Chapter 8
Climate-Smart Silviculture in Mountain Regions

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Abstract Mountain forests in Europe have to face recently speeding-up phenomena related to climate change, reflected not only by the increases in the mean global temperature but also by frequent extreme events, that can cause a lot of various damages threatening forest stability. The crucial task of management is to adapt forests to environmental uncertainties using various strategies that should be undertaken to enhance forest resistance and resilience, as well as to maintain forest biodiversity and provision of ecosystem services at requested levels. Forests can play an important role in the mitigation of climate change. The stand features that increase forest climate smartness could be improved by applying appropriate silvicultural measures, which are powerful tools to modify forests. The chapter provides information on the importance of selected stand features in the face of climate change and silvicultural prescriptions on stand level focusing to achieve the required level of climate smartness. The selection of silvicultural prescriptions should be also supported by the application of simulation models. The sets of the various treatments and management alternatives should be an inherent part of adaptive forest management that is a leading approach in changing environmental conditions.

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8.1 Introduction

European forests in mountain regions are particularly vulnerable to the impact of climate change that could endanger the provision of ecosystem services. Hanewinkel et al. (2013) showed that the expected loss of value of European forest lands due to the decline of economically valuable species, in the absence of effective countermeasures, varies between 14% and 50% by 2100, depending on the interest rate and climate scenario applied. Adaptive forest management can address environmental uncertainties with strategies that enhance forest resistance and resilience, maintain forest biodiversity, and provide ecosystem services at requested levels. Various types of adaptation can be distinguished (Locatelli et al. 2010; Yousefpour et al. 2017; Lindner et al. 2020): (1) anticipatory or proactive adaptation, which takes place before the impacts of climate change are observed, (2) reactive adaptation,

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which takes place after impacts of climate change have been observed, and (3) autonomous or spontaneous adaptation that does not constitute a conscious response to climatic stimuli, but is triggered by ecological changes in natural systems and by market or welfare changes in human systems. The selection of the adaptation strategy should be based on a profound analysis of environmental and socioeconomic circumstances at a local and regional level and requires planning and implementation of forward-looking adaptation measures considering projected climate change (Lindner et al. 2020).

Climate-Smart Forestry (CSF) defined as "*sustainable adaptive forest management and governance to protect and enhance the potential of a forest to adapt to, and mitigate climate change*" (Bowditch et al. 2020) can be characterized by selected criteria and indicators originating from sustainable forest management (SFM) indicators (Santopuoli et al. 2021). The stand features that increase forest smartness could be improved by silvicultural measure (e.g., horizontal and vertical spatial structure, mixed species composition, deadwood amount, etc.). This chapter presents possible silvicultural measures for CSF with analysis via simulation models to evaluate their reliability.

8.2 Risks to Forests Induced by Climate Change

Mountain forests are considered to be particularly vulnerable to the effects of climate warming as temperature determines the upper limit of the altitudinal range for plant communities (Lenoir et al. 2008). Most studies conducted in mountain areas predict an upward shift of forest communities in response to temperature increases (Guisan et al. 1998). However, other factors (i.e., land-use changes, disturbances, biotic interactions) also modulate these responses (Martín-Alcón et al. 2010; Ameztegui and Coll 2013; Ameztegui et al. 2016), which can lead to unforeseen dynamics such as downslope displacements (see Bodin et al. 2013).

The increasing occurrence of extreme drought and heat events is at the origin of many declining forests and tree mortality episodes worldwide (Allen et al. 2010; Martínez-Vilalta et al. 2012; Margalef-Marrase et al. 2020) and mountain forests are not an exception (Galiano et al. 2010; Linares and Camarero 2012). Climate warming is predicted to intensify the disturbance regimes to which these systems are exposed (Seidl et al. 2017). For example, in Mediterranean mountains, the combined effect of fuel flammability increases and fuel accumulation associated to land-abandonment is expected to have a high impact on fire risk (Pausas 2004) compromising the local persistence of some populations that do not present adaptive mechanisms to such events (Vilà-Cabrera et al. 2012). In temperate and boreal areas, warming is also expected to intensify the frequency and severity of windstorms events (Seidl et al. 2014), insect outbreaks (Weed et al. 2013; Biedermann et al. 2019), and pathogen attacks (Sturrock et al. 2011) and to modulate the interactions among different disturbances (Temperli et al. 2013; Seidl and Rammer 2017).

The intensification of disturbance regimes is particularly important in mountain areas where the occurrence of these events (e.g., extensive bark beetle attacks, crown fires) was rare in the past. Recent catastrophic events, such as the *Vaia* storm (that caused in October 2018 damages of millions of cubic meters in northern Italy) or the unprecedented outbreaks of bark beetles in central Europe, point to the need of implementing effective monitoring strategies and designing managing regimes accounting for increasing risks.

Large-scale natural disturbances usually are followed by salvage logging: the main aim of it is to reduce economic losses. Besides, sanitary and aesthetic reasons are of some importance with this respect, too. On the other hand, the salvage logging practices indicate its strong impact on the functioning of the forest ecosystem, such as ecosystem restoration due to deterioration of the regenerative capacity of forests (Pons et al. 2020) and to threats to biodiversity conservation (Thorn et al. 2018). In order to maintain populations of the saproxylic species, Lonsdale et al. (2008) strongly suggest reducing salvage logging intensity in damaged tree-stands.

8.3 Indicators that Could Be Modified by Silvicultural Measures at Stand Level (Silvicultural Indicators)

Criteria and Indicators (C&I) of CSF originated from C&I of Sustainable Forest Management (Forest Europe 2015; Bowditch et al. 2020; Santopuoli et al. 2021) may refer to the stand, landscape, or even regional/national level. In this chapter, we are focusing on "silvicultural indicators" of CSF, which are manageable by silviculture measures at the stand level. Their evaluation is based on classification of indicators, presented by Bowditch et al. (2020) (Table 8.1).

8.4 Silvicultural Treatments Improving Stand Adaptation

8.4.1 Forest Area (Afforestation)

In the last decades, European mountains have undergone important forest expansion processes associated with the abandonment of traditional agrosilvopastoral activities (Kozak 2003; Gehrig-Fasel et al. 2007; Ameztegui et al. 2010). These processes include the encroachment of woody vegetation in areas previously occupied by cultures or pastures, and the densification of pre-existing forest stands. The rate of forest expansion is not homogeneous and depends on several factors operating and different spatiotemporal scales such as the browsing pressure (Coop and Givnish 2007), physiographic factors (Poyatos et al. 2003), or local socioeconomic conditions (Dirnböck et al. 2003; Ameztegui et al. 2016), among others. The ecological consequences of these processes differ. The progressive regression of abandoned land is leading to a homogenization of the landscape, and to the loss of

Table 8.1 Criteria and indicators (Bowditch et al. 2020) considered as those that can be shaped by silvicultural treatments within adaptation and mitigation strategy at the stand level

Strategy	Criteria	Indicator	Label	Description
Adaptation	Forest resources and global carbon cycles	Forest area	Forest area	Area of forest and other wooded lands, classified by forest type and by availability for wood supply, and share of forest and other wooded lands in total land area.
		Age structure and/or diameter distribution	Forest structure	Age structure and/or diameter distribution of forest and on other wooded lands, classified by availability for wood supply.
	Forest health and vitality	Soil condition	Soil condition	Chemical soil properties (pH, CEC, C/N, organic C, base saturation) in forest and on other wooded lands related to soil acidity and eutrophication, classified by main soil types.
		Forest damage	Forest damage	Forest and other wooded lands with damage, classified by primary damaging agent (abiotic, biotic, and human induced).
	Productive functions of forests	Increment and felling	Increment/ felling	A balance between net annual increment and annual felling of wood in forest available for wood supply.
	Forest biological diversity	Tree species composition	Diversity	Area of forest and other wooded lands, classified by the number of tree species occurring.
		Regeneration	Regeneration	Total forest area by stand origin and area of annual forest regeneration and expansion.
		Naturalness	Naturalness	Area of forest and other wooded lands by the class of naturalness (“undisturbed by man,” “seminatural,” or “plantations”).
		Introduced tree species	New species	Area of forest and other wooded lands dominated by introduced tree species.
		Deadwood	Deadwood	The volume of standing deadwood and of lying deadwood in forest and on other wooded lands.
		Genetic resources	Genetic resources	Area managed for conservation and utilization of forest tree genetic resources (*in situ* and *ex situ* genetic conservation) and area managed for seed production.
		Threatened forest species	Threatened species	Number of threatened forest species, classified according to IUCN Red List categories to the total number of forest species.
	Protective function (soil and water)	Protective forests – soil, water, and other ecosystem functions, and infrastructures	Protective forests	Area of forest and other wooded lands designated to prevent soil erosion, preserve water resources, maintain other protective functions, protect infrastructure and managed natural resources against natural hazards.
	New indicators	Slenderness coefficient	Slenderness	The ratio of total tree height to stem diameter outside bark at 1.3 m above ground level.
		Vertical distribution of tree crowns	Vertical crowns	Distribution of tree crowns in the vertical space. It can be measured in terms of layers (one, two, multiple), or in terms of the ratio between tree height and crown length.
		Horizontal distribution of tree crowns	Horizontal crowns	Canopy space-filling and can be expressed in measure of the density of tree crowns, such as crown area, tree crown diameter. It can be also expressed in measure of the density of trees, such as trees per hectare, basal area per hectare (in this case, the horizontal distribution refers to the tree).

Table 8.1 (continued)

Strategy	Criteria	Indicator	Label	Description
Mitigation	Forest resources and global carbon cycles	Growing stock	Growing stock	Growing stock in forest and on other wooded lands, classified by forest type and by availability for wood supply.
		Carbon stock	Carbon stock	Carbon stock and carbon stock changes in forest biomass, forest soils, and in harvested wood products.
	Productive functions of forests	Roundwood	Roundwood	Quantity and market value of roundwood.

mosaic-type structure, which is important for maintaining high biodiversity (Edwards 2005). The increase of stand density causes both higher fuel accumulations in the stands and higher competition for growing resources among the individuals, thus increasing the vulnerability of these systems to wildfires and drought (Nocentini and Coll 2013). Forest expansion in the upper parts of catchments can induce significant streamflow reductions in semiarid regions (Gallart and Llorens 2004).

Reforestation programs took place in mountain areas of many European countries during the twentieth century. The primary objective of these actions was to avoid soil degradation and regulate the hydrological conditions of watersheds (Mansourian et al. 2005). Conifer species were mainly used due to its pioneer character and ability to establish in difficult environmental conditions (Ceballos 1960). Unfortunately, management after afforestation was not adequately conducted and, at present, they show excessive densification, growth stagnation, and generalized poor health status (Pausas et al. 2004). The current management of these stands (some of which are rather aged) represents a big challenge for forest practitioners due to the location (often in inaccessible areas) and their primary protective role (Brang et al. 2006).

8.4.2 Structure of Forest Stands (Age and Diameter Distribution, Vertical and Horizontal Distribution of Tree Crowns)

Age structure, diameter distribution, and vertical and horizontal distribution of tree crowns are closely interrelated. Structural diversity in forests encompasses different age cohorts, size classes of trees and the spatial arrangement of different patches of tree groups, and structural elements, such as large living and dead trees, coarse woody debris or seed-producing tree clusters on a stand level. These stand legacies provide essential ecosystem processes (e.g., seed dispersal, nutrient translocation) and preserve genetic information in the phase of an ecosystem's recovery after disturbance. They are important elements in the reorganization loop of the adaptive cycle (Drever et al. 2006; Bauhus et al. 2009). Furthermore, stand legacies enhance faunal species richness, for example, as antagonist species, which reduce forest vulnerability.

The multiaged stands with structural diversity have the potential to increase both the resistance and resilience to various-scale forest disturbances (improve response diversity of a forest) (Elmqvist et al. 2003; Brang et al. 2013; O'Hara and Ramage 2013; Spathelf et al. 2018) and also productivity (Torresan et al. 2020). Such structural diversity in a forest can be achieved using several ways during stand management.

Thinning may become increasingly important for adaptation in many forest types, reducing stand density and increasing the individual stability and stress resistance of the remaining best crop trees in the stand (Misson et al. 2003;

Rodríguez-Calcerrada et al. 2011; Sohn et al. 2013; Spathelf et al. 2018). The application of the selected method and type of thinning must be compatible with the silvicultural objectives. Among the various thinning methods, there are those as selective (Schädelin), classical differentiation thinning, interfering with all layers of the stand, and free or variable density thinning, belonging to the crown or all-layer thinning type, that contribute to the increased structural diversity in the stand (Leibundgut 1982; Schütz 1987; Helms 1998; Schütz 2001b; Spiecker 2004; O'Hara et al. 2012; Silva et al. 2018). All types of thinnings, besides the improvement of timber quality, can help to create a diversity of age classes; decrease the water, nutrient, and light competition; increase individual tree resistance to biotic and abiotic factors; and, in some cases, encourage a wider range of species, which is a way of reducing and dispersing of silvicultural risk (Silva et al. 2018). Thinning, especially accomplished in medium-aged and/or older stands, may create conditions for the establishment of natural regeneration of the same or different species, thereby introducing new young age-classes of trees into the stand. Such vertically structured stands are more resilient after disturbance, since advanced regeneration is going to be quickly released (Brang et al. 2013).

Diameter and age structural diversity of forest stands is also associated with the occurrence and severity of natural disturbances; for example, a study from the Julian Alps showed that occurrence of windthrow disturbances in forest stands is negatively related to the volume of small-diameter trees (<30 cm in diameter at breast height), and positively with the volume of medium- (30–50 cm in diameter at breast height) and large diameter trees (>50 cm in diameter at breast height), while a large amount of small-diameter trees (<30 cm in diameter at breast height) increased the likelihood for snow breakage occurrence (Klopčič et al. 2009). The integration of various-scale disturbances into forest management could be the way of achieving a multiaged, multilayer, and multispecies forests that can fulfil multiple purposes. A wide range of measures to promote uneven-aged stands and structural diversity including emulation of disturbances, planning salvage operations, and variable treatment intervals or intensities is presented by O'Hara and Ramage (2013). However, many of these measures can be applied on forest (a group of stands) or landscape scale leading to high structural diversity that reduces the probability of stand-replacing disturbances.

Forest stability, vitality, and resilience can be also enhanced through silvicultural activities making the best use of natural structures and processes. This includes proactive steering of natural successions instead of passive waiting for natural processes to occur (Silva et al. 2018; Lindner et al. 2020). These processes can supplement the structural diversity in terms of species composition, and vertical and horizontal stand diversification.

The application of some silvicultural systems is one of the possible measures leading to the formation of structurally diversified forests. At present, slightly less than 70% of forests in Europe are reported as even-aged, whereas uneven-aged forests appear to be the main forest type in South-West Europe (Forest Europe 2015). But this does not mean that all even-aged stands should be converted. The long-term process of transition from even-aged to uneven-aged stand could be performed only in those places where site, stand, and socioeconomic conditions allow its realization and where it is advisable. Several methods to achieve uneven-aged, structurally diversified stands composed of shade-tolerant tree species exist including silvicultural systems (irregular shelterwood system and its variations, selection system), thinning (selection with intervention in all stand layers), and other methods combining different felling schedule with various methods of natural and artificial regeneration (Schütz 2001b; Nyland 2003; Pretzsch 2019; Hilmers et al. 2020). The transformation from regular to irregular stands can be accomplished in the present stand with the sequence of differentiation thinning or in the following stand generation depending on the stability and irregularity of a stand (Schütz 2001b). The implementation of the methods depends on the silvicultural objectives, species composition and stability of existing stand, and site conditions. Irregular shelterwood system and its many variations are characterized by the greatest potential and versatility to shape uneven-aged forests composed of various tree species of functional traits (Puettmann et al. 2009; Raymond et al. 2009; D'Amato et al. 2011; Lussier and Meek 2014; Raymond and Bédard 2017).

8.4.3 Soil Condition

Forest cover is strongly influenced by soil productivity, which is partially governed by climate, but more significantly by bedrock composition and erosion rate (Hahm et al. 2014; Milodowski et al. 2015; Wolf et al. 2016). Forest soil productivity is crucial for sustainable forest management and is a function of soil potential properties and soil conditions. Soil potential properties are the ones that are not easily altered such as soil depth, stoniness, the content of organic matter, texture, porosity, clay mineralogy, while soil conditions can be altered more easily and are represented by soil thickness, porosity, and soil organic matter (Poff 1996). Soil depth, as a basic criterion of soil classifications, represents the depth from the surface of the soil to the parent material. Soil porosity is a combined volume of solids and pores filled with air and/or water. The size and interconnection of pores determine water infiltration and retention, gas exchange, biological activity, and rootable soil volume, thus representing an extremely important link in soil productivity.

Forest soils hold a substantial portion of terrestrial carbon and any alterations in carbon cycling are significant for forest productivity and ecosystem services (James and Harrison 2016). Change in quality or quantity of soil organic matter caused by climate change is probably one of the most important factors affecting forest soils (Raison and Khanna 2011), since soil organic matter, together with nitrogen and phosphorous, is one of the principal components of soils and has a crucial role in several biological, chemical, and physical properties (James and Harrison 2016). At large scale, the variability of soil organic carbon is mostly governed by climate, while on a local scale, it depends on forest management practices, type of bedrock, soil properties, and topography (Conforti et al. 2016).

Bedrock has a significant role in vegetation growth by regulating physical and chemical properties of soils (Hamh et al. 2014) and has a substantial influence on soil erosion processes (Milodowski et al. 2015). It is the source of mineral nutrients and influences soil texture characteristics controlling the water and nutrient retention capacity, but can also present a supply of heavy metals that have an adverse influence on plant growth (Jiang et al. 2020). Soil texture in forest soils determines soil water and aeration, both important for tree growth and microbial processes (Gomez-Guarrerro and Doane 2018). Soil degradation includes higher bulk density and lower hydraulic conductivity and extensive nutrient losses in soils (Hajabbasi et al. 1997). Loss of porosity leads to infiltration reduction, loss of soil volume, and enhances soil erosion. However, these effects might be happening at the same time and causal (Poff 1996). Similar is true for textural properties. Soils with high silt, low clay, and low organic matter content are generally considered to be more erodible. However, this is not straightforward and particle size distribution has to be considered in relation to other soil physical and chemical properties (Wischmeier and Mannering 1969).

Altieri et al. (2018) have experimentally shown that soil erosion is not a substantial problem in well-managed forests and minimal values of soil loss were reported in areas with high canopy cover and biomass. However, some authors indicate that new silvicultural treatments should be planned with care, since, as established by earlier studies, loss of forest cover, either due to deforestation or climate change, can impose a serious problem with long-term consequences. If the topsoil layer is disturbed due to natural or human-induced causes, such as wildfire, harvesting, and prescription burning, erosion rates can substantially rise. Relationship between soil disturbance and soil productivity is a complex interconnection among soil physical properties, nutrient cycling, and climate. The disturbance effect depends on soil local characteristics and microclimate, so mitigation solutions have to be site-specific (Elliot et al. 1996).

8.4.4 Forest Damages

Forest disturbances are, in many cases, inseparably related to climate change (Dale et al. 2000, 2001; Reyer et al. 2017; Seidl et al. 2017). Disturbances, human-induced and naturally caused mostly by wind, insects, fungi, fires, droughts, heatwaves, and their interactions, shape the forest ecosystems in terms of species composition, structure, and processes. Proactive disturbance-risk management should encompass adaptive silvicultural measures, being a part of the adaptive forest management, which enable using some strategies to counteract the effects of climate change resulting in forest disturbances. These possible actions should be undertaken considering uncertainties about climate change impacts on forests and their reactions (Lindner et al. 2014). The potential silvicultural disturbance-prevention measures include (1) the use of more climate-adapted tree species or their genotypes (Kauppi et al. 2018; Thurm et al. 2018), the introduction of economic alternatives to main species (Deuffic et al. 2020), management to facilitate the establishment of species outside of historical natural ranges and genomics-based assisted migration in reforestation (transformation) (Hagerman and Pelai 2018); (2) application of more diversified species composition of forests (mixtures of conifers with broadleaves, shade-tolerant with intolerant species, more drought-resistant with less-resistant species) involving also conversion from single-species to multispecies stands where site conditions permit (Kerr et al. 2010; Jactel et al. 2017), this allows to distribute silvicultural risk to many tree species in a stand; (3) managing for and/or increasing resilience (Hagerman and Pelai 2018); (4) more frequent and intensive thinnings (selective or differentiation) and shorter and/or diversified rotation length (Jactel et al. 2009; Silva et al. 2018; Deuffic et al. 2020); (5) shaping the diversified age structure of forests (uneven-aged/selection structure) (Schütz 2002; O'Hara 2014; Deuffic et al. 2020); (6) increasing stand stability and decreasing stand density (Knoke et al. 2008; Deuffic et al. 2020); (7) fire-smart landscape management techniques (Kauppi et al. 2018; Lindner et al. 2020). In addition, the realization of the concepts of close-to-nature silviculture (Schütz 1999; Brang et al. 2014; O'Hara 2014) and/or continuous cover forestry (Mason et al. 1999; Pukkala and Gadov 2012) seems to enhance adaptation to climate change and, to some extent, mitigate its effects on forests (Fig. 8.1).

Fig. 8.1 Forest damages caused by the windstorm Xynthia (2010) in la Val d'Aran (NE, Spain). (Photo: Álvaro Aunós)

8.4.5 Increment and Felling

In the case of close-to-nature mountain mixed and uneven-aged forests, comprised of species combination such as silver fir (*Abies alba* Mill.), Norway spruce (*Picea abies* (L.) Karst.) and European beech (*Fagus sylvatica* L.), all silvicultural operations attempt to achieve growth sustainability from one cutting cycle to the next and continuous forest cover for preventing soil erosion. To this end, the single and/or group selection (plenter) system can be used in mountain regions across Europe (Schütz 2001a). In the selection forest, the mixture of trees of different sizes (diameter at breast height and height), ages, and species, growing together in a small area (<0.1 ha) (Schütz 2001a; Bončina 2011a; O'Hara 2014), results in much more steady course of growth, in comparison to one species dominated even-aged stands, at both tree and stand level (Fig. 8.2). The higher resilience of stand growth to silviculturally induced density reductions in mixed, uneven-aged mountain forests can be observed as in this case, trees in the medium and lower canopy layers can compensate (buffer) for losses in the upper layer and vice versa (Mitscherlich 1952; Assmann 1970).

In the structural stable mountain forests (Fig. 8.3), the equilibrium state is achieved when standing volume remains relatively constant from one cutting cycle to the next; in the other words, the harvest volume equals increment. Therefore, the value of periodic volume increment may serve as an additional important parameter to consider, when regulating the long-term development of mixed, uneven-aged

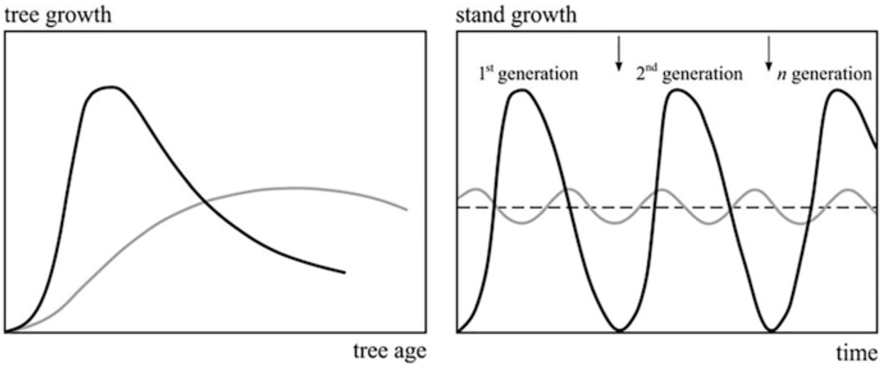

Fig. 8.2 Tree level and stand level growth pattern in two contrasting silvicultural systems: simple even-aged forests (black line) and complex uneven-aged forests (grey line), managed by a clearcutting and selection system, respectively. *In the first case, the growth* (e.g., *tree diameter left and stand volume increment right) follows the unimodal curve and changes more rapidly (up and down) over the time with a clear pick during the optimal developmental phase, to decreases, however, to zero at the time of the initiation of subsequent forest generation by clearcuttings or shelterwood cuttings with short regeneration period (10–20 years). In the selection forest, the combination of trees of many sizes, age classes, and species buffers the changes in the growth pattern and thus a steadier increment course can be observed.* (Adapted from Schütz (2001a) and Pretzsch et al. (2015) (after Assmann 1970) in case of a tree and stand level, respectively)

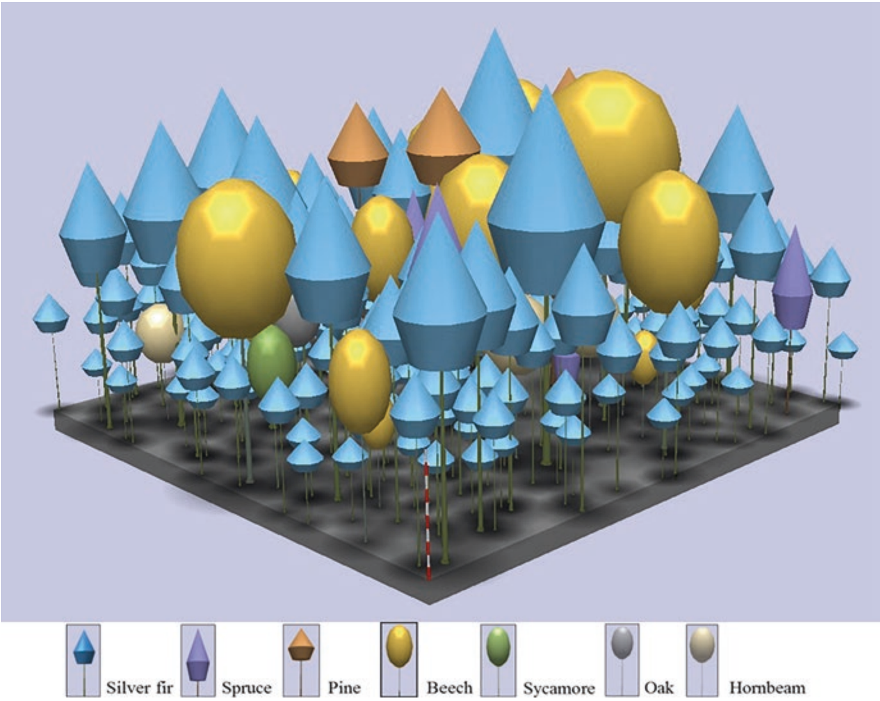

Fig. 8.3 The vertical and horizontal structural stable close-to-nature mountain mixed and uneven-aged stand, comprised of silver fir (Abies alba Mill.), European beech (Fagus sylvatica L.) and Norway spruce (Picea abies (L.) Karst.), as well as other minor tree species, managed by selection (plenter) system on the long-term experimental plot in the Zagnansk Forest District (Poland)

mountain forests. Schütz (2001a) described the plenter structure as being maintained through continuous control of the growing stock (standing volume); a growing stock over the equilibrium would lead to reduced regeneration and recruitment into smaller diameter classes, whereas levels of growing stock below the equilibrium would reduce total increment and the quality of trees as well as to overpromote the natural regeneration and recruitment rate. However, in some cases such as the current stand diameter structure deviates significantly from the equilibrium curve (e.g., when stands previously were managed by uniform or irregular shelterwood cuttings), the transformation by applying heavy structural differentiation thinning is recommended, to reduce mainly the density in the middle diameter at breast height classes and, therefore, as a consequence, also the stand productivity.

Finally, if the proportion of more light-demanding tree species in the stand is required, the irregular shelterwood system, emulating the natural gap dynamic pattern, would be also recommended in the scope of climate-smart silviculture. The main difference between selection and irregular shelterwood systems lies in an emphasis on individual trees in the former case versu. cohorts of trees in the latter (Schütz et al. 2016). Moreover, the irregular shelterwood system gives a free hand to the manager. On the one hand, it is possible to create larger openings by clearcutting (e.g., up to 0.5 ha) when regeneration of more light-demanding tree species (larch, pine, oak) is required, and on the other hand, the shelter or group and single selection cuttings may be applied for intermediate (sycamore, spruce, elm) and more shade-tolerant tree species (lime and beech) as well (cf. Schütz 2001a; Raymond et al. 2009). During the long regeneration period (e.g., 40–50 years), the volume increment of regeneration and overstorey add up. Thus, compared with an age-class forest, the total volume increment's oscillations are much less distinct (cf. Fig. 8.2). The more multilayered a stand and the more horizontally diverse it is, the higher is its growth resilience to natural and silvicultural interferences (Pretzsch et al. 2015).

8.4.6 Tree Species Composition

In production forests, the diversity of tree species composition positively affects other indices of biodiversity and shows close relationships with multiple ecosystem services (e.g., Gamfeldt et al. 2013; Bravo-Oviedo et al. 2014; Almeida et al. 2018). However, in the case of provisioning services, the forest development stage is also of great importance (Zeller and Pretzsch 2019).

None of the single tree species in a forest is able to safeguard a provision of the full set of ecosystem services. On the other hand, the provision of some services can be impossible, since they might be negatively correlated with each other. Therefore, in order of satisfying the society demands regarding multiple ecosystem services the production forests should be managed considering use of the various tree species. No doubt that tree species diversity positively influences ecosystem functioning, but in some cases, probably the effect of species identity is stronger compared with diversity (Nadrowski et al. 2010). If the dominating tree species is badly chosen, then changing it back to the former one might reverse the negative outcomes for biodiversity, production, and recreational values, as well as on stand vulnerability to a wind, frost, and drought damage, as well as on risks of pathogen or insect outbreak (Felton et al. 2019).

Tree species diversity of temperate mountain forests is much lower if compared to the tree species diversity in forests of lower vegetation belts (i.e., planar-hilly, sub-montane). Therefore, management options regarding tree species are much broader in lower areas. For the adaptation of mountain forests to climate change, it is highly important (1) to maintain/increase genetic variation in the species, (2) to increase structural diversity (Brang et al. 2014), and (3) to assure that all "natural" tree species of mountain forests are present in a forest stand. However, quite often, European mountain forests contain mainly three tree species – Norway spruce, silver fir, and European beech. Especially, recruitment of silver fir into these forests is often restricted or even prevented due to browsing pressure (e.g., Ficko et al. 2011; Simončič et al. 2019), which noticeably decreases adaptation capacity of forests to climate change. The additional characteristics if compared to the forests in lower elevations are that in mountain forests, many minor tree species (e.g., *Sorbus aucuparia*, *Salix* sp.) cannot compete or are economically less important for forest managers/owners, while some other with possible high economic value (i.e., *Ulmus glabra*, *Fraxinus excelsior*) suffer from diseases. Therefore, tree species from genus *Acer*, *Larix*, and *Pinus* gain higher importance for the adaptation of mountain forests to climate change.

There is increasing evidence that tree species mixtures positively influence forest functionality. Forest stands with tree species with different functional traits enhance forest fitness in the face of climate change as they include different "strategies" concerning plant establishment and competitiveness (Jactel et al. 2017). Moreover, in many cases, species-rich forests with high functional diversity are more productive than less diverse forests (Pretzsch et al. 2010). In stands where light demanding and shade-tolerant, canopy and understory or deep-rooting, and shallow-rooting species are combined, resources such as light, water, and nutrients can be spatially and temporarily used differently and thus more efficiently. Such forests are more resistant to various abiotic disturbance events, such as drought, fires, or storms (Bravo-Oviedo et al. 2014; Knoke et al. 2008; Schütz et al. 2006; Spellmann et al. 2011; Lebourgeois et al. 2013), and more resilient once a disturbance has occurred (Jactel et al. 2009, 2017). With an increasing number of functionally different species, the probability increases that some of these species can resist external disturbances or changing environmental conditions (i.e., the ecological insurance concept, according to Yachi and Loreau 1999). Examples are the bark beetle *Ips typographus* that attacks Norway spruce (*Picea abies*), but not broad-leaved species or silver fir (*Abies alba*) (Wermelinger 2004), or the ash dieback (*Hymenoscyphus pseudoalbidus*) affecting exclusively *Fraxinus excelsior* (Kjær et al. 2012). In addition to functional diversity, the redundancy of species increases the probability that one species can take over the role of another species that does not survive (Walker et al. 1999; Messier et al. 2019).

It is assumed that in the future also mountain forest ecosystems are severely affected by water shortage (Collin 2020). The admixture of broadleaved tree species in conifer stands can have positive effects on soil water availability, thus reducing water stress for the trees. There is evidence that interception losses are higher in pure conifer stands with Scots pine and Norway spruce compared to broadleaved or mixed stands with European beech (Barbier et al. 2009; Berger et al. 2009). In a study in northeastern Germany, Müller (2009) analyzed seepage rates in mixtures of Scots pine with European beech compared to pure Scots pine stands. The higher seepage in mixed stands is due to reduced interception losses and a higher stemflow on broadleaved trees compared to pine. Moreover, in pure (pine) stands, dense ground vegetation layers of the grass *Calamagrostis* lead to a further reduction of the soil water content (Müller and Bolte 2009).

Other complementarity effects with respect to water supply of species mixtures are reported, such as hydraulic redistribution or the different stomatal behavior of the trees. Thus, water availability in mixed stands can be positively influenced, although many effects are observed at the drier end of the gradients and are not yet quantified (Grossiord et al. 2014; Bauhus et al. 2017).

8.4.7 Regeneration

One of the most important practices to increase species richness is the choice of regeneration cut or the silvicultural system, respectively. Here, the future species composition of the forest can actively be changed by replacing tree species and/or tree individuals sensitive to climate change with trees of native or introduced species and/or species' provenances that are potentially better adapted to future climate conditions (called active adaptation; Martín-Alcón et al. 2016; Bolte et al. 2009). Examples for this strategy are the ongoing conversion of pure Norway spruce stands into mixed stands or silvicultural measures aiming at replacing species such as Norway spruce by other species of comparable economic value (e.g., Douglas fir). In Germany, the Bavarian State Department of Environment, Health and Consumer Protection published a regional climate program in November 2007 that includes an example for the application of the "active adaptation" concepts on species level. It is planned to convert about 200,000 ha of pure Norway spruce forests by 2030 in areas where a high risk of drought damage is assumed to less sensitive mixed forests, predominantly with European beech and oak (Stmelf 2018).

The concomitant natural establishment of diverse species can be controlled by creating large variations in light conditions, allowing both light-demanding and shade-tolerant species to regenerate (e.g., group selection or irregular shelterwood in combination with strip cuts). In young growth originating from natural regeneration, enrichment planting is a valuable practice to introduce additional species. Once young trees are established, species richness can be maintained by appropriate tending measures, such as precommercial thinning or thinning. Especially rare species or species with low competitiveness, in particular, if they are adapted to a warmer and drier climate, have to be released in this case (Brang et al. 2008). Finally, the successful establishment of species-rich stands depends very much on the control of ungulates (Gill 1992; Götmark et al. 2005; Ameztegui and Coll 2015). To achieve the optimal adaptive effect of species mixtures, large monospecific patches should be avoided as well as very intimate mixtures, which usually require high tending investments. Forest conversion encompasses the use of the many native (mostly broadleaved) tree species, and selected exotic tree species, respectively. From tree species trials and recent dendroecological analyses, we know that rare native species and non-native tree species can increase forest resilience in a landscape (Kunz et al. 2018; Vitali et al. 2017). Furthermore, provenances of native tree species from warmer regions of the species' distribution range could enrich forest diversity in mountainous regions. Especially these rear-edge populations (Hampe and Petit 2005) of native species often show desired adaptation traits, such as higher drought stress tolerance compared to provenances from the core distribution area of a species.

In the early stage of conversion, an interesting option to diversify forest stands is currently discussed in Germany. Some authors have recommended integrating early successional species, which seem to be more adapted to the drier site conditions into regular stand management in the tree species portfolio (Lüpke 2009). Early successional species quickly cover bare regeneration sites. They recover nutrients, which otherwise would likely be lost and are valuable elements for enhancing biodiversity (nurse crops). Moreover, there is growing evidence to abandon the practice of a merely local provenance choice. Assisted migration, the planned translocation of provenances and species beyond their natural occurrence range, has the potential to ensure that provenances or species are adapted enough to cope with the future warmer climate in the final stage of their development cycle. At the same time, stands must be robust enough to get along with still harsh climatic conditions in the establishment phase when they are young plants. Already well-developed recommendations for assisted migration transfer distances have been developed for Douglas fir in the Northwest USA (Sáenz-Romero et al. 2016).

8.4.8 Naturalness

The concept of naturalness has been broadly used in forestry. The naturalness of forest stands can be assessed by different indicators (e.g., Brumelis et al. 2011; Winter 2012); tree species composition is one of the most important. The naturalness of tree species composition for a given forest site is estimated by comparison between current tree species composition and the natural tree species composition of forest stand, which is a part of potential natural vegetation. Due to climate change, natural vegetation may change over several decades (e.g., Hickler et al. 2012).

The analyses of forest stands in the Dinaric mountain areas (Bončina et al. 2017) showed that the alteration of the natural tree species composition of forest stands is primarily the result of forest management and past land-use, conditioned either by topography or accessibility of forests. The portion of Norway spruce increased due to past forest management. A higher level of alteration of natural tree species composition of mountain forests significantly increases the susceptibility of forest stands to natural disturbances – mainly windthrows and insect outbreaks. Therefore, sanitary felling can be a few times larger than in stands with natural tree species composition (e.g., Pasztor et al. 2015; Bončina et al. 2017).

In general, for the introduction of new, exotic tree species and provenances, it is suggested to follow the order: (1) species that are already adapted on a larger scale in the planting region and tested non-autochthonous provenances, then (2) new species with knowledge on their behavior but no adaptation yet, and finally (3) completely new species (Spathelf and Bolte 2020). Currently, only a few forest owners have started to plant nonnative tree species other than Douglas fir, red oak, and grand fir on a larger scale. Nevertheless, around 10–20 “new” species are in the search of forest research institutes across Europe. Existing trials with non-native species are currently evaluated and new trials established (de Avila and Albrecht 2017; Brang et al. 2016; Metzger et al. 2012).

8.4.9 Introduced Tree Species

Tree species have been deliberately moved by humans for as long as humans have been cultivating land for food, fuel, and fiber. Indeed, it is highly probable that tree species were inadvertently moved by our hunter-gatherer ancestors, just as other primates do today (Chapman 1989). Long before the development of countries with borders and associated concepts, such as nativeness, immigration, introduction, or invasiveness, humans travelled and traded and the only considerations were what thrived where, and what value it had as a product. For trees, the main considerations would have been fruit and nut production, foliage and bark for animal fodder, firewood potential, and use as a building material.

As concepts of forest management developed, the choice of tree species for timber production has become increasingly sophisticated. It may even be that tree species selection was the first conscious silvicultural decision? Originally it is probable that some native tree species were preferred by foresters; for example, oak has long been advocated in Britain for ship-building (Evelyn 1664; Fisher 1763). Where one tree species is favored, others inevitably decline in abundance. In the past, when there was no concept of genetic diversity or origin, although native tree species might have been selected for planting, the seed itself might have been introduced from another country.

It is difficult to trace the widespread use of nonnative (also known as exotic or introduced) species in plantation forestry, but it is always likely to have been most prevalent in countries like the UK with low native tree species richness. For example, at least one introduced tree species, European larch (*Larix decidua*), has been being planted in the UK since the mid eighteenth century when medals and cash prizes were awarded to those who planted most trees by the Royal Society for the Encouragement of the Arts. In their first full transactions published in 1783, the summary of the activities since the inauguration in 1754 showed that a sum of 50 £ had been paid alongside the award of 45 gold and 14 silver medals for the "encouragement of planting to raise Timber" from a list of trees, including oak, but also larch (Anon 1783). Larch would have been included on the list, because softwood timber was considered best for ship's masts and the UK has only three native conifers, juniper (*Juniperus communis*), yew (*Taxus baccata*), and Scots pine (*Pinus sylvestris*), of which only one, Scots pine, can grow sufficiently straight and tall to be used as a ship's mast.

In the early nineteenth century, the great plant hunters, such as David Douglas, sent new coniferous tree species back to Britain, from the Pacific North-West, where the climate is similar to the Atlantic North-West of Europe. It was soon noted how fast and straight these species, particularly Sitka spruce (*Picea sitchensis*) and, of course, Douglas fir (*Pseudotsuga menziesii*) (Savill 1991) grew. Consequently, they were soon widely planted, not only in the UK but on suitable sites throughout north-western Europe. Sitka spruce now comprises more than a quarter of all forest trees in the UK (Forestry Statistics 2019).

In recent years, the recognition of the role of trees, woods, and forests in combating climate breakdown has led to an appreciation that introduced tree species may have a role to play in climate mitigation. Specifically, if introduced trees grow faster than native species, they are considered to sequester carbon faster in the forest via net photosynthesis. This means that they have faster biomass accumulation to provide a carbon substitution benefit from the forest sooner (as wood fuel, or by replacing building materials, such as concrete and steel that have higher carbon footprints). Whether faster-growing trees or more frequent harvests of biomass provide better climate mitigation as timber density, not just volume needs to be quantified, carbon transfer via roots into the mycorrhizae and soil needs to be measured and the effects of more frequent harvests on soil disturbance need to be included.

The concept of planting introduced tree species as a silvicultural treatment for improving forest stand adaptation to climate breakdown is novel. It has been recognized that long-lived, slow to reproduce, heavy seeded plants, including many native tree species, are unable to rapidly adapt to climate change by moving or adapting, so increased tree mortality and associated forest dieback is projected to occur in many regions over the twenty-first century (Field et al. 2014). Consequently, although the native tree species in a given country may be adapted to survive the current pests, diseases, and abiotic threats they face, they may not be resistant and resilient to future threats. As tree species in certain locations may be adapted to climatic conditions that are similar to the ones predicted to be faced in others, it can be argued that to maintain a forest structure, for commercial timber production and other ecosystem services, but also as a habitat/ecosystem for biodiversity, the introduction of tree species likely to thrive in the future climate is justified (Forestry Commission 2020).

The novel argument for the introduced tree species, as a silvicultural treatment to help forests adapt to climate breakdown and thus maintain the delivery of ecosystem services, including climate mitigation, is controversial. However, a CSF approach means putting climate adaptation and mitigation first among multiple sustainable forest management objectives and all options need to be considered (Bowditch et al. 2020). Research into the effectiveness of this approach is needed and indicators need to be developed to guide if, how, and where this is viable. For example, in commercial plantation forestry with introduced tree species, it is not a great issue to introduce others. However, in the current plantations of native species, it may need more careful consideration. In our most pristine native woodlands, introduced tree species may be viewed as too damaging to their integrity.

8.4.10 Deadwood

In forest ecosystems, deadwood influences the nutrient and water cycling, humus formation, carbon storage, fire frequency, natural regeneration and represents a crucial component of forest ecosystems for maintaining and improving forest biodiversity. Decaying wood, such as logs, snags and stumps, as well as rot holes, dead limbs and roots, heart rot and hollowing in living ancient or veteran trees, all of them are habitats for the specific species of fungi, flora, and fauna (Humphrey et al.

2004). Thus, the deadwood volume narrowly meant as the coarse woody debris (CWD), that is, logs and snags, has been selected as the main Pan-European SFM indicator regarding biodiversity, and it is also one of 15 main indicators of biodiversity as proposed by European Environmental Agency (Humphrey et al. 2004; Merganičová et al. 2012)., However, the ancient and veteran trees in all forests also are of key importance for rare and threatened saproxylic species, but, unfortunately, they arenot used for biodiversity monitoring (Humphrey et al. 2004).

Deadwood constitutes habitats for many species of cryptogams, such as bryophytes, lichens, and fungi (Humphrey et al. 2002; Lonsdale et al. 2008; Stokland and Larson 2011; Persiani et al. 2015; Preikša et al. 2015), invertebrates like saproxylic beetles (Martikainen et al. 2000; Franc 2007; Müller et al. 2008; Lassauce et al. 2011), as well as amphibians, birds, and mammals (Merganičová et al. 2012).

Wood-decaying fungi are essential for the functioning of forest ecosystems. They provide habitat for many other deadwood-dependent organisms and enable the regeneration of forests. There are plenty of examples of enhanced survival of seedlings of various forest tree species (mainly conifers) occurring on decaying deadwood (Lonsdale et al. 2008). To support the decaying fungi species of varying requirements, a wide range of CWD of different sizes and stages of decay is necessary (Lonsdale et al. 2008).

All types of deadwood are a substrate for the development of rare cryptogam species. The intermediate decay stages are extremely important for fungi, while bryophytes or lichens do not show such a clear preference. The highest number of cryptogam species is found on the deadwood of Common ash, English oak, and Norway spruce, while deadwood of other tree species hosts less than half cryptogam species (Preikša et al. 2015).

Throughout Europe, saproxylic beetle species have been identified as the most threatened community of invertebrates (Davies et al. 2008). Species richness in saproxylic beetles has a significant positive correlation with the main deadwood variables (Martikainen et al. 2000). It depends not only on deadwood amount, but also on other microhabitat factors, such as the richness of wood-inhabiting fungi, and, for the threatened saproxylic beetles – on the frequency of *Fomes fomentarius* (Müller et al. 2008).

Natural variation of deadwood niches – including decay stages, snag sizes, tree cavities, and wood-decaying fungi species – must be maintained to efficiently preserve the whole saproxylic beetle fauna. To better assess the quantitative relationships between deadwood and biodiversity of saproxylic beetles, apart from the deadwood volume, deadwood type or decay stage should also be considered (Lassauce et al. 2011).

Pieces of evidence have shown that climate change will speed up tree growth and accumulation ending up in a higher stock of deadwood available *in situ* (Mazziotta et al. 2014). However, due to increased decomposition rates, the time the deadwood stock is available for deadwood-associated species will diminish and the carbon stored in deadwood will return to the carbon cycle faster (Büntgen et al. 2019). Disturbances from fire, insects, and pathogens, in particular, are likely to increase in a warming world (Seidl et al. 2017), which could markedly modify the distribution of deadwood across the forested landscapes in time and space. Under such circumstances, it is going to become increasingly challengeable to manage deadwood in a sustainable way. Some authors recommend that the structure and dynamics of old-growth forests are used as a reference system for managed forests (Jandl et al. 2019). Based on modelling results, it was found that continuous cover forestry, based on emulating natural disturbances and leaving 10% of stands uncut with no deadwood extraction, will result in greater dendrobiotic birds habitat quality per unit of current volume increment under climate change (ARANGE 2020). However, it is not clear whether the carbon sink function will decrease or even stop when the forests get into a steady-state of carbon sequestration in biomass and soil organic matter and of carbon loss due to decomposition of deadwood debris and soil organic matter (Desai et al. 2005; Pukkala 2017). In all cases, forest owners should be flexible and prepared to diversify the silvicultural systems across forested landscapes. They will need to follow natural disturbances in a way that will guarantee the presence of enough deadwood to adequately address the various trade-offs between wood biomass production for carbon sequestration, on the one hand, and forest protective functions, on the other.

Forest management should mimic the natural stand dynamics, increasing the number of dead trees and the diversification of the vertical and horizontal tree layers, considering the good potential for restoring and increasing the diversity of saproxylic communities and their associated ecological functions. For monitoring the ecological sustainability of forest management, we must focus on threatened species (Müller et al. 2008).

For strategies to increase deadwood amount in managed forests, the best results will be achieved in areas close to existing reserves or other important habitats (Müller et al. 2008). Research into deadwood dynamics carried out in unmanaged forest ecosystems (Christensen et al. 2005; Persiani et al. 2015) has proved useful as a reference tool to implement rehabilitation criteria in sustainable management, to maintain and increase biodiversity and other ecosystem services provided by managed forests.

Sanitary cuttings, carried out mainly to avoid outbreaks of insect pest populations or to reduce risk of forest fires, are another measures leading to severe restriction of the capacity of managed forest ecosystems to provide habitats for saproxylic species (Humphrey et al. 2004). Since healthy, resistant, and resilient managed forest should, partly, consist of diseased or injured trees (Szwagrzyk 2020), their retention until natural death would allow accumulation of deadwood of various types and sizes, and representing all tree species that grow in a forest, that would create niches for all deadwood depending species.

To improve the status of the deadwood-depending organisms, the managed forests should maintain a long-term continuous provision of greater amounts of dead and decaying wood microhabitats that deadwood-depending organisms require for their survival (Christensen et al. 2005; Davies et al. 2008; Lonsdale et al. 2008). However, no simple deadwood stocking recommendations can be applied, due to the inherent complexity of all the stand, site, and management factors that drive deadwood dynamics (Persiani et al. 2015).

8.4.11 Genetic Resources

Since the advent of the population genetic studies based on molecular markers, it has been postulated that long-term survival and adaptation of species and populations to the changing environment strongly depends on the high genetic variation accumulated in the gene stock of populations over historical times (van Dam 2002). Natural forest tree populations tend to maintain a high level of genetic diversity along with the distribution range because of the high outcrossing rate, the long life span of individuals that besides preserving their highly heterogeneous genomes can fix beneficial alleles for a longer period (Petit and Hampe 2006). Several acting forces on population-level, however, can shape the uniform distribution of the genetic variation especially at the range periphery of the species where gene flow usually decreases or the environment reaches the tolerance limit of populations. Genetic drift and inbreeding acting at the range margins can cause differentiation by changing the frequency distribution of alleles and selecting population-specific alleles (Hampe and Petit 2005). These selected alleles might be beneficial in the local adaptation on the range margins, but can be also harmful forcing populations to counteract against their fixation (balancing selection). Differences between central versus peripheral populations and the role of the beneficial alleles helping population adaptation have been much discussed in different studies (Gibson et al. 2009; Logana et al. 2019).

Natural or human-induced fragmentation in species' distribution area can increase the effect of marginality and can cause isolation within the species' range, not only at the range periphery. Fragmentation increases genetic divergence and will promote the overall structuring of populations. If gene flow becomes limited among the fragmented sites, the long-lasting drift and inbreeding end up in pauperization of the gene stock causing a higher rate of homozygous individuals. Homozygote excess usually produces limited resilience and lower fitness, impeding population adaptation to the changing environment (Mátyás 2002; Allendorf et al. 2013). All these processes are strongly affecting populations in the time of the ongoing climate change that has an unpredictable impact on the structure of the ecosystems and populations therein.

Forest trees having a large genome and preserving a considerable amount of genetic variation are expected to have high resilience. Moreover, the high phenotypic plasticity allows them to withstand even large environmental fluctuations during their lifetime. However, researchers have expressed their concerns also. Tree species with long generation time due to the long-life span of individuals might be unable to react to the fast changes experienced due to the ongoing climate change. These events are too rapid relative to the tree's age and populations may not have adequate time to adapt or to disperse and colonize the newly available habitats. As many species are unlikely to migrate fast enough to track the rapidly changing climate in the future, their standing adaptive variation will likely play an increasingly important role in their response (Jump and Peñuelas 2005; Mátyás and Kramer 2016).

Studies have shown that the effect of climate change acts strongly at the species' ecological limits, as tolerance to climatic extremes is genetically determined. For example, European beech populations at the lower xeric limit of the species' distribution are more exposed to the impacts of climate change; hence, the decline of population in these territories has been anticipated (Mátyás and Kramer 2016). In turn, more recent studies have shown that beech, currently dominating lower elevations in mountain sites, has a high potential to advance to higher altitudes, where it can perform better in mixed stands than in monospecific stands (Pretzsch et al. 2020c).

Since the first utilization of forests ecosystems, humans intentionally or unconsciously have altered the gene pool of forest tree species (Buiteveld et al. 2007). The decrease in forest area size, habitat degradation, change in species composition, forest plantations, or tree breeding, all these, have influenced natural gene stock of forest communities. Most European forests are also affected by historical forest management and despite the intention of the last decades, to preserve sustainable silviculture, the long history in forestry has left strong imprints in the genetic makeup of forest tree species.

Genetic studies in forest tree populations using a large stock of genetic markers make it possible to reveal all these historical processes, to evaluate the composition and quality of the forest gene stock, and foresight future ability of communities to adapt to changing environmental conditions.

To mitigate the effects of the changing climate in forest ecosystems and to conduct CSF primarily, it is important to have deep insights into the genetic constitution of species and their populations. High levels of "standing genetic diversity" in populations is a prerequisite for species to face fast environmental changes as selection and fixation of new adaptive mutations take a comparatively longer time. In contrast to new adaptive mutations, standing variation most probably has already passed through a "selective filter" and might have been formerly tested by selection in past environments (Barrett and Schluter 2008). Selection of the new alleles in tree species with their long-life span needs comparatively more time; thus, lack of adaptation may end in the decline of the functional traits.

Former case studies on the phenotypic variation of forest trees, experiments in provenance trials beginning from the early 1970s, and many common garden experiments provided a large source of data helping in understanding the adaptive behavior of species. Although these were not designed to monitor the effects of climate change, they still provide insights into the aspects of the genetic variation and of the adaptive response to species to the acting environmental forces. A more recent study by use of field trials and modelling tools tried to determine the extent to which four widespread forest tree species in Europe (Norway spruce, Scots pine, European beech and sessile oak) may be affected by the climatic change (Mátyás and Kramer 2016).

To explore the impacts of environmental change on the adaptive potential of trees, functional phenotypic traits need to be assigned to allele composition. This is not always unequivocal as for many traits, there is still limited knowledge, likely because they are regulated by multilocus systems. Thus, a genome-wide scanning of the changes in population genetic diversity based on neutral markers represents another but a more conservative approach (Kramer et al. 2010). However, an increasing number of projects mapping complete genomes of mountain forest tree species (Mosca et al. 2019) and the development of gene-specific primers makes it already possible to identify nucleotide diversity in genes and candidates responsible for the adaptive variation.

To grant forest tree populations the ability to keep an adequate level of genetic diversity, to maintain viability, and to support long-term evolutionary potential, genetic aspects should be embedded in the forest management (Buiteveld et al. 2007). Forest ecosystems will only persist if genetic variation and allele composition of trees are maintained at a high level and this especially holds in view of the environmental changes. Therefore, studies on the genetic makeup of species and populations should be performed before starting the planning of any forest management activity. Moreover, selection and conservation of multiple genetic resources should be also in the focus.

A case study on the genetic variation of pure beech stands along species' distribution range was initiated within CLIMO (COST Action, CA15226). The novelty of this work is the coupling of the genetic data with other empirical measurements within the considered study plots. Dominant trees from 12 study plots were subjected to molecular evaluation based on six nuclear microsatellite markers. The overall high genetic variation of the stands was correlated to local climate variables. Among the genetic indices, the number of alleles and Shannon genetic diversity were shown to be highly correlated with daily temperature and the frequency of frost days (Höhn et al. 2021).

8.4.12 Threatened Forest Species

Because of the extensive studies that had been carried out for years, species diversity turned out to be the easiest aspect to implement among the main biodiversity components (Kraus and Krumm 2013). Species diversity remains a keystone, and the loss of species is the most recognizable form of biodiversity decline.

European Red List of Trees identifies those species that are threatened with extinction at the European level to inform about actions needed to improve their conservation status (Rivers et al. 2019). The list summarizes the results for the assessment of all known native European trees, a total of 454 species, of which 265 (over 58%) are endemic to continental Europe. In common with vascular plants (Bilz et al. 2011), some of the highest levels of endemism are found in the main mountain chains, such as the Alps, Pyrenees, Carpathians, Apennines, Dinaric Mts., and others. The mountain areas also represent the richest centers in Europe (Rivers et al. 2019).

Overall, 191 (42%) European tree species have been assessed as having a high risk of extinction, that is, assessed as critically endangered, endangered, or vulnerable. A further 13 tree species are assessed as almost meeting the criteria for a threatened category, while 216 of them are considered of not being of conservation concern (Rivers et al. 2019).

Each country may produce its national red lists (most often containing more species than in the European red list), which assess the risk of species' extinction within the country borders. The species considered of not being under threat in the European scale locally can be seen as critically endangered or vulnerable. Thus, several countries have developed management or action plans for various species and have legislation in place to protect certain species legally. Some examples of successful initiatives include the Regional Programme of Conservation and Restitution of *Sorbus torminalis* in Poland (Zwierzyński and Bednorz 2012), and the *Zelkova* global action plan (Kozlowski and Gratzfeld 2013).

Forest management can negatively affect several taxa by reducing the number of natural attributes, such as microhabitats bearing trees, deadwood quantity, and its diversity (Lafond et al. 2015). However, the opposite happens: for example, in European beech forests, no loss in vascular plant species restricted to forests occurred over the past 250 years despite forest management (Schulze et al. 2016). On the other hand, some field studies highlight a positive effect of forest management on the diversity of such taxa, as bird species related to open spaces and species of the understory vegetation (Lafond et al. 2015).

In uneven-aged forests, large tree retention has a positive effect on several structure and biodiversity indicators. In particular, it seems efficient to compensate for the negative impacts of increased harvesting intensity by limiting the decrease in tree sizes diversity. Through the analysis of compensating effects, it has been revealed the existence of possible ecological intensification pathways, that is, the possibility to increase management intensity while maintaining biodiversity through the promotion of nature-based management principles, that is, gap creation and retention measures (Lafond et al. 2015).

Nowadays, climate change represents an important driver of species extinction, the importance of which is increasing, when acting in synergy with habitat destruction and fragmentation. Climate change will cause many tree species to lose a part of their current habitat but will also enable them to colonize new habitats (e.g., del Río et al. 2018). As natural migration is often slow, forest managers are examining the option of assisted migration (Martín-Alcón et al. 2016; Gömöry et al. 2020). A main premise of SFM is that silviculture based on patterns and processes found in old-growth forests will maintain the provision of important habitats for biodiversity (Schütz et al. 2016). Although in some parts of Europe is implemented "close-to-nature management" (Schütz 1999; Brang et al. 2014; O'Hara 2016), the resulting forests lack the diversity in composition and structure of forest ecosystems that are driven by natural succession and dynamics so characteristic to old-growth forests (Kraus and Krumm 2013).

8.4.13 Protective Forests (Soil, Water, and Other Ecosystem Functions)

Protective forests as one of SFM indicators are presented here in the broader context, that is, including all FRA 2020 categories as described in Global Forest Resources Assessment (FAO 2020) for European countries, except Russian Federation and Turkey. Thus, apart from protective forests primarily designated for protection of soil and water, Table 8.2 contains data on areas primarily designated for other forest ecosystem functions, such as conservation of biodiversity, social services, and of multiple use. Such an approach enables a general understanding of the role of forest in maintaining also other forest functions.

Any undisturbed forest has a capacity to effectively sustain the maintenance of biological diversity and protection of soil and water, since these capacities/functions are integrated with natural processes ongoing in forest ecosystems. Forest functions as such, reflect a forest ecosystem capacity of sustaining expected people's demands, while forest ecosystem services account for the provision of requested benefits (Lesiński 2012). These benefits are of the basic value for peoples' welfare.

Forests primarily designated for various management objectives (FAO 2020), with exception of production, occur in as many as 30 out of 36 countries and they all together cover 42.55% of the total forest area in Europe (Table 8.2). The representativeness of the above FRA 2020 categories in individual countries and their average share in the total forest area in Europe are as follows:

- Multiple use: 18 countries; area – 19.50%
- Protection of soil and water: 21 countries; area 10.70%
- Conservation of biodiversity: 26 countries; area 10.03%
- Social services: 19 countries; area 2.32%

In 12 countries' data in the matter, production has not been at all mentioned as one of primarily designated forest management objectives, and in six of them, forests have not been designated for any other objective, either. The latter have been described either as other, none, or unknown (Table 8.2).

When it comes to distinguishing forest functions, two main approaches might be considered – the zoning approach and the integration approach (Bončina 2011b). The zoning approach takes place mainly in scarcely populated countries/regions, large forest areas, or mountain ranges (Lesiński 2012). Protective forests are typical examples of such an approach. In Europe, they cover large areas in the mountains of Switzerland, Austria, Romania, Moldova, and Norway (Table 8.2). The integrated approach suits much better in densely populated countries/regions (Lesiński 2012). Integration approach results in multifunctionality that is achieved by combining various objectives at a larger scale. France, Belgium, Netherlands, and Spain are very good examples of implementing such an approach (Table 8.2).

Table 8.2 Primary designated forest management objectives 2020 expressed by the percent of the total forest area in the country

Country	Production (%)	Protection of soil and water (%)	Conservation of biodiversity (%)	Social services (%)	Multiple use (%)	Other/none/ unknown (%)	Total forest area (1000 ha)
France	–	–	–	–	100.00	–	17,253.00
Belgium	–	–	1.80	–	98.20	–	689.30
Netherlands	0.82	–	39.31	–	59.87	–	369.50
Poland	–	20.29	10.29	10.77	58.65	–	9483.00
Slovakia	22.88	18.23	2.33	8.44	48.14	–	1925.90
Spain	19.14	17.63	22.18	–	41.05	–	18,572.17
Slovenia	47.38	19.17	0.90	2.05	30.50	–	1237.83
Bulgaria	38.53	9.38	18.08	5.78	28.23	–	3893.00
Iceland	33.30	29.97	0.33	12.85	23.38	0.17	51.35
Moldova	–	58.09	–	19.63	22.28	–	386.50
Denmark	80.23	–	5.08	–	14.69	–	628.52
Croatia	68.80	12.57	2.87	1.41	14.35	–	1924.98
Sweden	70.00	–	18.30	–	11.70	–	27,980.00
Lithuania	71.66	6.13	9.40	2.68	10.13	–	2201.00
Norway	46.55	37.84	6.02	0.16	9.43	0.21	12,180.00
Estonia	72.85	4.55	13.63	–	8.97	–	2438.01
Serbia	65.65	21.96	6.67	2.93	2.13	0.66	2722.65
Portugal	–	–	–	–	0.10	99.90	3312.00
Austria	61.50	37.27	–	1.23	–	–	3899.15
Albania	79.60	16.48	3.92	–	–	–	789.00
Latvia	76.31	6.41	15.26	2.02	–	–	3410.79
Czech Rep.	73.78	9.45	9.10	5.28	–	2.39	2677.09
Belarus	49.91	16.31	15.56	14.71	–	3.51	8767.60
Hungary	49.55	9.28	24.69	0.95	–	5.35	2053.01

(continued)

Table 8.2 (continued)

Country	Production (%)	Protection of soil and water (%)	Conservation of biodiversity (%)	Social services (%)	Multiple use (%)	Other/none/ unknown (%)	Total forest area (1000 ha)
Switzerland	27.51	54.60	8.43	2.65	–	6.81	1269.11
Ukraine	37.18	25.87	14.55	14.96	–	7.46	9690.00
Romania	42.71	36.30	4.63	–	–	16.36	6929.05
Ireland	44.38	–	5.55	0.07	–	50.00	782.02
Finland	–	–	12.64	2.36	–	89.00	22,409.00
Italy	3.43	–	0.97	–	–	95.60	9566.13
Germany	–	–	–	–	–	100.00	11,419.00
UK	–	–	–	–	–	100.00	3194.06
Greece	–	–	–	–	–	100.00	3901.80
Bosnia & Herzegovina	–	–	–	–	–	100.00	2187.91
Northern Macedonia	–	–	–	–	–	100.00	1001.49
Luxembourg	–	–	–	–	–	100.00	88.70
Total – area	59,101.37	21,476.84	20,147.61	4664.84	39,132.60	56,161.58	200,684.84
Total – %	29.45	10.70	10.03	2.32	19.50	28.00	100.00

Order of countries by Multiple use (descending) and by Other/None/Unknown (ascending)
Source: Reports for individual European countries; FAO (2020)

Protective forests are forests that have as their primary function to prevent soil erosion, to preserve water resources and protect infrastructure, people and/or their assets against the impact of natural hazards. The main tool used by these forests are standing trees, which act as obstacles to downslope mass movements such as rock falls, snow avalanches, erosion, landslides, debris flows, and floods. The protective effect of these forests is ensured only if the silvicultural system used and any natural disturbances that occur leave a sufficient amount of forest cover (Brang et al. 2006). Thus, the continuous cover approach maintaining uneven-aged tree-stands seems to be the best solution (Mason et al. 1999; Pukkala and Gadov 2012).

8.4.14 Slenderness Coefficient

The slenderness coefficient (H/D ratio) is commonly used in studies of the resilience of trees and forest stands to the destructive activity of wind and snow (Abetz 1976; Burschel and Huss 1997). This is a synthetic indicator describing the shape of the tree trunk (the stem taper, the opposite of slenderness). The slenderness coefficient is calculated by dividing the height (H) by the diameter at breast height (D). The higher the diameter at breast height of a tree with the same height (lower H/D values), the stronger the force necessary to bend the trunk. A low slenderness coefficient is found for a longer crown, lower center of mass, and better-developed root system due to the large space for growth. Free growth of the crown facilitates the increase of growth of the diameter at breast height and the increase of the stem taper (Petty and Worrell 1981). Expansion of the space for the growth of the crown also limits their asymmetric development (Petty and Swain 1985; Valinger et al. 1994), with trees with regular crowns less susceptible to leaning or swinging (e.g., due to an asymmetric snow load). A torsional moment is also observed less frequently. The tree roots in a loose stand tend to be better anchored in the soil, leading to increased resilience to tree damages (Nielsen 1995). Peltola et al. (1997) explain that with a low slenderness coefficient, snow is dislodged from the tree crowns by gusts of wind, and these two damaging factors do not accumulate.

In some of the publications, the authors use the slenderness coefficient as a measure of tree/stand stability. They formulate conclusions and assessments concerning the rules of tending stands, assuming that lower H/D values will result in higher resistance to the wind (snow) (Wang et al. 1998; Wilson and Oliver 2000; Castedo-Dorado et al. 2009; Vacchiano et al. 2013; Meng et al. 2017). A different methodological approach is the creation of models allowing the prediction of the fact or probability of wind damage emergence. In these models, the slenderness coefficient is one of the many variables. An analysis of efficiency, measures of fit, prediction, and classification capacity of these models indicates that the slenderness coefficient, as a single variable, is frequently of relatively limited value as an explanatory variable. The studies by Pukkala et al. (2016) show the slenderness coefficient as useful, but only in interaction with specific basal area and diameter at breast height values. In the work by Martín-Alcón et al. (2010), the slenderness coefficient influenced the prediction of the percentage of damaged trees in a stand, but only when divided by the basal area. Slenderness alone is not a good indicator of tree stand stability, one also requires a factor indicating mutual tree support within a forest stand (Schütz et al. 2006; Schelhaas et al. 2007; Martín-Alcón et al. 2010). The work by Albrecht et al. (2012) concludes that a variable tree height should be used in conjunction with H/D. The usability of the slenderness coefficient as a predictive measure was also critically assessed in studies covering historic data. Oliveira (1987) and Schütz et al. (2006) also do not recommend the usage of the H/D ratio as a single variable to predict resistance to wind damage. In the work of Díaz-Yáñez et al. (2017), the best prediction models use the slenderness coefficient to clarify the probability of damage at less than 10%. Tree height covered the majority of the probability of damage emergence.

Due to the common usage of the slenderness coefficient as the measure (indicator) of resistance against wind damage, many authors indicate its desired/critical values. In Germany, Abetz (1987) recommended a slenderness factor of ca. 80 as suitable for Norway spruce. Burschel and Huss (1997) suggest the following value scale for coniferous species: very unstable (H/D > 100), unstable (H/D 80–100), stable (H/D < 80), alone trees (H/D < 45). The values given by other authors are similar (Johann 1981; Cremer et al. 1982; Rottmann 1986; Becquey and Riou-Nivert 1987; Lohmander and Helles 1987; Peltola et al. 1997; Wilson and Oliver 2000). Summarizing their research, Wilson and Oliver (2000) conclude that there is no single H/D value guaranteeing stability. It depends on the wind strength. We are not able to eliminate the hazard of wind, and a lower H/D value leads to a reduction in value or production costs (pruning). Skrzyszewski (1993) concluded that promotion of trees (European spruce) with a low slenderness coefficient (below 80) during thinning leads to the emergence of tree stands with trees that have branches extending low over the ground, with a large share of ingrown knots and showing a very tapered stem and broad rings, resulting in low mechanical strength and reduced wood durability in coniferous species. Safe strategies are sparser planting spacing and/or high-intensity cuttings at a young age (sapling and pole stage), and in neglected forest stands – very low-intensity cuttings during many entries. An alternative may be growing all-age (selection) stands (Dobbertin 2002; Griess and Knoke 2011; Jaworski 2013; Hanewinkel et al. 2014; Pukkala et al. 2016).

The slenderness coefficient is the result (effect, derivative) of a specific vertical structure of a stand, its age (height), and density as well as the silvicultural treatments executed in the past. The quoted publications indicate that the slenderness coefficient should be analyzed in association with other variables (site conditions, soil humidity, exposure, slope, altitude and age as well as the height of neighboring tree stands).

8.5 Silvicultural Treatments Improving Stand Mitigation

8.5.1 Growing Stock

Keeping the average growing stock low is mentioned as one of the principles for climate change adaptation in forest management (Brang et al. 2014). This might be especially true in the case when forest fires cause stand damages, which is typical for Mediterranean forests. However, the high growing stock of forest stands can increase damage susceptibility also in mountain areas, especially in the case of wind throws or insect attacks, and much less for ice or snow breaks (Jalkanen and Mattila 2000; Klopčič et al. 2009).

At present, the average growing stock in European forests is 163 m^3/ha, ranging from 10 m^3/ha in Iceland to 352 m^3/ha in Switzerland (Forest Europe 2015). During the last decades, the rate of annual felling to wood volume increment has been relatively stable and remained under 80% for most countries in Europe. Such a ratio of wood utilization allows the forest stock to increase being a right action within mitigation of climate change. However, the ratio of harvesting to increment is assumed to increase in the nearest future since the demand for woody biomass as a renewable energy source is expected to increase (Forest Europe 2015). On the other hand, the higher growing stock as well the age of forest stands may increase the occurrence and the severity of insect outbreaks in forests prone to the pest damages (Pasztor et al. 2015). The crucial issue is to keep the average value of growing stock on a certain level allowing to control trade-offs between climate change mitigation actions focused on growing stock increase (increase carbon stores) and adaptation potentials focused on wood harvesting enabling to create multilayer and uneven-aged stands (D'Amato et al. 2011). For example, Pretzsch et al. (2014) and Schütz (2003) suggest to strive to achieve the average growing stock amounted to 300–400 m^3/ha on fertile site conditions and 250–350 m^3/ha in medium site conditions in mountain mixed-species forests.

8.5.2 Carbon Stock (Soil)

Soils are the largest carbon pools of most of the mountain forest ecosystems and are more stable and less exposed to sudden fluctuations than trees (Scharlemann et al. 2014; Achat et al. 2015). Nevertheless, forest management can affect soil organic carbon (SOC) stock and accumulation rate (Mayer et al. 2020; Tonon et al. 2011). The extent to which different silvicultural systems, harvest intensities (percentage of original biomass), frequency, modality (tree stem or whole tree collection), and the degree of mechanization might impact SOC is still debated due to the huge number of external factors (climate, species composition, litter quality, type of soil, and relative clay content) involved in the ecosystem response (Hoover 2011). In this context, although conflicting results can be found in literature, often linked to the different timescale of the experiments, some general indications can be provided to maximize the SOC stock of mountain forests. First, all forest management strategies with a positive effect on the ecosystem's net primary productivity have to be considered as putative strategies to increase SOC accumulation through the increased production of leaf, root, and woody litter. An example is a conversion of pure to mixed forest in central Europe (Aguirre et al. 2019; Pretzsch et al. 2020b; Torresan et al. 2020) with evident positive effects in terms of productivity and resilience to climate changes. Recent evidence shows that intensive forest harvest, based on the collection of tree stems and logging residues, has a deleterious effect on SOC content resulting in a consistent and lasting loss of carbon that affects both the surface and mineral soil layers, suggesting that residues management is one of crucial aspect to take into account in the context of the CSF (Achat et al. 2015). On the other hand, it has been shown that traditional forest harvest has a temporary effect limited to the forest floor without any serious impact on the mineral deeper soil layers (Achat et al. 2015; Mayer et al. 2020).

The transition from the traditional age class forestry to the continuous cover forestry toward the conversion of even-aged to irregular or uneven-aged forests is a further indication that emerged from the literature as a potential option with a positive long-term effect on soil and forest carbon storage (Seidl et al. 2008). This structural change can be reached through the application of selective harvesting, such as single-tree selection or small-group selection cutting. However, this transition is not possible everywhere as it requires local technical skills, suitable climate conditions to support natural regeneration under the typical uneven-aged forest microclimate, and a flexible timber market. In agreement with the ECCP-Working Group on Forest Sink (2003), 30 million hectares of forests at the European level could be converted to continuous cover forestry with an important carbon gain at the regional level. Where social, technical, or ecological reasons make the conversion to the continuous cover forestry inapplicable, the elongation of the rotation period of even-aged forest can be considered as a possible strategy to promote carbon sequestration. Indeed, the time that the system needs to recover to the pre-disturbance SOC content can largely vary according to climate, soil conditions, and magnitude of the perturbation. Therefore, the rotation period in even-aged forestry should be at least longer than the recovery time. An additional point supporting the elongation of the rotation period is the evidence that forest ecosystems are far to be carbon-saturated, since several old-growth forests are still accumulating carbon in the different compartment, soil included, at different altitudinal and latitudinal belts with an important accumulation rate (Gunn et al. 2014; Schrumpf et al. 2014; Vedrova et al. 2018; Badalamenti et al. 2019; Keeton 2019).

8.5.3 Roundwood (Timber Products)

One of the main effective measures to mitigate climate change is to retrieve carbon from the atmosphere and store it as long as possible in roundwood as a timber product resulting from the stand management expected by society (Kauppi et al. 2018). The roundwood timber can be used for many purposes in building construction and human everyday life. The best climate change mitigation effect is achieved when the wood is converted into long-lived products and where the same wood unit is used in several, successive product cycles (Jandl et al. 2018). Wood production belongs to the component of the CSF concept concerning mitigation related to sustainably increasing forest productivity and income and relies on using wood resources sustainably to substitute nonrenewable, carbon-intensive materials (Nabuurs et al. 2017; Verkerk et al. 2020). To produce the best quality and sufficient amount of different assortments of roundwood timber, many different optimal silvicultural measures depending on the species composition, site conditions, and societal expectations can be implemented. The most important silvicultural treatments are thinnings, both precommercial and commercial, which systematically applied in the stand can lead to the rotation age with high-quality crop trees. Some economic analyses have shown that properly made precommercial thinning often is the most rewarding long-term investment that can be made during the silvicultural prescription throughout a stand's life (Smith et al. 1997). The various methods of commercial thinning focused on the production of best timber products at the end of rotation are undertaken in Europe. The common thinning rule is the selective promotion of the best trees (taking into account vitality, social position, quality, stability) in the stand using the procedure based on Assmann concept (1961) of maintaining stand basal area allowing to maintain the growth at the level of at least 95% of the maximum periodic annual volume increment. Other models of thinnings are mostly adapted to the main tree species in the stand and their selection is also determined by the final production objective (target timber assortments) of stand management. Here we are presenting some examples of particular methods applied in stands consisting of main tree species: *Pinus sylvestris* stands – thinning leading to construction timber and/or valuable wood (Burschel and Huss 1997); *Picea abies* stands – the selection of thinning procedure based on the stand age, method of stand formation, origin, and stability (Burschel and Huss 1997); Abetz's model (1975) of thinning with a selection of a certain number of future best trees; *Abies alba* stands – selection of the best crop trees and selective thinning (Korpel 1975), differentiation thinning leading to the forest with selection structure (Schütz 1989; Korpel and Saniga 1993); *Quercus* stands – three methods to produce veneer, valuable, and sawn timber (Burschel and Huss 1997); *Fagus sylvatica* stands – selective thinning (Burschel and Huss 1997), qualitative group thinning (Kato and Mülder 1983; Röhring et al. 2006), thinning procedures by Freist and Altherr

(Burschel and Huss 1997). However, most of them were developed in the period when climate change was not an important issue, so they need to be modified according to the current state of knowledge and predicted climate change effect on timber production.

8.6 Application of Simulation Models for Development, Testing, and Improving Silvicultural Prescriptions

8.6.1 The Role of Models in Forest Science and Practice

Under changing environmental conditions, models become particularly important as they may provide information about the efficacy regarding the mitigation and adaptation potential of new silvicultural measures, which are so far not at all or not sufficiently covered by experiments.

For instance, experiments addressing the effects of thinning in mixed stands or the transition from even-aged monocultures to uneven-aged mixed stands are still rare; so, the potential of mixing and transitioning on the resistance, resilience, or recovery under drought stress may be explored by model scenarios.

The main role of models is the integration of existing knowledge for better understanding and regulation of well-analyzed ecosystems under stable conditions; however, in the view of unknown future development, their contribution to exploring future management strategies by scenario analyses gains an increasing importance. Ecophysiologically based models derive the system behavior from mechanistic relationships but often lack an interface to silvicultural management. Management models statistically reflect the tree stand growth observed on experimental and inventory plots, cover the practical relevant dendrometric variables but are less flexible regarding changing growing conditions. By deriving relationships between growth and water supply with ecophysiological models, statistically modelling such relationships and integrating them into management models pros and cons of both model approaches may be combined as shown by Schwaiger et al. (2018a, b).

Silvicultural prescriptions can serve as guidelines for tree and stand regulation. Their development starts with a participative target development (Pretzsch et al. 2008). Their quantitative development is based on expert knowledge, existing experimental plots, and inventory data and is also often based on scenario analyses with models. Models may reveal which treatment variants match the best with the ecosystem functions and services aimed at by the defined target state. In this case, models are used as virtual experiments for testing the long-term consequences of various treatment options. By comparing the model outcome with the target, the most promising and stable variants may be selected and formulated as easy-to-apply rules for use in forest practice.

8.6.2 Models as a Substitute for Missing Experiments

Strong stand density reductions, thinning in mixtures, transitioning from even-aged monocultures to uneven-aged mixed stands, gap cuts, or inner edges may be caused by silvicultural measures of mitigation or adaptation to drought and climate change. The effects of such measures on the long-term are unclear, as respective long-term experiments are hardly available (Pretzsch et al. 2020a).

However, spatially explicit individual tree models, parameterized based on long-term experiments with distant-dependent prognosis algorithms, are suitable for assessing the reactions of such silvicultural measures. Especially models that are based on basic rules such as allometric principles, self-thinning lines, density competition relationships, or dose–response functions for quantification of the relationships between growth and environmental conditions may be helpful. They interpolate or extrapolate beyond the range of stand and structure conditions covered by the parameterization data sets.

Starting with defined stands and environmental conditions, various treatment options may be tested by simulation runs. Link functions to various ecosystem services enable the assessment of the long-term consequences of various treatment options and their match with the target settings by management. The treatment options that meet the best the various criteria of the defined target are an appropriate basis for further development of practical guidelines.

Certainly, additional aspects, for example, of forest utilization, wood quality, and ecological and socioeconomic impact, need to be considered; however, the effects of the silvicultural measures on the natural production represent an essential pillar and step in the process of guideline development.

8.6.3 Models as Decision Support in the Case of an Unclear Future Development

In the last few decades, forest science and forestry were faced with the environmental impact on forest ecosystems such as acid rain, increasing atmospheric ozone concentration, and eutrophic deposition as well as climate change. There is hardly any previous experience from experiments or monitoring on how forestry may mitigate or adapt to such environmental changes. Chamber experiments are restricted to small trees and field experiments are costly and very long-lasting; they are important but not sufficient to quickly provide forest management with recommendations for decision making under environmental stress.

Model predictions and model scenarios are often the only alternatives for getting decision support, but their results should be interpreted and applied with due care for the following reasons: Mechanistic models that are based on general relationships between environmental conditions and ecophysiological processes. As they are hardly parameterized for the special stress conditions, they may provide false estimations of the stress reactions, acclimation, and adaptation reactions of forest ecosystems (Bréda et al. 2006). Statistical models that apply the "space for time" approach and derive the temporal reactions on different environmental conditions from the plant's reactions along spatial gradients may underestimate the time trees need to adapt to or recover from exposition to stress events. Individual tree models that estimate the tree's growth response on tree size and competition within the stands may be parametrized for even-aged stands where most trees have a similar history. If their functions are used to predict tree growth in mixtures, in heterogeneous stands or regeneration phases with gaps and edge trees, there may be false estimations as the trees are no longer growing similarly but have much more diverse courses of growth than in monospecific and even-aged stands. Thus, their growth courses can hardly be predicted just from their present size and competition. More information about past development, for example, about the inner and outer traits, may be necessary for appropriate predictions. Improved models are on the way (Sievänen et al. 2000; Pretzsch et al. 2002; Rötzer et al. 2012) and may be used for silvicultural scenario development for mitigation and adaptation measures (Hilmers et al. 2020).

8.6.4 Model Scenarios to Fathom Out the Potential of Adapting Forest Stands to Climate Change by Silvicultural Measures

Model scenario calculations may help to better adapt forest ecosystems to environmental changes and stress. We see mainly five lines of models' applications for the development of climate-smart silvicultural measures.

(i) Models, mainly ecophysiologically based models that consider the water cycle, may contribute to developing thinning strategies for mitigation and adaptation to drought stress (Rötzer et al. 2012).
(ii) Models may contribute to test the climate smartness of so far not well-known neglected domestic and introduced tree species. Of special interest is whether species, for example, such as sorb tree (*Sorbus domestica*) or Douglas-fir (*Pseudotsuga menziesii*) or Turkey oak (*Quercus cerris*), may be suitable substitutes for more drought-susceptible tree species.
(iii) Tree species mixing by mixing from the beginning on or underplanting may improve the stand growth stability due to risk distribution, improved recovery, or resistance to drought due to interspecific facilitation (Forrester and Bauhus 2016).
(iv) As there are still only a very few long-term experiments on transitioning from even-aged monocultures to uneven-aged mixed stands (Pretzsch 2019), models may support the derivation of suitable scenarios concerning multiple ecosys-

tem services. Of special interest are the effects of a transition to continuous cover forestry regarding the reduction of damages and growth losses by wind and storm, bark beetle, and drought stress.

(v) Reduction of the rotation length or introduction of midrotation mixture may reduce risks and increase productivity due to adaptation to the species-specific growth rhythms and damage susceptibilities and temporary tree species mixing.

In all cases, model scenarios may serve as a substitute or completion of information derived from experiments or observations.

8.6.5 Example of the Application of Models for the Development of Silvicultural Guidelines

Since early medieval times, mixed mountain forests of Norway spruce, silver fir, and European beech have been replaced by Norway spruce monocultures for wood supply of the salt works in Southern Germany (Hilmers et al. 2020). The resulting nonnatural and unstable secondary Norway spruce monocultures suffered various damages by snow and wind in the last decades. In addition, they are strongly affected by drought stress events and continuous climate warming (Pretzsch et al. 2020b).

To increase the stability of these historically destabilized and presently endangered forests, alternative management concepts are being intensively discussed and introduced. A promising option to restore the stability of these ecosystems is their transformation from pure Norway spruce stands into site-appropriate, sustainable, and stable mixed mountain forests of mainly Norway spruce, silver fir, European beech, and sycamore maple.

However, there are hardly any practical examples for such silvicultural transformations. As way out we used the stand growth simulator SILVA to develop and assess the results of various transformation scenarios. We also analyzed any trade-offs between the pros and cons of various scenarios and the resulting success criteria (Hilmers et al. 2020). Here we show the results of the test of seven different transformation scenarios (e.g., slit-coupes, shelterwood and gap-coupes, strip clear-cutting, do-nothing). We report their impact on the five evaluation criteria forest stand growth, economical results, carbon store, stand stability, and biodiversity.

Some of the results are visualized in Fig. 8.4. In essence, we found out that the scenarios applying gap or slit-coupes resulted in the most beneficial overall utility values regarding the five evaluation criteria. We could also show the best way to transform destabilized forests to sustainable and stable ecosystems. In this way, guidelines for restauration and transformation can be developed, compared, and ranked regarding their achievement, even if respective model stands are scarce or missing. A precondition is the availability of simulation tools that are based on basic principles of tree and stand dynamics and thus can realistically simulate also treatment options and development scenarios that were not used for the model parameterization (Pretzsch et al. 2008).

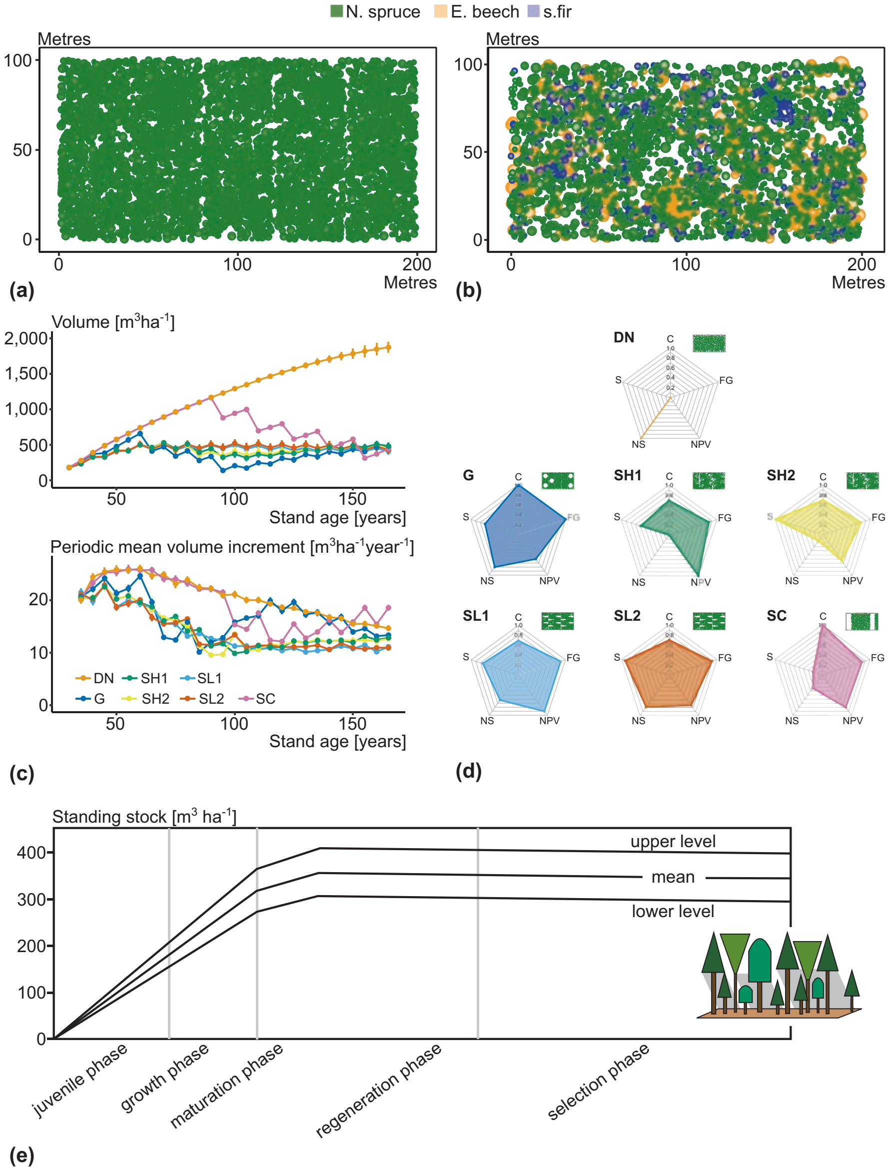

Fig. 8.4 The development of silvicultural guidelines by model scenario calculations comprises the (**a** and **b**) initial state and aim of the modelling exercise, (**c**) definition of the standing stock for the model scenarios (above) and the resulting growth (below), (**d**) the multicriteria overview of the long-term consequences of various options regarding ecological, economical, and socioeconomic criteria, (**e**) simplified silvicultural guidelines for practical application. (After Hilmers et al. 2020) **a and b visualizes the objective, that is, the transition from even-aged monospecific Norway spruce stands to uneven-aged mixed-species stands of mainly Norway spruce, silver fir, and European beech**

8.6.6 *From Models for Regulation and Optimization to Guidelines for Silvicultural Steering*

We showed that model-based scenario analysis may contribute to deriving an appropriate silvicultural option. For this purpose, various treatment variants may be implemented algorithmically in a growth model, for example, various tree number-mean tree height guidelines (N-h-curves) for steering the stand development over time. For each of $i = 1 \ldots n$ treatment options, the stand development can be simulated, and the result may be compared with the target state. The very treatment options that show the best approximation of the stand to the defined target may be of special interest as a suitable guideline for practical application. The derivation is often based on a combination of a normative, experimental, and simulation approach and also includes ecological and socioeconomic criteria.

The involvement of a scenario analysis requires a quantitative formulation of a set of treatment options and finally provides a quantitatively based treatment option that might be used for target-oriented silvicultural steering in practice.

This model-based derivation of a silvicultural treatment guideline represents a regulation process (Fig. 8.5). By definition (Berg and Kuhlmann 1993), regulation means that a development is controlled by the initial conditions, the time exogen variables and that the rules can be modified by a feedback between the current development and the applied rules (closed-loop-control).

The application of this once derived curve for stand treatment represents steering. Steering means that system development is controlled just by the initial conditions, the time exogen variables, and static rules (open-loop control). In this case, once derived and prescribed rules (e.g., guideline curve or threshold), provide the setpoint for stand characteristic (e.g., density, mixing proportion, number of future crop trees) to which the stand is adjusted by silvicultural interventions in defined intervals. In the case of steering, there is no feedback between the stand development and the once fixed guideline. The application of a given silvicultural treatment in forest practice, that is, the thinning of a stand based on a defined N-h curve, represents steering.

Fig. 8.4 (continued)

c shows the stand volume of the remaining stand in m^3 ha^{-1} (above) and the mean stand periodic annual volume increment in m^3 ha^{-1} year^{-1} (below) resulting from 30 simulations of the following variants: DN, do-nothing scenario; G, Gap-coupes with the planting of beech and fir; SH1, shelterwood-coupes with natural regeneration; SH2, shelterwood-coupes with the planting of fir and beech; SL1, slit-coupes with natural regeneration; SL2, slit-coupes with the planting of fir and beech; SC, strip clear-cutting with natural regeneration
d shows radar charts of the evaluation of multiple criteria. FG, forest growth; NPV, net present value; C, carbon sequestration; S, stability; NS, number of species. The scaled results of the respective factors of each criterion are shown. Results were scaled between 0 and 1. Results evaluated with 1 represent the best scenario in comparison to the other scenarios. Categories rated 0 show the worst scenario. DN, do-nothing scenario; G, Gap-coupes with the planting of beech and fir; SH1, shelterwood-coupes with natural regeneration; SH2, shelterwood-coupes with the planting of fir and beech; SL1, slit-coupes with natural regeneration; SL2, slit-coupes with the planting of fir and beech; SC, strip clear-cutting with natural regeneration
e represents the main aspect of the derived silvicultural guideline, the target standing stock for stands growing under different site conditions

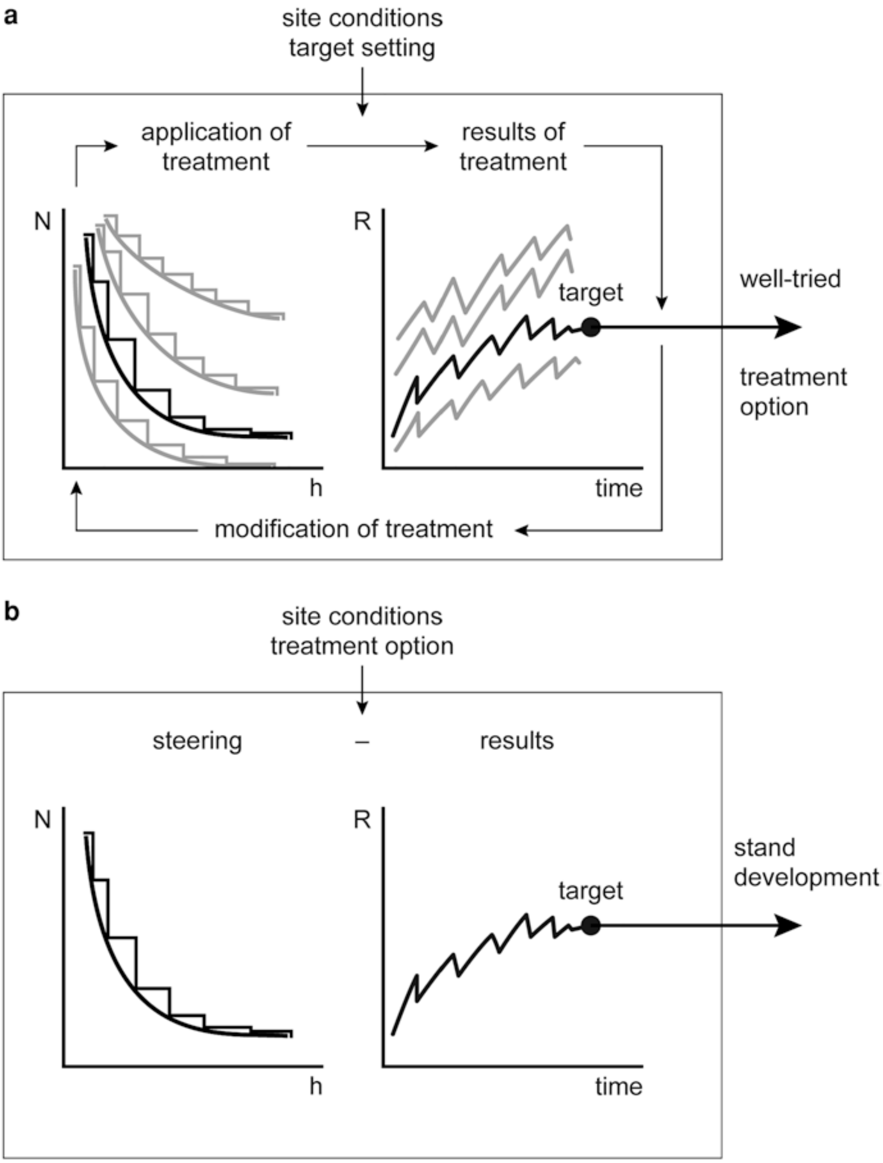

Fig. 8.5 Regulation (**a**) vs. steering (**b**) of systems after Berg and Kuhlmann (1993). (**a**) The effects of silvicultural treatments on stand development may be simulated by a model. The stand development can be compared with the management objective, and the repeated modification of the treatment and simulation generates a set of scenarios with some of them appropriate for reaching a defined management objective. The feedback between the simulated stand development and the treatment option makes this process a regulation. (**b**) A derived silvicultural treatment may be applied as a silviculture guideline for stand management. As there is no feedback between stand development and guideline characteristics, this process is called steering

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