as NOM. The combined area shares in BHQ2 and BHQ3 increased under BAU from 49 % in P1 to 64 % in P3, under the NOM regime even more to 77 % (Table 1). Under the climate change conditions of c1 and c5, the area shares in P3 increased to 68 and 83 % (under BAU) and to 81 and 89 % (NOM), respectively, in period P3 (Table 2; Fig. 4). Climate change scenarios, particularly scenario c5, favoured habitat quality because of more standing deadwood of large dimensions and canopy openings due to bark beetle damages. In NOM without active management, standing deadwood volume (SDWV) increased to 28.8 $m^3$ $ha^{-1}$ (P3, historic climate; Table 1). Under climate change scenarios, SDWV increased strongly (compare Fig. 3 and Table 2). Under NOM, the number of large trees (LLTN) increased under all climate scenarios except c5 where natural tree mortality and bark beetle infestations resulted in a decrease in LLTN. Under BAU, harvests and bark beetle damages reduced LLTN (Table 1; Fig. 3), the latter particularly under conditions of climate change (Table 2).

Landslide protection at the beginning of the simulation period was already well developed. In P1 under historic climate and BAU management, 93 % of the landscape area met the crown cover criterion for medium (LPI2) or good protection (LPI3) (Table 1). This share rose to 98 % in P3. Under the NOM regime, these shares were even higher, and in P3 100 % of the landscape area were classified as LPI3. The development was similar for protection against avalanches (API; see Table 1). Interestingly, both API and LPI were almost insensitive to climate change (compare Table 2).

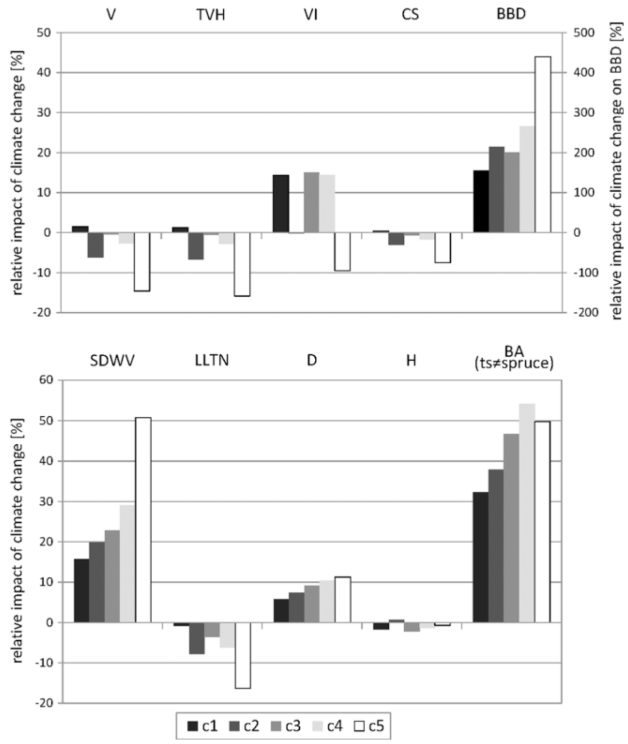


**Fig. 3** Impact of five climate change scenarios on ES indicators under BAU management in period P3 (2078–2110) in relation to historic climate (c0)

### Scale dependency of ecosystem service indicators

With three grain sizes, we explored the effect of the assessment scale on indicator estimates and spatial heterogeneity of indicators. Levene tests for heteroscedasticity of the ES indicators were non-significant and Shapiro–Wilks tests were significant for API, size diversity *H* and partly the large standing deadwood volume SDVW. After visual inspection of the data, we decided against transformation of variables because ANOVA is fairly insensitive to slight deviations from normality (McDonald 2014). ANOVAs for effects of grain size within periods and management regimes under historic climate were not significant ($\alpha = 0.05$) for V, CS, D, SDVW, LLTN and API (compare data in Table 2). Figure 4 shows the area shares for the BHQ and LPI categories for each of the three grain sizes. For bird habitat, the share of categories BHQ1 ("poor") decreases with increasing grain size, and for BHQ3 ("good") it is the opposite. This pattern is independent of the used climate scenario and the management regime. For LPI, however, this effect of grain size is similar for the NOM regime but not as apparent under BAU management. In general, for ordinal BHQ and API indicators the share of significant

**Table 2** Ecosystem service indicators for "business-as-usual" management (BAU) and the no-management regime (NOM) under climate change scenarios c1, c3 and c5 in period P3 (2078–2110)

| Acronym | Explanation | Unit | BAU | | | NOM | | |
|---|---|---|---|---|---|---|---|---|
| | | | c1 | c3 | c5 | c1 | c3 | c5 |
| **Timber production** | | | | | | | | |
| *V* | Standing volume living trees | $m^3\ ha^{-1}$ | 497.3 | 488.1 | 418.44 | 609.8 | 599.2 | 520.8 |
| THV | Annual volume harvested | $m^3\ ha^{-1}\ year^{-1}$ | 2.1 | 2.1 | 1.8 | 0 | 0 | 0 |
| VI | Annual net volume increment | $m^3\ ha^{-1}\ year^{-1}$ | 7.4 | 7.5 | 5.9 | 6.7 | 6.8 | 5.4 |
| BBD | Volume killed by bark beetles | $m^3\ ha^{-1}\ year^{-1}$ | 1.4 | 1.6 | 2.9 | 1.4 | 1.6 | 2.9 |
| **Carbon sequestration** | | | | | | | | |
| CS | Carbon (trees, standing deadwood, coarse woody debris and soil carbon) | $t\ ha^{-1}$ | 223.8 | 221.4 | 206.1 | 254.3 | 251.4 | 234.2 |
| **Biodiversity** | | | | | | | | |
| *D* | Tree species diversity | – | 1.31 | 1.35 | 1.37 | 1.27 | 1.31 | 1.34 |
| *H* | Tree size diversity (mean Shannon diversity of DBH and height) | – | 5.51 | 5.48 | 5.56 | 4.87 | 4.83 | 4.95 |
| AA | Basal area share of *Abies alba* | % | 4.0 | 4.3 | 4.7 | 4.2 | 4.6 | 5.6 |
| BL | Basal area share of broadleaves | % | 3.7 | 4.3 | 4.4 | 2.7 | 3.2 | 3.7 |
| SDWV | Standing deadwood volume (DBH > 20 cm) | $m^3\ ha^{-1}$ | 27.6 | 29.2 | 35.9 | 32.2 | 34.1 | 42.1 |
| LLTN | Large living trees (DBH > 50 cm) | $n\ ha^{-1}$ | 48.4 | 47.1 | 40.7 | 65.06 | 63.4 | 55.4 |
| BHQ | Bird habitat quality | % | [32/63/05] | [24/69/07] | [17/62/21] | [19/69/12] | [15/73/12] | [11/68/21] |
| **Protection against gravitational hazards** | | | | | | | | |
| API | Avalanche protection index | 0–1 | 0.98 | 0.98 | 0.97 | 0.98 | 0.98 | 0.98 |
| LPI | Landslide and erosion protection | % | [01/26/73] | [00/25/75] | [01/30/69] | [00/25/75] | [00/25/75] | [02/25/73] |

Classification categories for BHQ and LPI: 1 = bad, 2 = moderate, 3 = good, provided in percentage of area for categories [1/2/3]. Indicator values for BHQ, API and LPI are based on 1 ha samples and all other indicators based on HU simulation entities

Friedman tests ($\alpha = 0.05$) for effects of grain size was small, but larger under the NOM regime compared with BAU (see Table 3). The percentage of significant Friedman tests under historic climate c0 increased from P1 (BAU: 5 % for BHQ, 1 % for LPI; NOM: 6 % for BHQ, 1 % for LPI; not shown) to P3 (BAU: 11 % for BHQ, 1 % for LPI; NOM: 19 % for BHQ, 8 % for LPI).

The aggregate BHQ index responded sensitive to the assessment scale because the relatively rare structural elements required for good habitat quality (large snags, large live trees) are covered much better with larger grain size. LPI, in contrast, did not show this sensible response as it builds on canopy cover as such and is independent of tree size distribution. Please note that landscape structure (forest, non-forest area) remains constant throughout the simulation. Small 1 ha samples covering non-forest area shares are rather insensitive to changes in forest attributes.

In general, the variability of indicators expressed as coefficient of variation (CV) decreased with increasing grain size (see Fig. 5). However, magnitude and temporal development of the variability differed among indicators. Not unexpectedly, the number of large living trees (LLTN) varied greatly when sampled with 1 ha squares, but the CV decreased in P2 and P3. Estimates of total carbon storage showed a decreasing CV over the three assessment periods almost independent of climate scenario. Tree species diversity showed very low contrasts between grain sizes and climate scenarios which is mainly due to the dominating role of Norway spruce. Admixed tree species are rare and do occur in spatial clusters depending on site conditions and regeneration fellings. Particularly, the fellings along the skyline tracks initiate regeneration of Silver fir, mountain maple and beech which become effective in later decades of the assessment period. With small grain size these rare events are obviously missed in most samples which leads to low variation in *D*, while with larger sample size admixed species were more frequently hit, thus increasing the CV in *D*, particularly under warming scenarios. Similarly, API values are favourable throughout the landscape, resulting in a very low variation which even decreased further along time (Fig. 5).

## Grain size and multifunctionality

We decided to use carbon storage (CS), bird habitat provision (BHQ) and protection against avalanches (API) and landslides (LPI) to analyse the simultaneous provision of ecosystem services as these ES are represented unambiguously by one specific ES indicator. We categorized total carbon storage (category CS1: < 203 t ha$^{-1}$, CS2: 203–332 t ha$^{-1}$, CS3: >332 t ha$^{-1}$) and API (API1: 0–0.33, API2: 0.34–0.66, API3: 0.67–1.0) in three performance classes (spread of simulated indicator values divided by 3) so that they were comparable with the ordinally scaled indicators BHQ and LPI. For each grain size, the share of samples with two, three and four ES indicators being simultaneously rated at least "moderate" (categories 2 or 3) was determined to represent the "multifunctional" share of the landscape.

Figure 6 shows several distinct patterns for period P3 where effects of management as well as climate change scenarios became visible: (a) the tendency that the multifunctional area in the case study landscape decreased with increasing number of considered ES. This effect was particularly visible when moving from portfolios of two to three ES; (b) under the NOM regime the multifunctional area shares were not as strongly decreasing with increasing number of considered ES; (c) with increasing scale (i.e. grain) the multifunctional area increased in general; and (d) the increase in multifunctional area under the moderate warming of scenario c1 (also having a favourable distribution of summer precipitation) was higher in BAU compared with NOM. In both BAU and NOM, the multifunctional area was smaller under the severe climate change scenario c5 compared with c1, and sometimes even smaller than under the historic reference climate c0.

The multifunctional share of the landscape under BAU management in period P3 when two ES were combined varied between 53 % (combination of CS and API under c5 at 5 ha grain size) and 100 % (several ES combinations, particularly the combined protective services against avalanches and landslides and erosion). In the NOM scenario, these shares increased to 77 % (CS & BHQ, LPI and BHQ under historic climate c0 at 1 ha grain size) and 100 % (all possible combinations of 2 ES, in different climates and grain size combinations).

The multifunctional share of the landscape combining three ES ranged from 53 (API, CS and BHQ under climate scenario c5 with 5 ha grain size) to 86 % (LPI, CS, BHQ under climate scenario c1 and 10 ha grain size) under BAU management. In unmanaged conditions (NOM), the least compatible ES combination increased the multifunctional area to 76 % (LPI, CS, BHQ) and the most complementary ES combination to 100 % of the landscape (all ES triplets under historic climate and grain sizes larger 1 ha), respectively.

When four ES had to be integrated at small scale, a minimum of 53 % (5 ha grain size, climate scenario c5) and a maximum of 86 % (10 ha grain size, climate scenario c1) of the managed landscape (BAU) were considered as multifunctional. NOM generated between 76 %

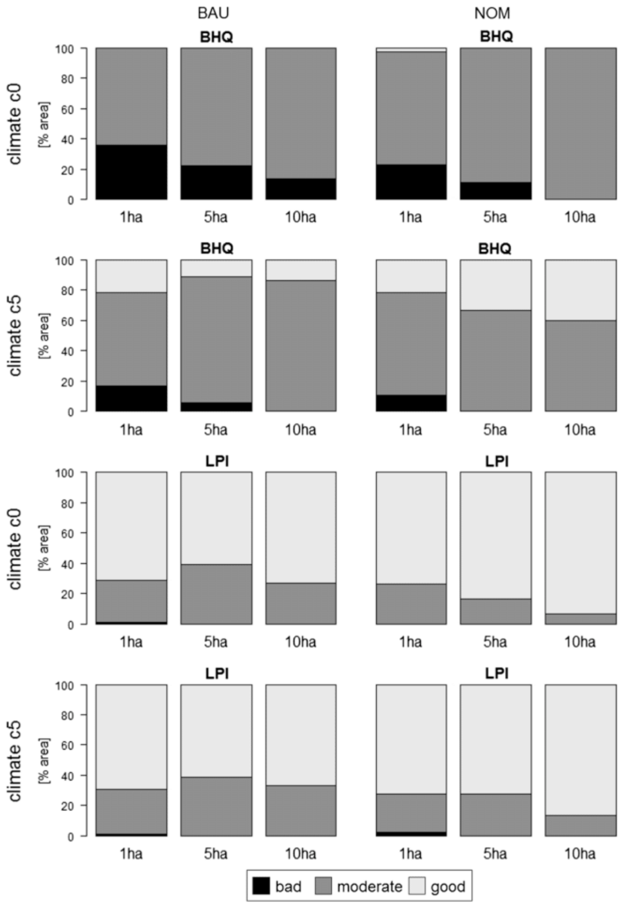


**Fig. 4** Effect of grain size on bird habitat quality (BHQ) and landslide and erosion protection index (LPI) in period P3 (2078–2110) under historic climate (c0) and strong warming (c5)

(under c0 and c5 at 1 ha grain size) and 100 % (under historic climate c0 at 10 ha grain size) of multifunctional landscape area.

Analysis of the temporal development of multifunctionality in the study landscape under historic climate c0 revealed that from periods P1 to P3 the area with sufficient ES provisioning increased with sample sizes of 1 and 10 ha and decreased or showed no trend at all with 5 ha grain size (not shown).

## Discussion and conclusion

In the presented study, we analysed the current "business-as-usual" (BAU) management regime and its effect on the provisioning of timber production, carbon sequestration, nature and habitat conservation and protection against gravitational hazards (avalanches, landslides) under historic climate and five climate change scenarios in a catchment of 250 ha in the Eastern Alps in Austria. Here, we scrutinize the analysis approach, the ability of BAU

**Table 3** Effect of grain size on ES indicators in BAU and NOM management under historic climate (c0) and climate change scenarios (c1, c5)

| Indicator | Grain size | BAU | | | NOM | | |
|---|---|---|---|---|---|---|---|
| | | 1 ha | 5 ha | 10 ha | 1 ha | 5 ha | 10 ha |
| *V* | c0 | 406.1 ± 118.9 | 386.4 ± 80.1 | 398.6 ± 58.8 | 500.3 ± 142.3 | 471.5 ± 86.2 | 490.4 ± 58.7 |
| | c1 | 411.4 ± 123.8 | 387.6 ± 83.1 | 398.7 ± 64.8 | 508.1 ± 150.5 | 476.3 ± 94.1 | 493.2 ± 69.4 |
| | c5 | 345.7 ± 122.2 | 319.7 ± 83.1 | 331.9 ± 68.6 | 434.0 ± 154.7 | 397.6 ± 100.3 | 415.7 ± 81.0 |
| CS | c0 | 226.6 ± 34.8 | 225.8 ± 23.6 | 227.1 ± 19.0 | 259.3 ± 43.0 | 252.2 ± 27.6 | 255.7 ± 21.2 |
| | c1 | 228.2 ± 36.1 | 226.5 ± 24.7 | 227.7 ± 20.8 | 261.2 ± 45.0 | 253.5 ± 30.1 | 256.7 ± 24.0 |
| | c5 | 212.4 ± 35.4 | 210.0 ± 25.2 | 212.1 ± 21.6 | 243.4 ± 45.0 | 234.6 ± 31.9 | 238.6 ± 26.6 |
| *D* | c0 | 1.26 ± 0.39 | 1.32 ± 0.47 | 1.37 ± 0.41 | 1.25 ± 0.38 | 1.30 ± 0.43 | 1.35 ± 0.40 |
| | c1 | 1.34 ± 0.44 | 1.42 ± 0.53 | 1.47 ± 0.48 | 1.31 ± 0.42 | 1.37 ± 0.48 | 1.43 ± 0.45 |
| | c5 | 1.40 ± 0.47 | 1.50 ± 0.56 | 1.57 ± 0.54 | 1.37 ± 0.46 | 1.46 ± 0.53 | 1.53 ± 0.52 |
| SDWV | c0 | 24.6 ± 7.3 | 23.5 ± 4.4 | 24.1 ± 2.9 | 29.5 ± 8.5 | 28.2 ± 4.8 | 28.8 ± 3.6 |
| | c1 | 28.1 ± 9.6 | 27.2 ± 5.6 | 27.8 ± 4.4 | 32.8 ± 10.2 | 31.4 ± 5.5 | 32.5 ± 4.6 |
| | c5 | 36.0 ± 12.6 | 35.1 ± 6.8 | 35.6 ± 5.2 | 42.8 ± 14.3 | 42.1 ± 7.8 | 42.2 ± 6.3 |
| LLTN | c0 | 48.2 ± 21.9 | 46.3 ± 13.3 | 46.6 ± 12.6 | 64.9 ± 28.8 | 61.4 ± 16.4 | 62.2 ± 15.4 |
| | c1 | 47.7 ± 22.7 | 45.2 ± 13.8 | 45.4 ± 13.5 | 65.0 ± 29.9 | 60.8 ± 18.0 | 61.5 ± 17.1 |
| | c5 | 40.2 ± 21.2 | 37.3 ± 13.5 | 38.2 ± 13.0 | 55.5 ± 28.8 | 50.8 ± 17.9 | 52.0 ± 17.0 |
| BHQ | c0 | **[36/64/00]**[1] | **[22/78/00]**[1] | **[13/87/00]**[1] | **[23/75/02]**[3] | **[11/89/00]**[3] | **[00/00/00]**[3] |
| | c1 | **[32/63/05]**[2] | **[06/94/00]**[2] | **[00/100/00]**[2] | **[19/69/12]**[4] | **[00/89/11]**[4] | **[00/93/07]**[4] |
| | c5 | [17/62/21] | [06/83/11] | [0.00/0.87/0.13] | **[11/68/21]**[5] | **[00/67/33]**[5] | **[00/60/40]**[5] |
| API | c0 | 0.98 ± 0.06 | 1.00 ± 0.00 | 1 ± 0.00 | 0.98 ± 0.07 | 1.00 ± 0.00 | 1.00 ± 0.00 |
| | c1 | 0.98 ± 0.07 | 1.00 ± 0.00 | 1 ± 0.00 | 0.98 ± 0.06 | 1.00 ± 0.00 | 1.00 ± 0.00 |
| | c5 | 0.97 ± 0.10 | 0.99 ± 0.05 | 1 ± 0.00 | 0.98 ± 0.08 | 1.00 ± 0.01 | 1.00 ± 0.00 |
| LPI | c0 | [01/27/71] | [00/39/61] | [00/27/73] | **[00/26/74]**[6] | **[00/17/83]**[6] | **[00/07/93]**[6] |
| | c1 | [01/26/73] | [00/28/72] | [00/27/73] | **[00/25/75]**[7] | **[00/11/89]**[7] | **[00/07/93]**[7] |
| | c5 | [01/30/69] | [00/39/61] | [00/33/67] | **[00/25/73]**[8] | **[00/28/72]**[8] | **[00/13/87]**[8] |

Indicator values are mean values for period P3 (2078–2110). BHQ and LPI provided in percentage of area for classification categories [1 = bad/2 = moderate/3 = good]. Bold entries indicate share of significant Friedman tests ≥5 % and share of significant Friedman tests: [1] 11 %, [2] 38 %, [3] 19 %, [4] 5 %, [5] 12 %, [6] 8 %, [7] 9 %, [8] 11 %

management to provide the demanded ES, the spatial heterogeneity of ES indicators within the case study landscape and finally the level of ES integration in providing multiple ecosystem services from the same parcel of forest.

## Analysis approach

The PICUS forest ecosystem model has already proven its ability as a valuable tool for climate change impact studies (Maroschek et al. 2015; Seidl et al. 2011; Lexer et al. 2002). The model has been evaluated in several validation experiments and has built up substantial credibility for applications in European mountain forests (e.g. Huber et al. 2013; Seidl et al. 2011; Didion et al. 2009; Seidl et al. 2005). Given the uncertainty in the data given by Neumann (1993) and the complex stand structures in two of the three validation stands, the validation against three 30-year records of stand development in the current study context further supported the credibility of the model. Essentially the generic parameterization from earlier studies generated 30-year stand development trajectories which matched reconstructed time series of key stand attributes well. Also the simulated volume productivity in the application matched internal records of the forest owner closely. Model adjustments for the current study were confined to establishment and height increment potentials of tree seedlings. Due to intense competition by grass and herb species, regeneration dynamics were much slower in reality compared with initial model simulations. In a generic model, not every ecosystem component and each process can be explicitly covered. However, assuming that the vitality of grass and herb species will not change under future climatic conditions, the implicit parameterization via adjusted germination rates seemed to be a reasonable approach.

In the current contribution, the size of a simulation unit approximately 20 ha simultaneously simulated forest area. This extent of the simulation entities allowed a consistent

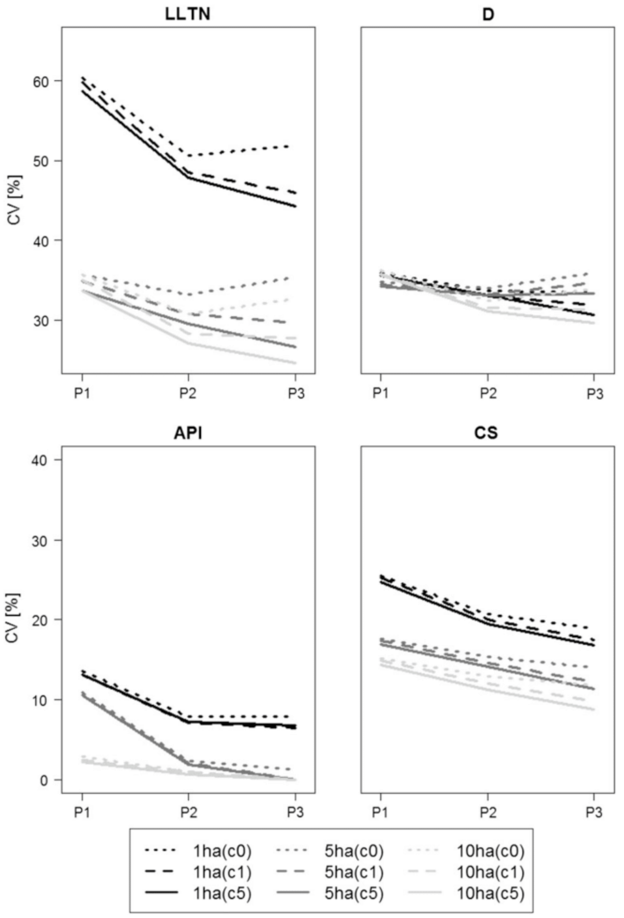


**Fig. 5** Coefficient of variation (CV) for selected ES indicators over the three assessment periods P1–P3 in dependence of grain size and three climate scenarios (historic climate c0, climate change scenarios c1 and c5)

implementation of felling regimes and disturbances by bark beetles (see Pasztor et al. 2014, 2015). Tree mortality caused by disturbance agents is highly influential regarding tree regeneration dynamics, habitat quality and protection against gravitational hazards. Storm damages have not been an important disturbance factor so far, and this is not expected to change in the future (Nikulin et al. 2011).

A further key feature for the analysis was the spatially explicit availability of a 250 ha forest landscape at the resolution of individual trees and 100 $m^2$ patches. This imposes the challenge of realistic model initialization. Roces-Díaz et al. (2015) point at the importance of remote sensing data to (1) overcome the data bottleneck for ES assessments and (2) to improve the spatial representation of ES supply. The employed forest initialization and projection approach in the current study supports both issues. For larger assessment areas, it would otherwise be impossible to provide realistic initial states for model simulations. The ability to map the trees from a set of simulation units into a digital terrain model offers huge potential to analyse structural and compositional features of a forested landscape (Maroschek et al. 2015). In the current study, we utilized this model feature to assess the effect of different grain sizes on the estimation of ES indicators and thus to shed light on the level of ES integration in a Central European mountain forest landscape.

According to Villa et al. (2014), ES assessment methods must be quantitative and scalable. The ES indicators used in the presented study are transparent and quantitative and allow the analysis of trade-offs among ES. The spatial simulation and assessment approach can be scaled continuously from tree to landscape level. To isolate the effect of management as such a non-intervention regime (NOM) has also been implemented and used for comparison with the "business-as-usual" (BAU) regime. We thus concluded that the assessment approach was well suited for the research objectives at hand.

## Landscape-level ES provisioning under BAU management

BAU management and the related cutting intensity have led to increasing volume stocks. From the mean annual increment of 5.7 $m^3$ $ha^{-1}$ $year^{-1}$, about 33 % were harvested. Another 10 % were lost to bark beetle induced tree mortality which, at mortality rates of 0.58 $m^3$ $ha^{-1}$ $year^{-1}$, could not be salvaged at reasonable costs and therefore largely remained in the forest. At the end of the 100-year simulation period, the mean standing volume stock of living trees was over 600 $m^3$ $ha^{-1}$ with the clear tendency to increase further. On the one hand, this increased the in situ carbon pools. On the other hand, however, it also increased the risk of damage through disturbances (Pasztor et al. 2014) which may then in turn negatively feedback on the climate change mitigation effect via in situ carbon storage. Due to the low felling intensity, avalanche (API) and landslide protection (LPI) improved over the 100-year simulation period to sufficient levels at almost the entire landscape area. This did not change significantly under climate change conditions. However, the potential for negative climate change impacts has been revealed under the severe climate change scenario c5 and the related increase of bark beetle damage to more than 500 % compared with historic climate under the NOM regime. It should be noted that in the current study the avalanche protection index did not include gap formation by disturbances or management in explicit form (compare Maroschek et al. 2015) and may therefore overestimate the protection effect. However, the 1 ha grain size used to calculate the API and LPI indicators can be considered as suitable for the detection of protection deficiencies.

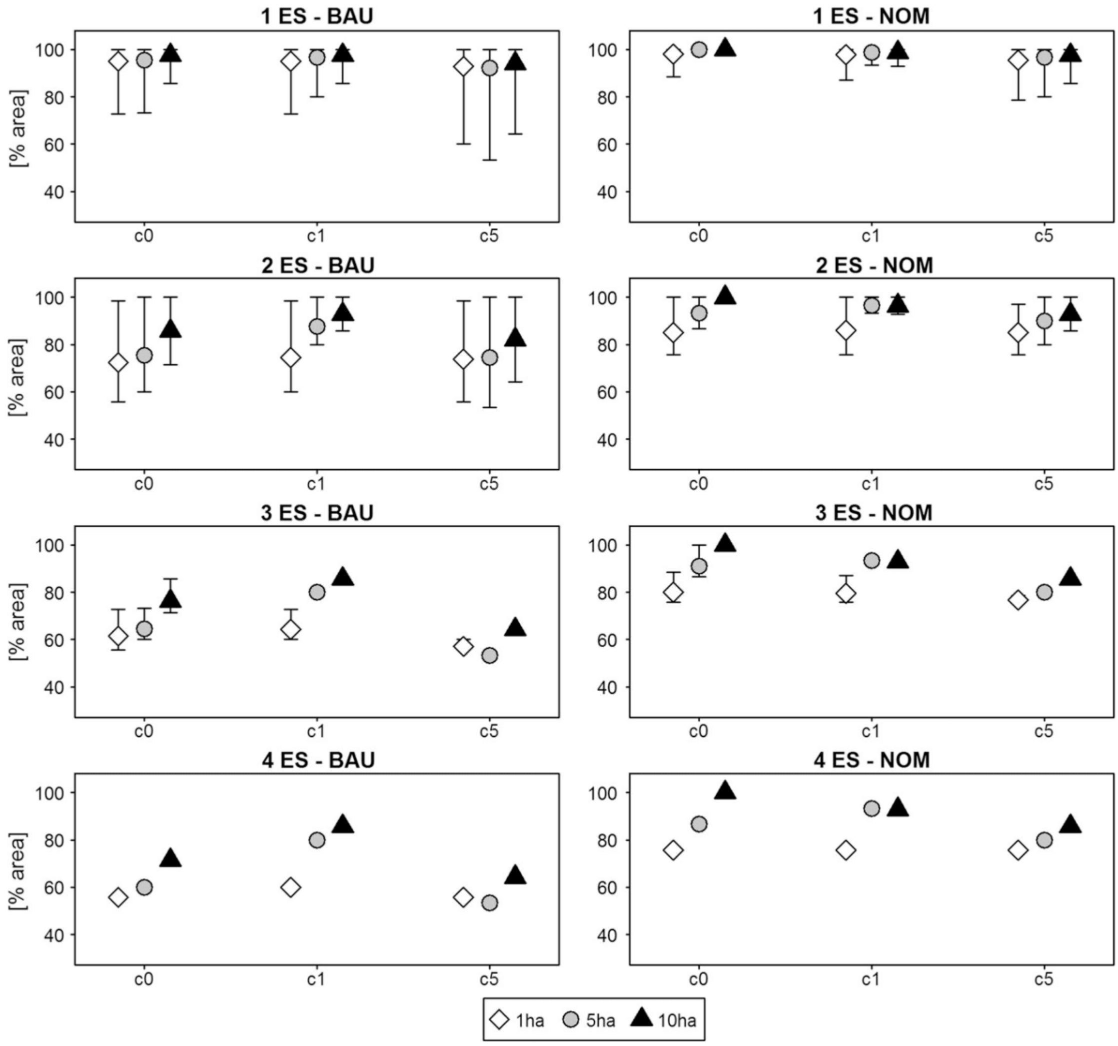


**Fig. 6** Multifunctional share of landscape area [bundles of ES in performance category 2 (moderately good) or 3 (good) in period P3 (2078–2110)] under historic climate (c0) and two climate change scenarios (c1, c5) under BAU ("business-as-usual") management and the no-management regime (NOM). Symbols represent mean area percentage, and *whiskers* indicate worst and best case within given situation. Considered ES: carbon storage, bird habitat, protection against avalanches and landslides

The bird habitat quality in initial phases of the analysis period is insufficient (bird habitat quality category BHQ1) on about one-third of the landscape. However, it must be noted that this is mostly due to missing large snags which could not be realistically initialized from available information. From period P2 onward, simulation results indicated that simulated tree mortality generates realistic numbers of snags in the forest.

## Effects of scale and multifunctionality

As a general pattern, our analysis revealed the largest variation for 1 ha, smallest for 10 ha grain size and a decrease in spatial variation over time for most ES indicators, independent of grain size. The fine-grained cutting pattern of BAU management homogenized most ES indicators in course of the twenty-first century, after an initial increase in spatial variation. In the NOM regime, this homogenizing effect is less visible. Of course, these results cannot be generally extrapolated to any other landscape because they largely depend on the interplay of harvesting pattern and major ecosystem processes (growth, regeneration, mortality). However, such studies are rarely reported in the literature (compare Grêt-Regamey et al. 2014) but can shed light on the importance of spatial scale in ES assessments.

The set of ES considered in the presented study is typical for European mountain forests: timber production, nature and habitat conservation and protection against gravitational hazards. In situ carbon sequestration may be a general public interest as well. In our case study area, BAU management sacrificed timber harvests in favour of carbon sequestration, protection against gravitational hazards and bird habitat provisioning. To which extent the protective services would be affected by intensified harvests is not easy to guess and would require the extended analysis of alternative silvicultural regimes. In our study, the most severe trade-offs seem to be related to timber harvesting. In general, the NOM regime showed better performance in all non-timber ES throughout the entire simulation period. The reduction in multifunctional area when the number of demanded ES is increased from two to four is surprisingly low (from about 80 to 60 % under BAU management and historic climate). Under the NOM regime, this area reduction is even smaller and may show no effect at all. However, what must be noted is that the share of admixed species is further reduced in NOM and that no tree regeneration is initialized. Thus, the mean tree age is increasing and in the long run NOM may enter a phase of reduced ES provisioning and lower resilience compared to BAU.

Compared with the BAU regime, the "multifunctional" area share was in general larger in NOM. Whether the non-timber ES are neutral or even complementary over longer time periods depends on several issues: first, the thresholds that define "sufficient level of ES provisioning". In our analysis, we used categories of equal width between minimum and maximum values of each ES indicator. In a real decision-making situation, this will depend mainly on stakeholder preferences. Second, silvicultural regimes can be tailored locally to meet the requirements of demanded ES portfolios. For instance, Boncina (2011) discussed approaches to integrate nature conservation values in forest management. Fuhr et al. (2015) show that protection against gravitational hazards can be provided also by ageing forests as long as forest structure is irregular and patchy.

With no explicit benchmarks available, it is difficult to qualify BAU management in the study area. It is obvious that timber production could be intensified. Protective effects against avalanche release, landslides and erosion appear as sufficient for large shares of the landscape. Habitat quality for various protected bird species is already maintained by current management on vast shares of the analysed landscape. The remaining question is whether it is possible to intensify timber harvests without jeopardizing (1) the other demanded ES and (2) the adaptation of the forest to conditions of climatic changes. Whether such a management approach can be identified will be the focus of further work. In the long run, increasing the shares of broadleaves and conifers which are less vulnerable under warmer climates would be a useful means to increase the resistance against disturbance agents. Detrimental to this adaptive management strategy was the high browsing pressure in the area which actually renders any significant shift in species composition as highly unrealistic.

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## REGIONAL ENVIRONMENTAL CHANGE

**Can current management maintain forest landscape multifunctionality in the Eastern Alps in Austria under climate change?**

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## Supplementary Material

### Model calibration

Table SM-1 shows the adjusted height growth potentials for species in height classes, the germination fraction of seed and the external seed input to the simulated patches. Note that for conifers and Fagus sylvatica seed comes from parent trees in the simulated forest.

**Table SM-1**. Species parameter for the regeneration module in PICUS. HC = height class. External seed input = annual seed input to every 100 m$^2$ patch from outside the simulated forest.

| **Species** | **Height growth potential** [cm] | | | | **germination fraction** | **external seed input** [n 100m$^{-2}$] |
|---|---|---|---|---|---|---|
| | **HC1 0-10cm** | **HC2 10-30cm** | **HC3 30-80cm** | **HC4 80-130cm** | | |
| Picea abies | 4 | 5 | 7 | 10 | 0.012 | 0 |
| Abies alba | 4 | 5 | 7 | 10 | 0.012 | 0 |
| Acer pseudoplatanus | 4 | 8 | 12 | 15 | 0.05 | 0 |
| Fagus sylcatica | 4 | 8 | 10 | 12 | 0.15 | 0 |
| Alnus incana | 4 | 8 | 12 | 12 | 0.05 | 20 |
| Alnus viridis | 4 | 8 | 12 | 12 | 0.05 | 32 |
| Betula pendula | 4 | 8 | 12 | 15 | 0.05 | 12 |
| Populus tremula | 4 | 8 | 12 | 15 | 0.10 | 20 |
| Sorbus aucuparia | 4 | 8 | 10 | 10 | 0.12 | 12 |

## Model evaluation

Table SM-2 shows the three plots established in 1991 as given in Neumann (1993). The stands are situated at altitudes of 950m (Stand 3), 1230m (Stand 2) and 1690m (Stand 1) a.s.l. Stands 1 and 2 were uneven-aged with huge spread in individual tree sizes, Stand three resembled characteristics of a more even-aged stand. The tree attributes DBH and height had been reconstructed based on the inventory in 1991, and tree ring and stem analysis data from a subsample of the trees on each plot. Plot size was 0.1 ha.

PICUS was initialized with the tree data and positions provided for 1961 and simulated over 30years until 1991. Halfway in each period a random thinning was implemented to control for stem density according to Neumann (1993).

Observed and simulated stand variables were visually compared in bivariate scatter-plots and by a regression analysis of observed vs. predicted values. The regression models were tested with a simultaneous F-test (Mayer et al. 1994, Vanclay and Skovsgaard 1997).

**Table SM-2:** Characteristics of research plots according to Neumann (1993).

| plot ID | Period [yrs] | Dom. height [m] | Avg. Height [m] | Avg. DBH [cm] | Stems [n/ha] | Basal area [$m^2$/ha] |
|---|---|---|---|---|---|---|
| **1** | 1961 | 26.7 | 19.7 | 27.9 | 880 | 53.83 |
| **1** | 1971 | 27.9 | 20.3 | 29.2 | 860 | 57.51 |
| **1** | 1981 | 29.1 | 21 | 30.6 | 820 | 60.23 |
| **1** | 1991 | 30.6 | 22.6 | 33.9 | 690 | 62.32 |
| **2** | 1961 | 29.1 | 25.1 | 25.5 | 570 | 29.58 |
| **2** | 1971 | 31.1 | 28.3 | 29.1 | 570 | 38.24 |
| **2** | 1981 | 33 | 29.9 | 32.1 | 570 | 46.26 |
| **2** | 1991 | 34.7 | 31.9 | 38.2 | 460 | 52.79 |
| **3** | 1961 | 31.9 | 32 | 36.2 | 460 | 47.41 |
| **3** | 1971 | 35.4 | 33.3 | 38.4 | 460 | 53.24 |
| **3** | 1981 | 38.1 | 35.1 | 42.4 | 350 | 49.37 |
| **3** | 1991 | 40.6 | 36.8 | 46.2 | 320 | 53.53 |

## Results

The numeric results of the validation experiment are displayed in Table SM-3. The graphs of the scatter-plots are shown in Figure SM-1 and Figure SM-2. Predicted mean DBH and basal area are not significantly different from the observations (state and periodic increment). Height predictions seem to be predicted least accurate. Looking at the states of dominant height and average height (Figure SM-1) and the respective increments (Figure SM-2) it is apparent that the largest deviation occurred for stand 1 at 1650m a.s.l. This stand was particularly heterogeneous in stand structure and thus tree attributes difficult to reconstruct. Considering the complex stand structures, particularly in stands 1 and 2, and the uncertainty in reconstructed stand attributes the results of the model validation exercise can be considered satisfactorily.

**Table SM-3.** Results of regression models observed vs predicted values and the simultaneous F-test. a = intercept, b = slope coefficient; F-dist. value (df1=1, df2=11, p=0.95) = 5.318. A significant F-test indicates that intercept and slope parameters do not significantly deviate from = 0 and 1, respectively. Asterisks indicate significance at alpha = 0.05. . Dominant height = mean height of 100 largest trees per ha, avDBH = quadratic mean diameter, average height = height of quadratic mean diameter.

| Parameter | $R^2$ | a | b | calculated F-value | significance |
|---|---|---|---|---|---|
| dominant height [m] | 0.93 | 11.530 | 0.695 | 40.454 | |
| average height [m] | 0.94 | -2.372 | 1.024 | 7.322 | |
| avDBH [cm] | 0.97 | -2.926 | 1.082 | 1.076 | ** |
| basal area [m²/ha] | 0.91 | 6.301 | 0.867 | 1.380 | ** |
| dom. height increment [m] | 0.16 | 3.801 | 0.685 | 6.460 | |
| av. height increment [m] | 0.19 | -0.391 | 0.819 | 5.637 | |
| avDBH increment [cm] | 0.20 | -0.172 | 1.262 | 0.653 | ** |
| basal area increment [m²/ha] | 0.62 | 2.512 | 0.536 | 5.176 | ** |

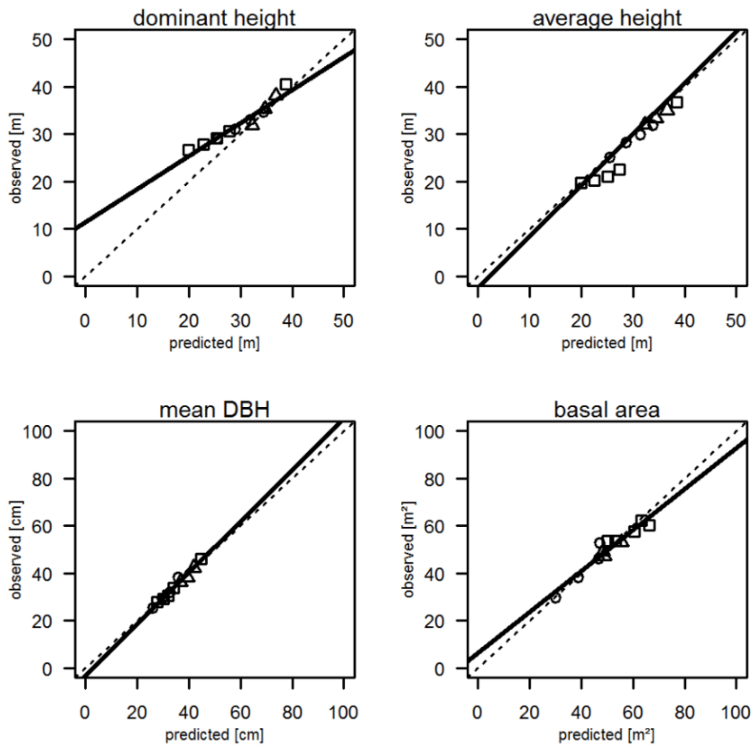


**Figure SM-1.** Predicted versus observed stand attributes of three validation plots in 1971, 1981 and 1991 from Neumann (1993). Squares = stand 1, circles = stand 2, triangles = stand 3.

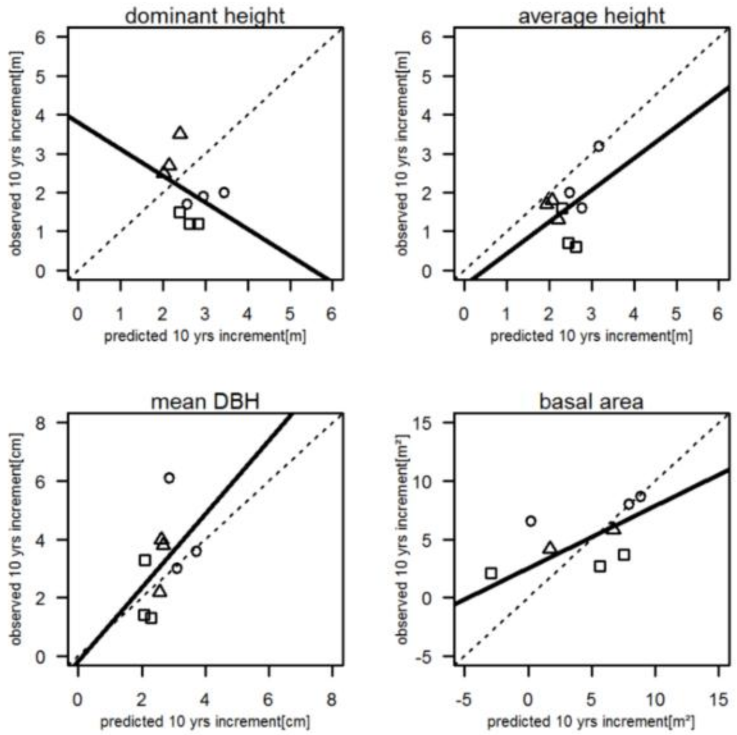


**Figure SM-2.** Predicted versus observed periodic increments (1961-1970, 1971-1980, 1981-1991) of stand attributes of three study plots in from Neumann (1993). Squares = stand 1, circles = stand 2, triangles = stand 3.

## 8.3 PAPER 3

Irauschek F, Rammer W, Lexer MJ (2017) Evaluating multifunctionality and adaptive capacity of mountain forest management alternatives under climate change in the Eastern Alps. European Journal of Forest Research 136:1051–1069.

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ORIGINAL PAPER

# Evaluating multifunctionality and adaptive capacity of mountain forest management alternatives under climate change in the Eastern Alps

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**Abstract** Future provisioning of ecosystem services (ES) from mountain forests is uncertain due to potential impacts of climate change. For a case study catchment in the Eastern Alps in Austria we analysed how management and climate change may affect the provisioning of four ES (timber production, carbon sequestration, biodiversity and bird habitat quality, and protection against gravitational hazards). We used the PICUS forest ecosystem model to project seven management alternatives that differed with regard to cutting pattern size (SLIT, PATCH, STRIP) and two harvesting intensity levels (in terms of return interval) under historic climate and five transient climate change scenarios over 100 years. In addition no management and sanitary management were simulated. In total twelve indicators were linked to model output to quantify ES provisioning. Results under historic climate showed increased volume and carbon stocks in low-intensity management, while high-intensity management decreased stocks. Bird habitat quality was maintained only by low-intensity management using SLIT and PATCH cuts. In particular rockfall protection decreased strongly under the STRIP cut scenario. Improved tree growth in warming scenarios was counterbalanced by increasing damage from bark beetle disturbances. Canopy openings and increased deadwood supply from disturbances partly fostered bird habitat quality in no-management alternatives. Overall none of the management alternatives performed best for all ES. PATCH and SLIT regimes at (currently practiced) low intensity appeared as compromise to achieve multifunctionality at small scale. As involved trade-offs among ES can be substantial, partial segregation with priority on specific services in designated zones is recommended.

**Keywords** Mountain forest management · Ecosystem services · Multifunctionality · Climate change · Simulation

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## Introduction

Forests in the European Alps provide a wide variety of ecosystem services (ES) for owners and society by creating income through timber production, protecting infrastructure from gravitational natural hazards, providing high-quality drinking water and mitigating climate change through the uptake and storage of carbon (Buttoud 2000; Price et al. 2000; Dorren et al. 2004; Nabuurs et al. 2014). Moreover mountain forests are important aesthetical assets for tourism (Nepal and Chipeniuk 2005). Despite intensive use for timber, fuelwood and livestock grazing since centuries, substantial shares of mountain forests in the Eastern Alps have been maintained in relatively natural state and are hotspots of biodiversity (Nagy et al. 2003; EEA 2010).

As a result of manifold demands for ecosystem services multifunctionality has been the main paradigm of forest policy and management in many European mountain ranges (e.g. Buttoud 2000; Hanewinkel 2011). However, the conceptual framework of multifunctional forest management aiming at integration of a bundle of ecosystem services at relatively small spatial scale (i.e. stand scale, comprising a few hectares) has rarely been made operational so far and lacks an explicit and transparent definition of management objectives and planning procedures for multiple ecosystem services (Suda and Pukall 2014). While recently an increasing number of studies on ES provisioning by mountain forests have become available, knowledge about ES trade-offs in dependence of forest management regimes is still limited (Briner et al. 2013; Häyhä et al. 2015; Uhde et al. 2015). While negative impacts of large-scale cutting pattern on place-based ES provision such as protection against rockfall and avalanche release are obvious (compare Dorren et al. 2004; Cordonnier et al. 2008), for other ES the relation between forest management and service provisioning levels may not be as straight forward.

Recently the potential impacts of climate change on forest growth, structure and composition and related ES provisioning have received a lot of attention (Carpenter et al. 2009; Rounsevell et al. 2010). Among various climate change impact mechanisms in particular abrupt tree mortality due to intensifying disturbance regimes has the potential to strongly impair the ability of forests to continuously provide ES such as timber production, carbon storage and protection against gravitative natural hazards (Thom and Seidl 2016). Homogeneous forests with high shares of conifers and poor structural and tree species diversity appear to be vulnerable to intensifying biotic disturbance regimes (Seidl et al. 2011; Griess et al. 2012; Temperli et al. 2012; Pasztor et al. 2014; Hlásny et al. 2015).

Current management regimes in European mountain forests are diverse and include, among others, stripwise clear cuts, shelterwood systems and various patch cutting approaches (cf. Nyland 1996; Burschel and Huss 2003). Local practices are often based on socio-economic and technological development and tradition (Heinimann et al. 2001; Brang et al. 2014; Bugmann et al. 2017). Current recommendations for the Eastern Alps emphasize small-scale cutting pattern to heterogenize the horizontal mosaic of stand structures (Ott et al. 1997; Motta and Haudemand 2000; Höllerl 2008) and are considering multiple ecosystem services at small spatial scale (i.e. stand scale, a few hectares). These recommendations have evolved as a result of practitioners' expertise and societal and stakeholders' demands towards forest ecosystems. However, under changing climatic conditions and intensifying disturbance regimes, experiences of the past may not be suitable any more. Because of low growth rates and high costs for management interventions in steep terrain, return intervals in mountain forests are longer compared to lowland forests. Considering the long lead times timely implementation of adaptive measures therefore is particularly crucial (e.g. Seidl et al. 2011) despite the uncertainty in magnitude and detailed properties of climate change as well as in future societal preferences for ES (e.g. Skourtos et al. 2010).

There is an urgent need to evaluate how different silvicultural regimes (i.e. cutting pattern for harvesting timber and tree regeneration) affect ES provisioning, and whether such management regimes are suitable to sustain ES provision under conditions of climate change. To achieve these goals and to provide science-based decision support in planning and decision-making, experiments utilizing advanced ecosystem modelling including disturbances and consideration of multiple ES are required (Wolfslehner and Seidl 2010; Muys et al. 2011).

In this contribution we set out to analyse the effects of alternative silvicultural management regimes and climate change on a set of ES (timber production, carbon storage, biodiversity and nature conservation, protection against gravitative hazards) for a catchment in the Eastern Alps in Austria employing a simulation-based scenario approach. Specifically the following questions will be addressed:

1. What are the effects of different silvicultural cutting patterns and turnover times on ecosystem services provisioning?
2. What is the impact of climate change on ecosystem service provisioning?
3. Are there interactions of climate change and management effects on ecosystem service provisioning?

## Methods and material

### Study area

The study area is situated in Vorarlberg in western Austria, in a side valley of the Montafon (47.08N; 9.82E). The forest is property of the Stand Montafon Forstfonds, an association owned by municipalities, that holds about 6500 ha of forest land in the region (Fig. 1a).

On crystalline and calcareous bedrock cambisols and rendzinas occur as main soil types. The current climate in the study region is characterized by low temperatures (long-term mean annual temperature at 1300 m a.s.l. is 6.2 °C) and annual precipitation of 1486 mm with 901 mm occurring in the growing season from April to September. The natural forest types along an elevation gradient are mixed broadleaf forests of beech (*Fagus sylvatica* L.) with admixed maple (*Acer pseudoplatanus* L.) and ash (*Fraxinus excelsior* L.) at the valley bottom (approx. 800–900 m), mixed conifer forests of Norway spruce

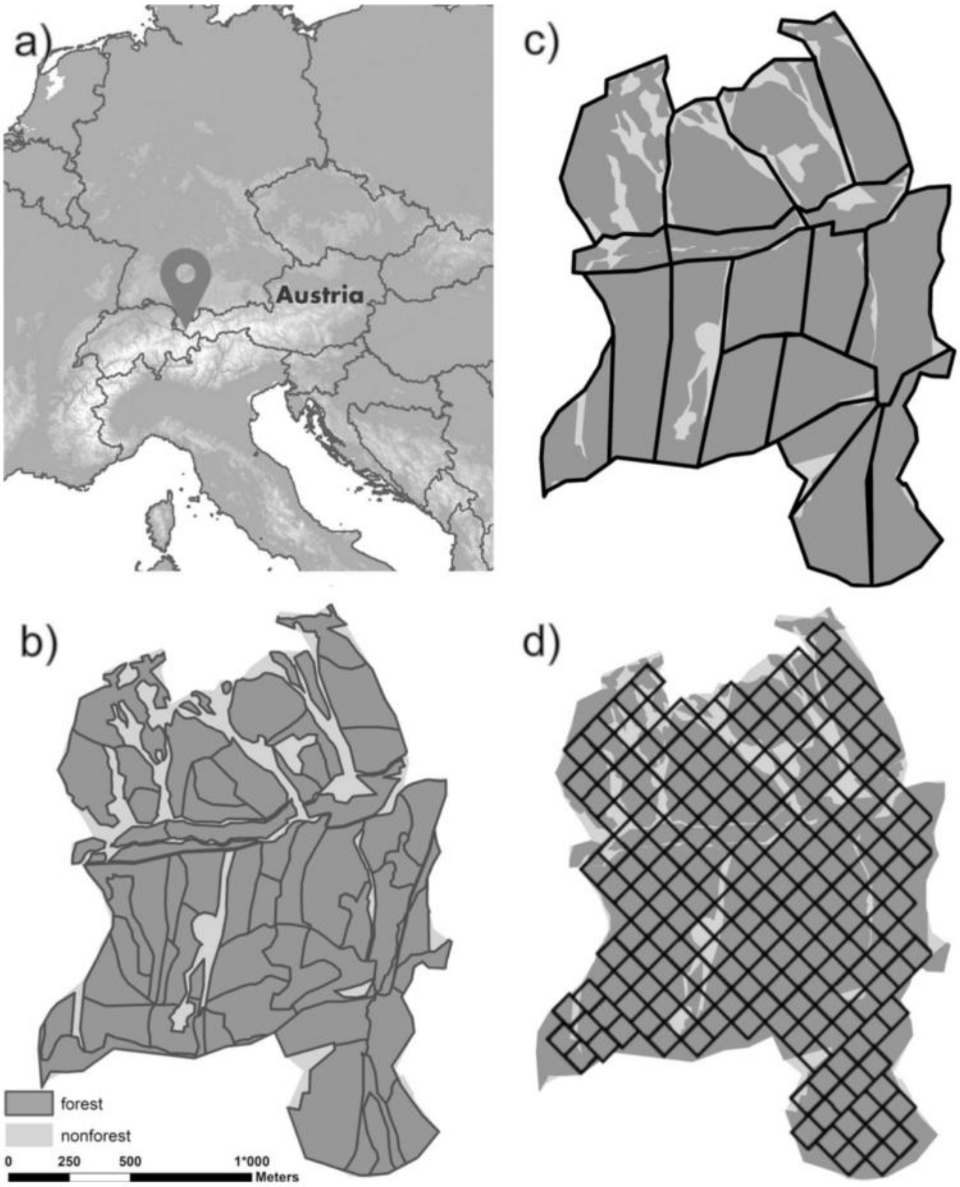


**Fig. 1** Location of the study area (**a**), 53 stand polygons for establishing initial forest conditions (**b**), 18 harvesting units (HU) (**c**), and 1-ha grid cells ($n = 220$) for the analysis of ecosystem service provisioning (**d**)

[*Picea abies* (L.) Karst.] and fir (*Abies alba* Mill.) with some admixed maple at intermediate elevation levels (approx. 900–1500 m), and spruce up to the timber line (approx. 1800 m).

The region has a forest management history since at least 500 years (Bußjäger 2007). Land use has shaped landscape and forest structure through timber harvests and livestock grazing, which shifted the tree line downhill for alpine pasturing. In the last decades forest grazing has been abandoned in the study area (Malin and Maier 2007). Traditional forest management has mainly used patch cuts for timber harvests and stand regeneration, which resulted in uneven-aged forest structures in most stands. Due to low management intensity (partial cuts with long return intervals) the share of large old trees is considerable. Steep slopes and rough topography require timber harvesting based on skyline logging techniques. Forest and game management has favoured spruce, which is currently holding 88% of basal area share. Due to considerable densities of wild ungulates browsing pressure on regeneration of admixed tree species (i.e. fir, beech and maple) is high.

Current management objectives are to generate income from timber harvests while securing protection against landslides, rockfall and snow avalanche release (Malin and Lerch 2007). Major shares of the forest area are under Natura 2000 regulations, which aim at protecting habitat for black woodpecker (*Dryocopus maritimus* L.) and three-toed woodpecker (*Picoides tridactylus* L.) (Grabherr 2000). For the current study a catchment of 270 ha at elevations

**Table 1** Climate scenario characteristics for the case study at elevation 1300 m a.s.l; c0 = historic climate data (1961–1990), c1–c5 = changes in mean annual temperature (°C) and precipitation (rel. change) in scenarios c1–c5 in relation to c0

| Climate scenario | Annual mean temperature | Annual precipitation | Summer precipitation | |
|---|---|---|---|---|
| **c0** | **6.2 (°C)** | **1486 mm** | **901 mm** | **CV = 16%** |
| c1 | +2.6 | +20% | +13% | 22% |
| c2 | +2.8 | +2% | −8% | 26% |
| c3 | +3.4 | +7% | −5% | 18% |
| c4 | +4.2 | −1% | −1% | 24% |
| c5 | +5.9 | −9% | −25% | 32% |

Summer precipitation = April to September, CV = coefficient of variation

between 1060 and 1800 m a.s.l. has been selected because of demand for various ES by the landowner and the public, existing interest in active management and because Norway spruce is by far the most dominant tree species (see Fig. 1b).

## Climate data

Historic climate data and five transient climate change scenarios covering 100 years up to 2100 were used as model input. They consist of daily values for temperature, precipitation, vapour pressure deficit and radiation. For the historic climate daily instrumental data for the period 1961–1990 from a nearby weather station were adjusted for site conditions in the case study regarding elevation and slope, utilizing algorithms from Thornton et al. (2000). Five climate change scenarios were based on simulations from the ENSEMBLES project (Hewitt and Griggs 2004), employing the A1B greenhouse gas emission scenario (IPCC 2007). For details on the downscaling approach see Bugmann et al. (2017). Climate scenario characteristics are shown in Table 1.

## Forest model

We used the forest ecosystem model PICUS v1.5 (Lexer and Hönninger 2001; Seidl et al. 2005). PICUS v1.5 (henceforth referred to as PICUS) is a hybrid of classical gap model components (PICUS v1.2; Lexer and Hönninger 2001)) and a process-based stand-level net primary production model (3PG; Landsberg and Waring 1997). In PICUS growth, regeneration and mortality of individual trees are simulated utilizing a grid of 10 × 10 m patches, topped by layers of crown cells of 5 m height each. A three-dimensional light model, used to estimate absorbed radiation for each tree, is a key component to connect the gap model elements to the production model. For detailed description of core model properties see Seidl et al. (2005). PICUS was successfully tested (Didion et al. 2009; Huber et al. 2013) and applied in various studies in Alpine forest ecosystems (Seidl et al. 2007b, 2011; Maroschek et al. 2015; Irauschek et al. 2017). The model provides accurate tree growth projections and furthermore captures key ecosystem processes such as spruce bark beetle (*Ips typographus* L.) disturbances, deadwood decay and tree regeneration including browsing by ungulates.

## Forest initialization

A total of 53 stand polygons were delineated on a high-resolution orthophoto for establishing the initial forest state. Based on a 50 × 50 m raster a terrestrial inventory was carried out with at least five sample points per polygon using angle-count sampling (Bitterlich 1952) to collect information about diameter distribution and height–diameter relationship of trees above 5 cm diameter at breast height (DBH). With concentric fixed radius plots regeneration structure and browsing damage of all smaller trees (>10 cm height and <5 cm DBH) were measured. Soil type, soil depth and water holding capacity were recorded from soil pits as described in Seidl et al. (2009). To establish a spatially explicit initial tree map for the 53 stand polygons (Fig. 1b), information from the inventory was combined with a LiDAR-based volume map and a normalized crown model (Hollaus et al. 2006, 2007). For each polygon trees are drawn from the related diameter distribution and assigned to that 10 × 10 m patch of the virtual forest where the overall match with regard to (a) maximum, minimum and mean tree height from the crown model is highest, (b) total accumulated tree volume per patch does not exceed the corresponding value from the volume map and (c) total crown projection area of all trees on a patch does not exceed a specified maximum value (cf. Maroschek et al. 2015). For this study the landscape was divided into 18 harvesting units with sizes of 4–20 ha (Fig. 1c), which served as simulation entities. Main rationale for the delineation of the harvesting units was topography, which determines location of skyline tracks for efficient timber harvesting.

## Forest management

Due to the steep terrain in the case study area trees are felled, delimbed and cut to length by chainsaw, and extracted to forest roads by cable yarding systems. A total of 131 skyline tracks were set out in the study area, considering topography and possible landing sites on forest roads. In all management scenarios timber harvests are bound to these skyline tracks. The suite of tested management alternatives included the currently practiced management approach as well as most of the currently discussed approaches for Norway spruce-dominated mountain forests (Ott et al. 1997; Streit et al. 2009; Brang et al. 2014). The management alternatives differ mainly in the spatial cutting pattern along the skyline tracks and the return interval. For a harvesting operation the skyline track of 5 m width is cleared of all trees >10 cm DBH, and logging is implemented in a corridor of 25 m maximum lateral skidding distance (see Fig. 2).

The currently practiced harvesting operations (PATCH) feature irregularly shaped patches of 1500–2000 m$^2$. To study effects of different cutting areas and pattern we included a slit cut scenario (SLIT), with small openings of 300 m$^2$, and a strip cut scenario (STRIP), where harvests are concentrated on 5000 m$^2$ strips along the skyline (Fig. 2; Table 2). Slit cuts are frequently recommended in mountain silviculture to initiate regeneration in protective forests (Streit et al. 2009), while strip cuts result in lower harvesting costs and more efficient artificial regeneration measures. We compared the three cutting patterns under current (LO) and increased (HI) harvesting intensity (cf. Table 2). In current management (PATCH-LO) each year on average 0.83 ha (i.e. 0–2 skyline tracks) is subject to regeneration fellings, resulting in a theoretical complete area turnover of 250 years and an average return interval to the same skyline track of 84 years. Scheduling and selection of harvesting operations depend on tree maturity. Hence areas with older trees and higher stocking volume are harvested earlier. To avoid strong visual impacts harvesting operations are not implemented in adjacent skylines until tree regeneration is safely established. To represent increased harvesting intensity (HI scenarios) the return interval is reduced, so that the required area turnover time is 150 years. The general schedule and layout of the skyline tracks are the same in all scenarios. Moderate changes in sequence and timing of skylines are due to the adjacency constraint.

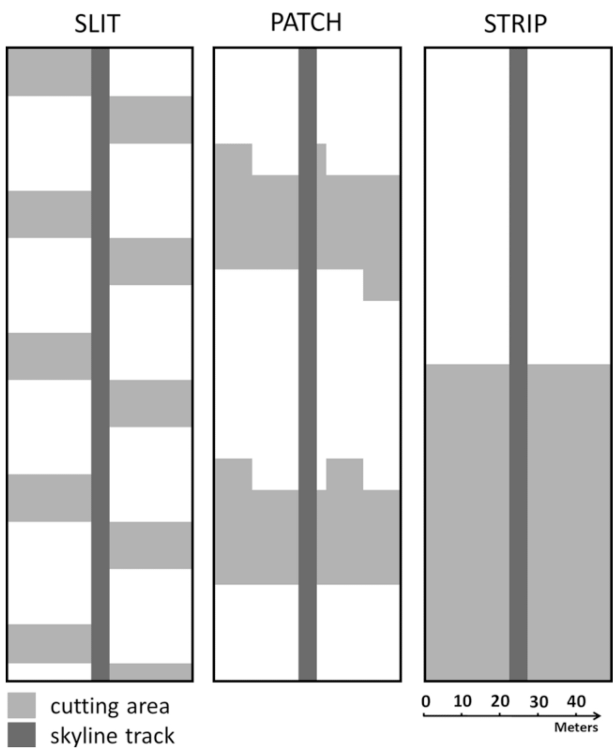


**Fig. 2** Simulated harvesting pattern along skyline corridors

Two management variants were simulated using artificial regeneration: in STRIP-HI-P a mix of spruce, larch (*Larix decidua* Mill.), maple and fir is planted after felling operations in regular raster spacing of 2 × 2 m. Browsing effects (see "Browsing of tree regeneration by ungulates" section) are switched off for planted seedlings to mimic tree protection measures. In a sanitary felling scenario (SAN) bark beetle-infested spruce trees are felled, but remain on site after debarking. In the resulting gaps sycamore maple is planted and protected against browsing. For comparison a no-management regime (NOM) is simulated without silvicultural interventions and assuming current browsing probabilities. Details of the management alternatives are shown in Table 2.

### Browsing of tree regeneration by ungulates

Based on the inventory (see "Climate data" section) the annual browsing probability for regeneration was calculated. In the simulation the browsing probability determines the fraction of simulated patches with available regeneration of a particular tree species that is browsed in any given year. Browsed patches are drawn randomly, and the height growth of the browsed species in that respective simulation year is set to zero. If a seedling is browsed in successive years (i.e. has zero height growth) it falls below the minimum growth requirement of the mortality algorithm and a fraction of saplings of the browsed species on that patch dies (see Table SM2-1). In all simulated management scenarios natural regeneration is subject to browsing.

### Ecosystem service indicators

Cordonnier et al. (2013) presented a comprehensive set of indicators for ES provisioning by mountain forests as a

**Table 2** Key characteristics of management scenarios. Share harvested per set-up = share of skyline corridor area harvested per entry

| Short name | Description | Cutting pattern ($W \times L$) [m] | Share harvested per set-up | Return interval | Area turnover [year] | Regeneration |
|---|---|---|---|---|---|---|
| SLIT-LO | Slit cuts low intensity | 30 × 10 | 1/3 | 84 | 250 | Natural |
| PATCH-LO | Irregular patch cuts low intensity | 50 × 30–40 | 1/3 | 84 | 250 | Natural |
| STRIP-LO | Strip cuts low intensity | 50 × 100 | 1/2 | 126 | 250 | Natural |
| SLIT-HI | Slit cuts high intensity | 30 × 10 | 1/3 | 50 | 150 | Natural |
| PATCH-HI | Irregular patch cuts high intensity | 50 × 30–40 | 1/3 | 50 | 150 | Natural |
| STRIP-HI | Strip cuts high intensity | 50 × 100 | 1/2 | 76 | 150 | Natural |
| STRIP-HI-P | Strip cuts high intensity + planting | 50 × 100 | 1/2 | 50 | 150 | Natural + planting spruce, larch, maple, fir |
| NOM | No management | – | – | – | – | Natural |
| SAN | Sanitary management | – | – | – | – | Natural + planting maple |

major outcome of the ARANGE FP7 project (Advanced Multifunctional Forest Management in European Mountain Ranges). For the current study a total of 12 ES indicators were selected to represent the provision of timber, carbon storage, biodiversity and bird habitat, and protection against the gravitational hazards rockfall, snow avalanches and landslides. Below a brief description of the indicators is provided; for details see Supplementary Material SM1.

Timber production was represented by volume of living trees (V), harvested commercial volume (TVH), periodic net volume increment (VI) and volume of trees killed by bark beetles (BBD). Carbon sequestration was assessed by total carbon (C), including carbon in tree biomass, standing deadwood, woody debris and soil carbon. Biodiversity and nature conservation were represented by tree species diversity (DSP), tree size diversity (DSI) and bird habitat quality (BHQ). The latter characterized habitat for woodpeckers which are considered as important umbrella species for nature conservation (Drever et al. 2008). BHQ is a composite indicator considering standing large deadwood, large living trees and canopy cover. It was provided on an ordinal scale (good, medium, poor). Protection against gravitational hazards was represented by four indicators: an avalanche protection index (API) expressed the ability of a stand to prevent avalanche release; a rockfall protection indicator (RPI) (for rock diameter 0.46 m) quantified the percentage of falling rocks passing through a forest stand; and a landslide protection index (LPI). API was based on basal area and mean DBH, RPI used stem number, basal area and mean DBH as stand characteristics for input, and LPI was calculated from canopy cover of trees >5 cm DBH. API and RPI were measured on interval scale [0–1], where values of 1 indicate an optimal protective effect; LPI was provided on ordinal scale (good, medium, poor).

In addition to these ES indicators a regeneration indicator (REG) was provided for all simulation runs characterizing regeneration dynamics and indicating availability of young trees for the further development of the forest. REG specified the area share of the landscape with at least 50 saplings per ha in height class 80–130 cm.

## Assessment approach

The nine management alternatives (Table 2) were each simulated under historic climate and five climate change scenarios (Table 1). To account for the variation in indicator results due to stochastic algorithms in the PICUS simulation environment, each simulation unit was simulated ten times. Altogether for each of the 18 simulation units 540 runs were conducted (9 × 6 × 10). Model output (single tree attributes) was mapped onto a digital elevation model and analysed utilizing the PICUS landscape assessment tool (see Maroschek et al. 2015). First the mean values from the ten replicates for all intermediate variables feeding into indicator calculation were prepared. Then the indicators for timber production and carbon sequestration were aggregated at the level of the 18 harvesting units (mean values), and finally an area weighted average was calculated at landscape scale. Indicators for biodiversity, nature conservation and protection against gravitational hazards which require a smaller spatial context were calculated for 220 1-ha pixels (see Fig. 1d). For the interval scaled protection indicators (API and RPI) and the tree diversity indicators (DSP and DSI), the 10th, 50th and 90th area percentiles (P10, P50, P90) were calculated to characterize the worst, the mean and the best pixel values of the catchment. For the categorical indicators LPI and BHQ the landscape area percentage of the three categories was calculated.

To represent the development of ES provisioning over time for further analysis, the indicators were provided in three assessment periods (period P1: 2000–2032, period P2: 2033–2066, period P3: 2067–2100), as periodic mean, modal or minimum value (for details see Table SM1-1).

Before analysing the effects of climate change and management on ES provisioning, we evaluated the model-induced variation in ES indicators. Simulated indicator values vary depending on the interplay of stochastic model elements, which in turn affect forest structure and composition. Per management × climate combination the coefficient of variation (CV) from the ten replicates was calculated for the indicators in period P3. The categorical indicators LPI and BHQ were included as the area share rated in category "good". To test the effects of management, climate and their interaction in P3 on (1) ES indicators and (2) CVs of indicators, we employed ANOVA tests using general linear models and Tukey tests. All statistical tests were conducted with the R language and environment (R Core Team 2016) utilizing the "TukeyC" package (Faria et al. 2016).

## Results

First we briefly compare the temporal development of ES indicators under different management alternatives and historic climate to reflect the transient behaviour of ES provisioning in the study landscape. Then we focus on the assessment period P3 and show how management and climate affect the long-term development of ES indicators. Finally we synthesize the results and compare all management alternatives with regard to simultaneous provisioning of all ES. An evaluation of the model-induced variation in ES indicators as a prerequisite to interpret differences in ES indicators among scenarios is provided in Supplementary Material (SM3).

### Temporal development of ES provisioning

Forests in the study region were not in steady state conditions, but evolved and showed transient behaviour over the 100-year analysis period. All scenarios started from the same initial forest conditions, and differences in ES indicator trajectories within a specific climate scenario are due to effects of management alternatives. Figure 3 shows the temporal development over three assessment periods for all indicators under historic climate. Harvested volume per hectare and year (TVH) increased over time in all management scenarios. Reasons were the predefined sequence and constraints of the skyline harvesting pattern, which fixed the amount of harvested area per period (see "Forest management" section), and the increasing mean volume stock on the scheduled cutting areas. Mean periodic volume increment (VI) increased as well because, as a result of the cutting activities and related regeneration, the average tree age decreased. In contrast in NOM and SAN alternatives (i.e. management scenarios without timber harvests) forest growth started to decline in the final assessment period due to overmature trees. Under the historic climate damage by bark beetles (BBD) was similar under all management alternatives (0.4–0.5 $m^3$ $ha^{-1}$ year$^{-1}$) and did not change much over time (Fig. 3). Stocks of volume (V) and carbon (C) increased over time in all low-intensity management alternatives as well as in the NOM and SAN scenarios. In high-intensity scenarios volume and carbon stocks were lower in period P3 compared to the initial period P1. Interestingly species diversity (DSP) showed a moderately decreasing trend under all management regimes except in the STRIP-HI-P alternative with planting. Tree size diversity (DSI) increased in general. However, in STRIP-HI and SAN management this development was dampened in P3, while in the NOM scenario tree size diversity did not increase at all due to missing active regeneration initiation. Bird habitat quality (BHQ) was maintained in PATCH-LO and SLIT-LO scenarios (approx. 50% of the area rated as "good"; Fig. 3), whereas in all other management alternatives bird habitat quality showed an unfavourable trend. While protective effects against avalanche release (API) and landslides (LPI) in general improved under all management regimes, rockfall protection (RPI) improved only in low-intensity and no-harvest variants (NOM and SAN). In STRIP-LO and all high-intensity scenarios rockfall protection decreased in the intermediate period P2 only to recover in period P3 to approximately the initial values. The regeneration indicator (REG) revealed one general trend: the area share with regeneration increased until period P2 and dropped then in most management scenarios to the level of period P1 or even less. Only the small-scale alternatives SLIT-HI and PATCH-HI maintained high area shares with regeneration in period P3.

### Long-term effects of management and climate on ES provisioning

Effects of management and climate scenarios on ES provisioning emerged and accumulated over time. ANOVA results showed that in the final assessment period P3 for all analysed indicators effects of management and climate, as well as the interaction of these two factors, were significant (Table SM4-1 in Supplementary Material). However, it is acknowledged that in many cases differences were rather small and from a management perspective not relevant.

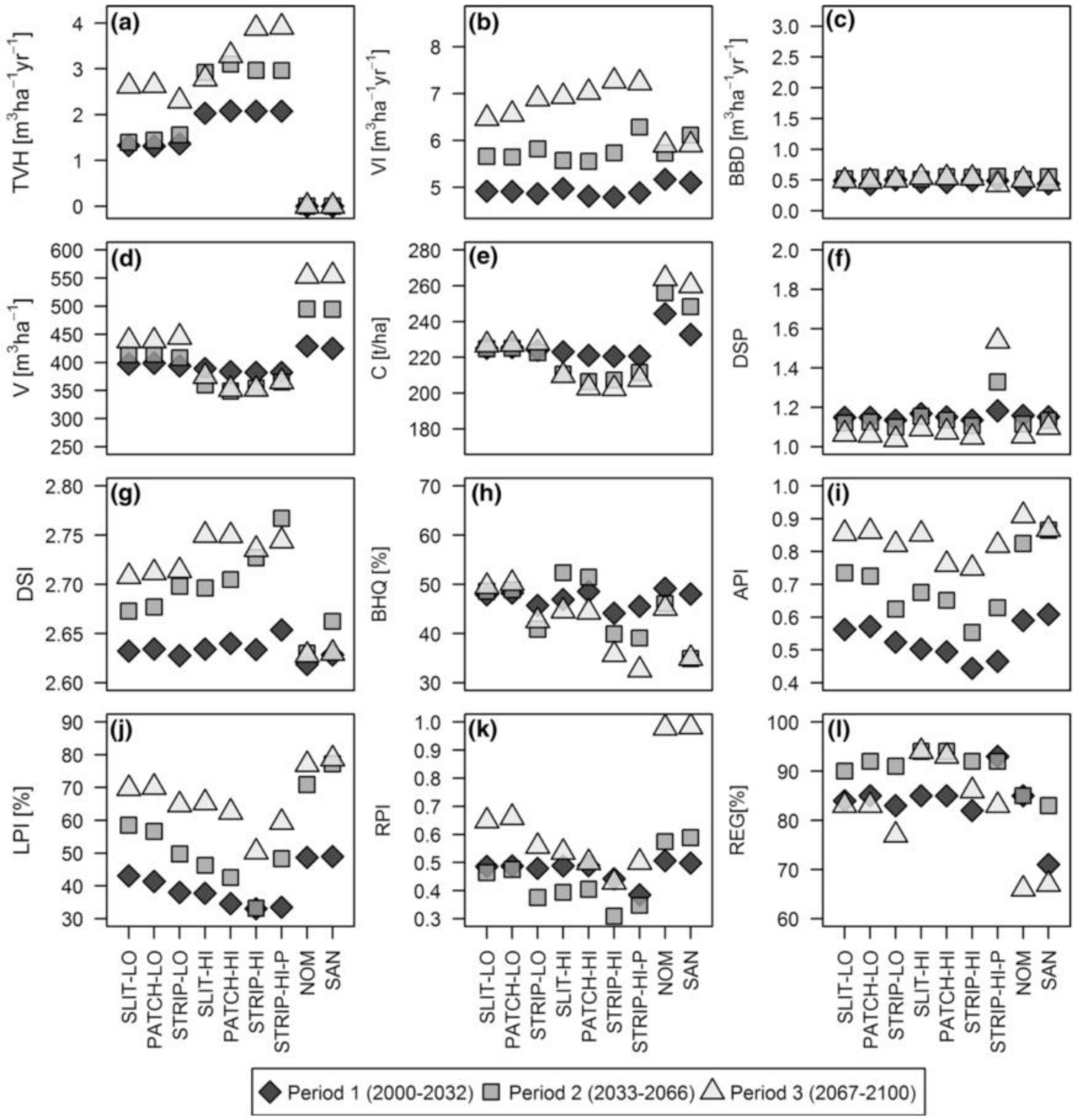


**Fig. 3** Temporal trend of ES indicators in nine management alternatives under historic climate (c0) over three assessment periods (P1, P2, P3) from 2000 to 2100. DSI is represented by 50th percentile, API and RPI by 10th percentile and LPI as share of area in category "good", all other indicators by mean values

Below results for all indicators are discussed by ES. Figure 4 shows all ES indicators in P3 for each management alternative under all climate scenarios. In Supplementary Material detailed indicator values are shown including Tukey test results for differences between management scenarios (Tables SM4-2 to SM4-7) and for differences between climate scenarios (Table SM4-8).

*Timber production*

In historic climate total harvested timber volume (TVH) over the entire 100 year assessment period under low management intensity ranged from 174 (STRIP-LO) to 180 $m^3$ $ha^{-1}$ (PATCH-LO). In general high-intensity scenarios harvested significantly more (on average 58–60%). Under climate change total harvested timber decreased, with the largest drop (i.e. 10%) occurring in climate scenario c5 (not shown). Looking at period P3 alone the same pattern emerged and harvested volume of high-intensity alternatives was on average between 39% (in historic climate) and 37% (climate scenario c5) higher than in the low-intensity alternatives (Fig. 3; Tables SM4-2 to SM4-7).

For volume increment (VI) two trends were apparent: First in high-intensity management productivity was significantly higher in all climate scenarios

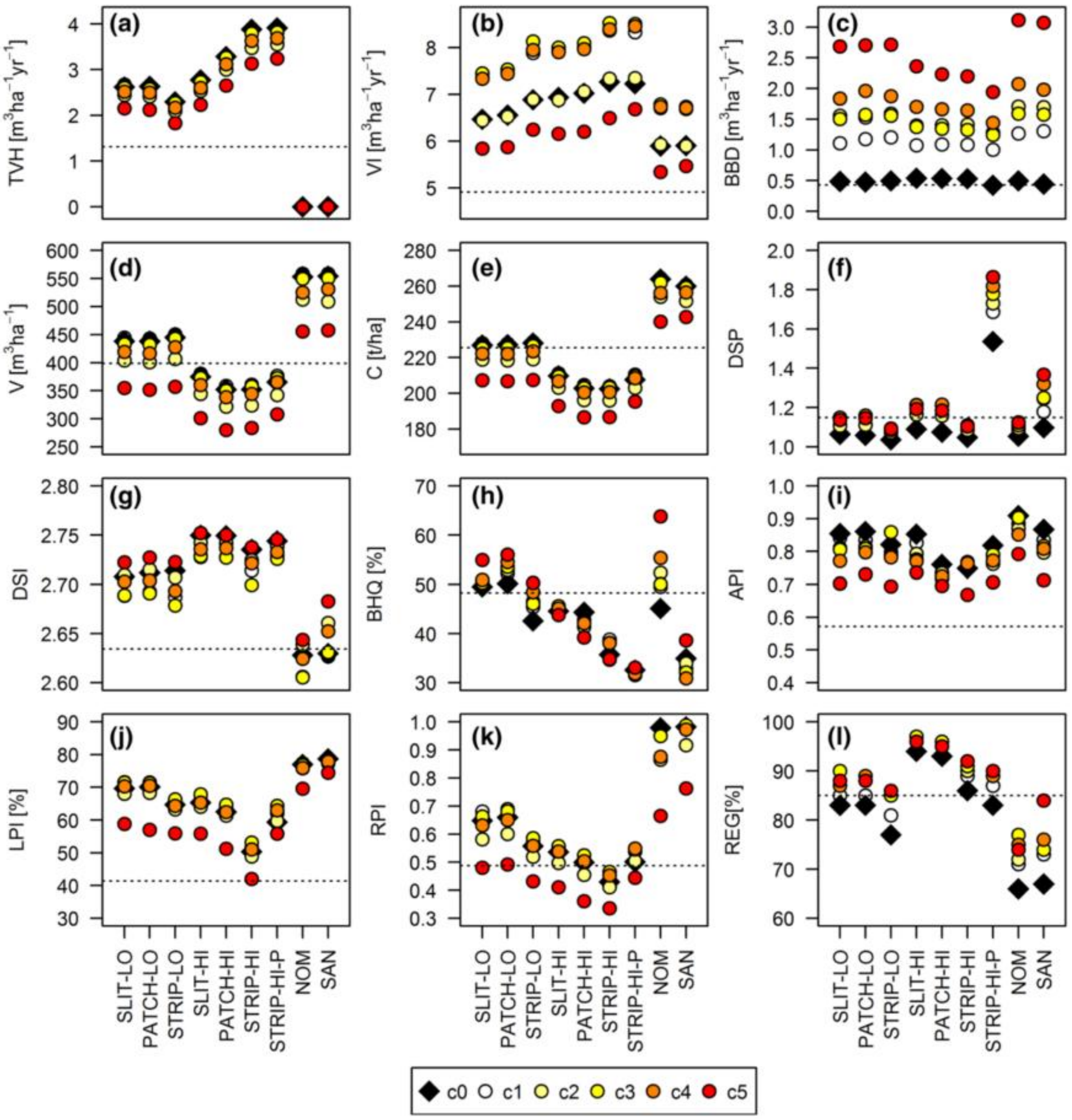


**Fig. 4** Ecosystem service indicators in nine management alternatives and six climate scenarios (c0 = current climate, c1–c5 are climate change scenarios). Shown are mean values for the period P3 (2067–2100). Corresponding values in P1 under PATCH-LO and historic climate c0 as reference are shown as dotted line. DSP and DSI are represented by 50th percentile, API and RPI by 10th percentile and LPI and BHQ as share of area rated as "good". (Color figure online)

[6.4 $m^3$ $ha^{-1}$ $year^{-1}$ (c5) to 8.3 $m^3$ $ha^{-1}$ $year^{-1}$ (c3)] compared to low-intensity management [6.0 $m^3$ $ha^{-1}$ $year^{-1}$ (c5) to 7.7 $m^3$ $ha^{-1}$ $year^{-1}$ (c3)] and to no-harvest alternatives NOM and SAN [5.5 $m^3$ $ha^{-1}$ $year^{-1}$ (c5) to 6.8 $m^3$ $ha^{-1}$ $year^{-1}$ (c3)]. The reason was the faster turnover of the initial tree population and consequently a resulting lower mean tree age. Second warmer temperatures of climate change scenarios fostered tree growth. Compared to historic climate net increment increased under all climate change scenarios except the strongest warming scenario c5, independent of the management alternative (Fig. 4). However, under the moderate warming scenario c2 gains were negligible and not significant (Table SM4-8).

Under historic climate damaged timber by bark beetles (BBD) was very similar for all management alternatives. Climate change conditions caused a strong increase in damages, particularly under scenario c5. There was a trend of higher damages in NOM and SAN, followed by low-intensity alternatives, while high-intensity management variants tended to have lower damages (Fig. 4). Overall the cutting pattern did not affect bark beetle damages significantly except in c5, where SLIT-HI had higher damages than PATCH-HI and STRIP-HI. Comparing NOM and

STRIP-HI-P it was obvious that the influence of management as a possible measure to reduce bark beetle damages [between 0.3 $m^3$ $ha^{-1}$ $year^{-1}$ (c1) and 1.2 $m^3$ $ha^{-1}$ $year^{-1}$ (c5) higher in NOM] was considerably lower than the expected damage increase due to climate change [between +0.7 $m^3$ $ha^{-1}$ $year^{-1}$ (c1) and +2.7 $m^3$ $ha^{-1}$ $year^{-1}$ (c5)] (Table SM4-2 to SM4-7).

Volume stock is closely related to all other timber indicators. Under historic climate NOM and SAN accumulated 553–554 $m^3$ $ha^{-1}$ volume stock, while all managed scenarios resulted in significantly lower standing volume, with high-intensity scenarios always lower than low-intensity scenarios (lowest volume in PATCH-HI and STRIP-HI: 352 $m^3$ $ha^{-1}$). Impacts of climate on volume stocks ranged from slight increases (+5 $m^3$ $ha^{-1}$ in scenario c1) to significantly reduced volume (scenario c2: −8%, c4: −4%, c5: −20%; average values over all management alternatives; see Table SM4-2 to SM4-8).

*Carbon storage*

In historic climate under low-intensity management carbon stocks were on average 227 t $ha^{-1}$. High-intensity alternatives reduced carbon stocks significantly (−8 to −11%), while no-harvest scenarios significantly increased stocks (+14 to +16%) (Table SM4-2).

Climate change resulted in small increases in carbon under scenario c1 and minor negative effects in climate scenarios c3 and c4 (Fig. 4). Under conditions of climate change scenarios c2 and c5 carbon stocks were significantly reduced (−2 to −4% in c2, and −6 to −9% in c5, depending on management). The decrease was significantly different between high- and low-intensity and no-harvest scenarios, with strongest reduction under the latter and smallest in high intensity management scenarios (Tables SM4-2 to SM4-8).

*Biodiversity and nature conservation*

Under historic climate the differences in tree species diversity (DSP) between management alternatives with natural regeneration were rather small, although partly significant [e.g. 50th percentiles (P50) of DSP from 1.04 to 1.09; Table SM4-2]. Independent of management intensity SLIT and PATCH cutting pattern promoted slightly higher species diversity compared to STRIP alternatives. Not surprisingly the alternatives including planting of broadleaves (STRIP-HI-P and SAN) yielded the highest species diversity (DSP P50 of 1.54 and 1.10, respectively). Species diversity in NOM was similar to the managed scenarios (DSP P50 of 1.05). In all management alternatives the highest species diversity occurred in climate scenarios c4 or c5 (Tables SM4-2 to SM4-7).

Also for tree size diversity (DSI) results under historic climate were fairly similar. In all low- and high-intensity scenarios DSI P50 ranged from 2.71 to 2.75. In NOM and SAN tree size diversity was moderately lower (DSI P50 of 2.63; Table SM4-2). Impact of climate change scenarios was very low (although partly significant; Table SM4-8), comprising of small decreases and small increases (the latter in NOM, SAN and LO alternatives under climate c5). For DSI the impact of climate change was generally lower than the effect of management.

Under historic climate PATCH-LO and SLIT-LO provided the highest share of suitable bird habitat (BHQ) with 49.6 and 50.2% of landscape area in the highest category (Table SM4-2). All other alternatives provided significantly less suitable habitat with STRIP-HI variants (35.7% for STRIP-HI and 32.6% for the planting variant STRIP-HI-P) and SAN (35.0%) producing the lowest area shares.

Climate change affected the area of suitable bird habitat differently depending on management. In general climate change caused more standing deadwood and canopy openings in dense areas of the landscape. As a result low-intensity alternatives, NOM and SAN had best habitat conditions under the strongest warming scenario c5. Interestingly under climate scenario c5, NOM provided highest shares of suitable bird habitat. In high-intensity variants highest area share of good bird habitat was generated under milder climate change scenarios (Table SM4-8).

*Protection against gravitational hazards*

While 50th and 90th percentiles of the avalanche protection indicator (API P50 and API P90) remained at optimum value of 1.0 under all management and climate scenarios, the worst 10% (API P10) responded negatively to climate change scenarios (Table SM4-8). NOM was best under all climate scenarios (API P10 between 0.88 in scenario c0 and 0.79 in scenario c5). The SAN scenario showed a reasonable performance under historic climate (API P10 of 0.87, not significantly different from NOM), but fell back under climate change scenario c5 (API P10 of 0.71). PATCH-HI and STRIP-HI were constantly the least performing management alternatives (Tables SM4-2 to SM4-7).

Management had pronounced long-term influence on landslide protection (LPI). Under historic climate management intensity and cutting pattern caused impacts of approximately the same order of magnitude. At both intensity levels STRIP produced significantly lower area share in category "good" compared to SLIT and PATCH management (Table SM4-2). Generally low-intensity variants provided better landslide protection. SLIT-HI was best among the high-intensity variants and not significantly different from the worst variant under low intensity (STRIP-LO). When comparing STRIP-HI and STRIP-HI-P significant positive effects of artificial regeneration were observed. However, landslide protection remained below PATCH and SLIT variants.

Compared to historic climate (c0) severe climate change (c5) yielded negative impacts on LPI under all management alternatives. Under conditions of c5 the NOM alternative still provided 69.6% of landscape area in the best LPI category (Table SM4-7).

As indicated by P50 and P90 of the rockfall protection indicator (RPI) forest conditions were sufficient to protect against rocks of 46 cm diameter on large shares of the landscape regardless of climate scenario (Tables SM4-2 to SM4-7). However, the worst 10% of the landscape area (i.e. P10) provided just low protective effect against falling rocks (RPI P10 between 0.66 under PATCH-LO in scenario c0 and 0.34 under STRIP-HI in scenario c5). High-intensity management had adverse effects on rockfall protection, and STRIP alternatives were the least preferable cutting patterns. In HI management SLIT was significantly better than PATCH. NOM and SAN maintained high protective effect (RPI P10 between 0.99 and 0.87) under all climate scenarios except c5. In climate scenario c5 RPI P10 decreased to 0.67 (NOM) and 0.76 (SAN), respectively.

### Effect of management regimes on multifunctionality

In Fig. 5 performance profiles of management alternatives under historic climate (c0) and one selected climate change scenario (c5) summarize the provision of ES in the study area. It is apparent that no single alternative was best in all indicators. High-intensity management was generally preferable from a timber production perspective. However, the harvest rates were still below the increment and it was just under strong warming scenarios (e.g. c5) that climate-related tree mortality, including bark beetle damaged stems, resulted in a net loss in volume stocks. For protection services, carbon sequestration and partly for the biodiversity and nature conservation indicators, high-intensity alternatives were the least preferable options. NOM and SAN maintained high carbon stocks and provided better protection against gravitational hazards than any of the other alternatives. SLIT-LO and PATCH-LO provided best bird habitat, but were outperformed in all other services and indicators by either the respective high-intensity variant (for timber production) or one of the no-harvest alternatives (NOM, SAN). Regarding tree species and structural diversity there were just minor differences among alternatives. However, in none of the ES indicators low-intensity variants ranked last.

Essentially the performance profile under severe climate change conditions of scenario c5 appears similar. However, a major shift in ranks occurred with regard to damaged timber from bark beetle disturbances and in bird habitat provisioning. In NOM and SAN damages caused by the intensified disturbance regime were higher than in any other management alternative. However, disturbances had also positive effects on bird habitat quality by generating deadwood and opening the dense canopy. From Fig. 5 it is also apparent that the effect of management intensity on ES provisioning was larger than the effect of cutting pattern under both historic climate c0 and strong warming scenario c5. STRIP cutting pattern were negatively impacting protection services and bird habitat quality at low- and high-intensity level. High-intensity alternatives generated the largest area with tree regeneration, as indicated by the REG indicator. However, due to strong browsing pressure on admixed fir and broad-leaved species, the tree species diversity did not increase much (under climate change scenarios) or not at all (in historic climate).

## Discussion

### Study design and model limitations

The comprehensive scenario matrix in our simulation study covered 54 management × climate combinations and provided insight in potential future pathways of forest management in steep coniferous mountain forests and related ES provisioning. The silvicultural cutting patterns to harvest timber and to regenerate the forest include all major feasible options for cable yarding terrain. Patch and strip cuts concentrate timber harvests in space and time and avoid the diffuse dispersal of single stem removals of individual tree selection and shelterwood systems (Burschel and Huss 2003; Weinfurter 2013). This keeps damage to residual stems low and increases harvesting efficiency (Stampfer 2000).

Recently several studies (e.g. Temperli et al. 2012; Villa et al. 2014) emphasized the importance of scaling to bridge the gap between stand-level indicators and landscape level ES provisioning. Based on spatially explicit simulations in the current study different approaches to calculate ES indicators were used. In particular indicators that do not scale linearly with area, such as bird habitat quality and protection service indicators, require a specific spatial context. The 1-ha size of the raster cells that were used to calculate most ES indicators in the current study can be evaluated from two perspectives. Cutting areas of simulated PATCH and SLIT regimes with 1500 and 300 $m^2$, respectively, were several times smaller than the 1-ha analysis raster cells. Thus these raster cells were large enough to capture the spatially heterogeneous texture of mountain forests. The 1-ha cells were, on the other hand,

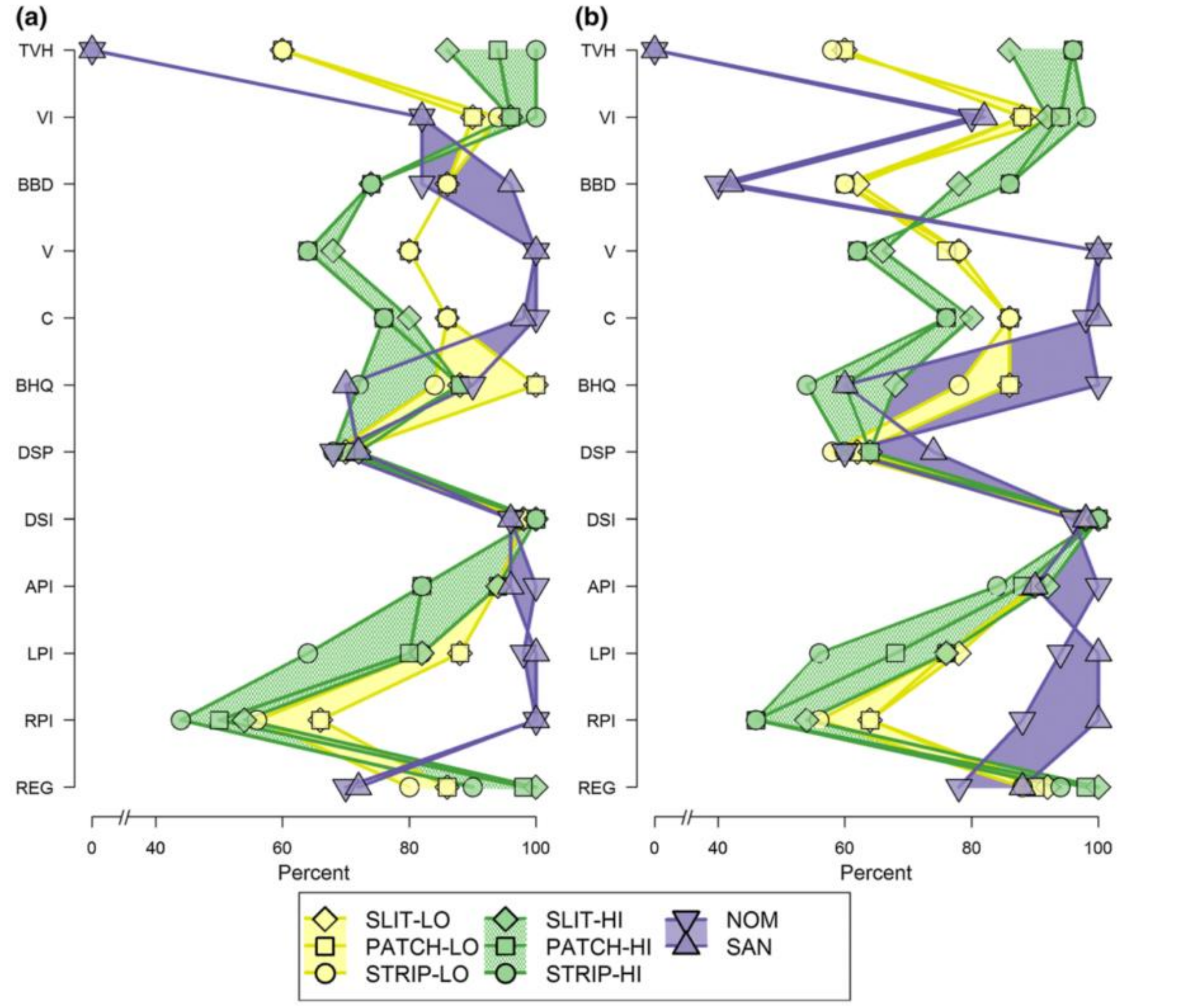


**Fig. 5** Ecosystem service indicators in period 3 relative to best management scenario according to preference direction. *Left plot* (**a**) shows results under historic climate and *right plot* (**b**) climate change scenario c5. *Colored band* depicts range for management intensities [low intensity (LO) = *yellow*, high intensity (HI) = *green* (pattern), NOM and SAN = *purple*). DSP and DSI are represented by 50th percentile, API and RPI by 10th percentile, LPI and BHQ as share of area rated as "good" and TVH as sum over 3 periods. (Color figure online)

small enough to provide a suitable spatial context for ES such as protection against gravitational hazards and habitat quality. However, representing the frequency distribution of 1-ha samples of an ES indicator by three different percentiles (P10, P50, P90) increased complexity and the need for more detailed explanations. In providing this spatial context in ES indicators we are in line with Grêt-Regamey et al. (2014) and Raudsepp-Hearne and Peterson (2016). Overall we believe that our approach improved ES quantification substantially compared to other studies (e.g. Marzluff et al. 2002; Elkin et al. 2013; Malek et al. 2015). Upscaling of stand-level ES indicators, by weighting indicator results with respective represented area in the landscape, does not provide unbiased estimates for ecosystem service indicators that do not scale linearly (e.g. Hlásny and Turčáni 2013; Hlásny et al. 2014). Other studies used non-spatial point models (Pabst et al. 2008; Mina et al. 2015) that cannot mimic effects of patch and slit cuts.

Regeneration dynamics are a key feature in mountain forests. In the PICUS model light is the major driver for the establishment and early growth of regeneration. The light regime on the forest floor is linked to management interventions and natural disturbances. While the light regime is considered at a high level of detail in the model, ground vegetation that may suppress tree regeneration is not considered explicitly. In mountain forests suitable regeneration niches are found at gap edges (e.g. Streit et al. 2009). If gaps in the canopy (i.e. regeneration cuts) get too large, the ratio of gap edges to gap area increases and most of the gap area will be populated with shade intolerant ground vegetation species such as *Rubus* sp. and grass species (e.g.

*Calamagrostis* sp.) that may impede tree regeneration. Particularly in the STRIP variants the simulated results for natural regeneration may thus be too optimistic.

### Disturbances as key driver of forest development in a warmer climate

Overall our study supports the theory that tree growth, in the cool–wet conditions observed in the subalpine region in the European Alps, is currently limited by temperature and may therefore profit from a warming climate (cf. Lindner et al. 2010; Elkin et al. 2013; Hartl-Meier et al. 2014; Helama and Sutinen 2016). Simulation results showed a significant increase in productivity in three out of five climate change scenarios compared to current climate. Nevertheless results for two climate change scenarios also revealed that changes in the precipitation regime can inhibit or even reverse this trend (cf. Büntgen et al. 2006). In addition the host–insect system of Norway spruce and the spruce bark beetle is highly sensitive to climatic changes, as both beetle development and host tree defences are dependent on climate variables (Netherer and Schopf 2010; Weed 2013). As a result bark beetle disturbances are expected to intensify in a warmer climate (Bentz et al. 2010) and have significant influence on development of spruce-dominated forests much earlier than direct effects of climate on forest growth. In the study area other disturbance agents, such as storms (cf. Schelhaas et al. 2003), have not been relevant so far and are not considered as important in the future (Kromp-Kolb et al. 2014). Our simulation results show that part of the warming-induced gain in volume increment was set off by tree mortality, particularly from bark beetle disturbances. We simulated a strong increase in frequency and intensity of disturbances in warmer climates, and in the strongest warming scenario (c5) disturbance damage at the end of the assessed period was as high as 45% of the periodic increment.

While under historic climate differences in damaged timber volume between management alternatives were negligible, under climate change scenarios the damages by bark beetles intensified and contrasts between management scenarios increased. Under climate change volume stocks and damages by bark beetle disturbances were positively correlated for three reasons: (1) in a warmer climate more beetles could develop because of faster life cycle completion, (2) higher stocks increased the susceptibility of stands to bark beetle damages (see Schroeder 2001; Stadelmann et al. 2013; Hlásny and Turčáni 2013) and (3) more volume is killed by beetles when stocks are high (e.g. Hanewinkel et al. 2011).

Nevertheless comparing the management scenarios with the lowest (STRIP-HI-P) and highest (NOM) bark beetle damages, only between 39% (c1) and 34% (c5) of the expected increase in damages may be prevented by forest management. Under climate change conditions NOM might not be a realistic option any more, due to legal requirements for proactive sanitary cuttings to prevent large-scale bark beetle outbreaks and spillover effects on nearby forests (Kautz et al. 2011b; Brůna et al. 2013). Large-scale implementation of selective sanitary management, as proposed in the SAN scenario, may be very costly and requires specialized personnel and support by remote sensing approaches to identify and locate bark beetle-infested trees (e.g. Stadelmann et al. 2013; Fassnacht et al. 2014).

Disturbances bear the potential to negatively influence provisioning of ecosystem services (Thom and Seidl 2016). Given the dominance of Norway spruce in the study area assessing future ES provisioning without consideration of the bark beetle disturbance regime would therefore provide unreliable results. However, in most of the more recent model-based climate change assessments in Central European spruce forests, bark beetle disturbances were not considered at all (e.g. Hlásny et al. 2014; Schuler et al. 2016; Mina et al. 2015). The PICUS bark beetle submodel used for this study (cf., Seidl et al. 2007a) is based on phenological principles of insect development and a predisposition approach to estimate damage risk. Compared to earlier PICUS stand-level applications (Seidl et al. 2011) the simultaneously simulated forest area was increased to about 20 ha in the current study. This corresponds well to the reported extent of consecutive bark beetle damages, where new infestations occur within distances of 250 m from previous attacks (Kautz et al. 2011a).

The simulated bark beetle-induced tree mortality was directly related to volume (e.g. Pasztor et al. 2014), which in turn correlates with carbon storage on site. Thus a similar trend over management alternatives and climate scenarios evolved. Compared to the average carbon pool of 201.1 t ha$^{-1}$ in Austrian forests (Weiss et al. 2000) only the high-intensity management alternatives under strong warming resulted in lower carbon storage in our study catchment. Under strong warming also no-harvest alternatives (NOM and SAN) were barely able to maintain initial carbon pools. Negative effects of climate change on carbon stocks through bark beetle infestations are partly buffered, because increased tree mortality first increased deadwood and subsequently soil carbon pools. Overall these results indicate that in situ mitigation potential of Norway spruce mountain forests under climate change is limited (compare Nabuurs et al. 2013).

In contrast to mean volume and carbon stocks impacts on protection services against gravitational hazards, bird habitat quality and tree diversity are much more depending on how damages (i.e. volume of trees affected per damage event) are distributed over the landscape and subsequently affect stand structure.

In the applied PICUS modelling environment tree mortality caused by bark beetle infestations and senescence was the main source of stochasticity in model results and thus in indicator uncertainty (see Supplementary Material). Accordingly simulation repetitions were necessary to analyse the variability of results. Under a warmer future climate beetle population build-up was faster and more homogeneous between years, and therefore, the variability of simulated damages was decreasing. This simulated behaviour of disturbance damage is supported by findings in Seidl et al. (2016).

Browsing of tree regeneration by wild ungulates was another disturbance factor in our analysis, and it turned out to be the main reason why tree species diversity was relatively insensitive to management intensity and cutting pattern. That selective browsing of admixed tree species can heavily affect species composition in the long run has been confirmed by many empirical (Klopcic et al. 2009; Schulze et al. 2014; Winter et al. 2015) and modelling studies (Didion et al. 2009; Cailleret et al. 2014). The employed browsing probabilities in all management alternatives were based on local inventory results and reflected current conditions in the study area. Browsing intensity was kept constant throughout the simulations. However, in the long run management changed habitat structure which, at same ungulate density, may change the browsing impact. In particular clear cutting in narrow strips has a high predisposition to game damage (Vospernik and Reimoser 2008). In contrast selective small-scale silvicultural interventions may result in less impact by ungulate game on forest vegetation, though hunting might be more difficult (Reimoser and Gossow 1996). A limitation of the analysis set-up was that browsing by ungulates was not implemented in a full factorial design as were management and climate. Without a dedicated analysis of different browsing intensities one can just speculate about effects of reduced browsing pressure on mid- and long-term forest development and related ES provisioning.

## Effects of management on ecosystem service provisioning

Timber harvests, performed in long return intervals (i.e. low intensity), are the current strategy of BAU management, generating income from timber while maintaining other ES at acceptable levels. However, it is not defined explicitly what those minimum levels are and which ES are prioritized at small scale. Our general findings show that none of the analysed alternatives is best for all ES. Below we discuss how our results could be interpreted in the context of adaptive management, if only one ES is prioritized at local scale while total ES value is accounted for at the regional level.

With high emphasizes on timber production higher management intensity is preferable. Under currently practiced low-intensity management the periodic increment is not fully utilized and timber stocks will increase further. If timber production was the only management goal, STRIP pattern would be the best option.

If carbon storage on site is considered, the NOM scenario obviously would be beneficial; however, less fibre is available to supply bioeconomy, feed the wood products pool and produce green energy (FAO 2016), and the carbon sink in the forest will be saturated once (Nabuurs et al. 2014). Considering carbon storage as a secondary ES to be maximized highly dispersed cutting patterns (SLIT) served best.

Protection efficiency against gravitational hazards benefitted from higher stand density under SAN and NOM, and all low-intensity management scenarios. However, protection against avalanche release, and particularly against rockfall, requires that the structure of the tree population remains balanced and comprises of large living trees but also of sufficient numbers of smaller trees (Dorren et al. 2004; Frehner et al. 2005). NOM and SAN are, relatively seen, the best alternatives for rockfall protection, however, at partly very low performance levels and decreasing resilience in the future. Regarding the cutting pattern in low-intensity management scenarios no significant improvement resulted from highly dispersed SLIT patterns compared to currently used PATCH management.

The ES nature conservation and biodiversity showed diverging results. If bird habitat needs to be provided, closed canopies in dense stands are detrimental. Thus interestingly the NOM alternative did not generate the highest shares of good bird habitat. SAN decreased the pool of large snags and was not beneficial for habitat provisioning either. Therefore, low-intensity management using dispersed cutting patterns (SLIT or PATCH) was the best options for bird habitat provisioning, as they kept the canopy layer open and provided enough large snags and veteran trees. Size diversity of the tree population was clearly positively affected by active management scenarios, where management intensity was much more influential than cutting pattern.

## Climate change and interaction effects on ecosystem services

Direct effects of temperature and precipitation on forest dynamics were clearly less important than effects of management. Indirectly climate change affected forest structures and related ES via the disturbance regime. Under a warmer climate damages from bark beetle disturbances increased substantially and thus provided large amounts of deadwood and canopy gaps. These effects supported bird habitat quality in dependence of management. For instance NOM in severe climate change scenarios provided best habitat quality, while under historic climate and mild climate change canopy closure was too high.

Further complex interrelationships among ES indicators do occur. For instance the interrelationship between available deadwood for bird habitat and bark beetle-induced tree mortality. Woodpeckers may play a significant role in regulating bark beetle populations. Especially for three-toed woodpeckers, strong response of bird numbers during a gradation of bark beetle was observed (Yaeger 1955; Pechacek 1994). If woodpeckers aggregate at a disturbance site, they may kill between 45 and 98% of bark beetles by direct (consumption) or indirect (debarking) effects (Fayt et al. 2005). But research also shows complex interactions between bird population dynamics at landscape scale and local consumer–resource relationships (see Fayt et al. 2005): the predatory impact of woodpeckers will depend mostly on the presence of juvenile woodpeckers across the landscape when beetle larvae reach an adequate size. Important factors are therefore reproductive success of the regional bird population and habitat connectivity. Furthermore under endemic conditions woodpeckers depend on pray other than bark beetle (e.g. longhorn beetle larvae), especially during reproduction season because of higher caloric content (Pechacek and Kristin 2004). As proposed in this study the presence of sufficient large standing deadwood and the presence of veteran trees in general may serve as important food source for woodpecker species during lower beetle population densities. Thus a high bird habitat rating across the landscape is expected to dampen disturbances and result in lower bark beetle-induced tree mortality.

The complexity of functional relationships among ES indicators and how these are affected by management and climate change has been discussed above for bird habitat quality. Several feedback relations exist also for the tree regeneration process. Simulated regeneration progressed slowly under all management alternatives. This is in line with empirical studies in Alpine mountain forests showing that with patch and slit cut approaches successful regeneration may require several decades (Brang 1998; Streit et al. 2009). A warmer climate will generally accelerate regeneration processes and improve the competitiveness of broad-leaved tree species. However, the strong browsing pressure prevented the increase in tree species diversity, regardless of the applied cutting pattern.

## Implications for multifunctional adaptive mountain forest management

The general finding of the presented study was that none of the analysed alternatives is best for all ES and that it requires a more in depth analysis of ES priorities to establish a long-term strategy for forest management. The dilemma in deciding for a management strategy is in the fact that speeding up the increase in species diversity and the patchiness of forest structure by reducing volume stocks and initiating canopy gaps as an adaptive strategy will, at least temporarily, negatively affect protective services. If spatial disentangling of required services is possible, zoning would be the best solution where in each zone a set of specific ES and priorities would be addressed (Nitschke and Innes 2008; Côté et al. 2010). Management in each zone is then focussed on a subset of predefined ES (Messier et al. 2009), and total ES value can be maximized on the regional level. Socially acceptable prioritization of ES demands is a prerequisite for such zoning approaches.

From our results we conclude that PATCH and SLIT regimes appear as compromise if multifunctionality at small scale is the goal. However, at currently practiced management intensity such management approaches would not be suitable to adapt forest composition and structure to a warming climate. Intensifying disturbances from bark beetles may increase dramatically and impede protective services. Results indicate that increased management intensities at the level of analysed HI variants may moderately reduce bird habitat quality and protective services, but will also reduce forest susceptibility for bark beetle disturbances and has the potential to speed up the increase in admixed species shares, which in turn will positively feedback on susceptibility and foster resilience of forests (e.g. Ammer 1996; Bolte et al. 2009; Didion et al. 2009; Schuler et al. 2016). As a prerequisite for the latter browsing intensity on tree regeneration must be reduced (Winter et al. 2015). In a warmer climate substantial sanitation efforts may be required to control large-scale bark beetle outbreaks (e.g. Triebenbacher 2014) and to keep resulting unstocked areas small. If browsing by ungulates is no longer the main limiting factor, regeneration dynamics are expected to benefit from a warmer climate and longer vegetation periods. In addition artificial regeneration can be used to reduce periods of limited ES provisioning on disturbed areas and to increase the share of admixed tree species in a targeted approach.

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EUROPEAN JOURNAL OF FOREST RESEARCH

**Evaluating multifunctionality and adaptive capacity of mountain forest management alternatives under climate change in the Eastern Alps**

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# Electronic Supplementary Material

## 1. SM1:Ecosystem service indicators

In Table SM1-1 all ecosystem service indicators used in this study are presented with details on measurement units, aggregation and preference direction.

**Tab SM1-1** Ecosystem service indicators and the regeneration indicator (REG) as used in this study.

| Acronym | Short description | unit | temporal aggregation | spatial grainsize | Preference direction |
|---|---|---|---|---|---|
| **Timber production** | | | | | |
| THV | Timber harvested | $m^3\ ha^{-1}\ yr^{-1}$ | mean | harvesting unit | more |
| VI | Volume increment | $m^3\ ha^{-1}\ yr^{-1}$ | mean | harvesting unit | more |
| BBD | Bark beetle damage | $m^3\ ha^{-1}\ yr^{-1}$ | mean | harvesting unit | less |
| V | Standing Timber | $m^3\ ha^{-1}$ | mean | harvesting unit | optimum |
| **Carbon sequestration** | | | | | |
| C | Carbon | $t\ ha^{-1}$ | mean | harvesting unit | more |
| **Nature conservation** | | | | | |
| DSP | Species Diversity | [Index] | mean | 1ha cell | more |
| DSI | Size Diversity | [Index] | mean | 1ha cell | more |
| BHQ | Bird Habitat Quality | [good, medium, poor] | mode | 1ha cell | More (area [good]) |
| **Protection against gravitational hazards** | | | | | |
| API | Protection against snow avalanches | [0-1] | minimum | 1ha cell | More |
| LPI | Protection against landslides | [good, medium, poor] | lowest | 1ha cell | More (area [good]) |
| RPI | Protection against rockfall | [0-0.99] | minimum | 1ha cell | More |
| ***Forest regeneration*** | | | | | |
| REG | area with regeneration | [%] | mean | 1ha cell | more |

Below follows a detailed description of the employed ecosystem service indicators (compare Table SM1-1).

**Timber production:**

**V:** Volume of life trees with minimum diameter at breast height (DBH) of 5cm.

**TVH:** Total annual volume of trees harvested with minimum DBH of 5cm.

**VI:** periodic annual volume production of trees with minimum DBH of 5cm.

**BBD:** Annual timber volume killed by bark beetle infestations with minimum DBH of 5cm.

**Carbon sequestration:**

**C:** Total carbon contained in tree biomass for trees with minimum DBH of 5cm (above and belowground), includes carbon contained in live trees, standing deadwood carbon, carbon contained in woody debris and carbon contained in soil organic matter.

**Nature conservation:**

**DSP:** Tree species diversity calculated as relative true diversity index according to Jost (2006), using basal area of trees with DBH >5cm. This index can be interpreted as an "equivalent number of species" as it equals tree species richness when all species in the stand are equally abundant. Otherwise, it is always below tree species richness (see Eq. 1)

$$DSP = \exp(H) \tag{1}$$

$$H = -\sum_{i=1}^{S} p_i \ln(p_i)$$

Where $S$ is the number of tree species and $p_i$ is the relative basal area share of species (i).

**DSI:** Tree size diversity calculated as the mean of the Shannon entropy indices applied to basal area in diameter classes and height classes respectively, as presented in Cordonnier et al (2013) (see Eq. 2). For diameter classes (5cm classes) minimum DBH is 5cm, for height classes (2m classes) minimum tree size is 4m.

$$DSI = \frac{H_{DBH} + H_H}{2} \tag{2}$$

$$H_{DBH} = -\sum_{m=1}^{N_{DBH}} p_m \ln(p_m)$$

$$H_H = -\sum_{n=1}^{N_H} p_n \ln(p_n)$$

Where $N_{DBH}$ is the number of 5cm DBH classes, $N_H$ is the number of 2m height classes, $p_m$ is the relative basal area within a DBH class and $p_n$ is the relative basal area within a height class.

**BHQ:** The bird habitat indicator assesses quality of forest habitat elements for bird species. The used indicator is a modification of the version presented in Cordonnier et al (2013). BHQ is measured on ordinal scale (categories: good, medium, poor) and is composed of standing deadwood (trees with DBH<30cm), large living trees (DBH>50cm) and canopy cover (DBH>5cm) of a forest (see Table SM2). If two of the three sub-indicators are classified as "good", the BHQ is "good", for "medium" two indicators have to be at least classified as "medium". Rating is "poor" if the criteria for "good" or "medium" are not met.

**Tab SM1-2** Sub-indicators for the Bird habitat Quality index (BHQ).

| **Sub-indicator** | **acronym** | **Requirements for BHQ categories** | | |
|---|---|---|---|---|
| | | **good** | **medium** | **poor** |
| Volume of standing dead trees [$m^3$ $ha^{-1}$] | DVW | DWV > 35 | 15 < DWV < 35 | DWV < 15 |
| Number living trees [n $ha^{-1}$] | LLTN | LLTN > 20 | 10 < LLTN < 20 | LLTN < 10 |
| Canopy cover [%] | CC | 60 < CC < 80 | $(80 < CC < 90) \cup (40 < CC < 60)$ | $(CC > 90) \cup (CC < 40)$ |

**Protection against gravitational hazards:**

**API**: The avalanche protection index indicates protection against snow avalanche release. It has been presented in Cordonnier et al. (2013) based on Frehner et al. (2005) and Gauquelin & Courbaud (2006). The index is calculated from mean slope, basal area and average diameter (for trees with DBH >5cm) (Eq. 3). API varies between 0 and 1, where 1 indicates efficient protection.

$$API = \min\left[\frac{G}{(0.2901 * \overline{DBH} + 1.494) \times (0.1333 * s - 3)};1\right] \quad (3)$$

Where $G$ is basal area of trees >5cm [$m^2$ $ha^{-1}$], $\overline{DBH}$ is mean DBH of trees >5cm [cm] and $s$ is slope [°].

**LPI**: The indicator for landslide and erosion protection has been presented in Cordonnier et al. (2013) based on Berger (1997) and Frehner et al. (2005). It builds on crown cover defined by canopy crown area cover for trees with DBH >5cm. LPI is ordinally scaled in three categories:

**Poor:** canopy cover < 30%

**Medium:** 30% ≤ canopy cover < 60%

**Good:** canopy cover ≥ 60%

**RPI:** The indicator for rockfall protection estimates protection against rockfall in transit and deposit zones. Formulae are based on the tool simulation Rockfornet presented in Berger and Dorren (2007). See also http://www.ecorisq.org/rockfor-net-en. RPI varies between 0 and 0.99, where 0.99 means that 99% of passing rocks are stopped by a forest. For this study we calculated indices for different rock sizes. RPI was calculated for rock diameter 0.46m and density 2400kg/m$^3$. Initial fall height was set to 20m. Eq. (4a) is used for cells with basal area >10m$^2$ha$^{-1}$, Eq. (4b) for cells with basal area <10m$^2$ ha$^{-1}$.

$$A = \frac{\left(\Phi_{rock} \times N \times 100 \times \cos(s)\right) \times \left(EvG + (DcD \times 1.7)\right) * 38.7 * \overline{DBH}^{2.31}}{3.352 \times 10^4 \times \left[0.5 \times \rho \times \pi \times \left(\Phi_{rock}/2\right)^3 \times \left[\min\left(\sqrt{\left(2 \times 9.81 \times \left(F_{ih} + \left(100/\cos(s)\right) \times \max(\tan(s) - 0.6; 0.00086)\right)\right)}; 0.64 \times s\right)^2\right] + 0.25 \times \rho \times \pi \left(\Phi_{rock}/2\right)^3 \times F_{ih}\right]} \quad (4a)$$

$$RPH = \max\left(0.01; 1 - A\right)$$

$$RPI = 1 - \max\left(0.01; 1\text{-}A\right)$$

$$B = \frac{\left(\Phi_{rock} \times N \times 100 \times \cos(s)\right) \times \left(EvG + (DcD \times 1.7)\right) * 38.7 * \overline{DBH}^{2.31}}{3.352 \times 10^4 \times \left[0.5 \times \rho \times \pi \times \left(\Phi_{rock}/2\right)^3 \times \left[\min\left(\sqrt{\left(2 \times 9.81 \times \left(F_{ih} + \left(100/\cos(s)\right) \times \max(\tan(s) - 0.6; 0.00086)\right)\right)}; 0.8 \times s\right)^2\right] + 0.25 \times \rho \times \pi \left(\Phi_{rock}/2\right)^3 \times F_{ih}\right]} \quad (4b)$$

$$RPH = \max\left(0.01; 1 - B\right)$$

$$RPI = 1 - \max\left(0.01; 1\text{-}B\right)$$

Where *N* is the number of trees >5cm , *s* is slope [°], *EvG* is basal area share of evergreen species [%], *DcD is* basal area share of deciduous species [%], $\overline{DBH}$ is mean DBH of trees >5cm [cm], $\Phi_{rock}$ is rock diameter [m], $\rho$ is rock density [kg m$^{-3}$] and $F_{ih}$ is initial free fall height of rocks [m]

**REG:** Regeneration indicator. It is calculated as the share of 1ha grid cells with at least 50 saplings per ha in height class 80-130cm.

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## 2. SM2: Model parameters for browsing algorithm

**Tab SM2-1** Model parameters for the PICUS browsing submodule. browsing probability = annual fraction of simulated patches experiencing browsing; mortality fraction = fraction of stems per simulated patch in height class (i) exceeding the allowed number of successive years with height growth below the threshold that die; required height growth = minimum height growth per year; years below threshold = number of successive years below the required height growth threshold;

| Model parameter | spruce | fir | maple | beech |
|---|---|---|---|---|
| **browsing:** | | | | |
| browsing probability | 0.31 | 0.68 | 0.41 | 0.38 |
| **mortality:** | | | | |
| mortality fraction | 0.63 | 0.57 | 0.57 | 0.63 |
| required height growth [cm/yr] | | | | |
| height class 0-10cm | 3 | 3 | 3 | 3 |
| height class 10-30cm | 3 | 3 | 5 | 3 |
| height class 30-80cm | 2 | 1 | 5 | 3 |
| height class 80-130cm | 2 | 1 | 5 | 3 |
| years below threshold [yr] | 5 | 5 | 2 | 3 |

## 3. SM3: Model-induced variability of ES indictors

In general, model-induced variability in indicators increased over time. Focusing on final assessment period P3, we observed some distinct patterns (see Fig. SM3-1): First, stochastic variation of indicators that scale linearly and are aggregated at the level of harvesting units (V, C, VI, and TVH) was very low (management-specific CVs <1.5% under current climate c0) and increased only slightly under the most severe temperature increase of scenarios c4 and c5 (CV = 2-2.5%), with VI as the only exception (highest CVs under c2). The diversity indicators DSP and DSI showed a very similar behaviour. Variation of DSI was particularly low (CVs <0.2%). For these indicators the effect of climate on indicator variability was significant for TVH, VI and V only (Table SM3-2). Second, management-specific coefficients of variation for the indicators BHQ, LPI, API, RPI and REG were in general moderately larger (CV = 2-8%). From all indicators the bark beetle induced damage in timber volume (BBD) clearly showed the highest variability (CV = 9-17% per management alternative). Within this indicator group the effect of climate on indicator variability was significant for the protection indicators LPI and RPI. Interestingly, variation in BBD decreased with increasing climate change severity with variability clearly being smallest under the warmest climate scenario c5. Detailed analysis revealed that with increasing severity of climate change damages by bark beetles occurred more regularly, and as a consequence variation between simulation runs decreased.

Overall, the absolute differences in CV between management alternatives were very small within a specific climate scenario (0.5-4%) with the exception of BBD, where they were as large as 8%. For DSP, DSI, BHQ and RPI, management affected indicator variation significantly (Table SM3-2). Most indicators exhibited highest variability under the NOM and SAN management alternatives, with the exception of RPI, LPI, C, V and BBD, that showed the largest variation under the PATCH-LO management. There was no interacting effect of climate and management on the variability of indicators.

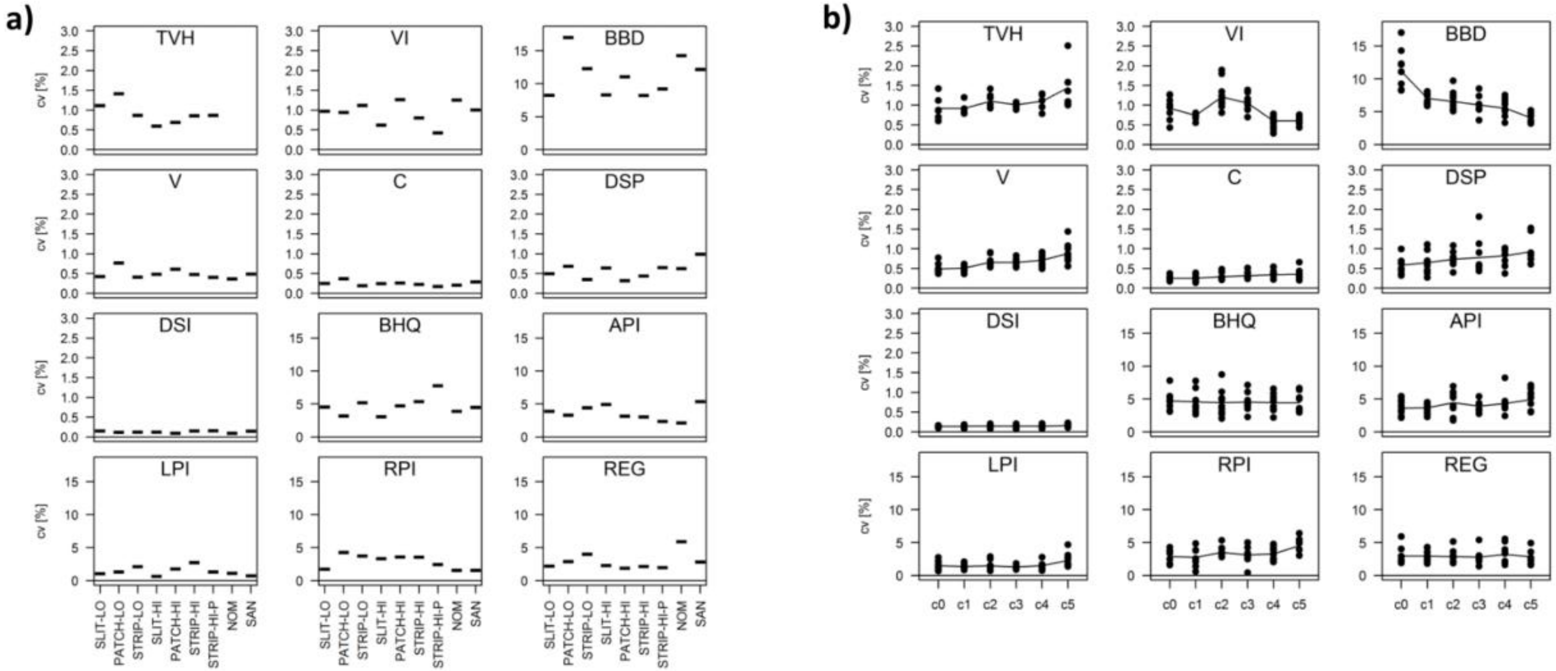


**Fig SM3-1** Panel a) Variability of indicators under historic climate (c0). Coefficient of variation (CV) in period P3 (2067-2100) based on 10 repetitions. Panel b) Variability of indicators under climate change (n per climate scenario: 9 management alternatives x 10 simulation repetitions); TVH = timber harvests, VI = annual volume increment, BBD = damage by bark beetle disturbances, V= timber volume stock, C= carbon stock, DSP= tree species diversity, DSI = tree size diversity, BHQ = bird habitat quality, API = avalanche protection, LPI = landslide protection, RPI= rockfall protection, REG = regenerated area. DSP and DSI are represented by 50th-percentile, API and RPI by 10th-percentile and LPI and BHQ as share of area rated as "good". Black line shows fitted cubic smoothing spline (DF=5).

**Table SM3-2** F-values for ANOVA test for difference of variation (calculated as CV) for model repetitions between management scenarios (DF=8), climate scenarios (DF=5) and interaction management x climate (DF=40). ***= Pr <0.001, **=Pr<0.01, *= Pr<0.05, n.s. = Pr>0.05; # = salvaged timber left on site is not included.

| | TVH# | VI | VBB | V | C | DSP | DSI | BHQ | API | LPI | RPI | REG |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **management** | 0.53 n.s. | 2.33 n.s. | 0.02 n.s. | 1.07 n.s. | 3.67 n.s. | 13.29 *** | 9.41** | 5.01* | 0.14 n.s. | 2.36 n.s. | 9.30** | 3.44 n.s. |
| **climate** | 4.37** | 14.35*** | 17.80*** | 7.73*** | 1.92 n.s. | 2.10 n.s. | 0.65 n.s. | 0.05 n.s. | 1.53 n.s. | 2.61* | 3.91** | 0.24 n.s. |
| **management:climate** | 2.14 n.s. | 1.49 n.s. | 0.56 n.s. | 0.90 n.s. | 0.89 n.s. | 1.46n.s. | 0.36 n.s. | 0.48 n.s. | 1.68n.s. | 0.06 n.s. | 1.00 n.s. | 0.10 n.s. |

## 4. SM4: Longterm effects of management and climate on ES provisioning

**Table SM4-1** F-values for ANOVA test for effects of management (DF=8), climate scenarios (DF=5) and interaction of management x climate (DF=40). P10, P50, P90 = 10th, 50th, and 90th percentile. ***= Pr <0.001, **=Pr<0.01, *= Pr<0.05, n.s.=Pr>0.05; # = salvaged timber not included; empty fields = test not applicable because at least one group showed no variation.

| Factor | Period | TVH# | VI | BBD | V |
|---|---|---|---|---|---|
| management | 1 | 462233.0*** | 4411.1*** | 4.9*** | 10308.2*** |
| climate | 1 | 135.3*** | 17305.6*** | 846.7*** | 657.1*** |
| management:climate | 1 | 8.4*** | 5.4*** | 1.3*** | 4.3*** |
| management | 2 | 198491.7*** | 1977.2*** | 19.6*** | 39844.4*** |
| climate | 2 | 355.5*** | 3186.8*** | 4976.3*** | 2069.2*** |
| management:climate | 2 | 22.5*** | 5.0*** | 5.1*** | 2.9*** |
| management | 3 | 146603.5*** | 4818.2*** | 182.1*** | 41906.4*** |
| climate | 3 | 3512.8*** | 10921.5*** | 5028.9*** | 12010.0*** |
| management:climate | 3 | 144.6*** | 24.3*** | 23.2*** | 32.5*** |

| Factor | Period | DSP P10 | DSP P50 | DSP P90 | DSI P10 | DSI P50 | DSI P90 | BHQ (poor) | BHQ (med) | BHQ (good) |
|---|---|---|---|---|---|---|---|---|---|---|
| management | 1 | | 376.3*** | 161.9*** | 522.4*** | 522.4*** | 169.7*** | 69.4*** | 58.4*** | 152.1*** |
| climate | 1 | | 313.9*** | 133.1*** | 38.7*** | 38.7*** | 130.2*** | 2.0 | 4.8*** | 8.9*** |
| management:climate | 1 | | 3.0*** | 1.7** | 4.2*** | 4.2*** | 7.0*** | 1.4* | 2.8*** | 4.0*** |
| management | 2 | | 5427.4*** | 2933.9*** | 4742.3*** | 4742.3*** | 10203.0*** | 186.4*** | 301.6*** | 666.4*** |
| climate | 2 | | 1814.5*** | 1062.3*** | 262.0*** | 262.0*** | 365.6*** | 29.8*** | 66.1*** | 144.2*** |
| management:climate | 2 | | 17.0*** | 8.1*** | 8.5*** | 8.5*** | 5.4*** | 3.7*** | 4.7*** | 7.1*** |
| management | 3 | | 23765.0*** | 8405.3*** | 3852.7*** | 3852.7*** | 7078.3*** | 336.2*** | 302.5*** | 1002.1*** |
| climate | 3 | | 1973.7*** | 2523.9*** | 559.0*** | 559.0*** | 718.4*** | 28.5*** | 61.1*** | 40.2*** |
| management:climate | 3 | | 120.1*** | 28.8*** | 10.4*** | 10.4*** | 21.9*** | 4.0*** | 7.9*** | 17.8*** |

| Factor | Period | API P10 | API P50 | API P90 | LPI (poor) | LPI (med) | LPI (good) | RPI P10 | RPI P50 | RPI P90 |
|---|---|---|---|---|---|---|---|---|---|---|
| management | 1 | 852.7*** | | | 1264.0*** | 2307.1*** | 5905.0*** | 336.4*** | | |
| climate | 1 | 24.3*** | | | 18.2*** | 6.5*** | 27.2*** | 59.6*** | | |
| management:climate | 1 | 2.8*** | | | 1.6* | 1.6*** | 2.1*** | 3.6*** | | |
| management | 2 | 793.7*** | | | 1283.6*** | 7077.1*** | 10273.9*** | 3205.6*** | | |
| climate | 2 | 62.4*** | | | 36.5*** | 425.9*** | 580.1*** | 158.0*** | | |
| management:climate | 2 | 2.6*** | | | 3.3*** | 12.1*** | 14.5*** | 2.6*** | | |
| management | 3 | 84.0*** | | | 264.9*** | 2874.5*** | 4074.5*** | 4661.4*** | | |
| climate | 3 | 124.3*** | | | 55.0*** | 910.4*** | 1275.8*** | 977.7*** | | |
| management:climate | 3 | 3.2*** | | | 3.5*** | 15.0*** | 19.3*** | 18.4*** | | |

**Tab SM4-2** Ecosystem service indicators for management alternatives under historic climate (c0) in assessment period P3 (2067-2100), P10, P50, P90 = 10th, 50th, and 90th percentile, mean of 10 simulation runs, Tukey tests for significant differences between management scenarios under historic climate conditions (c0), letters denote significant (alpha = 0.05) differences between management scenarios, NA = excluded from test

| Indicator | Unit | | SLIT-LO | PATCH-LO | STRIP-LO | SLIT-HI | PATCH-HI | STRIP-HI | STRIP-HI-P | NOM | SAN |
|---|---|---|---|---|---|---|---|---|---|---|---|
| THV | $m^3 ha^{-1} yr^{-1}$ | mean | 2.6 d | 2.6 d | 2.3 e | 2.8 c | 3.3 b | 3.9 a | 3.9 a | 0.0 NA | 0.0 NA |
| VI | $m^3 ha^{-1} yr^{-1}$ | mean | 6.5 d | 6.6 d | 6.9 c | 7.0 bc | 7.0 b | 7.3 a | 7.3 a | 5.9 e | 5.9 e |
| BBD | $m^3 ha^{-1} yr^{-1}$ | mean | 0.5 ab | 0.5 ab | 0.5 ab | 0.5 a | 0.5 a | 0.5 a | 0.4 b | 0.5 ab | 0.4 b |
| V | $m^3 ha^{-1}$ | mean | 438.2 c | 438.1 c | 444.9 b | 375.0 d | 352.1 f | 352.4 f | 365.6 e | 553.0 a | 554.3 a |
| C | $t ha^{-1}$ | mean | 226.9 d | 227.1 d | 228.1 c | 209.8 e | 202.8 g | 202.5 g | 207.6 f | 264.0 a | 260.0 b |
| *DSP* | - | P10 | 1.00 n.s. | 1.00 n.s. | 1.00 n.s. | 1.00 n.s. | 1.00 n.s. | 1.00 n.s. | 1.09 n.s. | 1.00 n.s. | 1.00 n.s. |
| | | P50 | 1.06 d | 1.06 d | 1.04 f | 1.09 b | 1.07 c | 1.05 e | 1.54 a | 1.05 de | 1.10 b |
| | | P90 | 1.67 c | 1.67 c | 1.59 d | 1.72 b | 1.68 c | 1.54 e | 2.39 a | 1.67 c | 1.70 bc |
| *DSI* | - | P10 | 2.66 d | 2.66 d | 2.67 c | 2.70 a | 2.71 a | 2.69 b | 2.71 a | 2.57 e | 2.57 e |
| | | P50 | 2.71 e | 2.71 de | 2.71 d | 2.75 a | 2.75 a | 2.74 c | 2.74 b | 2.63 f | 2.63 f |
| | | P90 | 2.74 d | 2.74 d | 2.75 c | 2.77 b | 2.77 ab | 2.77 ab | 2.78 a | 2.66 e | 2.67 e |
| *BHQ* | % | poor | 10.0 de | 9.2 e | 10.9 cd | 11.8 bc | 12.8 b | 15.3 a | 15.6 a | 9.1 e | 11.5 bc |
| | | medium | 40.5 NA | 40.6 NA | 46.5 NA | 43.6 NA | 42.8 NA | 49.0 NA | 51.9 NA | 45.7 NA | 53.6 NA |
| | | good | 49.6 a | 50.2 a | 42.6 b | 44.6 b | 44.4 b | 35.7 c | 32.6 d | 45.1 b | 35.0 cd |
| API | - | P10 | 0.86 bc | 0.86 bc | 0.82 c | 0.85 bc | 0.76 d | 0.75 d | 0.82 c | 0.91 a | 0.87 ab |
| | | P50 | 1.00 n.s. | 1.00 n.s. | 1.00 n.s. | 1.00 n.s. | 1.00 n.s. | 1.00 n.s. | 1.00 n.s. | 1.00 n.s. | 1.00 n.s. |
| | | P90 | 1.00 n.s. | 1.00 n.s. | 1.00 n.s. | 1.00 n.s. | 1.00 n.s. | 1.00 n.s. | 1.00 n.s. | 1.00 n.s. | 1.00 n.s. |
| LPI | % | poor | 5.9 c | 5.5 c | 5.5 c | 5.7 c | 6.5 b | 7.1 a | 5.5 c | 4.4 d | 4.1 d |
| | | medium | 24.4 NA | 24.4 NA | 29.8 NA | 29.0 NA | 31.0 NA | 42.5 NA | 35.2 NA | 18.6 NA | 17.2 NA |
| | | good | 69.7 c | 70.1 c | 64.7 d | 65.3 d | 62.5 e | 50.4 g | 59.4 f | 77.0 b | 78.7 a |
| RPI | - | P10 | 0.65 b | 0.66 b | 0.56 c | 0.54 c | 0.50 d | 0.43 e | 0.50 d | 0.98 a | 0.98 a |
| | | P50 | 0.99 n.s. | 0.99 n.s. | 0.99 n.s. | 0.99 n.s. | 0.99 n.s. | 0.99 n.s. | 0.99 n.s. | 0.99 n.s. | 0.99 n.s. |
| | | P90 | 0.99 n.s. | 0.99 n.s. | 0.99 n.s. | 0.99 n.s. | 0.99 n.s. | 0.99 n.s. | 0.99 n.s. | 0.99 n.s. | 0.99 n.s. |

**Tab SM4-3** Ecosystem service indicators for chosen management types under climate change (c1) in assessment period P3 (2067-2100), P10, P50, P90 = 10th, 50th, and 90th percentile, mean of 10 simulation runs, Tukey tests for significant differences between management scenarios under climate change conditions (c1), letters denote significant (alpha = 0.05) differences between management scenarios, NA = excluded from test

| Indicator | Unit | | SLIT-LO | PATCH-LO | STRIP-LO | SLIT-HI | PATCH-HI | STRIP-HI | STRIP-HI-P | NOM | SAN |
|---|---|---|---|---|---|---|---|---|---|---|---|
| THV | $m^3\ ha^{-1}\ yr^{-1}$ | mean | 2.7 e | 2.6 f | 2.3 g | 2.8 d | 3.3 c | 3.9 b | 3.9 a | 0.0 h | 0.0 h |
| VI | $m^3\ ha^{-1}\ yr^{-1}$ | mean | 7.4 d | 7.4 d | 7.9 c | 7.9 bc | 8.0 b | 8.4 a | 8.3 a | 6.7 e | 6.7 e |
| BBD | $m^3\ ha^{-1}\ yr^{-1}$ | mean | 1.1 cde | 1.2 bcd | 1.2 abc | 1.1 de | 1.1 cde | 1.1 de | 1.0 e | 1.3 ab | 1.3 a |
| V | $m^3\ ha^{-1}$ | mean | 444.5 c | 442.8 c | 450.3 b | 380.4 d | 358.5 f | 360.6 f | 376.5 e | 558.3 a | 558.5 a |
| C | $t\ ha^{-1}$ | mean | 228.0 c | 227.9 c | 228.7 c | 211.2 d | 204.6 e | 204.0 e | 210.4 d | 264.4 a | 260.7 b |
| *DSP* | - | P10 | 1.00 de | 1.00 d | 1.00 e | 1.02 b | 1.02 b | 1.00 de | 1.14 a | 1.00 de | 1.01 c |
| | | P50 | 1.11 d | 1.11 d | 1.07 f | 1.17 c | 1.16 c | 1.09 e | 1.69 a | 1.08 e | 1.18 b |
| | | P90 | 1.84 c | 1.84 cd | 1.69 e | 1.94 b | 1.92 b | 1.67 e | 2.72 a | 1.80 d | 1.91 b |
| *DSI* | - | P10 | 2.64 c | 2.64 c | 2.64 c | 2.68 a | 2.69 a | 2.67 b | 2.69 a | 2.55 e | 2.57 d |
| | | P50 | 2.69 c | 2.69 c | 2.69 c | 2.73 a | 2.73 a | 2.71 b | 2.73 a | 2.61 e | 2.63 d |
| | | P90 | 2.72 e | 2.72 de | 2.73 d | 2.76 b | 2.76 ab | 2.75 c | 2.76 a | 2.65 g | 2.66 f |
| *BHQ* | % | poor | 9.09 d | 8.82 d | 8.95 d | 11.00 c | 11.45 c | 14.59 b | 16.82 a | 8.55 d | 13.32 b |
| | | medium | 40.18 f | 39.00 f | 45.59 cd | 43.41 de | 45.77 cd | 46.64 c | 50.77 b | 41.91 ef | 53.86 a |
| | | good | 50.73 a | 52.18 a | 45.45 b | 45.59 b | 42.77 b | 38.77 c | 32.41 d | 49.55 a | 32.82 d |
| API | - | P10 | 0.84 b | 0.83 bc | 0.80 cd | 0.83 bcd | 0.74 f | 0.75 ef | 0.79 de | 0.88 a | 0.83 bc |
| | | P50 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 |
| | | P90 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 |
| LPI | % | poor | 5.64 cd | 5.36 cd | 5.41 cd | 5.73 bc | 6.23 ab | 6.50 a | 5.09 d | 4.32 e | 4.18 e |
| | | medium | 22.77 NA | 23.14 NA | 29.50 NA | 26.68 NA | 29.68 NA | 41.77 NA | 32.82 NA | 18.27 NA | 16.91 NA |
| | | good | 71.59 c | 71.50 c | 65.09 e | 67.59 d | 64.09 e | 51.73 g | 62.09 f | 77.41 b | 78.91 a |
| RPI | - | P10 | 0.68 b | 0.69 b | 0.57 c | 0.55 cd | 0.51 e | 0.45 f | 0.54 d | 0.98 a | 0.99 a |
| | | P50 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 |
| | | P90 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 |

**Tab SM4-4** Ecosystem service indicators for management types under climate change (c2) in assessment period P3 (2067-2100), P10, P50, P90 = 10th, 50th, and 90th percentile, mean of 10 simulation runs, Tukey tests for significant differences between management scenarios under climate change conditions (c2), letters denote significant (alpha = 0.05) differences between management scenarios, NA = excluded from test

| Indicator | Unit | | SLIT-LO | PATCH-LO | STRIP-LO | SLIT-HI | PATCH-HI | STRIP-HI | STRIP-HI-P | NOM | SAN |
|---|---|---|---|---|---|---|---|---|---|---|---|
| THV | $m^3\ ha^{-1}\ yr^{-1}$ | mean | 2.5 e | 2.4 e | 2.1 f | 2.5 d | 3.0 c | 3.5 b | 3.6 a | 0.0 NA | 0.0 NA |
| VI | $m^3\ ha^{-1}\ yr^{-1}$ | mean | 6.5 c | 6.6 c | 6.9 b | 6.9 b | 7.0 b | 7.3 a | 7.3 a | 5.9 d | 5.9 d |
| BBD | $m^3\ ha^{-1}\ yr^{-1}$ | mean | 1.6 ab | 1.5 bc | 1.6 ab | 1.4 cd | 1.4 cd | 1.4 cd | 1.3 d | 1.7 a | 1.7 a |
| V | $m^3\ ha^{-1}$ | mean | 403.8 bc | 400.8 c | 406.6 b | 344.5 d | 321.8 e | 324.0 e | 342.1 d | 512.7 a | 508.8 a |
| C | $t\ ha^{-1}$ | mean | 218.9 c | 218.4 c | 219.0 c | 203.1 d | 196.2 e | 196.0 e | 203.0 d | 254.0 a | 251.8 b |
| *DSP* | - | P10 | 1 d | 1 d | 1 d | 1.01 c | 1.01 c | 1 d | 1.15 a | 1 d | 1.02 b |
| | | P50 | 1.11 de | 1.11 d | 1.07 g | 1.16 c | 1.16 c | 1.09 fg | 1.73 a | 1.1 ef | 1.25 b |
| | | P90 | 1.9 d | 1.91 d | 1.76 f | 1.96 c | 1.95 c | 1.73 f | 2.83 a | 1.86 e | 2.05 b |
| *DSI* | - | P10 | 2.66 bc | 2.67 b | 2.66 c | 2.7 a | 2.7 a | 2.7 a | 2.71 a | 2.58 e | 2.6 d |
| | | P50 | 2.71 d | 2.72 c | 2.71 d | 2.74 a | 2.74 a | 2.72 b | 2.74 a | 2.64 f | 2.66 e |
| | | P90 | 2.74 c | 2.74 c | 2.74 c | 2.77 ab | 2.77 a | 2.76 b | 2.77 a | 2.67 e | 2.69 d |
| *BHQ* | % | poor | 10.0 cd | 8.5 d | 10.6 c | 13.5 b | 14.2 b | 16.2 a | 17.6 a | 8.7 d | 13.0 b |
| | | medium | 39.6 NA | 38.6 NA | 42.8 NA | 42.1 NA | 44.4 NA | 49.1 NA | 50.8 NA | 39.0 NA | 53.1 NA |
| | | good | 50.5 a | 52.9 a | 46.6 b | 44.5 b | 41.4 c | 34.7 d | 31.6 e | 52.4 a | 34.0 de |
| API | - | P10 | 0.81 b | 0.81 b | 0.79 bc | 0.79 b | 0.73 c | 0.77 bc | 0.76 bc | 0.87 a | 0.80 b |
| | | P50 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 |
| | | P90 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 |
| LPI | % | poor | 6.1 | 5.4 | 5.6 | 5.9 | 6.5 | 6.9 | 5.0 | 4.3 | 4.2 |
| | | medium | 25.8 NA | 26.3 NA | 31.2 NA | 30.1 NA | 32.1 NA | 44.3 NA | 35.3 NA | 19.9 NA | 18.4 NA |
| | | good | 68.1 c | 68.4 c | 63.3 d | 64.1 d | 61.4 e | 48.9 g | 59.8 f | 75.8 b | 77.4 a |
| RPI | - | P10 | 0.58 c | 0.6 c | 0.52 d | 0.5 d | 0.45 e | 0.41 f | 0.5 d | 0.87 b | 0.92 a |
| | | P50 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 |
| | | P90 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 |

**Tab SM4-5** Ecosystem service indicators for management types under climate change (c3) in assessment period P3 (2067-2100), P10, P50, P90 = 10th, 50th, and 90th percentile, mean results of 10 simulation runs, Tukey tests for significant differences between management scenarios under climate change conditions (c3), letters denote significant (alpha = 0.05) differences between management scenarios, NA = excluded from test

| Indicator | Unit | | SLIT-LO | PATCH-LO | STRIP-LO | SLIT-HI | PATCH-HI | STRIP-HI | STRIP-HI-P | NOM | SAN |
|---|---|---|---|---|---|---|---|---|---|---|---|
| THV | $m^3 ha^{-1} yr^{-1}$ | mean | 2.6 d | 2.6 d | 2.3 e | 2.7 c | 3.3 b | 3.8 a | 3.8 a | 0.0 NA | 0.0 NA |
| VI | $m^3 ha^{-1} yr^{-1}$ | mean | 7.5 d | 7.5 d | 8.1 b | 7.9 c | 8.1 b | 8.6 a | 8.5 a | 6.8 e | 6.8 e |
| BBD | $m^3 ha^{-1} yr^{-1}$ | mean | 1.5 a | 1.6 a | 1.6 a | 1.4 b | 1.4 bc | 1.3 bc | 1.2 c | 1.6 a | 1.6 a |
| V | $m^3 ha^{-1}$ | mean | 432.8 c | 431.8 c | 443.0 b | 373.1 d | 350.3 f | 357.5 e | 372.5 d | 549.3 a | 550.4 a |
| C | $t ha^{-1}$ | mean | 225.2 d | 225.3 d | 226.5 c | 209.5 e | 202.9 f | 203.1 f | 209.5 e | 261.9 a | 259.2 b |
| *DSP* | - | P10 | 1.01 c | 1.01 cd | 1 e | 1.03 b | 1.03 b | 1.01 cd | 1.18 a | 1.01 de | 1.03 b |
| | | P50 | 1.14 d | 1.15 d | 1.09 f | 1.21 c | 1.21 c | 1.1 e | 1.78 a | 1.11 e | 1.25 b |
| | | P90 | 1.92 c | 1.92 c | 1.75 e | 2.03 b | 2.03 b | 1.75 e | 2.86 a | 1.85 d | 2.03 b |
| *DSI* | - | P10 | 2.63 cd | 2.64 c | 2.63 d | 2.69 a | 2.68 a | 2.67 b | 2.69 a | 2.55 f | 2.57 e |
| | | P50 | 2.69 c | 2.69 c | 2.68 d | 2.73 a | 2.73 a | 2.7 b | 2.73 a | 2.61 f | 2.63 e |
| | | P90 | 2.72 c | 2.72 c | 2.72 c | 2.75 a | 2.76 a | 2.74 b | 2.76 a | 2.64 e | 2.66 d |
| *BHQ* | % | poor | 9.3 e | 8.4 e | 9.6 e | 12.0 d | 12.6 cd | 15.1 ab | 16.6 a | 8.2 e | 13.9 bc |
| | | medium | 40.4 NA | 38.1 NA | 44.3 NA | 42.4 NA | 45.3 NA | 46.8 NA | 50.4 NA | 41.7 NA | 54.0 NA |
| | | good | 50.3 b | 53.6 a | 46.1 c | 45.6 c | 42.1 d | 38.1 e | 33.1 f | 50. b | 32.1 f |
| API | - | P10 | 0.81 c | 0.81 c | 0.86 ab | 0.78 cd | 0.72 e | 0.76 de | 0.79 cd | 0.90 a | 0.82 bc |
| | | P50 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 |
| | | P90 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 |
| LPI | % | poor | 5.6 | 5.2 | 5.2 | 5.5 | 4.9 | 6.3 | 4.9 | 4.2 | 4.1 |
| | | medium | 22.9 NA | 23.5 NA | 28.5 NA | 26.7 NA | 29.5 NA | 40.6 NA | 30.7 NA | 19.1 NA | 17.5 NA |
| | | good | 71.5 c | 71.3 c | 66.3 e | 67.9 d | 64.7 f | 53.1 g | 64.4 f | 76.8 b | 78.4 a |
| RPI | - | P10 | 0.66 c | 0.68 c | 0.59 d | 0.56 de | 0.52 f | 0.47 g | 0.55 ef | 0.95 b | 0.99 a |
| | | P50 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 |
| | | P90 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 |

**Tab SM4-6** Ecosystem service indicators for management types under climate change (c4) in assessment period P3 (2067-2100), P10, P50, P90 = 10th, 50th, and 90th percentile, mean results of 10 simulation runs, Tukey tests for significant differences between management scenarios under climate change conditions (c4), letters denote significant (alpha = 0.05) differences between management scenarios, NA = excluded from test

| Indicator | Unit | | SLIT-LO | PATCH-LO | STRIP-LO | SLIT-HI | PATCH-HI | STRIP-HI | STRIP-HI-P | NOM | SAN |
|---|---|---|---|---|---|---|---|---|---|---|---|
| THV | $m^3\ ha^{-1}\ yr^{-1}$ | mean | 2.5 e | 2.5 e | 2.2 f | 2.6 d | 3.1 c | 3.6 b | 3.7 a | 0.0 NA | 0.0 NA |
| VI | $m^3\ ha^{-1}\ yr^{-1}$ | mean | 7.4 e | 7.5 d | 7.9 bc | 7.9 c | 8.0 b | 8.4 a | 8.4 a | 6.7 f | 6.7 f |
| BBD | $m^3\ ha^{-1}\ yr^{-1}$ | mean | 1.8 bc | 2.0 ab | 1.9 b | 1.7 cd | 1.7 d | 1.7 d | 1.4 e | 2.1 a | 2.0 ab |
| V | $m^3\ ha^{-1}$ | mean | 419.3 d | 416.8 d | 427.6 c | 360.4 f | 338.7 h | 344.5 g | 365.6 e | 525.0 b | 530.9 a |
| C | $t\ ha^{-1}$ | mean | 222.2 c | 222.1 c | 223.5 b | 206.9 e | 200.5 f | 200.6 f | 208.4 d | 256.3 a | 256.4 a |
| *DSP* | - | P10 | 1.01 d | 1.01 d | 1 f | 1.03 c | 1.03 c | 1.01 de | 1.18 a | 1.01 ef | 1.05 b |
| | | P50 | 1.15 d | 1.16 d | 1.09 f | 1.21 c | 1.21 c | 1.11 ef | 1.82 a | 1.13 e | 1.32 b |
| | | P90 | 1.97 de | 2 d | 1.81 f | 2.08 c | 2.06 c | 1.79 f | 2.93 a | 1.94 e | 2.15 b |
| *DSI* | - | P10 | 2.65 b | 2.66 b | 2.65 b | 2.69 a | 2.7 a | 2.69 a | 2.7 a | 2.57 d | 2.59 c |
| | | P50 | 2.7 c | 2.7 c | 2.69 d | 2.74 a | 2.74 a | 2.72 b | 2.73 a | 2.62 f | 2.65 e |
| | | P90 | 2.73 d | 2.73 d | 2.73 d | 2.76 bc | 2.77 a | 2.75 c | 2.76 b | 2.65 f | 2.68 e |
| *BHQ* | % | poor | 9.2 cd | 8.7 cd | 9.8 c | 13.1 b | 12.8 b | 16.4 a | 16.8 a | 7.2 d | 14.1 b |
| | | medium | 39.9 NA | 36.7 NA | 41.8 NA | 41.8 NA | 45.1 NA | 45.6 NA | 51.3 NA | 37.4 NA | 55.0 NA |
| | | good | 50.9 b | 54.9 a | 48.4 b | 45.1 c | 42.1 d | 38.1 e | 31.9 f | 55.4 a | 30.9 f |
| API | - | P10 | 0.77 bc | 0.80 b | 0.78 b | 0.77 bc | 0.72 c | 0.76 bc | 0.77 bc | 0.85 a | 0.81 ab |
| | | P50 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 |
| | | P90 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 |
| LPI | % | poor | 5.9 | 5.5 | 5.3 | 5.9 | 6.1 | 6.1 | 4.7 | 4.3 | 4.2 |
| | | medium | 24.1 e | 23.8 e | 30.3 c | 29.0 d | 31.0 bc | 43.0 a | 32.0 b | 20.0 f | 18.0 g |
| | | good | 70.5 c | 70.3 c | 64.4 de | 65.4 d | 62.4 f | 51.1 g | 63.0 ef | 75.9 b | 77.9 a |
| RPI | - | P10 | 0.63 c | 0.65 c | 0.56 d | 0.54 d | 0.5 e | 0.45 f | 0.55 d | 0.88 b | 0.97 a |
| | | P50 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 |
| | | P90 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 |

**Tab SM4-7** Ecosystem service indicators for management types under climate change (c5) in assessment period P3 (2067-2100), P10, P50, P90 = 10th, 50th, and 90th percentile, mean results of 10 simulation runs, Tukey tests for significant differences between management scenarios under climate change conditions (c5), letters denote significant (alpha = 0.05) differences between management scenarios, NA = excluded from test

| Indicator | Unit | | SLIT-LO | PATCH-LO | STRIP-LO | SLIT-HI | PATCH-HI | STRIP-HI | STRIP-HI-P | NOM | SAN |
|---|---|---|---|---|---|---|---|---|---|---|---|
| THV | $m^3\ ha^{-1}\ yr^{-1}$ | mean | 2.2 e | 2.1 e | 1.8 f | 2.2 d | 2.7 c | 3.1 b | 3.2 a | 0.0 NA | 0.0 NA |
| VI | $m^3\ ha^{-1}\ yr^{-1}$ | mean | 5.8 f | 5.9 e | 6.3 c | 6.2 d | 6.2 c | 6.5 b | 6.7 a | 5.4 h | 5.5 g |
| BBD | $m^3\ ha^{-1}\ yr^{-1}$ | mean | 2.7 b | 2.7 b | 2.7 b | 2.4 c | 2.2 d | 2.2 d | 1.9 e | 3.1 a | 3.1 a |
| V | $m^3\ ha^{-1}$ | mean | 354.7 bc | 351.6 c | 357.1 b | 301.4 e | 280.3 f | 283.5 f | 308.2 d | 455.9 a | 457.8 a |
| C | $t\ ha^{-1}$ | mean | 207.3 c | 206.8 c | 207.4 c | 192.9 e | 186.6 f | 186.8 f | 195.3 d | 240.2 b | 242.7 a |
| *DSP* | - | P10 | 1.01 d | 1.01 d | 1.00 d | 1.02 c | 1.03 c | 1.01 d | 1.19 a | 1.00 d | 1.06 b |
| | | P50 | 1.14 de | 1.15 d | 1.09 f | 1.19 c | 1.19 c | 1.10 f | 1.86 a | 1.12 e | 1.37 b |
| | | P90 | 2.03 ef | 2.04 de | 1.90 g | 2.08 cd | 2.08 c | 1.91 g | 3.04 a | 1.98 f | 2.24 b |
| *DSI* | - | P10 | 2.67 d | 2.68 c | 2.68 cd | 2.72 a | 2.72 a | 2.71 b | 2.71 ab | 2.59 f | 2.63 e |
| | | P50 | 2.72 d | 2.73 d | 2.72 d | 2.75 a | 2.75 ab | 2.74 c | 2.75 b | 2.64 f | 2.68 e |
| | | P90 | 2.75 cd | 2.75 c | 2.75 d | 2.77 ab | 2.78 a | 2.77 b | 2.77 b | 2.67 f | 2.71 e |
| *BHQ* | % | poor | 10.5 d | 10.0 de | 11.5 d | 14.5 c | 16.0 bc | 18.7 a | 17.3 ab | 7.8 e | 14.8 c |
| | | medium | 34.6 NA | 34.0 NA | 38.2 NA | 41.7 NA | 44.8 NA | 46.5 NA | 49.7 NA | 28.4 NA | 46.6 NA |
| | | good | 55.0 b | 56.1 b | 50.3 c | 43.8 d | 39.2 e | 34.8 f | 33.1 f | 63.8 a | 38.6 e |
| API | - | P10 | 0.70 bc | 0.73 b | 0.69 bc | 0.74 b | 0.70 bc | 0.67 c | 0.71 bc | 0.79 a | 0.71 bc |
| | | P50 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 |
| | | P90 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 |
| LPI | % | poor | 6.5 | 6.1 | 6.3 | 6.1 | 6.8 | 8.3 | 5.2 | 5.1 | 4.6 |
| | | medium | 34.7 d | 36.9 cd | 37.8 c | 38.0 c | 42.0 b | 49.7 a | 39.0 c | 25.3 e | 21.0 f |
| | | good | 58.8 c | 57.0 cd | 55.9 d | 55.9 d | 51.2 e | 42.0 f | 55.8 d | 69.6 b | 74.5 a |
| RPI | - | P10 | 0.48 c | 0.49 c | 0.43 d | 0.41 d | 0.36 e | 0.34 e | 0.44 d | 0.67 b | 0.76 a |
| | | P50 | 0.99 | 0.99 | 0.99 | 0.96 | 0.93 | 0.92 | 0.99 | 0.99 | 0.99 |
| | | P90 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 | 0.99 |

**Tab SM4-8** Tukey tests for significant differences between climate change scenarios in period P3 for each management alternative; letters denote significant differences. n.s. = results not significant. For empty fields ANVOA/Tukey tests where not applicable because at least one group showed no variation.

| Management | Climate | TVH | VI | BBD | V | C | DSP | | | DSI | | | BHQ | | API | | | LPI | | RPI | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | mean | mean | mean | mean | mean | P10 | P50 | P90 | P10 | P50 | P90 | poor | good | P10 | P50 | P90 | poor | good | P10 | P50 | P90 |
| SLIT-LO | c0 | b | c | e | b | b | | d | e | bc | b | b | n.s. | b | a | | | bc | c | bc | | |
| | c1 | a | ab | d | a | a | | c | d | d | d | d | n.s. | b | ab | | | c | a | a | | |
| | c2 | d | c | c | e | e | | c | c | ab | b | b | n.s. | b | bc | | | ab | d | d | | |
| | c3 | b | a | c | c | c | | b | c | d | d | d | n.s. | b | bc | | | c | ab | ab | | |
| | c4 | c | b | b | d | d | | a | b | c | c | c | n.s. | b | c | | | bc | bc | c | | |
| | c5 | e | d | a | f | f | | b | a | a | a | a | n.s. | a | d | | | a | e | e | | |
| PATCH-LO | c0 | a | c | e | b | a | | d | e | bc | b | b | ab | d | a | | | b | b | ab | | |
| | c1 | a | b | d | a | a | | c | d | d | d | d | ab | cd | ab | | | b | a | a | | |
| | c2 | d | c | c | e | d | | c | c | b | b | b | b | bc | b | | | b | c | c | | |
| | c3 | b | a | c | c | b | | ab | c | d | d | e | b | bc | b | | | b | a | ab | | |
| | c4 | c | ab | b | d | c | | a | b | c | c | c | ab | ab | b | | | b | ab | b | | |
| | c5 | e | d | a | f | e | | b | a | a | a | a | a | a | c | | | a | d | d | | |
| STRIP-LO | c0 | a | c | e | b | a | | c | e | a | b | a | ab | c | ab | | | b | abc | b | | |
| | c1 | a | b | d | a | a | | b | d | d | e | c | c | bc | b | | | b | ab | ab | | |
| | c2 | c | c | c | d | d | | b | c | b | c | b | ab | b | b | | | b | c | c | | |
| | c3 | a | a | c | b | b | | a | c | e | f | d | bc | b | a | | | b | a | a | | |
| | c4 | b | b | b | c | c | | a | b | c | d | c | bc | ab | b | | | b | bc | b | | |
| | c5 | d | d | a | e | e | | a | a | a | a | a | a | a | c | | | a | d | d | | |
| SLIT-HI | c0 | a | b | e | b | b | | d | d | b | a | a | cd | n.s. | a | | | ab | b | a | | |
| | c1 | a | a | d | a | a | | c | c | d | d | c | d | n.s. | ab | | | ab | a | a | | |
| | c2 | d | b | c | d | d | | c | c | b | b | b | ab | n.s. | bc | | | ab | c | b | | |
| | c3 | b | a | c | b | b | | a | b | d | d | d | bcd | n.s. | cd | | | b | a | a | | |
| | c4 | c | a | b | c | c | | a | a | c | c | c | abc | n.s. | cd | | | a | b | a | | |
| | c5 | e | c | a | e | e | | b | a | a | a | a | a | n.s. | d | | | a | d | c | | |
| PATCH-HI | c0 | ab | c | e | b | b | | d | d | b | a | b | bc | a | a | | | ab | b | b | | |
| | c1 | a | b | d | a | a | | c | c | d | c | d | c | ab | ab | | | bc | a | ab | | |
| | c2 | d | c | c | d | d | | c | c | bc | b | c | ab | bc | ab | | | ab | b | c | | |
| | c3 | b | a | c | b | b | | a | b | d | c | e | bc | ab | ab | | | c | a | a | | |
| | c4 | c | b | b | c | c | | a | ab | c | b | cd | bc | ab | ab | | | bc | b | ab | | |
| | c5 | e | d | a | e | e | | b | a | a | a | a | a | c | b | | | a | c | d | | |
| STRIP-HI | c0 | a | c | e | c | b | | c | e | b | a | a | b | b | a | | | b | bc | bc | | |
| | c1 | a | b | d | a | a | | b | d | c | c | c | b | a | a | | | bc | ab | ab | | |
| | c2 | d | c | c | e | d | | b | c | b | b | b | b | b | a | | | bc | c | c | | |
| | c3 | b | a | c | b | b | | a | c | c | d | d | b | a | a | | | bc | a | a | | |
| | c4 | c | b | b | d | c | | a | b | b | b | c | b | a | a | | | c | b | ab | | |
| | c5 | e | d | a | f | e | | a | a | a | a | a | a | b | b | | | a | d | d | | |
| STRIP-HI-P | c0 | a | c | e | c | d | d | f | e | b | a | a | n.s. | n.s. | a | | | a | c | b | | |
| | c1 | a | b | d | a | a | c | e | d | d | cd | cd | n.s. | n.s. | b | | | abc | b | a | | |
| | c2 | d | c | c | d | e | c | d | c | b | b | bc | n.s. | n.s. | b | | | bc | c | b | | |
| | c3 | b | a | c | b | b | b | c | c | d | d | e | n.s. | n.s. | ab | | | bc | a | a | | |
| | c4 | c | a | b | c | c | ab | b | b | c | c | de | n.s. | n.s. | b | | | c | ab | a | | |
| | c5 | e | d | a | e | f | a | a | a | a | a | b | n.s. | n.s. | c | | | ab | d | c | | |
| NOM | c0 | NA | c | e | ab | a | | e | e | b | b | b | a | d | a | | | b | ab | a | | |
| | c1 | NA | b | d | a | a | | d | d | c | c | d | ab | c | ab | | | b | a | a | | |
| | c2 | NA | c | c | d | d | | c | c | a | a | b | a | c | ab | | | b | c | b | | |
| | c3 | NA | a | c | b | b | | b | c | c | c | e | ab | c | a | | | b | abc | a | | |
| | c4 | NA | ab | b | c | c | | a | b | b | b | c | b | b | b | | | b | bc | b | | |
| | c5 | NA | d | a | e | e | | a | a | a | a | a | ab | a | c | | | a | d | c | | |
| SAN | c0 | NA | c | e | ab | ab | | e | e | c | d | d | b | b | a | | | b | a | a | | |
| | c1 | NA | b | d | a | a | | d | d | c | d | e | a | bcd | ab | | | b | a | a | | |
| | c2 | NA | c | c | d | d | | c | c | b | b | b | ab | bc | b | | | b | b | b | | |
| | c3 | NA | a | c | b | b | | c | c | c | d | de | a | cd | b | | | b | ab | a | | |
| | c4 | NA | b | b | c | c | | b | b | b | c | c | a | d | b | | | b | ab | a | | |
| | c5 | NA | d | a | e | e | | a | a | a | a | a | a | a | c | | | a | c | c | | |