Solving conflicts among conservation, economic and social objectives in boreal production forest landscapes; Fennoscandian perspectives

Mikko Mönkkönen, Daniel Burgas, Kyle Eyvindson, Eric Le Tortorec, Maiju Peura, Tähti Pohjanmies, Anna Repo & María Triviño

University of Jyvaskyla, Department of Biological and Environmental Sciences, POB 35, FI-40014 University of Jyvaskyla, Finland

Corresponding author: Mikko Mönkkönen, e-mail: mikko.monkkonen@jyu.fi

Table of Contents

Abstract

This chapter discusses challenges and possibilities involved in preserving biological diversity and the diversity of ecosystem services in the boreal zone and yet at the same time maintaining intensive timber extraction in boreal forests. Our focus is on Fennoscandian forests at the landscapes level, and we consider economic, social and ecological in the sustainability of forest management. We provide an outlook to boreal forest ecosystems and their history, and an overview of the forestry practices and policies that aim to ensure multi-functionality of Fennoscandian forests, i.e. diversity of efforts on sustaining biodiversity, timber production and other ecosystem services from forest landscapes. We review the current scientific understanding management effects on the structure and dynamics of the forest at different spatial, and the consequent repercussions on forest biodiversity and ecosystem services. Evidence suggests that many ecosystem services and biodiversity are in conflict with intensive timber production in boreal forests. We therefore present methods for assessing conflicts among alternative forest uses and for finding solutions for conflicts. We conclude the chapter by providing insights for future management aiming at sustainability from economic, ecological and social perspectives.

1. Introduction

The boreal biome encompasses almost one third of the world's forests (UNEP et al. 2009). Unlike tropical and temperate forests, boreal forests have remained relatively stable in area in recent decades (UNEP et al. 2009; FAO 2010). Overall, the region is characterized by a net gain in growing forest stock (FAO 2010). However, extensive tracts of boreal forests are actively managed and harvested for timber production, with changes to the structure and tree species composition of the forests and impacts on wildlife and ecosystem functioning (Östlund et al. 1997; Bradshaw et al. 2009; Kuuluvainen et al. 2012). Moreover, there is ongoing pressure to increase forest biomass use to reduce CO_2 emissions and increase renewable energy use according to set targets (Stupak et al. 2007).

Even though forest cover is still extensive, forestry has caused profound changes in the landscape and stand structure in the boreal zone (Bryant et al. 1997; Esseen et al. 1997). Boreal forests also harbor unique biodiversity (UNEP et al. 2009). However, habitat availability and resources for species associated with processes and structures characteristic for pristine forests has severely declined. Consequently, forest management is the most prominent threat for a number of species in the north. For a large proportion of the red-listed species for example in Finland and Sweden (totally about 1700 and 3500 species, respectively;) forestry is a main threat, and invertebrates and fungi are particularly common among forest-associated red-listed species (Rassi et al. 2010; Westling 2015).

Throughout the boreal region, intact forests are concentrated in the northernmost or otherwise inaccessible regions and, even still, are not extensively protected (Potapov et al. 2008). Protected areas cover only less than 5% of productive forest land in the Nordic countries. There is also a strong bias concerning protected areas so that most protected areas are situated in the north, and they consist, for large part, of forest types that are less productive than average (e.g. Angelstam and Pettersson 1997).

The multifunctional role of forests is widely recognized from scientific (Harrison et al. 2010) and policy points of view (e.g. the EU Forestry Strategy[1]). Forests are a major source of timber, but also provide a range of other goods and services that are essential to human societies (Vanhanen et al. 2012). In addition to providing jobs, income and raw material to industry, forests, for example, regulate and purify freshwater, prevent soil erosion, conserve biodiversity, protect against landslides, provide recreation and non-timber products, and acts as carbon sinks thus contributing to climate change mitigation (EC 2012). For both biodiversity and ecosystem services, it is crucially important how the forests outside the protected areas are managed. Forest exploitation has a long history in the Nordic countries dating back to the 17th century (Esseen et al. 1997). Intensive forestry practices such as clear-cut harvesting, soil preparation and planting new trees were developed in the mid 20th century to maximize the production of timber and pulpwood. Forest industry constitutes a large proportion of national gross production of Nordic countries and is very important to many local economies. This imposes a great challenge: is it possible to preserve biological diversity and the diversity of ecosystem services in the boreal zone and yet at the same time maintain intensive timber extraction from Fennoscandian boreal forests? In this chapter, we aim at providing an answer to this question based on the understanding provided by the research on the natural dynamics of the boreal biome, on management effects on the structure and dynamics of the forest at different spatial, and on multi-functionality of Fennoscandian forests. Our focus is on Fennoscandian forest landscapes where we scrutinize sustainability of forest management from economic, social and ecological perspectives.

We first provide a general overview of boreal forest ecosystems. We introduce main abiotic and biotic factors affecting the distribution and causing variation in boreal forest, to lay a basis for understanding factors affecting both biodiversity and ecosystem services. We provide an overview of the history of the biome, describe the natural dynamics of boreal forests, and provide an overview of biodiversity patterns.

---

[1] http://ec.europa.eu/agriculture/fore/forestry_strategy_en.htm

Further, we describe the main ecosystem services from boreal forest and identify ecosystem services that are typical for boreal forests, emphasizing differences with warm temperate forests. We provide insights how important these ecosystem services are globally, regionally and locally. We particularly discuss the so called every-man's-right tradition in Fennoscandia, and how this influences possibilities to benefit from forest ecosystem services.

Secondly, we describe the traditional and current forest management practices in Fennoscandia, and go on by describing how management mechanistically connects with biodiversity and the flow of ecosystem services. We review synergies and conflicts among alternative ecosystem services as well as biodiversity. Third, we provide an overview of the forestry practices and policies that aim to ensure multi-functionality of Fennoscandian forests, i.e. diversity of efforts on sustaining biodiversity, timber production and other ecosystem services from forest landscapes. Finally, we present alternative methods for assessing conflicts among different ecosystem services and for finding solutions for them. We conclude the chapter by providing insights for future management aiming at sustainability from economic, ecological and social perspectives.

It becomes obvious from our overview of boreal forest ecosystems that Fennoscandian forests are probably the simplest of all forest ecosystems on Earth. Because of long traditions in natural history and forest ecological research in all Nordic countries, Fennoscandian boreal forests are relatively well-known ecosystems. From global perspective, national economies of the Nordic countries are affluent, stable and predictable. We assert therefore that in comparative terms, Fennoscandia is an ideal test laboratory to find out if sustainable forest management, in ecological, economic and social terms, is an achievable goal (Mönkkönen 1999).

2. Boreal forest ecosystem

2.1. Distribution of boreal forests

The boreal biome is characterized by a relatively cold climate, large differences between summer and winter temperatures, and a persistent snow cover in winter. In some regions of the boreal, precipitation may fall mainly as snow. This combination, along with nutrient poor soils, favors the preponderance of conifer species (*Pinus* spp., *Picea* spp., *Abies* spp, *Larix* spp.), although species of broadleaved deciduous trees, particularly *Betula* spp., *Populus* spp. and *Salix* spp., are also common. Compared to lower latitudes, forests in the boreal biome are home to relatively few tree species. The boreal region meets the tundra to the north and at high altitudes, and the temperate forest to the south (Fig 1). The transition zone between the boreal and temperate regions, called the hemiboreal (or sometimes 'boreonemoral') zone, is characterized by a mixture of boreal and temperate elements (Nilsson 1997).

Olson et al. (2001) recognized 29 ecoregions within the boreal zone, eighteen in the Nearctic and eleven in the Palearctic region. Boreal forests of Fennoscandia and western Russia, stretching a latitudinal range of ~57° – 69° N, are one of the Palearctic boreal ecoregions. There are clear variations in climate at larger scales within the ecoregion, i.e. a gradient from a maritime climate in Norway to increasingly continental climates eastwards (Tuhkanen 1980), as well as a latitudinal and altitudinal gradient from the southern boreal zone to the northern boreal zone (Moen 1999). The location of Fennoscandia at the western edge of the Eurasian boreal forest belt makes the regions climatically different from other boreal ecoregions. Because of relatively maritime climate, annual variations in temperature and precipitation are less pronounced than in more continental areas in Siberia and the Nearctic. Also mean annual temperature is higher in the western Palearctic than at the same latitudes elsewhere, and consequently, the southern edge of the boreal zone in Fennoscandia is at the same latitude as the northern edge of the zone, for example, in Eastern North America.

The main gradients of variation influencing the local characteristics of natural boreal forests are (a) biogeographical patterns partly driven by climatic gradients, (b) the soil's nutrient and moisture gradients and their effects on tree growth and natural disturbances, and (c) time since disturbance (forest development). All these three gradients represent continuous variation rather than discreet classes. Nevertheless, classifying the gradients has large heuristic value in recognizing classes of forests and their developmental stages because they provide tools to handle underlying abiotic and biotic factors affecting both biodiversity and ecosystem services.

The biotic zonation of boreal forests in Fennoscandia has received much interest. Widely accepted subdivision scheme was created by Ahti et al. (1968), which identified four zones mainly based on the composition of plant communities in mesic sites. A main pattern is the increase of conifer tree dominance from hemiboreal zone towards the north boreal and concurrent decline of temperate broad-leaved trees. Likewise, the dominance of herbs in the field layer decreases and that of dwarf shrubs increases with increasing latitude. In the middle boreal zone, herbs are restricted to nutrient-rich sites, and even more so in the north boreal zone where bryophytes and lichens also become more abundant (Esseen et al. 1997).

Variation in nutrient availability and moisture also causes variation in forest community composition, which is summarized in forest site type classification systems (e.g. Cajander 1949; Arnborg 1990). Generally, soil type, soil moisture and vegetation in the European boreal forest are closely related. The vegetation of the understory of Fennoscandian boreal forest can be classified as follows. Very dry/nutrient-poor sites are lichen-dominated heaths. Dry-mesic and moderately rich sites are dominated by dwarf shrubs, e.g. *Calluna vulgaris*, *Empetrum hermaphroditum*, *Vaccinium vitis-idaea* and *V. myrtillus*. Moist-wet, moderately rich sites are characterized by *V. uliginosum* and *Ledum palustre,* while mesic, moderately rich by pleurocarpous mosses. In moist and rich sites herb layer with pteridophytes and e.g. *Maianthemum bifolium* is typical, and in moist-wet and very rich sites tall herb layer with e.g. *Filipendula ulmaria*.

Time since disturbance is yet another gradient of variation because boreal forests can be considered disturbance driven ecosystems (see below). This variation can be classified into forest successional stages based on stand structural characteristics (Shorohova et al. 2009).

2.2. History of boreal forests

Boreal forest biome is the youngest of the Earth's forest biomes. It has developed during gradual cooling that has characterized the global climate during the past 20 million years. During the early Oligocene, some 30 million years ago, tropical and subtropical forests thrived as far north as the Spitzbergen or Canadian (presently) artic. At the evolutionary time scale, the evolution of cold-hardiness in trees has been critical. The origin of the present boreal tree taxa likely locates in East Asian mountain regions, where frost-tolerance evolved in high altitude conifer and mixed forest during the gradual cooling of the Miocene climate. During the Pleistocene, these cold-hardy tree taxa then spread across the Palearctic and Neartic regions with further cooling of global climate (Latham and Ricklefs 1993). The current extent of the boreal zone is relatively recent phenomenon, developed during the past 2 million years when the global cooling culminated in the Pleistocene era. Thus, the boreal biome as we see it today is a child of the Ice Ages.

Pleistocene differs from the preceding era not only in terms of cooler global climates but also in terms of large climatic variations with recurrent glacial periods interrupted by warmer inter-glacial periods. Most of the area currently occupied by boreal forests, particularly in the western Palearctic and the Nearctic regions, was glaciated several times during the Pleistocene. Typically, tree species reacted to Pleistocene climatic oscillations in idiosyncratic ways, and thus forest zones or communities did not retract towards south during glaciations (Huntley and Birks 1983). Consequently, forests in the glacial refugia lack modern equivalents in the present boreal tree communities both taxonomically and structurally. The latest glacial period ended approximately 10,000 years ago, after which species gradually (re)colonized the region. As a consequence, boreal species assemblages can be considered relatively 'young' from a geological time perspective.

During the peak of glaciations, forests were almost completely wiped out from Europe except from the south- and sea-facing slopes of mountains in the Balkan, Apennine and Iberian peninsulas. Further north the climate was too cold and lower down in the Mediterranean region too dry for forest growth. Of the boreal tree species, glacial refugia of the Scots pine (*Pinus sylvestris*) and birches were much larger than those of the Norwegian spruce (*Picea abies*) that only existed in the Balkans and further east. In the Neartic and in eastern Palearctic, total forest area remained much more extensive than in the western Palearctic. In the Nearctic, both conifer and temperate deciduous zones extended across the entire continent as continuous belts (Cox and Moore 2010).

2.3. Natural disturbances in boreal forests

Boreal forests are shaped by a range of disturbances varying in size, severity and frequency, including fire, flooding, windthrow, snowbreak, avalanches, soil erosion, fungal diseases, outbreaks of defoliating insects, ungulate browsing, and the actions of beaver (*Castor fiber* in Eurasia) (Esseen et al. 1997). Disturbances are important natural drivers of forest ecosystem dynamics (Kuuluvainen and Aakala 2011), and have strong effects on the structure and functioning of forest ecosystems (Turner 2010). Particularly, fire has been one of the disturbances most studied because fires have been the most important factor shaping the structure and dynamics of natural boreal forests (Gromtsev 2002).

The boreal forest disturbance regimes range from natural succession following disturbances, such as severe stand-replacing fires and wind-storms, to small-scale dynamics associated to gaps in the canopy created by the loss of individual trees (Angelstam and Kuuluvainen 2004). Furthermore, boreal forest disturbance regimes may vary considerably depending on the characteristics of the dominant tree species, landscape, local site conditions and regional climate (Angelstam and Kuuluvainen 2004). According to Angelstam (1998), three main broad types of boreal forest can defined based on natural disturbance regimes in northern Europe: (1) Norway spruce dominated forest on wet and moist soils, characterized by internal gap dynamics, and often forming more or less narrow elements in the landscape along depressions and water courses; (2) successional forest following stand-replacing disturbance such as fire (commonly on mesic soils), characterized by a gradual change from open conditions to closed forest, and from more deciduous trees and pine shortly after disturbance to more Norway spruce after several decades; (3) Scots pine dominated forest on drier sites subjected to frequent low-intensity fires, with different cohorts of trees having survived past fire events (Fig. 2). Evidence suggests that in Fennoscandian conditions gap dynamics (1) and cohort dynamics (3) were much more common than successional dynamics following stand-replacing fires (2) under natural disturbance dynamics (Angelstam and Kuuluvainen 2004). Also simulations suggest that old-growth forests dominated the natural landscapes of northern Europe, including the European part of the Russian boreal zone (Gromtsev 2002; Pennanen 2002). Similarly in northeastern North America, pre-settlement forests were dominated by relatively frequent, low intensity disturbances that produced a heterogeneous mosaic, while large-scale stand-replacing disturbances were rare (Seymour et al. 2002). In more continental regions such as Siberia and central Canada, stand-replacing fires and subsequent even-aged successional forests have been the dominating natural dynamics (Cogbill 1985).

Many studies showed that natural disturbances have a positive effect on biodiversity. Therefore, emulation of natural disturbances has been proposed as a sustainable forest management (Kuuluvainen and Grenfell 2012). In boreal forest ecosystems where fire is the major natural disturbance agent, it has been suggested the application of prescribed burning as a measure to restore natural forest conditions (Toivanen and Kotiaho 2007) whereas in wind- and bark beetle-dominated disturbance regimes the creation of gaps of various sizes and shapes has been recommended to increase biodiversity (Kuuluvainen 2002).

In recent decades, forest disturbances have increased their frequencies and severity in many parts of the world (Chapin et al. 2000) and it is expected to increase even more in the future as a result of climate change (Seidl et al. 2014). As a consequence, it is very important to know the impacts of natural disturbances on boreal forests biodiversity and ecosystem services. A recent review by Thom and Seidl (2015a) carried out in temperate and boreal forests to evaluate the impacts of three of the most relevant disturbance factors (fire, wind and bark beetle) on 13 different ecosystem services and three indicators of forest biodiversity revealed a 'disturbance paradox' reporting that disturbances can have negative effects on ecosystem services while simultaneously facilitating biodiversity. Thom and Seidl (2015) found that under a high-severity disturbance event, it was expected a loss of 38% of carbon storage while an increase in species richness by 36%. However, since negative effects on carbon are rapidly reduced with time, the positive effects on biodiversity do not substantially change over time. Therefore, managing for a low to medium frequency disturbance regime would result in a reduced negative impact on ecosystem services while still benefiting biodiversity. This emphasises the need to put more attention on disturbance risk and resilience in ecosystem management and new strategies to address the disturbance paradox in management are needed.

2.4. Biodiversity in boreal forests

The deep evolutionary history and Pleistocene glaciations are etched in the species diversity of current temperate forest regions. For example, the effects of Pleistocene glaciations have been most drastic in the western Paleartic region where several tree species and genera went extinct, mostly due to diminishing area of forests. Consequently, forests in the western Palearctic are less diverse in tree species than Nearctic and eastern Palearctic forests (Huntley 1993). Fennoscandian boreal forests contain only two common conifer tree species (Scots pine and Norway spruce) whereas in eastern Siberia there are eight common conifer species, and in the eastern Nearctic typically 4-6 widespread conifers. Parallel differences among regions with the boreal zone are evident in forest-associated vertebrate species (Mönkkönen and Viro 1997). Compared to tropical forests where up to 300 tree species may occur in a single hectare (Gentry 1988) or temperate deciduous forest biome with almost 1200 tree species in total (Latham and Ricklefs 1993), boreal forests may appear dull. However, for some taxa such as beetles, the monotonous appearance is misleading, and consequently, Hanski and Hammond (1995) asserted that “if we do not soon recognize that boreal forests are not simple stands of pine and spruce, that is just what they may become”.

Disturbances and processes following them are important drivers of biodiversity in boreal forests. Large scale disturbances such as forest fires or storms create structural variation at the landscape scale, and boreal ecosystems under natural disturbance regimes are composed of forests representing different developmental stages (Syrjänen et al. 1994; Kuuluvainen and Aakala 2011) providing habitat and resources for species with varying preferences and requirements. Smaller scale disturbances resulting in replacing individual trees or group of trees by new individuals provide structural variation within forest stands, e.g. maintain mixed species tree communities and variation in tree ages, which is important from biodiversity perspective.

Tree deaths following disturbances influence a number of patterns and processes occurring in forest ecosystems (Franklin et al. 2002). Large quantities of decaying wood are one of the most characteristic features of natural, both young and old boreal forests. Decaying wood is important resource and habitat for a very large number of species. Both in Sweden and Finland, about 20-25% of all multicellular forest species (20000 in Finland, 25000 in Sweden) are associated with dead-wood, i.e. 5000 and 6500 saproxylic species in Finland and Sweden, respectively (Siitonen 2001; De Jong et al. 2004). Because of intensive forestry, dead-wood volumes in managed Fennoscandian boreal forests have decreased by some 95% compared with forests under natural disturbance regime. This is a prime example of the fact that forestry-induced disturbances typically differ from natural disturbances in their effects on structures and processes important for biodiversity (Fig. 2).

The non-linear relationship between the number of species and area is one of the few recurrent pervasive patterns in ecology (Rosenzweig 1995). When the area of a habitat type starts to diminish, for example, on old-growth forests, the changes in the species number are very small at first, but the rate of loss of species accelerates considerably with further habitat loss. Habitat loss is globally the main reason for increasing species extinction rates (MEA 2005). The proportion of old-growth forests in the present Fennoscandian landscape is getting so small in the modern landscape that we very likely operate on the steeply declining part of the species-area relationship for species strictly associated with old-growth forests. For example in Finland, forests more than 120 years in age comprise about 4% and 18% of forest land in southern and northern parts, respectively (Peltola 2014); in a natural boreal landscape this percentage is much higher, e.g. 70% in pristine NE European taiga in Russia (Syrjänen et al. 1994). Similarly, resulting from diminishing habitat availability for dead-wood associated species, they are disproportionally numerous among the threatened species in Fennoscandia: the reduction of large diameter dead-wood has been identified as the principal threat factor for more than 50% of forest species (Berg et al. 1994; Siitonen 2001).

A simple calculation suggests that many species associated with virgin forests and their characteristic forest structures have either become extinct or will be lost in near future, as these natural forests comprise an ever-decreasing fraction of the total land area in Fennoscandia (Hanski 2000). This is a sheer challenge for sustainable forest management. Because of pressures to increase forest biomass harvesting, future boreal landscapes will likely face further reductions in the area of old, mature forests. The sustainability challenge actually is to break the species-area relationship with careful landscape level planning, i.e. maintain species and their populations when their habitat is decreasing. This seems very challenging indeed, because with low proportion of original habitat and resources in the landscape, the secondary consequences of habitat fragmentation such as edge effects tend to draw down the species number even more than predicted by the species-area-relationship (Mönkkönen 1999).

2.5. Main ecosystem services from boreal forests

Ecosystem services can be divided into three main classes: *regulating*, *provisioning* and *cultural* services according to the Common International Classification of Ecosystem Services (CICES) (Haines-Young and Potschin 2011; Haines-Young and Potschin 2013). Regulating services are the benefits obtained from the regulation of ecosystem processes such as climate regulation and pest control. Provisioning services are the products obtained from ecosystems such as food and fresh water. Cultural services are non-material benefits that people obtain from ecosystems such as recreation and spiritual enrichment. Here, we provide an overview of the main ecosystem services in boreal forests for each class.

2.5.1. Regulating

Globally the most important *regulating* service in boreal forests is the climate regulation (Burton et al. 2010). Carbon storage and sequestration by boreal forests is hugely important for global climate change mitigation. Boreal forests contain more than 30 % of the global carbon storage and more than 20 % of the global carbon sinks in forests (Pan et al. 2011). In boreal forests, 60% of carbon is stored in soil and

20% in woody biomass whereas in other forests, such as tropical forests, the shares are opposite (56% in biomass and 32% in soil). In addition, especially old natural boreal forests store and also sequestrate carbon (Luyssaert et al. 2008) and contrary to temperate deciduous forests, some forested areas in the boreal zone still exist without human disturbance (Burton et al. 2010). Moreover, there are differences between forest biomes in climate regulation in terms of evapotranspiration and albedo effects (Snyder et al. 2004). In the boreal zone albedo effect is more important for climate change mitigation than in tropical and temperate forests where cooling effects by evapotranspiration is more important (Bonan et al. 2008).

Many regionally and locally important regulating services in forests are related to water and soil and are considered as public goods. Boreal forests are part of hydrological cycles and regulate water flows, they filter ground water and act as buffer zones for adjacent waters by retaining nutrients (Saastamoinen et al. 2014). Boreal regions have one of the largest freshwater supplies in the world (Schindler 1998). In addition, boreal forests retain nutrients and maintain soil productivity (Maynard et al. 2014), and provide resistance against natural disturbances such as pests, diseases, fire, wind and floods, which may become even more important in the future if climate change increases disturbances (Thom and Seidl 2015b). Furthermore, they provide habitats for many beneficial organisms, such as pollinators and decomposers, e.g., honeybees living in forests are pollinators of many commercially valuable crop species (Kettunen et al. 2012).

2.5.2. Provisioning

Timber production is the economically most important *provisioning* service in boreal forests (Burton et al. 2010; Vanhanen et al. 2012). Boreal forests contain approximately 45% of the world's stock of growing timber and approximately 25% of global exports of forest industry derives from boreal forests. Slow tree growth produces strong, narrow-ringed wood with excellent properties as construction timber and fiber for paper-making. Logging residues and small diameter trees are used for bioenergy production to substitute for fossil fuels (Repo et al. 2011, Lemprière et al. 2013).

The intensity of forestry varies greatly across boreal forests. Most of Alaska's boreal forests are beyond timber production whereas only 7 % of Finland's forestland is excluded from timber production. The share of the forest sector in the Gross Domestic Product (GDP) is approximately 1% in Russia, 2% in Canada and Sweden and even 4% in Finland (Vanhanen et al. 2012). Especially in Fennoscandia, many forests are privately owned and provide economic benefits from timber for private people. In addition, forest industry is an important employer and creates jobs for many local people.

In addition, non-timber forest products, such as mushrooms and berries are valuable provisioning services in boreal forests (Burton et al. 2010; Saastamoinen et al. 2014). These products are economically and culturally important especially for rural and aboriginal communities (Turtiainen et al. 2012; Natural resources of Canada 2016). Every-man's-right tradition in Fennoscandia allows all people to have access to forested areas and opportunity to collect products even in privately owned forests (Kettunen et al. 2012). This right makes berries and mushrooms a public good (Paassilta et al. 2009). The berries and mushrooms are mainly collected for household use but some species, such as bilberries (*Vaccinium myrtillus*, *Vaccinium corymbosum*), cowberries (*Vaccinium vitis-idaea*), and mushrooms, particularly *Lactarius* spp. and *Boletus* spp., are also commercially harvested. In Fennoscandia, approximately 5-10% of the berry and mushroom crops are harvested annually. Selling collectables is tax-free thus providing income for local people (Turtiainen et al. 2012).

Commercial harvesting of non-timber products of boreal forests and their international trade are increasing Finland and Sweden are the main exporters of bilberries (Paassilta et al. 2009) and Canada is the main producer of maple products (Natural resources of Canada 2016). Wild mushrooms from boreal forests, such as cep (*Boletus edulis*) and chanterelle (*Cantharellus cibarius*) species are exported to Central Europe and Asia (MARSI 2014; Natural resources of Canada 2016).

Moreover, hunting game species, such as moose (*Alces alces*), provide income for local people, and in Fennoscandian countries, the economic value of game meat is between 44 and 125 million euros per year (Kettunen et al. 2012). Nature tourism is both nationally and regionally important sector in the Nordic countries. For example, in Finland nature tourism provides employment for 32000 persons; forestry for 25000 persons (Ministry of Employment and the Economy 2014). In Finnish Lapland, nature tourism is the most important regional economy sector. Other regionally and locally important provisioning services distinct to boreal region are services such as reindeer herding and Christmas tree harvesting (Kettunen et al. 2012; Natural Resources of Canada 2016).

2.5.3. Cultural

Many non-timber forest products are also categorized as *cultural* ecosystem services because of their recreational and cultural value. Recreational berry and mushroom picking as well as hunting are popular activities for local people (Kettunen et al. 2012; Brandt et al. 2013). For example, more than half of the Finns annually participate picking berries and mushrooms (Turtiainen et al. 2012). Hunting has long traditions and it still has importance to local communities also in terms of cultural identity.

Boreal forests are used also for other recreational outdoor activities, such as hiking, camping and watching e.g., bird species. For example, almost half of the Norwegians go hiking in forests or mountains more than twice a month (Kettunen et al. 2012). Also the scenic beauty of forest landscape and national species have their own cultural and recreational values (Saastamoinen et al. 2014). Forests also provide possibilities to improve human health conditions. The results of a vast amount of research show that forest visits promote both physical and mental health by reducing stress (Karjalainen et al. 2010). In Fennoscandian countries, blueberry and cloudberry as well as birch sap are used for health related products and cosmetics (Kettunen et al. 2012). Aboriginal communities in Canada have long traditions using medical plant species (Uprety et al. 2012). In general, a great potential exist in the development and utilization of health benefits deriving from boreal forests.

3. Past and present forest use and management in the boreal zone

3.1. Historical use of boreal forests

From a global perspective, the boreal biome has sparse human population and relatively low anthropogenic impacts (Gauthier et al. 2015). In the boreal, only a minor proportion of the boreal forest has been converted to farmland, and land clearing for agriculture has mostly occurred in lowland areas along the seacoasts, near large lakes and in river valleys. Forest land is a dominating land use type across the entire boreal zone but still, large parts of the European boreal forest have been influenced by a long history of shifting slash-and-burn agriculture (Myllyntaus et al. 2002). Importantly, most of the European boreal forest has been subjected to some form of logging or management for wood production (Bryant et al. 1997). In Fennoscandia, the history of logging is relatively long compared to other parts of the boreal biome. Already in the 17th-19th centuries, forests were logged for charcoal production to supply the mining industry, and potash and tar production had large impacts on forests in some regions (Esseen et al. 1997; Östlund and Roturier 2010).

Extensive logging for saw timber – typically diameter-limit cutting targeting the largest trees – increased mostly from the 19th century, spreading gradually from the south into the north and inland of Fennoscandia (Imbeau et al. 2001a). Also the demand for pulp wood considerably increased in the first half of the 20th century. Since the 1950s, even-aged forestry involving clear-felling and thinning has been the dominating management regime throughout the boreal zone. In Russia, forest management has thus far been less intensive than in the Nordic countries from a silvicultural perspective (Elbakidze et al. 2013), although logging has affected most parts of the Russian boreal forest (Potapov et al. 2008). In the Nearctic boreal zone, large scale forest management and harvesting started in the mid-20th century, and harvesting for timber still largely considers pristine forests (Bryant et al. 1997; Imbeau et al. 2001b). Moreover, forest fires have been suppressed very successfully in the Nordic countries since the later parts of the 19th century (Zackrisson 1977), whereas they still are a relatively more common disturbance agent in Russia and in Canada.

3.2. Forestry

In today's Fennoscandia, most of the productive forest area is under management for timber production, while low-productive areas are typically less affected by forestry. At a larger scale, there are still some relatively intact forest landscapes in the boreal region, unlike the situation in most of temperate Europe. However, these landscapes are confined to the northern parts of European Russia (Potapov et al. 2008) as well as to areas along the Finnish-Russian border and the Scandinavian and Ural mountain ranges.

Forest management is the conduct of human actions to extract resources and modify the growing potential of the forest. From a recently harvested stand (or bare ground) there are regional level recommendations "best practice" guidelines that indicate which actions may be suitable to conduct on different forest structures (i.e. Äijälä et al. 2014). The set of management actions possible depend on the age and density of the stand, and the access to the site. On recently harvested stands, the primary interest is often to ensure the rapid establishment of an appropriate tree species. In Fennoscandia, there is generally access to the forest site, silvicultural treatments may be applied to assist in the establishment and early development of the stand. For instance, preparatory actions may be conducted to aid in regeneration and planting, to sowing seeds or leaving a selection of seed trees to allow for natural regeneration. Once the stand has been established, and has grown for several years, the option to conduct a pre-commercial thinning could increase growth by reducing competition between trees. Once the forest has matured, there are possibilities to extract timber resources. Two different methods of extracting timber are possible: to conduct thinnings (selective removal of trees) or to conduct a clear felling (removal of most trees in the stand), returning the stand to a 'bare ground' state. When access to the forest site is the key limitation (such as in Russia and Canada), road construction is required to do forest operations. This restricts the economically feasible management alternatives.

When access to site is not a limiting factor, the exact timing of the operations depends upon the ability of the forest to grow. The key input variables for growth models relate to conditions specific to the stand (Skovsgaard and Vanclay 2008). This includes the quantity of sunlight received, the soil conditions and water availability determines the fertility of the site. For instance, forests located on sites with reduced soil nutrition or water viability will require a longer growing period, resulting in a delayed harvest when compared to a site with improved soil nutrition and improved water availability.

In Finland, the legal restrictions have been removed with an update of the forest law (Finnish Forest Act 2014) in 2014, continuous cover forestry (CCF) or uneven aged forestry is now allowed. Currently there is substantial research being conducted to find the most appropriate method of conducting CCF operations (Pukkala et al. 2011; Rämö and Tahvonen 2014; Lundmark et al. 2016). Although the legal restrictions have been relaxed, the primary method to extract timber resources is through a clear felling.

In Fennoscandia, predictions of future forest resources are conducted through forest management software. These tools integrate a large number of models to predict development of the forest. Models predict the growth, mortality and ingrowth of the trees dependent on the specific site conditions (location, fertility, soil type). There is a wide variety of software options, for instance in Finland there are three primary options: MELA (Redsven et al. 2012), MOTTI (Hynynen et al. 2005) and SIMO (Rasinmäki et al. 2009), in Sweden the Heureka system (Wikström et al. 2011), and AVVIRK in

Norway (Eid and Hobbelstad 2000). In addition to generating predictions of future forest development, these software packages also include optimization tools (i.e. linear programming and heuristics; Kangas et al. 2015), which can be used to develop forest management alternatives.

4. Conflicts and synergies among ecosystem services and biodiversity

4.1. Mechanistic pathways from management to biodiversity and ecosystem services

The structure of forests under natural disturbance regime vs. managed forests differ at several spatial scales (Fig. 2). These are important to acknowledge because species possess adaptations to structures that they have encountered during the evolutionary past and may therefore have difficulties in coping with changes in forest structures due to forest management. The basic pattern is that natural disturbances such as forest fires and gap formation create much structural variation and functional diversity in forest ecosystems that are missing, severely diminished or altered in managed forests (Hansen et al. 1991; Esseen et al. 1997). These stem from differences in the disturbances: natural disturbances show much variation in their intensity and extent whereas forest management actions usually follow standard sets of measures with little variation among stands (Table 1). At the stand scale, the most significant differences are low diversity of tree species, low amount of dead wood and low abundance of very large tree individuals in managed stands compared with stands under natural dynamics. At the landscape scale, differences in size and age distribution of stands are large, managed forest landscapes having much less variation in the size of disturbed (managed) areas and more even age-distribution. Managed landscapes typically are patchworks of stands at different developmental stages having more sharp edges between stands and less connectivity between patches of similar habitat types.

These structural differences have important repercussions to biodiversity and potentially also to ecosystem services in managed forests because of the tight links between the structure and function of ecosystems. Structural differences have direct effects on species ability to find habitats, disperse in the landscape and eventually persist as viable populations, and thereby indirect effects on the functioning of forest ecosystem. Indeed, in boreal forests where extensive areas have been actively managed and harvested primarily for timber production for decades, changes in forest structure and negative impacts on biodiversity and ecosystem functioning are now apparent (Bradshaw et al. 2009; Kuuluvainen et al. 2012). For example, the clear-cutting of even-aged forest stands commonly used in boreal timber production can radically change both biotic and abiotic conditions within a very short time, leading to negative impacts on biodiversity. In Finland, where production forests have been intensively managed with clear-cutting for almost 100 years, within-stand forest structure has become relatively even-aged, and the amount of deadwood has been considerably reduced (e.g. Vanha-Majamaa et al. 2007). As a consequence species requiring e.g. dead-wood and very large (over-mature) trees have faced severe habitat loss.

The delivery of ecosystem services may be described as a process with four interacting stages: ecosystem structures, ecosystem functions, benefits experienced by humans, and finally the values assigned to these benefits (Haines-Young and Potschin 2010). Here, ecosystem structures refer to both the biotic and abiotic components of the ecosystem. As described above, in boreal forest ecosystems these characteristics (Fig. 4)) are heavily modified by forest management, and this has potential impacts on ecosystem functions. As these structures and functions are the basis of ecosystem services, the changes to the forest ecosystem following management interventions may thus extend to the various benefits required by different stakeholders (Fig. 4).

Species biodiversity, encompassing the diversity of species, the ecological functions that they possess, as well as their phylogenetic history, form the basis of ecosystem functions, which in turn provide ecosystem services to humans (Duncan et al. 2015). Both experimental and correlational research into the links between biodiversity and ecosystem functioning have shown that losses in biodiversity have clear negative impacts on ecosystem functioning (Cardinale et al. 2012; Tilman et al. 2012). For example, Costanza et al. (2007) showed a positive link between vascular species diversity and net primary productivity in North America. Similarly, Paquette and Messier (2011) showed that increased functional diversity of shrubs and trees were associated with increased forest productivity in temperate and boreal forests in Canada. In addition, studies looking at the links between biodiversity and ecosystem services have shown a similar pattern. For example, Gamfeldt et al. (2013) showed in Sweden that forests with a higher diversity of tree species showed higher levels of ecosystem services, namely biomass production, berry production and soil carbon storage. Similarly, Vilá et al. (2007) showed that in the Mediterranean tree species richness was positively associated with wood production. Maes et al. (2012) found that on a European scale, increased levels of biodiversity had a general positive influence on forest-associated ecosystem services such as timber production, carbon storage and recreation. Although previous research into the links between biodiversity and ecosystem functioning and services has focused on species richness, it has become increasingly clear that the diversity of functions processed by species plays the most important role (Díaz et al. 2007).

In general, the relation between biodiversity and ecosystem services is complex because biodiversity plays an important role at many levels of ecosystem service production (Mace et al. 2012). Multiple species are involved in producing ecosystem functions, and multiple ecosystem functions can be required to produce even a single ecosystem service. Biodiversity is also a multifaceted concept with several alternative elements (e.g. species) and attributes (e.g. amount or composition) whose role in providing any given ecosystem service can vary from indispensable to harmful. In addition, the links between biodiversity and ecosystem functioning, and biodiversity and ecosystem services are not always linear. For example, even though Gamfeldt et al. (2013) showed a positive association between biodiversity and several ecosystem services, in half of the cases the relation was hump-shaped. In practice this means that sites with six or more tree species had diminishing levels of biomass and bilberry production. Spatial scale also plays an important role since positive relationships between biodiversity and ecosystem services have been found at the global scale (e.g. Strassburg et al. 2010) but this relationship weakens at national or regional scales (e.g. Thomas et al. 2013).

Despite the uncertainties outlined above, the majority of evidence points to a positive impact of increased biodiversity on ecosystem functioning and services. Thus, forest management practices, such as timber felling, that have negative impacts on biodiversity can also negatively impact both ecosystem functioning and ecosystems services. In addition to final harvesting, other forest management practices can also impact ecosystem services. For example, the frequency and intensity of thinning play very important roles in timber production and carbon sequestration yet thinning also reduces structural diversity important to biodiversity. Indeed, simulation studies have shown that foregoing thinning altogether can have clear positive impacts on biodiversity and ecosystem functioning by resulting in 5-6 times more dead wood than green tree retention (Tikkanen et al. 2012).

4.2. Conflicts

As described above, forest management actions taken to increase timber yields affect also other forest functions and services. If stand management and harvesting cause losses in ecosystem services, these functions are in conflict and there are trade-offs between timber production and other benefits provided by the forest. These situations have been found to be common in boreal forests. For example, Pohjanmies et al. (unpublished) evaluated all of the pairwise conflicts among timber production and four other ecosystem services (bilberry production, carbon storage, pest regulation, and biodiversity conservation) using forest management simulations. They found strong conflicts between timber production and the other ecosystem services to be common, whereas conflicts among the four non-timber objectives were less severe (Fig. 4). For example, prioritizing timber production caused an average loss of 58% in the ecosystem service of pest regulation, while prioritizing carbon storage caused an average loss of only 9% in the same service. Similarly, prioritizing pest regulation caused average losses of 94% and 5% in timber production and carbon storage, respectively. According to these results, timber production and pest regulation are thus strongly conflicting but carbon storage and pest regulation are highly compatible management objectives. However, Triviño et al. (2016) showed that targeting high levels of timber revenues creates conflicts also between non-timber benefits (here, carbon storage and biodiversity).

The forestry activities with the most substantial effects on ecosystem services are those that most severely affect the structure of the forest ecosystem. These include, for example, intensive and/or extensive wood harvesting, management of stand structure and tree species composition, and mechanical soil preparation. However, the natural processes that generate ecosystem services may be very complex, and much information on forestry's effects on them is still lacking. The effects of timber production on ecosystem services may be highly site-dependent and varied – simultaneously harmful for some services and beneficial for others.

One of the most extensively studied ecosystem services with respect to its responses to forest management is climate regulation via carbon dynamics. Carbon storage and sequestration by boreal forests is hugely important for global climate change mitigation (Pan et al. 2011), and forestry may have substantial impacts on these functions. However, these impacts are not always straightforward. Timber production and climate regulation via carbon sequestration are in conflict if forestry operations decrease the system's ability to fix carbon or if they result in releases of carbon into the atmosphere from long-term storages in the forest ecosystem, e.g. from forest soils. Generally, management choices increasing tree growth have a positive effect on the balance whereas increasing the harvesting level results in negative effects (Liski et al. 2001). Management strategies to increase forest carbon stocks include, extending rotation lengths (Cooper 1983; Liski et al. 2001; Kaipainen et al. 2004; Triviño et al. 2015a), changes in initial stand density and thinning strategies (e.g. Niinimäki et al. 2013; Pihlainen et al. 2014) and forest fertilization (e.g. Boyland 2006). Extending forest rotation period allows trees to grow larger and forests to accumulate more litter and soil organic matter, whereas forest fertilization increases tree growth and litter input to the soil from living biomass and forest thinnings. Forest management choices can also reduce the carbon stocks and the carbon sink capacity of through the intensification of biomass harvests in the form of timber or forest harvest residues (Repo et al. 2011; Kallio et al. 2013; Sievänen et al. 2013) and the shortening of forest rotation period (Kaipainen et al. 2004). Forest management minimizing the disturbances in the stand structure and soil reduces the risk of unintended carbon losses (Jandl et al. 2007). However, there may be trade-offs between short-term and long-term carbon sequestration. On top of the direct effects of forest management on carbon-related ecosystem functions, forestry's contribution to carbon sequestration and/or emissions may crucially depend on the entire life-cycle of the forest product. Moreover, the role of forests in climate regulation is not limited to carbon dynamics, but includes processes such as water and energy fluxes (Naudts et al. 2016), surface albedo (Lutz and Howarth 2014), and production of aerosols that contribute to cloud formation (Spracklen et al. 2008).

Another extensively studied forest ecosystem service is regulation of surface water quality. Clear-cut harvesting and mechanical site preparation, which are common practices in boreal forestry, may have negative impacts on runoff water quality by increasing nutrient and organic matter load (Kreutzweiser et al. 2008). Also fertilization may increase the nutrient load from forests to surface waters (Laudon et al. 2011), while also affecting ground vegetation (Strengbom and Nordin 2008).

Alterations to the structure and composition of the stand and the physical properties of the soil may affect the site's suitability as habitat for beneficial species such as collectable forest products, pollinators, natural pest control agents, and decomposers. Here, it is particularly typical that forestry has contrasting impacts on the various benefits, because different species have differing habitat and resource requirements. For example, the recreationally and economically important wild bilberry (*Vaccinium myrtillus*) has declined in abundance in Fennoscandia due to extensive clear-cut harvesting and soil preparation activities. Conversely, species that thrive in young stands or benefit from increased canopy openness may benefit from forestry activities (Clason et al. 2008). Pest outbreaks may be controlled by removing naturally felled trees and thus minimizing the availability of food and breeding resources of pest species (Jactel et al. 2009). Then again, the lack of nesting resources created by natural disturbances has been also suggested to negatively affect pollinators (Rodríguez and Kouki 2015).

Some of the ecosystem services provided by production forests directly benefit forestry itself. These include ecosystem functions that maintain the productivity of the forest ecosystem or protect the forest property, such as nutrient cycling, erosion prevention, mitigation of storm damages, and regulation of pest outbreaks. Losses in these ecosystem services would eventually risk also timber yields. The impacts of forestry operations on the physical and chemical properties of soils that maintain productivity are highly site-dependent, but in many cases they have been estimated to be small in effect (Kreutzweiser et al. 2008). Boreal forests are typically able to recover from nutrient losses caused by biomass removal and increased leaching due to comparatively long rotation cycles as well as atmospheric deposition (Kreutzweiser et al. 2008). However, this ability may be threatened if productivity is artificially increased or biomass harvesting is intensified (Laudon et al. 2011). This would mean there is a conflict between these services and timber production, but only if the services are evaluated in a short time perspective.

The importance of the prevention and mitigation of natural disturbances is increasingly recognized as these disturbances are predicted to become more common in response to global change. The storm resistance of a stand may potentially be reinforced by choices regarding the structure of the stand and the surrounding landscape, that is, by stand diversification and minimization of height differences, gaps, and stand edges (Zeng et al. 2009; Zeng et al. 2010). Resistance to biotic disturbances may be more complicated to promote, because different pests may respond to stand management differently and because actions planned to control pest populations may also harm their natural enemies (Jactel et al. 2009).

Overall, it can be concluded that there is great variation between different ecosystem services in the extent to which the impacts of forestry on them are known. Changes in processes that involve interactions within species communities over various temporal and spatial scales are difficult to quantify and predict. The negative effects of forestry on numerous forest species and species groups have been recognized, but the long-term implications for the forest functions that rely on these communities are still poorly understood. Moreover, even when changes in the supply of ecosystem services are predicted, the consequences of these changes to human well-being are rarely evaluated. However, it is clear that intensive forestry has the potential to lead to substantial losses in several crucial forest ecosystem services.

Because of the linkages between management, biodiversity and ecosystem services it is obvious that economically, ecologically and socially sustainable forest management planning must simultaneously consider all these aspects. Management solely focusing on intensive timber production at the landscape scale will (and has) incur(red) severe ecological and societal costs in terms of loss of ecosystem structures and functions as well as species, and consequently, may put many ecosystem services at risk. Over the past 20-30 years, forest management has been in transition from mere exploitation to sustainability, and has adopted several methods to allow for more balanced use of the goods and benefits forests provide. In the next section we provide an overview of these methods. While debate on the most beneficial forest management methods is ongoing and important information is still lacking (Kuuluvainen et al. 2012), one general message can already be derived from the past research: we need more variation in management regimes. In the following section, we also provide insights how to combine alternative management regimes at the landscape level to figure out desirable management plans that will meet the potentially conflicting objectives in an optimal way.

We can also conclude from the overview we provider earlier that even if biodiversity is the basis for ecosystem functioning and services thereof and there tends to be a positive correlation between them, management for a full set of ecosystem services does not necessarily align with the management that most effectively supports biodiversity. For example, prevention and mitigation of natural disturbances is increasingly recognized as an important regulating ecosystem service but many studies have shown that natural disturbances are crucial to the maintenance of biodiversity. Thus, while management planning is increasingly incorporating measures to ensure multiple ecosystem services simultaneously, we cannot assume that this will be good for biodiversity as well.

5. Management for solving the conflicts

5.1. Management approaches for multi-use forestry management approaches for multi-use forestry

The currently dominant practice of even-aged management with clear felling was originally targeted to solely maximize the volume of harvested timber. However, there is a gradual recognition of shortcomings on the economic effectiveness of clearcutting (Hyytiäinen and Tahvonen 2002; Hyytiäinen et al. 2004; Kuuluvainen et al. 2012) and of the conflict of such approach with other services from forest habitats beyond wood extraction. To address these, there has been a range of alternatives proposed. The common denominator in the deviations from the general practice is the preservation of key structural and biological elements necessary for certain ecological processes of forest ecosystems (e.g. seedling, ground water regulation, connectivity between forest stands).

Monocultures are largely preferred in current intensive forestry (Clark and Covey 2012). For instance, in Finland and Sweden only 14% and 10% of the total forest land, respectively, is composed of mixed forests (Christiansen 2014; Peltola 2014) with the proportion of mixed forest in Finland seeing a reduction by half since the 1950's in favor of monocultures of coniferous species (Parviainen and Västilä 2011). The model of monoculture is attractive in intensive forestry for its conceptual and management simplicity while providing large amounts of timber by focusing on the most productive, commercially interesting tree species. Nonetheless, mixed woods have a series of advantages that can make them appealing to stakeholders. First, mixed stands are usually not so badly affected than monocultures by extreme conditions linked to climate change such as wind throw, pest outbreaks or drought (Kelty 1992; Felton et al. 2010; Felton et al. 2016). Second, mixed stands still enjoy relatively high timber productivity (Kelty 1992; Gamfeldt et al. 2013). This is due to broadleaved trees reducing soil acidification typical in monocultures of preferred coniferous species (Jönsson et al. 2003) and to niche partition (i.e. use of partly different resources) between species (Kelty 1992; Amoroso and Turnblom 2006). Eventually this translates into higher carbon sequestration. For instance, Gamfeldt et al., (2013) calculated that forest stands with five species incorporated 11% more carbon to the soil than monocultures. Third, the number of tree species in a stand is directly related to general species diversity (Felton et al. 2010; Gamfeldt et al. 2013; Felton et al. 2016). This is not surprising as several specialist forest species use resources from one or just a few species, while other species benefit from heterogeneity of resources found in mixed forests.

Another fundamental approach to reconcile wood production with other functions is through retaining trees in the harvesting interventions (so called green tree retention) (Rosenvald and Lõhmus 2008; Gustafsson et al. 2012; Fedrowitz et al. 2014). From the forestry perspective, leaving trees in the stand can be cost-efficient way to facilitate forest regrowth. Trees of the commercially interesting species are sometimes left as seeding trees or as shading trees for sun-intolerant species of interest. Also, given that a significant portion of trees retained it is possible prevent the rise of the water table in the stand that can be detrimental for forest regrowth. Additionally, some commercially less interesting tree species (e.g. Aspen) may be best left untouched to avoid sucker regeneration (Frey et al. 2003). Beyond timber production, tree retention can have important implications for other ecosystem services and biodiversity by preserving biological resources, promoting stand heterogeneity and connectivity. In this sense one can expect that the output of the tree retention option is directly related to what percentage of the forest is retained and to what tree species are spared. For instance, although stands with retained trees contain more threatened species than clear-cut forests, its relevance is related to how much is left (Vanha-Majamaa and Jalonen 2001; Hyvärinen et al. 2006; Aubry et al. 2009; Santangeli et al. 2013). Aggregating retained trees also seems to be more successful in retaining biodiversity value (Carlén et al. 1999; Aubry et al. 2009) as opposed to the preferred practice in silviculture of random tree retention when seeking seeding and shadow trees. It has also been recommended that green tree aggregations should be situated in areas rich in threatened species, typically moist sites (Vanha-Majamaa and Jalonen 2001) and in different woodland key habitats (Timonen et al. 2010) if they are present in the stand. Given that a large portion of species, especially invertebrates and fungi, are dependent on deadwood, it is also advisable to leave dead or decaying trees also because the economic value of those is rather small, and this may even have positive economic effects via better seedling establishment (Alaspää et al. 2015).

A special case of green tree retention is the continuous cover or uneven-aged management, where the upper stratum of the forest is removed (also called thinning from above) inducing faster growth of the lower strata (Pommerening and Murphy 2004). Mounting evidence shows that this approach can be equally or more worthy for forestry purposes as the conventional model with clearcuts (Laiho et al.

2011; Kuuluvainen et al. 2012; Rämö and Tahvonen 2014; Tahvonen 2015). Additionally, it is considered more resilient than the even-aged models to natural disturbances like windthrow, insect pests or fires partly because of the different age classes and more alternatives to recovery after disturbances (O'Hara and Ramage 2013; Felton et al. 2016; Pukkala 2016). It can also outperform the conventional approach when considering some other ecosystem services, e.g. climate regulation (Pukkala 2016) but not all, e.g. collectable goods (Peura et al. 2016). From a biodiversity point of view, the largest contribution of continuous cover forestry is on the grounds of habitat connectivity; a key threat to intensively managed landscapes (Fahrig 2003; Nordén et al. 2015). One has to note that despite the concept of continuous cover forestry has received more interest for restoring biodiversity it is obvious that a landscape covered by intensively managed uneven-aged forests cannot fulfill the demands of all species, particularly specialist species that rely on large amounts of deadwood or require closed forest structures. Uneven-aged management can, however, improve connectivity between more suitable patches for most of those more selective species. Still, applying permanent tree retention where some trees are allowed to grow, die and decay within the context of continuous cover forestry is likely to greatly improve potential of this management model.

The clear-cut approach in intensively managed forests often includes a number of interventions with thinning from below to select the tree species of commercial interest, to promote faster growth of the remaining trees and to prevent natural mortality (Daniel et al. 1979; Bamsey 1995). Despite of the fact that it takes longer for a non-thinned stand to reach equally large trees than a thinned stand, the additional dead wood generated in non-thinned stands can render non-thinning a cost-efficient strategy for promoting those species dependent on deadwood (Tikkanen et al. 2007; Tikkanen et al. 2012; Mönkkönen et al. 2014).

Under the risk of potential hazards to forest, managers might be moved to shorten the rotation times on the clear-cut model. However, while reducing rotation cycles may be good against some risk agents like windthrow, cambium feeders or root rot, at the same time this strategy increases the risks of regeneration pests and fire (Björkman and Nimelä 2015; Roberge et al. 2016). At the same time, shortening rotation typically has negative effects (vice versa for extending rotation) on relevant attributes for forest biodiversity (e.g., connectivity of old forest, reduction of dead wood, reduction in understory complexity), and regulating and supporting services (e.g., carbon storage, soil quality. hydrologic integrity) (Jandl et al. 2007; Pawson et al. 2013; Pihlainen et al. 2014; Triviño et al. 2015b; Felton et al. 2016; Roberge et al. 2016).

5.2. Policy tools to enforce and promote management for multi-objective forestry

There is an interlinked array of policy tools designed to promote sustainable forestry. National forest legislation set minimum standards for forest management. In addition, international forest certification standards are developed to promote sustainable forest management. The forest certification standards use the compliance with national laws as a starting point but have some elements that require more than the national legislation.

In Nordic countries two voluntary forest certification standards prevail: the Forest Stewardship Council (FSC) and the Programme for the Endorsement of Forest Certification schemes (PEFC). Both systems include certificates for forest management and for the chain-of-custody. The standard for forest management sets requirements for responsible forest management, whereas the chain of custody is a mechanism for tracking certified material from the forest to the final product to ensure that the wood, wood fiber or non-wood forest produce contained in the product or product line can be traced back to certified forests. In Finland, 17,6 million hectares of the 20,3 million hectares of productive forest land are certified with the PEFC and approximately a million hectares with the FSC. In Sweden 11,5 million hectares are certified with the PEFC, and FSC certificates are given to 12 million hectares, which corresponds to almost half of the productive forest land in Sweden. Globally 300 million hectares of forest are certified with the PEFC and 190 million with the FSC (FCS International 2017[2], PEFC International 2017[3]).

The FSC and the PEFC schemes consider ecosystem services and biodiversity. The standards do not use the concept of ecosystem services but explicitly account for multiple benefits of forests and their multifunctionality (FSC Finland 2010). A forest owner is required to acquire information on the occurrences of threatened species and plan the management activities safeguarding protection of rare, endangered species and their habitats. For instance, fellings are forbidden during the bird nesting season in both standards. Forest owners are also encouraged to ensure adequate resources for the protection of biological diversity, and soil and water resources. The standards also require consideration of recreation values, and forest owners are required to take into consideration routes and structures for ecotourism and recreational use when planning forest management activities. The PEFC standard also recognizes the carbon sequestration in forests and requires that forest stands should be preserved as carbon sinks (e.g. PEFC Finland 2014). In addition, the Nordic everyman's rights or the freedom roam is safeguarded in the standards. In environmental considerations the FSC is more demanding than the PEFC (Gulbrandsen 2005). The most noticeable difference is that FSC requires that at least 5% if the productive forest is permanently set aside (FSC Finland 2010; FSC Sweden 2010).

Forest certification can contribute to biodiversity conservation but the level and the effect depend on the requirements of the forest certification scheme, and its implementation (Elbakidze et al. 2011). According to Gullisson (2003), forest certification can generally benefit biodiversity conservation in three ways by

[2] https://ic.fsc.org/en (accessed 3.4.2017)
[3] www.pefc.org (accessed 3.4.2017)

1) reducing logging pressure on high conservation value, 2) preventing land use change, and 3) improving ecological value of certified forests for biodiversity. In Nordic countries, national Forest Acts together with other environmental legislation set minimum environmental requirements for forest management. Nieminen (2006) concludes that direct, additional ecological benefits from forest certification in Finland are small compared to the environmental requirements in national legislation. However, the Forest Acts in Finland and Sweden only require conservation of biodiversity values already present, whereas the FSC standard demands the creation of new values by creating snags, leaving retention trees and prescribed burning. These restoration measures are important in creating new structures such as old trees, deadwood, and deciduous trees and disturbances like fire that have decreased in managed forests (Johansson et al. 2013).

Although a direct cause-effect relationship between forest certification and environmental benefits is difficult to show (Rametsteiner and Simula 2003; Gulbrandsen 2005; Johansson and Lidestav 2011), studies conducted in Finland and Sweden indicate that forest certification can bring additional environmental benefits compared to requirements of the legislation. These benefits result mainly from criteria for retention trees, prescribed burning, areas set aside from management, and restrictions in management operations in valuable habitats (Nieminen 2006; Johansson et al. 2013). Tree retention aims to reduce the intensity of harvest in clear-cutting by leaving single trees or tree groups and has several important functions, it 1) can “lifeboat” species over the regeneration phase, 2) increases structural diversity in young production forests, 3) enchase the connectivity in forest landscape, 4) promote species dependent on dead wood and living trees in early successional environments and 5) sustain ecosystems functions such as nitrogen retention (Gustafsson et al. 2010; Kruys et al. 2013). The Finnish legislation has no specific requirements for retention trees whereas as according to the Swedish legislation 2 to 10% of timber value should be left after harvesting, prioritizing rare and broadleaved species as well as old trees and nesting trees (Johansson et al. 2013; Finnish Forest Act 2014). The FSC scheme gives specific quantitative and qualitative requirements for retention trees. Neither the Finnish nor the Swedish legislation has requirements for prescribed burning. The Finnish and Swedish Forests Acts list key habitats and require that these habitats are managed and used in such a manner that the general conditions for the preservation of these habitats important for the biological diversity of forests are safeguarded (Finland) or that damage from forestry is minimized or avoided (Sweden) whereas the FSC standard does not allow any commercial forest measures in these key habitats (FSC Finland 2010; FSC Sweden 2010; Johansson et al. 2013).

Forest certification has also indirect environmental benefits. Forest certification has harmonized forest management practices, improved communication among stakeholders and clarified practical instructions for forest management (Nieminen 2006). In addition, the certification schemes have increased environment awareness and consideration of environmental aspect among forest owners (Johansson and Lidestav 2011) and employees of the forest sector (Nieminen 2006). Studies on corrective action requirements issued by certification bodies indicate that the auditing process improves the management of existing forests because the corrective action requirements must be addressed to obtain the certificate (Gullison 2003; Auld et al. 2008).

Forest certification has positive but limited impact on sustainable forest management, maintenance of ecosystem services and biodiversity conservation. For example, tree retention may reduce harmful consequences of clear-cutting on biota but it cannot maintain the characteristics of intact mature forests (Gustafsson et al. 2010). In addition, the long-term benefits of tree retention to red-listed species are questioned (Johansson et al. 2013). The forest certification integrates conservation measures into production forests, which complies with sustainable land use strategy of the Millennium Ecosystem Assessment (2005) promoting different ecosystem services with the same land use. However, according to Johansson et al. (2013) forest certification together with environmental legislation does not guarantee biodiversity conservation in Sweden. This is because the levels of dead wood, the share of deciduous trees, areas for set aside and other environmental consideration in the standards do not meet the thresholds identified in the scientific literature (Johansson et al. 2013). In conclusion, forest certification can alleviate the negative effects of forest management and be seen as a complementary, but not substitutive, measure to formal forest protection (Rametsteiner and Simula 2003; Elbakidze et al. 2011; Johansson et al. 2013).

In addition to forest certification, new payment schemes have been proposed to guide forest management with economic incentives (e.g. Farley & Costanza (2010)). According to Engel et al. (2008) payments for ecosystem services (PES) are voluntary transactions for a well-defined environmental services that a service buyer acquires from a service provider, who secures services provision of this service. Besides traditional tax and subsidy instruments these new PES instruments can promote the provision of public goods in forestry (see review by Ollikainen (2016)). For example, subsidize-and-tax model and carbon rent models have been proposed as alternatives to implement payments for carbon sequestration and storage for forest owners (van Kooten et al. 1995; Lintunen et al. 2016). An interest to participate to carbon offset mechanisms has been shown, for example, among Norwegian forest owners (Håbesland et al. 2016). Previous studies on offset mechanism for ecosystem services and biodiversity have introduced new, promising instruments for policy-makers. However, the development of operational PES systems in forestry is in its infancy (Ollikainen 2016). One example of existing PES system in boreal forests is Southern Finland Forest biodiversity Programme METSO. The METSO programme is a voluntary-based conservation program aiming to halt the ongoing decline in the biodiversity of forest habitats and species, and establish stable favourable trends especially in Southern Finland’s forest ecosystems. The program offers monetary compensation for forest owners for permanent or temporary protection of forest, or support for nature management in forest habitats (METSO 2015).

5.3.Evaluating conflicts

Conflicts exist when there is a need to balance between the desired outcomes of different objectives. Conflicts between different functions and services of the forest ecosystem can be evaluated in a variety of fashions. One common way to evaluate conflicts between two services or functions is by evaluating their trade-offs (King et al. 2015). The key idea behind the trade-off analysis is to gain an understanding of loss in one objective that must be endured for the benefit of a second objective. The trade-off analysis can be done through an optimization framework, which seeks to find the maximum of one objective while a second objective is constrained to a specific proportion of the specific objectives maximum. Though an iterative method of adjusting the proportion required, a frontier can be established, and the trade-offs which occur between the objectives can be examined.

With a two dimensional problem, the conflict between each objective is easy to graph and describe to policy makers. When the problem involves the consideration of conflicts between multiple ecosystem services, describing and portraying the conflicts become complicated. By analyzing the potential change in management prescriptions, the conflicts can be examined through a multi-objective optimization problem (Miettinen 1999). Through an interactive process (such as Nimbus; Ojalehto et al. 2007) it is possible for stakeholders to define their preferences between the selected set of criteria. These kinds of interactive processes work well when there is a clear decision maker who can accurately define his/her preferences. When considering the potential stakeholders in landscape level planning, each stakeholder may provide a different perspective of how to evaluate the environment. Some of the stakeholders may not be able or willing to accurately define their preferences (i.e. Nordström et al. 2009). In these group decision-making processes, the focus is to promote understanding between stakeholders and improve the acceptability of the management plan (Hjortsø 2004).

Rather than simply evaluating alternative management plans, is it also possible to quantify the conflicts between objectives (Mazziotta et al. 2017). When evaluating ecological objectives, the case may exist where some objectives are compatible and maximizing one objective causes only a small decrease in the possible maximum of the other objective. This is, for example, the case with the habitat availability of the capercaillie (*Tetrao urogallus*) in Finland which can be maximized with fairly small reductions in timber production (Mönkkönen et al. 2014). Alternatively, when the objectives are not compatible, maximizing one objective will cause a large decrease in the possible maximum of the other objective; this is the case e.g. between timber production and carbon storage. Mazziotta et al. (2017) developed an index of compatibility, which is essentially the percentage of an objective (x) possible when maximizing a second objective (y). These compatibility indices are not symmetrical, so the compatibility index of x when maximizing y, does not necessarily equal the compatibility index of y when we maximize x.

From decision-making and forest management planning perspectives, trade-off analysis can be divided into two classes (Felton et al. 2017). The problem can be formulated as a 'how to' questions focusing on identifying a single or limited set of management alternatives from a larger set, which meet a specific set of objectives and desired constraints. Often this approach provides a mean of determining the optimal values for a set of objectives. In contrast, a 'what-if' approach aims at understanding the implications of different scenarios regarding 'what' will be the implications for the objectives, 'if' this policy or management intervention takes place.

Along the lines of how-to approach, it is possible to develop an objective function, which finds a suitable compromise solution for a set of objectives. Mazziotta et al. (2017) suggest the use of a compromise programming formulation (Yu 1973; Zeleny 1982), where the idea is to minimize the maximum deviations from the ideal value for all objectives under consideration, i.e. to minimize total losses due to a decision. The solution produced demonstrates how maximizing a set of objectives requires that no objective obtains their maximum potential values. One must consider the preferential interpretation of the objective function (Diaz-Balteiro et al. 2012).

Quantification of conflict between ecological objectives can enhance our understanding of trade-offs between various conservation objectives. However, the key conflict is between economic and ecological objectives. Mazziotta et al. (2017) highlighted how much the ecological objectives would suffer by requiring 95% of the maximum net present value be extracted from the forest. As expected, the achievement of all objectives decreased substantially. On average, the values of the ecological objectives decreased about 20% from the case when only ecological objectives were considered. From a conservation perspective, this is a substantial decline. To resolve the conflicts caused by the economic requirement, a reduction to 2/3rds of the maximum NPV would be required (Mazziotta et al. 2017). From an economic perspective, this may not be an acceptable reduction. From this point, the trade-off between ecological objectives and economic demands are evident, and will require some compromise.

Alternatively, rather than comparing specific management alternatives to evaluate the conflicts between criteria, we can view the issue through shifts in policy. Policy suggestions can be evaluated by analyzing the statements, objectives and goals using the ‘what-if’ approach. Optimized forest management plans can be made by making specific assumptions which correspond to the policy statement. The conflict between criteria of interest can then be evaluated through the systematic relaxation of the assumptions made, i.e. adjusting the ‘if’ part. So rather than finding a specific solution which best fits the policy, we can find a set of solutions (which are Pareto optimal; any positive adjustment in one objective must be offset by a negative adjustment in another objective) which highlights the potential conflicts between criteria.

5.4. Consequences of increasing timber harvesting rate - A case study

The Finnish government is currently promoting the growth of the bioeconomy, where the use of renewable resources is encouraged (Ministry of Employment and the Economy 2014). For forestry in Finland, this is being implemented through an increase towards the maximum annual allowable harvest. Currently, forest operations in Finland are not near the maximum annual allowable harvest (Peltola 2014) but because of conflicts, increases in harvests will most likely cause harm to other ecosystem services such as climate regulation, recreational use and biodiversity. Through modeling and simulation by use of forest management software, it is possible to predict future forest resources according to different management regimes and then evaluate the ecological performance of the forest. By adjusting the management alternatives for different portions of the forest, we can predict how the increased use of forest resources will impact the forest's other uses. To conduct the analysis in a more systematic fashion, optimization can be used to determine the optimal spatial allocation of the specific management regimes.

For this example, we evaluate the trade-offs and potential conflicts for a variety of criteria at a watershed level when there is a requirement for even-flow of timber harvested. We apply a what-if approach with the policy statement of maximizing the even-flow of timber over the duration of the planning horizon. We compare this policy shift to the case were only a proportion of the maximum even-flow of timber will be harvested. In principle, the amount of timber provided by the forest will be constant for all periods during the planning horizon. However, the required timber harvested will be less than the theoretical maximum.

For this example, we analyze a watershed located in central Finland. The region consists of slightly over 5,060 ha (composed of 3,356 stands) of managed boreal forests. This region is composed of primarily three tree species, by Norway spruce (*Picea abies (L.) Karst.*) composes 57% of the total volume, Scots

Pine (*Pinus sylvestris L.*) is 26% and Silver birch (*Betula pendula*) is 16.5%. The remaining (0.5%) component consists of other deciduous trees (*Betula pubescens,* and *Populus tremula*). Figure 6 describes the age distribution of the forest, which is rather even – with a slight bulge in the age classes of 60- 80.

To predict the possible future states of the forest holding, we used the forest management software SIMO (Rasinmäki et al. 2009). For each stand, 19 alternative management regimes were simulated. All stands included the option of not conducting any actions in the forest, otherwise to set the stand aside. Other regimes followed the recommendations from the forest management guide (Äijälä et al. 2014), including slight variations on these recommendations (such as to delay or speed up the timing of the management actions, Table 2). Additionally, a management regime corresponding to a method of conducting continuous cover forestry (CCF) was included (Pukkala et al. 2013).

Once the alternative management alternatives were simulated, the next step is to find the combination of management actions which best fulfills the objectives of the stated policy. For this case, the policy is to increase harvests to the maximum allowable sustainable harvestable yield. At the country level, this value is computed by the authority responsible for natural resources, and take into account the growth rate of the forest, protected forested areas to evaluate what is the maximum sustainable harvestable yield. For a watershed level, the maximum sustainable harvestable yield could be comparable to the maximum even-flow of timber for the area under consideration. Finding the maximum even-flow of timber is an optimization problem which maximizes the first period harvested yield, subject to a constraint that for all other periods under consideration, the yield cannot be less than the first period yield. The object function is [Model 1]:

$$Max\ z = \sum_{k=1}^{K}\sum_{j=1}^{Jk} c_{kj1}x_{kj} \qquad [1]$$

subject to:

$$\sum_{k=1}^{K}\sum_{j=1}^{J_k} c_{kj1}x_{kj} \leq \sum_{k=1}^{K}\sum^{J_k} c_{kji}x_{kj}, i = 2, \dots, I \qquad [2]$$

$$\sum_{j=1}^{J_k} x_{kj} = 1, k = 1, \dots, J \qquad [3]$$

$$x_{kj} \geq 0\ \forall\ k = 1, \dots, K, j = 1, \dots, J_k \qquad [4]$$

where K is the total number of stands of the forest holding, J_k is the number of management regimes for stand k, c_{kji} is the amount of timber which is available by managing stand k according to management regime j at time period i, x_{kj} is the decision to manage a specific proportion of stand k according to management regime j, I is the total number of time periods under consideration and z will represent the maximum amount of even-flow timber for the duration of the planning horizon. Constraint [2] represents the requirement for even flow, constraint [3] is an area constraint, requiring that the sum of the decisions must equal 1, constraint [4] is requires that the decision variable is not a negative number. In forest management, this type of problem is referred to as a model I problem (Johnson and Scheurman 1977), and one of the key features is that the spatial integrity of the stands is maintained. This allows for easy mapping of which management regimes are proposed for which stand.

Once we have evaluated the maximum amount of even-flow of timber, we can then analyze the possible impacts of relaxing the specific constraints. In this specific case, we will reduce the required annual timber harvested and simultaneously maximize a set of normalized ecosystem services. Other than timber provided, we are interested in promoting the production of bilberries (*Vaccinium myrtillus*), increasing the amount of carbon stored in the forest, increasing the amount of deadwood in the forest and increasing the habitat suitability for a set of species. We included six vertebrate species representing a wide spectrum of habitat associations and also conservation and social values ranging from game birds (capercaillie, hazel grouse) to red-listed (Siberian flying squirrel) and indicator species (three-toed woodpecker, lesser-spotted woodpecker and long-tailed tit) (Mönkkönen et al. 2014).

To do this, we employ a different optimization model. This model optimizes the normalized set of ecosystem services, while introducing an additional constraint to ensure that the first period timber harvest meets a proportion of the maximum even-flow of timber (z). The objective function of this model is [Model 2]:

$$Max \sum_{k=1}^{K}\sum_{j=1}^{J_k}\sum_{i=1}^{I}\sum_{l=1}^{L}\frac{d_{kjil}x_{kj}}{y_l} \quad [5]$$

subject to:

[6] $$\sum_{k=1}^{K}\sum_{j=1}^{J_k} c_{kj1}x_{kj} \geq z * p$$

and constraints 2, 3 and 4, where d_{kjil} is the value of the specific ecosystem service (l) of interest from the set of all ecosystem services under consideration (L), y_l is the normalization constant for the ecosystem service l, and p is a parameter which represent the desired proportion of the maximum even-flow of harvest. The objective function [5] maximizes the normalized set of ecosystem services while constraint [6] requires that the first period timber is at a specific proportion of the maximum even-flow. Constraint [2] ensures that all future timber harvest is not lower than the first period harvests.

By running Model 2 iteratively, it is possible to evaluate how the relaxation of the even-flow constraints will impact the other ecosystem services. For those ecosystem services which are negatively impacted by increased harvests, by relaxing the even-flow constraint, it is expected that those ecosystem services will increase. Alternatively, for those ecosystem services which are positively impacted by increased harvests, relaxing the even-flow constraint will reduce those ecosystem services. To provide an example, we have conducted this analysis by adjusting p from 0.6 to 0.95 in increments f 0.05 and with p =0.975.

To provide an elaborative description of how tightening the timber requirement impacts the selected ecosystem services, we set the case where p = 0.6 to be the starting point, and all other iterations are compared to that case. The current harvest rate in Finland varies annually between 60-75%, and thus the starting point roughly corresponds with the current situation.

Figure 7 highlights the trade-off between increasing timber harvests and the biodiversity and ecosystem service indicators of interest. With an initial small increase in required timber harvests (i.e. from p = 0.6

to $p = 0.65$), the decrease in the ecosystem services other than timber is rather small; however, with a continued increase in timber harvested the impact on the ecosystem services becomes rather severe. Alternative indicators of ecosystem services and biodiversity, nonetheless, show different patterns of decline. Carbon storage linearly declines monotonically with increasing harvest rate (proportion of the maximum even-flow of harvest) whereas bilberry yield shows first slight increase and very steep decline at very high levels of timber harvesting. Thus, each increment in harvest rate results in equal reduction in carbon storage but bilberry yields can be maintained at the initial levels until timber harvesting exceeds 95% of the maximum. Bilberry benefits from open forest structures, e.g. after thinning, but suffers from clear-cutting (Peura et al. 2016) explaining the non-linear response to increased timber harvesting.

The shift of increased sustainable harvest implies an intensification of forest use. When we move towards the maximum sustainable harvest, a greater proportion of forest area moves from being 'set aside' towards other management regimes (Fig. 8). When the required sustainable harvest is relatively low, the majority of the area that is harvested is done using the management regime of CCF. As the required level of harvest increases, all management regimes which conduct harvests increase and forest managed according to CCF also increases. Only at the maximum sustainable harvest level does the traditional management regime become the dominant method of managing the forest. However, there is still a large component of the forest being managed with CCF, and still some forest being managed without the conduct of thinnings prior to final felling.

With optimization processes, finding an optimal solution to a particular case is rather straight forward, the difficulty remains in being able to implement the solution. In Scandinavia, being able to implement these kinds of solutions requires motivation for the policy makers, but more importantly requires acceptance and a majority of compliance to be able to achieve these kinds of results.

6. Towards sustainable management of boreal forests – landscape level planning

Natural boreal forests are disturbance driven ecosystems. Some consequences of industrial forest harvesting resemble the effects of natural disturbances such as forest fire. Although the analogy between forest management and fire disturbance in boreal ecosystems has some merit, it is important to recognize that natural disturbance and human-induced disturbances differ considerably (Bergeron et al. 2002; Table 1).

Research has shown that many ecosystem services and biodiversity are in conflict with intensive timber production in boreal forests. The conflicts stem from the changes forest management causes to the structure and dynamics of forests. The conventional regeneration forest management including site preparation, planting trees and 1-3 thinning operations before final felling by clear-cutting (see section 3.2.), if applied consistently on the entire landscape, causes ecological (Mönkkönen et al. 2014) and social costs (Triviño et al. 2015a). Biodiversity losses are because a proportion of species do not have adaptations to cope with changes in resource availability, habitat structures and their spatial configuration. Also ecosystem functioning is altered. There are some conflicts among non-timber ecosystem services and moreover, some between biodiversity and ecosystem service indicators but these conflicts are generally weaker than for timber provision (Pohjanmies et al. 2017, Triviño et al. 2017). Because alternative management regimes such as continuous cover forestry (Peura et al. unpublished) or retaining from thinnings (Mönkkönen et al. 2014) are more beneficial for some objectives but worse for some others than the intensive Fennoscandian forest management regime, no management regime alone is optimal. Therefore, the best option for multiple objectives would be to diversify management, but finding an efficient balance among alternative management options requires careful planning.

Forest management planning can be conducted for a wide range of interests and for varying spatial scales. For forest owners who own small parcels of land, their interests may be purely financial or they may be interested in managing the forest with an aim to enhance the ecological functions of the forest. A key issue in forestry planning is the spatial scale. At stand scale, reconciling alternative economic, ecological and social objectives is difficult because only one management regime can be applied at a time. But at the landscape scale, it may be possible to find plans, i.e. combinations of management regimes, that provide acceptable compromise solutions because of flexibility provided by increasing combinations of stand level management decisions. Large scale planning would be desired also because some ecological functions depend upon stand structures operate at scales larger than a single holding (i.e. habitat requirements of species; Mönkkönen et al. 1997; Kurki et al. 2000). The costs of increasing the scale are obviously the increasing complicatedness of the problems and the difficulty of putting plans into practice when they encompass several forest holdings. Because of the dynamism of the forest ecosystems, forest planning should also consider long time perspectives and future generations: the consequences of today's management decisions may realize only after several decades.

Resolving the conflicts necessitates applying a mixture of management regimes on a landscape, i.e. applying the conventional regeneration harvest regime on a proportion of stands and alternative regimes on others to better achieve multiple objectives. The desired combination of alternative management regimes depends on the decision maker's preferences and objectives. Even when the decision maker aims at maximizing timber revenues, she should not apply the recommended management consistently but only on a fraction of stands (Mönkkönen et al. 2014; Fig 8). With decreasing emphasis on timber production and increasing emphasis on non-timber benefits and biodiversity, the utility of the conventional regeneration harvest management further decreases. Retaining from intermediate thinnings, extending stand rotations, and increasing the amount of area set aside from forestry seem necessary to safeguard biodiversity and non-timber ecosystem services (see e.g. Triviño et al. 2016).

The good news is that typically pair-wise conflicts are solvable at relatively low cost if land-use planning is done at the landscape scale (Pohjanmies et al. unpublished). For example, giving up 5% of the maximum timber revenues enabled maintaining up to 278 % more habitats for species (Mönkkönen et al. 2014) or increasing the landscape's capacity to store carbon by 9% and to sequester carbon by 23% (Triviño et al. 2015a).

The bad news is that attaining to high values for more than two objectives at the same time seems very difficult. In fact, the objective of having high timber harvest rates aggravates the conflicts among non-timber objectives as shown by Triviño et al. (2016). Triviño et al. (2016) aimed at reconciling timber production with provisioning of other ecosystem services (i.e. store carbon for climate regulation) while maintaining suitable habitat for forest biodiversity. They applied seven alternative forest management regimes, ranging from the current recommended regime to set aside, using a forest growth simulator in a large boreal forest production landscape. With multi-objective optimization, they identified the optimal combination of forest management regimes to minimize the trade-offs between timber harvest revenues, carbon storage and biodiversity maintenance. Results indicate that it was not possible to achieve high levels of carbon storage or biodiversity if the objective of forest management was to maximize timber harvest revenues. However, with small reductions of timber revenues (1- 5%) it was possible to greatly increase the multifunctionality of the landscape, especially the biodiversity indicators (47-90% of the maximum deadwood and 65-88% of the habitat availability) (see Fig. 9). For more severe reductions in timber harvest revenues, e.g. 80-95%, it was possible to almost achieve the maximum levels for both carbon storage and biodiversity indicators (see Fig. 9). Even with modest levels of timber objective, there was a strong trade-off between carbon storage and biodiversity objectives and both objectives remained far from their maximum values. When timber objectives were relaxed, close to maximum levels for both carbon storage and biodiversity objectives could be achieved.

The results also showed that no management regime alone is able to maximize timber revenues, carbon storage and biodiversity individually or simultaneously, and that a combination of different regimes is needed to resolve the conflicts among these objectives (see Fig. 10). Forest management actions, alternative to the conventional regeneration harvest, such as reducing thinning intensity, extending the rotation period and increasing the amount of area set-aside from forestry may be necessary to safeguard biodiversity and non-timber ecosystem services in Fennoscandia. They concluded that it is possible to reduce the trade-offs between different objectives by applying diversified forest management planning at the landscape-level.

The example above suggests that strong emphasis on timber production at the landscape scale makes it impossible to simultaneously achieve high levels in more than one other objective no matter how landscape is managed. Intensifying timber production results in increasingly strong conflict biodiversity protection and climate regulation via carbon storing, even though initially biodiversity and carbon-related ecosystem services were not in conflict. We can therefore conclude that the current objective of bioeconomy policies to considerably increase timber flow from forest is not sustainable in Fennoscandian forests (see also section 5.4.) because already at current timber harvest rates biodiversity is at risk (Hanski 2000; Mönkkönen et al. 2014), many ecosystem services have declined (Pohjanmies et al. unpublished), and even with very careful landscape level planning and management, optimization resolving the conflicts among several objectives is not possible.

Two alternative forest harvesting strategies are proposed to meet timber demands with other objectives: land sharing, which combines timber extraction with other objective protection across the entire concession; and land sparing, where high intensity harvesting in one place is combined with the low intensity or no harvesting somewhere else (Edwards et al. 2010). Discussion above clearly suggests that sustainable forest use via landscape sharing would require rather low overall intensity of timber production. Thus, if timber requirements are large landscape sparing becomes a desired option for economic and social reasons. Segregating the role of landscapes is justified also from the mere nature conservation point of view. Several ecological processes underpinning both ecosystem services as well as biodiversity have a minimum threshold that is context dependent. Ecological research has concluded that if a limited area of species habitats can be protected, they should be protected in spatially aggregated clusters rather than as randomly scattered fragments. This will generally increase the conservation benefits for a given total area protected (Rybicki and Hanski 2013). The Nagoya agreement recognizes the need to focus on 'especially areas of particular importance for biodiversity and ecosystem services, … ecologically representative and well-connected systems of protected areas … integrated into the wider landscapes' (http://www.cbd.int/doc/strategic-plan/2011-2020/Aichi-Targets-EN.pdf, strategic goal C, target 11). Thus, we conclude that regional forest resource management planning should start differentiating landscapes where environmental and social objectives have priority over timber production landscapes.

Finding a balance between timber production landscapes and multiuse landscapes is yet another challenging objective for natural resource management. Hanski (2011) suggested the third-of-third rule of thumb, which implies that conservation landscapes would cover one-third of the total region, and within conservation landscapes one-third of the total area be protected resulting in roughly 10% level of set asides. This is less than the target set in Nagoya, but this 10% would be additional to the existing protected areas. Further, this 10% figure is only a double amount of set asides required by FSC certification standards. Within conservation landscapes, biodiversity and people coexist, and the ecosystem services provided by biodiversity and natural habitats is an integral part providing direct benefits to local communities and to the society at large.

In boreal forest settings, Hanski's (2011) suggestion would mean concentrating timber production on two-thirds of the total area within a region. Also in timber production landscapes, applying multiple management regimes, including extended rotations, refraining from thinnings and continuous cover forestry, is necessary for maximum timber values. Multi-use landscapes would cover the remaining one third. Here set-asides comprise a prominent proportion of area but still managed forest for multiple purposes dominate. There is a growing support for management in production forest to recreate conditions found within a given region by natural disturbances with the rationale that the process and species are adapted to such conditions (Attiwill 1994; Burton et al. 2003; Kuuluvainen 2009; Kuuluvainen and Aakala 2011; Larocque 2016). However, vast forest areas of the boreal biome have much lower natural disturbance rate than that required from forestry sectors. Therefore, it has been suggested to conduct functional zoning in which one part of the region focuses on protection, another part is managed according to natural disturbance dynamics, and the rest is devoted to intensive forestry (Seymour and Hunter 1992; Côté et al. 2010; Strengbom et al. 2011; Lindenmayer et al. 2012; Tittler et al. 2016).

According to mitigation hierarchy, avoidance is often the easiest, cheapest and most effective way of reducing potential negative impacts of any development (http://www.thebiodiversityconsultancy.com/approaches/mitigation-hierarchy/). Therefore, development, e.g. forestry, should be concentrated on areas where negative impacts can be avoided. Kareksela et al. (2013) developed the method of inverse spatial prioritization and applied it to land-use allocation for peat-land mining. This approach can also be used to identify multi-use landscape vs. timber production landscapes within a region. In practice, this means finding landscapes with the lowest timber but highest biodiversity and non-timber ecosystem service values (multi-use landscapes), and conversely, landscapes with the highest timber production potential and lowest biodiversity and non-timber ESS values (timber production landscapes). We suggest that a regional approach where timber production landscapes are separated from multi-use landscapes using systematic zoning tools, such as the inverse spatial prioritization, would be a promising way of reconciling multiple conflicting objectives for boreal forests and their management.

7. Concluding remarks

In this chapter, we have provided a general overview of boreal forest ecosystems and their management for timber production, maintaining biodiversity and ensuring the flow of non-timber ecosystem services. We showed evidence that in boreal production forests the conflicts between the primary provisioning service of timber and other benefits are real, severe, and challenging to solve. This in line with the more general finding that unbalanced focus on one or few provisioning ecosystem services typically results in severe trade-offs (Howe et al. 2014). Research into the processes affecting the supply of different forest ecosystem services may assist in designing forestry practices and planning management regimes that protect diverse forest benefits. We need to understand the mechanisms causing trade-offs and recognize situations where they likely occur when we want generate solutions to these trade-offs.

Forestry's effects on ecosystem services may be generated at various spatial scales (e.g. a single stand, a landscape) that are relevant for different forms of forest ownership and management (e.g. a private forest holding, a state-owned forest). We concluded that long-term landscape level planning that simultaneously takes into account alternative objectives and the capacity of each land parcel (stand) to meet these objectives is a necessary, but not sufficient, condition for sustainable forest use. In addition, regional level planning where the roles of landscapes are differentiated is needed. Maintaining a diverse set of forest services requires coordination of activities among forest owners. Therefore, we need incentives for landowners to make decisions that reflect the value of ecosystem services and biodiversity conservation in general. This is, however, particularly challenging in several parts of the boreal forests like Fennoscandia where ownership of forest land propriety is heavily divided.

An extra challenge in regional or landscape level forest planning stems from the fact that biodiversity or alternative ecosystem services provide goods and benefits to different stakeholders. Some commodities such as timber are considered private property, benefiting primarily the landowner while others are considered public goods. For example, climate change mitigation provides a global benefit by reducing atmospheric CO_2 levels, while water quality regulation, and recreational use, natural collectable products (e.g., berries and mushrooms) profit mostly the local community. Private landowners typically lack the incentive to manage land to provide ecosystem services and biodiversity conservation benefits in cases where the benefits produced on their land accrue to others. A recent review (Howe et al. 2014) showed that trade-offs among ecosystem services are especially likely to occur when one of the services is a provisioning service and one of the stakeholders involved has a private interest in the services. In summary, besides new management practices and planning tools, new regulations and/or incentives are required to improve the protection of public interests in boreal production forests.

Literature cited

Ahti T, Hämet-Ahti L, Jalas J (1968) Vegetation zones and their sections in northwestern Europe. Ann Bot Fenn 5:169–211.

Alaspää K, Muukkonen P, Mäkipää R (2015) Lahopuun merkitys kasvualustana eteläboreaalisen vanhan luonnon - tilaisen kuusimetsän uudistumisessa. Metsätieteen Aikakauskirja 237–245.

Amoroso MM, Turnblom EC (2006) Comparing productivity of pure and mixed Douglas-fir and western hemlock plantations in the Pacific Northwest. Can J For Res 36:1484–149.

Angelstam P, Kuuluvainen T (2004) Boreal forest disturbance regimes, successional dynamics and landscape structures – a European perspective. Ecol Bull 51:117–136.

Angelstam P, Pettersson B (1997) Principles of present Swedish forest biodiversity management. Ecol Bull 46:191–203.

Angelstam PK (1998) Maintaining and restoring biodiversity in European boreal forests by developing natural disturbance regimes. J Veg Sci 9:593–602. doi: 10.2307/3237275

Arnborg T (1990) Forest types of northern Sweden. Vegetatio 90:1–13.

Attiwill PM (1994) The disturbance of forest ecosystems: the ecological basis for conservative management. For Ecol Manage 63:247–300.

Aubry KB, Halpern CB, Peterson CE (2009) Variable-retention harvests in the Pacific Northwest: A review of short-term findings from the DEMO study. For Ecol Manage 258:398–408.

Auld G, Gulbrandsen LH, McDermott CL (2008) Certification schemes and the impacts on forests and forestry. Annu Rev Environ Resour 33:187–211.

Bamsey CR (ed) (1995) Innovative silviculture systems in boreal forests. Proceedings, Symposium, 2–8 October 1994, Edmonton, Alberta. Clear Lake Ltd., Edmonton, Alberta

Berg A, Ehnstrom B, Gustafsson L, Hallingback T, Jonsell M, Weslien J (1994) Threatened plant, animal, and fungus species in Swedish forests: distribution and habitat associations. Conserv Biol 8:718–731.

Bergeron Y, Leduc A, Harvey BD, Gauthier S (2002) Natural fire regime: A guide for sustainable management of the Canadian boreal forest. Silva Fenn 36:81–95.

Björkman C, Nimelä P (eds) (2015) Climate change and insect pests. CABI.

Boyland M (2006) The economics of using forests to increase carbon storage. Can J For Res 36:2223–2234.

Bradshaw CJA, Warkentin IG, Sodhi NS (2009) Urgent preservation of boreal carbon stocks and biodiversity. Trends Ecol Evol 24:541–548.

Bryant D, Nielsen D, Tangley L (1997) The Last Frontier Forests: Ecosystems and Economies on the Edge. World Resource Institute, Washington.

Burton PJ, Bergeron Y, Bogdanski BEC, Juday GP, Kuuluvainen T, McAfee BJ, Ogden A, Teplyakov VK, Alfaro RI, Francis DA, Gauthier S, Hantula J (2010) Sustainability of boreal forests and forestry in a changing environment. For Soc responding to Glob drivers Chang 247–282.

Burton PJ, Messier C, Smith DW, Adamowicz WL (eds) (2003) Towards Sustainable Management of the Boreal Forest. NRC Research Press.

Cajander AK (1949) Forest types and their significance. Acta For Fenn 56:1–71.

Cardinale BJ, Duffy JE, Gonzalez A, Hooper DU, Perrings C, Venail P, Narwani A, Mace GM, Tilman D, Wardle DA, Kinzig AP, Daily GC, Loreau M, Grace JB, Larigauderie A, Srivastava DS, Naeem S (2012) Biodiversity loss and its impact on humanity. Nature 486:59–67.

Carlén O, Mattsson L, Atlegrim O, Sjöberg K (1999) Cost efficiency in pursuing environmental objectives in forestry. J Environ Manage 55:111–125.

Chapin FS, Mcguire AD, Randerson J, Pielke R, Baldocchi D, Hobbie SE, Roulet N, Eugster W, Kasischke E, Rastetter EB, Zimov SA, Running SW (2000) Arctic and boreal ecosystems of western North America as components of the climate system. Glob Chang Biol 6:211–223.

Christiansen L (ed) (2014) Swedish Statistical Yearbook of Forestry 2014. Swedish Forest Agency, Jönköping, Sweden.

Clark JA, Covey KR (2012) Tree species richness and the logging of natural forests: A meta-analysis. For Ecol Manage 276:146–153.

Clason AJ, Lindgren PMF, Sullivan TP (2008) Comparison of potential non-timber forest products in intensively managed young stands and mature/old-growth forests in south-central British Columbia. For Ecol Manage 256:1897–1909.

Cogbill C (1985). Dynamics of the boreal forests of the Laurentian Highlands, Canada. Can J For Res, 15:252–261.

Cooper CF (1983) Carbon storage in managed forests. Can J For Res 13:155–166.

Costanza R, Fisher B, Mulder K, Liu S, Christopher T (2007) Biodiversity and ecosystem services: A multi-scale empirical study of the relationship between species richness and net primary production. Ecol Econ 61:478–491.

Côté P, Tittler R, Messier C, Kneeshaw DD, Fall A, Fortin M-J (2010) Comparing different forest zoning options for landscape-scale management of the boreal forest: Possible benefits of the TRIAD. For Ecol Manage 259:418–427.

Cox CB, Moore PD (2010) Biogeography. An ecological and evolutionary approach, 8th edn. Wiley, New York

Daniel TW, Helms JA, Baker FS (1979) Principles of silviculture. McGraw-Hill Book Company, New York, USA

De Jong J, Dahlberg A, Stokland JN (2004) Död ved i skogen: Hur mycket behövs för att bevara den biologiska mångfalden? Sven Bot Tidskr 98:278–297.

Díaz S, Lavorel S, de Bello F, Quétier F, Grigulis K, Robson TM (2007) Incorporating plant functional diversity effects in ecosystem service assessments. Proc Natl Acad Sci 104:20684–20689.

Diaz-Balteiro L, González-Pachón J, Romero C (2012) Goal programming in forest management: customising models for the decision-maker's preferences. Scand J For Res 7581:1–8.

Duncan C, Thompson JR, Pettorelli N (2015) The quest for a mechanistic understanding of biodiversity–ecosystem services relationships. Proc R Soc B Biol Sci 282:20151348.

EC (2012) Commission staff working document accompanying the document communication on innovating for sustainable growth: a bioeconomy for Europe. European Commission, Brussels.

Edwards D P, Gilroy JJ, Woodcock P, Edwards FA, Larsen TH, Andrews DJR, Derhé MA, Docherty TDS, Hsu WW, Mitchell SL, Ota T, Williams LJ, Laurance WF, Hamer KC, Wilcove DS (2014) Land-sharing versus land-sparing logging: reconciling timber extraction with biodiversity conservation. Glob Change Biol 20: 183–191.

Eid T, Hobbelstad K (2000) AVVIRK-2000: A large-scale forestry scenario model for long-term investment, income and harvest analyses. Scand J For Res 15:472–482.

Elbakidze M, Andersson K, Angelstam P, Armstrong GW, Axelsson R, Doyon F, Hermansson M, Jacobsson J, Pautov Y (2013) Sustained yield forestry in Sweden and Russia: how does it correspond to sustainable forest management policy? Ambio 42:160–73.

Elbakidze M, Angelstam P, Andersson K, Nordberg M, Pautov Y (2011) How does forest certification contribute to boreal biodiversity conservation? Standards and outcomes in Sweden and NW Russia. For Ecol Manage 262:1983–1995.

Engel S, Pagiola S, Wunder S (2008) Designing payments for environmental services in theory and practice: an overview of the issues. Ecol Econ 65:663–674.

Esseen P-A, Ehnström B, Ericson L, Sjöberg K (1997) Boreal forests. Ecol Bull 46:16–47.

Fahrig L (2003) Effects of habitat fragmentation on biodiversity. Annu Rev Ecol Evol Syst 34:487–515.

FAO (2010) Global Forest Resources Assessment 2010. FAO For Pap 378.

Farley J, Costanza R (2010) Payments for ecosystem services: from local to global. Ecol Econ 69:2060–2068.

Fedrowitz K, Koricheva J, Baker SC, Lindenmayer DB, Palik B, Rosenvald R, Beese W, Franklin JF, Kouki J, Macdonald E, Messier C, Sverdrup-Thygeson A, Gustafsson L (2014) Can retention forestry help conserve biodiversity? A meta-analysis. J Appl Ecol 51:1669–1679.

Felton A, Ranius T, Roberge J-M, Öhman K, Lämås T, Hynynen J, Juutinen A, Mönkkönen M, Nilsson U, Lundmark T, Nordin A (2017). Projecting biodiversity and wood production in future forest landscapes: 15 key modeling considerations. J Environ Manage (in press).

Felton A, Gustafsson L, Roberge J-M, Ranius T, Hjältén J, Rudolphi J, Lindbladh M, Weslien J, Rist L, Brunet J, Felton AM (2016) How climate change adaptation and mitigation strategies can threaten or enhance the biodiversity of production forests: Insights from Sweden. Biol Conserv 194:11–20.

Felton A, Lindbladh M, Brunet J, Fritz Ö (2010) Replacing coniferous monocultures with mixed-species production stands: An assessment of the potential benefits for forest biodiversity in northern Europe. For Ecol Manage 260:939–947.

Finnish Forest Act (2014) (1093/1996; amendments up to 567/2014 included).

Franklin JF, Spies TA, Pelt R Van, Carey AB, Thornburgh DA, Berg DR, Lindenmayer DB, Harmon ME, Keeton WS, Shaw DC, Bible K, Chen J (2002) Disturbances and structural development of natural forest ecosystems with silvicultural implications, using Douglas-fir forests as an example. For Ecol Manage 155:399–423.

Frey BR, Lieffers VJ, Landhäusser SM, Comeau PG, Greenway KJ (2003) An analysis of sucker regeneration of trembling aspen. Can J For Res 33:1169–1179.

FSC Finland (2010) Standard for Finland. FSC-STD-FIN-(Ver1-1)-2006 Finland natural forests. FSC Finland, Helsinki. 66.

FSC Sweden (2010) Swedish FSC Standard for Forest Certification including SLIMF indicators FSC-STD-SWE-02-02-2010 Sweden Natural, Plantations and SLIMF EN. FSC Sweden, Uppsala. 99.

Gamfeldt L, Snäll T, Bagchi R, Jonsson M, Gustafsson L, Kjellander P, Ruiz-Jaen MC, Fröberg M, Stendahl J, Philipson CD, Mikusiński G, Andersson E, Westerlund B, Andrén H, Moberg F, Moen J, Bengtsson J (2013) Higher levels of multiple ecosystem services are found in forests with more tree species. Nat Commun 4:1340.

Gauthier S, Bernier P, Kuuluvainen T, Shvidenko AZ, Schepaschenko DG (2015) Boreal forest health and global change. Science (80-) 349:819–822.

Gentry AH (1988) Tree species richness of upper Amazonian forests. Proc Natl Acad Sci U S A 85:156–9.

Gromtsev A (2002) Natural disturbance dynamics in the boreal forests of European Russia: A review. Silva Fenn 36:41–55.

Gulbrandsen LH (2005) The Effectiveness of Non-State Governance Schemes: A Comparative Study of Forest Certification in Norway and Sweden. Int Environ Agreements Polit Law Econ 5:125–149.

Gullison RE (2003) Does forest certification conserve biodiversity? Oryx 37:153–165.

Gustafsson L, Baker SC, Bauhus J, Beese WJ, Brodie A, Kouki J, Lindenmayer DB, Lõhmus A, Pastur GM, Messier C, Neyland M, Palik B, Sverdrup-Thygeson A, Volney WJA, Wayne A, Franklin JF (2012) Retention forestry to maintain multifunctional forests: a world perspective. Bioscience 62:633–645.

Gustafsson L, Kouki J, Sverdrup-Thygeson A (2010) Tree retention as a conservation measure in clear-cut forests of northern Europe: a review of ecological consequences. Scand J For Res 25:295–308.

Haines-Young R, Potschin M (2013) Common international classification of ecosystem services (CICES): consultation on version 4, August-December 2012.

Haines-Young R, Potschin M (2011) Common International Classification of Ecosystem Services (CICES): 2011 Update. Expert Meet Ecosyst Accounts 1–17.

Haines-Young RH, Potschin MB (2010) The links between biodiversity, ecosystem services and human well-being. In: Raffaelli DG, Frid CLJ (eds) Ecosystems Ecology: a new synthesis. Cambridge University Press.

Hansen AJ, Spies TA, Swanson FJ, Ohmann JL (1991) Conserving biodiversity in managed forests. Bioscience 41:382–392.

Hanski I (2000) Extinction debt and species credit in boreal forests: modelling the consequences of different approaches to biodiversity conservation. Ann Zool Fennici 37:271–280.

Hanski I (2011) Habitat loss, the dynamics of biodiversity, and a perspective on conservation. Ambio 40:248–55.

Hanski I, Hammond P (1995) Biodiversity in boreal forests. Trends Ecol Evol 10:5–6.

Harrison PA, Vandewalle M, Sykes MT, Berry PM, Bugter R, de Bello F, Feld CK, Grandin U, Harrington R, Haslett JR, Jongman RHG, Luck GW, da Silva PM, Moora M, Settele J, Sousa JP, Zobel M (2010) Identifying and prioritising services in European terrestrial and freshwater ecosystems. Biodivers Conserv 19:2791–2821.

Hjortsø CN (2004) Enhancing public participation in natural resource management using Soft OR — an application of strategic option development and analysis in tactical forest planning. 152:667–683.

Howe C, Suich H, Vira B, Mace GM (2014) Creating win-wins from trade-offs? Ecosystem services for human well-being: A meta-analysis of ecosystem service trade-offs and synergies in the real world.

Glob Environ Chang 28:263–275.

Huntley B (1993) Species-richness in north-temperate zone forests. J Biogeogr 20:163–180.

Huntley B, Birks HJB, Harry JB (1983) An atlas of past and present pollen maps for Europe, 0-13,000 years ago. Cambridge University Press.

Hynynen J, Ahtikoski A, Siitonen J, Sievänen R, Liski J (2005) Applying the MOTTI simulator to analyse the effects of alternative management schedules on timber and non-timber production. For Ecol Manage 207:5–18.

Hyvärinen E, Kouki J, Martikainen P (2006) Fire and Green-Tree Retention in Conservation of Red-Listed and Rare Deadwood-Dependent Beetles in Finnish Boreal Forests. Conserv Biol 20:1710–1719.

Hyytiäinen K, Hari P, Kokkila T, Mäkelä A, Tahvonen O, Taipale J (2004) Connecting a process-based forest growth model to stand-level economic optimization. Can J For Res 34:2060–2073.

Hyytiäinen K, Tahvonen O (2002) Economics of forest thinnings and rotation periods for Finnish conifer cultures. Scand J For Res 17:274–288.

Håbesland DE, Kilgore MA, Becker DR, Snyder SA, Solberg B, Sjølie HK, Lindstad BH (2016). Norwegian family forest owners' willingness to participate in carbon offset programs. For Policy Econ 70:30–38.

Imbeau L, Mönkkönen M, Desrochers A (2001) Long-term effects of forestry on birds of the Eastern Canadian boreal forests: a comparison with Fennoscandia. Conserv Biol 15:1151–1162. doi: 10.1046/j.1523-1739.2001.0150041151.x

Jactel H, Nicoll BC, Branco M, Gonzalez-Olabarria JR, Grodzki W, Långström B, Moreira F, Netherer S, Orazio C, Piou D, Santos H, Schelhaas MJ, Tojic K, Vodd, Vodde F, Hervé J, Bruce CN, Manuela B, Ramon J, Wojciech G, Bo L, Francisco M, Sigrid N, Christophe O, Dominique P, Helena S, S MJ, Karl T, Floor V, Jactel H, Nicoll BC, Branco M, Gonzalez-Olabarria JR, Grodzki W, Långström B, Moreira F, Netherer S, Orazio C, Piou D, Santos H, Schelhaas MJ, Tojic K, Vodde F, Hervé J, Bruce CN, Manuela B, Ramon J, Wojciech G, Bo L, Francisco M, Sigrid N, Christophe O, Dominique P, Helena S, S MJ, Karl T, Floor V, Jactel H, Nicoll BC, Branco M, Gonzalez-Olabarria JR, Grodzki W, Långström B, Moreira F, Netherer S, Orazio C, Piou D, Santos H, Schelhaas MJ, Tojic K, Vodde F (2009) The influences of forest stand management on biotic and abiotic risks of damage. Ann For Sci 66:701.

Jandl R, Lindner M, Vesterdal L, Bauwens B, Baritz R, Hagedorn F, Johnson DW, Minkkinen K, Byrne KA (2007) How strongly can forest management influence soil carbon sequestration? Geoderma 137:253–268.

Johansson J, Lidestav G (2011) Can voluntary standards regulate forestry? — Assessing the environmental impacts of forest certification in Sweden. For Policy Econ 13:191–198.

Johansson T, Hjältén J, de Jong J, von Stedingk H (2013) Environmental considerations from legislation and certification in managed forest stands: A review of their importance for biodiversity. For Ecol Manage 303:98–112.

Johnson KN, Scheurman H (1977) Techniques for prescribing optimal timber harvest and investment under different objectives - discussion and synthesis. For Sci (Monogr.) 18.

Jönsson U, Rosengren U, Thelin G, Nihlgård B (2003) Acidification-induced chemical changes in coniferous forest soils in southern Sweden 1988–1999. Environ Pollut 123:75–83.

Kaipainen T, Liski J, Pussinen A, Karjalainen T (2004) Managing carbon sinks by changing rotation length in European forests. Environ Sci Policy 7:205–219.

Kallio AMI, Salminen O, Sievänen R (2013) Sequester or substitute—Consequences of increased production of wood based energy on the carbon balance in Finland. J For Econ 19:402–415.

Kangas A, Kurttila M, Hujala T, Eyvindson K, Kangas J (2015) Decision Support for Forest Management. Springer International Publishing.

Kareksela S, Moilanen A, Tuominen S, Kotiaho JS (2013) Use of inverse spatial conservation prioritization to avoid biological diversity loss outside protected areas. Conserv Biol 27:1294–1303.

Karjalainen E, Sarjala T, Raitio, H (2010) Promoting human health through forests: overview and major challenges. Environ Health Prev Med 15:1–8.

Kelty MJ (1992) Comparative productivity of monocultures and mixed-species stands. In: Kelty MJ, Larson BC, Oliver CD (eds) The Ecology and Silviculture of Mixed-Species Forests. Springer, pp 125–141.

Kettunen M, Vihervaara P, Kinnunen S, D'Amato D, Badura T, Argimon M, Ten Brink P (2012) Socio-economic importance of ecosystem services in the Nordic Countries. Synthesis in the context of The Economics of Ecosystems and Biodiversity (TEEB).

King E, Cavender-Bares J, Balvanera P, Mwampamba TH, Polasky S (2015) Trade-offs in ecosystem services and varying stakeholder preferences: evaluating conflicts, obstacles, and opportunities. Ecol Soc 20:art25.

Kreutzweiser DP, Hazlett PW, Gunn JM (2008) Logging impacts on the biogeochemistry of boreal forest soils and nutrient export to aquatic systems: A review. Environ Rev 16:157–179.

Kruys N, Fridman J, Götmark F, Simonsson P, Gustafsson L (2013) Retaining trees for conservation at clearcutting has increased structural diversity in young Swedish production forests. For Ecol Manage 304:312–321.

Kurki S, Nikula A, Helle P, Lindén H (2000) Landscape fragmentation and forest composition effects on grouse breeding success in boreal forests. Ecology 81:1985–1997.

Kuuluvainen T (2002) Natural variability of forests as a reference for restoring and managing biological diversity in boreal Fennoscandia. Silva Fenn. 36: 97–125.

Kuuluvainen T (2009) Forest Management and Biodiversity Conservation Based on Natural Ecosystem Dynamics in Northern Europe: The Complexity Challenge. AMBIO A J Hum Environ 38:309–315.

Kuuluvainen T, Aakala T (2011) Natural forest dynamics in boreal Fennoscandia: A review and Classification. Silva Fenn. 45:823–841.

Kuuluvainen T, Grenfell R (2012) Natural disturbance emulation in boreal forest ecosystem management — theories, strategies, and a comparison with conventional even-aged management. Can J For Res 42:1185–1203.

Kuuluvainen T, Saaristo L, Keto-Tokoi P, Kostamo J, Kuuluvainen J, Kuusinen M, Ollikainen M, Salpakivi-Salomaa P, Hallanaro E-L, Jäppinen J-P (2004) Metsän kätköissä : Suomen metsäluonnon monimuotoisuus. Edita, Helsinki.

Kuuluvainen T, Tahvonen O, Aakala T (2012) Even-aged and uneven-aged forest management in boreal Fennoscandia: a review. Ambio 41:720–37.

Laiho O, Lahde E, Pukkala T (2011) Uneven- vs even-aged management in Finnish boreal forests. Forestry 84:547–556.

Larocque GR (ed) (2016) Ecological forest management handbook.

Latham R, Ricklefs R (1993) Continental comparisons of temperate-zone tree species diversity. In: Species diversity in ecological communities. Historical and geographical perspectives. University of Chicago Press, Chicago, pp 291–314.

Laudon H, Sponseller R, Lucas R, Futter M, Egnell G, Bishop K, Ågren A, Ring E, Högberg P (2011) Consequences of More Intensive Forestry for the Sustainable Management of Forest Soils and Waters. Forests 2:243–260.

Lindenmayer DB, Franklin JF, Lõhmus A, Baker SC, Bauhus J, Beese W, Brodie A, Kiehl B, Kouki J, Pastur GM, Messier C, Neyland M, Palik B, Sverdrup-Thygeson A, Volney J, Wayne A, Gustafsson L (2012) A major shift to the retention approach for forestry can help resolve some global forest sustainability issues. Conserv Lett 5:421–431.

Lintunen J, Laturi J, Uusivuori J (2016) How should a forest carbon rent policy be implemented? For Policy Econ 69:31–39.

Liski J, Pussinen A, Pingoud K, Mäkipää R, Karjalainen T (2001) Which rotation length is favourable to carbon sequestration? Can J For Res 31:2004–2013.

Lundmark T, Bergh J, Nordin A, Fahlvik N, Poudel BC (2016) Comparison of carbon balances between continuous-cover and clear-cut forestry in Sweden. Ambio 45:203–213.

Lutz D a., Howarth RB (2014) Valuing albedo as an ecosystem service: implications for forest management. Clim Change 124:53–63.

Luyssaert S, Schulze E-D, Börner A, Knohl A, Hessenmöller D, Law BE, Ciais P, Grace J (2008) Old-growth forests as global carbon sinks. Nature 455:213–215.

Mace GM, Norris K, Fitter AH (2012) Biodiversity and ecosystem services: a multilayered relationship. Trends Ecol Evol 27:19–26.

Maes J, Paracchini ML, Zulian G, Dunbar MB, Alkemade R (2012) Synergies and trade-offs between ecosystem service supply, biodiversity, and habitat conservation status in Europe. Biol Conserv 155:1–12.

MARSI (2014) Luonnonmarjojen ja – sienten kauppaantulomäärät vuonna 2014. Raportteja ja selvityksiä 3/2015.

Mazziotta A, Podkopaev D, Triviño M, Miettinen K, Pohjanmies T, Mönkkönen M (2017) Quantifying and resolving conservation conflicts in forest landscapes via multiobjective optimization. Silva

Fenn 51 (1) article id 1778. https://doi.org/10.14214/sf.1778

MEA (2005) Millennium Ecosystem Assessment. Ecosystems and Human Well-being: Synthesis. Island Press, Washingthon, DC.

METSO 2015 http://www.metsonpolku.fi/en-US accessed 6.4.2017

Miettinen K (1999) Nonlinear Multiobjective Optimization. Kluwer Academic Publishers, Boston.

Ministry of Employment and the Economy (2014) Sustainable growth from bioeconomy. The Finnish Bioeconomy Strategy.

Moen A (1999) National Atlas of Norway – Vegetation. Norwegian Mapping Authority, Hønefoss.

Myllyntaus T, Hares M, Kunnas J (2002) Sustainability in Danger?: Slash-and-Burn Cultivation in Nineteenth-Century Finland and Twentieth-Century Southeast Asia. Environ Hist Durh N C 7:267.

Mönkkönen M (1999) Managing Nordic boreal forest landscapes for biodiversity: ecological and economic perspectives. Biodivers Conserv 8:85–99.

Mönkkönen M, Juutinen A, Mazziotta A, Miettinen K, Podkopaev D, Reunanen P, Salminen H, Tikkanen O-P (2014) Spatially dynamic forest management to sustain biodiversity and economic returns. J Environ Manage 134:80–89.

Mönkkönen M, Reunanen P, Nikula A, Inkeroinen J, Forsman J (1997) Landscape characteristics associated with the occurrence of the flying squirrel Pteromys volans in old-growth forests of northern Finland. Ecography (Cop) 20:634–642.

Mönkkönen M, Viro P (1997) Taxonomic diversity of the terrestrial bird and mammal fauna in temperate and boreal biomes of the northern hemisphere. J Biogeogr 24:603–612.

Natural resources of Canada (2016) Non-timber forest products. Available from http://www.nrcan.gc.ca/forests/industry/products-applications/13203 [accessed 26 May 2016].

Naudts K, Chen Y, McGrath MJ, Ryder J, Valade A, Otto J, Luyssaert S (2016) Europe’s forest management did not mitigate climate warming. Science (80-) 351:597–600.

Nieminen A (2006) Metsäsertifioinnin ekotehokkuus. Metlan työraportteja (Working Papers of the Finnish Forest Research Institute) 39: 1-85.

Niinimäki S, Tahvonen O, Mäkelä A, Linkosalo T (2013) On the economics of Norway spruce stands and carbon storage. Can J For Res 43:637–648.

Nilsson S (1997) Forests in the temperate-boreal transition: natural and man-made features. Ecol Bull 61–71.

Nordén B, Dahlberg A, Brandrud TE, Fritz Ö, Ejrnaes R, Ovaskainen O (2015) Effects of ecological continuity on species richness and composition in forests and woodlands: a review. Ecoscience 21:34–45.

Nordström E-M, Romero C, Eriksson LO, Öhman K (2009) Aggregation of preferences in participatory forest planning with multiple criteria: an application to the urban forest in Lycksele, Sweden. Can J For Res 39:1979–1992.

O'Hara KL, Ramage BS (2013) Silviculture in an uncertain world: utilizing multi-aged management systems to integrate disturbance. Forestry 86:401–410.

Ojalehto V, Miettinen K, Mäkelä M (2007) Interactive Software for Multiobjective Optimization: IND-NIMBUS, WSEAS. Trans Comput 6:87–94.

Ollikainen M (2016) Forest management, public goods, and optimal policies. Annu Rev Resour Economics 8:207–226.

Olson DM, Dinerstein E, Wikramanayake ED, Burgess ND, Powell GVN, Underwood EC, D'amico JA, Itoua I, Strand HE, Morrison JC, Loucks CJ, Allnutt TF, Ricketts TH, Kura Y, Lamoreux JF, Wettengel WW, Hedao P, Kassem KR (2001) Terrestrial Ecoregions of the World: A New Map of Life on Earth. Bioscience 51:933.

Paassilta M, Moisio S, Jaakola L, Häggman H, Oulu University press (2009) Voice of the Nordic wild berry industry. A survey among the companies. 84.

Pan Y, Birdsey RA, Fang J, Houghton R, Kauppi PE, Kurz WA, Phillips OL, Shvidenko A, Lewis SL, Canadell JG, Ciais P, Jackson RB, Pacala SW, McGuire AD, Piao S, Rautiainen A, Sitch S, Hayes D (2011) A large and persistent carbon sink in the world's forests. Science (80-) 333:988–993.

Paquette A, Messier C (2011) The effect of biodiversity on tree productivity: from temperate to boreal forests. Glob Ecol Biogeogr 20:170–180.

Parviainen J, Västilä S (eds) (2011) State of Finland´s Forests 2011. Ministry of Agriculture and Forestry & Finnish Forest Research Institute (Metla).

Pawson SM, Brin A, Brockerhoff EG, Lamb D, Payn TW, Paquette A, Parrotta JA (2013) Plantation forests, climate change and biodiversity. Biodivers Conserv 22:1203–1227.

PEFC Finland (2014) Criteria for PEFC Forest Certification PEFC FI 1002:2014. PEFC Finland, Helsinki.

Peltola A (ed) (2014) Finnish Statistical Yearbook of Forestry 2014. Forest Research Institute, Tampere, Finland.

Pennanen J (2002) Forest age distribution under mixed-severity fire regimes – a simulation-based analysis for middle boreal Fennoscandia. Silva Fenn 36:213–231. doi: 10.14214/sf.559

Peura M, Triviño M, Mazziotta A, Podkopaev D, Juutinen A, Mönkkönen M (2016) Managing boreal forests for the simultaneous production of collectable goods and timber revenues. Silva Fenn 50 article id 1672. https://doi.org/10.14214/sf.1672

Pihlainen S, Tahvonen O, Niinimäki S (2014) The economics of timber and bioenergy production and carbon storage in Scots pine stands. Can J For Res 44: 1091–1102.

Pommerening A, Murphy S (2004) A review of the history, definitions and methods of continuous cover forestry with special attention to afforestation and restocking. Forestry 77:27–44.

Potapov P, Yaroshenko A, Turubanova S, Dubinin M, Laestadius L, Thies C, Aksenov D, Egorov A, Yesipova Y, Glushkov I, Karpachevskiy M, Kostikova A, Manisha A, Tsybikova E, Zhuravleva I (2008) Mapping the world's intact forest landscapes by remote sensing. Ecol Soc 13:Artn 51.

Pohjanmies T, Triviño M, Le Tortorec E, Mazziotta A, Snäll T, Mönkkönen M (2017) Impacts of forestry on boreal forests: An ecosystem services perspective. Ambio (in press) doi:10.1007/s13280-017-0919-5

Pukkala T (2016) Which type of forest management provides most ecosystem services? For Ecosyst 3:9.

Pukkala T, Lähde E, Laiho O (2013) Species interactions in the dynamics of even- and uneven-aged boreal forests. J Sustain For 32:371–403.

Pukkala T, Lähde E, Laiho O, Salo K, Hotanen J (2011) A multifunctional comparison of even-aged and uneven-aged forest management in a boreal region. 862:851–862.

Rametsteiner E, Simula M (2003) Forest certification—an instrument to promote sustainable forest management? J Environ Manage 67:87–98.

Rasinmäki J, Mäkinen A, Kalliovirta J (2009) SIMO: An adaptable simulation framework for multiscale forest resource data. Comput Electron Agric 66:76–84.

Rassi P, Hyvärinen E, Juslén A, Mannerkoski I (eds) (2010) The 2010 Red List of Finnish Species. Ympäristöministeriö & Suomen ympäristökeskus, Helsinki.

Redsven V, Hirvelä H, Härkönen K, Salminen O, Siitonen M (2012) MELA2002 Reference Manual, 2ND edn. The Finnish Forest Research Institute.

Repo A, Tuomi M, Liski J (2011) Indirect carbon dioxide emissions from producing bioenergy from forest harvest residues. GCB Bioenergy 3:107–115.

Roberge J-M, Laudon H, Björkman C, Ranius T, Sandström C, Felton A, Sténs A, Nordin A, Granström

A, Widemo F, Bergh J, Sonesson J, Stenlid J, Lundmark T (2016) Socio-ecological implications of modifying rotation lengths in forestry. Ambio 45 Suppl 2:109–23.

Rodríguez A, Kouki J (2015) Emulating natural disturbance in forest management enhances pollination services for dominant Vaccinium shrubs in boreal pine-dominated forests. For Ecol Manage 350:1–12.

Rosenvald R, Lõhmus A (2008) For what, when, and where is green-tree retention better than clear-cutting? A review of the biodiversity aspects. For Ecol Manage 255:1–15.

Rosenzweig M (1995) Species diversity in space and time. Cambridge University Press.

Rybicki J, Hanski I (2013) Species-area relationships and extinctions caused by habitat loss and fragmentation. Ecol Lett 16:27–38.

Rämö J, Tahvonen O (2014) Economics of harvesting uneven-aged forest stands in Fennoscandia. Scand J For Res 29:777–792.

Saastamoinen O, Matero J, Horne P, Kniivilä M, Haltia E, Vaara M, Mannerkoski H (2014) Classification of boreal forest ecosystem goods and services in Finland. Publications of the University of Eastern Finland. Reports and Studies in Forestry and Natural Sciences Number 11, University of Eastern Finland, Faculty of Science and Forestry, School of Forest Sciences.

Santangeli A, Wistbacka R, Hanski IK, Laaksonen T (2013) Ineffective enforced legislation for nature conservation: A case study with Siberian flying squirrel and forestry in a boreal landscape. Biol Conserv 157:237–244.

Schindler D (1998) Sustaining aquatic ecosystems in boreal regions. Conserv Ecol 2:18.

Seidl R, Schelhaas M-J, Rammer W, Verkerk PJ (2014) Increasing forest disturbances in Europe and their impact on carbon storage. Nat Clim Chang 4:806–810.

Seymour RS, Hunter MLJ (1992) New forestry in eastern spruce-fir forests: principles and applications to Maine. Miscellaneous Publication 716. Maine Agricultural Forest Experimental Station.

Shorohova, E., Kuuluvainen, T., Kangur, A & Jogiste, K. (2009) Natural stand structures , disturbance regimes and successional dynamics in the Eurasian boreal forests: a review with special reference to Russian studies. Annals of Forest Science 66:1–20.

Sievänen R, Salminen O, Lehtonen A, Ojanen P, Liski J, Ruosteenoja K, Tuomi M (2013) Carbon stock changes of forest land in Finland under different levels of wood use and climate change. Ann For Sci 71:255–265.

Siitonen J (2001) Forest management, coarse woody debris and saproxylic organisms: Fennoscandian boreal forests as an example. Ecol Bull 11–41.

Skovsgaard JP, Vanclay JK (2008). Forest site productivity: a review of the evolution of dendrometric concepts for even-aged stands. Forestry 81:13–32.

Spracklen D V, Bonn B, Carslaw KS (2008) Boreal forests, aerosols and the impacts on clouds and climate. Philos Trans A Math Phys Eng Sci 366:4613–26.

Strassburg BBN, Kelly A, Balmford A, Davies RG, Gibbs HK, Lovett A, Miles L, Orme CDL, Price J, Turner RK, Rodrigues ASL (2010) Global congruence of carbon storage and biodiversity in terrestrial ecosystems. Conserv Lett 3:98–105.

Strengbom J, Dahlberg A, Larsson A, Lindelöw Å, Sandström J, Widenfalk O, Gustafsson L (2011) Introducing intensively managed spruce plantations in Swedish forest landscapes will impair biodiversity decline. Forests 2:610–630.

Strengbom J, Nordin A (2008) Commercial forest fertilization causes long-term residual effects in ground vegetation of boreal forests. For Ecol Manage 256:2175–2181.

Stupak I, Asikainen A, Jonsell M, Karltun E, Mizaraitė D, Pasanen K, Pärn H, Raulund-Rasmussen K, Röser D, Schroeder M, Varnagirytė I, Vilkriste L, Callesen I, Gaitnieks T, Ingerslev M, Mandre M, Ozolincius R, Saarsalmi A, Armolaitis K, Helmisaari H-S, Indriksons A, Kairiukstis L, Katzensteiner K, Kukkola M, Ots K, Ravn HP, Tamminen P (2007) Sustainable utilisation of forest biomass for energy—Possibilities and problems: Policy, legislation, certification, and recommendations and guidelines in the Nordic, Baltic, and other European countries. Biomass and Bioenergy 31:666–684.

Syrjänen K, Kalliola R, Puolasmaa A (1994) Landscape structure and forest dynamics in subcontinental Russian European taiga. Ann Zool Fenn 31:19–34.

Tahvonen O (2015) Economics of rotation and thinning revisited: the optimality of clearcuts versus continuous cover forestry. For Policy Econ 62:88–94.

Thom D, Seidl R (2015) Natural disturbance impacts on ecosystem services and biodiversity in temperate and boreal forests. Biol Rev 91:760–781.

Thomas CD, Anderson BJ, Moilanen A, Eigenbrod F, Heinemeyer A, Quaife T, Roy DB, Gillings S, Armsworth PR, Gaston KJ (2013) Reconciling biodiversity and carbon conservation. Ecol Lett 16:39–47.

Tikkanen O-P, Heinonen T, Kouki J, Matero J (2007) Habitat suitability models of saproxylic red-listed boreal forest species in long-term matrix management: Cost-effective measures for multi-species conservation. Biol Conserv 140:359–372.

Tikkanen O-P, Matero J, Mönkkönen M, Juutinen A, Kouki J (2012) To thin or not to thin: bio-economic analysis of two alternative practices to increase amount of coarse woody debris in managed forests. Eur J For Res 131:1411–1422.

Tilman D, Reich PB, Isbell F (2012) Biodiversity impacts ecosystem productivity as much as resources, disturbance, or herbivory. Proc Natl Acad Sci 109:10394–10397.

Timonen J, Siitonen J, Gustafsson L, Kotiaho JS, Stokland JN, Sverdrup-Thygeson A, Mönkkönen M

(2010) Woodland key habitats in northern Europe: concepts, inventory and protection. Scand J For Res 25:309–324.

Tittler R, Messier C, Goodman RC (2016) Triad Forest Management: Local Fix or Global Solution. In: Laroque GR (ed) Ecological forest management handbook. CRC Press, pp 33–45

Toivanen T, Kotiaho JS (2007) Mimicking natural disturbances of boreal forests: the effects of controlled burning and creating dead wood on beetle diversity. Biodivers Conserv 16:3193–3211.

Triviño M, Juutinen A, Mazziotta A, Miettinen K, Podkopaev D, Reunanen P, Mönkkönen M (2015) Managing a boreal forest landscape for providing timber, storing and sequestering carbon. Ecosyst Serv 14:179–189.

Triviño M, Pohjanmies T, Mazziotta A, Juutinen A, Podkopaev D, Le Tortorec E, Mönkkönen M (2017) Optimizing management to enhance multifunctionality in a boreal forest landscape. J Appl Ecol 54:61–70.

Tuhkanen S (1980) Climatic parameters and indices in plant geography. Svenska växtgeografiska sällskapet.

Turner MG (2010) Disturbance and landscape dynamics in a changing world. Ecology 91:2833–2849.

Turtiainen M, Saastamoinen O, Kangas K, Vaara M (2012) Picking of Wild Edible Mushrooms in 1997-1999 and 2011. Silva Fenn 46(4):569–581.

UNEP, FAO, UNFF (2009) Vital Forest Graphics. UNEP/GRID-Arendal.

Uprety Y, Asselin H, Dhakal A, Julien N, Shrestha K, Haddad P, Haddad P, Arnason J, Steffany A, Bennett S, Prentki M, Bennett S, Arnason J, Haddad P, Arnason J, Haddad P (2012) Traditional use of medicinal plants in the boreal forest of Canada: review and perspectives. J Ethnobiol Ethnomed 8:7.

van Kooten GC, Binkley C, Delcourt G (1995). Effect of carbon taxes and subsidies on optimal forest rotation age and supply of carbon services. Am J Agr Econ 77:365–374.

Vanha-Majamaa I, Jalonen J (2001) Green tree retention in Fennoscandian forestry. Scand J For Res 16:79–90.

Vanha-Majamaa I, Lilja S, Ryömä R, Kotiaho JS, Laaka-Lindberg S, Lindberg H, Puttonen P, Tamminen P, Toivanen T, Kuuluvainen T (2007) Rehabilitating boreal forest structure and species composition in Finland through logging, dead wood creation and fire: The EVO experiment. For Ecol Manage 250:77–88.

Vanhanen H, Jonsson R, Gerasimov Y, Krankina ON, Messier C (2012) Making Boreal Forests Work for People and Nature. 15.

Westling A (ed) (2015) Rödlistade arter i Sverige 2015. Artdatabanken, Uppsala.

Wikström P, Edenius L, Elfving B, Eriksson LO, LäMåS T, Sonesson J, ÖHMAN K, Wallerman J, Waller C, Klintebäck F (2011) The Heureka forestry decision support system: an overview. Math Comput For Nat Sci 3:87–94.

Vilà M, Vayreda J, Comas L, Ibáñez JJ, Mata T, Obón B (2007) Species richness and wood production: a positive association in Mediterranean forests. Ecol Lett 10:241–250.

Virkkala R, Rajasarkka A, Vaisanen RA, Vickholm M, Virolainen E (1994) The significance of protected areas for the land birds of southern Finland. Conserv Biol 8:532–544.

Yu PL (1973) A class of solutions for group decision problems. Manage Sci 19:936–946.

Zackrisson O (1977) Influence of forest fires on the North Swedish boreal forest. Oikos 29:22.

Zeleny M (1982) Multiple Criteria Decision Making. McGraw-Hill, New York.

Zeng H, Garcia-Gonzalo J, Peltola H, Kellomäki S (2010) The effects of forest structure on the risk of wind damage at a landscape level in a boreal forest ecosystem. Ann For Sci 67:111.

Zeng H, Peltola H, Väisänen H, Kellomäki S (2009) The effects of fragmentation on the susceptibility of a boreal forest ecosystem to wind damage. For Ecol Manage 257:1165–1173.

Äijälä O, Koistinen A, Sved J, Vanhatalo K, Väisänen P (2014) Metsänhoidon suositukset. [The good practice guidance to forestry, (in Finnish)]. Metsäkustannus Oy, Forestry Development Centre Tapio, Helsinki.

Östlund L, Roturier S (2010) Forestry historical studies in the province of Västerbotten, Northern Sweden: a review of Lars Tirén (1937). Scand J For Res 26:91–99.

Östlund L, Zackrisson O, Axelsson a-L (1997) The history and transformation of a Scandinavian boreal forest landscape since the 19th century. Can J For Res 27:1198–1206.

Tables

Table 1. Comparison of disturbance dynamics between natural disturbance dynamics and modern intensive forestry (managed forests). Modified from Kuuluvainen et al. (2004).

Disturbances	**Natural forest**	**Managed forest**
Number of different disturbance factors	Large	Small
Qualitative variation of disturbances	Large	Small
Proportion of trees dying during disturbance	0-100 %	95-100 % [1]
Proportion of dead trees remaining as dead wood on site	100 %	0-5 %
Recurrency of disturbances	Every 10 to 500 years	Every 70-130 years [2]
Occurrence of disturbances	Irregular	Regular
Area disturbed	0.001 →100.000ha	0.5-10 ha

[1] Proportion of trees harvested at final felling

[2] Average forest rotation length in Fennoscandia

Table 2. Alternative management regimes simulated for stands in the landscape. The brackets after the abbreviation indicate the number of years final felling is sped up (-) or delayed (+).

Management regime	**Description**
Tapio (BAU)	Conventional regeneration harvest regime (see section 3.2). Simulated sped up (-5) and delayed (5, 10, 15, 30) final fellings.
Tapio harvesting w/o thinnings (BAU wo thin)	Otherwise similar to Tapio but no thinnings applied before final felling by clear-cutting. Simulated sped up (-5) and delayed (5, 10, 15, 30) final fellings.
Tapio harvesting w thinnings (BAU w thin)	Otherwise similar to Tapio but thinnings are always applied before final harvest by clear-cutting. Simulated sped up (-5) and delayed (5, 10, 15, 30) final fellings.
Tapio Seed Tree (GTR, GTR w thin)	Otherwise similar to Tapio but regeneration after harvest is through seed trees, rather than planting (with and without thinning). Simulated sped up (-10) and delayed (20) final fellings.
Continuous cover forestry (CCF)	Rather than conducting harvests, only thinnings are conducted, depending on the specific stand properties. Follows suggestions by Pukkala et al. 2013.
Set-aside (SA)	No management

Figure legends

Figure 1. Map showing the global extent of the boreal zone (outlined) and estimated cover of woody vegetation greater than 5 meters in height in 2010. Darker colours represent more dense forest cover. Data source: Tree Canopy Cover, 2010, Global Land Cover Facility, www.landcover.org.

Figure 2. Illustration on how the three types of natural disturbance regimes are not totally distinct but form a continuum in terms of size, severity and repeatability of disturbance. Gap disturbances are frequent at landscape scale, small in size and often low in severity leaving most of the vegetation alive. Partial disturbances in cohort dynamics is in the middle of the gradient. Succession is caused by severe disturbances such as fire or strong wind, which often cover large areas, but may occur relatively seldom. Figure taken from Angelstam and Kuuluvainen (2004).

Figure 3. Forests under natural disturbance regime vs. under intensive management. Pictures on the left show early successional (top), middle-aged (appr. 80 years old), and old-growth boreal forest (bottom), and on the right, there are corresponding developmental stages in managed forests. Notice the higher level of structural diversity in forests under natural disturbance regime, particularly at early and late successional stages. All photos by Timo Kuuluvainen.

Figure 4. Framework linking forest management activities via forest structures and functions to final benefits and values experienced by humans. Modified from Pohjanmies et al. (2017).

Figure 5. Pairwise conflicts between five ecosystem services measured as the level of one service when forest management has been planned to maximize another service. The black points show the average value of the services across nearly 30,000 forest stands in Central Finland. These values are expressed as percentage of the maximal achievable value; units on all axes are thus percentages (%). Dashed grey

lines have been added to all plots at y = 100 and x = 100 for graphical comparison. The closer to the point (100, 100) the two points are, the weaker is the conflict between the two services. Modified from Pohjanmies et al. (unpubl.).

Figure 6. Age distribution of the forest within the watershed.

Figure 7. Represents the normalized change in the five criteria under consideration. Income increases linearly (due to the flow constraint), while the remaining criteria vary dependent on the optimization.

Figure 8. Proportion of management regimes as the proportion of timber harvest is increased. BAU refers to alternative clear-cut based management regimes with variable thinning intensities and rotation lengths (GTR = green tree retention, w thin = with thinnings before clear felling, wo thin = with no thinnings). CCF refers to continuous cover forestry with not final felling by clear-cut, and set aside denotes permanent protection (no management). For description of management regimes see Table 2.

Figure 9. Multi-objective optimization results: curves representing the trade-offs between carbon storage and two biodiversity indicators (deadwood index and combined habitat availability of 6 vertebrate species) for different levels of timber harvest revenues. The black star in each Pareto optimal set indicates the compromise management plan. Figure adapted from Triviño et al. 2016.

Figure 10. Changes in percentage of area in the landscape allocated for the different management regimes for the compromise outcome in the Pareto optimal set (the black stars from Fig. 9) at decreasing levels of timber harvest revenues (from 99.9% to ‘no constraints’). The acronyms of the management regimes are: BAU (Business as usual); EXT10 (Extended rotation by 10 years); EXT30 (Extended

rotation by 30 years); GTR30 (Green tree retention); NTSR (No thinning short rotation); NTLR (No thinning long rotation and SA (Set aside). Figure adapted from Triviño et al. 2016.