Chapter 14
Forest Management

Peter Meyer and Christian Ammer

Abstract From an ecological perspective, forestry interventions can be defined as disturbances actively implemented at different spatial scales with the aim of obtaining a variety of forest-based ecosystem services. By changing the spatial and temporal distributions of resources, they alter competition between trees at the individual, species, and generational levels. In addition to silvicultural measures in the strict sense, drainage, liming, and pest control also constitute ecological disturbances. The diversity of stand and landscape structures that result from natural and forestry-initiated disturbances has important consequences for biodiversity. Forestry-initiated and natural disturbances have many similarities and also major differences. Ecologically oriented forestry practices are those that integrate the essential elements and attributes of the natural disturbance regime into forest management.

Keywords Emulating natural disturbances · Forest dynamics and management · Forestry-initiated disturbances · Landscape effects · Stand regeneration · Silvicultural systems · Tree removal

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T. Wohlgemuth et al. (eds.), *Disturbance Ecology*, Landscape Series 32,
https://doi.org/10.1007/978-3-030-98756-5_14

14.1 Importance of Disturbance Ecology for Forest Management

Forestry practices in Central Europe have changed profoundly, especially since the 1980s. Instead of the management of pure, even-aged stands with the primary goal of timber production, forestry has been increasingly understood as the management of complex ecosystems with multiple objectives at different spatial scales (Kohm and Franklin 1997). With the increasing recognition that the complexity and diversity of forests, and thus the ecosystem services they provide, depend largely on the impacts of disturbances, an understanding of the effects of natural and management-related disturbances has become of central importance.

In forestry, the emulation or integration of natural disturbances is often seen as a promising approach for maintaining biodiversity at different levels (from the gene pool level to the species level to the ecosystem level) (Franklin et al. 2002; North and Keeton 2008) and to reduce management efforts according to the principle of "biological rationalization" (Schütz 1996; Puettmann et al. 2009). Close-to-nature forestry practices have a long tradition in Central Europe, especially in Germany (Gayer 1886; Möller 1923), and the corresponding management concepts have still developed further (Pommerening and Murphy 2004; Schmidt 2009). Recently, new concepts have evolved which give priority to natural processes over explicit production targets (Sturm 1993; Otto 1995 with critical discussion, Puettmann et al. 2009).

Studies of European primeval forests (Leibundgut 1993; Korpel 1995) as well as the establishment and monitoring of set-aside forests (Bücking 1997; Meyer 1997) have led to improved understanding and greater appreciation of natural disturbances for the dynamics of forest ecosystems (Meyer et al. 2004; Brang 2005; Svoboda et al. 2012; Trotsiuk et al. 2014; Hobi et al. 2015; Winter 2015). Such insights have formed the basis of close-to-nature forest management, even if, due to climate change, the disturbance regime of the past may be of limited applicability as a reference for the future (Puettmann et al. 2014).

14.2 Historical Changes in Forest Management

Central European forests have a long history of use, dating back thousands of years. Ever since the Neolithic period, forests have been cleared for the establishment of agricultural land and the extraction of firewood and timber and to increase pastureland and hunting grounds (Grober 2013). The different forms and intensities of forest utilization resulted in changes in the extent, duration, and type of anthropogenic disturbances, and they have changed the tree species composition of forests (Firbas 1949; Ellenberg et al. 2010) (Fig. 14.1). In the Middle Ages, complex management systems, including the establishment of coppice and coppice with standard forests, were already oriented toward sustainable multiple use and ensured simultaneously firewood and timber supply as well as pastureland. With the upswing of

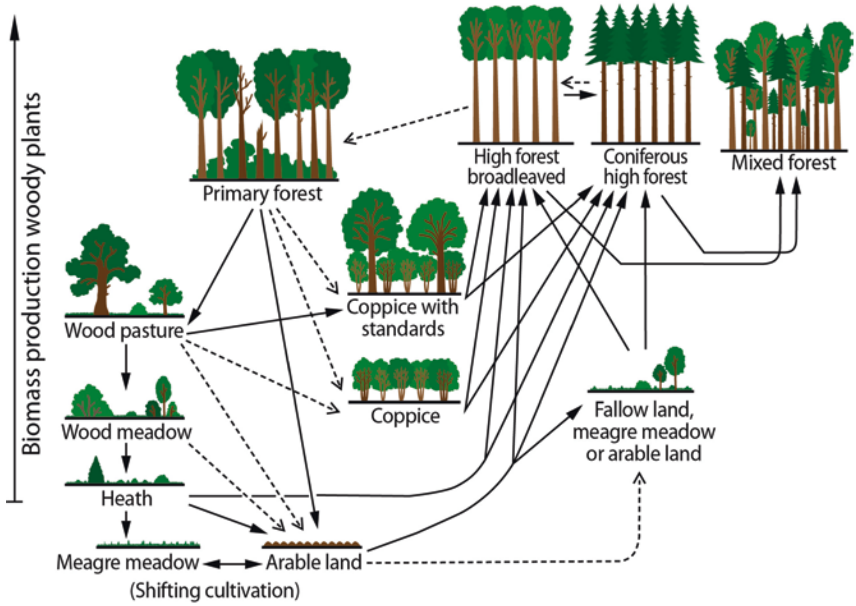

Fig. 14.1 The development of historical and recent forest and agroforestry management systems. The predominant system in use today is high forests. Continuous arrows indicate significant development and dotted arrows less significant development. (Adapted from Ellenberg 1996)

modern forestry during industrialization, the locally and regionally heterogeneous management systems were transformed to high forests on a large scale and managed predominantly for wood production.

14.3 Evaluation of Disturbances in the Forest

Generally, the management of natural resources aims at minimizing the disruptions to production. In particular, the simplification of forest structure in the form of even-aged pure stands and the standardization of interventions aimed at controlled and predictable wood supply (Puettmann et al. 2009). However, it gradually became clear that this management strategy resulted in forests that were more vulnerable to unanticipated natural disturbances (Holling and Meffe 1996). Modern approaches to forest utilization, therefore, aim at diversifying forest area units, interventions, and structure (Wagner 2007). Moreover, the spectrum of tolerated forest disturbances has changed considerably in recent decades. Thus, in the 1970s, area-wide windthrows and bark beetle infestations were perceived exclusively as catastrophes (Kremser 1973), whereas today such disturbances are handled in a more nuanced manner (BUWAL 2000; Brang et al. 2015). After the drought period of 2018–2020, extensive areas of bark beetle infestations characterize the landscape, not only in

Fig. 14.2 Bark beetle infestations at Mount Lusen, Bavarian Forest National Park (Germany). (Photo: NW-FVA)

protected areas (Fig. 14.2) but also in production forests. At the same time, there is less acceptance of larger anthropogenically caused disturbances. The increasingly critical view of activities such as clear-cutting and shelterwood cutting (Fig. 14.3) motivated the development of wood certification systems to ensure that wood production and forest management are carried out in a sustainable manner with respect to economic, environmental, and social criteria and that take into account the interests of various stakeholder groups (FSC Arbeitsgruppe Deutschland 2012).

14.4 Comparison of Natural and Anthropogenic Forest Disturbances

The similarities and differences between natural and anthropogenic forest disturbances can be assessed based on various criteria, especially the type of disturbance, the strength of the disturbance (affected biomass per unit area), the spatial extent of the disturbance, the frequency of recurrence in the same area, and the consequences for the tree population (Tables 14.1 and 14.2).

Both forestry practices and natural disturbances often result in the removal of trees from the living stand and thus reduce stand density. However, while in forestry a large part of the biomass is removed, in natural disturbances all of the biomass typically remains in the ecosystem. Thus, a major difference between natural and

Fig. 14.3 A beech stand after shelterwood cutting favouring regeneration of oak. The image was originally labelled in Kaiser et al. (2012) as representing a clear-cut. (Photo: Andreas Varnhorn/ Greenpeace)

anthropogenic forest disturbances is the amount of deadwood left in the forest. For natural disturbances that do not lead to the death of trees but only to a reduction in their vitality, such as diseases or irregular flooding, there are very few analogies in modern forest management systems. An exception is prescribed burning to reduce combustible biomass or initiate natural regeneration, the effect of which on the soil and tree populations may resemble that of natural fires (Pyne et al. 1996; Kraus and Zeppenfeld 2013). Some forestry measures, such as the "snapping" of trees during thinning (see Sect. 14.6.2), do not directly cause tree death and have a natural equivalent. Conversely, forest interventions like road building, drainage, and soil cultivation have no natural counterparts.

In most cases, natural disturbances occur irregularly and create heterogeneous spatial patterns (Turner 2010). This is only partially the case for forest interventions, which instead tend to create more homogeneous spatial effects (however, see Sect. 14.6). Natural disturbances typically create wide, irregular transition zones (ecotones) between disturbed and undisturbed areas, whereas anthropogenic forest disturbances such as road building have largely linear effects with narrow transition zones. Also, the designation of management units results in linear borders.

In addition, the phases of forest development following natural disturbances differ from those that result from anthropogenic disturbances (Fig. 14.4) (Leibundgut 1993). For many years, trees of advanced age as well as uncleared, deadwood-rich windthrows, and areas affected by bark beetle infestations did not occur in regularly managed high forests. However, silvicultural systems are increasingly being

Table 14.1 Typology of natural disturbances in forests compiled in view of the concepts of Oliver and Larson (1990), Richter (1997) and Roberts (2004)

Type of disturbance	Intensity	Spatial extent of effects (Single tree / Tree group / Stand / Landscape)	Affected layer (Canopy / Stem space / Roots)	Frequency (Frequent / Sporadic / Rare)	Affected by management	Direct effects on the tree stand
Storm	◉	Decreasing: Single tree – Stand	Range: Canopy – Roots	Decreasing: Frequent – Sporadic	●	Windthrow and breakage of whole trees
Hurricane	●	Increasing: Tree group – Landscape	Range: Canopy – Roots	Increasing: Sporadic – Rare	◉	Areawise windthrow and breakage
Ground fire	○	Decreasing: Single tree – Landscape	Range: Stem space – Roots	? (Sporadic)	●	Lower vitality; higher mortality
Crown fire	●	Increasing: Stand – Landscape	Range: Canopy – Roots	Increasing: Sporadic – Rare	●	Lower vitality; much higher mortality
Flood*1	○-●	Increasing: Stand – Landscape	Range: Roots	Increasing: Sporadic – Rare	-	Lower vitality; higher mortality
Landslide	●	Range: Stand – Landscape	Range: Canopy – Roots	Increasing: Rare	●	Areawise mortality
Rockfall	●	Range: Stand – Landscape	Range: Canopy – Roots	Increasing: Rare	-	Areawise mortality
Avalanche	●	Range: Stand – Landscape	Range: Canopy – Stem space	Increasing: Rare	○	Areawise windthrow and breakage
Disease	○-◉	Range: Single tree – Landscape	Range: Canopy – Roots	Range: Frequent – Rare	◉	Lower vitality; higher mortality
Pests	○-●	Range: Single tree – Landscape	Range: Canopy – Roots	Increasing: Frequent – Rare	○	Lower vitality; higher mortality

*1 Regular flooding in floodplain forest is not considered a disturbance

Impact/importance: ○ = Low, insignificant, rare, slow ◉ = Medium ● = Strong, important, frequent, fast

↔ = Range ◣ = Decreasing ◢ = Increasing

implemented in which old trees are not harvested but are instead retained as legacy trees (see Gustafsson et al. 2012; Sect. 14.6.3.1). Given the shortened life cycle of a commercial forest, the temporal distribution patterns of forest disturbances will differ depending on the silvicultural system. Harvesting, as the largest disturbance, usually starts in the "optimum" development phase (Fig. 14.4) and may extend over several decades (Fig. 14.5). In natural broadleaved and mixed forests of the temperate zone, disturbances are rare during this phase (Fig. 14.4), and tree mortality is typically low (Holzwarth et al. 2013).

14.5 Silvicultural Systems as Anthropogenic Disturbances

Silvicultural interventions are characterized by the type, strength, and cycles of tree removals or tree enrichments carried out over the course of a stand's life for the purpose of stand maintenance, timber harvesting, and the establishment of tree

Table 14.2 Typology of forest-related disturbances in high forests

Type of disturbance	Intensity	Spatial extent of effects: Single tree	Tree group	Stand	Landscape	Affected layer: Canopy	Stem space	Roots	Frequency: Frequent	Sporadic	Rare	Direct effects on the tree stand
Weeding	○-◉	◣	◣			⟷	⟷	⟷			◥	Decreased competition of ground flora
Improvement thinning	○	◣	◣			⟷	⟷	⟷			◥	Change in competition relationships
Thinning from below	○-◉	◣	◣	◣		⟷	⟷	⟷	◣	◣	◣	Change in competition relationships
Thinning from above	○-◉	◣	◣	◣		⟷	⟷	⟷	◣	◣	◣	Change in competition relationships
Light felling	◉-●	◣	◣	◣		⟷	⟷	⟷			◥	Strong reduction of stand density
Harvesting	○-●	⟷	⟷	⟷	⟷	⟷	⟷	⟷			◥	Reduction of old tree coverage
Ploughing	●			⟷				⟷			◥	Improved establishment of seedlings/saplings
Soil scarification	○-◉			⟷				⟷			◥	Improved establishment/growth of seedlings/saplings
Fertilization	○-◉	⟷	⟷	⟷				⟷			◥	Improved nutrient supply
Liming	○			⟷				⟷		◣	◣	Compensation of acid rain; change in nutrient supply
Road building	◉-●			⟷	⟷	⟷	⟷	⟷			◥	Linear thinning; interruption of water flow
Skid road construction	○-◉			⟷		⟷	⟷	⟷	◣	◣		Linear thinning
Drainage	◉-●			⟷	⟷			⟷			◥	Water excess less frequent
Planting of non-native tree species	○-●			⟷	⟷	⟷	⟷	⟷			◥	Change in competition

Impact/importance: ○ = Low, insignificant, rare, slow ◉ = Medium ● = Strong, important, frequent, fast
⟷ = Range ◣ = Decreasing ◥ = Increasing

regeneration. A distinction is usually made between coppice, coppice with standards, and high forest silvicultural systems.[1] In all three systems, the average living biomass during the course of the management cycle is generally less than that

[1] According to Vergani et al. (2017) the systems are defined as follows:

coppice: the cutting of the stems of young trees or shrubs close to the ground, causing them to resprout and to re-establish the canopy, or an area so treated.

coppice with standards: forest or stand consisting of coppice among which a number of trees (standards), that are generally of seedling origin, are retained on a long rotation to provide large material and seeds to regenerate the forests.

high forest: a forest management system which allows the trees to grow to at least two-thirds of their ultimate height, as opposed to earlier cutting or coppicing where a much lower canopy is formed.

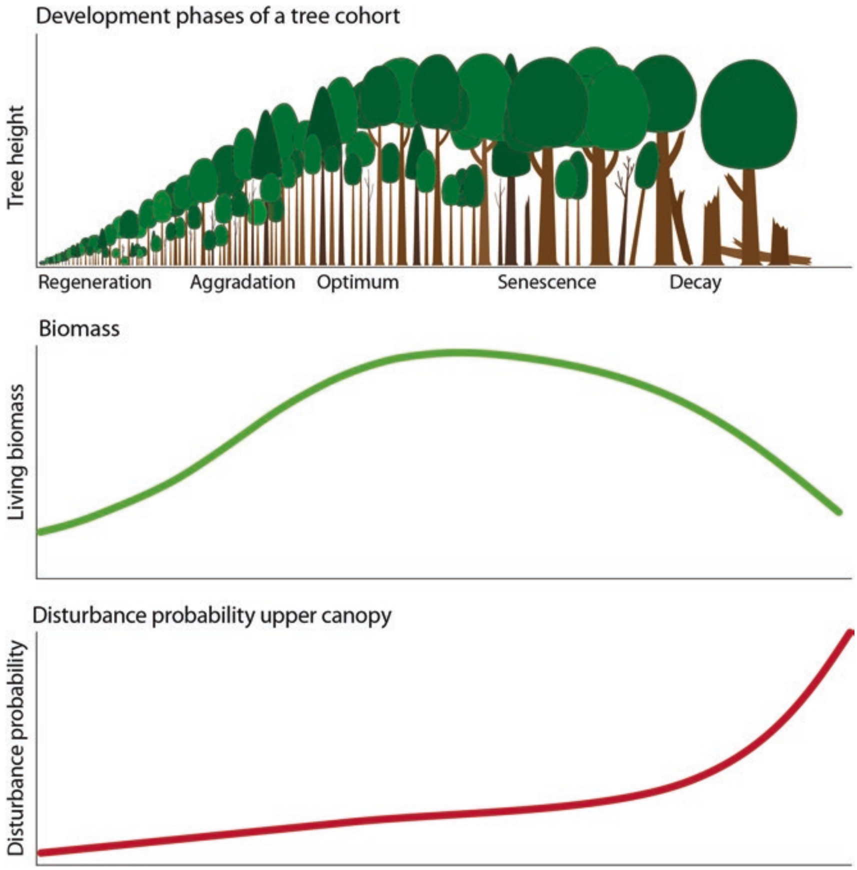

Fig. 14.4 Development of biomass (center) and frequency of natural disturbances (bottom) during the different developmental phases of a tree cohort (top) in broadleaved and mixed broadleaved–coniferous forests of the temperate zones. Derived from conceptual models developed by Watt (1947), Korpel (1982) and Oliver and Larson (1990)

of an undisturbed forest stand (Fig. 14.4). Actually, “the coppice system involves reproduction by [stump] shoots or suckers. When felled near ground-level, most broadleaved species, up to a certain age, reproduce from shoots sent up from the stump” (Troup 1928). The coppice and coppice with standard systems are typically managed in the form of a spatially coherent system of cutting (felling) areas in which a proportion of the area is cut each year and managed on a rotational basis. Since timber is harvested from a different area each year, the number of fellings corresponds to the length of the rotation period. For coppice forests, rotation periods of 15–20 years are common and for coppice with standards 20–30 years. Coppice forests are mainly used to produce small poles and firewood as the tree stems are typically small.

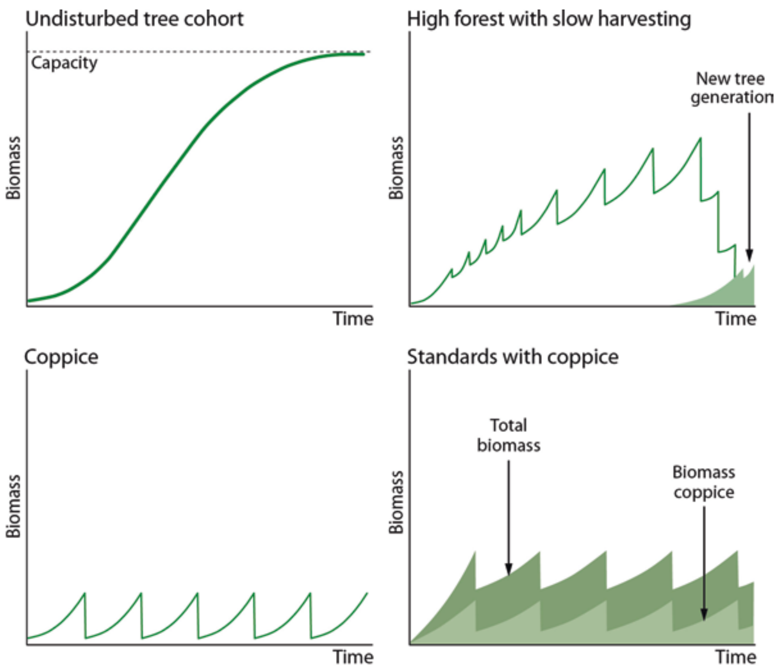

Fig. 14.5 Schematic development of the aboveground woody biomass of an undisturbed tree cohort (**a**) compared to that of a high forest with a temporally extended harvest (**b**), a coppice (**c**), and a coppice with standards (**d**)

Coppice with standards is composed of the vegetatively regenerated coppice layer and individual trees that emerged from seeds. The seedling trees eventually form a loose sub-canopy whose crowns cover 30–50% of the area (Fig. 14.6). Firewood resulting from the coppice layer and sawlogs from canopy trees are the main products of these forests. In the past, interventions were often combined with the establishment of pastures below the canopy trees. In Central Europe, the average aboveground biomass in coppice forests has been estimated at 60 t ha^{-1} (±38 t ha^{1}) and in coppice with standard forests 103 t ha^{-1} (±58 t ha^{-1}) (Albert and Ammer 2012). High forests are distinguished by a significantly higher maximum biomass of >400 t ha^{-1}, depending on the tree species and with large fluctuations over time. Wood growth is significantly higher in high forests than in coppice and coppice with standard forests (Albert and Ammer 2012). The primary goal in high forests is the production of sawtimber.

Because of the mosaic-like divisions resulting from the different felling areas, coppice and coppice with standards offer a wealth of edges and different developmental phases that within a small area provide a high species and structural richness (Schröder 2009; Fartmann et al. 2013). However, the practices that give rise to these areas have become rare in Central Europe. Moreover, this forest management regime cannot be compared to natural disturbances, as neither the regular spatial

Fig. 14.6 Typical structure of a coppice with standards forest 1 year after cutting of the shrub layer, Liebenburg forest of Lower Saxony in the northern Harz region of Germany. (Photo: NW-FVA)

pattern of successive areal disturbances nor the vegetative regeneration within these areas is found in natural forests in Europe. The exceptions are subalpine bushes and mountain pine forests, both of which are exposed to regular rockfall and small avalanches. In addition, consistent coppice and coppice with standard forest management give sprouting tree species a clear competitive advantage (Matula et al. 2012). There is also a certain resemblance between the canopy layer of coppice with standard forests and windthrow areas, where 30–50% of the stand may survive. However, the quality of the remaining trees is very different: In the coppice with standards stand vital trees with large crowns predominate, whereas in natural windthrow areas trees in poor and intermediate condition remain together with partially damaged trees (Fig. 14.7).

A much greater similarity exists between managed high forests and natural broadleaved and coniferous forests. Trees in high forests nearly reach their maximum natural height, and by the time stands are harvested, they can achieve a comparably high biomass. For example, the biomass of a 60-year-old Douglas fir (*Pseudotsuga menziesii* [Mirb.] Franco) stand may be already half that of a 450-year-old natural forest of Douglas fir and western hemlock (*Tsuga heterophylla* [Raf.] Sarg.) (Barnes et al. 1998). Note that a typical rotation for Douglas fir in Central Europe is 80 years. In German high forests of European beech (*Fagus sylvatica* L.), the volume stock at 120 years (yield table of Wiedemann 1931) corresponds to the average stock of a primeval European beech forest (582 m^3 ha^{-1}; Hobi et al. 2015). The average stock of 30- to 120-year-old beech high forests of equal area is about 350 m^3 ha^{-1}. In these comparisons, the respective removals in the course of management and the amounts of deadwood left in the forest must be taken into account. In

Fig. 14.7 An 8-ha windthrow that occurred in 2014 in the primeval beech forest Havešová, eastern Slovakia. (Photo taken in 2015; © Peter Meyer, NW-FVA)

the primary beech forests of the Western Carpathians investigated by Hobi et al. (2015), the amount of deadwood was 162.5 ± 8.4 m^3 ha^{-1}.

The regionally predominant natural disturbance regime will influence the assessment of the closeness to nature of the different types of cutting. While small-scale disturbances dominate in temperate broadleaved forests (Fig. 14.8; Seymour et al. 2002; Hobi et al. 2015), large-scale windthrows, fires, and insect infestation are much more common in boreal forests. In high-elevation montane and subalpine locations, landslides and avalanches also lead to large-scale disturbances (Bebi et al. 2009). However, these can also occur at lower elevations, such as in the European beech forests of southeastern Europe (Nagel et al. 2014; Hobi et al. 2015). Storm-damaged areas in European beech primeval forests (Fig. 14.7) and in natural forest reserves have also been documented (Willig 2002; Schmidt and Meyer 2015). Rare strong interventions over a larger contiguous area also contribute to a greater closenes to nature in nemoral broadleaved forests and are congruent with naturally occurring disturbances (Foster and Boose 1992). In principle, in terms of the amount of disturbed area, no cutting method can be considered unnatural as long as it does not occur significantly more or less often than naturally occurring disturbances (Fig. 14.8). Thus, silvicultural cutting methods do not create light conditions different from those that can result from natural disturbances. However, in all other respects, such as deadwood dynamics, the consequences of almost all interventions in silvicultural systems differ significantly from those of natural disturbances.

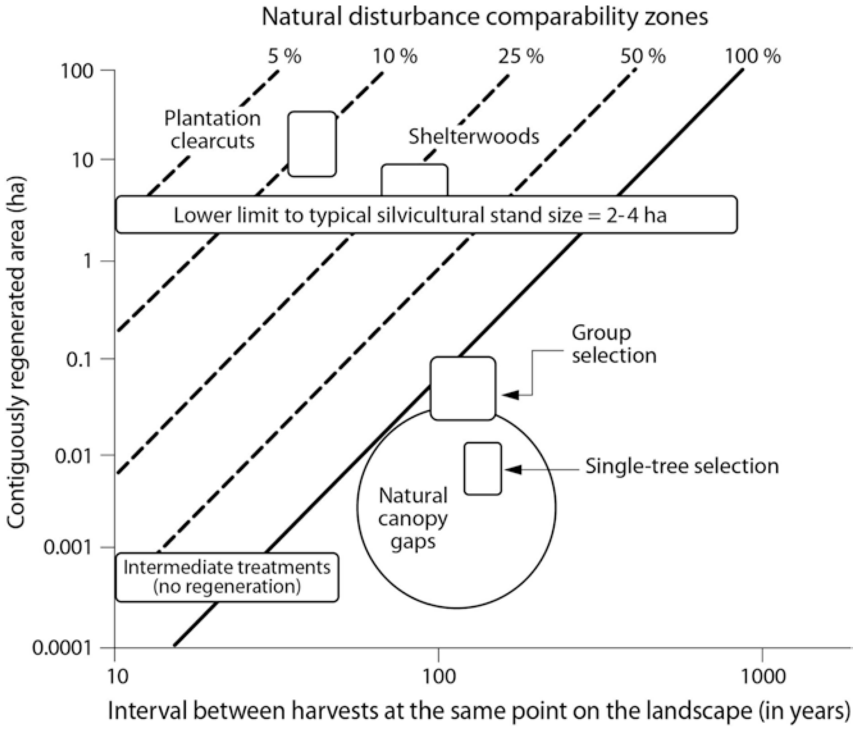

Fig. 14.8 Frequency and extent of the disturbances caused by different cutting regimes compared to natural disturbances in northeast American temperate broadleaved and mixed broadleaved–coniferous forests (Seymour et al. 2002). The comparison zones reflect the recurrence of cutting events compared to the recurrence of natural disturbances in an area of the same dimensions, for example, a recurrence of group shelterwood cutting ("Femelschlag") events every 100 years would be comparable to the natural disturbance regime

14.6 Disturbance Effects of Individual Forest Measures

In the assessment of disturbances, it is primarily their effects on the structures and processes of the concerned ecosystems that are decisive rather than whether they are anthropogenic or natural in their origin (Bazzaz 1983). Disturbances can abruptly change the density and thereby the competitive interactions in forest stands as well as the growth and mortality of trees and regeneration dynamics; consequently, they typically lead to a reorganization of the forest ecosystem (Fig. 14.9). Few forest management measures are so extreme that their impact exceeds that of the natural (undisturbed and disturbed) fluctuation range of the ecosystem. In the long run, many pioneer tree species can survive only if major disturbances occur, as they are unable to successfully regenerate in closed forests.

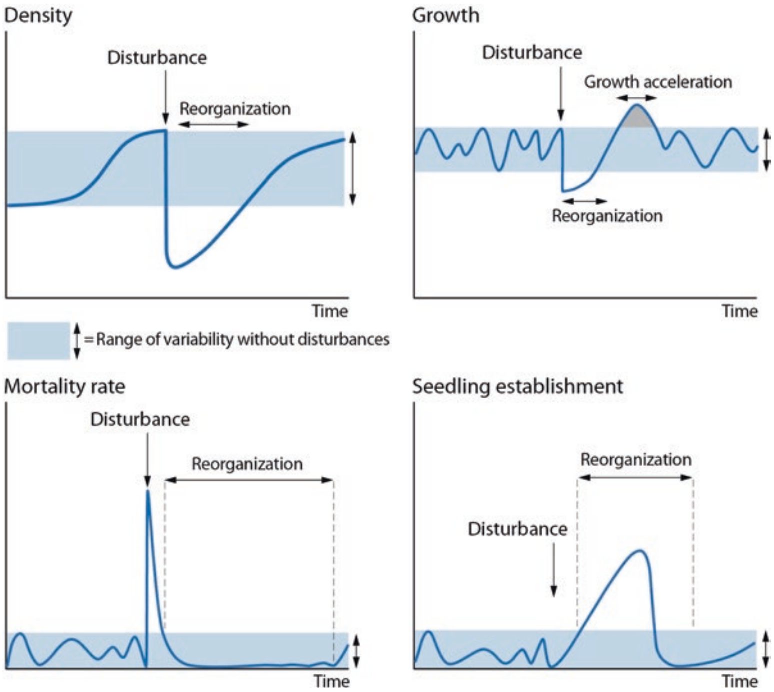

Fig. 14.9 Assumed changes in stand density and population dynamics in forests induced by disturbances: density (**a**), growth (**b**), mortality (**c**), and establishment of seedlings (**d**)

In general, forestry measures reflect two fundamental management decisions: (1) the selection of tree species and (2) the regulation of stand density (Schall and Ammer 2013). From an ecological viewpoint, the selection of tree species may lead to a significant deviation from the species involved in forest development under natural conditions at the same site, while the regulation of stand density influences intra- and interspecific competition and causes a redistribution of resources among the remaining trees (Ammer 2008). The interventions differ depending on a stand's development phase. During the initial phase, they aim at regeneration establishment and securing sapling growth. In this case, the disturbances created by management may also address the competing vegetation. In young stands, trees will be removed by tending measures either because their stem quality is poor or to obtain a desired mixture of tree species. Here, the silvicultural disturbance disrupts natural intra- and inter-species competition because it does not always select trees that would have prevailed in the absence of human interference. In subsequent thinnings, the best quality trees or individuals of certain tree species are promoted by removing competitors. Once the production target is reached, the stand is harvested. The effects of this type of disturbance will vary depending on the progress of the interventions and the amount of wood removed (see Sect. 14.6.3).

14.6.1 Vegetation Management and Tending of Young Stands

Interfering ground vegetation is eliminated in favour of planted, sown, or naturally seeded trees (Davis et al. 1998). In contrast to North America and other parts of the world, where herbicides are frequently used for this purpose, in Central Europe management of ground vegetation is usually done mechanically (Ammer et al. 2011). The redistribution of resources due to vegetation management favours the growth of the desired young trees (Harrington et al. 1999) and leads to a change in biomass allocation. Thus, an increase in the amount of light is usually accompanied by an increase in root biomass in relation to the total biomass (Shipley and Meziane 2002; Schall et al. 2012).

14.6.2 Cleaning and Thinning

The number of plants in a fully stocked even-aged stand decreases as the mean plant biomass increases. This fundamental relationship, the self-thinning line, is independent of human influence (Reineke 1933; Yoda et al. 1963). This density-dependent mortality can be preempted by sufficiently strong interventions. During the thicket phase, that is, from canopy closure to the beginning of natural pruning which usually corresponds to a diameter at breast height of dominant trees of about 15 cm, few interventions are carried out to stop the natural dying-off process, especially in the case of broadleaved trees; instead, clearing consists only of the removal of single, qualitatively unsatisfactory individuals. In the case of coniferous trees, however, for reasons of stability (e.g. to avoid snow-induced breakage), a reduction in the stem number is often carried out already at this stage.

The effects of thinning depend on the frequency, strength, and type of thinning. So-called “thinning from below” essentially intervenes in the suppressed stand layer, so that changes in the competitive environment of dominant trees are largely insignificant. In contrast, interventions in the dominant stratum (“thinning from above” or “crown thinning”) have a much larger impact, as they increase the amount of resources available to the remaining trees. Depending on the age of the trees and the growth dynamics of the particular tree species, the remaining trees respond with an increased crown surface. After a certain time lag, this is reflected in a stand productivity that may be higher than at maximum density, assuming that the density has not been excessively reduced (Assmann 1961; Pretzsch 2004; “growth acceleration” in Fig. 14.9b). However, it remains unclear how long this increase in production is maintained and whether the productivity achieved over the entire life span of a stand will be higher than that of an untreated stand (Curtis et al. 1995; Zeide 2001). The stand density at which maximum productivity is achieved depends on the age and mean diameter of the stand (Zeide 2004) and is lower for light-demanding than for shade-tolerant tree species (Pretzsch 2005). Thinning accelerates individual plant development such that a certain target diameter will be reached more quickly. In addition, crown thinning can ensure the survival of suppressed trees as well as the conservation of less competitive species. Examples include the promotion of oaks (*Quercus* spp.) or of other broadleaved tree species such as European ash (*Fraxinus excelsior* L.) and maple species (*Acer pseudoplatanus* L., *A. platanoides* L.) in European beech stands (Nüsslein 1995). This corresponds to natural forests where certain tree species are also favoured, for example, by the emergence of stand gaps during the regeneration phase (Poulson and Platt 1996).

As recent studies have shown (Kohler et al. 2010; Gebhardt et al. 2014), thinning also leads to temporarily reduced drought stress. This is because water loss due to interception is reduced, which allows a larger amount of precipitation to reach the forest floor (Stogsdili et al. 1992; Simonin et al. 2007) and also because stand transpiration decreases significantly. The end result is an increase in the amount of available water (Aussenac and Granier 1988; Gebhardt et al. 2014). Analogous to light extinction, these effects are not proportional to stand density (Bréda et al. 1995). Thus, the effectiveness of an intervention depends on its intensity. However, very strong interventions can also lead to a lush growth of the ground vegetation (Son et al. 2004). The positive effect of thinning on a forest stand continues for several years after a drought event. In the case of more frequent drought events (Lindner et al. 2014), repeated, strong interventions are among the effective measures allowing the adaptation of forest stands to climate change (Ammer 2017).

The effects of thinning on the mechanical stability of the stand and thus on its sensitivity to future disturbances in the forms of wind and snow can be classified in two successive phases. Immediately after thinning, there is a period where the stability of the stand decreases (Richter 2003; Albrecht et al. 2012); however, this is followed by a second, longer period of greater stability, as the morphological adaptations of individuals increase their resistance to mechanical stress. As a result of thinning, the height–diameter ratio decreases, and the crown length increases; both of these characteristics are thought to reduce the risk of windthrow (Mayer and Schindler 2002).

14.6.3 Final Harvest, Stand Regeneration

Harvest of Mature Stands

Depending on the intensity of harvesting and the size of the affected area, harvesting mature trees is accompanied by changes in the abiotic conditions of varying degrees, as the amount of light, the temperature regime, and the availability of belowground resources are affected. The removal of mature trees from a stand is usually the starting point for the establishment and/or development of tree regeneration.

The impact of harvesting operations differs mainly with regard to the affected area, the number of trees removed per intervention, the number of interventions, and

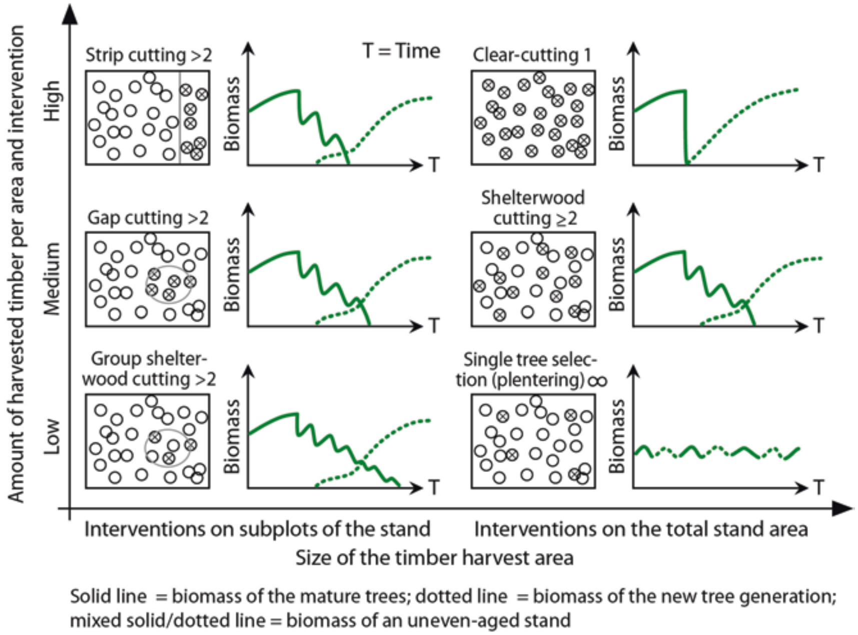

Fig. 14.10 Classification of the typical cutting methods used in harvesting wood and regenerating forest stands according to the amount of wood removed per sub-area, intervention, and the area affected by the intervention. The numbers indicate how often the interventions are carried out until the stand is fully harvested. For all cutting methods, the affected area (each circle represents a tree, circles with a cross are the trees removed during an intervention) and the development of the stand's biomass (solid line: biomass of the mature trees (in case of plentering: total stand biomass), dotted line: biomass of the new tree generation) over time are shown (© Abt. Waldbau und Waldökologie der gemäßigten Zonen der Fak. f. Forstwissenschaften und Waldökologie der Georg-August-Univ. Göttingen)

the distribution of the removed trees (Fig. 14.10). The strongest changes are imposed by clear-cutting, in which no mature trees remain on the harvested area. The resulting changes in abiotic conditions are similar to those caused by large-scale natural disturbances (Fig. 14.11) and include increased differences between day and night temperatures, changes in wind speed and light conditions, changed amount of water reaching the forest floor, increased evaporation from the soil surface and transpiration by vegetation on the felled area, decreases in air and soil moisture, and an increase in the litter decomposition rate, which may be accompanied by humus losses. In addition, the rate of nitrogen mineralization increases (Chen et al. 1993; Carlson and Groot 1997), which, depending on the rate at which the ground vegetation is re-established, may be associated with temporary nitrogen losses (Lindo and Visser 2003; Weis et al. 2006; Klinck et al. 2013). In terms of stand regeneration, these conditions favour early successional species and/or those that tolerate high irradiation but also frost, such Norway spruce (*Picea abies* [L.] Karst.) and Scots pine (*Pinus sylvestris* L.) (Fisichelli et al. 2014).

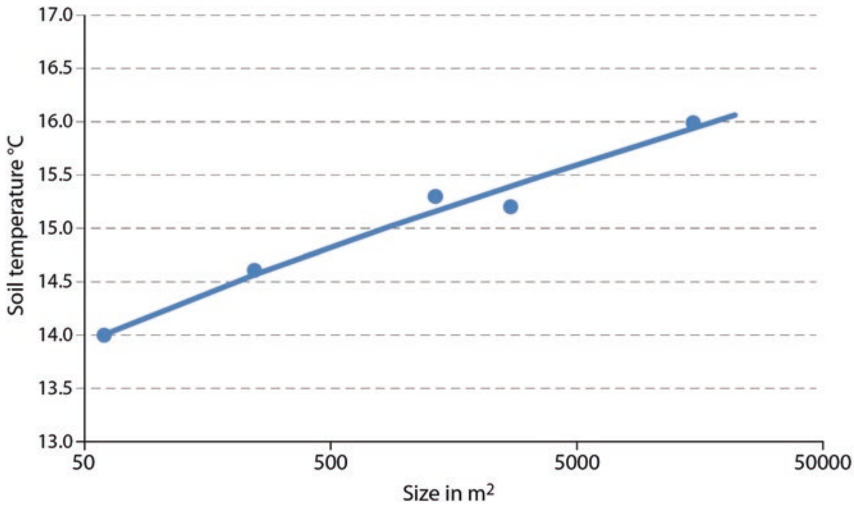

Fig. 14.11 Change in the soil temperature (5 cm depth, average values from June to September) as a function of the size of the area affected by the intervention. Note the exponential scale on the x-axis. The largest area represents a 1.5-ha clear-cut, the smallest a 60-m^2 gap. (© Abt. Waldbau und Waldökologie der gemäßigten Zonen der Fak. f. Forstwissenschaften und Waldökologie der Georg-August-Univ. Göttingen)

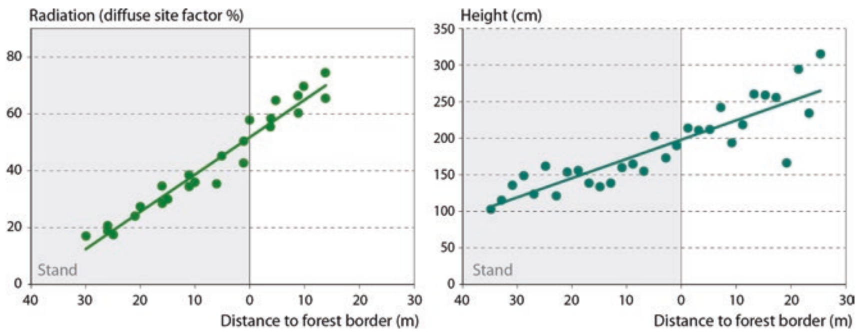

Fig. 14.12 Radiation (DIFFSF = diffuse site factor in %) and the heights of regenerated beech in the inner and outer areas demarcated by a margin. (Figure adapted from Wagner and Spellmann 1993)

The complete removal of trees on a small area is carried out during certain group and strip felling silvicultural systems, such as gap cutting (German: Lochhieb; typical gap diameter of approximately 30 m; see Wagner 1999; Moslonka-Lefebvre et al. 2011) and strip cutting (German: Saumhieb; the width of the strip where all mature trees are removed usually corresponds approximately to the height of the trees; see Röhrig et al. 2006). In these cases, the change in environmental conditions is not as drastic as in clear-cutting. Both gap cutting and strip cutting lead to a gradient of ecological conditions that allows tree species with different light requirements to regenerate simultaneously (Fig. 14.12). In the case of strip cutting, the amount of radiation on the forest floor increases progressively from within the stand to the stand edge to the harvested area. The availability of belowground resources increases accordingly. In the case of gap cutting, significant increases in the amount of light and in belowground resource availability occur toward the center of the gap (Ritter et al. 2005; Herrmann 2014). In the Northern Hemisphere, the highest radiation will be found at the northern edge of the gap, assuming that the ground is level (Wagner 1999). For both gap and strip cutting, it takes years to decades until all mature trees of a stand are removed.

Also in the case of shelterwood (German: Schirmschlag) and group shelterwood (German: Femelschlag) cutting, tree harvest requires several interventions carried out over the same area and usually extending over several decades. As shelterwood cutting is always applied on the entire stand area, it creates relatively homogeneous ecological conditions, whereas in a "Femel" cutting, the initial tree removal is carried out in discrete areas (usually between 500 and 1000 m^2) and results in more heterogeneous resource conditions. In both cases, some mature trees are not removed during the initial interventions: in shelterwood cutting, the remaining trees are distributed over the entire area, and in a group shelterwood cutting, they can be found on both the discretely harvested and the untouched remaining areas of the stand. These mature trees reduce the resources available to seedlings (Ammer 2002; Petriţan et al. 2011). The effect of cuttings on canopy closure is mainly used to control competition within tree regeneration and to enhance stem quality of saplings. Von Lüpke and Hauskeller-Bullerjahn (2004) drew on the example of a very unevenly exposed 160-year-old beech stand (light availability at the forest floor ranging from 6% to 67% of open field conditions) to demonstrate the importance of canopy closure on the relationship between the height growth of oak and beech and thus the control of tree species composition in the tree regeneration layer. Five-year-old oak seedlings exposed to radiation above a certain threshold reached a greater height than did young beech of a similar age. However, only 3 years later oak needed a much higher threshold radiation level to outgrow beech, indicating that the amount of light required by oak to remain competitive with beech increases with age. An increase in the amount of light can be guaranteed by harvesting mature trees (Lüpke and Hauskeller-Bullerjahn 2004). Another important function of canopy density is that tree seedlings and sapling trees below the canopy of mature trees form fewer and thinner side branches because of the reduced light availability; this results in more trees with a straight and clear (i.e., with few and thin side branches) stem, meaning an increase in the future value of the timber from the stand (Weidig et al. 2014). Another positive effect of shading mature trees is that the establishment of ground vegetation which competes with tree regeneration is limited (Kuuluvainen and Pukkala 1989; Ammer 1996).

A method in which only a few individual trees are removed and in single interventions is the so-called plentering or single-tree selection (Fig. 14.10; Schütz 1994). By definition, in plentering, the growing stock of a stand should not exceed nor fall below a certain value, as either would result in the loss of the typical multi-layered structure of the stand. The disturbance regime corresponds to a large extent to the small-scale gap dynamics of natural mixed broadleaved forests made up of shade-tolerant tree species. Interruption of canopy closure presumably ensures the continuous regeneration of the stand.

The light and temperature conditions resulting from application of the plentering system are more uniform than those produced by the previously mentioned approaches for harvesting high forests (Burschel and Huss 2003; Ehbrecht et al. 2017). While even-aged stands, for example, resulting from the shelterwood system, give rise to a mosaic of different developmental phases and differ between stands but much less within stands on the landscape level, Plenter forests are characterized by a high within-stand heterogeneity but a low between-stand heterogeneity. It has been shown that the uneven-aged Plenter forests are economically feasible and less frequently damaged by storm events than even-aged forests (Knoke 1998). However, since conditions on the forest floor are relatively dark, the species diversity of various taxa (other than trees) was found to be lower on the landscape level when compared to the even-aged systems, which provide a greater diversity of abiotic conditions (Schall et al. 2018).

Old and large trees are crucial for biodiversity, as they frequently provide microhabitats (Vuidot et al. 2011; Larrieu and Cabanettes 2012), which are important habitats for rare species (Hofmeister et al. 2016). This is taken into account in the so-called retention method (Gustafson et al. 2007; Aubry et al. 2009), in which not all of the trees of a mature stand are harvested, but rather some are left permanently in the area, mostly in groups (approximately 10 trees ha^{-1}), and continue to mature. After their natural death, their positive effects on biodiversity remain in the form of standing or lying deadwood (Müller and Bütler 2010; Seibold et al. 2015). Thus, these trees are in part taken up by the next forest generation. This approach is more close to nature than cutting methods characterized by complete harvesting, and it can contribute to the preservation of the forest biodiversity that relies on old trees (Rosenvald and Lõhmus 2008; Fedrowitz et al. 2014).

Regeneration Measures The harvest of mature timber is usually accompanied by measures supporting tree regeneration. There are, for example, occasionally additional measures, many of which are a disturbance for the ground vegetation, including tillage and slash removal. While, especially in Scandinavia, soil preparation to encourage establishment of regeneration over large areas is common (Örlander et al. 1996), in the temperate zones of Europe, it is used only occasionally, for example, to support the natural regeneration of pine (Lehnigk and Ammer 2012). In the past, ploughing, such as conducted in agriculture, was carried out to encourage tree regeneration, whereas today only the humus layer is "scarified" (i.e., partly removed to expose the mineral soil). Slash removal is usually carried out after cutting procedures that produce a large quantity of wood per operation if the harvested area must immediately be made accessible and plantable, or for reasons of pest control (Lobinger 2006). Whereas the "slash" (i.e., the logging debris left after removal of stems during harvesting—foliage, smaller branches, etc.) was previously collected into large piles or strips in the stand, today it is occasionally removed and burned to generate heat and energy. Depending on the nutrient content of the soils, slash removal can involve a considerable loss of nutrients (Meiwes et al.

2008); consequently, slash removal is often regulated in certification guidelines. As a rule, however, the slash is left in the forest. Planting measures under the canopy of mature trees— to close gaps caused by disturbances through planting fast-growing (possibly non-native) tree species or to convert a pure stand of one species into a mixed-species stand by establishing shade-tolerant tree species—are not disturbances in a strict sense, but they do influence the response of a forest ecosystem to a disturbance.

14.6.4 Indirect Effects of Forest Management

Forest Road/Track Network In actively managed forest landscapes, there is a permanent network of paved, truck-compatible access roads as well as rough tracks within the stands. The creation of a paved road results in several disturbances: An inner edge of the forest is created, the groundcover is removed, road construction material is brought in, and a side ditch is usually dug to provide drainage for the road. In the root zone, the water flow is altered to some extent. In addition, the infrastructure also changes the water flow at the landscape level and has negative effects on running and still waters (Lindenmayer and Franklin 2002). In particular, on forest sites with a high groundwater table, roadside ditches serve as drainage for the adjacent forest stands. The smaller tracks may likewise act as drainage channels during heavy rainfall events and therefore accelerate surface runoff (Witzig et al. 2004). In acidic forest landscapes, roads made of alkaline limestone gravel frequently affect the flora adjacent to the road (Mrotzek et al. 2000). Furthermore, forest roads facilitate the distribution of ruderal species and neophytes, as their seeds (and other propagules) are transported by vehicles (Ehbrecht et al. 2017) and by game stopping along tracks to feed (Heinken et al. 2005).

Unpaved rough tracks have a less pronounced effect because they are narrower and no material is brought in and do not usually have associated ditches. However, the use of heavy machinery (such as harvesters and transport vehicles) may result in soil compaction, which for sensitive soils leads to a restriction of the oxygen supply, waterlogging, and a reduction of the nitrogen supply (Hildebrand 2008; Ehbrecht et al. 2017). Compaction of loamy and clayey soils or of acidic soils with low biological activity is almost irreversible (Ebrecht and Schmidt 2005; von Wilpert and Schäffer 2006). Ebeling et al. (2016) showed recovery of the soil structure of biologically active soils on limestone and on loess over sandstone 20 years after compaction caused by heavy machinery whereas sandy-loamy podzols showed little recovery over the same period.

Drainage Forest stands close to groundwater or on waterlogged sites are often continuously drained to allow their profitable management (see above). In the nineteenth and twentieth centuries, close-meshed drainage systems were frequently installed (Fig. 14.13). Until the 1980s, waterlogged sites in Germany were in many cases ploughed before planting (Dertz 1972; Fig. 14.14), but this practice has since then been discontinued for reasons of soil protection.

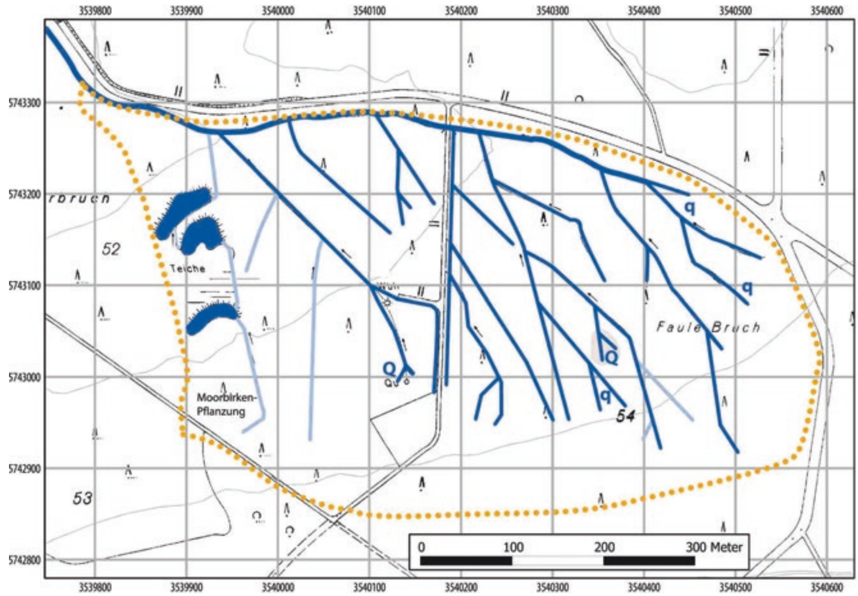

Fig. 14.13 The drainage system (blue lines) in a forest stand in the Solling Mountains, Lower Saxony (Germany). (Figure adapted from Küchler 2011)

Fig. 14.14 Ploughing using a disc harrow to prepare the conversion of a mire in the Reinhardswald (Germany) into a spruce stand. (Photo taken in 1928; photographer unknown; archive Prof. Dr. Gisbert Backhaus)

On peatland sites, drainage results in strong ground subsidence and peat mineralization (Kuntze 1993), and on mineral sites drainage induces large changes both in water and nutrient balances and in the species composition of the ground vegetation. On periodically wet soils, drainage reduces the extremes of water levels. A more consistent water supply can have a positive effect on the vitality of forest trees, by shortening the phases of oxygen deficiency in the topsoil. This may alter the balance of competition since tree species widely differ in their tolerance of waterlogging/flooding (Glenz et al. 2006). However, while drainage increases the growth success of planted saplings, extends the time window when soil-friendly timber harvesting can take place, and thus indirectly reduces soil damage, its effects are often negative with respect to carbon storage, landscape water balance, and nature conservation (Niemelä et al. 2005).

Fertilization and Liming Harvesting of whole trees in combination with nutrient losses caused by acidic nutrient inputs can cause nutrient levels to fall below the amounts needed for a sustainable nutrient supply (Stüber et al. 2008; Waldner et al. 2015). Until the 1980s, application of synthetic fertilizers in forests was not uncommon, and its effectiveness in increasing the vitality and productivity of forests had been demonstrated in numerous studies. From a nature conservation point of view, however, fertilization is a nonnatural measure that can abruptly change the nutrient balance and chemical conditions of the soil and thus cause an unnatural alteration of the species composition.

Compensatory liming can be used to counteract the effects of acid rain (Hüttl 1989). However, the previous practice of widespread liming has been replaced by a more targeted approach (Janssen et al. 2016). Although liming runs counter to the preservation of naturally nutrient-poor ecosystems (Reif et al. 2014), leads to nitrogen release (Kreutzer 1995), and causes changes in the ground vegetation (Schmidt 2002), it nonetheless can compensate for acids in the soil, induce biological activation of the mineral soil, and increase the supply of nutrients to trees (Grüneberg et al. 2017).

Pesticide Use The application of pesticides has been largely reduced in the forests of Central Europe (Kogan 1998; Ammer et al. 2011). However, if insect densities threaten the existence of the forest, insecticides can be applied to control an otherwise self-perpetuating disturbance and thus stabilize the forest development. Nevertheless, the use of pesticides remains controversial, especially because of their effects on nontarget organisms (Petercord and Lobinger 2010).

14.7 Effects of Forest Management at the Landscape Level

14.7.1 Changes in the Spatial Distribution of Forests

Deforestation leads to a fragmentation of forested areas and an increase in the amount of forest edge (Harris 1984). The effects of the latter are strongly dependent on the shape of the edge, the site conditions, and the tree species composition (Murcia 1995). From the interior of the stand toward the edge of the forest, the physical and ecological conditions increasingly take on the characteristics of open space. Investigations based on the microclimate and on water and nutrient balances have shown that this transition zone may be relatively wide (Keenan and Kimmins 1993; Klinck et al. 2013). Open-land species (Schmidt et al. 2011) may invade the previously closed forest stand. For species characteristic of closed forests, both the available habitat areas and the possibilities for dispersal are reduced. Consequently, a formerly large, spatially coherent population may disintegrate into subpopulations that are isolated from each other. In forest borders, the increased amounts of light and heat together with reduced competition among the trees lead to an improvement in the growth conditions of the shrub and herb layers and to increases in abundance and species diversity, especially of arthropods and birds (Reif and Achtziger 2000). The intersections of open-land and forest ecosystems at forest margins can act as dispersal axes for animal and plant species. Given the positive effects of forest margins on biodiversity, their targeted creation and maintenance may serve as important nature conservation measures (Coch 1995).

In addition to increased edge effects, deforestation also results in the increasing isolation of subpopulations and a reduction of habitat area (Schmidt et al. 2011). The latter has a larger impact on populations than does habitat isolation (Bailey 2007; Fahrig 2013). Deforestation also influences water and nutrient budgets and erosion processes within a landscape. For example, forest clearances in Central Europe during the Middle Ages led to an increase in groundwater levels, flooding, and a sharp rise in erosion (Bork et al. 1998; Ellenberg et al. 2010). The widespread loss of fertile soils that followed the St. Mary Magdalene's flood in 1342 is an impressive example of the potentially negative consequences of the agricultural use of what were previously forested landscapes (Bork and Kranz 2008).

Changes in land use can result not only in a decrease in forest area but also an increase in forest area. For example, the raised bogs of northwest Germany became forested after peat cutting and drainage. Forests have also established on post-mining landscapes and on previous military training grounds. Globally, the increasing concentration of human settlements and economic activity along with the intense land use in preferred locations have led to large areas of traditionally cultivated landscapes becoming fallow and subsequently developing into forests (Poyatos et al. 2003).

14.7.2 Landscape Effects of Stand Treatments

Studies in agricultural landscapes indicate that landscape effects are often more important for biodiversity than effects at the stand level (Gámez-Virués et al. 2015). In forests, the different cutting regimes (see Sect. 14.6) lead to the development of different landscape patterns. While the small-scale removal of single stems or groups of stems creates a fine-grained mosaic of different age groups, shelterwood or clear-cutting creates more coarse-grained patterns (Shifley et al. 2008). In a forest landscape with different forest owners and enterprises, the heterogeneity of targets and subsequent management concepts often lead to a high diversity of forest stands in terms of tree species composition, density, and structure (Gustafson et al. 2007; Schaich and Plieninger 2013).

A large-scale survey of older beech forests in a number of German state forest enterprises showed how the landscape pattern is influenced by the cutting regime (Meyer et al. 2016). Under natural conditions, the dominant climax community in many places would be closed old beech forests (Kaiser and Zacharias 2003; Meyer and Schmidt 2008), but today the stands of older beech forests are highly fragmented, and the canopy density is much reduced as a result of harvesting (Fig. 14.15). However, under current beech management practices, legacy trees are usually retained (see Sect. 14.6.3.1) thereby extending marginal zones and increasing the fine-grained character of the stand mosaic while reducing the isolation of habitats of species that depend on older forests.

Different species presumably react differently to the size and spatial distribution of the remaining forest (Fedrowitz et al. 2014). In the absence of reliable and locally

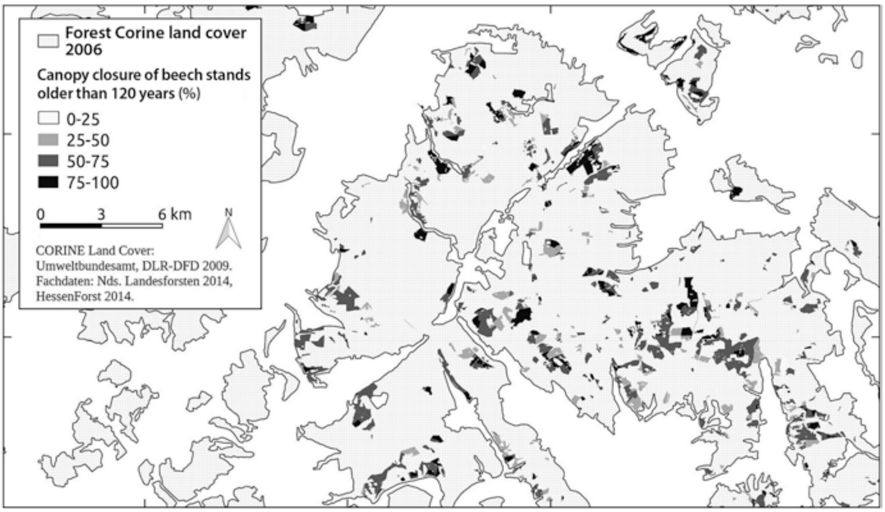

Fig. 14.15 Degree of canopy closure (%) and spatial distribution of >120-year-old beech forests in Solling, Lower Saxony (Germany)

transferable results, a broad range of silvicultural approaches is recommended with the integration of existing biodiversity hotspots as a key element (Meyer et al. 2015).

Overall, the application of a mixture of different silvicultural systems and protection concepts within a landscape is likely to be the best strategy for preserving biodiversity, as it maximizes the heterogeneity of environmental conditions. In their comparison of shelterwood and single-tree (plenter) cutting in beech forests, Schall et al. (2018) showed that the shelterwood systems resulted in a significantly higher diversity for 6 of 15 species groups (vascular plants, beetles, spiders, weavers, birds, and lichens) according to at least one diversity measure. Interestingly, for vascular plants and spiders, this trend was also significant when species restricted to forests were considered. There were no significant differences for bats, mosses, deadwood fungi, lacewings, Hymenoptera, bugs, ectomycorrhiza, and bacterial RNA, while the diversity of bacterial DNA was higher in the Plenter forest (Schall et al. 2018). A possible explanation for these findings is that single-tree harvesting creates relatively homogeneous structures at the landscape level. By contrast, in shelterwood-cut forests, there is little variation in the abiotic conditions within stands, whereas between the stands of different ages, the differences are significant.

14.8 Conclusions

Forest management is characterized by a large variety of disturbances, some of which are similar to natural disturbances. However, managed and unmanaged forest landscapes/stands differ considerably (Lindenmayer and Franklin 2002).

Understanding of the causes and consequences of natural and anthropogenic disturbances is essential for sustainable forest management and nature conservation. In the development of a strategy for the conservation of biological diversity, forestry must be guided by natural disturbances since many species and ecosystems depend on them for their continued survival (Spies and Turner 1999). However, as a direct blueprint, natural disturbances are rarely suitable since the objectives of forestry include economic and other societal interests as well as nature conservation (Lindenmayer and Franklin 2002). Natural disturbances can support but also impede those objectives. Disturbance ecology has enhanced our understanding of the different developmental pathways followed by natural ecosystems and the significantly altered conditions that may arise after random events (DeAngelis and Waterhouse 1987; Perry and Amaranthus 1997). Larger areas of disturbances are of crucial importance for the conservation of biodiversity. Ecologically oriented, close-to-nature forestry is therefore characterized not by uniform and small-scale interventions of varying intensity, but rather by a wide range of interventions of varying intensity and by the integration of essential elements of natural disturbance regimes.

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