

How to quantify forest management intensity in Central European forests

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Abstract Existing approaches for the assessment of forest management intensity lack a widely accepted, purely quantitative measure for ranking a set of forest stands along a gradient of management intensity. We have developed a silvicultural management intensity indicator (SMI) which combines three main characteristics of a given stand: tree species, stand age and aboveground, living and dead wooden biomass. Data on these three factors are used as input to represent the risk of stand loss, which is a function of tree species and stand age, and stand density, which is a function of the silvicultural regime, stand age and tree species. Consequently, the indicator consists of a risk component (SMI_r) and a density component (SMI_d). We used SMI to rank traditional management of the main Central European tree species: Norway spruce (*Picea abies* [Karst.] L.), European beech (*Fagus sylvatica* L.), Scots pine (*Pinus sylvestris* L.), and oak (*Quercus robur* L. and *Quercus petraea* L.). By analysing SMI over their whole rotation period, we found the following ranking of

management intensity: oak < beech < pine << spruce. Additionally, we quantified the SMI of actual research plots of the German Biodiversity exploratories, which represent unmanaged and managed forest stands including conifer forests cultivated outside their natural range. SMI not only successfully separate managed from unmanaged forests, but also reflected the variability of forest management and stand properties across the entire sample and within the different management groups. We suggest using SMI to quantify silvicultural management intensity of stands differing in species composition, age, silvicultural system (even-aged vs. uneven-aged), thinning grade and stages of stand conversion from one stand type into another. Using SMI may facilitate the assessment of the impact of forest management intensity on biodiversity in temperate forests.

Keywords Land-use intensity · Silvicultural systems · Biodiversity exploratories · Thinning · Unmanaged

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Introduction

Land-use intensity is known to be a major driver of biodiversity. However, although the relationship between land use and biodiversity has recently become a hot spot of ecological research, the actual impact of land-use intensity on various taxa is not known. Thus, for agricultural systems and for forests, there is an increasing interest in whether or not, and to what extent, biodiversity is influenced by management intensity (Fischer et al. 2010). For forests, such analyses require a large set of stands that differ in attributes thought to be relevant for stand management, such as tree species composition, stand age, and stand density. Additionally, the stands need to be positioned along a management intensity gradient. However, as

the following literature review shows, a purely quantitative measure for the assessment of forest management intensity of single stands is lacking, although several approaches exist. The same is true for silvicultural regimes, which describe the planned sequence of silvicultural interventions in stand development phases.

The objectives of this study are therefore (1) to review existing approaches, (2) to present a purely quantitative measure of silvicultural management intensity' (SMI), (3) to quantify the components of SMI for main tree species in Central Europe, (4) to demonstrate that the indicator is applicable both for silvicultural regimes as well as for specific developmental phases in these regimes (i.e. forest stands at a given age).

Literature review

Conceptual approaches for assessing land-use intensity

From a systems point of view, land-use intensity can be described either by the output obtained from a unit of land in terms of natural or monetary yield (Turner and Doolittle 1978) or by the input invested into managing the unit of land to achieve output in terms of human effort, materials, chemicals, energy, etc. or money (Herzog et al. 2006; Arano and Munn 2006). Theoretically, the output approach is more appropriate when comparing a single crop under different management regimes, whereas the input approach is preferred when key input factors exist across the land-uses studied, for example fertilization (Dormann et al. 2007; Kleijn et al. 2009) or pesticide application (Geiger et al. 2010; Gibbs et al. 2009). Management intensity includes different management components. Thus, it is taken into account (1) by defining discrete categories such as managed versus natural (e.g. Paillet et al. 2010), and intensive versus extensive regimes (Batary et al. 2007; Bowman et al. 2000; Meier et al. 2005), (2) by using an ordinal gradient of 3–4 levels (Flynn et al. 2009; Müller et al. 2007a; Oehl et al. 2003), (3) by deriving an index (Herzog et al. 2006; Liira et al. 2007; Mas and Dietsch 2003; Wilson et al. 2003), (4) by using the main axis of a principal component analysis of management components (Kerr and Cihlar 2004) or (5) by simply treating the single management components separately in multivariate statistics (e.g. Dormann et al. 2007; Billeter et al. 2008; Grandchamp et al. 2005; Kleijn et al. 2009).

Approaches which measure the output of forest systems, that is yield or harvests, irrespective of stand state or management regime, may be influenced by changes in the efficiency of land use. Thus, for example, an increase in productivity may be due to improved management skills, increased productivity of crops through selective breeding

or by recovery of previously over-utilized ecosystems. Concepts based on the assessment of management input are challenged by a multitude of qualitatively different management practices between and within land uses. Therefore, many approaches rely on categorical, dichotomous definitions of land use. Forests, for example, are classified as unmanaged or managed, and close-to-nature forestry contrasts with even-aged, or age class-based, forestry. Ordinal gradients may distinguish between undisturbed forests, disturbed forests, and timber plantations. However, qualitative categories are broad. Additionally, the position of a given land-use category along a land-use intensity gradient is ambiguous. In the case of forests, these categories do not adequately address interactions of tree species composition, stand age, and logging intensity.

Approaches for assessing forest management intensity

Approaches designed to assess *land-use intensity* across the full spectrum of land-use types are based on artificial concepts which do not allow direct measurement, like hemeroby (Jalas 1955; Sukopp 1969; Naveh and Lieberman 1994; Hill et al. 2002; Grabherr et al. 1998), or use quantities difficult to measure, like human appropriation of net primary production (HANPP: Erb et al. 2009; Haberl et al. 2007; Vitousek et al. 1986), or quantify the difference between the actual state of a system and a reference state (often based on vegetation composition) like naturalness (Dierschke 1984). The HANPP concept is one of the few non-monetary quantitative concepts that focuses on system output by quantitatively assessing the amount of net primary production (NPP) harvested or destroyed during harvest. However, HANPP also includes a component which relates to the system state because the production opportunity induced by humans through for example land cover change, irrigation, fertilization and soil degradation is also considered by relating the NPP of the actual vegetation to the potential NPP of a reference state (i.e. natural vegetation). In contrast, the hemeroby concept adopts a blend of criteria related to system input, output, and state (e.g. fertilization, biomass removal, soil properties, and vegetation composition) to assess qualitatively the degree of anthropogenic influence on a land unit, especially the vegetation component, using an ordinal scale.

The approaches developed to assess *forest management intensity* vary considerably in terms of the (1) conceptual approach in general, (2) indicators used, and (3) scale of intensity. Three main approaches can be identified (Table 1).

The naturalness approach, also referred to as the reverse hemeroby approach (e.g. Winter et al. 2010), had already been suggested 52 years ago (Sukopp 1969; Jalas 1955). It tries to quantify the deviance in the actual system state of a

Table 1 Overview of approaches characterizing forest management intensity grouped according to the main characteristics of land use (hemeroby/naturalness, disturbance, and management) and the number of indicators employed (multi-criteria indicator systems or classifications, key-indicator approaches, key-factor approaches)

Hemeroby and naturalness approaches

Multi-criteria systems

Scoring systems (Šmelko and Fabrika 2007; Bartha 2004; Grabherr et al. 1998; Winter et al. 2010)

Key-indicator-related approaches

State-related key-indicators (Luyssaert et al. 2011)

Disturbance approaches

Multi-criteria systems

Anthropogenic disturbances (Kohv and Liira 2005; Liira and Sepp 2009; Liira et al. 2007)

Key-indicator-related approaches

State-related key-indicators (Seymour et al. 2002; Leniere and Houle 2006)

Management approaches

Multi-criteria classifications

Principal management alternatives (Bell et al. 2008; Duncker et al. 2008; Mason and Perks 2011; Wulder et al. 2007)

Distinct management types (e.g. Müller et al. 2007a, b; Verwer et al. 2008)

Key-factor-related approaches (rate related)

Related to financial inputs (Arano and Munn 2006)

Related to actual relative productivity and harvest of biomass (Haberl et al. 2007; Vitousek et al. 1986)

Related to harvest history (Zenner et al. 2006; Sippola et al. 2004; Storaunet et al. 2005; Uotila et al. 2002; Aguilar-Amuchastegui and Henebry 2007)

Related to thinning and harvest history (Kueffer and Senn-Irlet 2005)

We use the term factor when a causal link between the topic and the indicator is given, for example the absolute or relative number of trees harvested per unit area and time causally describes management

forest stand from a natural reference state, which is based on properties observed in primary forests (i.e. largely undisturbed forests in case of European temperate forests; e.g. Tabaku 1999; Meyer et al. 2003; Christensen et al. 2005; Drössler and von Lüpke 2005, 2007) or ascribed to the so-called potential natural vegetation (Tüxen 1956; Leuschner 1997; Bohn and Neuhäusl 2004). This approach is often applied to temperate forests, where a number of indicators of naturalness have been suggested. The most important of these are tree species composition, quantity and decay status of dead wood, spatial structure, number of very old trees, gap size, etc. To undertake a comprehensive assessment of forest state, that is by employing a number of selected indicators, naturalness approaches tend to be formalized as multi-criteria scoring systems (Winter et al. 2010; Šmelko and Fabrika 2007; Grabherr et al. 1998; Bartha 2004), the most extensive being that presented by Grabherr et al. (1998). These authors describe a specific reference state for each forest type in Austria. Another approach measuring forest land-use intensity by relating actual state to a natural reference state was recently presented by Luyssaert et al. (2011). In this case, two indicators, stand density and the diameter of the mean basal area tree in a given stand, are used and their deviation from the self-thinning line determined. However, this approach does not distinguish between tree species and site.

The disturbance approach follows different concepts: On the one hand, the occurrence and strength of anthropogenic disturbances in forest stands may be quantified by measurements which do not require a reference state (e.g. cut stumps, forest tracks, neighbouring clear-cut areas, ditches, etc.; Kohv and Liira 2005; Liira and Sepp 2009; Liira et al. 2007). However, the general suitability of this approach is not yet clear because studies so far focus on specific forest types in small geographic regions. Again, no distinction between different tree species is made. On the other hand, the temporal and spatial deviance between natural and anthropogenic disturbances is assessed for contiguous areas treated by single silvicultural interventions. The deviance is interpreted as a measure of naturalness of forest management regimes (Seymour et al. 2002). However, the approach of Seymour et al. (2002) addresses the landscape level rather than single stands.

The management approach either directly quantifies the intensity of forest management operations in forest stands (Aguilar-Amuchastegui and Henebry 2007; Kueffer and Senn-Irlet 2005; Storaunet et al. 2005; Uotila et al. 2002; Sippola et al. 2004; Zenner et al. 2006; Arano and Munn 2006) or assesses the forest management regime applied in forest stands as a whole (Müller et al. 2007a, b; Verwer et al. 2008; Wulder et al. 2007; Duncker et al. 2008; Bell et al. 2008). In contrast to the other two approaches, which

address the effect or evidence of land use respectively, the management approach quantifies the intensity of management itself. Different management practices therefore represent factors rather than indicators. Some relevant papers have a general focus and use multi-criteria classifications based on management decisions and practices (e.g. choice of tree species; type of regeneration; extent of soil preparation, machine operation, fertilization and liming, application of pesticides and herbicides; integration of succession elements, protection and development of habitats, harvested biomass compartments, final harvest system, rotation length) (Duncker et al. 2008; Bell et al. 2008; Mason and Perks 2011; Edwards et al. 2011). Others are based on factorial accuracy, using a metric scale of key management factors for a limited set of management practices (e.g. annual investment expenses (Arano and Munn 2006), harvest history based on tree number, basal area, or volume (Zenner et al. 2006; Sippola et al. 2004; Storaunet et al. 2005; Uotila et al. 2002; Aguilar-Amuchastegui and Henebry 2007) and time since the last silvicultural thinning or harvest operation (Kueffer and Senn-Irlet 2005)). However, there is no widely accepted concept on how to combine these diverse factors into a generally applicable, simple, yet accurate measure of forest management intensity.

Conclusions for designing a silvicultural management intensity index

Silvicultural stand management affects the compositional, horizontal and vertical structure of forest stands as well as

the fluxes of organic matter and nutrients in several ways. Thus, basically any measure referring to forest structure, to the export of organic matter and nutrients or to (size and) changes in organic matter and nutrient pools may be said to characterize ‘intensity’ for good reason. However, for actual forest management in Central Europe, which ensures sustainability of wood harvests, wood and nutrient exports may not be the most relevant indicators for stand-level intensity. In fact, presently, the harvest yield is generally lower than the average increment (German Federal Ministry of Food 2006; Ciaia et al. 2008), and soil fertility does not decrease even without fertilization on many sites if whole tree harvests are not conducted (Blanco et al. 2005; Meiwes et al. 2008; Saarsalmi et al. 2010).

Management decisions influencing SMI on the strategic level

We suggest that, in Central European forestry, the most prominent features distinguishing stands from each other in terms of intensity are the crop species selected, the stand age and the reduction in growing stock and removal through tending, thinning, and final harvest. The selection of the crop species is strategically the most fundamental management decision because it shapes the stand for the length of the rotation period (Table 2). Employing more productive tree species is commonly perceived as intensification, as the yield increases and the less productive tree species, which may have been growing under natural conditions, were replaced. Thus, for example, management intensity might increase when beech is replaced by spruce

Table 2 Qualitative relationship between silvicultural management intensity and components of strategic and operational forest stand management

Forest management intensity increases with	Ecological interpretation
<i>Strategic level</i>	
Selecting/raising more productive species	Change in species composition
Selecting/raising species more susceptible to natural disturbances	Decreased system stability and change in species composition
Decreasing length of rotation period	Decreased system maturity
Larger size of management units	Spatial homogenization with temporal synchronization
More uniform silvicultural management (esp. even-aged forest management)	Temporal synchronization with spatial homogenization
<i>Operational level</i>	
Increasing harvest ratio (unit harvest per unit production)	Increased matter, energy and nutrient export
Increasing control of non-crop species	Change in species composition
Increasing productivity of merchantable timber without species change	Unknown (perhaps: increased allocation towards wood or increased NPP)
Increasing strength of thinning and tending	Increased intensity of disturbances
Increasing frequency of thinning and tending	Increased frequency of disturbances
Increasing share of artificial regeneration	Decreased natural self-reproduction
Decreasing density of artificial regeneration when no natural regeneration is intended	Decreased on site selection capacity

or, subsequently, spruce by Douglas fir. However, as shown for oak and Scots pine (Table 3), not every change in species composition results in increased productivity. We therefore suggest that stand compositional changes should not be equated with intensification per se, for example by determining the difference between natural and actual tree species composition, nor by increased yield. However, tree species selection not only affects the expected yield but also the susceptibility of a stand to natural disturbances, for example wind throw and biotic pathogens. With increasing susceptibility, a stand needs to be managed more frequently and more intensively in order to minimize the probability of stand loss. Examples include the repeated removal of wood residues, which are used as breeding places by harmful insects, or repeated thinnings to improve single tree stability against windthrow. Nonetheless, all these measures signal increased management intensity. Finally, tree species selection also affects rotation length due to the species-specific stand growth characteristics (Table 3). A shorter rotation length generally signals increasing management intensity as the stand is replaced more frequently and investments are bound for a shorter period of time. Also, within species, the rotation length may be reduced by modern silvicultural management strategies like final harvest before culmination of mean annual increment (MAI). Rotation length may also decrease with an increase in individual tree’s growth rates through repeated heavy thinning from above.

Management decisions at the strategic level that are independent of tree species are confined to spatial aspects

such as extent and patchiness of silvicultural management operations. Larger forest management units may be associated with an increase in management intensity per se, because, in larger forests, more options exist for rationalizing workflow, and fostering spatial homogenization and temporal synchronization. However, the size of management units in Central Europe largely depends on forest ownership. While large public (state, municipalities) and private (usually private enterprises with inherited property) landowners establish management units for their economical and ecological advantage within a contiguous forest property, management options of small woodland owners are constrained by forest parcel size. This interaction of size and ownership, in combination with the small variability in stand size of large landowners, puts the (potentially) indicative value of spatial aspects of management units into perspective.

The forest management system employed, that is age class versus uneven-aged management, represents another strategic decision important for SMI. The gradient of uniform to patchy forest management is characterized by the extremes of clear-cut and single tree selection system (or plenter forests). If patchiness, duration and extent of overlap of the overstorey, and regeneration are taken into account, other silvicultural systems relying on retention trees, for example shelterwood cuttings or group selection systems lie between these two extremes. Even-aged management, however, is not exclusively related to the regeneration system but may also distinguish uniform schemes of thinning and harvesting, for example regularly spaced

Table 3 Characteristics from German yield tables site class 1 for European beech, sessile and pedunculate oak, Norway spruce and Scots pine (Schober 1987)

	European beech		Sessile and pedunculate oak		Norway spruce	Scots pine
Yield table	Schober 1987		Jüttner 1955		Wiedemann 1936/42	Wiedemann 1943
Site index $h_{100}(100)$ (m)	33		28		35	29
Thinning grade	Moder.	Heavy	Moder.	Heavy	Moderate	Moderate
Total wood increment (I_{V-100}) ($m^3 ha^{-1} 100 years^{-1}$)	781	756	666	699	1,219	779
First age tabulated (years)	30		20		20	25
Last age tabulated (years)	150		200		120	140
Min basal area G ($m^2 ha^{-1}$)	14.2	14.2	15.4	15.5	26.0	25.6
At age (years)	30		20		20	25
Max basal area G ($m^2 ha^{-1}$)	33.8	26.1	27.4	23.0	48.3	33.8
At age (years)	150		200		100	95
Maximum natural basal area $G_{nat,max}$ ($m^2 ha^{-1}$)	42.8		35.5		58.9	41.4
Due to h_{100} (m)	38.4		34.2		37.3	32.1
At age (years)	150		200		120	140
Maximum natural stocking density	0.79	0.61	0.77	0.65	0.82	0.82

Yield tables describe thinnings from below every 5 years. Maximum natural basal area $G_{nat,max}$ was estimated according to Sterba (1981, 1987) with parameters given by Spellmann et al. (1999). Maximum basal area, the main setting for determining the density component of SMI, is highlighted in bold

cuttings and strip-cutting, from non-uniform strategies, for example selective thinning (Table 2).

In summary, we conclude that two factors are relevant for management intensity at the strategic level: (1) stand age and tree species-specific risk of stand loss and (2) abundance of mature overstorey trees over the whole regeneration phase. The risk approach avoids an assessment of tree species per se, for example by adopting a productivity gradient, as well as by defining a reference state relative to natural vegetation. The unbiased nature of both assessments may be difficult to ensure. Thus, it is not clear how the difference in productivity of different tree species could be transferred into a consistent indicator of SMI. Although there might be a correlation between naturalness and management intensity, a quantitative assessment of naturalness at a stand level against an assumed reference state is difficult to achieve because adequate references other than mountainous Central European forests do not exist anymore. Instead, we suggest that the choice of a certain tree species indicates the acceptance of a higher risk, which in turn reflects a strategic decision to increase management intensity. With regard to the stand age-related risk component, we suggest that management intensity should logically decrease with stand age, since existing older stands have survived thus far. For the second factor, quantifying the temporal and spatial overlap of mature trees and regeneration, we propose to measure the relative deviance between actual biomass and biomass carrying capacity, with carrying capacity being species and site specific.

Management decisions influencing SMI at the operational level

Silvicultural management intensity at the operational level is mostly related to fellings through tending, thinning, and harvest operations, which reduce stem number, basal area, growing stock (merchantable wood volume), and biomass. Fellings may also affect tree species composition. As the felled trees partly remain in the forest as harvest residuals (e.g. small trees, branches, stem sections with defects), removals do not equal fellings. In the case where trees are lost due to natural events (e.g. ageing, windthrow) and remain in the stand, the discrepancy between “fellings” and removals becomes even more evident. Therefore, removals are more indicative of silvicultural management intensity than trees lost due to natural or silvicultural reasons. Thus, we propose to measure removals by the deviance between actual biomass of living and dead trees and maximum biomass, with maximum biomass being species, age and site specific.

Stand regeneration by planting and sowing is generally considered to reflect more intensive silvicultural operations than by promoting natural regeneration. Aside from the active

introduction of seedlings or seeds, artificial regeneration may require additional measures like soil preparation and weed control. Consequently, management intensity would increase with increasing number of trees planted. However, we assume that exactly the opposite applies as a low plant density results in a reduced potential of natural selection. The reduction in planted seedling density to the lowest possible number for still high-quality timber production is therefore seen as an attribute of higher silvicultural intensity. Thus, we suggest assessing silvicultural management intensity in artificially regenerated young stands analogously to fellings in later developmental phases. It could therefore be defined as the deviation in tree density from a reference value representing natural regeneration density. However, since the effect of tree removal on silvicultural management intensity is already very high in young stands due to low biomass, the additional absolute effect of low planting densities is negligible. In contrast to regular plantings, enrichment plantings and planting in advance under the canopy of mature stands are not regarded as intensive measures. These plantings or direct seedings normally aim at the conversion of pure (often conifer) stands into mixed stands for various ecological and economical reasons (Ammer et al. 2008; Knoke et al. 2008). Such efforts therefore contribute to future stand stability and do not affect present stand management intensity.

Materials and methods

Definition of SMI

Based on the considerations above, we suggest that SMI can be described by two components, risk of stand loss and stand density, which theoretically are independent of one another. The risk component defines the combined effect of tree species selection and stand age on SMI. The stand density component quantifies the effect of removals and regeneration method using actual biomass related to a reference. Both components are applicable to stand management regimes as well as to individual stands. Here, we use basal area as a proxy of biomass, because the dependency of basal area on species, age and site quality is well documented. However, the following definitions also apply to biomass and to other proxies like stem volume.

Risk component

We define the risk component of silvicultural management intensity (SMI_r) as the age-dependent probability of a stand at age t_0 to be lost before reaching a specific reference age t_r ($t_r > t_0$). As such, it reflects the effects of stand age and species selection on the probability of stand loss. According to survival analysis (Sachs and Hedderich 2009;

Crawley 2007), the survivor function S is defined as $S(t) = P(T > t)$ where t is time, T is a random variable denoting the time of an event, in this case stand loss and P represents probability. Thus, the survivor function represents the probability that stand loss occurs after a specified time. S decreases monotonously with $S(0) = 1$ and $\lim_{t \rightarrow \infty} S(t) = 0$. The lifetime distribution function F , defined as the complement of the survivor function $F(t) = P(T \leq t) = 1 - S(t)$, represents the probability that an event occurs at or before a specified time. Consequently, the probability of the stand loss at or before age t_r , given survival to age t_0 , is

$$P(T \leq t_r | T > t_0) = \frac{P(t_0 < T \leq t_r)}{P(T > t_0)} = \frac{F(t_r) - F(t_0)}{S(t_0)}, \quad (1)$$

and the special case of probability of stand loss at or before the reference age of 180 years for stands of age t_0 ($180 > t_0$) $P_{180}(t_0)$, i.e. SMI_r , is

$$\begin{aligned} SMI_r = P_{180}(t_0) &= P(T \leq 180 | T > t_0) = \frac{F(180) - F(t_0)}{S(t_0)} \\ &= \frac{S(t_0) - S(180)}{S(t_0)} = 1 - \frac{S(180)}{S(t_0)}. \end{aligned} \quad (2)$$

For stands older than 180 years, SMI_r is considered to be zero. $P_{180}(t_0)$ decreases monotonously with $P_{180}(0) = F(180)$ and $P_{180}(180) = 0$. Thus, we define the risk component of silvicultural management intensity as the age-dependent probability of losing a stand before it reaches 180 years. A reference age of 180 years is proposed because, at this age, managed forests also show features of old-growth forests (Moning and Müller 2009). However, other reference ages result in only slightly changed assessments (Supplementary material, Fig. 2). Note that this definition of SMI_r should not be confused with hazard rate h ($h(t) = F'(t)/S(t)$ where the event density function F' is derivation of F). Hazard rate h describes the probability that a stand will be lost at a specific age t conditional to survival to that age. The causes of stand loss represented by the survivor function are calamities such as wind throw and pest damage, but not natural stand decay due to senescence. The approach may be extended to assess species mixtures. We suggest that the risk for the main stand will not increase by admixing species of higher risk. In this case, the risk of the mixed stand is defined by the risk of the main stand, because a failing of admixed species does not affect total stand survival. Moreover, the specific risk of the admixed species may even decrease (Griess et al. 2012). However, when species comprising a lower risk are admixed to a main stand, then the risk of stand loss of the mixed stand will be reduced proportionally, since admixed species should not be affected by calamities in the same extent as the main stand.

For the assessment of silvicultural regimes, we propose the mean SMI_r over the length of the rotation period rp (which can be quantified numerically):

$$SMI_{r, \text{regime}} = \frac{1}{rp} \cdot \int_0^{rp} SMI_r(t_0) \cdot dt_0. \quad (3)$$

SMI_r of the silvicultural regime thus reflects the species-related survival probabilities and rotation periods.

Quantifying SMI_r

Survival functions of calamities for the main Central European forest tree species were compiled by Staupendahl (2011), who conducted a literature review on stand survival probability for Austria, Germany and Switzerland based on studies by Brücken (1984) (Norway spruce), Dittrich (1986) (European beech, sessile and pedunculate oak, Norway spruce, and Scots pine) and König (1996) (Norway spruce). Original studies on survival probabilities were also conducted by Kouba (2002) and Hanewinkel and Holecý (2005) (Norway spruce, European silver fir). However, most studies were focused on Norway spruce. Only Dittrich (1986) also investigated European beech, sessile and pedunculate oak, and Scots pine. His work, however, covered a period (1971–1981) and region (GDR, i.e. former East Germany) known for its high air pollution levels, which were well above that of the present day. Recently, Staupendahl and Zucchini (2011) developed survival functions for European beech, sessile and pedunculate oak, Norway spruce, Scots pine and Douglas fir based on inventory data systematically sampled in the Rhineland-Palatinate, Germany, between 1994 and 2008. For European beech, oaks and Scots pine, stand ages range from young stands up to 180 years and older, while for Norway spruce, the oldest stands were about 150 years old. As the forests of Rhineland-Palatinate are mainly located in the submontane zone, yet also comprise a considerable fraction in low to middle montane and colline zones, these findings are believed to be representative for the present day Central Europe.

Staupendahl and Zucchini (2011) assumed T to be Weibull-distributed (Kouba 2002; Holecý and Hanewinkel 2006), so that the survivor function S is given as:

$$S(t) = \exp \left[- \left(\frac{t}{\beta} \right)^\alpha \right] \quad (4)$$

where α denotes the form parameter, and β the scale parameter of the Weibull-distribution. Survival functions of the tree species were found to deviate widely (Supplementary material, Table 1 and Fig. 1) with European beech and oaks having the lowest probability of stand loss. Stands of these species are expected to reach 100 years of age with a probability of 97 %. The high life expectancy at stand establishment of European beech highlights the fact that

Staupendahl and Zucchini (2011) only considered calamities and not natural stand decay. Compared to deciduous species, the probability of stand loss for Scots pine and Norway spruce is higher for stands older than about 80 years, and about 40 years respectively. For Norway spruce, the probability of reaching 100 years is 73 %.

The risk component of silvicultural management intensity SMI_r , determined with the parameters of Staupendahl and Zucchini (2011) (Supplementary material, Table 1) by using formula 2 with S substituted by formula 4, is highest for Norway spruce, followed by Scots pine, sessile and pedunculate oak, and European beech (Fig. 1). Between the deciduous species, the difference is relatively small. SMI_r for Scots pine is twice as high as for the two deciduous species. For Scots pine, sessile and pedunculate oak and European beech, SMI_r declines gradually with stand age. For Norway spruce, the decline increases when stand age approaches the reference age. Order and relative distance between the species are largely independent of the reference age (Supplementary material, Fig. 2), which makes the quantification of SMI_r robust against differences in ranges in stand age between species in the original data and uncertainties in the definition of the reference age. With increasing reference age, the distance between Norway spruce and the other species even decreases slightly. Silvicultural regimes with rotation periods of for example 120 years for Norway spruce, 150 years for European beech, 140 years for Scots pine, and 180 years for sessile and pedunculate oak result in a $SMI_{r,regime}$ of 0.773, 0.067, 0.244 and 0.103 respectively.

Stand density component

The change in basal area (G in $m^2 ha^{-1}$) over time of unmanaged even-aged stands undergoing self-thinning is characterized by natural basal area (G_{nat}) (Assmann 1961; Assmann and Franz 1965; Pretzsch 2009) and eventually

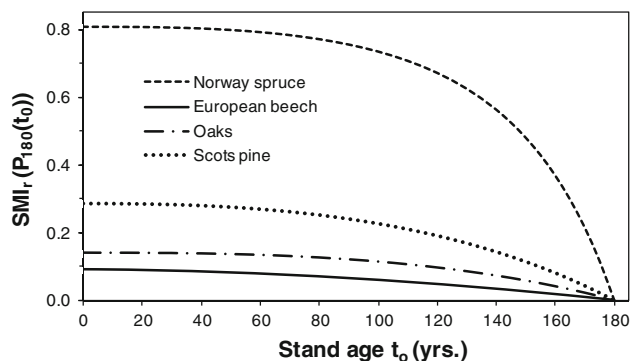


Fig. 1 Risk component of silvicultural management intensity (SMI_r) for the main Central European tree species, based on a survival analysis for Rhineland-Palatinate (Staupendahl and Zucchini 2011)

approaches the maximum natural basal area ($G_{nat,max}$), which defines the carrying capacity of a site in terms of G (Odum 1969; Pretzsch 2009). We quantify the SMI_d as the relative deviation of G from $G_{nat,max}$ (Formula 5). For even-aged stands, the deviation is composed of two separate additive effects, namely the age class effect ($G_{nat,max} - G_{nat}$) and thinning effect ($G_{nat} - G$), which reflects Assmann's (1961) natural stocking density (G/G_{nat}). The effects clearly depend on stand age, with the age class effect dominating in young stands and the thinning effect in old ones (Fig. 2). In uneven-aged stands (e.g. plenter forests), the age class and thinning effects intermingle, so that SMI_d , but not its two components, can be quantified.

$$SMI_d = \frac{(G_{nat,max} - G_{nat}) + (G_{nat} - G)}{G_{nat,max}} = \frac{G_{nat,max} - G}{G_{nat,max}} = 1 - \frac{G}{G_{nat,max}} \quad (5)$$

The approach can also be extended to address mixed species stands by defining the subtrahend term in Formula 5 for each species. Because stand G of mixed stands is given by the sum of species-specific G 's, SMI_d for mixed stands is defined by the difference between unity and the sum of basal areas (G_x) of single species ($x = 1 \dots S$) relative to their specific maximum natural basal area ($G_{nat,max,x}$):

$$SMI_d = 1 - \sum_{x=1}^S \frac{G_x}{G_{nat,max,x}} \quad (6)$$

This definition for mixed stands does not, however, consider mutual facilitation of species, which may lead to

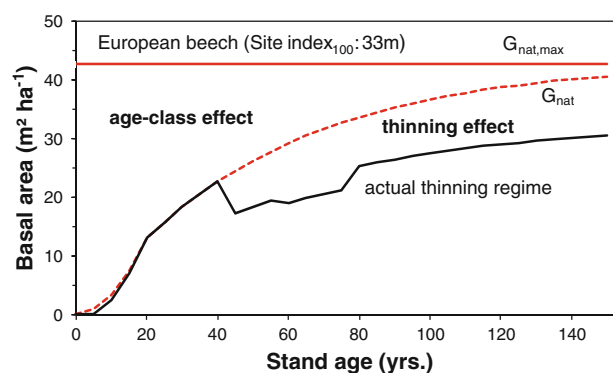


Fig. 2 Density component of silvicultural management intensity (SMI_d) with age class effect and thinning effect for an even-aged stand of European beech with site index 33 m. The actual thinning regime follows standard stand treatment prescriptions for Lower Saxony (Spellmann et al. 1999; Röhrig et al. 2006), which recommend a target value for natural stocking density depending on dominant height. Change in natural basal area over time according to Spellmann et al. (1999)

transgressive over-yielding (Pretzsch and Schütze 2009) and therefore a G and/or G_{nat} , and $G_{nat,max}$ higher than expected from pure stands of a given species.

Besides actual stand density, or any other past or future density, silvicultural regimes may also be assessed by SMI_d . We quantify silvicultural management intensity of a silvicultural regime by the mean SMI_d over the length of the rotation period rp :

$$SMI_{d,regime} = \frac{1}{rp} \cdot \int_0^{rp} SMI_d(t) \cdot dt. \tag{7}$$

Quantifying SMI_d

For Central Europe, natural basal area G_{nat} is usually estimated based on data from long-term experimental plots of unthinned stands and lightly thinned stands (A grade thinning) in which only dead and suppressed trees showing symptoms of dying have been harvested. Basically two approaches have been used. Assmann and Franz (1965) estimated $G_{nat,max}$ and G_{nat} dependent on site index (i.e. dominant height at the age 100 years; $h_{dom}(100)$) and yield level (Assmann 1961; Skovsgaard and Vanclay 2008) by an equation system including additional definitions and constraints. However, this approach was only applied to Norway spruce growing in Bavaria. In contrast, Sterba (1981, 1987) combined the self-thinning rule (see Reineke 1933; Kira and Sihdei 1967; Yoda et al. 1963) with Eichhorn’s rule (Eichhorn 1904; Skovsgaard and Vanclay 2008) to estimate G_{nat} from dominant height h_{dom} , ignoring yield levels. Sterba’s approach is based on the relationship between quadratic mean DBH dg and stand density N for stands of constant dominant height h_{dom} :

$$dg = \frac{1}{A \cdot N + B}, \quad \text{with } A = a_0 h_{dom}^{a_1} \quad \text{and} \quad B = b_0 h_{dom}^{b_1} \tag{8}$$

where a_0 , a_1 , b_0 , and b_1 are parameters which can be estimated by nonlinear regression. As Sterba (1981, 1987) demonstrated that stand density at the maximum basal area for constant dominant height is given by $N_{G_{max}} = B/A$, the relationship between G_{nat} and h_{dom} becomes:

$$G_{nat} = \frac{\pi}{16 \cdot A \cdot B} = \frac{\pi}{16 \cdot a_0 \cdot b_0 \cdot h_{dom}^{a_1+b_1}}. \tag{9}$$

Spellmann et al. (1999) determined the parameters of Eq. (9) for European beech, sessile and pedunculate oak, Norway spruce and Scots pine growing in north-western Germany. Döbbeler and Spellmann (2002) extended this approach to other areas in Germany by considering differences in yield levels between four regions (Supplementary material, Fig. 3). For European beech, yield levels were found to be higher in Northwest and Southeast Germany compared to East and Southwest.

For even-aged stands, G_{nat} may be estimated using actual h_{dom} and $G_{nat,max}$ using the actual site index, which depends on stand age and actual h_{dom} . Site index curves (Pretzsch 2009; Skovsgaard and Vanclay 2008) could be used to assess expected maximum h_{dom} . For uneven-aged stands (plenter forests), $G_{nat,max}$ may also be calculated via site index, but using height growth of even-aged stands on comparable sites as reference. For silvicultural regimes, $SMI_{d,regime}$ may be determined by estimating the age dependency of G_{nat} via the growth of h_{dom} with stand age for a specific site index, and, subsequently, relating G_{nat} and $G_{nat,max}$ to the prescribed time course of G (Eq. 7).

Aggregation of components

SMI_r and SMI_d range between 0 and 1. For the density component SMI_d , the lowest intensity represents a fully stocked mature stand, and the highest an unstocked stand just after a clear-cut or a young stand not yet reaching breast height (caliper threshold 1 cm). For the risk component SMI_r , the lowest intensity is found in stands that have reached (or passed) the reference age. However, a value of 1 will be achieved only if it is (nearly) assured that stands will be lost due to calamities before the reference age. To keep both components balanced, it thus may be appropriate to scale SMI_r to the species with highest risk of stand loss, as:

$$SMI_{r,scaled} = \frac{SMI_r}{\max F(t_r)} \tag{10}$$

where $\max F(t_r)$ denotes the maximum probability of stand loss before or at the reference age between the species considered.

Although SMI_r and SMI_d describe two distinctly different components of silvicultural management intensity and thus should preferably be addressed separately (or together) in further analyses, both may equally be aggregated to a single parameter of silvicultural intensity. We propose combining the components by viewing them as vectors which are arranged either straight or orthogonal, that is as sum of vectors or as the length of the Euclidean vector. For convenience, sum of vectors and length of the Euclidean vector may be replaced by their equivalents that scale the potential maximum to 1, the arithmetic and the quadratic mean.

Application of SMI

SMI of actual forest stands

In order to demonstrate the response and sensitivity of SMI and its components to actual forest management, we employed the measure to the 30 so-called very intensive

plots studied within the Biodiversity Exploratories (Fischer et al. 2010; see Appendix). These plots represent the range of land-use intensity (for details, see Fischer et al. 2010) found within the three exploratories Schäubische Alb, Hainich-Dün and Schorfheide-Chorin, namely: managed, even-aged forest with natural vegetation replaced by conifers (coniferous forest:); managed even-aged and uneven-aged forest without species change (beech forest and selection forest); formerly managed forests left unmanaged for decades (unmanaged forest). Stand characteristics are summarized in Supplementary material, Table 2.

Species and diameter at breast height (dbh) of trees (caliper limit dbh >7 cm) growing on the plots (VIPs) were surveyed in late 2012. 26 VIPs were sampled using five circular subplots of size 500 m² (12.62 m in radius) per plot. The subplots were located on a diagonal cross within each quadratic one ha plot. The distance between the centres of the diagonal subplots was randomly chosen between 35 and 50 m each. For 3 VIPs, species and dbh of all trees were obtained by the local management teams of the Biodiversity Exploratories. Additionally, basal area was estimated for one VIP covered by a thicket of European beech. We used this stand inventory data together with records of the forest administration to determine SMI. For the majority of plots stand age, the age of different stand layers or of admixed species was obtained from records from the various forest administrations. However, for unmanaged and selection forests in the Hainich-Dün stand age, assessed as age of the overstorey, was estimated from diameter of the largest 30 trees per ha based on data which Mund (2004) sampled in nearby selection forests. Maximum natural basal area $G_{nat,max}$ for plots was estimated from species (absolute or relative) site indices using region specific yield tables (Scots pine in Schorfheide-Chorin: Lembcke et al. 1975; European beech in Schorfheide-Chorin and Hainich-Dün: Dittmar et al. 1986; European beech in the Schwäbische Alb: Schober 1972; Norway spruce in Hainich-Dün: Wenk et al. 1984; Norway spruce in the Schwäbische Alb: Wiedemann 1936/42). For Norway spruce growing in Hainich-Dün and the Schwäbische Alb, maximum natural basal area was directly obtained from Assmann and Franz (1965). Beforehand, the site indices of Wenk et al. (1984) and Wiedemann (1936/42) were paralleled with Assmann and Franz (1965) using index height and mean annual increment. The best site qualities for European beech and Scots pine $G_{nat,max}$ for the specific region were estimated according to Döbbeler (2004) using the maximum top heights tabulated in the respective yield tables (at a stand age of 130 years for pine and 160 years for beech). These maximum natural basal areas were found to be about 30 % larger than the standard stocking tabulated for European beech and about 24 %

larger for Scots pine; a finding that corresponds well with the highest stocking degrees considered by Dittmar et al. (1986, p. 55: 1.3) and Lembcke et al. (1975, p 58: 1.15). Thus, for lower site qualities, $G_{nat,max}$ was estimated by adding 30 and 24 %, respectively. For oak, which is admixed on only some plots, we generally fixed $G_{nat,max}$ to 80 % of that of beech (Röhrig et al. 2006). Other species, which occasionally occur in some plots, were appended to European beech when broad-leafed, to Norway spruce when coniferous and growing in the Schwäbische Alb or Hainich-Dün and to Scots pine when coniferous and growing in Schorfheide-Chorin.

Risk and density components of silvicultural management intensity were calculated as described above (Eqs. (2), (4) with parameters from Staupendahl and Zucchini (2011), Supplementary material, Table 1 for SMI_r and Eqs. (5), (6) for SMI_d). SMI_d was related to basal area of living trees only, as information on deadwood amount was not available. SMI was calculated as average of risk and density components.

SMI of silvicultural regimes

We determined SMI_r , SMI_d and SMI for silvicultural regimes using German yield tables for site class 1 describing moderate thinning (B grade) for European beech, sessile and pedunculate oak, Norway spruce and Scots pine and heavy thinning (C grade) for European beech and oaks (Schober 1987, Table 3). The yield tables are based on silvicultural regimes characterized by thinning from below with an intervention frequency of about 5 years. Under moderate thinning, intermediate fellings comprise about 50 % of total volume production, in the case of oak even 60 %. For heavy thinning, intermediate fellings are higher and stocking degree is decreased (Table 3).

Change over time of G_{nat} was determined from dominant heights h_{100} (tabulated for stand ages) using Eq. (9) with parameters given by Spellmann et al. (1999). $G_{nat,max}$ was estimated by adding 5 % to G_{nat} of the oldest stand age tabulated for European beech, and Scots pine in order to account for some limited additional growth. For oaks and Norway spruce, the correction was 3 and 8 %, as the period tabulated differed. Below the youngest age tabulated, we interpolated G and G_{nat} linearly so that values of 0 were achieved for stands just established, although G is known to grow sigmoidally with a time delay until regeneration reaches breast height. This simplification was deliberately applied here as it affects the thinning effect only marginally and prevents an overestimation of the age class effect.

Risk and density components were calculated as described for actual forest stands (see above).

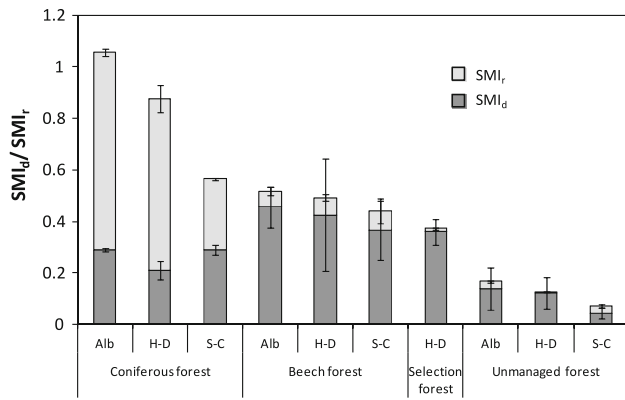


Fig. 3 Silvicultural management intensity of managed and unmanaged forests in the Exploratories Schwäbische Alb (Alb), Hainich-Dün (H-D) and Schorfheide-Chorin (S-C). Coniferous forest: even-aged, managed secondary forests; beech forest: even-aged, managed forests; selection forest: uneven-aged, managed forests; unmanaged forest: formerly managed forests left unmanaged for decades. Error bars the standard error for density and risk components ($N = 3$ per group)

Results

Silvicultural management intensity indicator of the very intensive plots of the Biodiversity Exploratories (see Supplementary material, Table 3; Fig. 4) was found to vary considerably, with an unmanaged beech forest in the Hainich-Dün comprising the lowest value (0.002) and a young Norway spruce forest in the Schwäbische Alb the highest (0.547). SMI_d ranged from 0 (unmanaged beech) to 0.837 (beech thicket), and SMI_r varied between 0 (old managed beech forest) and 0.769 (young Norway spruce forest) (Supplementary material, Table 3 and Fig. 4).

Unmanaged beech forests showed the lowest SMI_r , SMI_d , and SMI (0.102, 0.020, and 0.061) of all management types, with only small variability between and within exploratories (Fig. 3). Within managed beech forests,

uneven-aged forests (selection forests) showed lower SMI than even-aged forests (0.186 vs. 0.242), mainly due to lower values for the risk component SMI_r (0.013 vs. 0.068). In even-aged beech forests, a large variability of the density component SMI_d was found, which can be attributed to the large spread in stand age in this group reflecting the age class effect on SMI_d . Highest silvicultural management intensity SMI (0.416) and highest risk component SMI_r (0.569) were found for coniferous forests. Here, the tree species effect on risk separates Schorfheide-Chorin (Scots pine) from the other exploratories. As coniferous forest plots do not much differ in stand age, the variability of density and risk components was low (Fig. 3, Supplementary material Table 2 and Fig. 4).

Lowest SMI of the moderate thinning regime was found for sessile and pedunculate oak and highest for Norway spruce (Table 4). European beech showed slightly higher SMI_{regime} than oak, and SMI_{regime} of Scots pine was slightly higher than that of European beech. The risk component of the silvicultural regime, $SMI_{r,regime}$, was lowest for European beech (0.067) and increased for sessile and pedunculate oak, Scots pine and Norway spruce by a factor 1.5, 3.6 and 11.5 respectively (Table 4). Compared to the large spread of the risk component between species, the density component of the moderate thinning regime, $SMI_{d,regime}$, was found to have a narrow range (0.310–0.450). Scots pine showed the lowest value, followed by oak, Norway spruce and European beech. The high value of the density component for European beech was not due to a higher thinning grade but to its species-specific slow growth of G_{nat} compared to the other tree species. Therefore, in European beech, the age class effect contributed more to $SMI_{d,regime}$ than the thinning effect (0.338 vs. 0.112 or 75 vs. 25 %), in absolute as well as in relative terms, compared to oak (0.247 vs. 0.125 or 66 vs. 34 %). With the heavy thinning regime, only the thinning effect was increased (European beech: 0.197; oak: 0.213).

Table 4 Silvicultural management intensity for silvicultural regimes represented in German yield tables of site class 1 with thinning from below (Schober 1987)

	European beech		Sessile and pedunculate oak		Norway spruce	Scots pine
Yield table	Schober 1987		Jüttner 1955		Wiedemann 1936/42	Wiedemann 1943
Thinning grade	Moder.	Heavy	Moder.	Heavy	Moderate	Moderate
Rotation period (years)	150		180		120	140
Maximum natural basal area $G_{nat,max}$ ($m^2 ha^{-1}$)	42.8		35.5		58.9	41.4
Mean basal area G ($m^2 ha^{-1}$)	23.5	19.9	22.3	19.2	36.8	28.6
Mean deviation from $G_{nat,max}$ ($m^2 ha^{-1}$)	19.3	22.9	13.2	16.3	22.1	12.8
$SMI_{r,regime}$	0.067		0.103		0.773	0.244
$SMI_{d,regime}$	0.450	0.535	0.372	0.460	0.375	0.310
SMI	0.259	0.301	0.237	0.281	0.574	0.277

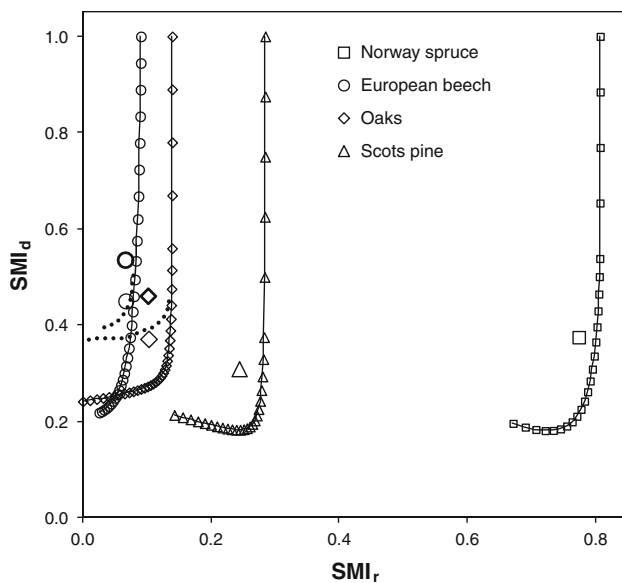


Fig. 4 Trajectories of silvicultural management intensity for silvicultural regimes represented in German yield tables (site class 1, light and heavy thinning from below; Schober 1987). Trajectories begin with regeneration after clear-cut (age = 0) and continue in time steps of 5 years over the rotation length (European beech: 150 years, sessile and pedunculate oak: 180 years, Norway spruce: 120 years, and Scots pine: 140 years). *Solid and dotted lines* light and heavy thinning grade. *Enlarged and enlarged bold symbols* the arithmetic mean for the silvicultural regime for moderate and heavy thinning grade

However, the effect was sufficient to raise SMI_{regime} of European beech and oak above that of Scots pine managed under moderate thinning (Table 4).

The risk and density component of silvicultural management intensity were age dependent for all tree species (Fig. 4). However, SMI_r of the conifer species did not decrease with age to the lowest possible levels since the Norway spruce and Scots pine stands were traditionally harvested long before the reference age t_r (Eq. 2). The development of SMI_d was characterized by species-specific growth and the thinning regime.

Discussion

While for agricultural land, the management inputs and the harvest outputs can be reliably measured within short periods of time, both components vary largely within a forest management cycle. Harvests (output) may be very low in young stands compared to mature stands, whereas, for management inputs, the converse may be true. Consequently, two temporally different forest management intensity measures need to be distinguished: (1) the actual state of a stand (e.g. by addressing stand density, basal area, biomass, stocking volume, diameter distribution) and

(2) the management regime where the actual state of the stand is regarded as one state in a chronological sequence of states. In short, the stand may be viewed as a single observation at a given point in time or as a temporal realization of a management regime. Both measures are addressed by the SMI.

The approach to quantify SMI presented in this study combines three main stand characteristics of a given stand: tree species, age, and biomass. These factors are represented by different risks, which are a function of tree species and age, and different stand densities, which are a function of the silvicultural regime and stand age. The results showed that by regarding components, SMI_r and SMI_d , not only silvicultural regimes, but also actual stands, could be successfully assessed. Unfortunately, the degree of appropriateness of this ranking cannot be tested as there is no absolute, “true” management intensity. We believe that SMI adequately addresses the most important components of any forest management decision: the choice of tree species and its treatment, both of which are addressed by the two components of SMI.

In this study, SMI was calculated by using basal area as a proxy for biomass for practical reasons (basal area of a stand is much easier to determine than stand volume or biomass). Due to the stand age-dependent nonlinear relationship between basal area and biomass (Pretzsch 2009), the age class effect on SMI_d would be more pronounced if related to biomass (or volume) instead of basal area. However, the direction of the response to stand growth and silvicultural interventions is independent of the measure used. Dead wood, which should contribute to SMI_d , could not be considered in this study because such data were not yet available from the biodiversity exploratories.

A forest manager’s perspective

A forest manager must make two basic decisions. First, which species should be planted/sown/regenerated naturally? Second, how will the stand be treated throughout the rotation period? These two questions include various sub-questions such as: How often, and how intensively should the stand be thinned?; Which cutting system should be used in the final harvest, that is a clear-cut or selective cuttings that extend the final harvest over decades?; and, will all mature trees be removed in the final cuttings or will some retention trees be kept as habitat and/or seed trees? Another important question might be whether or not the stand should be converted from an even-aged into an uneven-aged stand. The forest manager’s answers to these questions are very much influenced by his economic expectations. A risk-averse person will most likely consider species characterized by a low risk. In contrast, a risk taker may choose a highly productive species, taking a higher uncertainty of the expected high income into account, that is

appreciating a high variation in financial returns which might be compensated for by an exceptionally high return (Knoke et al. 2005). However, accounting for higher risks means intensification. Thus, a tree species such as Norway spruce which is very productive but vulnerable to abiotic (for example storms, see von Lüpke and Spellmann 1999; Polomski and Kuhn 2001) and biotic (for example bark beetles, see Eriksson et al. 2007; Bolte et al. 2010) threats must be managed much more intensively than a less vulnerable species. In this context, measures such as repeated tendings and thinnings might be necessary to increase single tree stability, salvage cuttings, bark beetle control, etc. Decisions on the frequency and intensity of thinnings are also very much driven by economic considerations. They depend, for example, on the liquidity of the woodland-owner. Even thinnings aiming at increasing stand resilience and resistance can be seen as investments to secure future income.

Rating pure and mixed stands

In general, SMI can be applied to pure and mixed stands, but for mixed stands assessments may be more uncertain. SMI_d for mixed stands is calculated as the weighted sum of pure stands (Eq. 6). However, mixing may increase (overyielding) or decrease (underyielding) biomass carrying capacity (Pretzsch and Schütze 2009). Presently, reliable information on the effect of species mixtures on stand yield is only available for Norway spruce-European beech stands (Pretzsch et al. 2010). However, these studies revealed that overyielding is very much dependent on site quality. For other species, namely mixtures of various broadleaved species, the picture is still unclear. In fact, Jacob et al. (2010) found no over- or under-yielding effects within stands differing in tree species diversity. Accordingly, no general patterns of maximum stand density of mixed stands versus pure stands were found by Woodall et al. (2005).

Unfortunately, the results on the effect of mixtures on stand stability are even more inconsistent than for productivity. Whereas, for example, König (1996) and von Lüpke and Spellmann (1999) did not find a positive effect of tree mixture on stand resistance in the case of Norway spruce and European beech and concluded that stand type is not as important as soil type. Schütz et al. (2006) and Griess et al. (2012) reported evidence of European beech stabilizing admixed Norway spruce in mixed stands. This finding was considered for SMI_r of mixed stands. While admixed species of lower risk proportionally decrease the risk of stand loss, admixed species of higher risk do not affect the risk of the main tree species.

To the best of our knowledge, there are no comprehensive quantitative survival functions derived for the main European tree species within the same region other than those by

Staupendahl and Zucchini (2011). However, Eqs. (2) and (4), defining these functions, could easily be adjusted if other regional information or consistent information about mixed stands were available.

Rating even-aged and uneven-aged stands

Even-aged stand management traditionally leads to a final harvest removing the mature stand in one (clearcut) or several (shelterwood) steps. The result, a cleared area or planted with seedlings, thus reflects a past but very intensive measure, namely the final cut(s). In contrast, modern continuous-cover concepts, selective cutting systems, and uneven-aged stand management never reduce the growing stock to zero. Even though the same amount of timber may be harvested within a given period of time by these alternative approaches, they do not alter the environmental conditions of the forest as intensely as the traditional systems. Moreover, there is no indication that biomass carrying capacity is affected by the age structure of a forest. The different approaches of stand management are represented by SMI_d . Thus, the more constant and high the growing stock, the less intense the interventions have been in the past. In uneven-aged stands, the risk of stand loss, SMI_r , is quantified using the stratum of the oldest stem members. This approach is based on the finding that the risk of stand loss is lower under uneven-aged management (Rottmann 1986).

Measures such as slash removal, soil preparation, fertilization, planting or direct seeding and weeding are usually not necessary in continuous-cover-systems. However, as we focused on silvicultural practices in the narrower sense (thinning, harvests), SMI may be of limited applicability for silvicultural systems which rely very much on the above mentioned measures. For such systems, additional terms quantifying these management measures might be a useful extension of the present approach (Blüthgen et al. 2012). Nevertheless, we believe that the ability of SMI to contrast the different systems and all stages of transformation from one system to another or to rate different thinning concepts is advantageous. It takes the basic elements of the conceptually well-established HANPP approach into account (Vitousek et al. 1986; Haberl et al. 2007). However, HANPP has not yet been elaborated at the stand level for Central European forests managed at varying intensities.

SMI and biodiversity

As outlined in the introduction, there is an increasing interest in the impact of land-use intensity on biodiversity. There are already a number of quantitative and qualitative indicators representing forest structure (see the review of

McElhinny et al. 2005). Some of these indicators, as for example the amount of deadwood, have been used as a measure for management intensity (Müller et al. 2007b). In fact, structural indicators such as the number of trees with nesting holes and trees above a threshold minimum diameter etc. have been found to be closely related to the diversity of various taxa, that is wood-inhabiting fungi, saproxylic beetles, birds and lichens (Moning and Müller 2009; Moning et al. 2009; Müller and Büttler 2010; Müller et al. 2010). Why then suggest another index which may be used to investigate the effect of management intensity on biodiversity? We believe that many different stands could only be adequately contrasted by using a large set of structural measures. Thus, a very dense 10-year-old European beech thicket and a 100-year-old mature stand, both of which contain no dead wood due to the complete removal of the felled biomass, would need a different set of structural parameters than two stands of different species of equal age. However, to measure various attributes is laborious. In contrast, SMI is based on three stand characteristics which are easy to obtain: tree species, stand age and growing stock, which is heavily influenced by logging intensity. Interactions of the three factors are known to be of major importance for biodiversity issues (Bengtsson et al. 2000; Bagnaresi et al. 2002; Atlegrim and Sjöberg 2004; Ishii et al. 2004; Eriksson and Hammer 2006; Schroeder 2007; Wilson and Puettmann 2007; Rosenvald and Löhmus 2008; Smith et al. 2008; Michel and Winter 2009; Felton et al. 2010; Quine and Humphrey 2010).

Another approach for the quantification of forest land-use intensity, which is also based on easily measurable variables, was recently suggested by Luyssaert et al. (2011). In contrast to our approach, the index of these authors does not take into account differences between tree species and stand age. Moreover, it does not consider different site conditions, which SMI accounts for by using h_{dom} in Eqs. (8) and (9). Instead, Luyssaert et al.'s index (2011) is based on the two components: (1) the deviation of the actual stem number of a given stand from a reference stem number, which was derived from nine Slovakian old-growth forests and (2) the deviation of the mean quadratic diameter from the maximum mean diameter indicated by a self-thinning line which does not differentiate between species and site. However, the parameters a and b of the self-thinning equation ($\ln N = a - b \ln dg$), where N is the stem number of living trees and dg the quadratic mean DBH, are known to vary considerably for different site conditions (parameter a) and tree species (parameter b) (Pretzsch and Biber 2005).

It should be noted that SMI is not a measure of naturalness. Instead, it tries to quantify how intense a given stand has been managed so far, irrespective of the question whether or not the species in question was cultivated within our outside its natural range. This means that if biodiversity

data are analysed against SMI, covariates representing habitat properties or naturalness may also be taken into account. In addition, the spatial configuration and share of land-use units and other landscape elements may be additional, important components affecting biodiversity (Tschardt and Brandl 2004).

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