Chapter 3
Natural Disturbances from the Perspective of Forest Ecosystem-Based Management

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Abstract Natural disturbances drive forest dynamics and biodiversity at different spatial and temporal scales. Forests in the boreal biome are shaped by several types of disturbance, including fire, windthrow, and insect outbreaks, that vary in frequency, extent, severity, and specificity. In managed forests, disturbances also affect the amount and quality of available timber. Ecosystem management uses information on disturbance regimes as a guide to finding a balance between ecological, economic, and social viewpoints. In this chapter, we review current knowledge on disturbance regimes in boreal forests and discuss some implications for managing the impact and risk of disturbances in the context of forest ecosystem management and restoration.

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Box. 3.1 Definitions of Terms Used in the Chapter

A *disturbance* is defined as a relatively discrete event that affects the structure of an ecosystem, community, or population and that modifies resources, substrate availability, or the physical environment (Pickett & White, 1985).

A *disturbance regime* consists of a combination of all characteristics generated by one or several disturbance agents acting within a given land area. Some principal descriptors related to natural disturbance regimes are listed below.

Intensity: the physical force of the event per area per unit of time (e.g., heat, wind speed)

Severity: the impact of the disturbance on an organism, community, or ecosystem (e.g., tree mortality)

Duration: the time (minutes to years) from the beginning to the end of a single disturbance event

Frequency: the proportion of area affected annually. Return interval = 1/frequency

Specificity: the selective nature of a disturbance agent toward one or several types of habitat or species

3.1 Fire

Fire is a dominant disturbance in circumboreal forests (Gauthier et al., 2015b), and it has been the basis for many emulation and restoration strategies. Circumboreal fire regimes are, however, highly variable (Buryak et al., 2003; Furyaev, 1996; Gromtsev, 2002; Rogers et al., 2015; Sofronov & Volokitina, 1990). In North American boreal forests, crown fires dominate (Rogers et al., 2015; Wooster & Zhang, 2004), although fire severity, i.e., the magnitude of the impact of fire on living plants and the soil organic layer, varies within and between events as well as within a fire season (April to October; Guindon et al., 2021). Eurasian boreal forests are shaped by mixed-severity fire regimes, where variation in fire severity is driven by climate and weather, vegetation, and characteristics of the soil and bedrock (Gromtsev, 2008; Sofronov & Volokitina, 1990; Valendik & Ivanova, 2001). Flammability is similarly dependent on the above factors and is often inversely related to severity, i.e., easily ignited areas

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are often subject to low-intensity surface fires. Flammability and fire severity can also be influenced by the occurrence of other disturbances, which affect the quality and quantity of fuel; for example, when poorly flammable forests are disturbed by an insect outbreak, they may become more flammable and, consequently, the resulting fire may display a crown-fire behavior.

As the characteristics of current fire regimes vary widely between forest regions in Canada and Eurasia, we detail below the regimes for Canadian and Eurasian boreal forests separately. We further distinguish the Fennoscandian boreal forests of Finland, Sweden, and Norway, as their respective fire regimes and fire-related management challenges differ greatly from the rest of the boreal zone.

3.1.1 Current Fire Regimes

3.1.1.1 Canada

The annual area burned varies markedly in Canadian boreal forests (Boulanger et al., 2014; Hanes et al., 2019). On average, 8,000 fires burn around 2 million ha of forest across the country each year (Gauthier et al., 2015a; Hanes et al., 2019). For fires larger than 200 ha (data covering 1959–2015), 85% were ignited by lightning (Hanes et al., 2019), whereas smaller fires may include a greater share of human-caused fires (Cardil et al., 2019). The regional annual burn rates (i.e., the fraction of the region that burns on average every year, compiled for 1959–1999) can vary from approximately 0.05% to 0.1% per year in northern and eastern regions to 1.5% per year in western and central Canada; this corresponds to return intervals of 2000 and 67 years, respectively (Boulanger et al., 2014). This relative interregional difference is expected to persist with climate change, whereas the total area burned is predicted to increase (Boulanger et al., 2014). Most fires are small, whereas a few large lightning-ignited fires are responsible for most of the area burned (Hanes et al., 2019).

In Canada, debates continue in regard to fire frequency and the influence of stand age, fuel types, and site conditions versus that of climate and weather (Bessie & Johnson, 1995; Cumming, 2001; Erni et al., 2018; Héon et al., 2014; Lefort et al., 2003). Under a given regional fire regime, deciduous forests are less likely to burn than coniferous ones (Bernier et al., 2016), and young and low-biomass forests are less likely to burn than older and high-biomass ones. In regions having the highest burn rate (1.5% per year), the return interval—the inverse of burn rate—is 66 years. Young (<30 years) deciduous forests and old (>90 years) coniferous forests have burn rates of 0.14% and 2.82% per year, respectively, whereas in regions experiencing the lowest regional burn rate (0.05% per year and a 2000-year return interval), the respective burn rate would be between 0.005% and 0.09% per year. As most future projections of fire burn rate are based only on future climatic conditions, accounting for this variation in fire selectivity can markedly change the outcome of projections, notably in areas where the burn rate is projected to be above 1% (Boulanger et al., 2017).

The occurrence of successive fires at short intervals may cause the regeneration failure of many tree species, contributing to a shift from a closed forest cover to an open woodland (Payette & Delwaide, 2018). This scenario occurs, for example, when young forests burn before a propagule bank—which can ensure post-disturbance forest recovery—has been constituted. With the projected increase in fire frequencies across Canada, regeneration failure may become more common in some stand types (Baltzer et al., 2021; Splawinski et al., 2019).

3.1.1.2 Russia

The annual area burned in Russia is considerably greater than that in North American boreal forests. In 2020, for example, 35,134 forest fires burned 16.44 million ha. However, there is a strong geographic gradient with over 90% of burned areas situated east of the Ural Mountains, i.e., in the Asian portion of Russia (Sofronov & Volokitina, 1990). In Siberia, 83% of fires occur in eastern Siberia and the Far East, whereas 17% occur in western Siberia. Western Siberia is characterized by a low-frequency fire regime. Variations in climate and vegetation drive these differences; frequent surface fires characterize the eastern part with easily flammable *light coniferous* forests dominated by Scots pine (*Pinus sylvestris* L.) and Siberian larch (*Larix gmelinii* Rupr. and *L. sibirica* Ledeb.) (Buryak et al., 2003; Korovin, 1996), whereas the western region burns less intensively, consisting mostly of *dark coniferous* forests dominated by Siberian spruce (*Picea obovata* Ledeb.), Siberian fir (*Abies sibirica* Ledeb.), and Siberian pine (*Pinus sibirica* Du Tour.). Climate also imposes a latitudinal gradient within the Siberian region; for example, in Siberian larch forests, the mean fire return interval increases with latitude, from 80 years at 64°N to about 200 years near the Arctic Circle and about 300 years near the northern range limit of larch forests (71°N) (Kharuk et al., 2016a). Among vegetation types, recently harvested southern boreal forests are considered as the most flammable of all Siberian forests, mainly because logging slash burns easily (Valendik et al., 2013). These differences in fire frequency are inversely related to fire severity; forest types that burn often are mostly subject to surface fires, characterized by low fuel loads and tree species adapted to survive frequent fires. However, fire severity varies greatly even within a given landscape type (Fig. 3.1a). Whereas surface fires are generally more common than crown fires, patchy crown fires can represent 50% of the total area burned during severe fire seasons (Belov, 1976; Valendik & Ivanova, 2001).

West of the Ural Mountains, in the European boreal forests of Russia, fire return intervals can vary among landscapes from 40 to more than 200 years, depending on site conditions (Melekhov, 1971; Zyabchenko, 1984), dominant tree species, landforms, and bedrock (Gromtsev, 2008). The variation in natural fire regimes is driven by differences in superficial deposits and topography that create a landscape mosaic with varying flammability and fuels. Similar to the Siberian part of the boreal forest, pine-dominated forests burn with a higher frequency, typically as surface fires but occasionally as crown fires. The most fire-prone pine forests tend to burn at least twice per century as surface fires and three to four times per millennium as crown

Fig. 3.1 Varying fire severity in boreal forests. **a** Six years after a mixed-severity fire of ca. 4,000 ha in the dark coniferous forests of the Eastern Sayan Mountains, Siberia, Russia; **b** crown fire in the coniferous boreal forest of eastern Canada; **c** patchily burned area six years after a surface fire in the northern boreal primeval rocky Scots pine forest, Karelia, Russia; **d** burning for ecological restoration in a southern boreal Scots pine forest, Finland. *Photo credits* **a** Ilkka Vanha-Majamaa, **b** Société de Protection des forêts contre les feux (SOPFEU), **c** Daria Glazunova, **d** Erkki Oksanen/LUKE archive

fires, whereas the less fire-prone pine forests burn with higher severity as crown fires, one to three times every 300 years (Gromtsev, 2008). Forests dominated by Norway spruce (*Picea abies* (L.) H. Karst.) burn as either intense crown fires triggered by severe droughts at a mean return period of once or twice per millennium or as more frequent but lower-severity ground fires (Gromtsev, 2008).

Fires in Russian boreal forests are ignited by lightning strikes or by humans. In Siberia, the occurrence probability of lightning-ignited fires varies with the type of terrain (Shishikin et al., 2012). In European Russia, where there is a higher human population density and easier accessibility to the forest than in Asian Siberia, humans are responsible for igniting more than 65% of fires (Conard & Ivanova, 1997; Shishikin et al., 2012); however, regional variation in the causes of ignition is great. For example, in northern larch stands, about 90% of wildfires are of natural origin (Ivanova & Ivanov, 2004), whereas in southern boreal forests, notably in the Khakasia region, 80% of fires are caused by campfires, the burning of logging slash on harvested areas, and the agricultural burning of grasses (Shishikin et al., 2012).

3.1.1.3 Norway, Sweden, and Finland

The boreal part of Fennoscandian forests outside Russia (i.e., Norway, Sweden, and Finland; NSF) has a natural fire regime resembling that of the adjacent Russian Fennoscandia, where diverse landscape conditions result in variable fire regimes in terms of fire frequency, size, and severity (Engelmark, 1987; Gromtsev, 2008). However, the current fire regime differs significantly from that of the boreal forest of European Russia and from the more active fire regime of the past because of human influence (Pinto et al., 2020; Rolstad et al., 2017). Forest fires were previously common in all three countries but have declined considerably in frequency from historical levels 150–250 years ago (Chap. 2; Rolstad et al., 2017; Wallenius, 2011). These changes were not associated with climatic shifts (Aakala et al., 2018; Rolstad et al., 2017) but rather with changes in cultural practices and land tenure. The mechanization of firefighting and the development of a dense forest road network have also influenced the efficacy of the active suppression of surface fires (Wallenius, 2011).

The number of fires and the area burned is currently small, having declined dramatically during the twentieth century. For example, from 2007 to 2016, the average annually burned area was 496 ha in Finland, 842 ha in Norway, and 2,876 ha in Sweden, corresponding roughly to 0.002, 0.007, and 0.01% of the forested area of each country (San-Miguel-Ayanz et al., 2018). The area burned annually, however, varies considerably. A notable example is the two peak fire years in Sweden in 2014 and 2018, during which 12,600 and 22,400 ha burned, respectively (MSB, 2020). Although human influence on fire ignition has declined over the last century, most fires are still ignited by humans.

3.1.2 Fire and Forest Management

Fires and forest management are linked in various ways, including fuel management and the use of fires to guide forest ecosystem management. Fire suppression strategies and forest fire policies differ markedly around the circumboreal region, as does the role of fires in managing forest ecosystems.

3.1.2.1 Fire Suppression Policies and Practices

In Canada, fire management agencies have been established in every province and territory. In regions where forest management licenses are active, these agencies aim to minimize the number of large fires and their adverse effects on people, property, and timber (Stocks, 2013; Stocks & Martell, 2016). With early fire detection systems to locate small fires, e.g., infrared satellite and aerial flyover monitoring, and the use of initial attack strategies to contain fires, the aim is to extinguish fires at a small final size (2–4 ha; Martell & Sun, 2008). Despite such fire management systems in

place, slightly more than 3% of fires become larger than 200 ha and are responsible for almost 97% of the area burned (Hanes et al., 2019). Cardil et al. (2019) showed that fire suppression success is greater in regions of mixed boreal forest because of the presence of deciduous species that are less flammable than resinous ones, with 82% to 92% of fires extinguished before they reach 3 ha compared with 53% to 77% in regions of coniferous boreal forest. Large fires occur on extreme fire-weather days when fuels are dry and winds favor fire spread. Since 1959, the area burned by larger fires has increased on average by about 350 ha per year (Hanes et al., 2019). By 2100, the annual area burned in Canada is projected to increase by two to four times (Boulanger et al., 2014; Coogan et al., 2019). Such situations may overwhelm the capacity of fire management agencies (Wotton et al., 2010) and result in a substantial increase in fire management expenditures (Hope et al., 2016).

In Russia, forest fire monitoring is based chiefly on satellite-derived information compiled and analyzed by the Federal Forest Service (since 2001) and on reports from the Federal State Agency "Central Base for aerial forest protection Avialesookhrana." It should be noted that up to 100 million ha of unused agricultural lands are now overgrown by forests in Russia. Such forests represent approximately 10% of all forests in Russia, although they are not officially referred to as forests (Shmatkov & Yaroshenko, 2018). Because these patches do not have an official forest status, fires within these forests are not classified as forest fires and are not officially monitored, with a notable exception being volunteer monitoring organized by Greenpeace.

Regional forest fire centers and aerial forest protection offices are responsible for forest fire protection in Russia; however, these organizations lack resources, both in terms of labor and equipment. During severe fire situations, local forest organizations—including forest companies—are obliged to participate in extinguishing fires. Forest fire protection in protected areas, urban forests, and military forests is organized respectively by the staff of protected areas, regional authorities, and the Russian Ministry of Emergency Situations. Responsibility for fire protection on former agricultural lands has not been allocated to any entity, except in cases deemed as "high emergency."

Currently, fires in about 45% of Russian boreal forests are not extinguished because of their remoteness and often low accessibility. Regional authorities define these *control zones* without explicit, law-based principles. The proportion of fires left to burn without intervention by firefighters in the spring–summer of 2020 varied regionally and monthly, ranging between 50 and 98% of the total number of forest fires.

In NSF, the detailed implementation of fire detection and suppression differs among the countries; however, the overall aim is to actively suppress all fires. Satellite detection and reconnaissance flights are used for the early detection of fires. Given that most forest areas are easily accessible because of the dense network of forest roads, fire suppression is generally efficient, which is reflected by the limited area that burns annually (see above).

3.1.2.2 Fire as a Driver for Forest Management

At a global level, ecosystem management and habitat restoration increasingly seek inspiration from natural disturbances, particularly fire, to support both an economically viable forest industry and biodiversity in managed forests. Four main descriptors of fire regime (annual burn rate, fire size, severity, and, more recently, specificity) form the basis of ecosystem management and restoration in boreal forests (Gauthier et al., 2009; Koivula & Vanha-Majamaa, 2020; Lindberg et al., 2020; Shishikin et al. 2012). In Canada, the annual burn rate helps define management targets in terms of even-aged vs. uneven-aged forests within landscapes. The variation in annual burn rates also strongly influences the amount of old-growth forest present in a given landscape (Bergeron et al., 2001; Weir et al., 2000), which in turn influences forest composition at the landscape level. Therefore, the amount of old-growth forest to be maintained in different boreal regions is defined on the basis of past fire regimes (Bouchard et al., 2015; DeLong, 2007). Fire-size distribution provides insights into the spatial configuration of different forest types and ages across the landscape (Gauthier et al., 2004; Perron et al., 2009), whereas variation in fire severity influences the retention strategies applied in harvested areas.

In Russia, the mosaic of forest patches and landscapes stemming from variable fire return intervals and burn severities (Kharuk et al., 2016a) leads to differences in the economic and nature conservation value of forests. Whereas emphasis has been traditionally placed on fire suppression, there are now calls for region-specific forest fire management policies. Fire policies should recognize the beneficial functions of fire for pyrophilous and deadwood-dependent species as well as its role in forest successional processes (Furyaev, 1996). This shift would imply replacing the current fire exclusion policy with a policy that allows for natural low-intensity fires and prescribed burning to reduce fire hazards and promote biodiversity (Davidenko et al., 2003; Goldammer, 2013). From a biodiversity perspective, implementing fire management strategies in protected areas, where actions may vary from fire prevention and suppression to doing nothing, is of particular importance (Kuleshova, 2002; Shishikin et al., 2012). Regional policy guidelines for fire should be based on the scientific knowledge of the (1) landscape-specific fire regimes; (2) regional- and landscape-specific effects of fire and postfire succession on biodiversity; and (3) socioeconomic conditions, including human population density, road networks, economic factors, agricultural use of fire, and forestry activities. Depending on the region or landscape, different strategies can be prescribed: (1) fire prevention, e.g., through establishing fire breaks and education; (2) suppression, including the control, monitoring, and fighting of fires whenever possible; (3) localization of ignited fires; (4) controlled or prescribed burning; and (5) regulation of postfire successional processes by applying different restoration measures. Multilevel educational actions designed for target groups from preschool children to university students and local people also play an important role in fire management (Kuleshova, 2002; Shishikin et al., 2012).

In NSF, there have been a few instances in which fire ecology has been used as a guide for developing sustainable forest management strategies; these are similar to the

Canadian ecosystem management approach. Perhaps the most well-known approach in the European context has been the ASIO model (Angelstam, 1998), which divides landscapes into four categories according to how frequently the forest burns under natural conditions: rarely (Aldrig), seldom (Sällan), infrequently (Ibland), or often (Ofta). In the mixed-severity fire regime, this frequency is often inversely related to fire intensity. The idea is that the forest management strategy applied in a given area is tailored according to the category in which the area is classified, emulating the stand age structure that would naturally occur in the area. This model has been applied in forest management planning by some large forest owners in Sweden (Angelstam, 1998). In Finland, ASIO has been used in conjunction with the landscape ecological planning of public lands. However, its role has been small and limited primarily to identifying parts of the landscape that almost never burn (Karvonen et al., 2001). Recently, Berglund and Kuuluvainen (2021) outlined a refined version of the ASIO model relying on an improved understanding of how forest fires shape the boreal forests of NSF.

In practical forest management in NSF, understanding fire ecology has been more commonly used as a silvicultural tool and for managing biodiversity rather than as a management template or guideline. The use of fire in silviculture was common after the 1950s when prescribed burning following clear-cutting was widely used as a regeneration tool. However, controlled prescribed burning is expensive; therefore, its popularity has declined. For example, the area of annual prescribed burns in Finland was around 35,000 ha in the 1950s, whereas it is currently only a few hundred hectares (Lindberg et al., 2020). As a consequence, habitats and structures previously maintained by frequent fires—mostly early successional habitats with abundant legacies such as burnt wood—have greatly declined (Kontula & Raunio, 2019). These types of fire-dependent habitats are currently being created by the prescribed burning of single or groups of retention trees in clear-cut areas—used to promote biodiversity and soil preparation—and through restoration burning (Lindberg et al., 2020). However, despite the benefits of fire for biodiversity, the areas burned annually in NSF remain small relative to the past natural fire regime.

Over the years, salvage logging of burned areas has globally gained importance to compensate for the impact of fire on timber availability (Nappi et al., 2004; Thorn et al., 2018). This practice negatively impacts the diversity of species occupying postfire forests, adding to the negative impacts of fire on diversity (Cobb et al., 2011). Habitat conditions, e.g., shadiness, and associated species communities appear altered less by insects than by fire or windthrow, whereas subsequent salvage logging renders these environments similar. Moreover, salvage logging reduces forest-species richness more in insect-disturbed than in fire- or windthrow-disturbed forests—reductions to about 57% versus 70–75% from the post-disturbance level (Thorn et al., 2020). Guidelines for retention within salvage logging areas have been proposed for reducing the negative impacts of such practices (Nappi et al., 2011; Thorn et al., 2020).

3.2 Wind

3.2.1 Susceptibility to Wind Damage

Windthrow occurs when wind speed is strong enough to override tree-root resistance to uprooting or stem resistance to trunk breakage. Wind is a common disturbance in a variety of biomes, from boreal (Ulanova, 2000) to temperate (Canham et al., 2001; Fischer et al. 2013) to tropical forests (Putz et al., 1983). Wind-induced disturbances vary in frequency, size, and severity both between and within biomes (Everham & Brokaw, 1996). In different parts of the boreal forest, windthrow return intervals vary from decades to a few hundred or thousand years (De Grandpré et al., 2018; Smolonogov, 1995; Waldron et al., 2013).

Wind damage can recur regularly at small scales and low severity or occur less often but at a large scale and high severity (Miller, 1985). Severe damage is associated with infrequent major storms and has led to significant efforts to document damaged areas and timber losses (Grayson, 1989; Ruel & Benoit, 1999; Valinger et al., 2014). Nonetheless, although less spectacular and often poorly documented, small-scale windthrow events can have significant consequences for forest management (Rollinson, 1987). In a compilation covering 29 European countries, Seidl et al. (2014) estimated that wind damaged 32.3 million $m^3 \cdot yr^{-1}$ of timber during the first decade of the twenty-first century.

Windthrow severity is influenced by interactions between wind speed, topographic and edaphic conditions, disturbance history, and the current characteristics of forest stands and landscapes (Everham & Brokaw, 1996; Ruel et al., 1998; Saad et al., 2017). Shallow and poorly drained soils restrict rooting depth, which leads to lower tree resistance to uprooting. However, soil properties interact with tree species and stand attributes; for example, jack pine (*Pinus banksiana* Lamb.) is more resistant than black spruce (*Picea mariana* (Mill.) BSP) on relatively deep soils but not on shallow soils, which prevent the development of deep roots. On the other hand, black spruce is inherently shallow-rooted and is thus better adapted to shallow soils (Élie & Ruel, 2005). Old-growth Norway spruce–dominated stands on rich soils consist of large trees with flagged crowns and a shallow root system. Consequently, they are more vulnerable to windthrow than, for instance, Scots pine or birch forests that are more deeply rooted (Karpachevsky et al., 1999; Skvortsova et al., 1983; Ulanova, 2000).

The most important characteristics influencing stand vulnerability to windthrow are tree species composition, size, and age structure. Tree-pulling studies allow a quantitative comparison of species resistance to windthrow (Achim et al., 2005; Nicoll et al., 2006; Peltola et al., 2000). Wood properties and the presence of decay (notably because of *Heterobasidion* fungi) strongly influence the resistance of trees to stem breakage (Rich et al., 2007). In eastern Canada, for instance, balsam fir (*Abies balsamea* (L.) Mill) has been consistently ranked as the most windthrow-prone tree species in large part owing to a high level of decay (Ruel, 2000). Among European boreal tree species, Norway spruce is the most sensitive to uprooting, whereas aspen

(*Populus tremula*) is damaged mainly by stem breakage (Skvortsova et al., 1983). In primeval European boreal forests, susceptibility to windthrow decreases with tree age structure from even-aged to all-aged stands, increases with the proportion of deciduous species, and decreases with site fertility (Fedorchuk et al., 2012; Karpachevsky et al., 1999; Shorohova et al., 2008).

3.2.1.1 Windthrow Impacts

Immediate and long-term windthrow impacts include (1) an abrupt or continuous change of forest structure with an increased share of broken and/or uprooted trees and deadwood; (2) pedoturbation (soil-mixing) with the creation of pit-and-mound systems (Fig. 3.2); (3) a change in microclimate; and (4) a change in stand vulnerability to subsequent disturbances (Chap. 4; Fischer et al., 2013; Šamonil et al., 2010; Schaetzl et al., 1989; Skvortsova et al., 1983; Ulanova, 2000). In old-growth forests, pit-and-mound systems may cover an area of up to 90% and remain visible for up to 200–500 years. In high-severity windthrow, environmental conditions, notably light availability and soil moisture, are strongly modified, and water balance can even change across the entire landscape (Karpachevsky et al., 1999). At the landscape scale, low- or moderate-severity windthrow results in a scattered pattern tree mortality of various modes (uprooting, stem breakage, or the formation of snags) (Fig. 3.3), a complex fine-scale mosaic of living and dead trees, and windthrow gaps that vary from 0.05 ha to a few hectares in size and have a variable pit-and-mound topography (Fedorchuk et al., 2012; Schaetzl et al., 1989; Shorohova et al. 2008; Skvortsova et al., 1983).

Spatial patterns of wind-induced tree mortality lead to multiple post-windthrow successional pathways in forest ecosystems, depending on the interplay between windthrow severity and stand attributes, including tree age structure, tree species composition, and site productivity (Meigs et al., 2017). Biotic and abiotic factors influence the succession of post-windthrow regeneration (Fischer et al., 2013; Girard et al., 2014; Ulanova, 2000). Coniferous tree species successfully regenerate where less than 60% of trees die in a stand (Petukhov & Nemchinova, 2015) and the surface area of windthrow pits covers less than 15% (Ulanova & Cherednichenko, 2012).

3.2.2 Wind and Forest Ecosystem Management

Windthrow generates timber loss due to falls and wounds on trees, and windthrow often results in subsequent biotic disturbances, such as bark beetle outbreaks or fungal infestations. In mountain regions, windthrow may increase the risk of avalanche and rockfall and consequently threaten human settlement and infrastructure (Schönenberger et al., 2005). Although extended rotation and partial cutting are important ecosystem management strategies (Bélisle et al., 2011; Montoro Girona et al., 2016), increasing rotation length can lead to more windthrow. Thus, wind damage
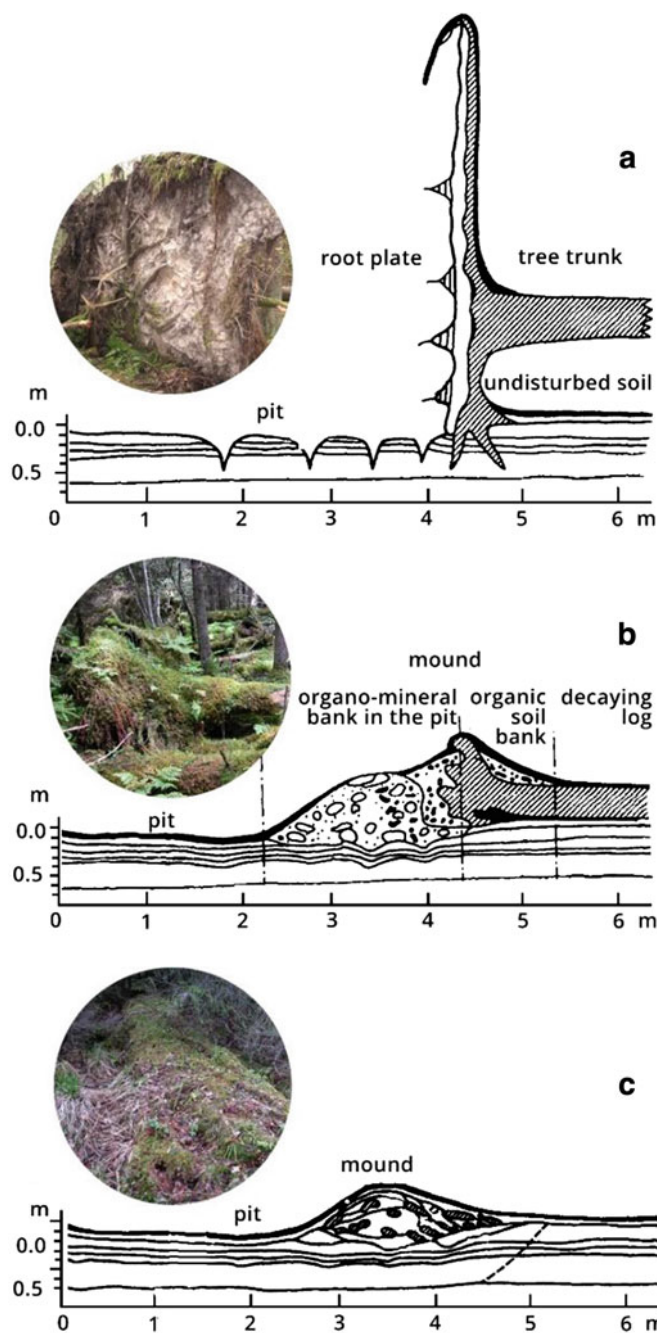
Fig. 3.2 Pit-and-mound complexes at **a** 20 to 30 years, **b** 50 to 60 years, **c** more than 100 years post-windthrow in southern boreal forests. Modified from (Ulanova, 2000) with permission from Elsevier. *Photo credits* **a** Aleksandr Gladyshev, **b** Ilkka Vanha-Majamaa, **c** Anna Ruokolainen

Fig. 3.3 Diverse coarse woody debris after a patchy windthrow in a primeval forest in the Vepssky Reserve, Russia. *Photo credit* Alexandr Korepin

tends to increase with tree age because of the increased tree height and incidence of decay (Ruel, 1995), posing a constraint when applying extended rotations to short-lived species. Increasing intertree spacing through partial cutting heightens the wind load on trees (Gardiner et al., 1997). There are many examples of increased wind damage after partial cutting (Chap. 16; Becquey & Riou-Nivert, 1987; Cremer et al., 1982; Montoro Girona et al., 2019; Ruel & Gardiner, 2019). Windthrow losses can, however, be minimized through windthrow-hazard evaluation tools, including decision keys, empirical models, or hybrid/mechanistic models (Gardiner et al., 2008). In recent decades, progress has been made in developing such tools, especially in the modeling of complex stands (Gardiner et al., 2008; Hale et al., 2012).

In post-disturbance situations, forest management strategies include (1) allowing natural successional development; (2) salvage logging followed by natural regeneration; and (3) salvage logging with subsequent soil preparation and tree planting (Brang et al., 2004; Fischer & Fischer, 2012; Fischer et al., 2002; Lässig & Močalov, 2000; Močalov & Lässig, 2002; Schönenberger, 2002; Soukhovolsky et al., 2012). A study from a Bavarian national park in Germany, comparing successional dynamics after windfall on permanent plots, demonstrated that salvage logging triggers natural secondary succession through intermediate phases having a dominance of birch or aspen (Fischer et al., 2002). The costliest silvicultural treatments allow for the regeneration of mixed conifer–deciduous forests, although without predisturbance natural mosaics (Fischer & Fischer, 2012; Lässig & Močalov, 2000). In mountain forests, "doing nothing" may ensure natural protection against snow avalanches and rockfall (Schönenberger et al., 2005).

The natural landscape–specific regime of wind disturbance can be considered as a basis, or a reference, for ecosystem-based forest management and ecological restoration. In landscapes dominated by small- and medium-scale windthrow, gap felling or variable retention felling can be recommended (Koivula et al., 2014). If salvage logging must be used for some economic or public safety reasons, post-windthrow attributes of known ecological importance, such as deadwood, living trees, and microtopography, should be retained within salvaged cutblocks, with some proportion of windthrow exempted from logging operations (Thorn et al., 2020; Waldron et al., 2013). Mimicking partial windthrows in wind-prone forests by conducting partial cuts can increase the likelihood of subsequent wind damage. However, a widely shared opinion among foresters in Finland is that damage risks are generally higher in even-aged than in uneven-aged management regimes, with the notable exception of root-rot infestations in Norway spruce forests (Nevalainen, 2017).

An additional challenge in incorporating wind disturbance into forest management is related to alterations of future disturbance regimes. Storms characterized by high wind speeds are more common in autumn and early winter in northern Europe and eastern Canada, periods where the frozen topsoil "anchors" trees in the ground, thereby decreasing the chances of treefall. Because of climate warming, however, periods of unfrozen soil are predicted to lengthen, resulting in a poorer anchoring of trees in a season of severe winds. Moreover, the frequency of autumn or early-winter windstorms may increase; thus, windstorm-caused timber damage could become more common and widespread (Gregow et al., 2011; Saad et al., 2017). Indeed, in Europe, the level of damage by wind, reported by Seidl et al. (2014) for the first decade of the twenty-first century, increased 140% compared with wind damage between 1971 and 1981. Between 1950 and 2000, more than 50% of natural tree mortality in Europe was due to windthrow, whereas biotic factors were responsible for 16% (Schelhaas et al., 2003). Although biotic factors appear relatively minor from this perspective, they can be locally devastating (Hlásny et al., 2019). These percentages are likely to change in the near future, however, as windthrow, drought, and insect outbreaks are predicted to increase, particularly for the boreal region (Seidl et al., 2020).

3.3 Insects

3.3.1 Insect Outbreaks and Their Characteristics

Forest insect outbreaks occur in all major forest ecosystems throughout the world but cause the most damage in high-latitude forests. Unlike fire and wind disturbances, insects are often specific in nature, such that only a limited number of host-tree species—usually a single genus or family—are affected (Bentz et al., 2020). This specificity also implies that certain attributes (frequency, size, severity) used to characterize fire regimes do not apply directly to insects (De Grandpré et al., 2018). For

example, although insect outbreaks may affect a larger total area than fire or windstorm, being specific to certain host-tree species, qualities, and sizes, these events lead to partial mortality except in pure host-species stands (Raffa et al., 2015). Thus, the losses of timber volume may be less than after windthrow or fire (Kneeshaw et al., 2015).

Contrary to wildfire, the return interval (the inverse of frequency) for insect outbreaks is usually calculated on the basis of insect population dynamics rather than the time required to affect a given area. Spruce budworm (*Choristoneura fumiferana*) outbreaks in eastern Canada occur every 30 to 40 years (Jardon et al., 2003; Morin et al., 2009; Navarro et al., 2018), a return interval similar to that of the mountain pine beetle (*Dendroctonus ponderosae*) (Alfaro et al., 2010). For insect species usually affecting only small areas, population return intervals are rarely calculated. Examples of such species include the European spruce bark beetle (*Ips typographus*), the gypsy moth (*Lymantria dispar*), and the oak processionary moth (*Thaumetopoea processionea*) (Bentz et al., 2020).

The severity of an insect outbreak can be expressed as the number or proportion of infected trees. Aerial surveys of areas affected by the spruce budworm give stand-level severity estimates based on annual defoliation. In Québec, these classes are 0–33% (light), 34–66% (moderate), and 67–100% (severe) (MFFP, 2019). If defoliation is less than 33%, tree growth is minimally affected (Chen et al., 2017). As the spruce budworm only eats current year (new) foliage and trees carry five to seven years of foliage, multiple subsequent years of infestation are required for the spruce budworm to kill a tree (Lavoie et al., 2021). For example, removing all foliage on a tree requires five years of 100% defoliation of new foliage, although a tree may die before the cumulative defoliation reaches 500%. This rule of thumb is useful for translating defoliation into mortality. Severity has also been measured through dendrochronological records by inspecting reductions in tree growth rings (Robert et al., 2018; Thomas et al., 2002).

Tree mortality is another useful indicator of outbreak severity. In mild outbreaks, only growth reduction may occur, whereas severe outbreaks result in detectable tree mortality. There is no accepted standard of the level of mortality required for an outbreak to be considered severe. The mountain pine beetle, for example, feeds on the phloem of living trees but can only successfully reproduce if it kills the tree and eliminates its defenses (Safranyik et al., 2010). In contrast, many other insect species can reach high population numbers (and thus outbreak conditions) while primarily affecting only tree growth. The forest tent caterpillar (*Malacosoma disstria*), the jack pine budworm (*Choristoneura pinus*), and the oak processionary moth, for example, rarely directly kill their host trees (Man & Rice, 2010; Sands, 2017). Thus, outbreaks causing any mortality may be considered severe for these species (Cooke et al., 2012).

Given the host specificity of herbivorous insects and their feeding preferences (defoliation of some or all leaves versus feeding on phloem or xylem), insects cause various forms of damage to trees. Hence, forest management based on the emulation of tree structure and microclimatic conditions resulting from insect disturbances must focus on parameters other than the impacted area or return interval. For the

spruce budworm, Baskerville (1975) suggested that the insect acts as a *super silvi-culturist* in releasing advance, i.e., pre-established, regeneration. Bouchard et al. (2006), Kneeshaw and Bergeron (1998), Reinikainen et al. (2012), and Burton et al. (2015) showed that outbreaks of defoliators are essential for maintaining the structural diversity of forests. Other authors have also evaluated the influence of insects on tree regeneration and, therefore, the future composition of forests within various site types. The mountain pine beetle, for instance, can act as an agent that removes and kills large older lodgepole pine (*Pinus contorta* Douglas ex Loudon) and, in turn, releases space and resources for the smaller stems of lodgepole pine or favor the recruitment of other tree species (Kayes & Tinker, 2012).

3.3.2 Forest Ecosystem Management and Insect Outbreaks

Lessons from insect outbreaks suggest that if forest management aims to emulate tree structures resulting from these outbreaks, forest managers should avoid monocultures and even-tree-size stands and favor tree diversity. These features would benefit wildlife diversity and decrease the likelihood of future outbreaks, as suitable host trees for these specialists would be less abundant. Koivula et al. (2014) suggest that partial cutting could emulate insect disturbances as most insect disturbances cause only partial mortality. Currently, forest managers preferentially harvest the most valuable companion tree species at maturity (Blais, 1983; Kneeshaw et al., 2021; Sonntag, 2016). Recent work suggests that insects tend to attack large contiguous blocks of host-tree species with greater synchrony and severity; therefore, breaking up such large blocks may be an effective pest management strategy at the landscape scale (Robert et al., 2012, 2018, 2020). As the ranges of many insects are currently expanding, managers should be aware that large blocks of monocultures should be eliminated or reduced to avoid increasing forest vulnerability to outbreaks (Kneeshaw et al., 2021).

Climate change may affect the population dynamics of different insect species, alter outbreak frequencies, and facilitate range shifts to more northern latitudes and higher elevations. Range expansions of forest insect pests may lead to widespread mortality of trees within the insect's new range. However, they may also be associated with contractions in other parts of the range (Régnière et al., 2012). Insect population density is regulated by density-dependent and density-independent factors, such as weather conditions and forest ecosystem characteristics (Isaev et al., 2017). Increases in temperature, especially in winter months, and drier conditions may contribute to increases in bark beetle populations and the ability of these beetles to overcome the defense mechanisms of trees (Raffa et al., 2015; Romashkin et al., 2020). Droughts have also been implicated by stressing trees and rendering them more vulnerable to bark beetle attacks, as has been observed for European spruce bark beetle outbreaks (Maslov, 2010). However, drought effects on defoliators remain equivocal (Itter et al., 2019; Kolb et al., 2016). Recent reviews have attempted to predict the effects of climate change on future insect outbreaks (Jactel et al., 2012; Kolb et al., 2016;

Pureswaran et al., 2018). These studies indicate that, despite expectations of greater outbreaks, responses are complex, and positive and negative feedback will probably occur (Haynes et al., 2014). In other words, some outbreaking insects may cause more damage whereas others will cause less, and this—combined with range contractions and expansions—adds much uncertainty to projections of future insect influence on forests.

3.4 Pathogens

Many pathogens influence trees by reducing tree growth and vitality (Hicke et al., 2012) by acting as predisposing agents to a number of other disturbances. Several pathogen species also kill trees directly. Because of its harsh climate, the boreal zone has previously been beyond the distribution of many pathogens. Consequently, their role in the disturbance regimes of natural forests has been overlooked. Certain species of fungi may play a significant role in the dynamics of old-growth forests in northern Fennoscandia (Lännenpää et al., 2008) in causing the small-scale mortality of individual trees or small groups of trees. Hence, at the landscape scale, pathogens occur frequently, but their impacts are of low severity and spatially scattered.

Many pathogen species are strictly host specific (Zhou & Hyde, 2001). Partly because of this host specificity, their role in intensively managed, monospecific, and structurally homogeneous forests appears greater than in natural forests (Storozhenko, 2001). However, trees in continuous-cover forest management appear to suffer from *Heterobasidion* infestations to a greater degree than those growing in standard even-aged management because of logging-caused damage to retained trees (Piri & Valkonen, 2013) and difficult root and stump removal. Fungi of the genera *Heterobasidion* and *Armillaria* are considered particularly problematic for forestry in the boreal zone (Garbelotto & Gonthier, 2013); as they spread through roots, trees in the next generation are easily infected.

The most aggressive fungal pathogen causing root rot in naturally regenerated coniferous boreal forests is *Armillaria borealis* Marx. & Korh. (Pavlov, 2015). Soil conditions determine the activity of and disturbance severity caused by *Armillaria* and *Heterobasidion* spp. (Fig. 3.4; Pavlov, 2015).

In European Russia, the bacterial dropsy diseases on birch (*Betula* spp.) and coniferous tree species, caused by *Erwinia multivora* Scz.-Parf., have increased during the last decades (Voronin, 2018). These bacterial diseases are triggered by drought and anomalous thaw events, causing fungal outbreaks in Siberian fir and pine forests (Voronin, 2018).

Climate is an important driver of disease outbreaks, influencing the disturbance agent directly or indirectly through host susceptibility (Sturrock et al., 2011). Changing climate may generate conditions favorable to pathogens by extending periods of growth and reproduction or causing phenological changes that may result in a greater overlap of host susceptibility and pathogen aggressiveness. *Heterobasidion* and *Phytophtora* species are expected to benefit from a warming climate
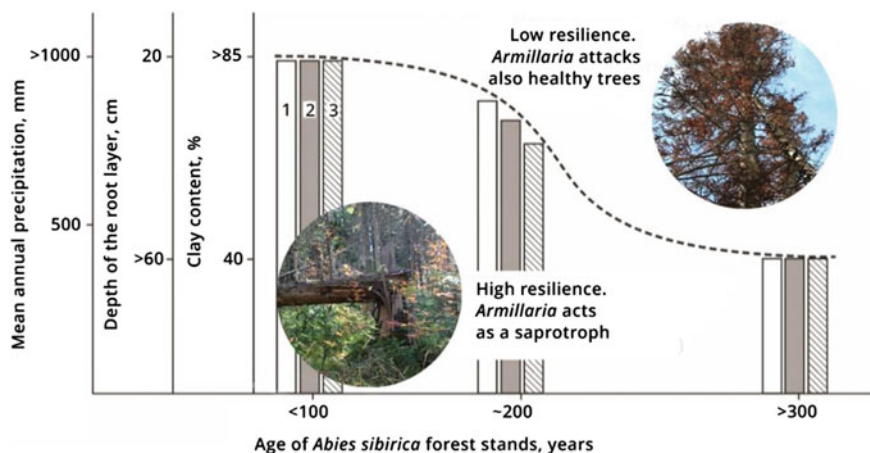
Fig. 3.4 Effects of drought and soil conditions on the resilience of Siberian fir trees against the pathogenic fungus *Armillaria mellea* s.l. Redrawn by permission from Springer Nature from Pavlov (2015). *Photo credit* Ilkka Vanha-Majamaa

(Pavlov, 2015). Similarly, the widespread *Armillaria* has the potential to increase in significance in boreal forests (Dempster, 2017). Like insects, pathogens are also candidates for invasive spread through human influence (Dukes et al., 2009).

3.5 Drought-induced Forest Decline

Whereas past disturbance studies have rarely considered drought, it is now recognized as a potent disturbance agent that can reduce tree growth (Itter et al., 2019), increase the vulnerability of trees to defoliation (Cooke & Roland, 2007), and drive tree mortality (De Grandpré et al., 2019). Mechanisms of drought-induced mortality include hydraulic failure, xylem embolism, and increased vulnerability to biotic disturbance agents, such as insects, fungi, and bacteria (Anderegg et al., 2013; Kharuk et al., 2016b; Voronin, 2018). Repeated drought events can weaken trees and decrease their resilience to subsequent drought events and secondary disturbance agents (DeSoto et al., 2020; Haynes et al., 2014; Pavlov, 2015).

Cases of drought-induced disturbances of varying severity have been reported across the boreal region (Chaps. 11 and 30; Michaelian et al., 2011; Pavlov, 2015; Zamolodchikov, 2012). In boreal forests, patchy drought-induced mortality is typical, especially in spruce-dominated primeval forests (Aakala & Kuuluvainen, 2011; Khakimulina et al., 2016) (Fig. 3.5). Similar patterns of decline and mortality of the "dark conifers" *Abies sibirica* and *Pinus sibirica* have been recorded in the southern Siberian Mountains and Baikal Mountains (Kharuk et al., 2013a). Birch

mortality, caused by prolonged drought, has been documented within the Trans-Baikal forest–steppe (Kharuk et al., 2013b). Notably, all reported cases of mortality of "dark conifers" in Russia have coincided with drought episodes, often accompanied by insect outbreaks (Kharuk et al., 2016b). However, interactions between drought and insect outbreaks are complex, especially for defoliating insects. Haynes et al. (2014) showed that outbreaks of only one of five forest insect pests in Germany were influenced by drought over the past centuries. Similarly, in North America, Itter et al. (2019) could not find an interaction between the growth reductions caused by two different defoliators and drought. On the other hand, De Grandpré et al. (2019) suggested that drought preceded spruce budworm–caused mortality. Another study indicates that bark beetles respond directly to climate change, whereas the evidence for defoliators is equivocal (Kolb et al., 2016). In addition to stressing host trees, drought can impact the insect itself. Thus, the effect of climate change on future insect outbreaks is difficult to predict. Outbreaks could become more severe if the trees are more negatively affected than the insect pests, or outbreaks could decrease in amplitude and severity if insects are more negatively affected than trees (Pureswaran et al., 2018).

From a forest management perspective, species response to drought is a key issue. In European boreal forests, the shallow-rooted overstory Norway spruce, which suffers from drought events over large spatial scales, is predicted to be strongly affected in the future (Kharuk et al., 2016b). In North America, jack pine is considered one of the least vulnerable tree species to drought (Peng et al., 2011). An inclusion of drought-resistant trees in forest management has been proposed as a strategy to mitigate the impacts of drought on forest productivity. However, before advocating large-scale switches from drought-sensitive to drought-tolerant species, it is

Fig. 3.5 Patchy drought-induced mortality of Norway spruce trees in a primeval forest of the Vepssky Forest Reserve, Russia. *Photo credit* Aleksandr Korepin

necessary to point out the complexity of these relationships. Aubin et al. (2018) used traits to identify drought-resistant trees in western Canada and classified trembling aspen (*Populus tremuloides* Michx.) as being highly sensitive to drought, whereas in eastern Canada, trembling aspen is considered one of the most drought-tolerant species (D'Orangeville et al., 2018; Héon-Grenier, 2020). Moreover, D'Orangeville et al. (2018) showed that although species differ in drought sensitivity, the marked intraspecific variability in this respect underlines the overriding effect of site. The severity of drought and other disturbances is also related to elevation, terrain topography, slope steepness, and aspect (Kharuk et al., 2013a). Even the least vulnerable trees will experience high rates of growth loss and mortality following drought if they are growing on shallow soils.

Deep snowpacks in boreal forests ensure that soil water is recharged annually; thus, cumulative soil moisture deficits may be limited and, consequently, minimize the effects of drought on boreal trees (Oogathoo et al., 2020; Léger-Beaulieu et al. In-Review). The timing of dry conditions during a season is also an essential factor to consider. D'Orangeville et al. (2016) have shown that spring droughts can benefit boreal soils subject to cold and wet conditions, whereas summer droughts can have more negative effects. Sánchez-Pinillos et al. (2022) also show that subsequent low-severity droughts can cause greater mortality than severe droughts.

Drought has been an increasingly common phenomenon over recent decades and is projected to be even more frequent and severe in the future. However, its effects are complex, especially its interactions with other disturbances. As tempting as it is to identify and favor drought-resistant species, site factors should be the primary consideration when predicting future impacts. Thus, from a management perspective, foresters should learn from forest vulnerability to drought. In particular, species and site conditions should be considered in silvicultural decisions, as certain sites may be at high risk of drought and should not be managed for timber production.

3.6   Snow and Ice

Snow and ice are often included in the list of typical disturbance agents of the boreal forest, but their effects have rarely been quantified. Ice storms are a major meteorological hazard in midlatitude regions (Cheng et al., 2007). They occur when freezing rain accumulates on trees, and the weight of the accumulated ice breaks the branches and stems. This can cause widespread damage in temperate forests, but these events are less common in boreal forests. Nonetheless, Markham et al. (2019) documented such an event in jack pine forests in Manitoba, where over 2,000 km^2 were damaged by ice in 2010. Similar events have been recorded in Manitoba in 1930 and 1958, showing that ice storms are also a potentially important disturbance agent in parts of the boreal zone.

The impacts of snow and ice on trees and forests resemble those of windstorms (Peltola et al., 1999; see also Sect. 3.2) in that they mechanically cause tree boles and branches to break. The breaking can also occur in interaction with high winds that

exert further forces on the stem. At the tree level, tree architecture and wood properties play a role. Ice storms and snow do not usually kill all trees in a stand (Markham et al., 2019), but they may change species composition, size structure, and stand spatial structure (Jalkanen & Konocpka, 2007; Nykänen et al., 1997). The accumulation of snow and ice and the resulting damage on trees across spatial scales from stands to regions depend on weather, e.g., cold and warm fronts, precipitation, air temperature, wind speed and direction, and locality, e.g., continentality, topography, altitude, and water table height (Barry & Chorley, 2010).

With changing climate, snow damage patterns are predicted to change (Kilpeläinen et al., 2010). As climate change brings about more extreme weather events and warmer conditions in the early winter and spring, the occurrence of ice storms is also likely to increase in the North American boreal forest (Cheng et al., 2007).

3.7 Concluding Remarks

Temporal and spatial descriptors of all disturbance types in the boreal forest vary in time and space and are thus difficult to emulate, predict, and control in an ecosystem management framework. The relative importance of different disturbance agents and the variability of current and future disturbance regimes within the boreal region require developing programs for ecosystem management and ecological restoration at a regional level. Decades of research have shown that the landscape scale should be better considered in ecosystem management (Patry et al., 2017). For instance, the current level of harvesting may, at least locally, be close to (or even beyond) the capacity of the system to cope with the combined effects of fire and harvesting, let alone climate alterations. Future climatic conditions are projected to become more conducive to several disturbance types, including fire, windthrow, insect outbreaks, and drought. Hence, maintaining the current level of harvesting in the future may be challenging (Boucher et al., 2018; Gauthier et al., 2015a). Assessments of the implementation of ecosystem management approaches are crucial in mitigating the future impacts of increasing disturbance frequency on forest ecosystems.

Partial harvesting, especially with the retention of deadwood and *habitat trees* (exceptionally large, usually scattered, individual trees in a stand), can maintain structural forest features similar to stands affected by insects or windstorms, and these features are crucial for hundreds of threatened forest species (Gustafsson et al., 2020; Kneeshaw et al., 2011; Koivula & Vanha-Majamaa, 2020). Descriptors of severity and specificity may provide a template for developing policies for maintaining biological legacies in post-harvest and salvage-logged forests (De Grandpré et al., 2018; Nappi et al., 2011).

Episodic disturbances may foster ecosystem adaptations to the effects of ongoing and future climatic change by increasing structural diversity with cascading positive effects on biodiversity, edaphic conditions, biogeochemical cycles, and increased heterogeneity across various spatial scales. Allowing some forests to be shaped by

natural processes may be congruent with multiple goals of forest management, even in densely settled and developed countries (Kulakowski et al., 2017). Emulating natural disturbances and successional dynamics at landscape and regional scales should be used to maintain the natural variability in old-growth attributes over time (Shorohova et al., 2011).

Addressing all forest ecosystem services calls for developing regional strategies to integrate disturbances into ecosystem management, with actions varying from prevention, control, and post-disturbance management to passive "rewilding" to the active emulation of disturbances. These actions have the combined goal of restoring ecosystem resilience by maintaining tree stand composition, age-class distribution, and natural-like structures.

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