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Tähti Pohjanmies

Trade-Offs Among Intensive Forestry, Ecosystem Services and Biodiversity in Boreal Forests

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ABSTRACT

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Finnish forests are used extensively for timber production but are also providers of other ecosystem services and harbor unique biodiversity. The ecosystem services approach has so far been used marginally in the context of Finnish forestry; however, due to the multiple values associated with Finnish forests and the impacts forestry operations have on forest ecosystems, it is clearly applicable in this context. In this thesis, I studied the occurrence and severity of trade-offs among ecosystem services and biodiversity conservation in Finnish forests. I used forest inventory data, forest growth simulations, and multi-objective optimization to reveal how the severity of the trade-offs varies among combinations of ecosystem services, across spatial scales, and across time, and how the trade-offs could be mitigated by forest management planning. Overall, the results showed that there are clear and challenging conflicts between intensive forestry and ecosystem services in Finland. Ecosystem services provided by forests were found to diminish when the forests were used intensively as a source of timber, whereas reducing or refraining from harvests maintained comparatively high levels of multiple non-timber services and biodiversity. Non-timber services and biodiversity were also shown to recover from intensive forestry the slower the longer intensive forestry was continued, suggesting that forestry's negative impacts may be long-lasting. The use of optimization tools can help planners to identify management strategies that balance conflicting objectives as well as possible, especially if the analyses are conducted at large enough scales. However, the fact that there are trade-offs means that losses in some objectives are inevitable. It is left to forest managers and other stakeholders to consider which of these losses they are willing to accept.

Keywords: Conflicts; Finland; forest management; multi-objective optimization; sustainability; timber production.

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LIST OF ORIGINAL PUBLICATIONS

The thesis is based on the following original papers, which will be referred to in the text by their Roman numerals I–IV.

I 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* 46: 743-755.

II Pohjanmies T., Triviño M., Le Tortorec E., Salminen H. & Mönkkönen M. 2017. Conflicting objectives in production forests pose a challenge for forest management. *Ecosystem Services* 28, Part C: 298-310.

III Pohjanmies T., Eyvindson K., Triviño M. & Mönkkönen M. 2017. More is more? Forest management allocation at different spatial scales to mitigate conflicts between ecosystem services. *Landscape Ecology* 32: 2337-2349.

IV Pohjanmies T., Eyvindson K., Peura M., Triviño M. & Mönkkönen M. 2017. Loss of resilience and multifunctionality in a production forest. Manuscript.

The table shows the contributions of the authors to the original papers.

	I	II	III	IV
Planning	TP, MT, TS, MM	TP, MT, MM	TP, KE, MT, MM	TP, KE, MM
Data	TP	TP, MT, HS, MM	TP, KE, MT	TP, KE, MP, MT
Analyses	TP	TP	TP, KE	TP, KE
Writing	TP, MT, ELT, AM, TS, MM	TP, MT, ELT, MM	TP, KE, MT, MM	TP, KE, MT, MM

TP = Tähti Pohjanmies, MT = María Triviño, ELT = Eric Le Tortorec, AM = Adriano Mazziotta, TS = Tord Snäll, MM = Mikko Mönkkönen, HS = Hannu Salminen, KE = Kyle Eyvindson, MP = Maiju Peura

1 INTRODUCTION

1.1 Thesis background

When humans use natural resources they modify their environment in ways that have both intentional and unintentional impacts on it (Kareiva *et al.* 2007). Intentional impacts include, for example, the increased production of a desired renewable resource, and unintentional impacts include pollution, increased risk of natural hazards, and species endangerment. Ecosystems used for the continuing production of food or raw materials, such as agricultural areas or production forests, are a setting of ongoing, close interactions between humans and nature, and the goods harvested from them by humans are a product of both natural and anthropogenic inputs. Ecosystems like these typically have a long history of human presence and interference, and likewise humans have come to attach multiple meanings and values to them beyond the products harvested from them. Yet, growing demand for resources has led to the intensification of their exploitation, intensifying also the unintended impacts on them. As a result, the natural inputs and processes underlying production may become threatened, as well as the other benefits humans gain from these ecosystems. Additionally, even the most human-modified landscapes are home to various kinds of wildlife, and the fate of countless individuals and species is influenced by the actions humans take to alter these ecosystems to their own ends.

The situation described above is exemplified in the forests of Finland. Finland is the most forested country in the world with approximately 73 % of its land area covered with forest (Anon. 2015a), and has a long history of exploiting its timber resources but also of close cultural connections between people and forests (Parviainen 2015). Currently, only 13 % of Finland's forests are protected or have forestry use restrictions (Peltola 2014). Future visions of "bioeconomy" as presented by, among others, the Finnish government, involve intensifying even more the use of forest resources (Anon. 2015b). At the same time, the environmental impacts of forestry have long been a source of concern and criticism, including issues such as harmful impacts on surface water quality (Kortelainen and Saukkonen 1998), harmful impacts on the abundance of non-timber forest products such as wild berries (Miina *et al.* 2009), loss of recreational opportunities and landscape aesthetics (Gundersen and Frivold 2008), and, perhaps most notably, endangerment and extinction of species and habitats (Esseen *et al.* 1997, Siitonen 2001, Kouki *et al.* 2001, Tikkanen *et al.* 2006, Junninen and Komonen 2011). Approximately 36 % of endangered species in Finland are forest species, with the impacts of forestry considered as the primary cause of their endangerment (Rassi *et al.* 2010). What is more still, the role of forests in climate regulation and the role of forest use or protection in climate change mitigation have emerged to the center of the debate on how forests should be managed.

Finnish forests are thus faced with a multitude of expectations by humans: they are a source of biomass to be harvested with an increasing intensity, they play crucial parts in regulating the quality of humans' environment that should be maintained, they are an integral part of global climate dynamics and thus a pawn in national and international climate politics, they are a source of recreation and traditionally utilized products which Finns continue to value, they are still the cornerstone of the Finnish mental landscape, and they are home to the majority of Finnish wildlife. These roles may not be automatically compatible, making people's objectives and expectations conflicting and creating a challenge for forest policy and management. In this thesis I study what kinds of conflicts there are among management objectives that are associated with Finnish forests, can they be reconciled with each other, and how. The main focus of the management of most forests in Finland is on their exploitation for timber harvesting (Äijälä *et al.* 2014). If timber harvesting is in conflict with other forest uses or values, its sustainability and acceptability become problematic: Under intensive forestry, can important ecological processes be maintained? Can public or non-monetary values be preserved? Can forest wildlife be safeguarded? I approach these questions through the ecosystem services framework and apply forest growth simulations and multi-objective optimization methods to find answers to them.

1.2 The ecosystem services approach: from ecosystem structure to human wellbeing

Humans depend on nature for their wellbeing, and anthropogenic environmental change has feedback effects on humans themselves (Díaz *et al.* 2006, Kareiva *et al.* 2007, Carpenter *et al.* 2009, Cardinale *et al.* 2012a). One attempt at revealing these interdependencies in order to improve the sustainability of natural resource use has been the development of the concept of ecosystem services. Ecosystem services are typically defined along the lines of "the benefits people obtain from ecosystems" (e.g., Seppelt *et al.* 2011). The ideas behind the concept – that humans are part of ecosystems and dependent on ecosystem functions, and that the economic development of human communities and the state of the environment are connected – go back several decades in scientific literature and conservation practice (Tallis *et al.* 2008, Vihervaara *et al.* 2010). The concept of ecosystem services emerged in scientific and policy literature in the 1990s as an attempt to crystallize and promote these ideas (Costanza *et al.* 1997, Daily 1997). The Millennium Ecosystem Assessment (MEA), published in 2005, popularized the concept and brought it to the forefront of applied ecology and environmental science. In the MEA, a classification of ecosystem services was presented consisting of four categories: provisioning, regulating, supporting, and cultural services. Provisioning services are products obtained from ecosystems, regulating services are ecosystem processes that regulate environmental conditions, cultural services are nonmaterial benefits obtained from interactions with nature, and supporting services are the fundamental biophysical processes underlying all other ecosystem services. Ecosystem services thus range from the goods obtained from ecosystems to the processes underlying their production and to the regulation of humans' environment via species interactions and biophysical processes (Anon. 2003a).

Haines-Young and Potschin (2010) describe the delivery of ecosystem services as a production chain or a "cascade" that links ecosystem structures and processes to human well-being. Biophysical structures and processes, i.e. the interactions among living organisms and their environment, result in ecosystem functions and products which benefit humans. The populations, species, functional groups, or habitat types that are most strongly associated with the functions that underlie ecosystem services are sometimes called ecosystem service providers (Kremen 2005). When the variability of ecosystem service supply is studied, the aim is typically to identify the key ecosystem service providers and their responses to environmental change. One ecosystem service may have multiple providers that operate over different spatial and temporal scales and respond to environmental changes differently (Kremen 2005). The ecology behind ecosystem services is thus complex and remains insufficiently understood in many contexts (Luck *et al.* 2009, Balvanera *et al.* 2014).

A specific, still open-ended issue directly related to the ecology of ecosystem services is the relationship between biodiversity and ecosystem services. The origin of the ecosystem services concept was in a perceived need to reveal the links between biodiversity and human well-being in order to justify the importance of nature conservation as well as to guide environmental policy and management (Haines-Young and Potschin 2010). Ecosystem service research still has strong links to biodiversity-ecosystem function research (Kremen 2005, Cardinale *et al.* 2012b, Duncan *et al.* 2015, Isbell *et al.* 2015). Biodiversity itself is a broad concept that encompasses a range of aspects: it refers to all of the variability among living organisms, including within and between species, and within and between ecosystems (Mace *et al.* 2012). As described by Haines-Young and Potschin (2010), the basis of all ecosystem services is in communities of living organisms, their interactions with their environment, and the resultant ecosystem functions. Higher diversity (in terms of any of the different aspects of biodiversity – genes, species, functional groups, etc.) can increase ecosystem function or its stability (Elmqvist *et al.* 2003, Cardinale *et al.* 2012b, Harrison *et al.* 2014, Tilman *et al.* 2014, Duncan *et al.* 2015). However, the exact forms of the dependence of ecosystem services on specific aspects of biodiversity, or the impact of biodiversity loss on ecosystem service supply, remains an open question in most contexts (Harrison *et al.* 2014). It is worth noting that biodiversity is sometimes considered an ecosystem service in itself (Mace *et al.* 2012), either falling into the category of supporting services or something with cultural value. In the conceptualization of ecosystem services of Haines-Young and Potschin (2010) as well as implicitly in the MEA (Anon. 2003a), biodiversity is not considered an ecosystem service in itself. In this thesis, the same approach is followed and biodiversity is considered as a quality of the ecosystem that contributes to ecosystem functioning and the provision of ecosystem services.

The past two decades have seen a rapid increase in the popularity of the ecosystem services concept, illustrated by a huge increase in the numbers of published scientific papers using it (Vihervaara *et al.* 2010, Abson *et al.* 2014, Costanza *et al.* 2017). The spread of the concept was followed by the realization that it is, in fact, highly ambiguous and used in different ways by different people. Numerous review papers have synthesized how the concept has been applied and conceptual papers have called for consistency, often suggesting their own way to apply it systematically (e.g., Fisher *et al.* 2009, Seppelt *et al.* 2011, Nahlik *et al.* 2012, Crossman *et al.* 2013, Villamagna *et al.* 2013). For example, some interpretations limit the concept to refer to direct benefits, final products, or measurable goods to the production of which ecosystems contribute (e.g., Boyd and Banzhaf 2007). Others include under it also indirect benefits or intermediate services – a wide range of ecosystem functions that contribute to human well-being in some way, even if the contribution is difficult to measure quantitatively (e.g., Daniel *et al.* 2012). The exact definition of ecosystem services that is adopted also influences the indicators that are used to measure them. For example, a provisioning service can be measured as the amount that is produced in an ecosystem (e.g., growth of trees) or as the amount that is used by humans (e.g., trees harvested). The different measures reflect different aspects of the process of ecosystem service delivery: the former may better capture the contribution of ecosystem processes to the production of the goods, while the latter may demonstrate their benefit and value to humans.

In addition to the debate on its full meaning and exact definition, the concept of ecosystem services has been severely criticized from a range of viewpoints questioning its very foundation (Cornell 2011, Luck *et al.* 2012, Schröter *et al.* 2014). Questions have been raised within the research field not only about how ecosystem services should be defined but whether such a concept is useful or appropriate to begin with (McCauley 2006, Morelli and Møller 2015). The most critical writers argue that in its anthropocentric utilitarianism the concept can erode people's feeling of obligation to protect nature and thus the basis for nature conservation (McCauley 2006, Gómez-Baggethun and Ruiz-Pérez 2011). Proponents of the approach argue that it continues to be necessary to reveal how humans benefit from nature and that the approach is complementary, not substitutive, to traditional arguments for conservation for nature's sake (Pearson 2016). A compromise position may be one that recognizes both the concept's potential usefulness and harmfulness and seeks transparency of definitions and underlying values in order to maximize the former and minimize the latter.

In this thesis, I follow the common definition of ecosystem services as the benefits that people obtain from nature and interpret it broadly similarly to the MEA (Anon. 2003a). Ecosystem services encompass both measurable ecosystem goods and immaterial benefits derived from ecosystem structures and functions. I adopt a broad, inclusive definition of the concept because it allows for the flexible identification of values and objectives attached to ecosystems. In this way, the ecosystem service approach can be used to analyze the supply of final products as well as the supply of functions that contribute to them. In the ecosystem services framework, an ecosystem function or product becomes an ecosystem service only when there are humans benefiting from it (Haines-Young and Potschin 2010). That said, I note that without an expressed or acknowledged human beneficiary, an ecosystem function may be considered a potential ecosystem service or an ecosystem to provide a potential supply of ecosystem services. The benefits to humans may also be highly diverse and the values associated with them can be experienced in different ways (Mononen *et al.* 2016), further encouraging flexibility in defining the services.

Literature describing the delivery of ecosystem services shares the view that it is a complex process that involves the dynamics of both ecological and social systems (Carpenter *et al.* 2009, Villamagna *et al.* 2013). In simple terms, the ecological system creates the potential supply of ecosystem services, and the social system is the source of demand for and use of the services. The two sides are constantly interacting, with human activities, including ecosystem service demand and use, creating pressures and drivers of ecosystem change, and these changes feeding back to societies via the dependencies of human activities on the environment. In this thesis, I focus on the side of the ecological system, studying the potential supply of ecosystem services from Finnish forests.

1.3 Conflicts among ecosystem services

The purpose of ecosystem service studies is often to inform decision making about the potential feedback effects of natural resource use and ecosystem management on human well-being. In various situations, ecosystem management targets the enhancement of one or a few ecosystem services. For example, managers of agricultural or forestry systems typically aim to control the site's vegetation to promote the growth of desired resources, such as crops or trees of certain species. Carbon storage may be increased by afforestation and forest protection (Jandl *et al.* 2007). Wetlands may be restored with the purpose of creating a natural water purification system (Zedler 2003). However, any attempt at ecosystem manipulation for the sake of some ecosystem services may negatively affect other services. In ecosystem service literature, these kinds of unintended and undesirable impacts of ecosystem management are often described as ecosystem service conflicts or ecosystem service trade-offs (Rodríguez *et al.* 2006, Bennett *et al.* 2009). A trade-off between ecosystem services means that the use of a service or the management of an ecosystem to increase the supply of a service leads to the deterioration of another service. Trade-offs between ecosystem services can be evaluated quantitatively by indices that incorporate the loss in one service under management that targets other services (Bradford and D'Amato 2012, Rodríguez-Loinaz *et al.* 2014). It is also possible that management that increases one service increases other services – these cases can be described as synergies between ecosystem services or win-win situations.

Trade-offs between ecosystem services were in a central role already in the MEA, which reported that at a global scale humans have transformed ecosystems to increase the supply of a few services, particularly the production of food and raw materials, simultaneously causing declines in almost all other services (Anon. 2005). Since then, multiple studies have shown the same pattern of trade-offs being particularly common between provisioning and other types of ecosystem services (Rodríguez *et al.* 2006, Carpenter *et al.* 2009, Howe *et al.* 2014). An extreme case of such a trade-off is the conversion of a natural ecosystem that supplies multiple services into a managed monoculture that supplies only one product. In reality, there is much variation in the quality of managed production ecosystems in terms of their naturalness and their ability to supply multiple ecosystem services, and even with continuous human intervention these systems can be important sources of multiple ecosystem services (Swinton *et al.* 2007, Power 2010, Edwards *et al.* 2014b). Nevertheless, in all production ecosystems management and harvesting activities related to the provisioning service that is of main interest may alter the supply of other ecosystem services. Indeed, provisioning services such as the production of food and raw materials differ from other ecosystem services in that not only management to enhance their supply but also their use directly alters the ecosystem. For example, in a production forest silvicultural activities such as fertilization or pruning of the developing stand aim to promote tree growth and can, by altering the conditions in the forest, affect the supply of other ecosystem services. However, an even more substantial change takes place when the trees are harvested: the ecosystem changes overnight from a mature forest to a clearing that can host a completely different species community and perform different ecosystem functions. In the case of provisioning services, it may thus be warranted to consider the production and the extraction of the goods as separate stages of the ecosystem service delivery that may interact with other ecosystem services in different ways (cf. the discussion on the definition and indicators of provisioning services in section 1.2).

The relationships between ecosystem services may be nonlinear, and complex, multi-scale ecosystem feedbacks and dynamics can make them difficult to reveal and predict. Ecosystem service trade-offs can occur at different spatial and temporal scales (Rodríguez *et al.* 2006). The effects of actions taken in one location over a limited area may extend far beyond that area, and the same is true for the temporal scale. Bennett *et al.* (2009) differentiate ecosystem service relationships based on whether the services are affected by the same drivers or whether they interact with each other directly. In the former case, changes in ecosystem services take place because of a separate driver that affects all of them irrespective of each other. For example, harvesting wood from a forest may affect the suitability of the forest as a habitat for pollinators as well as the amount of carbon stored in the forest and thus its contribution to climate change mitigation, but pollination and carbon storage are not interacting directly with each other. In the latter case, changes in ecosystem services are caused by direct interactions among the services. An example of a strong direct ecosystem service interaction is the synergistic relationship between erosion regulation and primary production: vegetation prevents erosion and protects soil fertility, and soil fertility supports the production of vegetation. If services are affected by the same drivers, the effects may be opposite. For example, the increasing density of living trees in a forest affects positively the amount of carbon stored in the forest, but at the same time it may affect negatively the yield of forest berries that benefit from canopy openness. Also if the services interact directly, the effects of the interaction may be opposite in direction and asymmetric in magnitude. Understanding the mechanisms behind observed ecosystem service relationships is necessary to manage them effectively (Bennett *et al.* 2009).

A trade-off can also occur between the supply of an ecosystem service in the short term and in the long term. An obvious example is the extraction of resources from an ecosystem faster than the resources renew, eventually leading to their depletion (e.g., Pauly *et al.* 2002). However, such a trade-off can also occur indirectly: for example, ecosystem management for maximizing provisioning services can cause declines in regulating and supporting services on which the provisioning services themselves ultimately depend (Anon. 2005). Indeed, the simplification of production ecosystems under the aim of maximizing provisioning services has led not only to declines in other ecosystem services but to a loss of stability and resilience of production systems themselves (Kareiva *et al.* 2007). Isbell *et al.* (2015) propose that ongoing biodiversity loss and associated extinction debt can translate into an ecosystem service debt: due to habitat loss, climate change, species invasions, and other environmental changes the maintenance of biodiversity is not guaranteed even in protected ecosystems, and so the sustained supply of ecosystem services that are based on this biodiversity is not guaranteed either. Promoting system stability and resilience can be a management aim in itself (Bennet and Balvanera 2007, Biggs *et al.* 2012), and if it conflicts with other management aims – such as short-term gains of resources – the trade-off is worth revealing and analyzing.

The relationships between ecosystem services and biodiversity conservation can be described similarly to the relationships among ecosystem services. Like the relationships among ecosystem services, the relationship between ecosystem services and biodiversity conservation may be difficult to reveal due to complex ecosystem dynamics and the multi-scale interactions among the ecosystem components that are involved. As described above in section 1.2, the relationships between different aspects of biodiversity and ecosystem services are still largely unknown, but also here trade-offs or win-wins can occur. A trade-off between ecosystem services and biodiversity conservation takes place if conservation measures reduce the supply of an ecosystem service or, conversely, if management for an ecosystem service leads to a loss of biodiversity. A win-win situation is one where desired ecosystem services and conservation outcomes are achieved by the same measures; for example, replacing conifer monocultures with mixed-species stands can benefit both ecosystem services and the richness of several species groups (Felton *et al.* 2016). It is possible that management for ecosystem services promotes the conservation of taxa that may otherwise be overlooked (Mori *et al.* 2017). Then again, several authors who are critical of the ability of the ecosystem services approach to contribute to biodiversity conservation have raised concerns that the approach values more or less inevitably only a part of biodiversity, and thus even without a direct trade-off with conservation, management for ecosystem services does not guarantee the protection of biodiversity (Ridder 2008, Vira and Adams 2009).

1.4 Reconciling ecosystem service conflicts

Acknowledgement of ecosystem service and biodiversity trade-offs has spurred efforts to develop management strategies that aim to alleviate them. The different management strategies to address ecosystem service trade-offs in production ecosystems are illustrated by the concepts of land-sharing and land-sparing – two approaches that originated as contending alternatives to reconcile biodiversity conservation and the growing demand for food (Green *et al.* 2005, Fischer *et al.* 2008, Phalan *et al.* 2011, Kremen 2015). Land-sharing strategies refer to agricultural systems where low-intensity production allows for habitats for wildlife to exist within agricultural areas. Land-sparing strategies involve the separation of land into farmland that is managed intensively for high yields and protected areas that are set-aside for wildlife. The discussion on the two approaches has extended from agricultural contexts to other human land-use and conservation conflicts as well as to other ecosystem services beyond commodity production (e.g., Lin and Fuller 2013, Edwards *et al.* 2014a). In such general sense, land-sharing can refer to management strategies that are designed to support multiple ecosystem services and biodiversity within the same site, and land-sparing to the allocation of land into areas where single objectives are prioritized so that the overall outcome is as good as possible. It should be noted that the distinction between the two is dependent on the spatial scale at which they are defined as well as the criteria as to what constitutes "spared" land – indeed, Ekroos *et al.* (2016) argue that many land-sharing strategies are ultimately land-sparing at a small scale. Because species depend on habitats and ecosystem processes depend on conditions across multiple scales, there is no one correct scale at which to conduct land-sparing, but addressing the conflicts may require land-sparing activities at multiple, hierarchical scales (Kremen 2015, Ekroos *et al.* 2016). In addition, some communities and ecosystem services are so local that their maintenance requires measures also within all sites, for example, maintenance of soil communities and the functions they perform within agricultural fields (Ekroos *et al.* 2016).

If information is available on the responses of ecosystem services to management actions, strategies to reconcile conflicting demands can be identified with multi-criteria decision analysis or multi-objective optimization tools (Moilanen *et al.* 2005, Mendoza and Martins 2006, Wolfslehner and Seidl 2010, Mazziotta *et al.* 2017). Methods based on multi-objective optimization involve formulating the management question into an optimization problem and solving it with mathematical methods. Optimization methods identify the best possible solutions to the problem. With optimization tools, alternative scenarios can be created and evaluated to find the ones that best balance multiple benefits (e.g., Wainger *et al.* 2010, Miina *et al.* 2010, Schwenk *et al.* 2012, Blattert *et al.* 2017). The complete set of mathematically optimal solutions found among alternative scenarios can be explored to characterize the severity of the conflict (Seppelt *et al.* 2013, Mazziotta *et al.* 2017). Optimization analyses to evaluate management alternatives can be conducted at a site-level or across multiple sites at a higher spatial scale. In the latter case, interactions among sites can be included in the analysis, for example in land-use optimization for conservation area networks (Cabeza and Moilanen 2001). Management optimization at a site-level is analogous to finding a land-sharing solution to the conflict, while optimization across multiple sites can result in a land-sparing type solution or a combination of the two types of strategies.

As described in section 1.2, the delivery of ecosystem services involves both ecological processes that form the potential ecosystem service supply and social system processes that create ecosystem service demand and use. These two aspects can be distinguished also with respect to ecosystem service trade-offs. For example, if logging a forest causes a decrease in the land's water retention ability, a clear conflict exists between timber harvesting and water flow regulation. However, if there is no demand for the timber from the area, the conflict may never manifest itself as a management problem that needs to be solved. Just like an ecosystem function becomes an ecosystem service only when there are humans benefiting from it (Haines-Young and Potschin 2010), an ecosystem service trade-off becomes a problem only when opposing human demands are involved. Yet, this may be the case more and more often as the human population increases and so does the demand for ecosystem services (Anon. 2005). According to the review of Howe *et al.* (2014), ecosystem service trade-offs occur most often when one of the services is a provisioning service with a private beneficiary and the other services are public benefits. No clear pattern was found as to when synergies between ecosystem services occur. Based on their findings, Howe *et al.* (2014) suggest that synergies may be most effectively created by tackling the socio-economic conditions that tend to lead to trade-offs, such as management failures, stakeholder exclusion, and narrow-minded prioritization of provisioning services.

In short, conflicts and synergies among ecosystem services and between ecosystem services and biodiversity conservation are created by the interactions within and between ecological and social systems. Human demand for certain services leads to the modification of ecosystems to promote those services, and the potential side effects of the ecosystem changes reflect back onto human societies and human well-being. Trade-offs between ecosystem services may be addressed either by technical solutions, that is, by developing the ways in which ecosystems are managed, or by changing the aims of ecosystem management and the socio-economic conditions that create situations where few benefit at the expense of others. In any case, managing ecosystem service conflicts should be a key part of ecosystem management (Rodríguez *et al.* 2006, Kareiva *et al.* 2007, Bennett *et al.* 2009). This requires that the ecosystem service trade-offs as well as opportunities for synergies are acknowledged and understood.

1.5 Aims of the thesis

In this thesis, I explore the occurrence and severity of conflicts among ecosystem services and biodiversity conservation in Finnish forests. Finnish forests are extensively used for timber production but also for other ecosystem services. Due to the growing pressures to intensify the utilization of timber resources combined with the wide range of values people associate with forests in Finland, there is high potential for ecosystem service conflicts to occur in Finnish forests. Then again, the long history of forest management in the country has created a wealth of information and expertise that may benefit the development of forestry practices that fulfill people's various demands and improve forestry's ecological sustainability (Moen *et al.* 2014). Earlier work has shown that there are trade-offs between timber harvesting and biodiversity, carbon storage and sequestration, and forest collectables in Finnish forests, and that these trade-offs can be to some extent alleviated by forest management planning (Mönkkönen *et al.* 2014, Triviño *et al.* 2015, Peura *et al.* 2016). However, open questions remain in regard to the occurrence and severity of trade-offs among non-timber forest ecosystem services, and the dependence of the trade-offs on spatial and temporal scales.

First, in an overview of literature I explore the applicability of the ecosystem services framework in the context of Finnish forestry as a tool to identify diverse benefits derived from forests as well as potential conflicts among them. Second, I use forest inventory data, forest growth simulations, and multi-objective optimization in three case studies to identify and characterize conflicts among forest ecosystem services and biodiversity conservation. I aim to reveal how the severity of the conflicts varies among combinations of ecosystem services, across spatial scales, and across time. I use multi-objective optimization to explore how the conflicts could be mitigated by forest management planning – how good are the solutions that are found, and what do they entail from a forestry perspective. Specifically, the thesis aims to resolve the following questions:

1. How has the ecosystem services framework been previously used in the context of Finnish forestry? What is known about the impacts of forestry on ecosystem services in Finland and what are the main knowledge gaps that remain? (I)
2. How severe are the conflicts among multiple ecosystem services in Finnish production forests? How can they be mitigated by forest management? (II)
3. How does the perceived severity of an ecosystem service conflict depend on the spatial scale of observation? What is the most efficient scale of management planning aiming to mitigate an ecosystem service conflict? (III)
4. Is there a conflict between intensive forestry and the resilience of multiple forest ecosystem services to forestry disturbances? (IV)

2 FINNISH FORESTRY IN AN ECOSYSTEM SERVICES PERSPECTIVE

2.1 Forestry in Finland

Finnish forests are part of the boreal forest biome. Unlike other forests, boreal forests have not been severely threatened by deforestation (Anon. 2015a), and this is also the case in Finland where forest growth currently exceeds forest harvesting (Peltola 2014). Large parts of the boreal forest have, however, been substantially modified by humans (Bradshaw *et al.* 2009). Income from forests has traditionally been an important part of rural populations' livelihoods in Finland (Siiskonen 2007), and the majority of Finnish forests is still privately owned in comparatively small holdings (Peltola 2014). After a long history of forest exploitation, there is very little pristine forest left in Finland, approximately 1 % of forest area (Anon. 2015a). An extensive forest road network enables utilization of forests throughout rural areas. Approximately 13 % of forest area is protected (Peltola 2014), but the protected forests are disproportionately located in northern Finland on remote lands of low productivity. The uneven distribution of protected areas in terms of location as well as forest type and management history severely limits the ability of the protected area network to sustain biodiversity (Kuuluvainen 2009, Hanski 2011). The management and use of production forests is thus of crucial importance for the conservation of forest biodiversity in Finland.

The roots of modern Finnish forestry are in the development of the forest industry and, as a response, of intensive forest management and forest policies to promote it during the 20th century (Siiskonen 2007, Kotilainen and Rytteri 2011, Kuuluvainen *et al.* 2012). The currently dominant forestry regime in Finland became widely adopted after the Second World War. It is based on stand management for an even-aged, single-species monoculture that is clear-cut once a certain timber stock is reached, after which the stand is regenerated naturally or artificially (Äijälä *et al.* 2014). The desired stand structure, aiming to promote tree growth, is achieved by pruning and thinning of the developing stand. Growth conditions may also be improved by drainage or fertilization of the site. In the final felling the stand is clear-cut and harvest residues and stumps may also be collected. A typical rotation time from regeneration to final felling is about 80-100 years (Äijälä *et al.* 2014). Before regeneration, the site may be prepared mechanically. Forestry operations are regulated by law (Anon. 2014a), but there is flexibility to their timing and intensity (e.g., thinnings) or whether they are conducted at all (e.g., fertilization). What kinds of operations are conducted in a stand depends on the properties of the site and the goals of the forest manager (Äijälä *et al.* 2014).

Stand management and forest harvesting alter the structure of the forest both at the stand scale (e.g., age distribution, tree species distribution) and at the landscape scale (e.g., spatial continuity, size and arrangement of clear-cuts). The disturbance and successional dynamics created by forestry significantly differ from those taking place in natural conditions (Kuuluvainen 2002, 2009). In Finland, intensive forestry has led to substantial changes in forest ecosystems and landscapes as compared to pristine forests. The proportions of old forests as well as natural early successional forests have declined, conifers have been favored in place of deciduous trees, and the amount of deadwood within forests has decreased (Siitonen 2001, Kuuluvainen 2009). These changes have directly contributed to the habitat degradation and endangerment of forest species (Tikkanen *et al.* 2006).

Towards the end of the 20th century the role of the forest industry in Finland's economy declined at the same time as awareness of the environmental degradation caused by intensive forestry increased (Siiskonen 2007, Kotilainen and Rytteri 2011). Currently, interest in alternative forest management systems such as those based on selective harvesting is growing due to environmental and social concerns related to even-aged forestry as practiced predominantly for the past decades (Kuuluvainen *et al.* 2012). Maintenance of biodiversity and ecosystem services has become a stated goal of forest management and policy (Äijälä *et al.* 2014, Anon. 2015b), and some measures to improve this aspect have been adopted. For example, approximately 85 % of production forests in Finland are part of the PEFC forest certification system (Anon. 2017), the criteria of which require among other things the protection of habitats of endangered species and the retention of some living and dead trees during final felling to promote biodiversity (Anon. 2014b). Funds have been allocated to forest biodiversity conservation through the Forest Biodiversity Programme for Southern Finland (METSO), where forest owners can make voluntary agreements for the protection of valuable habitats on their land and get compensation for lost income. However, the effectiveness of these measures in safeguarding biodiversity has been questioned and remains uncertain (Mönkkönen 1999, Roberge *et al.* 2015, Kotiaho 2017). The habitat demands of many threatened species, such as old-growth forests, old trees, and large-diameter deadwood, may ultimately be highly incompatible with intensive timber production (Tikkanen *et al.* 2006). In that case, forest exploitation inevitably causes changes in forest biodiversity and hence modifies forest structures and functions. These changes may, in turn, manifest as changes in ecosystem services.

2.2 Forest ecosystem services

In the Millennium Ecosystem Assessment (Anon. 2005), ecosystem services associated with forests span all service categories and include the provisioning of wood for fuel and fiber, wild foods, ornamental resources, genetic resources, and fresh water; the regulation of climate, air quality, and water quality, erosion control, storm protection, biological control, and pollination; and a range of cultural values such as recreation and aesthetics. In addition to the MEA, a widely used ecosystem service typology is the Common International Classification of Ecosystem Services (CICES). CICES overlaps with the MEA for a large part but is more systematic in structure and perhaps more comprehensive in content. Basing their work on CICES, Saastamoinen *et al.* (2014) classified ecosystem services relevant to Finnish forests. They identified services from all categories: provisioning (logwood, pulpwood, wood fuels, forest industry by-products, organic materials, wild foods, fresh water, and genetic diversity), regulation and maintenance (mediation of waste, air and water filtration, erosion mediation, hydrological regulation, storm protection, pollination and seed dispersal, pest and disease control, soil formation, and global and regional climate regulation), and cultural services (recreational, intellectual, spiritual, and symbolic interactions). Some of the ecosystem services have global benefits, such as global climate regulation, while the benefits derived from others are local, such as water filtration. Some of the services directly benefit forestry itself by maintaining the site productivity or by protecting the stand, such as nutrient cycling, storm protection, and pest control.

According to Saastamoinen *et al.* (2014), many of the provisioning services from Finnish forests are well-known and relatively easily quantified from existing statistics, while many of the regulation and maintenance services, albeit clearly provided by Finnish forests, have not been similarly monitored and are more challenging measure. Finnish forests produce approximately 100 million m^3 of new wood annually, of which approximately 75 % is currently harvested (Peltola 2014). The harvested wood is used in wood products, pulp production, and energy production. The economic value of timber production in Finland is indicated by the total share of the forestry sector in Finland's gross domestic product, which in recent years has been 3–4 % (Peltola 2014). In addition to timber, Finnish forests provide other goods such as berries, mushrooms, and herbs, collectively termed non-timber forest products. The wild berry crop harvested from Finnish forests annually can reach tens of millions of kilos and the mushroom crop also several millions of kilos (Peltola 2014). Over half of Finns pick berries or mushrooms as a recreational activity (Peltola 2014). Of the regulating services provided by Finnish forests, carbon storage is the only one directly quantified in Finnish forest statistics (Peltola 2014). Forest carbon storage has a critical role in global climate regulation and climate change mitigation (Pan *et al.* 2011). The annual growth of Finnish forests currently exceeds the timber harvests and forms a net carbon sink of some 35 million CO_2 equivalents per year (Peltola 2014). Other environmental data are collected in Finland that are linked to some of the other regulating services provided by forests, such as monitoring water quality and forest health, but they do not directly quantify the related ecosystem services.

Many of the benefits provided by forests have been recognized long before they were labeled ecosystem services (Saastamoinen *et al.* 2014). For example, the term "multiple-use forestry" goes back several decades in Finland and has been thought to include, similarly to forest ecosystem services now, at least wood and non-wood forest products, grazing, recreation and hobbies, cultural and scenic values, protective functions and pollution abatement, and nature conservation (Saastamoinen *et al.* 1984, Hytönen 1995). Also similarly to the current debate around ecosystem services, the concept of multiple-use forestry sparked discussion about its interpretation by different stakeholders, its various dimensions, and the relationship between multiple-use forestry and nature conservation (Hytönen 1995). "Relations between different forest uses", as written about by Saastamoinen *et al.* in 1984, is parallel to the current discussion on conflicts and synergies among forest ecosystem services. It is thus evident that there is a long-run recognition of not only the multiple benefits derived from forests but of the fact that their simultaneous maintenance may not be guaranteed.

Despite the long-run recognition of forests' diverse benefits to humans, the extensive research on the impacts of production forestry on forest landscapes and ecosystems, and the already relatively long history of ecosystem service research, a survey of existing literature conducted as part of this thesis showed that the ecosystem services framework has been used in a boreal production forestry context to a very limited extent (I). While large bodies of literature exist on the impacts of boreal forestry on some ecosystem functions that are linked to ecosystem services, such as carbon dynamics, soil retention, and nutrient cycling, the use of the ecosystem services terminology within this literature has so far been marginal (I). In an extensive literature review, Abson *et al.* (2014) found that literature on forest ecosystem services has so far been focused on tropical forests. In general, the ecosystem services concept originated essentially as a new terminology to describe an old idea (Vihervaara *et al.* 2010). However, the approach has its perceived merits (e.g., Thompson *et al.* 2011), including providing a link between science and policy and a consistent framework to apply across diverse contexts to describe human-nature dependencies. If these merits are accepted, the ecosystem services framework is highly applicable also in the context if Finnish forestry.

2.3 Potential effects of forestry on ecosystem services

The basis of ecosystem services is in ecosystem structures and functions, i.e. the interactions between living organisms and their environment (section 1.2). Impacts of human activities on species communities and habitats may translate into effects on ecosystem services. Although the ecosystem service concept itself has been used so far marginally in this context, potential effects of forestry on ecosystem services in Finland can be identified from literature describing the environmental and social impacts of Finnish forest management. An overview of relevant literature shows that intensive forestry as practiced in Finland, and throughout the boreal region, may have impacts on the supply of numerous ecosystem services (I). These impacts may be negative or positive, depending on the case. Much information is still missing, and the processes underlying some ecosystem services and forestry's impacts on them are better known than others.

The most widely studied forest ecosystem services and related ecosystem functions are maintenance of soil productivity, regulation of water flow and quality, and climate regulation or, specifically, carbon dynamics. These services may be affected especially by biomass removal and harvesting, management of stand structure, and soil preparation. Maintenance of soil productivity is naturally in the interest of forest managers and impacts on soil have thus been widely studied. Soil impoverishment is not considered a real problem in boreal forests (Grigal 2000, Kreutzweiser *et al.* 2008), but some of the impacts remain poorly understood, particularly the long-term consequences of changes in soil communities (Hartmann *et al.* 2012). Reduced nutrient and soil retention ability may also impact negatively on the quality of waters adjacent to the forest site (Kortelainen and Saukkonen 1998). One way to address these harmful effects is by leaving unfelled buffer zones to protect the waters (Gundersen *et al.* 2010). Carbon sequestration associated with production forests is a result of forest growth, wood harvesting, and the fate of the carbon fixed in the harvested wood (Liski *et al.* 2001). Forestry reduces the forest's contribution to climate change mitigation if it results in releases of carbon into the atmosphere from long-term storages in the forest ecosystem, for example if timber from old-growth forests is used in short-lived wood products, or if soil carbon storages are disturbed by forestry activities. Then again, forest management may increase carbon sequestration by promoting tree growth.

Management of stand structure by thinning, tree species selection, and ultimately final felling also affect the stand's suitability as habitat for various ecosystem service providers, such as non-timber forest products, pollinators, decomposers, and natural control agents, and may consequently affect the supply of the services they provide. The responses of some non-timber forest products to forestry, such as wild berries, have been rather extensively studied in Finland, and both negative and positive impacts have been identified (Miina *et al.* 2009). Forestry's potential effects on pollinators, decomposers, and natural control agents, then again, are still narrowly studied. For example, the three-toed woodpecker (*Picoides tridactylus*) is a predator of spruce bark beetles and so a potential natural control agent of this forest pest, and the ability of the species to stabilize pest populations is suggested to be affected by the structure of its forest habitat (Fayt *et al.* 2005). However, the ability of the woodpecker to respond to the population changes of its prey depends on complex, multi-scale population processes, and their responses to the alterations in ecological conditions are difficult to identify (Fayt *et al.* 2005).

Stand and landscape structure may severely affect also the recreational and aesthetic values of the forest. These values depend on variable, individual preferences perhaps more heavily than the values associated with any other ecosystem services, and as such are challenging to assess. Some generalizations have, however, been suggested. For example, based on a review of preference surveys from the northern Europe, Gundersen and Frivold (2008) found commonly preferred forest features to include factors such as naturalness and accessibility, whereas clear-cuts and other signs of forestry operations were commonly disliked.

In short, production forestry may have diverse impacts on forest ecosystems services. The effects of forestry activities on ecosystem services may be highly site-dependent as well as case-dependent – for example, simultaneously harmful for some services and beneficial for others. Overall, the highest potential for ecosystem service conflicts appears to occur when forestry activities are intensive and disturbances to the state and functioning of the forest are acute, such as when the forest is clear-cut and the soil is disturbed. In addition, all forest management choices that affect the structure of the forest may alter the community that can occupy it and the functions it can perform. Many knowledge gaps remain with respect to the responses of ecosystem services to forestry activities. Perhaps the most notable of them is the lack of understanding on the role of community structure and diversity in the provision of ecosystem services and, subsequently, on the potential effects of community changes or loss of diversity on ecosystem service supply. Uncertainties also remain with respect to the ability of production forests to provide diverse ecosystem services in the long-term future, when they are faced with pressures such as intensifying exploitation and climate change (Lindner *et al.* 2010, Laudon *et al.* 2011).

3 CASE STUDIES

3.1 Methods

3.1.1 Study areas and forest data

To explore the relationships among forest ecosystem services, forest inventory data from Finnish production forests were used together with forest growth simulations and ecosystem service models. The Finnish Forest Centre produces forest inventory data for privately owned forests in Finland using remote sensing and field measurements. Forest data collected with remote sensing methods are highly available in Finland and have been used to map biodiversity and ecosystem properties (Vehmas *et al.* 2009, Vihervaara *et al.* 2015).

Forest inventory data from a total of 17 study areas were used: the municipality of Hankasalmi in central Finland (II) and up to 16 small catchment areas from southern and central Finland (III, IV). The data was structured as stand-level, where a stand is a parcel of forest of relatively homogenous structure and type. Stands defined in this way are the basic operational units of forestry planning and management. The Hankasalmi area comprised nearly 30,000 stands, and the small catchment areas between approximately 400 and 3,500 stands. All of the areas represent typical Finnish production forests: the distribution of current stand age is skewed towards low values, the stands consist principally of spruce (*Picea abies*), pine (*Pinus sylvestris*), and birch (*Betula pendula* and *Betula pubescens*), and the landscape is dominated by forest with smaller proportions of settlements, agricultural areas, peat lands and lakes (Fig. 1). Individual stands are, on average, relatively small with a mean size of approximately 1.5 hectares. The data from the Hankasalmi area have been used in previous work and are described in more detail in associated publications (Mönkkönen *et al.* 2014, Triviño *et al.* 2015).

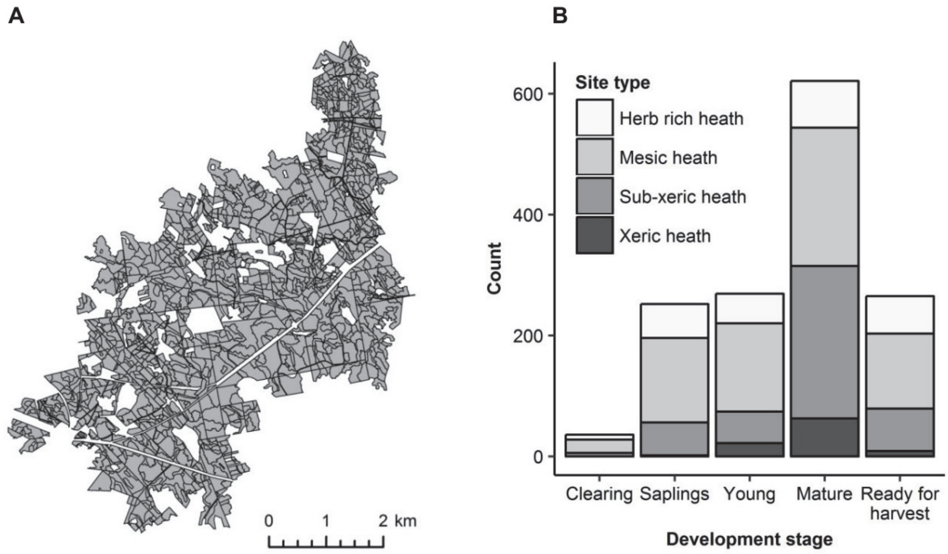

FIGURE 1 Example of a study area: the catchment area used in study IV. Shown are the 1,400 stands drawn as a map to show their size and coverage of the area (A), and their distribution as classified into current development stage (x axis) and site type (bar fill) (B). The development stage 'Young' refers to a stand with an average diameter at breast height of 8–16 cm, and 'Mature' to a stand with an average diameter at breast height greater than 16 cm but that is not yet ready for final harvest. The development stage 'Ready for harvest' is defined according to current Finnish recommendations for the timing of final felling (Äijälä *et al.* 2014). The site types are presented according to the Finnish forest classification system (Hotanen *et al.* 2008). The development stages and site types have been defined during the data collection by the Finnish Forest Centre. Panel (B) reprinted from chapter IV.

3.1.2 Forest simulations

Two different forest simulators were used, MOTTI (Hynynen *et al.* 2002, Salminen *et al.* 2005) (II) and SIMO (Rasinmäki *et al.* 2009) (III, IV). Both MOTTI and SIMO are software in which forest growth models are used to predict stand development based on the stand's initial characteristics and the forestry operations taken in the stand. The timing and intensity of forestry operations are determined by decision rules that can be adjusted to reflect different forest management regimes in order to create alternative future scenarios. In MOTTI, a simulation period of 50 years was used (II), and in SIMO a simulation period of 100 years (III, IV). In all studies the simulation period was divided into 5-year time steps so that the simulator predicted stand development at each time step.

Forest stands were simulated with a selection of alternative management regimes in order to explore the different, achievable futures of the study forests as comprehensively as possible. The MOTTI simulations were performed for earlier work (Mönkkönen *et al.* 2014, Triviño *et al.* 2015), and encompassed 7 alternative management regimes (II). The SIMO simulations in turn included a total of 19 alternative regimes (III, IV). Both MOTTI and SIMO simulations included a regime reflecting the current Finnish stand management recommendations (Äijälä *et al.* 2014), the recommended regime with modifications, and set-aside. The recommended regime is based on even-aged rotation forestry and consists of one or more thinnings, final felling by clear-cut, and stand regeneration by planting or seeding. In the modified versions of the recommended regime, the timing of final felling was postponed, thinnings were omitted, or green tree retention at final felling was increased. The MOTTI simulations included 5 and the SIMO simulations 16 such modified versions. In the set-aside option, no forestry operations were conducted and no timber was harvested from the stand. In addition to these regimes, the SIMO simulations included a continuous cover forestry regime, where regular, selective harvesting of large trees was conducted instead of final felling and new trees were generated naturally (Pukkala *et al.* 2012).

The forest management regimes included in the simulations were intended to reflect the real-world options available to forest managers. They create stands that differ from each other in terms of their structural features such as the number of standing trees, the distribution of tree age and size, and the amount of deadwood, and in terms of the development of these features over time.

3.1.3 Ecosystem service models

The potential supply of and the relationships among a total of six ecosystem services were analyzed. These were: timber production, carbon storage, pest regulation, bilberry production, cowberry production, and scenic beauty. This selection represents different types of ecosystem services (provisioning, regulating, and cultural), a range of spatial scales from local to global in which the benefits from the services are experienced by humans, and different types of beneficiaries with public or private interests in the service. For example, the benefits of scenic beauty are highly local as they are tied to the exact location of the forest, while the benefits of carbon storage are global via global climate change mitigation. In a privately owned forest, the forest owner has a private, monetary interest in the ecosystem service of timber production, while the benefits of, for example, bilberry and cowberry production are public as collecting the berries is permitted to anyone in Finland by the so called everyman's right. All six services were evaluated using previously developed models that relate forest features to the ecosystem services. This way, predictions of stand structure and development produced by the forest simulations could be directly used to predict ecosystem service supply.

Timber production was evaluated with three different measures: total biomass of harvested commercial timber (II), discounted income from harvested timber (III), and net present value of the forest (IV). Total biomass of harvested commercial timber (m^3) consisted of pulpwood and saw logs collected during thinnings and final felling. Discounted income from harvested timber (€) was calculated by multiplying the harvested quantities of different timber assortments by their recent average prices (Peltola 2014). A discount rate of 3 % was used to discount income generated in the future. Net present value of the forest (€) was calculated as the sum of income from timber harvests, the expected revenue from standing timber and bare soil, and the costs of silvicultural operations. Here, a 1 % rate was used to discount future income.

Carbon storage (metric tons) was measured as the total amount of carbon fixed in living and dead tree biomass in the forest and in extracted timber (II) or as carbon fixed in living trees, deadwood, and the forest soil (III, IV). Carbon fixed in living, dead, and extracted wood was estimated as 50% of the biomass (Anon. 2003b). Soil carbon was estimated with the Yasso07 models (Liski *et al.* 2005, Tuomi *et al.* 2009, 2011) for mineral soils and a carbon flux model (Ojanen *et al.* 2014) for peatland soils.

Pest regulation was measured using habitat availability for three-toed woodpecker (*Picoides tridactylus*) as an indicator (II). Three-toed woodpecker is an important predator of bark beetles and has the potential to stabilize bark beetle populations and regulate the occurrence of outbreaks (Fayt *et al.* 2005). Habitat availability for three-toed woodpecker was estimated with a habitat suitability model developed by Mönkkönen *et al.* (2014). The preferred habitat of three-toed woodpecker is mature forest with abundant deadwood. The model of Mönkkönen *et al.* (2014) estimates the suitability of a stand as three-toed woodpecker habitat based on the total basal area of recently died trees and the total stem volume of living trees. The model results in an index that varies between 0 and 1 and is related to the probability of the presence of the species in a stand.

The production (kg) of bilberry (*Vaccinium myrtillys*) and cowberry (*Vaccinium vitis-idaea*) were estimated using the models of Miina *et al.* (2009) and Turtiainen *et al.* (2013), respectively (II, IV). In both cases species coverage and berry yield is predicted based on stand characteristics, for example, dominant tree species, stand age, and stand basal area.

Scenic beauty was evaluated using the index developed by Pukkala *et al.* (1988) (IV). The index incorporates stand properties such as forest age, structure, and tree species composition to estimate the recreational and aesthetic attractiveness of the forest.

In addition to the six ecosystem services, one biodiversity indicator, the availability of deadwood resources, was included in the studies (II, IV). Availability of deadwood resources was measured as the total amount of deadwood (m^3) multiplied by the diversity of deadwood types. A total of 20 different deadwood types were considered, given by the 4 most common tree species (pine, spruce, and two birch species) and 5 decay stages (Mäkinen *et al.* 2006). Diversity over the 20 types was measured with Simpson's diversity index.

3.1.4 Multi-objective optimization

Forest management trade-offs were studied using multi-objective optimization to reveal what kind of levels of multiple ecosystem services could be simultaneously achieved (II), and how the achievable levels were affected by the spatial scale of optimal management allocation (III) and by the length of time of timber-focused forestry preceding multi-objective management (IV). Forest management was optimized to produce outcomes that were Pareto optimal in terms of two or more ecosystem services. A solution to a multi-objective optimization problem is Pareto optimal if it cannot be improved in terms of any of the objectives without being deteriorated in terms of some of the other objectives (Miettinen 1999). Pareto optimal solutions are thus a subset of all possible solutions. They make up a Pareto frontier, the properties of which can be used to describe the severity of conflicts between objectives (Seppelt *et al.* 2013, Mazziotta *et al.* 2017). For two objectives, a Pareto frontier can be graphically presented as a curve (Fig. 2), for three objectives as a surface, and so on. According to Mazziotta *et al.* (2017), a conflict between objectives can be evaluated by examining two types of points along the Pareto frontier: the extremes, where each single objective is maximized, and the "compromise" point, where all objectives are simultaneously as close as possible to their respective maximums (Fig. 2). Additionally, the shape of the frontier can inform about the marginal benefits that can be gained in some objectives by allowing for losses in others.

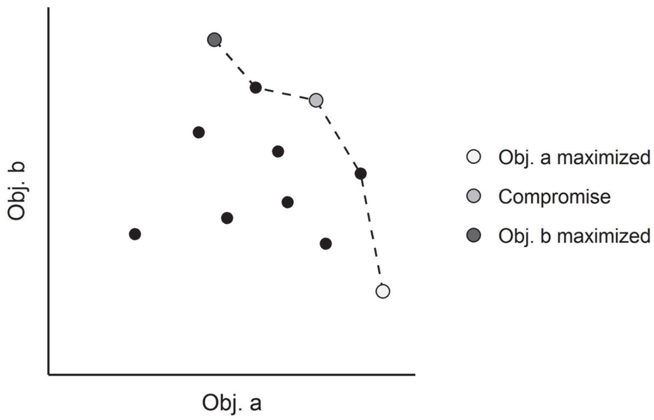

FIGURE 2 Illustration of the Pareto optimal solutions to an optimization problem involving a trade-off between two objectives ('Obj. a' and 'Obj. b'). The points show the outcomes of the two objectives under alternative scenarios. The points connected with the dashed line form the Pareto frontier, i.e., show the solutions that form the Pareto optimal set. The two extremes of the frontier show the outcomes when the objectives are respectively maximized, and the 'Compromise' point shows the outcome when the objectives are simultaneously as close as possible to their respective maximums.

To evaluate the conflicts among ecosystem services, the methodology of Mazziotta *et al.* (2017) was applied. Forest management planning was formulated into an optimization problem where the objective was to simultaneously achieve as high levels of multiple ecosystem services as possible. At the stand level, the available alternative management regimes form the solution space to the problem. At a landscape level, spanning multiple stands, the solution space is made up of all possible combinations of regimes applied across the stands. Every solution leads to an outcome that is characterized in terms of the achieved levels of the targeted ecosystem services. The Pareto optimal set among the solutions can be identified and used to characterize the trade-off between the ecosystem services as described above: by examining the extreme points on the Pareto frontier, the compromise point, and the shape of the frontier.

To identify and measure the conflicts among multiple forest management objectives, pairwise conflicts between the ecosystem services of timber production, carbon storage, bilberry production, pest regulation, and the biodiversity indicator of deadwood availability were measured as their tolerances of each other and as the goodness of compromise solutions to the pairwise conflicts (II). One objective's tolerance of another objective is defined as the proportion of its potential maximal level that can be achieved when the other objective is maximized (Mazziotta *et al.* 2017). A pair of objectives is thus characterized by two measures of tolerance: the first objective's tolerance of the second objective, and vice versa. The tolerances may be unequal, reflecting an asymmetric conflict between the objectives (Mazziotta *et al.* 2017). The compromise solution corresponds to the compromise point on the Pareto frontier, that is, the solution that results in as small losses as possible in both objectives. Its goodness is evaluated similarly to the measures of tolerance: as the proportion of potential maximal value of each objective that is achieved. Both tolerance and the goodness of the compromise solution are thus expressed in the same unit, percentage of the potential maximal value of an objective. The severity of conflicts among pairs of objectives was measured and the optimal management regimes were identified at the stand level. The distributions of the management regimes that were identified as providing the compromise solutions were examined to draw conclusions about practical management opportunities for multi-objective forestry.

The conflict between one of the pairs of ecosystem services, timber production and carbon storage, was further studied by examining the shape of the Pareto frontier and the goodness of the compromise solution when management was optimized over increasing spatial scales (III). A total of 28,900 forest stands from the 16 catchment areas were grouped together at hierarchical spatial scales. The groupings were designed to reflect administrative and/or natural boundaries and were, in increasing order by size, individual stands, small holdings, large holdings, catchment areas, and regional scale. The small holdings were defined according to real forest property boundaries. The large holdings were made up of adjacent small holdings so that each of them consisted of approximately 10 small holdings. The catchment areas were as defined by the Finnish Environment Institute. The regional scale included all of the stands. The set of Pareto optimal solutions for the joint production of timber and carbon storage was identified for each group of stands, i.e. across the different spatial scales. At the largest scale (region), this meant optimizing the management allocation over all stands. At the smaller scales, management allocation was optimized within each group and the levels of the two objectives from each group were then summed together to produce the overall outcome, so that the results across scales could be compared with each other. The aim was to reveal whether increasing the spatial scale of management optimization improves the joint production of the two services and so addresses the conflict between them more effectively. This was expected to be the case as the more stands are included in the area over which management is optimized, the more possible solutions there are and thus the higher chance of finding an efficient solution.

Finally, the compromise point found by multi-objective optimization for six forest management objectives was used as a representation of forest multifunctionality, and its resistance and resilience to intensive forestry were analyzed (IV). The six objectives included here were timber production, carbon storage, bilberry production, cowberry production, scenic beauty, and deadwood availability. The study area was one catchment area consisting of 1,400 stands. Forest growth simulations and multi-objective optimization were used to create alternative future scenarios where the study forest was managed for maximal timber production, for maximal multifunctionality, or first timber production and then multifunctionality for five consecutive 20-year planning periods. First, multifunctionality achievable under multifunctionality-focused management was compared with that achievable under timber-focused management to evaluate its resistance to intensive forestry. Second, scenarios where the management focus was changed from timber production to multifunctionality were compared with consistent targeting of multifunctionality in order to reveal if timber-focused management reduces forest multifunctionality that is achievable in the future. This corresponds to the resilience of forest multifunctionality, that is, its ability to recover from intensive forestry. Third, to illustrate the outcomes of multifunctionality-focused and timber-focused management, they were compared in terms of the resultant structure of the forest and the levels of the six objectives individually.

3.2 Results and discussion

3.2.1 Trade-offs between timber production and other ecosystem services are the most severe

The outcomes of forest management prioritizing a single objective and management aiming to reconcile two objectives were compared to assess the pairwise conflicts between the objectives (II). A total of ten pairs of management objectives, formed by four ecosystem services (timber production, bilberry production, carbon storage, and pest regulation) and one biodiversity indicator (availability of deadwood resources) were analyzed. There was high variability in the severity of conflicts among the pairs. Pairwise conflicts between ecosystem services were analyzed at the stand level, so when the same type of forest management maximized both of the ecosystem services in the same stand, there was no conflict between them in that stand. The share of stands where there was no conflict between a pair of objectives was low when one of the objectives in the pair was timber production, (1.4–31.2 % of stands, depending on the other objective in the pair) or bilberry production (25.1–34.7 % of stands). For all other pairs, involving carbon storage, pest regulation, and deadwood availability, there was no conflict in a majority of stands (80.5–90.8 %).

Where there was a conflict between a pair of objectives, its severity was measured by pairwise tolerance indices. Timber production showed the greatest level of conflict with the rest of the evaluated objectives, as the levels of harvested timber were very low when another objective was maximized (median values of 0–17.3 % of timber production's potential maximum; Fig. 3A–D). In turn, maximizing timber production led to losses in the other objectives as shown by their low tolerances of timber production. In particular, pest regulation had a low tolerance of timber production (median value of 39.3 %; Fig. 3B). There were strong conflicts also between bilberry production and pest regulation and bilberry production and deadwood availability (median values when bilberry production was maximized were 45.7 % for deadwood availability and 26.2 % for pest regulation; Fig. 3E–F). For all of the rest of the pairs, even when there was a conflict, it was less severe as the tolerance indices were higher (52.0–82.4 %; Fig. 3G–J).

Also the compromise solutions to the pairwise conflicts were very variable in terms of their goodness with respect to the different objectives. For the most severe conflicts between timber production and the other objectives, the compromise solutions were markedly unbalanced: they were very favorable for timber production, as its median values under the compromise solutions were 100 % for all pairs, but hardly different from timber-focused management for the other objectives, as their median values were only some percentage points higher than their respective tolerances of timber production (Fig. 3A–D). For the pairs not involving timber production, the compromise solutions were more balanced with all objectives reaching median values ranging between 80.3 % and 100 % (Fig. 3E–J). The pairwise conflict between pest regulation and carbon storage shows an example of an efficient compromise solution: when carbon storage was maximized, pest regulation could reach only about half of its potential maximum (median value of 52.0 %). Under the compromise solution, pest regulation could reach 100 % of its maximum while at the same time 97.0 % of maximal carbon storage was maintained (Fig. 3I).

The distribution of management regimes providing the compromise solutions also varied greatly between the pairs. Most notably, when timber production was not one of the objectives in the pair, the solutions were dominated by the set-aside regime, meaning the protection of the stand with no timber harvesting. When timber production was one of the two objectives, the solutions were more or less evenly distributed among the regime following current Finnish stand management recommendations and its modified versions (increased green tree retention, postponed final harvesting, or no thinnings).

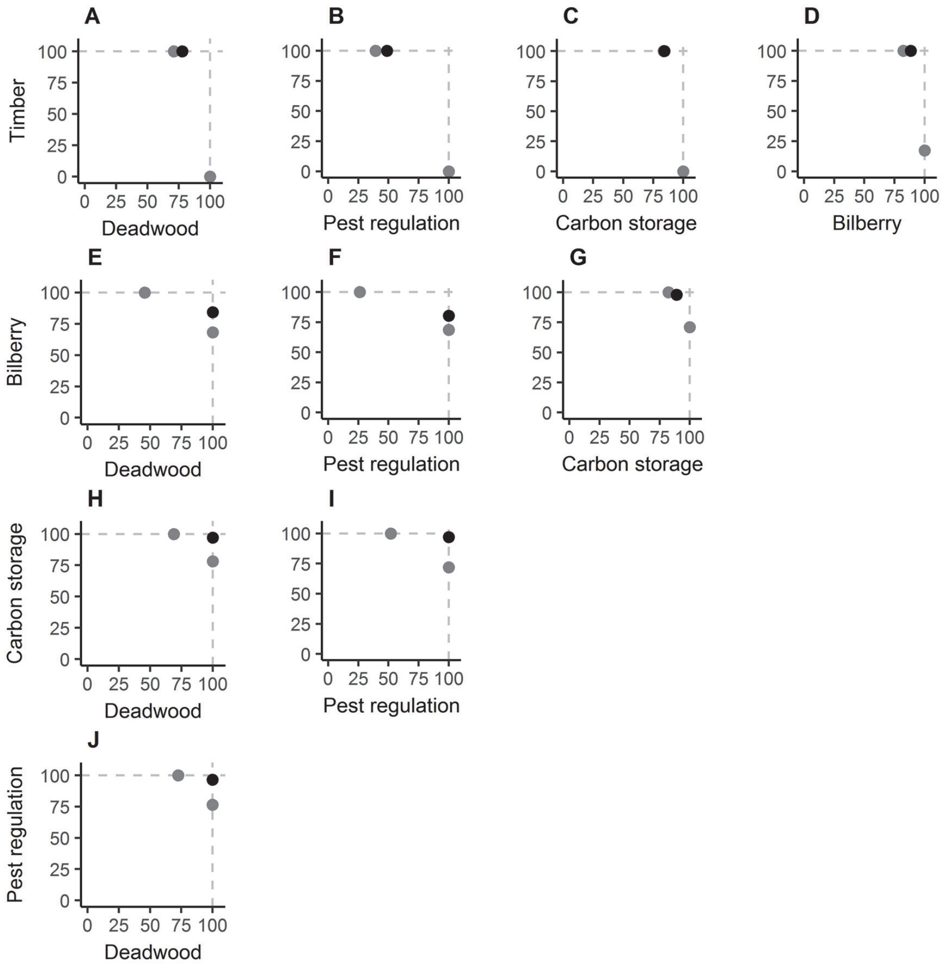

FIGURE 3 Pairwise tolerance indices (grey dots) and compromise solutions (black dots) for all pairs of objectives. The dots show the median values across the stands where there was a conflict between the pair of objectives (see text). The unit on all axes is percentage of the potential maximal value of an objective (%). Dashed grey lines have been added at y = 100 % and x = 100 % for graphical comparison. Reprinted from chapter II.

The patterns discovered in this study were as expected based on previous literature (Gamfeldt *et al.* 2013, Mönkkönen *et al.* 2014, Triviño *et al.* 2015, Peura *et al.* 2016, Pukkala 2016, Vauhkonen and Ruotsalainen 2017): there were conflicts between timber production and other objectives, and they were more common and more severe than the conflicts among the non-timber objectives. In addition, the conflicts between timber production and the other objectives were difficult to solve in balanced ways. What is more, the set-aside regime was superior in maximizing the non-timber objectives and as a solution to conflicts among them when there was one. By definition, under the set-aside regime the stand is not harvested at all and timber production is thus zero – hence the low tolerance of timber production to the non-timber objectives. Conversely, the low values of the non-timber objectives under the compromise solutions with timber production suggest that in most stands the management regimes other than set-aside were more or less equally bad for the non-timber objectives. True land-sharing solutions were thus not found among the management alternatives included in this study. Overall, the results show that in Finnish production forests the conflicts between the primary provisioning service of timber and other forest management objectives are common, severe, and challenging to solve while non-timber objectives including biodiversity conservation either are not in conflict or can be relatively easily reconciled with each other. The apparently inevitable losses in non-timber benefits caused by timber harvesting poses a challenge for the development of forest management practices and policies.

3.2.2 Forestry planning at relatively small scales enables good compromises

The effects of the spatial scale of analysis on the perceived severity of and solutions to the trade-off between timber production and carbon storage were examined (III). The extreme solutions where either timber production or carbon storage was prioritized were the same regardless of the spatial scale of management optimization and showed a clear conflict between the two ecosystem services: the losses in one when the other was maximized were considerable. When timber production was maximized, carbon storage reached 66 % of its potential maximum, and when carbon storage was maximized, the net present income for timber production reached only 5 % of its potential maximum. Beyond the two extremes, there were indeed differences in the apparent severity of the conflict between the services depending on the spatial scale of management optimization. The patterns were as expected: the smaller the spatial scale of management optimization, the steeper was the resulting production possibility curve and thus the stronger the conflict (Fig. 4). As the scale of the management optimization was increased, the compromise solution improved in terms of both objectives. At the stand scale, both objectives could simultaneously reach approximately 76–77 % of their maximal values, at the small holding scale 82–84 %, and at the three largest scales 84–85 %. Beyond the large holding scale, corresponding to approximately 100 forest stands or 200 ha of forest land, the improvements in increasing the scale became negligible. The results thus indicate that this scale is large enough to effectively mitigate the conflict between timber production and carbon storage.

In general, increasing the spatial scale over which management allocation is optimized increases the computational complexity of solving the optimization problem (Martin 2001). In addition, it potentially reduces the feasibility and acceptability of implementing the solution in the real world if it entails an uneven distribution of costs and benefits among land owners (Kurttila *et al.* 2002, Jumppanen *et al.* 2003). The results of this study, indicating that planning over relatively small forest areas can mitigate ecosystem service trade-offs effectively, are thus tentatively encouraging for the use of multi-objective optimization tools in real-world forest management. It should be noted, however, that the ecosystem services included in this study, timber production and carbon storage, provide benefits more or less irrespective of the size or location of the site. This is not true for many other ecosystem services, such as pollination, water filtration, and recreation, that are the products of ecosystem processes linked to fixed spatial scales or connectivity patterns (Mitchell *et al.* 2015, Kukkala and Moilanen 2016). Accounting for the supply of such ecosystem services in forest management planning may make it more difficult to address the conflicts, with larger areas required for efficient compromise solutions.

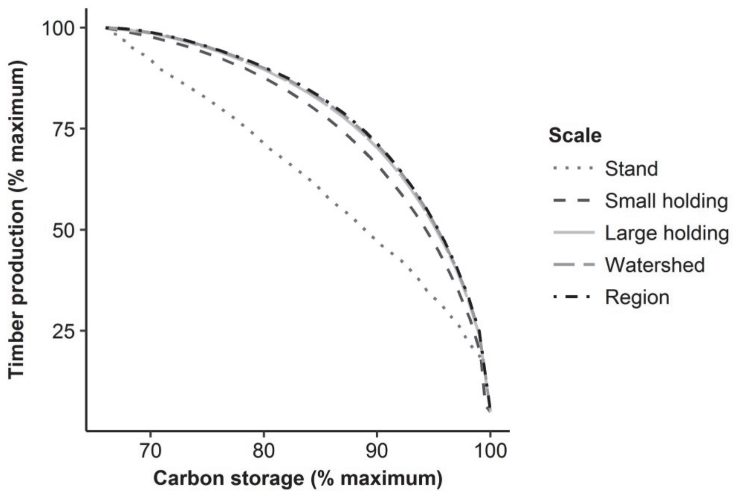

FIGURE 4 Production possibility curves showing the simultaneously achievable levels of timber production and carbon storage summed over the entire study area when optimal management allocation was determined at different spatial scales. The values of timber production and carbon storage are shown as relative to their potential maximal values. Reprinted from chapter III.

3.2.3 Loss of multifunctionality and resilience under intensive forestry

The resistance and resilience of forest multifunctionality to intensive forestry was analyzed by comparing alternative future scenarios created for the study forest (IV). Forest multifunctionality was defined as the condition where six forest management objectives (timber production, carbon storage, bilberry yield, cowberry yield, scenic beauty, and deadwood availability) are simultaneously as close as possible to their potential maximal values. It thus corresponds to the compromise point found by multi-objective optimization for the six objectives. The value of multifunctionality was expressed as the proportion (%) of their respective maximal values that all of the objectives could reach at the same time. When the study forest was managed consistently with the aim of maximizing multifunctionality, all of the objectives could reach up to over 60 % of their potential maximums (Fig. 5). When management aimed for maximal timber production, multifunctionality was considerably lower, varying between 20–30 % across the planning periods (Fig. 5). The large difference means that multifunctionality has low resistance to intensive forestry.

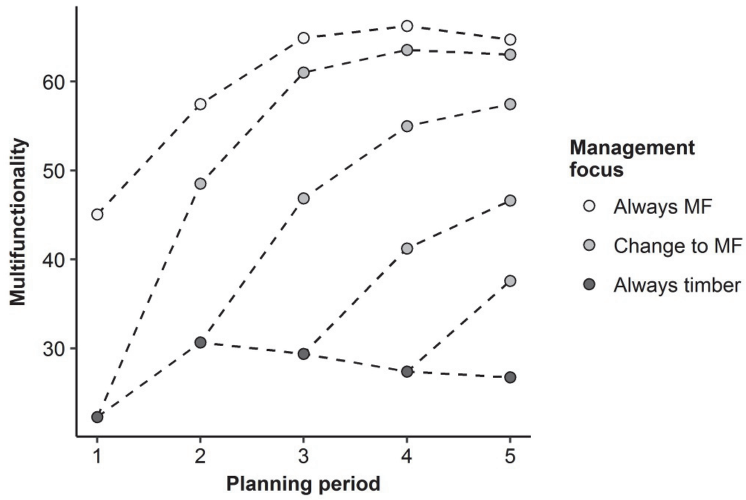

FIGURE 5 Forest multifunctionality across time under alternative future scenarios. The dashed lines have been added to connect the points to visualize the progression of the alternative paths. In the legend, 'MF' refers to multifunctionality. The value of multifunctionality shows the proportion (%) of their respective maximal values that all of its components can reach at the same time. Reprinted from chapter IV.

When the aim of management was changed from maximal timber production to maximal multifunctionality between planning periods, multifunctionality began to increase (Fig. 5). However, in all of the scenarios where the forest had first been managed with a timber production focus, multifunctionality remained lower than under consistently multifunctionality-focused management. That is, it did not fully recover from intensive forestry within the timeframe of the study. What is more, the longer the forest had been managed with a timber production focus before the change to a multifunctionality focus, the lower were the values of multifunctionality achieved throughout the time following the change and the longer it took for multifunctionality to recover. The decrease in the rate of multifunctionality's recovery caused by timber-focused management can be interpreted as a loss of its resilience.

There were clear differences in the outcomes of multifunctionality-focused and timber-focused management in terms of the supply of individual objectives as well as the structure of the forest. Multifunctionality-focused management was consistently more favorable for carbon storage, scenic beauty, and deadwood availability and almost always more favorable for bilberry production than timber-focused management. Timber-focused management was more favorable for cowberry production. Multifunctionality-focused management was surprisingly beneficial in terms of the economic value of the forest: under multifunctionality-focused management the forest's economic value increased consistently and was higher than under timber-focused management already after the first planning period (Fig. 6A). However, clearly less, if any, timber was actually harvested from the forest under multifunctionality-focused than under timber-focused management (Fig. 6B). Under multifunctionality-focused management the standing stock of timber increased in time and so did the expected revenue from harvesting it, but these harvests hardly ever actually took place. Conversely, under timber-focused management high harvests in the first planning period reduced the standing stock and by consequence the expected revenue in the future. The development of the forest confirms the tendency towards minimal timber harvests under multifunctionality-focused management: here, average stand age increased in time up to 110 years, whereas under timber-focused management it reached at most only 59 years (Fig. 6C). Also average stand basal area was consistently higher under multifunctionality-focused management than under timber-focused management (up to 18 m^2 ha^{-1} as compared with 9 m^2 ha^{-1}; Fig. 6D). Because the income from harvests did not carry over across planning periods but the value of standing timber did, the calculated economic value of the forest became higher under multifunctionality-focused than timber-focused management (Fig. 6A). However, this value does not reflect realized income to the forest owner.

In summary, the results of this study indicate that intensive forestry leads not only to a substantial loss of forest multifunctionality but also to a loss of its resilience. The revealed patterns indicate that these impacts are due to the negative effects of timber harvesting on non-timber ecosystem services and biodiversity. They highlight that the conflict these non-timber objectives have with timber production is, specifically, with timber harvesting, not tree growth or economic value. The decrease in the resilience of forest multifunctionality caused by intensive forestry brings into question the long-term sustainability of forestry – choices made now may have long-lasting impacts. If high levels of non-timber benefits are to be maintained in the long-term, the effects of forest management on them should be considered already now. Forests develop and respond to management activities with long timeframes. It is therefore of great importance to understand and promote the ability of these ecosystems to resist and recover from changes (Chapin et al. 2007; Reyer et al. 2015).

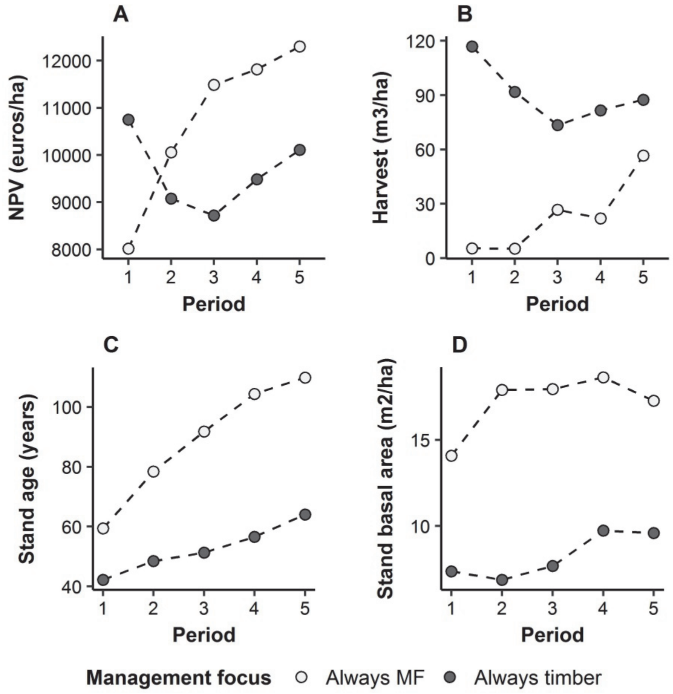

FIGURE 6 The economic value of the forest (net present value, 'NPV'; panel A), the amount of timber harvested (B), the average stand age (C), and the average stand basal area (D) under five consecutive 20-year planning periods of multifunctionality-focused management ('Always MF') or timber-focused management ('Always timber'). In (A) and (B), shown are the per-hectare values averaged across the study area. In (C) and (D), shown are the values averaged across the stands. Reprinted from chapter IV.

4 CONCLUSIONS: THE SUSTAINABILITY OF FINNISH FORESTRY IN QUESTION

Based on the results of this thesis, there are clear and challenging conflicts between intensive forestry and ecosystem services and biodiversity in Finland. Ecosystem services provided by Finnish forests are diminished when the forests are used intensively as a source of timber, whereas reducing or refraining from harvests maintains comparatively high levels of multiple non-timber services (II, IV). Moreover, the negative impacts of forestry may be cumulative and long-lasting (IV). The use of scenarios and optimization tools can help planners to identify forest management strategies that balance conflicting objectives as well as possible, especially if the analyses are conducted and the plans implemented at large enough scales (III). However, the fact that there are trade-offs among management objectives means that losses in some objectives are inevitable.

The case studies conducted as part of the thesis were based on forest management being a shared driver that impacts multiple ecosystem services via its alterations to the structure of the forest. A limitation of the analyses is that interactions among ecosystem services or ecosystem services and biodiversity were not considered. In particular, regulating and supporting services may increase the supply and stability of provisioning and cultural services (Bennett *et al.* 2009). Similarly, biodiversity can be linked to the magnitude and stability of ecosystem functions (e.g., Tilman *et al.* 2014). The comprehensive, long-term consequences of biodiversity loss and other ecosystem alterations on ecosystem services in Finnish forests, including timber production itself, remain for a large part an open question (I).

The ecosystem services approach has strong links to the concept of sustainability (Abson *et al.* 2014): the utilization of natural resources in ways that do not threaten the needs of current and future generations. In the context of Finnish forestry, the interpretation of sustainability and perceptions of the relative importance of its different aspects have changed through time. According to Kotilainen and Rytteri (2011), sustainable forestry has been understood in Finland as the protection of forests from deforestation in the latter half of the 19th century, as the active manipulation of the forest ecosystems for maximal tree growth during the 20th century, and as a combination of economic, ecological, and social aspects at the turn of the millennium. Despite the recent emergence of ecological and social concerns, the foundation of forest management and policy is still in the industrial use of forests (Kotilainen and Rytteri 2011), and sustainable forest use is still predominantly interpreted as a sustained yield of wood biomass (Kotiaho 2017). If social acceptability and the maintenance of biodiversity are considered as components of sustainability that are at least as important as economic development (Kuhlman and Farrington 2010), the sustained yield of biomass does not suffice to deem forestry sustainable. Instead, sustainable forestry would ideally meet the demand for timber without endangering non-timber benefits, with no harm to forest wildlife, and with maximal contribution to climate change mitigation. From decades of research documenting the harmful impacts of forestry on biodiversity it is already known that this does not reflect reality. However, the consequences of these impacts for human well-being are not straightforward.

Like the concept of sustainability, the concept of ecosystem services is open to differences of interpretation. Ecosystem services are not fundamental properties of ecosystems but are defined by humans and as such based on human preferences and perceptions (Haines-Young and Potschin 2010) – the concept of ecosystem services is thus inherently normative and subjective (Menzel and Teng 2009, Jax *et al.* 2013). Some authors argue that the ecosystem services approach can help identify and appreciate the value of previously overlooked ecosystem functions and species groups and so work in favor of environmental protection and conservation (e.g., Thompson *et al.* 2011, Saastamoinen *et al.* 2014, Mori *et al.* 2017). The concept of ecosystem services has so far been used in the context of Finnish forestry to a comparatively limited extent (I). Therefore, the usefulness of the approach in this context cannot yet be determined. I note, however, that many if not all of the goods and functions classified as ecosystem services in the MEA, in CICES, and even in this thesis are no different from ones that have been written about for decades or even centuries (Saastamoinen *et al.* 2014), and yet, forest degradation has continued. This suggests that if linking ecosystems and their functions to human well-being is sensible to begin with (for a critical view see, e.g., Batavia and Nelson 2016), the ecosystem services approach may have to move beyond conventional ideas to make use of this potential in the context of Finnish forests. A part of this may be the explicit acknowledgement and inclusion of all of the different types of values ecosystem services have, such as relational values – the appropriateness and meaningfulness of how we relate to and interact with others (Chan *et al.* 2016).

In this thesis, the potential supply of ecosystem services from Finnish forests under alternative scenarios was evaluated. Losses in non-timber ecosystem services were shown to be more or less inevitable under intensive forestry, bringing into question the sustainability of forest exploitation. The case of Finnish production forests is representative of two broader contexts: the vast boreal region, and managed production ecosystems in general. Across the boreal region, extensive tracts of forests are under forestry use with impacts on ecosystem functions and biodiversity causing concern (Bradshaw *et al.* 2009, Moen *et al.* 2014, Gauthier *et al.* 2015). Boreal forests are comparatively similar in their ecology and management, suggesting that effects parallel to those found in this thesis may take place throughout the region. In terms of production ecosystems, Finnish production forests exemplify a system where environmental degradation has to be weighed against the procurement of an in-demand material resource. In this thesis, the values of ecosystem services to humans or the losses in human well-being resulting from declines in the services were not evaluated. It is thus not possible to draw definitive conclusions about the significance of these losses in terms of the sustainability or, in particular, the social acceptability of forest exploitation. The methods used in this thesis to identify trade-offs between forest management objectives and to reconcile them as effectively as possible could be adopted in real-world forest management planning. It is, however, the consequences of forest ecosystem change to human well-being and their acceptability and reflection in decision-making processes that ultimately determine what kinds of management strategies are actually adopted. The links among forest ecosystem change, human well-being, and decision-making should all be explored further in order to deepen our understanding of the controversial sustainability of forest exploitation and, ultimately, to improve it.

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YHTEENVETO (RÉSUMÉ IN FINNISH)

Metsätalouden, ekosysteemipalveluiden ja luonnon monimuotoisuuden väliset ristiriidat boreaalisissa metsissä

Suomalaiset metsät ovat moninaisten hyötyjen lähde: niitä käytetään puuntuotantoon, niistä kerätään marjoja ja muita keruutuotteita, ja niissä ulkoillaan ja virkistytään. Lisäksi metsät puhdistavat ilmaa ja vettä, suojelevat maaperää sekä varastoivat hiiltä ja siten ehkäisevät ilmastonmuutosta. Metsien ihmisille tuottamia hyötyjä voidaan kutsua ekosysteemipalveluiksi. Ekosysteemipalvelut kattavat erilaiset luonnosta korjatut tuotteet, luontoon liitetyt kulttuuriset merkitykset sekä luonnon prosessit, jotka ylläpitävät ihmiselle suotuisia ympäristöoloja. Lukuisten ihmiselle tärkeiden ekosysteemipalveluiden lisäksi metsät ovat merkittäviä suomalaisen luonnon monimuotoisuudelle. Metsät pitävät sisällään suuren kirjon erilaisia elinympäristöjä, joissa elää lukemattomia eliölajeja. Monet suomalaisten metsien elinympäristöistä ja lajeista ovat kuitenkin uhanalaisia tai vaarantuneita. Suurimpana syynä tähän pidetään metsätaloutta ja sen aiheuttamaa elinympäristöjen häviämistä.

Valtaosa suomalaisista metsistä on metsätalouden piirissä. Talousmetsiä käsitellään aktiivisesti ja niistä korjataan puuta. Metsänkäsittely ja puunkorjuu muokkaavat metsän rakennetta ja toimintaa erilaiseksi kuin luonnontilaisissa metsissä. Vaikuttamalla puulajistoon, puuston rakenteeseen ja ikään, maaperään sekä metsämaiseman rakenteeseen metsätalous vaikuttaa myös metsien sopivuuteen erilaisten lajien elinympäristöksi sekä metsäekosysteemien toimintoihin. Eliölajit, ekosysteemin elottomat rakenteet sekä niiden väliset vuorovaikutukset ovat ekosysteemipalveluiden perusta. Talousmetsien metsänhoidon ensisijainen tavoite on tuottaa mahdollisimman paljon puuta. Vaikuttamalla metsien lajistoon, rakenteisiin ja toimintoihin metsänhoito voi kuitenkin vaikuttaa myös muiden ekosysteemipalveluiden tasoon. Kestävän metsänhoidon tulisi turvata puuntuotannon lisäksi muiden metsien ekosysteemipalveluiden korkea taso sekä luonnon monimuotoisuuden säilyminen. Väitöskirjassani tutkin näiden metsänhoidon tavoitteiden – puuntuotannon, muiden metsien ekosysteemipalveluiden sekä metsäluonnonsuojelun – välisiä ristiriitoja.

Tutkimuksessa käytettiin laajaa metsävarojen inventaarioaineistoa, metsän kasvun simulaatioita ja monitavoitteista optimointia. Tutkimusalueet edustivat tyypillisiä suomalaisia talousmetsiä ja kattoivat tuhansia metsikkökuvioita. Metsikkökuvio on puustoltaan verraten yhtenäinen alue, joka on määritelty metsän käsittely-yksiköksi. Metsävarojen inventaarioaineisto sisältää tiedon metsikön ominaisuuksista ja tämänhetkisestä puustosta, kuten kasvupaikkatyypin, pääpuulajin, puuston keskimääräisen iän ja eri puuositteiden tilavuuden. Inventaarioaineistoa voidaan käyttää ohjelmistoissa, jotka simuloivat metsikön tulevaa kasvua ja kehitystä ottaen huomioon siinä toteutetut metsänhoidolliset toimenpiteet. Yhdistämällä tulevaisuuden simulaatiot monitavoitteiseen optimointiin voidaan arvioida vaihtoehtoisten metsänkäsittelyjen ja niiden yhdistelmien kykyä tasapainottaa keskenään vastakkaisia metsänhoidon tavoit-

teita. Menetelmällä voidaan esimerkiksi paljastaa, miten hyvin muut tavoitteet voivat toteutua, kun yksi tavoite maksimoidaan, tai miten lähellä maksimaalisia tasojaan kaikki tavoitteet voivat olla samanaikaisesti.

Tutkimusta varten toteutetuissa metsän kasvun simulaatioissa oli mukana valikoima vaihtoehtoisia metsänkäsittelyjä suositusten mukaisesta tasaikäisrakenteisen metsän kasvatuksesta metsän jatkuvaan kasvatukseen ja pysyvään suojeluun. Nämä vaihtoehtoiset metsänkäsittelyt tuottavat rakenteeltaan erilaisia metsiköitä, jotka ylläpitävät eri ekosysteemipalveluita vaihtelevasti. Usean metsikön muodostaman maiseman tuottamat ekosysteemipalvelut voidaan pyrkiä maksimoimaan optimoimalla eri metsänkäsittelyjen toteuttamista erilaisilla metsikkökuvioilla eli painottamalla yksittäisiä tavoitteita niille sopivimmilla kuvioilla. Tutkimuksessa oli mukana kuusi ekosysteemipalvelua, joita voidaan mitata metsikön rakenteen perusteella: puuntuotanto, hiilen varastointi, tuholaisten säätely, mustikan tuotanto, puolukan tuotanto sekä maiseman esteettisyys. Ekosysteemipalveluiden lisäksi metsäluonnonsuojelun tavoitetta edusti metsässä olevan lahopuun määrä. Tutkimuksessa arvioitiin näiden tavoitteiden välisten ristiriitojen vakavuutta sekä metsänhoidon suunnittelun vaikuttavuutta ristiriitaisten tavoitteiden yhteensovittamisessa. Lisäksi arvioitiin, miten pysyviä intensiivisen metsätalouden vaikutukset ekosysteemipalveluihin ovat.

Puuntuotannon ja muiden ekosysteemipalveluiden sekä puuntuotannon ja luonnonsuojelun välillä havaittiin selviä ristiriitoja. Intensiivinen metsätalous heikensi useiden ekosysteemipalveluiden tasoa, kun taas puunkorjuun vähentäminen tai lopettaminen ylläpiti monia ekosysteemipalveluita verraten hyvin. Intensiivisen metsätalouden haitallisten vaikutusten havaittiin myös olevan pitkäkestoisia ja jopa kasautuvia: mitä pidempään intensiivistä metsätaloutta jatkettiin, sitä hitaammin muut ekosysteemipalvelut toipuivat sen päätyttyä. Optimoimalla erilaisten metsänkäsittelyjen jakautumista niin, että eri ekosysteemipalveluita painotettiin niiden tuotantoon sopivimmilla metsikkökuvioilla, löydettiin keskenään vastakkaisia tavoitteita tasapainottavia kompromissiratkaisuja. Monet ekosysteemipalvelut sekä lahopuun määrä jäivät kuitenkin kauas enimmäistasoistaan, kun puuntuotanto oli yksi optimoinnin tavoitteista. Menetelmän kyky löytää hyviä kompromisseja oli myös riippuvainen siitä, miten suuren metsäalueen yli metsänkäsittelyjen yhdistelmää optimoitiin.

Ihmiselle tärkeiden ekosysteemipalveluiden turvaaminen sekä luonnon monimuotoisuuden säilyttäminen ovat metsien kestävän hoidon ja käytön julkilausuttuja edellytyksiä. Väitöskirjani tulosten perusteella näiden tavoitteiden toteutuminen samanaikaisesti metsien intensiivisen hyödyntämisen kanssa on erittäin epävarmaa. Metsien kasvun simulaatioita ja monitavoitteista optimointia voidaan hyödyntää käytännön metsänhoidon suunnittelussa metsien käytön haitallisten vaikutusten minimoimiseksi. Tulokseni viittaavat kuitenkin siihen, että menetykset joissakin tavoitteissa ovat väistämättömiä. Se, miten metsiämme hoidetaan ja hyödynnetään, riippuu lopulta siitä, mitkä menetykset katsotaan saavutettujen hyötyjen arvoisiksi.

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ORIGINAL PAPERS

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IMPACTS OF FORESTRY ON BOREAL FORESTS: AN ECOSYSTEM SERVICES PERSPECTIVE

by

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REVIEW

Impacts of forestry on boreal forests: An ecosystem services perspective

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Abstract Forests are widely recognized as major providers of ecosystem services, including timber, other forest products, recreation, regulation of water, soil and air quality, and climate change mitigation. Extensive tracts of boreal forests are actively managed for timber production, but actions aimed at increasing timber yields also affect other forest functions and services. Here, we present an overview of the environmental impacts of forest management from the perspective of ecosystem services. We show how prevailing forestry practices may have substantial but diverse effects on the various ecosystem services provided by boreal forests. Several aspects of these processes remain poorly known and warrant a greater role in future studies, including the role of community structure. Conflicts among different interests related to boreal forests are most likely to occur, but the concept of ecosystem services may provide a useful framework for identifying and resolving these conflicts.

Keywords Conflict · Forest management · Sustainability · Timber production · Trade-off

INTRODUCTION

Boreal forests account for approximately one-third of the world's forest cover (UNEP et al. 2009). These forests are a major source of timber products, but also provide a range of other goods and services that are essential to human well-being (Vanhanen et al. 2012; Brandt et al. 2013; Gauthier et al. 2015). In general, the multifunctional role of forests is widely recognized within science (Harrison et al. 2010) and policy (e.g., the EU Forestry Strategy[1]). Boreal forests have a crucial role in global climate regulation and climate change mitigation (Pan et al. 2011). They also harbor unique biodiversity, and the biome includes some of the world's largest areas of intact primary forest (UNEP et al. 2009). Therefore, the development of boreal forests in the coming decades is of great importance for both humans and global biodiversity.

Unlike tropical and temperate forests, boreal forests as a whole have remained relatively stable in area in recent decades (UNEP et al. 2009; FAO 2015). In several boreal countries, forest conversion is discouraged by regulatory measures, and overall, the region is characterized by a net gain in growing forest stock (FAO 2015). However, extensive tracts of boreal forests are actively managed and harvested for timber production, with changes to the structure of the forests and impacts on wildlife and ecosystem functioning (Bradshaw et al. 2009; Kuuluvainen et al. 2012; Venier et al. 2014). Throughout the boreal region, even though intact forests are concentrated in the northernmost or otherwise inaccessible regions, still they are not extensively protected (Potapov et al. 2008). Moreover, there is ongoing pressure to harvest more forest biomass, for example, to increase the use of renewable energy according to set targets. The suggested ways of intensifying forest biomass production to achieve this (e.g., fertilization, tree species choice, and whole-tree harvesting) may further aggravate forestry's impacts on ecosystems (Laudon et al. 2011).

The concept of ecosystem services (Millennium Ecosystem Assessment 2005) provides a framework for describing the multifunctional role of ecosystems, for

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[1] http://ec.europa.eu/agriculture/forest/strategy/index_en.htm.

assessing the impacts of ecosystem management comprehensively, and for planning management strategies that balance conflicting interests. Ecosystem services are defined as the benefits human populations obtain directly or indirectly from the ecosystem structures and functions (Costanza et al. 1997; Millennium Ecosystem Assessment 2005). Besides timber, boreal production forests are actively used as a source of collectable goods and recreation, and provide a range of other ecosystem services, including climate regulation, water purification, maintenance of soil productivity, and air-quality regulation (Vanhanen et al. 2012; Brandt et al. 2013). The widespread acknowledgement of forests as major providers of ecosystem services is illustrated by the common use of forest cover as an indicator of several ecosystem services (e.g., Maes et al. 2016) or assignment of high values of service supply to forests compared with other land cover types (e.g., Vihervaara et al. 2010). However, recent work has emphasized the theoretical and practical importance of the relationships among ecosystem services, which may range from synergistic via neutral to conflicting and change in response to management (Bennett et al. 2009; Carpenter et al. 2009; Raudsepp-Hearne et al. 2010). In particular, trade-offs between provisioning and other services have been suggested to be common and driven by management that aims to maximize production (Millennium Ecosystem Assessment 2005). The main goal of forest management in commercial forestry in the boreal zone is typically to maximize timber production, as timber is the only or the primary source of revenue from the forest to the landowner. However, if management focuses disproportionately on this productive function, other important benefits may be degraded or lost.

Boreal countries are committed to sustainable management of forests and to the preservation of forest services, e.g., through the EU Forestry Strategy, the Montréal Process,[2] and the Convention on Biological Diversity.[3] The long history of forestry in boreal countries means that there are well-established systems of and accrued expertise in forest management, which may be seen as an opportunity for the development and implementation of management practices that promote diverse benefits and biodiversity (Moen et al. 2014). However, debate on the most beneficial forest management methods is ongoing, and important information is still lacking (Kuuluvainen et al. 2012). The forest models and indicators of sustainable forest management that are currently used as management and policy tools describe several forest ecosystem services insufficiently (MCPFE 2002; Mäkelä et al. 2012). Yet, forest structure, function, and biodiversity, which are all modified by forest management, are linked to the total supply of ecosystem services (Thompson et al. 2011). It is clear that the effects of forest management may extend to the level of multiple goods and services provided by the system, and because of the extent of forestry in boreal countries, the preservation of forest ecosystem services is dependent on production forests (Kuuluvainen 2009; Mönkkönen et al. 2011).

There is an abundance of empirical research on the effects of boreal forestry on certain ecosystem functions and properties, such as hydrology and soil conditions (Kreutzweiser et al. 2008), disturbance dynamics (Kuuluvainen 2009), stand structure (Brassard and Chen 2006), and certain species groups (e.g., Niemelä 1997). However, a comprehensive overview of the implications of these effects in terms of ecosystem services has to our knowledge not been performed. This is contrary to, for example, the environmental impacts of tropical forestry (e.g., Edwards et al. 2014) or agriculture (e.g., Power 2010).

In this paper, we review and synthesize our current knowledge on the environmental and social impacts of boreal forestry by applying the ecosystem services framework. The aims of this paper are (1) to investigate the previous use and potential applicability of the ecosystem services framework in this context, (2) to review the impacts intensive forestry may have by assembling literature on a range of well-acknowledged forest ecosystem services, and (3) to identify the ecosystem services and the aspects of the forestry–ecosystem services relationship that are still poorly known. As this is a wide range of issues and the space here is limited, our goal is to provide an overview of boreal forestry's potential effects on ecosystem services, rather than to survey the entire literature for quantitative estimates of the overall magnitude of these effects.

We first briefly discuss how the environmental impacts of boreal production forestry may be fitted into the ecosystem services framework and assess how widely the framework has been used in this context. Next, we describe the links between common forest management practices and a range of ecosystem services. Following the classification of the Millennium Ecosystem Assessment (2005), we present examples of forestry's impacts on regulating services (climate change mitigation, maintenance of soil productivity and water quality, resistance to natural hazards, and pollination), provisioning services (non-timber forest products), and cultural services (recreation, landscape aesthetics, and sociocultural values). We note that the environmental impacts of forestry include various effects generated during the entire life cycle of forest products, but here we focus on changes to the structure and functioning of the forest ecosystem that may, in turn, affect the supply of ecosystem services from the forest. We also

[2] http://www.montrealprocess.org/.

[3] http://www.cbd.int/.

note that biodiversity is sometimes considered an ecosystem service in itself, for example, with cultural value (Mace et al. 2012). Here, we consider biodiversity as a quality of the ecosystem, which contributes—often fundamentally—to ecosystem functioning and provision of ecosystem services (Cardinale et al. 2012; Harrison et al. 2014). Finally, we discuss the emerging patterns and the potential contribution of the ecosystem service framework with respect to sustainable forest management as well as recommendations for future research efforts.

BOREAL FORESTRY IN THE ECOSYSTEM SERVICES FRAMEWORK

Introduction to boreal production forestry

The circumpolar boreal zone is the most northerly of the world's major terrestrial biomes, encompassing about 1.890 billion ha of land mainly located within Russia, North America, and Fennoscandia (Brandt et al. 2013). The boreal zone is characterized by forests, which throughout the zone share several environmental characteristics and similar taxa. However, there is some variation within the region in the management history and current state of the forests: in Fennoscandia, boreal forests have been harvested for longer and more intensively than forests in North America and Siberia (Ruckstuhl et al. 2008; Elbakidze et al. 2013), and there is considerably less primary forest left in northern European countries than in the rest of the boreal zone (Table 1). In Canada, much of the timber harvesting is currently done in primary forests (Conference Board of Canada 2013). In northern Europe, most forests are privately owned, whereas in Canada and Russia, most forests are owned by the state or other communities (Brandt et al. 2013; Elbakidze et al. 2013).

The predominant means of timber production in boreal forests is based on clear-cut harvesting of even-aged stands. After a clear-cut, the stand is regenerated either naturally or artificially by seeding or planting. Before regeneration, the site is often prepared mechanically or by prescribed burning to ensure the establishment of a new stand. Under intensive management, regeneration may be followed by pruning and thinning of the developing stand to promote tree growth, and growth conditions may be improved by fertilization. The time of the final harvest may be determined by a planned schedule or a desired timber stock, and may aim at optimal cutting at the stand's maximal growth or at efficiency of operations over a larger area. Harvest residues and stumps may also be collected. Dead or living retention trees may be left in the logged area to promote biodiversity and soil nutrients. Forestry planning thus comprises the selection of silvicultural treatments applied to the site as well as the size, timing, and arrangement of harvests across the landscape. It is influenced by the conditions of the stands, including their accessibility, and the aims of the forest manager. In general, due to factors like management history and ownership structure, forest management in northern Europe is characterized by intensive management of relatively small stands, and in North America and Russia by extensive harvesting of larger areas (Gauthier et al. 2015). Besides clear-cutting regimes, alternative forest management systems such as those based on selection harvesting are used to a lesser extent, but interest in these systems is growing due to environmental and social concerns related to even-aged forestry (e.g., Kuuluvainen et al. 2012).

Table 1 Forest statistics of boreal countries (data from FAO 2015). It should be noted that these country-level statistics may include other forest types besides boreal forest

	Forest area (1 000 000 ha)	Forest of land area (%)	Primary forest (% of forest area)	Forest within protected areas (% of forest area)
Finland	22.2	73.1	1.0	17.7
Norway	12.1	39.8	1.3	4.8
Russia	814.9	49.8	33.5	2.2
Sweden	28.1	68.4	8.6	7.1
Canada	347.1	38.2	59.3	6.9

Delivery of forest ecosystem services

Understanding the effects of human activities on ecosystem services requires knowledge of the ecosystem processes producing the services as well as methods to quantitatively assess the state of the service supply. In general, the delivery of ecosystem services may be described as a process originating in the interactions among living organisms and their environment, leading to relevant ecosystem structures and functions, and ending with the benefits and values experienced by humans. This conceptualization is referred to as the cascade model (Haines-Young and Potschin 2010). In reality, the processes described by the model are not linear, and the stages defined in it are interconnected; however, it provides a typology for analyzing the links between ecosystem properties and human well-being in a systematic way (Haines-Young and Potschin 2010). As described above, intensive forestry comprises several management actions applied to forests throughout a rotation. These alter the biotic and abiotic structures of the forest ecosystem with potential impacts cascading through species communities, ecosystem functions, and the benefits obtained by humans. The effects of forest management on ecosystem services may thus also be comprehensively depicted and analyzed in terms of the cascade model (Fig. 1).

Indicators of ecosystem services may be defined based on any of the stages of ecosystem service generation, as enabled by the understanding of the phenomena or availability of data (Fig. 1) (e.g., Mononen et al. 2016). It may be recommendable to develop and use indicators that describe the state of ecosystem service supply at every step of their generation, because this can provide a more balanced and reliable view of the phenomenon than a single indicator, especially for monitoring and impact assessment purposes (Mononen et al. 2016). Quantification of the losses or gains in ecosystem services caused by forest management requires indicators that are intricate enough to capture the variation created by management at the different stages of ecosystem service delivery. Ideally, the effects of forest management should also be monitored or modeled over several decades or entire stand rotations, because a forest provides different ecosystem services depending on its age and structure (Schwenk et al. 2012; Zanchi et al. 2014).

Ecosystem services and boreal forestry in existing literature

The environmental impacts of boreal forestry have long been a subject of research and there are large amounts of published literature on some of these impacts, also with respect to the implications to human benefits (e.g., Webster et al. 2015; Roberge et al. 2016). In order to produce estimates of how widely boreal forestry's impacts on different ecosystem services have been studied, we conducted the literature searches in the ISI Web of Science database using search terms related to boreal forestry and different ecosystem services and recorded the numbers of results returned by each search (see Online Appendix S1 for the full list of search terms and further details). We then filtered these search results with the additional search term "ecosystem service*" to estimate how widely the concept of ecosystem services has been used in this field. The results of these simple searches indicate that there is great variation in the amount of existing literature among the different ecosystem services (Fig. 2). The numbers of articles related to maintenance of soil productivity, regulation of water flow and quality, and climate regulation are manifold compared with, for example, resistance to natural hazards, pollination, or provision of non-timber forest products. In addition, by filtering this literature with the search term "ecosystem service*", it becomes apparent that the use of the ecosystem service terminology has so far been marginal in this context (Fig. 2). This finding is supported by the extensive review by Abson et al. (2014), who reported ecosystem service literature from forest ecosystems to be focused on tropical forests. Few, model-based studies have examined the effects of boreal forest management on ecosystem services (Miina et al. 2010; Zanchi et al. 2014; Triviño et al. 2017), but the set of

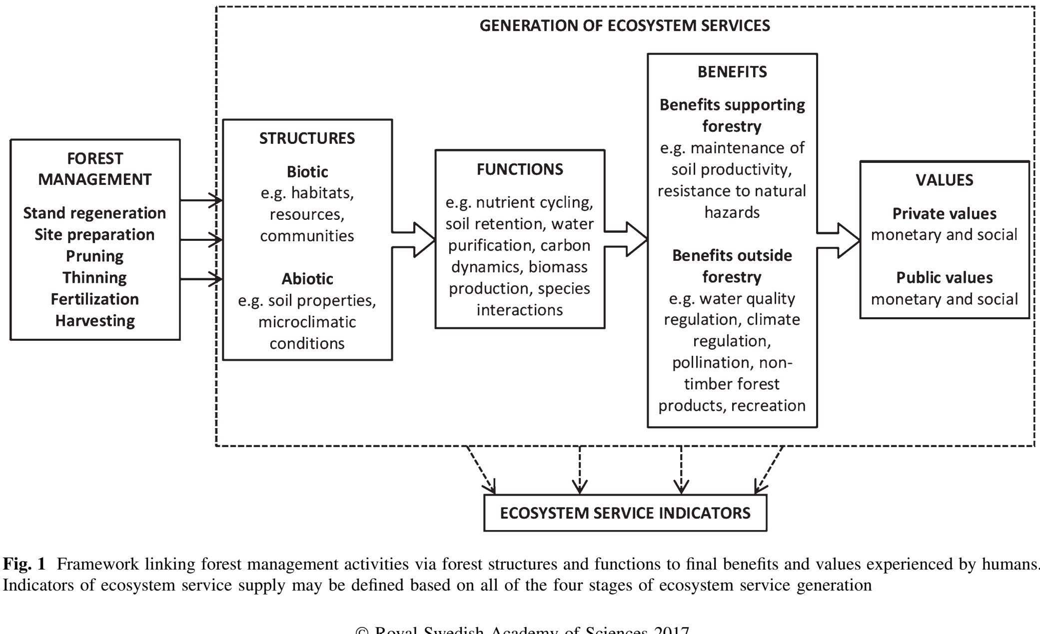

Fig. 1 Framework linking forest management activities via forest structures and functions to final benefits and values experienced by humans. Indicators of ecosystem service supply may be defined based on all of the four stages of ecosystem service generation

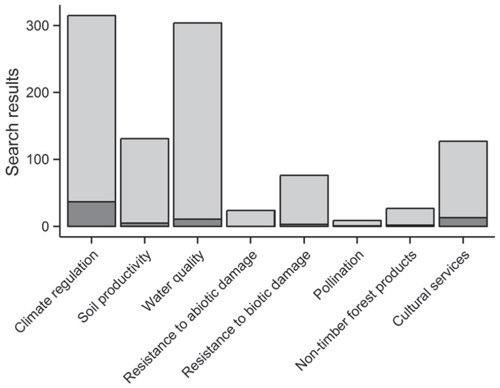

Fig. 2 Numbers of results returned by the literature searches using search strings related to boreal forestry and different phenomena associated with specific ecosystem services. Each ecosystem service had its own predefined set of search terms. The dark grey part of each bar shows the portion of the search results returned when the additional search term "ecosystem service*" was used. A detailed description of the literature searches, including a full list of search terms, is given in Online Appendix S1

ecosystem services included also in these studies is limited compared with the wide range of benefits that boreal forests provide. It is clear that the existing literature, particularly literature building on the ecosystem services framework, does not yet cover the full range of boreal forestry's potential consequences for human benefits.

EFFECTS OF FOREST MANAGEMENT ON ECOSYSTEM SERVICES

Regulating services

The role of boreal forests in climate regulation is one of their most widely studied functions. "The second lung of the planet" (Warkentin and Bradshaw 2012), boreal forest, contributes greatly to global air-quality and climate regulation. Carbon storage and sequestration by boreal forests is hugely important for global climate change mitigation (Pan et al. 2011), but the effects of forestry on these functions are complex. Forestry has a negative impact on climate change mitigation if it decreases the system's ability to fix carbon or if it results in releases of carbon into the atmosphere from long-term storages in the forest ecosystem, for example via disturbances to soils where most of the carbon resides (Jandl et al. 2007; Bradshaw and Warkentin 2015). Conversely, human interference may safeguard carbon storage, e.g., by preventing forest fires (Kurz et al. 2008), and forest management may increase carbon sequestration, e.g., by promoting tree growth via tree species choice or fertilization (Hyvönen et al. 2007). Whether production forests act as carbon sources or sinks may critically depend on the fate of the carbon fixed in harvested wood products (Liski et al. 2001). Moreover, forests contribute to climate regulation in other ways besides carbon dynamics, such as surface albedo (Lutz and Howarth 2014) and production of aerosols that contribute to cloud formation (Spracklen et al. 2008). The total effect of forest management on climate regulation is thus a result of several complex processes, many of which remain poorly understood.

At local scales, some of the most important ecosystem services from forests are related to water and soil quality. As shown above, these are also some of the most widely studied forest functions. Forest vegetation retains water, nutrients, and soil, both maintaining the productivity of the soil and regulating the quality of adjacent waters. In terms of nutrient cycling, undisturbed boreal forests are a comparatively closed system, and naturally occurring nutrient leaching from boreal forests is relatively low (Mattsson et al. 2003; Maynard et al. 2014). Forestry activities have direct impacts on soil physical properties and decomposer communities, alter the conditions in the forest, and disturb the nutrient cycling processes, and may thus change the ability of the forest to maintain soil productivity (Grigal 2000; Kreutzweiser et al. 2008; Hartmann et al. 2012). Harvesting, fertilization, and soil preparation activities typically increase nutrient availability and loss by leaching (Mattsson et al. 2003; Kreutzweiser et al. 2008), and road construction and use of heavy machinery may increase erosion and reduce the productivity of the site (Grigal 2000). In addition, nutrients are lost from managed forests in harvested biomass, with the amount of nutrients lost depending on harvesting intensity. Nutrient losses caused by biomass removal and increased leaching are variable, but have in many cases been estimated to be small in effect, and boreal forest soils appear to recover from them relatively rapidly (Kreutzweiser et al. 2008). However, forestry operations also have effects on soils that are not yet fully understood, such as changes in the composition of soil communities. These changes may, in fact, be more persistent than changes in soil nutrient pools, but their functional implications remain to be determined (Hartmann et al. 2012).

The consequences of reduced nutrient retention capacity in managed forests may be greater for water quality than those for soil fertility (Kreutzweiser et al. 2008; Webster et al. 2015). Nutrient and organic matter loads from forestry contribute to water eutrophication and increased turbidity, and some forestry operations may increase the transport of toxic compounds like methyl mercury into surface waters (Webster et al. 2015). Out of all silvicultural operations, clear-cut harvesting combined with mechanical site preparation is considered to have the strongest effect on runoff water quantity and quality, but the magnitude of the effect is heavily site dependent (Kreutzweiser et al. 2008). Because forests can retain nutrients arriving from upstream sources, leaving unfelled forests as buffers between waters and clear-cuts can be an effective way to mitigate the effects of forestry on water quality (Gundersen et al. 2010), although their effectiveness may depend on factors like the intensity of harvesting and the exact configuration of hydrologic pathways (Webster et al. 2015). Indeed, forests act as water-quality regulators most importantly when they are adjacent to waters and can act as buffer zones, or when they grow on nutrient-rich sites where the potential for nutrient leaching is high. In these sites, activities that reduce the forest's nutrient-retention ability may cause the most substantial losses in the service of water-quality regulation.

Besides regulation of climate, water, and soils, forest ecosystems perform functions that regulate the occurrence of natural disturbances. Natural disturbances to forests are biotic (pests and pathogens) and abiotic (fire, wind, floods) hazards that severely alter forest structure and function (Jactel et al. 2009). Resistance to natural disturbances and mitigation of their effects may be considered as ecosystem services that protect the timber stock. By regulating stand structure, tree age distribution, species composition, and tree growth, forest management may significantly alter the forest's susceptibility to both biotic and abiotic hazards (Schelhaas et al. 2003; Jactel et al. 2009). For instance, resistance to wind damage may be reinforced by planning stand rotations to smooth out height ratios among neighboring forest stands (Zeng et al. 2009), and by planning clear-cut size, placement, and density over the landscape to reduce the total length of stand edges (Zeng et al. 2010). Biotic hazards may be mitigated by minimizing the availability of alternative food and breeding resources of pest species (Jactel et al. 2009). Natural resistance to pests and pathogens may also be increased by managing stand composition to create natural barriers or by providing resources for natural control agents (Jactel et al. 2009). Increased stand diversity is often presented as a way to promote stand resistance to pests, but such effects in boreal forests have been questioned due to lack of empirical evidence (Koricheva et al. 2006).

Overall, production forests offer habitats for a range of beneficial organisms that provide important regulating services, such as natural enemies of pests, pollinators, and decomposers. These are the forest ecosystem services that seem to be the least studied and the most poorly understood, especially with respect to their responses to forest management. For example, it is suggested that predators such as three-toed woodpeckers (*Picoides tridactylus*) may contribute to stabilizing the population dynamics of forest pests, but their ability to do so depends on complex multiscale interactions that are not fully understood (Fayt et al. 2005). Pollinators inhabiting production forests contribute to the production of forest berries and to crop production in adjacent agricultural areas. In Finland, for example, pollination of several agricultural crops and forest berries is heavily dependent on bumblebees, and, despite extensive forestry, the Finnish forest-inhabiting bumblebee species populations are estimated stable or increasing (Paukkunen et al. 2007). However, lack of natural disturbances and the nesting resources that disturbances create has been also suggested to negatively affect pollinators (Rodríguez and Kouki 2015). Taki et al. (2011) found forest management to reduce the habitat and resource quality of forests and, in turn, the presence and abundance of pollinators in adjacent areas in an agriculturally dominated landscape in Japan. However, these relationships seem not to have been studied in the boreal region. Information is thus lacking on the effects of forest management on local populations of pollinators as well as other beneficial organisms.

Provisioning and cultural services

Production forests are a source of several products besides timber, such as berries, mushrooms, and herbs, collectively termed non-timber forest products. These products may have great economic and cultural importance especially in Aboriginal and rural communities (Duchesne and Wetzel 2002). Several factors independent of forest management affect the abundance of non-timber forest products, such as site type, climate, and weather conditions (e.g., Miina et al. 2009; Turtiainen et al. 2013). However, several forest characteristics that are altered by management, such as tree species composition, canopy openness, understory vegetation, and soil structure, moisture, and nutrient status (discussed above), also affect the suitability of a site as a habitat for species, and thus the availability of related products for humans (Miina et al. 2009; Gamfeldt et al. 2013). These effects may be positive or negative; for example, clear-cut harvesting has been reported to increase (Nybakken et al. 2013) or decrease (Atlegrim and Sjöberg 1996) the abundance of bilberry (*Vaccinium myrtillus*), depending on the characteristics of the site. Naturally the direction of these effects depends also on the requirements of the focal species. When the non-timber forest products are from species that thrive in young stands or benefit from increased canopy openness, their production may be particularly compatible with production forestry (e.g., Clason et al. 2008).

Abundance of several non-timber forest products is a component of cultural ecosystem services because of the high recreational and cultural value of activities like berry picking. Forests also offer opportunities for several other recreational and educational activities such as hiking, camping, and wildlife observation (Vanhanen et al. 2012). Where there is public access to production forests (e.g., the so-called "everyman's right" in Finland, Norway, and Sweden), they may be traditionally highly valued as a source of recreation (Parviainen 2015). Landscapes viewed as attractive or natural also have recreational and cultural value as such (Millennium Ecosystem Assessment 2005). Cultural services are often considered to be some of the most challenging ecosystem services to measure, and even though they appear to be among the most widely studied within boreal forests (Fig. 2), this literature reflects the complexity of the matter. For example, the recreational and scenic value of forest landscapes depends on individual preferences that may be variable. However, a review of preference surveys from the northern Europe concluded that factors such as accessibility, naturalness, and biodiversity typically increase the experience of recreational and aesthetic value, whereas obvious signs of forestry operations reduce it (Gundersen and Frivold 2008).

Production forests also have sociocultural value to forest owners and other stakeholders that may be affected by management and policy. In Finland, for example, the top-down instituted 'scientific' forest management in the mid-20th century led to dissent from forest owners because it conflicted with their economic interests, experience of independence, and aesthetic and cultural values attached to their forests (Siiskonen 2007). Many aspects of Aboriginal cultures depend in distinctive ways on forests and access to diverse forest lands and resources (e.g., in Canada; Sherry et al. 2005). To address this, forest planning and management systems may be developed to better incorporate Aboriginal interests and traditions (e.g., Wyatt 2008; Asselin et al. 2015).

DISCUSSION

In boreal production forests, the main focus of management is usually to enhance timber production. Our review suggests that intensive production forestry may have substantial effects on numerous ecosystem services (Fig. 3), and that these effects may be harmful or beneficial (Table 2). As described by the cascade model (Haines-Young and Potschin 2010), these effects are the result of changes caused by forestry to forest structures and functions that underpin ecosystem services. The evaluation of these changes from the perspective of ecosystem services is an emerging research path that may provide valuable insights for sustainable forest management. In order to do so, it must aim at clarifying the numerous ecological processes involved in the forestry–ecosystem services relationship that are still poorly understood (Mori et al. 2016), as well as the social processes that influence forest management decisions, demand for non-timber forest benefits, and the valuation of these benefits (Sandström et al. 2011; Filyushkina et al. 2016).

Overall, the forest's capacity to provide ecosystem services appears to be typically weakened when forestry activities are intensive and disturbances to the natural state and functioning are acute and severe. The extent and intensity of harvesting and site preparation seem to be among the most important management choices, as these operations have major potential for deteriorating several services simultaneously (e.g., climate regulation, maintenance of soil productivity, regulation of water quality, storm damage resistance, and aesthetic values). In addition to the harvesting method, tree species selection, thinning intensity, and regeneration method fundamentally affect the structure of the forest with impacts on, for example, habitat suitability for pollinators, abundance of forest collectables, and recreational attractiveness. In some situations, forest management may enhance the supply of an ecosystem service compared with the natural state, e.g., by creating suitable habitat for desired organisms. Identifying the forestry practices that contribute the most to the deterioration of ecosystem services and the types of forest sites that are particularly vulnerable to them are important research avenues that can inform the development of management practices that support production forests' role as ecosystem service providers (cf. Sandström et al. 2011; Edwards et al. 2014; Filyushkina et al. 2016). In order to secure diverse ecosystem services from forests, the suitability of management options to different stands and landscapes should be evaluated using broad criteria and long-term impact assessments (Laudon et al. 2011; Schwenk et al. 2012; Mönkkönen et al. 2014; Asselin et al. 2015).

Among the ecosystem services we reviewed, the least well understood with respect to forestry's potential impacts on them are the maintenance of soil productivity by soil communities, natural pest control, and pollination (Fig. 2). The existing literature on the impacts of boreal forestry on ecosystem services and related ecosystem functions is dominated by biophysical processes such as soil conditions, hydrology, and carbon storage and sequestration. Despite calls for research that would shed light on the ecological basis of ecosystem services (e.g., Kremen 2005), substantial knowledge gaps remain about the role of community structure in ecosystem functions and the provision of ecosystem services in forests (Mori et al. 2016). As a consequence, even though the negative impacts of boreal forestry on biodiversity are established for several species groups (e.g., Niemelä 1997; Venier et al. 2014), the implications of this biodiversity loss for the supply of forest ecosystem services are still poorly understood. The

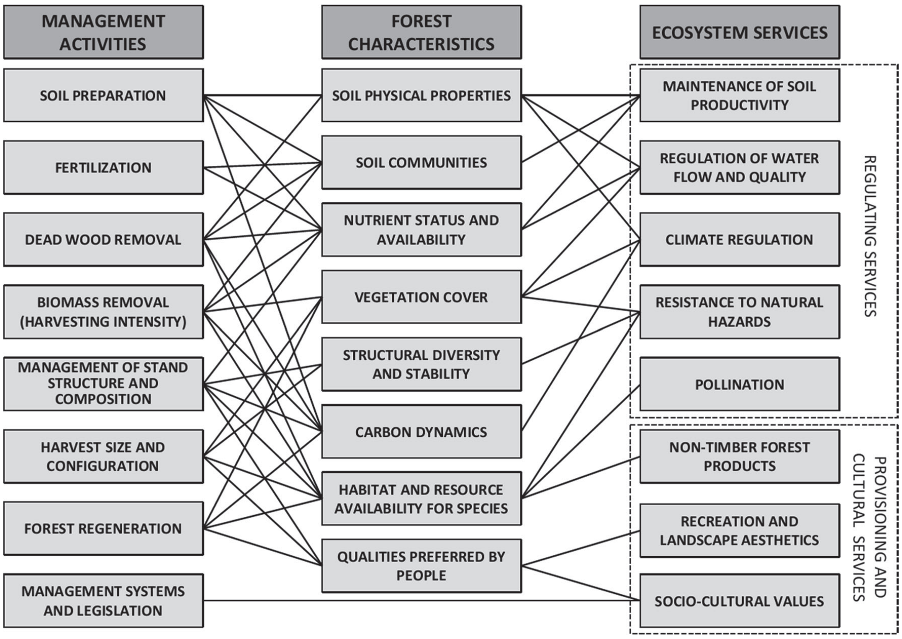

Fig. 3 Summary of some of the main connections between forest management activities, forest characteristics, and ecosystem services. *Lines connecting the boxes* in the columns show the impacts of management via forest characteristics on ecosystem services. These connections also show how identification and assessment of ecosystem services may guide management choices

Table 2 Changes in the supply of forest ecosystem services caused by production forestry as compared with undisturbed forest based on an overview of existing literature. Downward arrows indicate negative changes and upward arrows positive changes

Ecosystem service	Reported impacts
Maintenance of soil productivity	↑, ↓
Regulation of water flow and quality	↓
Climate regulation	↑, ↓
Resistance to biotic hazards	↓
Resistance to abiotic hazards	↓
Pollination	↑
Non-timber forest products	↑, ↓
Cultural services	↑, ↓

links between the diversity of forest communities and the maintenance of ecosystem services should be a major focus of future work (Thompson et al. 2011; Mori et al. 2016). Many ecosystem services are the product of complex ecological processes (as described by the cascade model), and it seems typical that forestry's effects on one or a few components of these processes are understood, but the overall effect on the final ecosystem service and the benefits and values derived by humans is not. This is the case even for the most widely studied ecosystem services, such as climate regulation (Landry and Ramankutty 2015). In addition, uncertainties remain about the long-term ability of the actively harvested and managed forests to provide also the widely studied ecosystem services, for example, regulation of water quality (Webster et al. 2015). Across various contexts, a good understanding of ecosystem service provision and its response to ecosystem change over different spatial and temporal scales is still lacking (Biggs et al. 2012; Mace et al. 2012).

Trade-offs between provisioning and other services are suggested to be frequent (Millennium Ecosystem Assessment 2005; Carpenter et al. 2009; Raudsepp-Hearne et al. 2010; Gamfeldt et al. 2013), and in production forests this situation is realized in the cases where activities intended to increase timber harvests cause other ecosystem services to deteriorate. These trade-offs may become more severe in the upcoming decades in response to the efforts to raise wood production to increasingly replace fossil fuels with forest energy and to sustain the demand for new wood-fiber based products and bio-materials. Whether this is achieved by subjecting more forest areas to harvesting, increasing forest productivity, or increasing the amount of biomass harvested, there are likely to be consequences in terms of the supply of forest ecosystem services. For example, increased biomass harvesting may lead to increasingly consequential nutrient losses from the system (Kreutzweiser et al. 2008), causing decreases in soil productivity and carbon sequestration capacity. With careful planning, however, it may be possible to design forest management to mitigate the trade-offs and promote the win–win situations among various objectives. This may require increased diversity in the adopted management regimes (e.g., Kuuluvainen et al. 2012) and care in the application of management activities to explicitly target multiple ecosystem services (e.g., Triviño et al. 2017).

Even though there is a long research tradition of linking forestry with ecosystem functioning, the terminology of ecosystem services has so far been used only marginally in the context of assessing the environmental impacts of boreal forestry. This is contrary to its common adoption in policy (e.g., the EU Forestry Strategy) and its rapidly growing use in other academic literature (Abson et al. 2014). The advantages and disadvantages of the concept are under ongoing debate (see e.g., Schröter et al. 2014). However, its widespread use suggests that at least some of its merits are widely accepted and that it is seen as policy relevant (e.g., Thompson et al. 2011). If the merits of the ecosystem services framework are accepted then its application in the context of boreal forestry is highly appropriate. It is based on a holistic socioecological system approach (Millennium Ecosystem Assessment 2005; Bennett et al. 2009; Carpenter et al. 2009), and may thus be well suited for analyzing the environmental impacts of forestry that are variable in direction, intensity, scale, and persistence. Boreal production forests are often associated with strong cultural values and identities by local people and play crucial roles in global biophysical processes. Therefore, evaluation of forestry's impacts on forest communities and ecosystem functions from the perspective of human benefits and values may be considered relevant in this context. Central to the ecosystem service approach is that it links ecosystem function and condition directly to the interests of different stakeholder groups and to political decision-making (Thompson et al. 2011), and may guide and promote conservation of taxa and ecosystems that may otherwise be overlooked (Mori et al. 2016). The concept has an inherent aim of advancing the sustainability of natural resource use (Millennium Ecosystem Assessment 2005). Thus, it is relevant with respect to developing sustainable forest management, which aims to reconcile multiple interests related to forests (Rametsteiner and Simula 2003; Mäkelä et al. 2012). Then again, it is worth noting that the ecosystem services approach is only one way to describe human–environment relationships and that additional or alternative formulations can be more advantageous, depending on the aims and the situation (Raymond et al. 2013). Researchers using the ecosystem service terminology should be aware of its implicit assumptions and the limitations that come with them (Raymond et al. 2013; Schröter et al. 2014).

An important issue that is beyond the scope of this work is the preservation of biodiversity in boreal forests. Biodiversity may co-occur with or fundamentally underlie ecosystem services, but these links are not guaranteed, especially for all services and all aspects of biodiversity (Mace et al. 2012; Harrison et al. 2014). In the upcoming decades, the management choices concerning boreal forests will likely have crucial implications to global efforts of biodiversity conservation (Moen et al. 2014). In order to secure preservation of boreal biodiversity and to meet international conservation targets, impacts on biodiversity must also be taken into account in the planning and evaluation of forest management strategies.

The multiple pressures facing boreal production forests are likely to intensify in the upcoming decades. In addition to production objectives and forest management choices, the future of forest ecosystem services depends on climate change and its effects. This myriad of intensifying, interconnected environmental and socioeconomic pressures facing boreal forests poses a great challenge to their management. The state of existing literature suggests that the framework of ecosystem services has so far been used in the context of boreal forestry to a very limited extent. However, it may be considered very applicable in this context because of the diverse benefits boreal forests provide globally as well as locally, and because, as this review shows, the supply of these benefits can be greatly affected by forest management actions. Major knowledge gaps remain regarding these processes, and we highlight especially the following research needs:

- The role of biodiversity and community structure in ecosystem functions and the generation of forest ecosystem services
- Impacts of biodiversity loss on the provision of forest ecosystem services
- Impacts of forestry on the long-term resilience of forest functions and the sustained supply of ecosystem services
- The drivers of demand for diverse forest ecosystem services
- Management strategies to balance conflicting demands and policy tools to implement them.

These issues mirror research needs identified by other authors (Moen et al. 2014; Filyushkina et al. 2016; Mori et al. 2016). By addressing these open questions, the ecosystem service approach may be a valuable tool in assessing the sustainability of forestry practices and in resolving conflicts between the various interests related to boreal forests.

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II

CONFLICTING OBJECTIVES IN PRODUCTION FORESTS POSE A CHALLENGE FOR FOREST MANAGEMENT

by

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Conflicting objectives in production forests pose a challenge for forest management

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ABSTRACT

Conflicts among different ecosystem services have been shown to be common and potentially exacerbated by management interventions. In order to improve the sustainability of natural resource use, the occurrence of these conflicts and the effects that management actions have on them need to be understood. We studied the conflicts between ecosystem services and the potential to solve them by management choices in boreal production forests. Our study area consisted of nearly 30,000 forest stands which were simulated for 50 years into the future under alternative management scenarios. The study included four ecosystem services – timber production, bilberry production, carbon storage, and pest regulation – and one biodiversity conservation objective defined as availability of deadwood resources. We 1) measured the conflicts among each pair of objectives, and 2) identified a compromise solution for each pairwise conflict defined as one which simultaneously minimizes the losses for both objectives. Our results show that conflicts between timber production and other objectives are typical, severe, and difficult to solve, while non-extractive benefits including biodiversity conservation can be more easily reconciled with each other. To mitigate the most severe conflicts in boreal forests, increased diversity in management regimes is required.

1. Introduction

Evaluating ecosystem services, or the diverse benefits people obtain from nature, may produce information that assists ecosystem managers in balancing the multiple, often conflicting, interests that people place on the environment (Albert et al., 2014; Bennett et al., 2009). Critical aspects of these evaluations include the co-occurrence of multiple ecosystem services, their interactions, and the impacts human activities have on their supply. The complexity of the relationships among different ecosystem services, aspects of biodiversity, and social objectives was recognized already in the Millennium Ecosystem Assessment (MEA, 2005), and considerable effort has since gone into conceptualizing and clarifying these processes (e.g. Kremen, 2005; Bennett et al., 2009).

A key first step in improving the sustainability of natural resource use is to identify patterns of trade-offs and synergies among ecosystem services and how they are driven by management interventions. A trade-off between ecosystem services occurs when the increased utilization of one service leads to a loss in another service, and they may take place at varying spatial and temporal scales (Rodríguez et al., 2006). The Millennium Ecosystem Assessment (MEA, 2005) established that ecosystem management to increase the supply of one ecosystem service may deteriorate the supply of other services, and that these negative trade-offs are particularly common between individual provisioning services and between provisioning and other types of ecosystem services (regulating, supporting, and cultural services). An extreme case is the conversion of natural ecosystems into managed monocultures, but also the extractive use of resources from a (semi-)natural ecosystem may, by altering the structures and functions of the ecosystem, cause more or less persistent changes in other ecosystem services.

Several recent studies have examined the relationships among ecosystem services and the effects of management on their supply in forests, where timber harvesting and other management activities cause changes in ecosystem structures and functions (e.g. Bradford and D'Amato, 2012; Edwards et al., 2014b; Brandt et al., 2014). Forests provide many important ecosystem services: they are a source of food and raw materials, provide recreational opportunities, hold cultural meanings, harbor a variety of beneficial organisms, regulate air, soil, and water quality, and play an important role in climate regulation. Even where forest loss is not a major threat, forests are affected by increasing pressures, such as

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a rising demand for forest biomass, the urgency to utilize forest ecosystems in climate change mitigation, and the need to safeguard biodiversity. Additionally, forests undergo natural disturbances that are expected to intensify in response to global change (Lindner et al., 2010; Seidl et al., 2016). These challenges create multiple objectives for forest management as well as a mounting need to resolve the conflicts among them (Bradford and D'Amato, 2012).

Boreal forests are extensively used for timber production, but are also a source of many locally and globally important ecosystem services. Earlier studies from boreal forests have shown that conflicts between timber production and other ecosystem services are common (e.g. Gamfeldt et al., 2013; Pohjanmies et al., 2017) and that stand management can affect trade-offs among forest services (Triviño et al., 2015; Zanchi et al., 2014). Specifically, maximizing timber harvests has been found to reduce forests' capacity to provide other services, while less intensive use of timber resources can lead to compromise solutions where intermediate levels of several objectives are maintained (Triviño et al., 2015; Zanchi et al., 2014). However, these impacts may be dependent on the ecosystem services in question and the properties of the forest (Biber et al., 2015). Moreover, few studies have examined the occurrence of conflicts among non-timber benefits from managed forests.

In this study, we study the occurrence and severity of conflicts between ecosystem services in a large production forest in Finland. Earlier studies in this landscape have shown that conventional, intensive forest management may cause severe trade-offs between timber production and biodiversity (Mönkkönen et al., 2014), climate regulation (Triviño et al., 2015), and forest collectables (Peura et al., 2016). Here, we measure the conflicts between timber production and non-timber forest benefits but also among non-timber benefits. We thus aim to resolve whether the most severe conflicts are those between a provisioning service (here, timber production) and other objectives, while non-extractive benefits including biodiversity conservation can be more easily reconciled with each other.

Earlier work conducted in our study area has also shown that considerable benefits in terms of biodiversity and ecosystem services can be gained by diversifying forest management regimes and optimizing their application across the landscape (Mönkkönen et al., 2014; Triviño et al., 2015). In these studies, forest management has been optimized at the scale of the entire landscape, recognizing the possibility that only some forest stands can produce high levels of several objectives simultaneously, while some can be disproportionally good for targeting a single objective. Optimal management across the landscape may thus be a combination of 'land-sharing' and 'land-sparing' strategies (e.g. Triviño et al., 2015), the former referring to a high supply of multiple ecosystem services from the same stand and the latter to prioritization of a single ecosystem service in a stand (e.g. Edwards et al., 2014a; Maskell et al., 2013). In our study, we focus on 'land-sharing' strategies and measure the severity of conflicts among pairs of objectives in each individual forest stand. We thus explore how achievable 'land-sharing' strategies are at the stand level. The achievability of good 'land-sharing' solutions at the stand level provides additional information on the severity of the pairwise conflicts and is important from a practical point of view. First, as a stand is the basic operational unit of practical forestry (Mäkelä and Pekkarinen, 2004), the stand level is the most relevant for forest managers. Second, management plans that allow for single-objective prioritization in parts of the target area may be misguided if demand for the objectives is not considered, that is, prioritization of an objective may be assigned to an area where there is no demand for it or *vice versa*. For example, while it may make little difference exactly where the benefits are generated in the case of some ecosystem services such as carbon storage, some ecosystem services may have very local demand (e.g. recreation, forest collectables, and some regulating services). Finally, minimizing trade-offs in every parcel of the landscape may help protect those objectives that are affected by the quality of neighboring stands; particularly, conservation of biodiversity that requires both patches of high-quality habitat and a relatively good-quality matrix (Kremen, 2015).

Our study includes five forest management objectives: four ecosystem services (timber production, bilberry production, carbon storage, and pest regulation) and one biodiversity conservation objective, defined as availability of deadwood resources. First, we measure the supply of each objective and the conflicts among all pairs of objectives under alternative forest management regimes. Second, we identify a compromise management solution for each pairwise conflict, defined as one which simultaneously minimizes the losses in both objectives. Finally, we examine the distributions of alternative forest management regimes among the compromise solutions and infer management recommendations for maintaining diverse benefits. Specifically, we address the following questions: 1) How strong are the conflicts between all pairs of objectives? 2) How efficiently can the pairwise conflicts be solved by optimizing management? 3) What kind of forest management may be required to secure high levels of multiple ecosystem services and biodiversity?

2. Materials and methods

2.1. Forest data and simulations

Our study area is a typical Finnish production forest landscape located in central Finland with forest covering the majority of the land and the rest consisting of a mosaic of lakes, peat lands, small settlements, and cultivated fields (Fig. 1). The total forest area is 431 km^2 and consists of nearly 30,000 individual stands. The stands are dominated by pine (*Pinus sylvestris*), spruce (*Picea abies*), birch (*Betula pendula* and *Betula pubescens*), or a mix of the four species. Most of the landscape has been under active forest management for several decades, and this is reflected in the current condition of the forest. Specifically, the age distribution of the stands is asymmetric with over 30% of the stands being younger than 20 years, over 60% younger than 50 years, and only about 5% older than 100 years.

In order to account for the long-term ability of the forest to provide ecosystem services, we simulated the development of the stands under different management regimes for 50 years into the future. The initial stand-level data was compiled from forest inventory data administered by the Finnish Forest Centre (Finnish Forest Centre, 2016) to include the variables needed for the simulations, e.g. basal area of trees, tree species composition, ages of tree cohorts, and site fertility. Forest growth simulations were implemented with the MOTTI stand simulator (Hynynen et al., 2002; Salminen et al., 2005). MOTTI predicts the development of a stand based on its initial characteristics and the forestry operations applied during the simulation. In MOTTI, a set of empirical-statistical models are integrated into software that predicts the growth and mortality of trees on the basis of the quality of the site, the growth potential of the tree and the competition effects imposed by other trees. We simulated each stand under seven alternative management regimes that form a gradient of management intensity (Table 1): the recommended regime for private forestry in Finland or 'business-as-usual' (Hyvän metsänhoidon suositukset, 2006); the recommended regime modified by increased green tree retention, postponed final harvesting (two options), or no thinnings (two options); and set-aside. The recom-

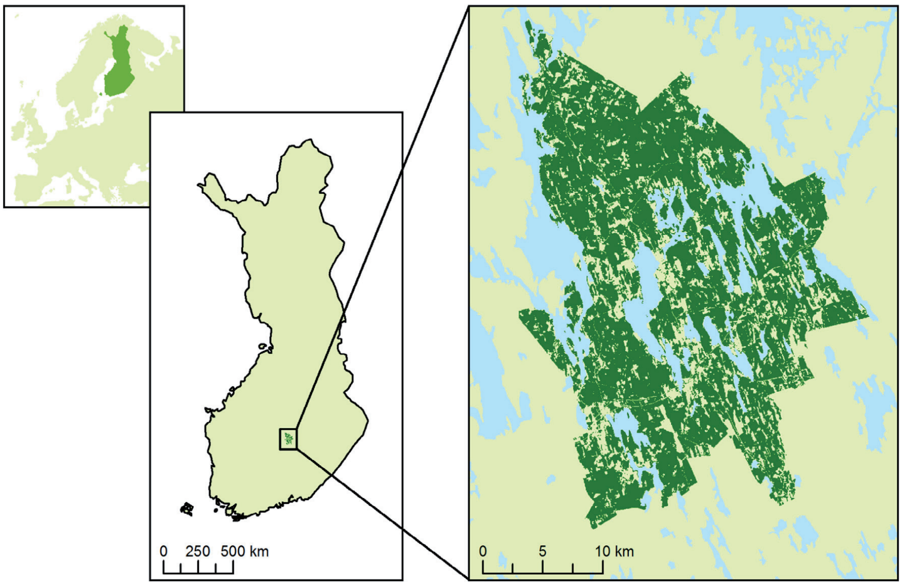

Fig. 1. Map showing the location of the study area in Finland (in dark green color). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Table 1
The seven alternative management regimes used in the forest growth simulations. The development of each stand in the study area was simulated under all of the alternative regimes (adapted from Mönkkönen et al., 2014).

Management regime	Acronym	Description
Business-as-usual	BAU	Recommended management: average rotation length 80 years; site preparation, planting or seedling trees; 1–3 thinnings; final harvest with green tree retention level of 5 trees/ha
Green tree retention	GTR30	BAU with 30 green trees retained/ha at final harvest
Extended rotation (10 years)	EXT10	BAU with final harvest postponed by 10 years (i.e. average rotation length 90 years)
Extended rotation (30 years)	EXT30	BAU with final harvest postponed by ≥30 years (i.e. average rotation length 115 years)
No thinnings (minimum final harvest threshold values)	NTSR	BAU with no thinnings & final harvest adjusted so that rotation does not prolong: average rotation length 77 years
No thinnings (final harvest threshold values as in BAU)	NTLR	BAU with no thinning & final harvest allowed to be delayed: average rotation length 86 years
Set-aside	SA	No silvicultural operations, no harvest

mended regime and the set-aside represent the extremes among the alternatives, while the other five regimes correspond to currently implemented strategies to mitigate biodiversity loss in commercial forests in Finland (Mönkkönen et al., 2014). Postponing the final harvesting, refraining from thinnings, and applying green tree retention are intended to increase the amount of deadwood and enhance the structural diversity within the forest. In general, they may lead to losses in harvest revenues due to delayed harvests (postponed final harvesting), reduced harvest volumes (green tree retention) or smaller sized trees (no thinnings). The simulation period was divided into 5-year time steps, giving a total of 11 model runs. For more details on the forest growth simulations, see Appendix A.

2.2. Ecosystem service and biodiversity indicators

We measured the provision of ecosystem services and biodiversity under each management regime. We considered four ecosystem services: timber production, bilberry production, carbon storage, and pest regulation. This selection represents all ecosystem service categories (provisioning, regulating, and cultural) and a range of spatial scales in which the benefits are realized (local – global). The objective of biodiversity conservation was measured as the availability of deadwood resources.

As timber production is the primary source of revenue to the forest owner, it is usually the main focus of forest management. Timber production was measured as the total amount of harvested commercial timber. This consisted of both pulpwood and saw logs collected during the first and intermediate thinnings as well as final harvesting, as applied in the forest growth simulations. Harvesting of energy wood (e.g. stumps and branches) was not considered.

Bilberry (*Vaccinium myrtillus*) is one of the economically most important wild berries in Finland and bilberry picking has a provisioning as well as a recreational function (Vaara et al., 2013). We used the data on bilberry yield estimates from Peura et al. (2016), where bilberry production was estimated using the models of Miina et al. (2009). Carbon storage by boreal forests has an important role in global climate regulation and maintaining this function is essential for climate change mitigation (Moen et al., 2014; Pan et al., 2011). We used the carbon storage data from Triviño et al. (2015), where the amount of carbon stored in a stand was calculated as the amount of carbon in living trees, deadwood, and extracted timber. We used habitat availability for three-toed woodpecker (*Picoides tridactylus*) as a proxy for pest regulation, as the species is an important natural predator of bark beetles and has been found to have a potentially significant role in regulating bark beetle outbreaks (Fayt et al., 2005). Additionally, three-toed woodpecker is suggested to be an indicator of bird species richness in Finnish forests (Pakkala, 2012). Estimates of habitat availability for three-toed woodpecker were taken from Mönkkönen et al. (2014), where habitat availability was calculated with a model that estimates a habitat suitability index related to the probability of presence of the species based on stand characteristics.

Availability of deadwood resources was selected as the measure of the biodiversity objective given the strong evidence of deadwood as an indicator of broad biodiversity (Gao et al., 2015), and because loss of deadwood is estimated to be the most common cause of species endangerment in Finnish forests (Rassi et al., 2010; Tikkanen et al., 2006). Availability of deadwood was described as the product of its total volume and its diversity, which was measured as the Simpson diversity of different deadwood types (different tree species and decay stages). By taking into account both the volume and the diversity of deadwood, the measure is more likely to be a genuine indicator of deadwood dependent biodiversity (Lassauce et al., 2011). For further details on the calculations of the ecosystem service and biodiversity indicators, see Appendix B.

Timber production was measured across the entire simulation period, i.e. as the total amount of harvested timber over the 50 years. All of the other measures were calculated for each time step of the simulation period and then averaged across time. These average values were used in the analyses.

2.3. Measures of conflicts and compromise solutions

We measured the pairwise conflicts between the objectives listed in the previous section using the methodology of Mazziotta et al. (2017). This method describes a pairwise conflict between objectives *a* and *b* as their tolerance of each other. Tolerance is measured as the proportion of objective *a* that can be achieved while objective *b* is maximized, and *vice versa* (Fig.2A). The method thus results in two values, *a*'s tolerance of *b* and *b*'s tolerance of *a*, recognizing that the conflicts may be asymmetric as management actions may affect different objectives in different, even opposite ways (Mazziotta et al., 2017). The conflict between objectives *a* and *b* is asymmetric if, for example, maximizing *a* leads to a substantial loss in *b*, but maximizing *b* leads only to a small loss in *a*. We measured the pairwise conflicts among all five objectives, resulting in 20 (5×4) pairwise tolerance indices.

For example, to calculate timber production's tolerance of bilberry production, we identified the forest management regime out of the seven alternatives that maximizes bilberry production and compared the amount of timber production under this regime to timber production's potential maximum. The tolerance index is thus the percentage of maximal timber production (achieved under timber-focused management) that is achieved under bilberry-focused management. If this percentage is low, the conflict between the two objectives is severe.

Following the methodology of Mazziotta et al. (2017), the conflict between two objectives may be solved by finding a compromise solution: an optimal management plan that simultaneously minimizes the losses in both objectives when compared to their maximal values (Fig.2B). We implemented this method to identify the compromise management option for each stand and for each pair of objectives. We then compared the values achieved under the compromise management to the maximal values of the objectives.

For each objective, we thus obtained two metrics: the value when another objective was maximized, and the value under compromise management with another objective. Both were expressed as percentage of the maximal achievable value. In order to evaluate conflict severity we obtained the frequency distribution of these conflict measurements by pooling information from all stands. Additionally, we examined the distributions of the management regimes that were identified as providing the compromise solutions.

3. Results

3.1. Ecosystem service potential of the landscape

The forest stands in the study area were highly variable in the potential to provide the measured objectives (Fig. 3). The distributions of the maximal values of all of the objectives were more or

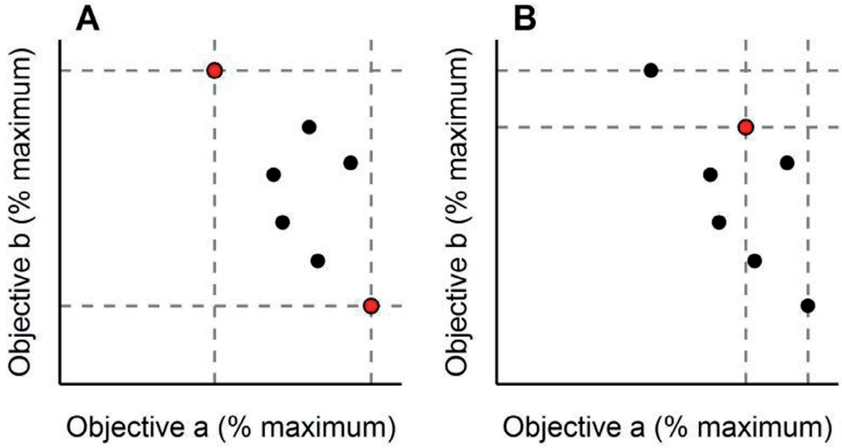

Fig. 2. Illustration of the method used to calculate pairwise tolerance indices and to identify compromise solutions for pairs of objectives. The points show the outcomes of two objectives (*a* and *b*) under different management scenarios. The two red points in (A) show the solutions that maximize the two objectives, respectively. The ratio between the dashed vertical lines measures objective *a*'s tolerance of objective *b*, and the ratio between the dashed horizontal lines measures objective *b*'s tolerance of objective *a*. The single red point in (B) shows the compromise outcome, i.e. the solution that minimizes the losses in both objectives when compared to their maximal values. Here, the ratios between the dashed vertical and horizontal lines measure the goodness of the compromise solution in terms of objective *a* and *b*, respectively.

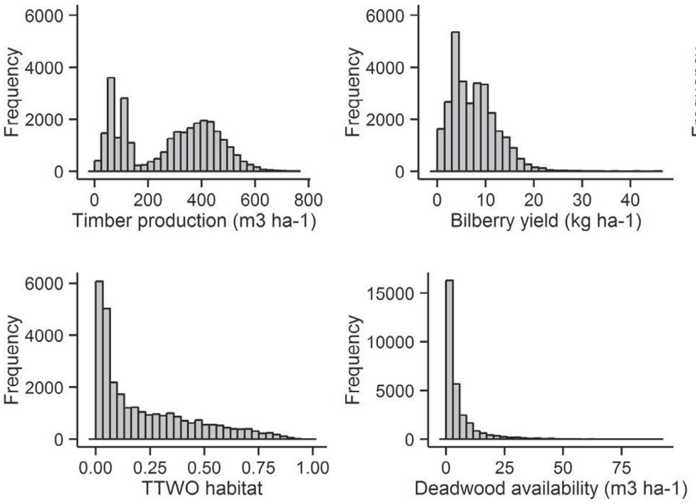

Fig. 3. Distributions of the maximal achievable stand-level values of the five objectives. The values for timber production show the total amount of harvested timber over the 50-year simulation period. The values for the other four measures show the average yearly value over the simulation period. The acronym TTWO refers to three-toed woodpecker. We note that deadwood availability was measured as the volume of deadwood multiplied by its diversity; its unit is thus $m^3 ha^{-1}$ but the values have been weighed by the diversity index and as such do not tell the true volume of deadwood in the stand.

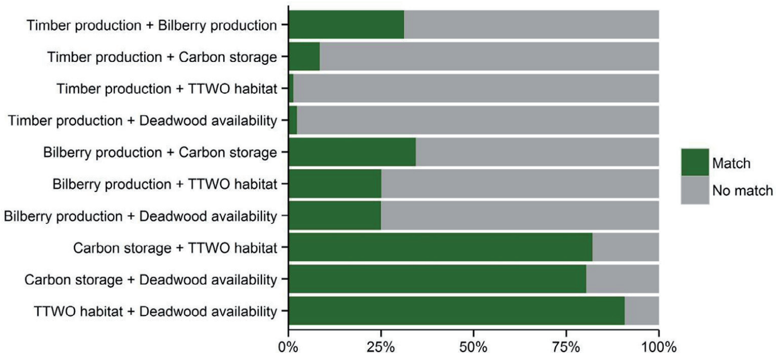

Fig. 4. Proportions of stands where two objectives were ('Match') or were not ('No match') maximized by the same management regime. The acronym TTWO refers to three-toed woodpecker.

less skewed towards low values, but particularly so were the values of the biodiversity objective of deadwood availability. This is likely due to the history of the landscape as production forest (cf. Siitonen, 2001).

3.2. Strength of the pairwise conflicts

There was high variability in the strength of the conflicts among pairs of objectives. A first indicator of the severity of the conflicts was the compatibility of optimal management regimes among pairs of objectives, i.e. the share of stands where both objectives could be maximized by the same management regime. This share of stands was low when one of the objectives in the pair was timber production (1.4%–31.2% of stands; Fig. 4) or bilberry production (25.1%–34.7% of stands; Fig. 4). For all other pairs the same management regime was the most favorable in a majority of stands (80.5%–90.8%; Fig. 4).

When the same management regime could maximize both objectives in a given stand, there was no conflict between them in that stand. To focus on the cases where the objectives were not completely compatible and could thus potentially be reconciled by management choices, we report here the pairwise tolerance indices and the compromise solutions only for those stands where the two objectives required different management regimes to reach their maximal values. This means 9.2%–98.6% of the stands, depending on the pair of objectives (Fig. 4). The results for the full set of stands are reported in Appendix C.

Measured by the pairwise tolerance indices, timber production showed the greatest level of conflict with the rest of the evaluated objectives: very low values of harvested timber were reached as compared to its achievable maximum when another objective was maximized (median values of 0%–17.3%; Fig. 5A-D). Likewise, maximizing timber production led to losses in the other objectives as shown by their low tolerances of timber production (median

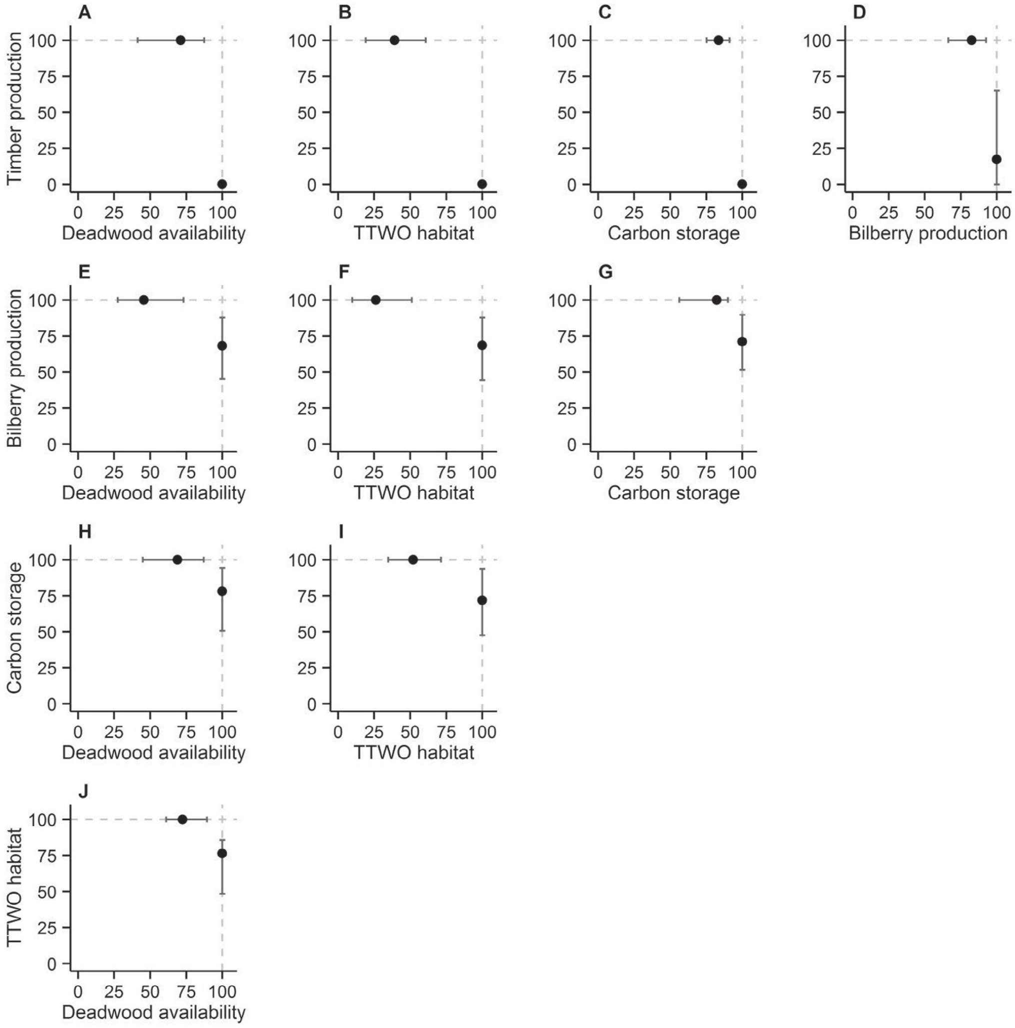

Fig. 5. Pairwise tolerance indices for all pairs of objectives. The black points show the median values and the error bars the second and third quartiles of stand-level values. The tolerance indices are expressed as percentage of the maximal achievable value; units on all axes are thus percentages (%). Dashed grey lines have been added to all plots at y = 100% and x = 100% for graphical comparison. Shown are the results for the stands where both objectives were not maximized by the same management regime. The acronym TTWO refers to three-toed woodpecker.

values ranging between 39.3% and 83.6%; Fig. 5A-D). The second strongest conflict was between three-toed woodpecker habitat and bilberry production (median value for three-toed woodpecker habitat when bilberry production was maximized was 26.2%, Fig. 5F). The tolerance indices for the rest of the pairs were not notably higher (45.7%–82.4%; Fig. 5E, G-J), but, as explained above, these values correspond only to a small proportion of the stands, whereas in a majority of stands the tolerance indices for these pairs were 100% (Fig. C1 in Appendix C).

Some of the observed conflicts between the objectives could be mitigated by finding compromise solutions. In particular, the compromise solutions were very favorable for timber: the median values of timber under the compromise solutions were 100% for all pairs (Fig. 6A-D). However, when compromised with timber production, the solutions were notably less favorable for the other objectives as they could reach values that were only some percentage points higher than their respective tolerances of timber production (median values of 48.7%–88.7%; Fig. 6A-D).

The compromise solutions were slightly more balanced for the second most conflicting pairs, i.e. those involving bilberry production, as here both objectives reached median values ranging between 80.3% and 100% (Fig. 6E-G). For the rest of the pairs among carbon storage, three-toed woodpecker habitat, and deadwood availability, which had shown moderately strong conflicts, the compromise solutions were outstandingly good (median values of 96.5%–100%; Fig. 6H-J). For example, when carbon storage was maximized, three-toed woodpecker habitat could reach only 52.0% of its maximum (Fig. 5I). Under the compromise solution,

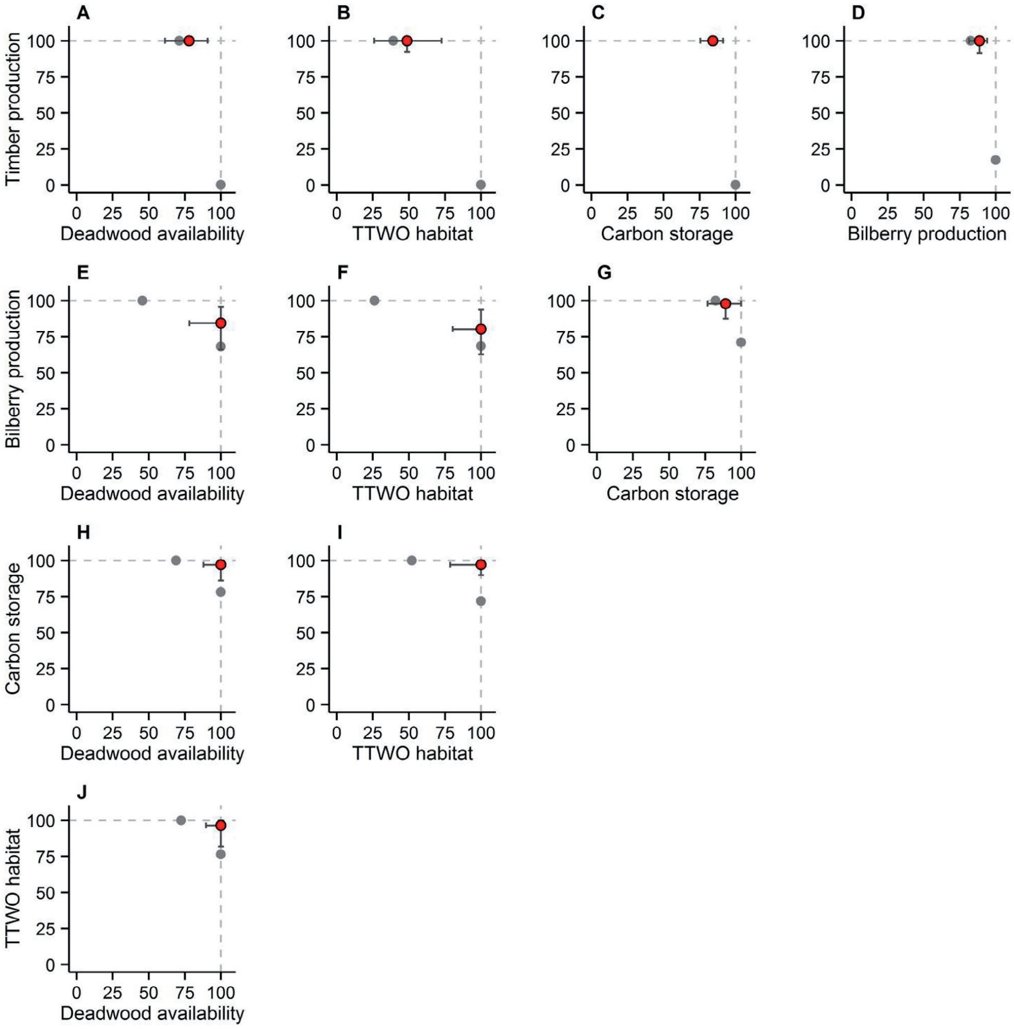

Fig. 6. Compromise solutions for all pairs of objectives. In each plot, the red point shows the median value and the error bars the second and third quartiles of stand-level values of the two objectives under the compromise solution. The two grey points show the median values of the tolerance indices for comparison (same as the black points in Fig. 4). The closer to the point (100, 100) the compromise is, the better it is in terms of the two objectives. Units on all axes are percentages (%). Dashed grey lines have been added to all plots at y = 100% and x = 100% for graphical comparison. Shown are the results for the stands where both objectives were not maximized by the same management regime. The percentage at the bottom corner of each plot shows the proportion of the entire set of stands meeting this condition. The acronym TTWO refers to three-toed woodpecker.

three-toed woodpecker habitat could reach 100% of its maximum while at the same time 97.0% of maximal carbon storage was maintained (Fig. 6I).

3.3. Management regimes

Identifying a single management regime as the compromise solution for two objectives was not always possible due to the fact that two or more management options could lead to similar outcomes. This was usually because, despite being different by definition, in practice they included the same combination of management actions (thinnings and final harvest) during the 50-year simulation period. Some patterns in the compromise solutions between pairs of objectives nevertheless stood out clearly. When timber production was one of the two objectives, the compromise regimes were more or less evenly distributed among 'business-as-usual' (the regime following current Finnish stand management recommendations) and the modified versions of 'business-as-usual' (increased green tree retention, postponed final harvesting, or no thinnings) with proportions ranging between 4% and 35% (Fig. 7). For the pairs not including timber production, the solutions were dominated by the set-aside option with set-aside identified as the compromise regime in 46%–95% of the stands (Fig. 7).

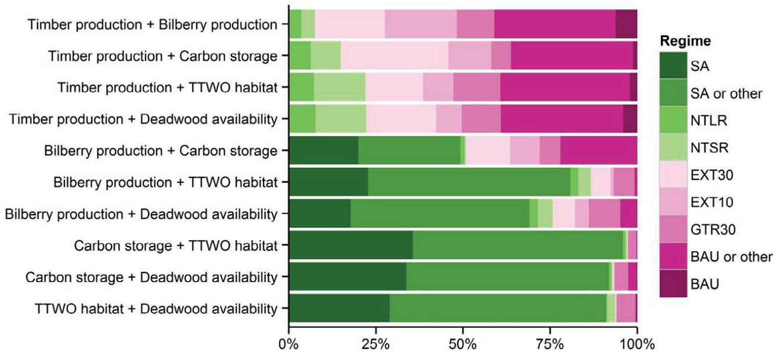

Fig. 7. Distribution of optimal management regimes (compromise solutions) among stands for different pairs of objectives. The regime acronyms stand for: SA – set-aside; NTLR and NTSR – no thinnings; EXT10 and EXT30 – extended rotation time; GTR30 – green tree retention; BAU – business-as-usual. More details are given in Table 1. 'SA or other' refers to cases where set-aside and one or more regimes gave equal outcomes, and 'BAU or other' or business-as-usual and one or more regimes gave equal outcomes (see text). The acronym TTWO refers to three-toed woodpecker.

4. Discussion

In this study, we compared the outcomes of forest management targeting a single objective and management aiming to reconcile two objectives. We measured the supply of each objective when another objective was maximized ('tolerance') and when management was optimized to simultaneously minimize the losses in both objectives ('compromise solution'). Both measures characterize the conflict between the two objectives. We found severe conflicts between timber production and the other objectives, moderate conflicts between bilberry production and the other objectives, and weak conflicts among carbon storage, pest regulation, and biodiversity conservation. Compromise management could mitigate the conflicts, but to varying extents.

Based on previous findings of trade-offs between timber production and other forest benefits (e.g. Duncker et al., 2012; Gamfeldt et al., 2013; Schwenk et al., 2012), we expected the conflicts between timber production and other objectives to be the most severe, and this was indeed the case. When any of the non-timber objectives were prioritized, timber production reached very low values, and when timber production was prioritized, the other objectives reached low (three-toed woodpecker habitat, deadwood availability) to moderate values (bilberry production, carbon storage) (Fig.5A-D). As generalized by the MEA (2005), intensive management for a single provisioning service changes the ecosystem and results in losses in other ecosystem services. Out of the non-timber objectives, bilberry production was the most compatible with timber production (Fig.5D). Other studies have also found non-timber forest products like berries to benefit from stand management activities under certain conditions (Clason et al., 2008; De-Miguel et al., 2014; Nybakken et al., 2013). The high tolerance of carbon storage to timber production (Fig.5C), then again, was likely affected by the calculation method for this objective. Carbon fixed in the biomass of extracted timber was one of the carbon pools included into the measure of total carbon storage, producing values that may favor timber-oriented management unrealistically as the fate of this carbon is not actually known. The fate of carbon fixed in harvested wood products may critically affect whether production forests act as carbon sources or sinks (Liski et al., 2001). Like timber production, bilberry production was in strong conflict particularly with three-toed woodpecker habitat and deadwood availability (Fig.5E-F). Conversely, carbon storage, three-toed woodpecker habitat, and deadwood availability were all highly compatible with each other (Fig.5H-J).

The compromise solutions showed the most prominent improvements in terms of timber production. Timber production reached very low levels when other objectives were prioritized (Fig.5A-D) but very high levels under the compromise solutions (Fig.6A-D). However, these results should not be misinterpreted as indicating an efficient solution to the conflict, because there was notable asymmetry in the goodness of the compromise solutions between objectives. The levels of the other objectives under the compromise solutions were only slightly higher than their tolerances of timber production whereas timber production was at or close to its maximum (Fig.6A-D), meaning that the compromise solutions were, in fact, only slightly different from prioritizing timber production. The non-timber objectives could be reconciled with each other with more balanced outcomes, with particularly carbon storage, three-toed woodpecker habitat, and deadwood availability all reaching very high levels (Fig.6H-J). The conflicts between timber production and the other objectives were thus not only the most severe but also the most difficult to solve.

It should be noted that the results of our study are influenced by the selection of management regimes included in the study. In particular, timber production's low tolerance of carbon storage, three-toed woodpecker habitat, and deadwood availability is probably for the most part due to the predominance of the set-aside regime in maximizing these objectives. In this regime, by definition, the stand is not harvested at all and timber production is thus zero. Consequently, when timber production was among the objectives, the set-aside regime was not identified as the compromise solution in any of the stands, and timber production reached much higher levels. The low values of the non-timber objectives under the compromise solutions suggest that in most stands the management regimes other than set-aside are more or less equally bad in terms of these objectives, limiting the possibilities to find efficient compromises and true 'land-sharing' solutions. This is further indicated by the high proportion of the set-aside option as the management to solve the conflicts between bilberry production and other objectives.

Another important consideration is that the measures of the conflicts used here are calculated based on the maximal achievable level of each objective under the alternative management regimes over the 50-year simulation period. They are thus conditional to what is achievable under these management regimes and over this time period. For example, the maximal total volume of deadwood as predicted by our simulations was on average $10 m^3 ha^{-1}$, which seems very low as there may be manifold amounts of deadwood in natural old-growth forests (Siitonen, 2001). If the simulation time had been longer and the forests had had more time to recover from their past as production forest, even higher maximal values could have been achievable for deadwood availability and the other non-timber objectives, and as a consequence the conflicts between them and timber production would have potentially appeared even more intense.

In short, in most of the stands in our study area, comparatively high levels of bilberry production, carbon storage, three-toed woodpecker habitat, and deadwood availability could be simultaneously achieved by permanent set-aside of the stand, whereas substantial losses in particularly three-toed woodpecker habitat and deadwood availability appear inevitable when timber production is also targeted. The situation is further aggravated by findings suggesting that targeting timber production may increase the conflicts between other objectives (Triviño et al., 2017). Naturally, permanent set-aside of large parts of the production forest may not be in the interest of the land-owners. However, as explained above, the selection of alternative management regimes most likely affects the results. Securing high levels of ecosystem services and biodiversity while also harvesting for wood may be possible, but require adoption of stand management regimes that differ from the current 'business-as-usual' even more strongly than the alternative regimes included here (for example, selective logging; Kuuluvainen et al., 2012; Pukkala, 2016). The next step is to identify the exact mechanisms causing the losses in ecosystem services to inform development of forestry regimes that minimize these harmful impacts so that better compromises can be achieved. This requires a deeper understanding of the nature of the relationships between the different objectives, including whether they are interacting with each other or just affected by the same drivers (Bennett et al., 2009).

Earlier work has shown that conflicts among timber production and other ecosystem services may be mitigated by optimizing the application of management regimes at a landscape level (Miina et al., 2010; Mönkkönen et al., 2014; Schwenk et al., 2012; Triviño et al., 2015). This may involve, for example, setting aside the stands that are most favorable for biodiversity objectives, and applying intensive management in the stands that are best at producing timber. These types of 'land-sparing' options were explicitly not considered in the present study. The effect of spatial scale and the effects of other spatial factors (e.g. location of demand for the services) on the achievability of good compromise solutions remain questions for future research. Besides spatial, also the time frame of the study and the temporal variation of supply and demand of different ecosystem services should be considered. Here, for example, we considered the supply of each objective averaged over the simulation time, but not its evenness over time. Of particular importance may be, for example, the evenness of timber supply, or the temporal continuity of deadwood resources for biodiversity conservation (Jonsson et al., 2005; Siitonen et al., 2000).

Our study shows that in boreal production forests the conflicts between the primary provisioning service of timber and other benefits are real, severe, and challenging to solve. Research into the processes affecting the supply of different forest ecosystem services may aid in the design of forestry practices and planning management regimes that protect diverse forest benefits. Here, forestry policies may also play an important role. A recent review by 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 benefits. This is exactly the case in privately owned production forests, where the financial gains of the forest owner may contrast with public benefits such as recreational use, water quality regulation, and climate change mitigation. Besides new management practices, new regulations and/or incentives such as certification schemes or payments for ecosystem services (e.g. Patterson and Coelho, 2009) may be required to encourage the adoption of more sustainable management practices and to improve the protection of public interests in boreal production forests.

Acknowledgments

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Appendix A. Details on the forest growth simulations

We simulated the development of the forest stands with the MOTTI stand simulator (Hynynen et al., 2002; Salminen et al., 2005). MOTTI projects the development of a stand based on its initial characteristics and the forestry operations applied during the simulation. The system core comprises distance-independent tree-level models for growth and mortality operating at 1–5 year steps. The parameterization of tree growth in MOTTI is based on extensive field data from sample plots on forestry land in Finland and representing prevailing growing conditions and management regimes. The uncertainty of predictions increases when individual models are used outside their intended range. This may be the case when simulating the most extreme options.

The stand development was simulated for 50 years into the future. We selected the simulation time of 50 years as it is a compromise between the typical rotation length in the area and the validity of MOTTI simulations.

It should be noted that the effects of climate change on forest growth were not taken into account in the simulations. Climate change is expected to have significant effects on forest growth; however, these are expected to become evident only towards the end of the 21st century and remain inconsequential within the next 50 years (Kellomäki et al., 2008). For this reason we did not take them into account in the simulations.

Appendix B. Additional details on the calculations of the ecosystem service and biodiversity indicators

Bilberry production

Bilberry (*Vaccinium myrtillus*) production data was taken from Peura et al. (2016), where bilberry production was estimated using the models and methods of Miina et al. (2009, 2010). These models predict bilberry coverage and yield based on stand characteristics (e.g. dominant tree species, stand age, and stand basal area).

Carbon storage

The carbon storage data was taken from Triviño et al. (2015). Here, carbon storage was calculated as the amount of carbon stored in living trees, deadwood, and extracted timber at a given time (for each time step of the simulation period). The total tree biomass (living, extracted, and cutting residues) was estimated from the MOTTI predictions of timber volume, and the amount of carbon in the biomass was calculated by multiplying it by 0.5. For deadwood and cutting residues, the decaying rate of the biomass was taken into account. For further details, see Triviño et al. (2015).

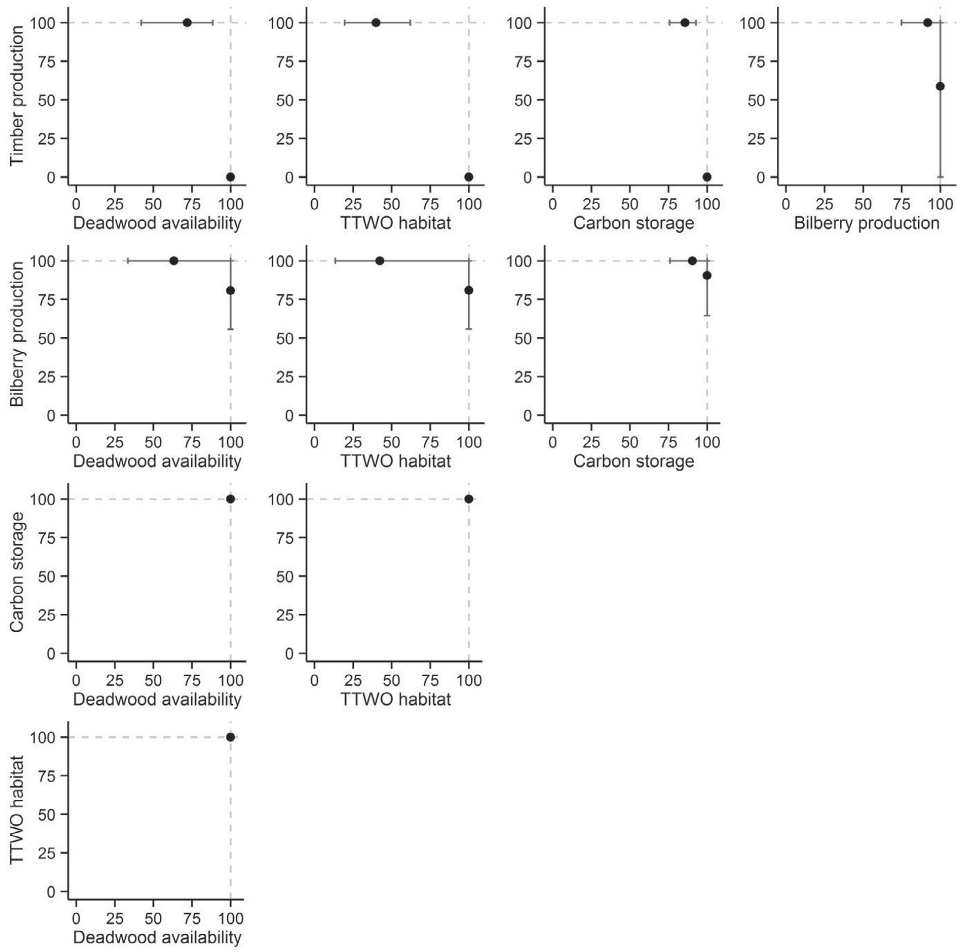

Fig. C1. Pairwise tolerance indices for all pairs of objectives. The black points show the median values and the error bars the second and third quartiles of stand-level values. The tolerance indices are expressed as percentage of the maximal achievable value; units on all axes are thus percentages (%). Dashed grey lines have been added to all plots at y = 100% and x = 100% for graphical comparison. TTWO stands for three-toed woodpecker.

Habitat availability for three-toed woodpecker

Estimates of habitat availability for three-toed woodpecker (*Picoides tridactylus*) were taken from Mönkkönen et al. (2014). Three-toed woodpecker prefers mature forests with abundant fresh deadwood to use as feeding and nesting resources. Mönkkönen et al. (2014) calculated estimates of habitat availability for the species with a model that estimates a habitat suitability index related to the probability of presence of the species. It is estimated based on the total basal area of recently died trees (*BA*) and the total stem volume of living trees (*V*). The model combines the logistic regression model constructed by Roberge et al. (2008) linking *BA* and occurrence of the species, and threshold values for site quality measured as *V* suggested by Pakkala et al. (2002). Mönkkönen et al. (2014) calculated the habitat suitability index as the product of these two models, so that the value of the index varies between zero and one. It gets a value of zero if *V* is <60 m^3, increases as *BA* and *V* increase, and is close to one when *BA* is >2.5 $m^2\ ha^{-1}$ and *V* is >200 m^3.

Availability of deadwood resources

The MOTTI simulations produce estimates of the volume of deadwood (kg/ha) in a stand at each time step. Deadwood volume is estimated separately for 20 different deadwood types, given by four tree species × five decay stages. The four tree species are the four most dominant species in the region: pine (*Pinus sylvestris*), spruce (*Picea abies*), and birch (*Betula pendula* and *Betula pubescens*). The five decay stages are based on Mäkinen et al. (2006) and are the following ones: 1) recently dead tree; 2) weakly decayed; 3) medium decayed; 4) very decayed; and 5) almost decomposed. We measured the diversity of deadwood using Simpson's diversity index (D) calculated over the 20 deadwood types. The availability of deadwood resources (*DWA*) in a stand was then

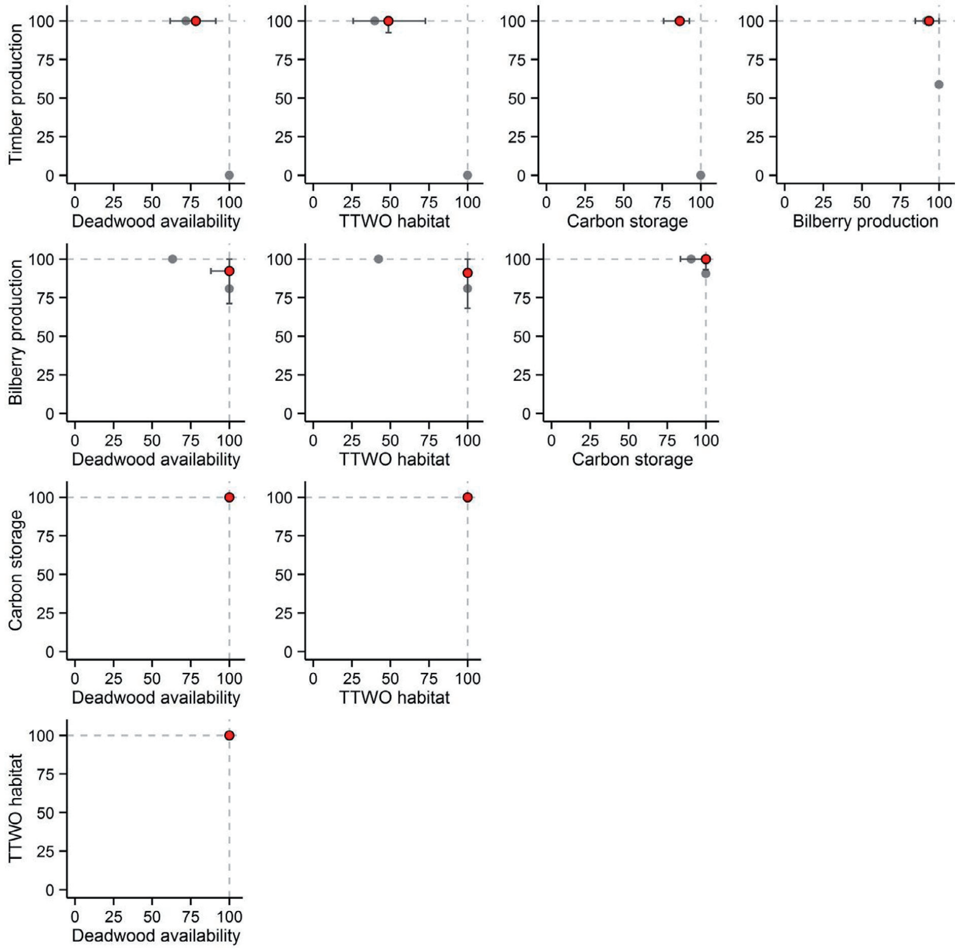

Fig. C2. Compromise solutions for all pairs of objectives. In each plot, the red point shows the median value and the error bars the second and third quartiles of stand-level values of the two objectives under the compromise solution. The two grey points show the median values of the tolerance indices for comparison (same as in Figure S1). Units on all axes are 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 compromise is, the better it is in terms of the two objectives. TTWO stands for three-toed woodpecker.

calculated as the product of the total volume of deadwood and its diversity:

$$DWA = (1 - D)\sum_{i=1}^{20} Vol_i.$$

Appendix C. Results for the full data set

When the stands in which the same management regime could maximize both objectives were not excluded, the pairwise tolerance indices were naturally higher and the compromise solutions better, and this was especially evident for the pairs of objectives which were the most compatible (i.e., those among carbon storage, TTWO habitat, and deadwood availability). Still, the conflicts were the strongest between timber production and the other objectives, and the second strongest for pairs involving bilberry production (Fig. C1). Conversely, the tolerance indices were high (median values of 100%, i.e. no conflict in majority of the stands) for all pairs among carbon storage, three-toed woodpecker habitat, and deadwood availability (Fig. C1). The goodness of the compromise solutions followed the same patterns: the most severe conflicts could not be solved with balanced outcomes, whereas the compromises were very good when the conflicts between the objectives were weak to begin with (Fig. C2).

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III

MORE IS MORE? FOREST MANAGEMENT ALLOCATION AT DIFFERENT SPATIAL SCALES TO MITIGATE CONFLICTS BETWEEN ECOSYSTEM SERVICES

by

Tähti Pohjanmies, Kyle Eyvindson, María Triviño & Mikko Mönkkönen 2017

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RESEARCH ARTICLE

More is more? Forest management allocation at different spatial scales to mitigate conflicts between ecosystem services

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Abstract

Context Multi-objective management can mitigate conflicts among land-use objectives. However, the effectiveness of a multi-objective solution depends on the spatial scale at which land-use is optimized. This is because the ecological variation within the planning region influences the potential for site-specific prioritization according to the different objectives.

Objectives We optimized the allocation of forest management strategies to maximize the joint production of two conflicting objectives, timber production and carbon storage, at increasing spatial scales. We examined the impacts of the extent of the planning region on the severity of the conflict, the potential for its mitigation, and the strategies that were identified as optimal.

Methods Using forecasted data from a forest simulator, we constructed Pareto frontiers optimizing the joint provision of the objectives in production forests in Finland. Optimization was conducted within increasing hierarchical spatial scales and outcomes were compared in terms of the severity of the conflict and the solution to mitigate it.

Results The trade-offs between timber production and carbon storage appeared less severe and could be mitigated more effectively the larger the planning regions were, but the improvements became minor beyond the scale of 'large forest holding'. The results thus indicate that this scale, approximately 100 stands or 200 ha, is large enough to effectively mitigate the conflict between timber production and carbon storage.

Conclusions Management planning over relatively small forest areas (200 ha) can mitigate ecosystem service trade-offs effectively. Thus the effective use of multi-objective optimization tools may be feasible even in small-scale forestry.

Keywords Carbon storage · Timber production · Land-sharing · Land-sparing · Landscape extent · Multi-objective optimization · Finland

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Introduction

The nature of the relationships among resource extraction, ecosystem services and biodiversity conservation is a central question in applied ecological research, landscape ecology, and sustainability science (Carpenter et al. 2009; Cimon-Morin et al. 2013; Abson et al. 2014). It is often of interest to promote all of these three objectives within the same land area, for example in agriculture (e.g., Swinton et al. 2007) or forestry (e.g., Edwards et al. 2014). However, they have been found to be commonly conflicting (Tallis et al. 2008; McShane et al. 2011), particularly resource extraction with the other two (MEA 2005; Burger 2009; Power 2010). Potential solutions to these conflicts include carefully designed management systems that are able to support multiple objectives in the same site (e.g., Miina et al. 2010), and the identification of optimal land-use allocation which prioritizes individual objectives in the sites that are the most favorable to them (e.g., Cordingley et al. 2016). These are sometimes termed 'land-sharing' and 'land-sparing' strategies, respectively (Maskell et al. 2013; Edwards et al. 2014). Both of them have the potential to lead to compromise outcomes, where the simultaneously achieved levels of multiple, conflicting objectives are maximized.

Several recent studies have explored the conflicts between resource extraction, ecosystem services and biodiversity in production forests, where timber harvesting and other forestry-related activities modify ecosystem structures and functions (Duncker et al. 2012; Schwenk et al. 2012; Gamfeldt et al. 2013; Mönkkönen et al. 2014; Zanchi et al. 2014; Triviño et al. 2015; Peura et al. 2016; Vauhkonen and Ruotsalainen 2017; Triviño et al. 2017). These studies have shown that management trade-offs between timber harvests, ecosystem services and biodiversity objectives are common but can be highly case and site dependent. As a consequence, some have suggested that a diversity of management approaches applied at a landscape level may be recommendable (Duncker et al. 2012; Schwenk et al. 2012). Indeed, optimization studies (Mönkkönen et al. 2014; Triviño et al. 2015; Peura et al. 2016; Triviño et al. 2017) have shown that a combination of different management regimes across the landscape, ranging from intensive forestry to permanent set-aside, is required to most efficiently mitigate the trade-offs. In other words, a combination of 'land-sharing' and 'land-sparing' types of management may be required to achieve the best compromises to balance conflicting objectives. This is because forests are not of uniform quality with respect to different objectives or to the system's responses to management activities (Gamfeldt et al. 2013).

Conducting ecosystem service assessments, management planning or land-use prioritization always involves decisions about spatial scales, such as the size of the study area or planning region (landscape extent), the size of planning units (landscape resolution), and data resolution and coverage (e.g., Mills et al. 2010). Because ecosystem services and other management objectives are typically unevenly distributed (e.g., Egoh et al. 2008; Raudsepp-Hearne et al. 2010), these decisions may be highly consequential to the results of assessments and the management recommendations that are drawn from them. For example, Blumstein and Thompson (2015) found the spatial scale at which ecosystem service hotspots are identified (state, watershed, or town) to strongly affect their perceived abundance and distribution. Similarly, Anderson et al. (2009) and Raudsepp-Hearne and Peterson (2016) found the resolution of the analysis to influence the observed patterns in the supply of ecosystem services and the strength and direction of correlations between the services. In a conservation planning context, for example Rodrigues and Gaston (2002) showed that the definition and size of the planning region may significantly affect perceived species rarity and subsequent site prioritization.

The above examples show that if management objectives are unevenly distributed, the extent of the planning region affects the identification of the best sites for different individual objectives as well as the sites where good 'land-sharing' outcomes are achievable. Finding a balance for conflicting objectives requires allocating management regimes to where they perform best, for instance sites which can provide high levels of multiple objectives should be managed according to the 'land-sharing' concept. Therefore, the extent of the planning region may influence exactly what can be achieved by optimizing management allocation, and how. With a larger planning region more options are available to efficiently assign each site to the most appropriate management regime for the problem at hand. This is indeed why land-use optimization is recommended to be done at large scales (e.g., 'landscape scale': Duncker et al. 2012; Schwenk et al. 2012; Mönkkönen et al. 2014; Triviño et al. 2015). Then again, the larger the planning region is, the more computationally demanding it is to solve the optimization problem (Martin 2001). Additionally, land management based on large-scale plans may be difficult to implement if there are no administrative systems in place at corresponding scales, for example if coordination of activities by several land-managers is required but there are no existing systems to support it. What is more, landscape-level management plans that are optimal for multiple objectives may result in an uneven distribution of costs and benefits among land owners which can hurt the plan's acceptability (e.g., Kurttila et al. 2002; Jumppanen et al. 2003). To reconcile the benefits of planning at large spatial extents and the feasibility and implementation of more local land-use plans, it is useful to explore the consequences of how the planning region is defined and to identify the smallest scale at which conflicts among management objectives can be mitigated effectively.

In this paper, we test for the effect of the spatial scale at which management allocation is optimized on the achieved outcomes in production forests in Finland. We do this by constructing Pareto frontiers optimizing the joint provision of two objectives, timber production and carbon storage, with varying spatial scales within which optimal management allocation is identified. Pareto frontiers comprise a set of multi-objective solutions that cannot be improved in terms of any one objective without deteriorating in terms of at least one of the other objectives (Miettinen 1999). The use of Pareto frontiers in land-use planning can provide important information on land-use trade-offs and their possible solutions (Seppelt et al. 2013). The forest management alternatives included in our analyses encompass 19 management regimes. These regimes have different amounts of harvestable timber as well as carbon stored in the system depending on productivity of the site (Äijälä et al. 2014). Promoting tree growth and thus timber production is typically the main focus of management planning in production forests, while carbon-related functions are gaining increasing attention on policy agendas in order to mitigate climate change and to ensure the carbon neutrality of forest-based energy sources (e.g., Ministry of Agriculture and Forestry 2015). Previous work (Hynynen et al. 2005; Triviño et al. 2015; Pukkala 2016) has shown that there are conflicts between timber production and carbon storage in Finnish forests, but that the conflicts can be mitigated by landscape-level planning and prioritization. However, the definition of the required 'landscape-level' has remained imprecise and the effect of the spatial scale unknown.

We use hierarchical spatial scales to look for the most efficient scale of management planning: individual forest stand, small forest holding, large forest holding, watershed, and region. These scales reflect real administrative and/or natural boundaries, with increasing numbers of forest owners included within each scale. In Finland, the majority of production forests are privately owned and forest holdings are comparatively small with an average size of approximately 30 ha (Peltola 2014). Large-scale, landscape-level forest management planning may thus require the cooperation of several forest owners and potentially compensation systems that make the plan acceptable for all of them (Kurttila et al. 2001). We work with the assumption that due to both computational and real-world practicality it is desirable to minimize the spatial scale of management planning.

The main questions we aim to answer are: (1) Are there differences in levels of timber production and carbon storage that can be simultaneously achieved when management is optimized at different scales? (2) Are there differences in the distribution of different kinds of management regimes when management is optimized at different scales?

Methods

Forest data and forest growth simulations

The basic unit of forest management planning is a forest stand, i.e., a parcel of forest of relatively uniform structure and type. Our study areas, located in southern and central Finland, comprise a total of 28,886 forest stands, covering nearly 44,000 ha. The forests are of variable ages and mainly mesic heaths, herb rich heaths and sub-xeric heaths dominated by spruce (*Picea abies*), birch (*Betula pendula* and *Betula pubescens*) and pine (*Pinus sylvestris*) (Fig. 1). Stand-level forest inventory data from these areas, produced by the Finnish Forest Centre, were used as input data in the forest growth simulator SIMO (Rasinmäki et al. 2009). SIMO forecasts the development of a stand based on its initial condition and the forest management actions applied to it during the simulation period. We simulated the development of the stands for one hundred years into the future under 19 alternative management regimes. These included the management regime that is currently recommended in Finland (Äijälä et al. 2014), a total of 16 modified versions of the recommended regime, continuous cover forestry,

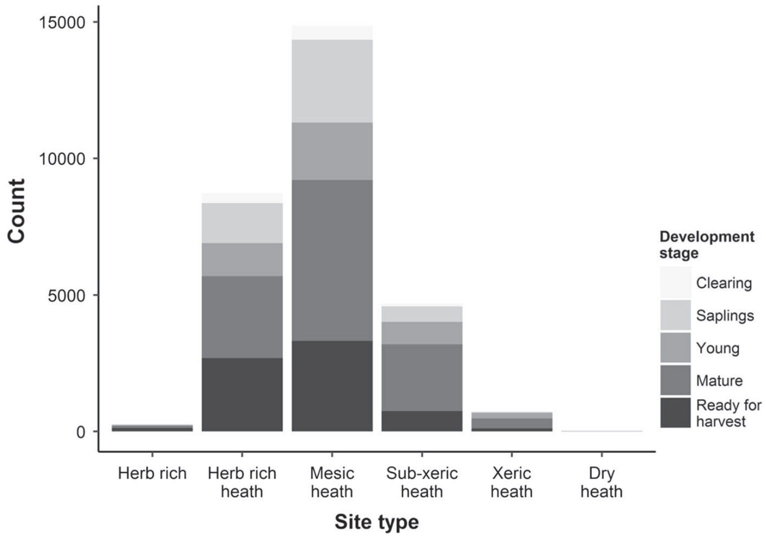

Fig. 1 The distribution of the site types (x axis) and current development stages (bar fill) of the forest stands included in the study areas. Site types are presented according to the Finnish forest classification system (Hotanen et al. 2008). The development stage 'Young' refers to a stand with an average diameter at breast height of 8–16 cm, and 'Mature' to a stand with an average diameter at breast height greater than 16 cm but that is not yet ready for final harvest. The development stage 'Ready for harvest' is defined according to current Finnish recommendations for the timing of final felling (Äijälä et al. 2014)

and permanent set-aside. The currently recommended regime, henceforth termed business-as-usual, consists of commercial thinnings, a final felling timed to achieve the stand's maximal growth, and regeneration of the stand by planting or seeding after final felling. In the SIMO simulations the timing of these operations is determined by decision rules regarding the site type, the height of the dominant tree species and the age of the stand. The modified versions of business-as-usual were implemented by adjusting these rules and included alterations of the timing of the final felling (postponing it by 5–30 years), conducting thinnings before and/or after final felling, refraining from thinnings completely, and adopting green tree retention for natural regeneration. These modifications are intended to reflect real-world variation in possible forest management choices and have corresponding policy incentives according to which forest owners are encouraged to modify management for multiple objectives (for further details, see Mönkkönen et al. 2014). Continuous cover forestry differs from business-as-usual in that it is based on regular, selective harvesting of large trees and no final felling (Pukkala et al. 2012). It has been suggested to have the potential to maintain forest biodiversity and ecosystem services better than conventional rotation forestry (Kuuluvainen et al. 2012). The set-aside option then again corresponds to protection of the forest, i.e., no management actions are taken and no timber is harvested. The simulation period of 100 years was divided into 20 five-year time steps, with the simulator producing predictions of stand development at each time step as output.

Measurement of objectives

Two objectives were measured throughout the simulation period based on the forecasted stand properties: timber production and carbon storage. Timber harvests are the primary source of income to a forest owner from their land and as such typically the main focus of forest management. We used net present income as the measure of timber production. This was calculated by multiplying the recent average prices for different timber assortments (Peltola 2014) by the quantity of each assortment harvested during thinnings and/or final felling. We used a discount rate of 3% to discount income generated in the future. We used this rate as it has been traditionally used in forest economics (e.g., Gren et al. 2014; Asante and Armstrong 2016).

Carbon storage is a crucial ecosystem service contributing to climate change mitigation. Carbon stored in the forest was calculated as the sum of the estimated amounts of carbon fixed in living wood, dead wood and soil. Carbon contained in living and dead wood was estimated as 50% of the biomass. Carbon stored in soil was estimated using two models depending on the soil type: the Yasso07 models (Liski et al. 2005; Tuomi et al. 2009, 2011) were used for mineral soils and the carbon flux models of Ojanen et al. (2014) were used for peatland soils. Carbon storage was estimated for each time step of the simulation period and the average over the time steps was used in the optimization analysis.

Spatial scales

The 28,886 forest stands were grouped together at different spatial scales that correspond to administrative and/or natural boundaries: small forest holdings, large forest holdings, watersheds, and regional scale (Fig. 2; Table 1). These were hierarchical so that small holdings made up the large holdings, large holdings made up the watersheds, and the regional scale included all of the watersheds. The smallest scale groupings, small holdings, were created based on real forest property data. The large holdings were created by grouping together adjacent small holdings so that each large holding contained approximately 10 small holdings. The watershed scale was defined by the boundaries of third-level catchment areas as delineated by the Finnish Environment Institute (SYKE 2010). The average area and the average number of stands included in each of these levels are given in Table 1. We used these four hierarchical scales as planning regions, i.e., as boundaries within which management allocation was optimized in terms of the two objectives (timber production and carbon storage). The resolution was the same at every scale, i.e., the stand level.

Optimization analysis

If management objectives are conflicting, their maximal levels cannot be reached at the same time, i.e., by the same type of management. Multi-objective optimization tools can be used to find management plans that solve these types of conflicts as efficiently as possible (Miettinen 1999), for example maximize one

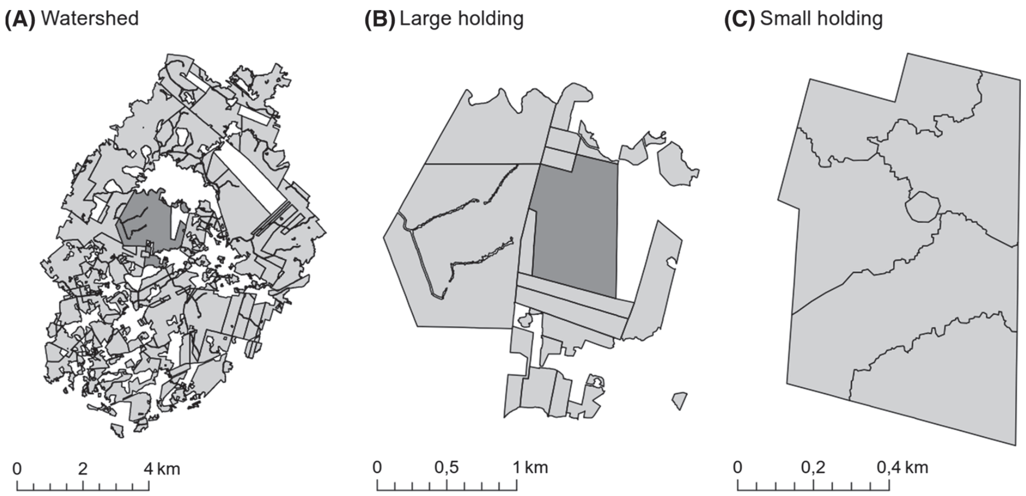

Fig. 2 Examples of the sub-regional scales analyzed. **a** An example of a watershed. The large forest holdings contained within the watershed are drawn on the map with black outlines. The dark grey area marks the large holding shown in **b**. **b** An example of a large forest holding with small forest holdings drawn with black outlines and the small holding shown in **c** highlighted in dark grey. **c** An example of a small forest holding with individual forest stands drawn with black outlines

Table 1 Total number of groups of stands used as planning regions, average number of stands in a group, and average area of a group as defined at the different spatial scales

Scale	Total number of groups	Mean number of stands	Mean area (ha)
Stand	28,886	1	1.5
Small holding	2537	11	17.3
Large holding	228	127	192.9
Watershed	16	1805	2748.1
Region	1	28,886	43,970.2

objective given the constraint that another objective stays above a set target level. A Pareto optimal solution to a multi-objective optimization problem is one with an outcome that cannot be improved with respect to any of the objectives without causing losses in some of the other objectives. Pareto optimal solutions are a subset of all feasible solutions and make up a Pareto frontier, which for two objectives can be graphically presented as a production possibility curve. The steepness of the curve reflects the severity of the trade-off, because the steeper the curve the greater the loss in one objective caused by an increase in another objective.

For each group of stands, defined at the different spatial scales, we identified the set of Pareto optimal solutions that maximized the joint production of timber and carbon storage. We maximized the value of timber production under the constraint that carbon storage was above a set target level, and by adjusting this required level we were able to build the Pareto frontier. Each Pareto optimal solution comprised a selected management regime for each stand in the group, i.e., the optimal allocation of management regimes within the group. The stands were thus treated as individual planning units that contribute to the overall outcome across the group but do not interact with each other. At the largest scale (region), management allocation was optimized over all stands. At the smaller scales, management allocation was optimized within each group and the levels of the two objectives from each group were then summed together to produce the overall outcome. The optimization analyses were carried out using the IBM ILOG CPLEX optimizer, version 12.6.2 (https://www.ibm.com/developerworks/downloads/ws/ilogcplex/). In addition to the four stand groupings, we carried out the optimization for each stand individually by selecting the management regime among the 19 alternatives considered here that maximized the joint production of the two objectives in that stand. Similar to the sub-regional scales, the stand-scale values of the two objectives were summed together across stands and the total values were compared to the levels of the objectives that could be achieved when planning at the larger scales. We note that better solutions may be achievable by more detailed stand management optimization than what is allowed for by the 19 management alternatives, but the stand-scale results obtained here are meant to serve primarily as comparison with planning over larger forest areas.

The trade-off between two objectives may be further characterized by three points on the production possibility curve that illustrate how compatible the objectives are: the two extreme ends of the curve and the 'compromise' point (Mazziotta et al. 2017). The compromise point is defined as the solution in the Pareto optimal set that minimizes the maximum loss in the objectives (Mazziotta et al. 2017). Roughly, this corresponds to the solution that is closest to an ideal solution where all objectives are maximized at the same time. In the general case for two objectives, the steeper the production possibility curve is, the farther the compromise solution is from the ideal solution. We identified the compromise solutions for timber production and carbon storage for the different spatial scales and used them to further examine the differences in the perceived severity of the trade-off and the optimal management allocations between the different scales (see below). We note that the compromise solutions identified here are optimal only in a mathematical sense, not in a 'social' sense. Selecting the 'socially optimal' solutions would require information on societal goals and weights assigned to different objectives.

As explained above, the compromise outcome is one point on the Pareto frontier and it is achieved by a set of management regimes, one for each stand. This set may consist of 'land-sparing' management regimes (either one of the objectives is maximized in a stand) and 'land-sharing' management regimes (both objectives reach moderate levels in the same stand). Within the compromise solutions for the different spatial scales, we recorded the proportions of 'land-sparing' and 'land-sharing' regimes. We identified these two types post hoc based on the regimes' outcomes with respect to the two objectives, so that 'land-sparing' was defined as maximizing only one of the objectives and 'land-sharing' as both objectives being below their potential maximums. Additionally, we recorded the proportion of 'win–win' outcomes, defined as cases where both objectives were maximized by the same regime. We compared the total area of each of these options (land-sharing, land-sparing for timber, land-sparing for carbon, win–win) under the compromise solutions at different spatial scales.

Results

Joint production of timber and carbon storage at different spatial scales

The maximal value of timber production measured as net present income over the entire study area was 433 million euros, or 9800 euros per ha, and the maximal carbon storage was 11.18 million tons, or 254 tons per ha. It was not possible to achieve both of these maximums at the same time, but the losses in carbon storage when timber production was maximized and vice versa were considerable: when timber production was maximized, carbon storage was 7.38 million tons (i.e., 66% of its potential maximum), and when carbon storage was maximized, the net present income for timber production was 21 million euros (i.e., 5% of its potential maximum). These two outcomes correspond to the extreme ends of the production possibility curve (Fig. 3). They are the same regardless of the spatial scale of the analysis, because the solution that is optimal in terms of only one objective consists of the best regime for that objective in each stand and so does not depend on the spatial scale within which management alternatives are allocated.

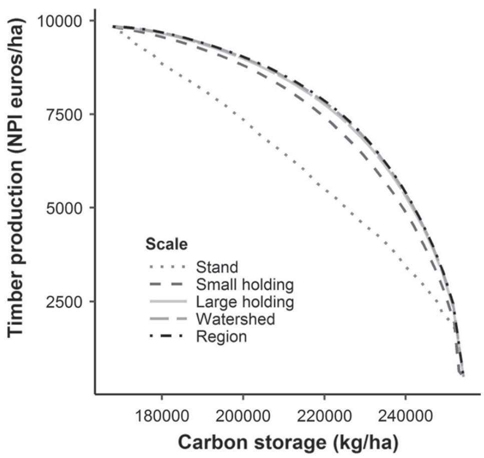

Fig. 3 Production possibility curves showing the simultaneously achievable levels of timber production and carbon storage over the entire study area when optimal management allocation was determined at different spatial scales

The smaller the spatial scale of management optimization, the steeper the resulting production possibility curve was (Fig. 3). The results obtained by selecting optimal management at the scale of individual stands stood clearly apart from results obtained by optimizing management over multiple stands (Fig. 3). Beyond this, the difference was the most notable between the 'small holding' scale and the three largest scales, while the differences among the three largest scales were very small. Differences in the outcomes that were achieved when management was optimized at the different spatial scales can be expressed as the difference in the value of one objective when the other objective is held at a set level. Excluding the stand-scale, this difference ranged up to 13%, depending on the objective, the required level of the constraint, and the scales compared. For example, when carbon was required to reach at least 95% of its maximal value and the optimization was conducted at the 'small holding' scale, timber production could reach 45.8% of its maximum, but when the optimization was conducted at the larger scales with the same carbon constraint, timber production could reach 51.0, 51.8, or 52.1% of its maximum at the 'large holding', watershed, and regional scale, respectively. In absolute terms, the difference of 6 percentage points between the 'small holding' and regional scale meant an increase of over 27 million euros.

Under the compromise solutions, i.e., where the losses in both objectives were simultaneously minimized, both objectives reached at least 76% of their maximal values at the stand scale and at least 82% of their maximal values at the larger scales (Table 2). When the scale of the management optimization was increased from the 'small holding' scale to larger scales, the simultaneously achievable values increased by an average of 1.1 percentage points. For the three largest scales, the compromise solutions were very similar and the values of the two objectives were within an average of 0.3 percentage points of each other.

Optimal management regimes

The two objectives differed substantially with respect to the forest management regimes that were optimal to them. Carbon storage was most often the highest under set-aside, while timber production was maximized by a combination of regimes, primarily continuous cover forestry, business-as-usual and business-as-usual without thinnings (Fig. 4). In the compromise solutions the distribution of management regimes was dominated by business-as-usual without thinnings, continuous cover forestry, set-aside, and to a lesser extent business-as-usual with extended rotation time (Fig. 4). When management was optimized at larger scales, more area was set-aside than at smaller scales (Fig. 4).

The share of 'land-sharing' management in the compromise solution decreased as the spatial scale of the management optimization was increased (Fig. 5). At most, when the scale was increased from the small holding scale to that of the entire region, the number of stands where a 'land-sharing' management was chosen decreased by 4%. This decrease was made up by an increase in the share of stands where only one of the objectives was prioritized. The share of 'win–win' outcomes, i.e., cases where the same kind of management maximized both objectives, was 2% and did not depend on the spatial scale of the analysis.

Discussion

In this study, we examined how the mitigation of ecosystem service trade-offs by land-use optimization is affected by the scale of the planning regions in Finnish production forests. Our results show that by optimizing management allocation, trade-offs between timber production and carbon storage can be mitigated, and that this is done the more effectively the larger the planning regions are—but up to a certain point. Optimizing over several stands was substantially more effective than selecting optimal management for each individual stand separately. Apart from individual stands compared with multiple stands, the largest improvements could be achieved by increasing the scale of the planning regions from approximately 10 stands to approximately 100 stands. Beyond this scale the improvements were minor.

The trade-off between timber production and carbon storage indicated by our results is consistent with previous findings from production forests (Schwenk et al. 2012; Triviño et al. 2015): maximal levels of the two objectives cannot be reached at the same time, but if even small losses in one are permitted, the

Table 2 The compromise solutions in terms of the two objectives in absolute values and relative to their maximal values

Scale	Timber production (million €)	Carbon storage (million tons)	Percent of maximal timber production (%)	Percent of maximal carbon storage (%)
Stand	331.46	8.70	76.59	77.33
Small holding	363.44	9.20	83.97	82.32
Large holding	362.17	9.39	83.68	84.00
Watershed	364.55	9.39	84.23	84.00
Region	365.99	9.39	84.56	84.00

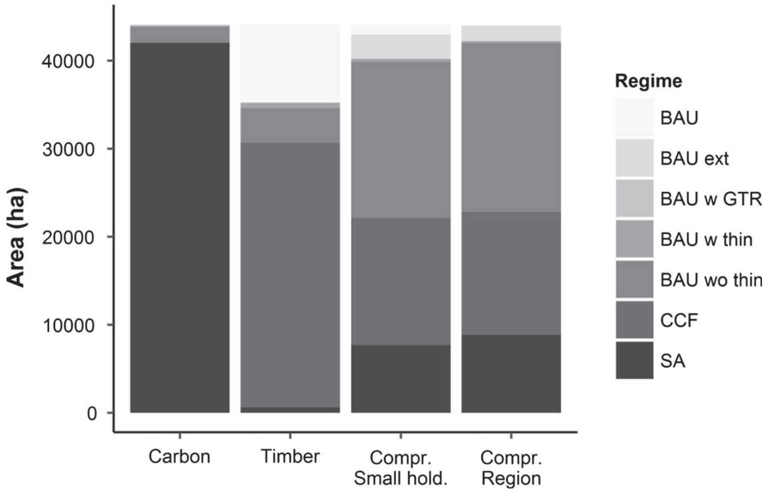

Fig. 4 The distributions of management regimes that maximize the two objectives ('Carbon' and 'Timber') or provide the compromise outcome (*Compr. Small hold.* compromise solution at the small holding scale, *Compr. Region* compromise solution at the regional scale). For visual clarity, the 16 modified versions of business-as-usual (see "Methods") have been grouped into 4 categories based on their defining features (extended rotation time, green tree retention, thinnings before final felling, or no thinnings). The abbreviations in the legend refer to: *BAU* business-as-usual, *BAU ext* business-as-usual with extended rotation time, *BAU w GTR* business-as-usual with green tree retention, *BAU w thin* business-as-usual with thinning before final felling, *BAU wo thin* business-as-usual without thinnings, *CCF* continuous cover forestry, *SA* set-aside

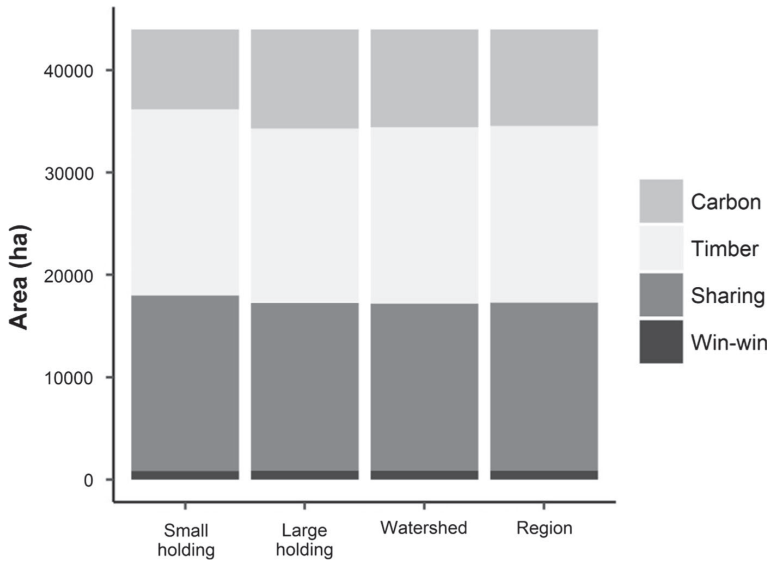

Fig. 5 The distributions of management regimes in the compromise solution characterized as prioritizing a single objective ('Carbon' or 'Timber'), producing a land-sharing outcome ('Sharing') or producing a win–win outcome where both objectives are simultaneously maximized ('Win–win') when optimal management allocation is determined at different spatial scales. The compromise solution refers to a solution where the losses in both objectives are minimized

supply of the other can be improved considerably. The trade-off was severe particularly in terms of timber production, which reached only 5% of its potential maximum when carbon storage was maximized. When timber production was maximized, carbon storage could reach 66% of its maximal value. The low value of timber production when carbon storage was maximized is due to the fact that the set-aside regime provided the highest level of carbon storage in a majority of the stands. This has been shown also in previous work (Triviño et al. 2015). By definition, the set-aside regime does not produce any income from timber production because no timber is harvested. At best, both objectives could simultaneously reach 82-84% of their respective maximums. These compromise outcomes were achieved by a diverse combination of alternative management regimes that differed clearly from management targeting only either one of the two objectives.

Several studies have shown that interactions among ecosystem services can be dependent of the scale of observation (e.g., Anderson et al. 2009; Raudsepp-Hearne and Peterson 2016; Hou et al. 2017). Likewise, we found that the severity of the trade-off between timber production and carbon storage, as indicated by the steepness of the production possibility curves, was affected by the spatial scale of the planning region. The curve was further from origin, i.e., better joint production outcomes of the two objectives could be achieved, when the planning regions were larger. We hypothesize this improvement is due to an uneven distribution of the potential supply of the two objectives, i.e., variation among stands in features affecting them. However, the improvements were notable only between the smallest scale and the three larger scales, while the differences between the three largest scales were very small. If the effect of the planning region's size stems from the amount of variation that is included within it, our results indicate that the variation in forest features that are relevant for the objectives considered here stops increasing beyond the scale of approximately 100 forest stands. Identifying the relevant features is beyond the scope of this study, but for example Triviño et al. (2015) suggested at least initial stand age and tree species distribution to affect forests' potential for joint provision of timber and carbon services. Gamfeldt et al. (2013) found tree species richness to have a positive or positively hump-shaped relationship with soil carbon storage and tree biomass production, with high proportions of birch and spruce in particular having a positive effect on both services.

We defined 'land-sharing' management of a stand as providing a less than maximal level of both of the objectives. This kind of management was more common as a part of the optimal solutions when the spatial scale of the optimization was smaller. In other words, the larger the planning regions were, the more stands were dedicated to the production of a single objective. This increased single-objective prioritization then enabled an improvement in the joint production of the two objectives over the entire study area. It is worth noting here that the concept of 'land-sharing' itself is scale-dependent (Ekroos et al. 2016). While optimizing management allocation at larger scales led to less 'land-sharing' management applied at the stand level, it enabled a better 'land-sharing' outcome at the level of the entire study area because higher levels of both objectives were achieved at the same time.

In short, our results indicate that the scale of 'large forest holding', consisting of approximately 100 stands, may be large enough to mitigate the conflict between timber production and carbon storage as efficiently as possible. This means that the 'correct' or at least sufficient spatial scale for analyzing and solving the management trade-off between timber production and carbon storage may be much smaller than the scale of thousands of stands that has been used in previous work (Triviño et al. 2015). The spatial scales we compared ranged in area from 17 ha to over 43,000 ha, and the increases in the utility of the optimization exercise that were gained from increasing the scale beyond some hundreds of hectares were minimal. Assuming larger-scale analyses are more time and resource intensive to conduct, studies like the current work can inform how resources for planning can be used most efficiently. However, in the current study we considered only the two objectives, yet production forestry and stand management practices may affect the supply of a range of other ecosystem services (Pukkala 2016; Pohjanmies et al. 2017). Moreover, ecosystem service management that is actually carried out is a product of not only ecological but also social and institutional factors.

In particular, if a larger range of ecosystem services is considered, the issue of finding the 'correct' scale for ecosystem service management may be greatly complicated by the fact that services may be the products of ecosystem processes taking place at different scales (Kremen 2005). For example, in our case carbon storage may be measured per any land unit irrespective of scale, but some other ecosystem services may be provided by processes that are linked to fixed spatial scales or connectivity patterns (Mitchell et al. 2015; Kukkala and Moilanen 2016), such as processes regulating water quality and supply (Brauman et al. 2007), the movements of mobile organisms (Kremen et al. 2007), or the experience of aesthetic landscapes (Gobster et al. 2007). In our study system, this means that the condition of areas covering several stands or interactions among stands may affect the supply of some ecosystem services and may need to be considered to secure high levels of all of them. Finding the optimal land-use allocation would then involve spatial optimization, which makes the problem more challenging to formulate and solve.

The ‘correct’ scale of ecosystem service management is further related to how the spatial scale of service provision matches with institutional and administrative scales (Hein et al. 2006; Schwerdtner Máñez et al. 2014). Mismatch among the scales at which ecosystem services are produced, at which their benefits accrue, and at which relevant management decisions are made may hinder their effective and equitable management. Even if the ecologically correct spatial scale of ecosystem service management is identified, it may be difficult to operate at it. For example, in forest stands timber production and carbon storage both depend on tree growth and are thus generated at the same spatial scale. However, they differ with respect to the institutional scales at which their benefits are realized: the benefits of timber production accrue at the forest ownership scale, while the benefits of carbon storage are global. They also differ with respect to the administrative scales where desired outcomes are defined: forest management for timber production, albeit influenced by regulations, is decided by the forest owner, while goals related to carbon storage and climate change mitigation are set at higher administrative scales. In a review of ecosystem service studies, Howe et al. (2014) concluded that when stakeholders are acting at different scales and have conflicting private and public interests in ecosystem services, trade-offs between the services are especially likely to occur. It should be noted, however, that also private forest owners may attach diverse, non-economic values and goals to their forests and seek to manage them accordingly.

Because of challenges concerning the practicality of optimizing land-use and implementing the resulting management plans, it may be beneficial to identify the smallest scales where such endeavors are maximally useful. Our results show that there are clear benefits to considering an area beyond a single stand or a few stands in forest management planning, but they also indicate that moderately small spatial scales, corresponding in area to a few typical Finnish forest holdings, are enough to provide the best possible compromise outcomes for timber production and carbon storage in Finnish production forests. They thus suggest that cooperation of only a few forest owners is required. This is tentatively encouraging for the practicality of real-world implementation of multiobjective optimization tools in landscape-level forest management planning. A more substantial obstacle may be lack of incentives for forest owners to target high levels of public non-timber benefits from their forests. For carbon storage, possibilities include payments to forest owners for carbon services or regulation of forest management practices to restrict carbon losses (e.g., Pohjola and Valsta 2007; Cao et al. 2010).

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IV

LOSS OF RESILIENCE AND MULTIFUNCTIONALITY IN A PRODUCTION FOREST

by

Tähti Pohjanmies, Kyle Eyvindson, Maiju Peura, María Triviño & Mikko Mönkkönen 2017

Manuscript

Loss of resilience and multifunctionality in a production forest

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Abstract

The management of production landscapes is increasingly recommended to target system resilience and multifunctionality: the sustained provision of high levels of multiple benefits and biodiversity. This may entail a switch of management focus from intensive exploitation to a more diverse set of approaches. However, demand for resources produced in these systems is also increasing, for example for forest biomass. Intensive resource exploitation places high pressure on ecosystems, and their ability to sustain ecosystem services and biodiversity under such conditions is uncertain. The ecological concepts of resistance and resilience provide a way to examine this ability. Resistance refers to a system's ability to avoid displacement due to a disturbance, and resilience to its ability to return to the state that preceded the disturbance. In this study, we used forest inventory data, forest growth simulations, and multi-objective optimization to create alternative future paths for a 2,200 ha production forest in Finland. In the alternative paths, the forest was managed for maximal timber production, for forest multifunctionality, or first timber production and then forest multifunctionality. We compared the paths to analyze the resistance and resilience of forest multifunctionality to intensive forestry and evaluated how the forest recovered after having been used for intensive forestry for varying lengths of time. Our results show the landscape to have high potential for multifunctionality. However, this potential was diminished by approximately half under intensive forestry. That is, multifunctionality was not resistant to intensive forestry. When the focus of management was changed from timber production to multifunctionality, multifunctionality started to increase, but it did not fully recover within the 100-year timeframe of the study and recovered the slower the longer intensive forestry had been continued. In other words, intensive forestry weakened the resilience of multifunctionality. The results suggest that intensive forestry not only reduces the supply of non-timber benefits but hinders their recovery once intensive forestry is terminated. They thus bring into question the long-term sustainability of intensive forest management as practiced today.

Keywords: Boreal forest; Finland; ecosystem services; forest management; optimization; resistance; stability; sustainability.

Introduction

In response to the global biodiversity loss crisis, progressing climate change, and growing demand for ecosystem services, the management of production landscapes is increasingly urged to target system stability and landscape multifunctionality (Bennet and Balvanera 2007). Ecosystems that are managed intensively for the production of food, raw materials, and bioenergy tend to become homogenized and simplified as a result of efforts to increase the efficiency of production (MEA 2005, Kareiva et al. 2007). This has led to loss of biodiversity, ecosystem services, and ecosystem stability (Folke et al. 2004, Thompson et al. 2011). The design of multifunctional landscapes and the development of management practices that maintain biodiversity and ecosystem functions have been proposed as a countermeasure to these harmful impacts. Calls have been made for studies and tools that take into account the temporal dynamics of ecosystems to inform management choices that are sustainable in the long term (Kremen 2005, Chan et al. 2006, Carpenter et al. 2009, de Groot et al. 2010, Heydinger 2016).

Stability of biological communities and ecosystem processes describes their sensitivity to perturbations through time. Resistance and resilience are two commonly differentiated aspects of stability. Resistance refers to the system's ability to avoid displacement due to a disturbance, and resilience to its ability to return to the state that preceded the disturbance (although overlapping definitions of the terms have been used; see e.g., Carpenter et al. 2001). While the concept of stability has its roots in basic ecology, it has become central in the study of socio-ecological systems (Holling 1973, Ives and Carpenter 2007, Biggs et al. 2012). Enhancing system resistance and resilience is important for sustainable ecosystem management where the goal is to maintain diverse communities and ecosystem functions over time, and in particular in the face of intensifying disturbances such as resource exploitation, pollution, and climate change (Bennet and Balvanera 2007, Seidl et al. 2016).

While in basic ecology the appropriate definition of stability may depend on the study system or community and the factors that may disturb it (Ives and Carpenter 2007), in sustainability research it is of interest to define it according to the desired condition of the socio-ecological system and the potential threats to it (Carpenter et al. 2001, Biggs et al. 2012). In the case of ecosystem services, the focus may be on the variability in the supply of ecosystem services in addition to or rather than the variability in the occurrence of species and habitats that provide the services (cf. Kremen 2005). For example, the stability of agroecosystems may be examined as the ability of the system to sustain food production in the face of changing circumstances (e.g., Lin 2011). Here, 'stability' is thus defined not as stability of the system's structure but as a lack of fluctuation in its output. Threats to the desired supply of ecosystem services include unexpected disturbances as well as slow, ongoing changes in conditions, and they may be natural or societal (Peterson 2000, Biggs et al. 2012, Heydinger 2016).

The Earth's land surface is dominated by intensively managed production ecosystems (Foley 2005, Creutzig 2017). Then again, the value of ecosystem services and biodiversity is increasingly recognized (Díaz et al. 2006, Mace 2014). For these reasons, the desired state of production landscapes is more and more often considered to be that of multifunctionality: the joint production of multiple ecological, environmental, social, and economic functions in a given land area. Landscape multifunctionality and ecosystem stability are linked because they are both thought to be built upon structural and biological diversity at different scales (Elmqvist et al. 2003, Fischer et al. 2006, Biggs et al. 2012). Additionally, the two concepts are linked by the temporal aspect: stability is ultimately the maintenance of functions over time, and landscape multifunctionality is often thought to cover fundamental ecological functions that increase system stability (Lovell and Johnston 2009).

Landscape multifunctionality is a concept that aims to reconcile potentially conflicting ecosystem services and conservation needs (Lovell and Johnston 2009, Reyers et al. 2012). Ecosystem services have been found to be commonly conflicting with each other so that ecosystem management to maximize one service causes losses in other services (MEA 2005, Rodríguez et al. 2006, Tallis et al. 2008). This has been found to be the case, for example, in intensively managed production forests, where humans modify the forest ecosystem with the aim of enhancing wood production (Duncker et al. 2012, Schwenk et al. 2012, Lutz et al. 2016, Triviño et al. 2017, Pohjanmies et al. 2017b). In this context, forestry activities such as tree harvesting can be considered as disturbances which the system either can or cannot absorb, respectively meaning that the supply of desired non-timber benefits is unaffected or is reduced. This corresponds to resistance. As the disturbance passes and the succession of the forest continues, the benefits may or may not return to their pre-disturbance levels. This corresponds to resilience.

Finnish forests are an example of an intensively exploited ecosystem with a recognized need of multifunctionality and resilience (Finnish Forest Research Institute 2011, Ministry of Agriculture and Forestry 2015). Finland has an extensive forest cover, around 73% of its land area (FAO 2015), and 86% of its total forest area – over 26 million hectares of forest – is under forestry use (Peltola 2014). Because of the extent of forestry, the preservation of forest biodiversity and ecosystem services in Finland depends on production forests (Kuuluvainen 2009). The dominant forestry system in Finland is based on clear-cut harvesting and regeneration of even-aged stands with optional operations including commercial thinnings and fertilization (Äijälä et al. 2014). Intensive forestry substantially alters the forest ecosystems, for example by homogenizing stand structure and tree species composition, reducing the amount of old-growth forests, and reducing the amount of dead wood in the forest (Siitonen 2001, Kuuluvainen 2009). The declines in resource and habitat availability caused by forest exploitation are considered to be the primary causes of biodiversity loss in Finnish forests (Rassi et al. 2010). The dominant forestry practices may also affect the supply of several ecosystem services (Pohjanmies et al. 2017a). Indeed, intensive forestry has been found to be in conflict with the provision of several non-timber ecosystem services in Finland (Triviño et al. 2015, Peura et al. 2016, Pukkala 2016, Vauhkonen and Ruotsalainen 2017, Pohjanmies et al. 2017b). The situation is similar across the boreal region, where extensive tracts of forests are under forestry use with impacts on crucial ecosystem services and biodiversity (Bradshaw et al. 2009, Moen et al. 2014, Gauthier et al. 2015, Pohjanmies et al. 2017a). However, while several studies have shown forestry activities to reduce the supply of non-timber benefits from forests, to our knowledge none have examined the ability of these benefits to recover from said activities.

In this study, we use the concepts of resistance and resilience to examine how the provision of multiple ecosystem services is sustained under and recovers from intensive management aiming to maximize the production of a single service. Our study area is a large production forest landscape in Finland. We use forest growth simulations and multi-objective optimization methods to create alternative future scenarios where our study forest is managed either for maximal timber revenues or for forest multifunctionality. By employing forest simulations we are able to explore the long-term outcomes of alternative management choices (Felton et al. 2017), and by switching the focus of management planning from timber revenues to multifunctionality after varying lengths of time we are able to examine both system resistance and resilience. Specifically, we aim to resolve the following questions: 1) When the forest is managed for timber production, how much multifunctionality is lost ('resistance')? 2) If the forest is first managed for timber production for varying lengths of time and after that for multifunctionality, how is the achievable multifunctionality affected ('resilience')?

Methods

Study area and forest growth simulations

Our study area is located in Central Finland. It covers approximately 2,200 ha and consists of 1,475 forest stands of varying age, site type, and tree species composition (Figure 1). The current age of the stands ranges between zero and 125 years with an average of 45 years, and the most common tree species are pine (*Pinus sylvestris*), spruce (*Picea abies*) and birch (*Betula pendula* and *Betula pubescens*). We used stand-level forest inventory data for our study area produced by the Finnish Forest Centre as input data in the forest growth simulator SIMO (Rasinmäki et al. 2009) to create projections of the growth and development of the stands under alternative forest management regimes. Using forest growth models, SIMO produces projections of future stand development based on site type, initial conditions, and forestry operations applied to the stand. Use of forest growth simulations and the projections produced by them is common in forestry planning to inform management choices (Kangas et al. 2015).

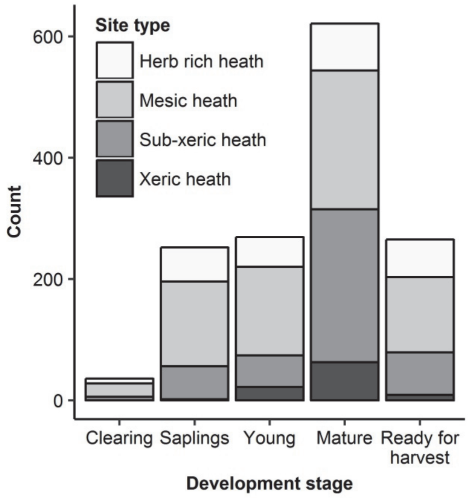

FIGURE 1 Distribution of the forest stands included in the study area classified into present development stages (x axis) and site types (bar fill). The development stage 'Young' refers to a stand with an average diameter at breast height of 8–16 cm, and 'Mature' to a stand with an average diameter at breast height greater than 16 cm but that is not yet ready for final harvest. The development stage 'Ready for harvest' is defined according to current Finnish recommendations for the timing of final felling (Äijälä et al. 2014). Site types are presented according to the Finnish forest classification system (Hotanen et al. 2008) and range from fertile mixed-species stands (herb rich heath) to less fertile, pine-dominated stands (xeric heath).

We simulated the development of the stands under alternative management regimes, which were intended to cover a wide range of different ways to conduct and time management operations and harvests. These included rotation forestry based on clear-cut harvesting of even-aged stands (Äijälä et al. 2014), continuous cover forestry based on selective harvesting (Pukkala et al. 2012), and protection or set-aside of the stand. Rotation forestry with clear-cut harvesting is the currently dominant mode of timber production in Finland. It consists of commercial thinnings, a clear-cut felling timed to achieve the stand's maximal growth, and regeneration of the stand after the clear-cut. We created several versions of rotation forestry with varying frequencies of thinning, amounts of tree retention, and rotation lengths for our simulations to reflect real-world variation in the implementation of the regime. Continuous cover forestry differs from rotation forestry in that it is based on regular, selective harvesting of large trees instead of a one-time clear-cut felling. Finally, if the stand is set-aside no forestry activities are carried out but the stand is left to develop naturally. We used a simulation period of 20 years and simulated the stands for five such periods, a total of 100 years into the future. Each 20-year simulation period was divided into five-year time steps, with the simulator producing predictions of stand development at each time step as output.

Ecosystem services and biodiversity features

We produced estimates of the stand-level supply of six objectives based on the structure and properties of each stand. These included five ecosystem services: timber production, carbon storage, bilberry yield, cowberry yield, and scenic beauty. In addition, we considered one biodiversity feature, availability of deadwood resources. These objectives were chosen based on their relevance in Finland to people and nature as well as on the availability of data and models required to evaluate them. Because all of the objectives are modeled based on the structure of the stand, they are affected by the same drivers, i.e. the forest management activities conducted in the forest, and the analyses should thus capture when they are truly co-produced rather than just co-occur.

Timber production was measured as net present value of harvest revenues (NPV, €). NPV consisted of revenue from wood harvested during the planning period and expected revenue from standing timber and bare soil at the end of the planning period, minus the costs of silvicultural operations. Revenues and costs were based on recent stumpage prices and costs (Peltola 2014). Future revenues were discounted using a moderate 1 % interest rate and were always discounted to the start of the planning period, so that the planning periods would be directly comparable with each other. To illustrate the result of the applied management and to discern the components of net present value (specifically, the value of harvested wood from the value of standing timber), we also recorded the amount of harvested timber (m^3) during each 20-year planning period as reported by the simulator.

Forest carbon storage has a critical role in global climate regulation and climate change mitigation (Pan et al. 2011). Carbon storage (kg) was measured as the sum of the predicted amounts of carbon fixed in living wood, dead wood and soil. Carbon fixed in living and dead wood was estimated as 50 % of the wood biomass. To estimate soil carbon, we used two models depending on the soil type: the Yasso07 models (Liski et al. 2005, Tuomi et al. 2009, 2011) for mineral soils, and the carbon flux models of Ojanen et al. (2014) for peatland soils. Bilberry (*Vaccinium myrtillys*) and cowberry (*Vaccinium vitis-idaea*) are the two most common wild berries in Finland with both high commercial and recreational value (Vaara et al. 2013). Bilberry yield (kg) was estimated using the models of Miina et al. (2009) and cowberry yield (kg) using the models of Turtiainen et al. (2013). Both predict species coverage and berry yield based on stand characteristics (e.g. dominant tree species, stand age, and stand basal area). Forests dominate the landscape in Finland, and forest structure impacts on their perceived scenic beauty and recreational use (Silvennoinen et al. 2001, Gundersen and Frivold 2008). Scenic beauty (no unit) was measured by the index developed by Pukkala et al. (1988), which estimates the recreational and aesthetic attractiveness of a forest based on forest age, structure and tree species composition.

Finally, we included the availability of deadwood resources as a biodiversity feature because lack of deadwood resources is estimated to be the most common cause of species endangerment in Finnish forests (Tikkanen et al. 2006, Rassi et al. 2010). In addition, there is strong evidence of deadwood as an indicator of broad biodiversity

(Gao et al. 2015). Availability of deadwood resources was measured as the total amount of deadwood (m^3) multiplied by the diversity of different deadwood types, comprising different tree species and decay stages (Triviño et al. 2017). Diversity was measured with the Simpson diversity index.

Except for NPV, we calculated the average levels of the objectives across the time steps of the simulation periods and these averages were used in the analyses. The stand-level values were multiplied by stand area and summed together to produce landscape-level values.

Multifunctionality

We measured the ability of the forest area to maintain high levels of all ecosystem services and biodiversity as forest multifunctionality. We defined forest multifunctionality as a condition where all ecosystem services and biodiversity features are simultaneously as close as possible to their potential maximal values. The definition is analogous to the 'compromise solution' found for conflicting management objectives described by Mazziotta et al. (2017). We calculated the potential maximal levels of the objectives by simulating the development of the stands for 100 years into the future under the alternative management regimes and identifying their maximal achievable levels during that time. To maximize forest multifunctionality, a management plan was identified for the study area where a management regime was selected for each stand so that the loss in the total level of each individual objective (ecosystem services and biodiversity) across the landscape was minimized. A loss in an objective from its maximum under a management plan was calculated as

$$loss = \frac{max_{tot} - \sum_i^n x_{i,j}}{max_{tot}}$$

where x_i is the value of the objective in stand i, n is the total number of stands, j indicates a management regime selected for stand i under the management plan, and max_{tot} is the potential maximum of the objective.

Maximal multifunctionality was then found by solving the optimization problem:

$$\text{minimize } \max(loss_1, loss_2, \dots, loss_k)$$

where k is the number of objectives. When multifunctionality was maximized, $k = 6$ after the six objectives described above. When management was planned to target timber production, forest multifunctionality was maximized under the constraint that timber production (NPV) reached its maximal value (now, $k = 5$ as timber production is considered separate from multifunctionality). This maximal value of NPV was calculated separately for each 20-year planning period, always discounting to the start of the planning period. The optimization model was created using the Pyomo software (Hart et al. 2012) and solved with the IBM ILOG CPLEX optimizer, version 12.6.2

(https://www.ibm.com/developerworks/downloads/ws/ilogcplex/). As a quantitative measure of multifunctionality (*MF*), we used

$$MF = 1 - \max(loss_1, loss_2, \ldots, loss_k)$$

where $max(loss_1, loss_2, \ldots, loss_k)$ is the value found by the optimization model. This measure directly shows how large a portion each objective reaches of its potential maximum. For example, if MF = 0.5, each objective reaches at least 50% of its potential maximal level.

Resistance and resilience

In order to examine the resistance and resilience of forest multifunctionality to forestry-related disturbances, we designed a simulation tree of alternative future paths where each node represents a choice between targeting maximal timber production or maximal multifunctionality (Figure 2). Choices were made at 20-year time steps, representing realistic management planning timeframes (Kangas et al. 2015). If forest multifunctionality was targeted in a given planning period, it was also targeted in all following periods. If timber production was targeted in a planning period, in the following period a choice was again made between timber production and multifunctionality. The consecutive planning periods formed a total of six alternative future scenarios, or paths (Table 1).

The resistance of forest multifunctionality to intensive forestry was examined by comparing multifunctionality that is achievable under timber-focused management and under multifunctionality-focused management (Figure 2). If timber-focused management causes a loss of multifunctionality as compared with multifunctionality-focused management, multifunctionality is not resistant to intensive forestry. Similarly, the resilience of multifunctionality was evaluated by comparing its value between consistently multifunctionality-focused management and multifunctionality-focused management following timber-focused management (Figure 2). If multifunctionality remains lower after the forest has been used intensively than when the forest has been consistently managed for multifunctionality, it is not resilient to intensive forestry.

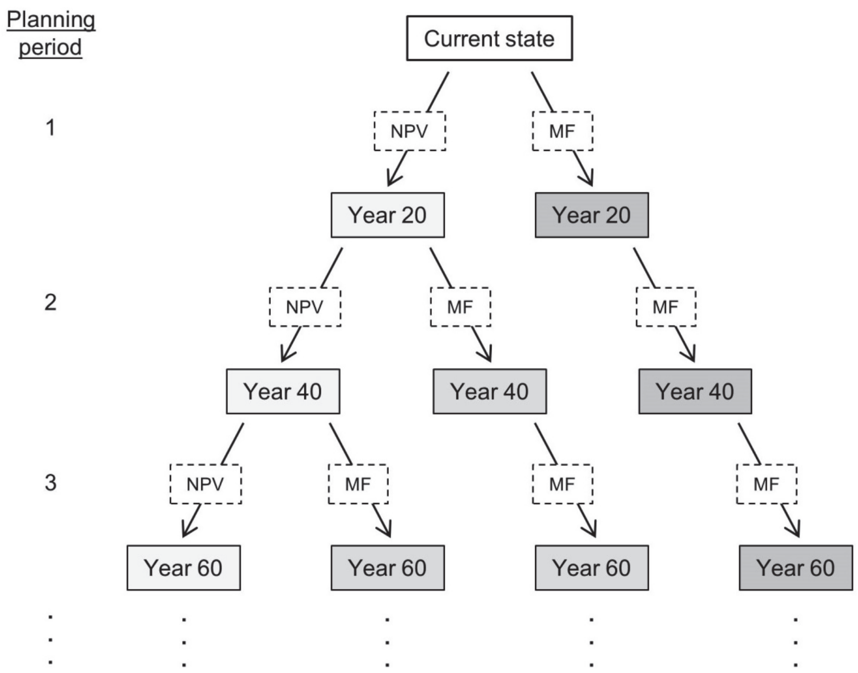

FIGURE 2 The simulation tree designed to create alternative future scenarios used to examine system resistance and resilience. Each arrow represents a step where the development of the forest is simulated for 20 years into the future (one planning period) under a range of alternative management regimes, out of which the set is then identified by multi-objective optimization that maximizes either timber production ('NPV) or forest multifunctionality ('MF.').

TABLE 1 Alternative future paths composed of consecutive planning periods where either timber production ('T') or forest multifunctionality ('MF') is targeted.

Path name	**Target in period 1**	**Target in period 2**	**Target in period 3**	**Target in period 4**	**Target in period 5**
MF5	MF	MF	MF	MF	MF
T1MF4	T	MF	MF	MF	MF
T2MF3	T	T	MF	MF	MF
T3MF2	T	T	T	MF	MF
T4MF1	T	T	T	T	MF
T5	T	T	T	T	T

Lastly, to examine multifunctionality-focused and timber-focused forest management in terms of the forest landscape they create, we compared the landscape-level forest structure between the alternative paths and over the planning periods. We examined forest structure in terms of the mean and the variation of stand age and stand basal area at the end of each planning period. Stand age was calculated as the average age of trees in the stand. Stand age and stand basal area represent different aspects of stand structure but are both directly affected by the silvicultural activities carried out in the stand and, in general, are the lower the more timber is harvested from the stand.

Results

Forest multifunctionality

When managed consistently for multifunctionality, the study forest showed high potential for the joint production of all of the objectives: after 40 years of multifunctionality-focused management, multifunctionality was above 0.6, that is, all objectives could reach over 60 % of their potential maximums (dark grey dots in Figure 3). When management consistently prioritized timber production, forest multifunctionality varied between 0.2–0.3 across the planning periods (white dots in Figure 3). When the focus of the management planning was switched from timber production to multifunctionality between planning periods, the values of multifunctionality landed between the extremes of these two paths, being the lower the longer the forest had been managed with a timber production focus (Figure 3).

When management consistently targeted multifunctionality, better outcomes were reached most of the time also in terms of individual objectives as compared with targeting timber production. When multifunctionality was the focus, carbon storage, scenic beauty and deadwood availability reached higher values than under timber-focused management in every planning period (Figure 4E-G) and bilberry yield in all but the first planning period (Figure 4C). Even NPV reached higher values under multifunctionality-focused management than under timber-focused management after the first planning period (Figure 4A). However, in terms of harvested wood, timber-focused management always led to higher levels of timber production than multifunctionality-focused management (Figure 4B). This indicates that under multifunctionality-focused management much of the calculated value of NPV consisted of standing timber that was never harvested (i.e., growing stock). In the NPV calculations the income gained from harvests in earlier planning periods did not carry over to following planning periods but the value of standing timber did. This is likely why the NPV of the forest became higher under multifunctionality-focused than timber-focused management. In addition to harvested wood, cowberry yield reached higher values under timber-focused management than under multifunctionality-focused management (Figure 4B).

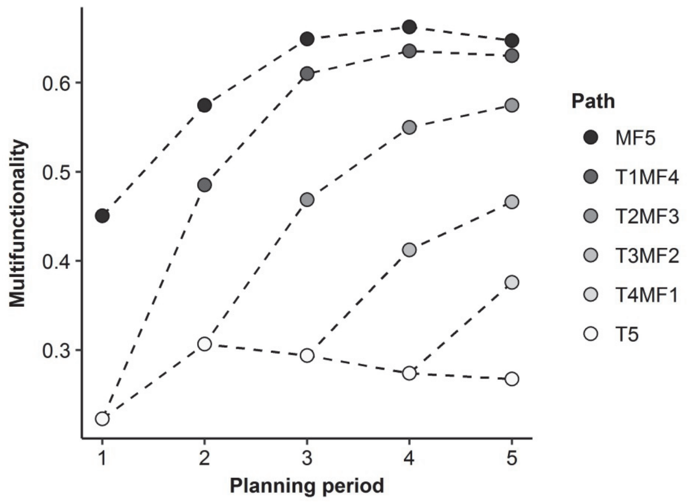

FIGURE 3 Forest multifunctionality across time under the alternative management paths. In the abbreviations in the legend, 'MF' refers to multifunctionality, 'T' to timber production, and the numbers following the letters to the number of planning periods in which multifunctionality or timber production was targeted. For example, 'T1MF4' means timber production was targeted in the first planning period and multifunctionality in the remaining four periods. The dashed lines have been added to connect the points to visualize the progression of the alternative paths.

Resistance and resilience

The difference in multifunctionality between multifunctionality-focused management and timber-focused management was considerable with multifunctionality being approximately twice as high under the former as under the latter. This means multifunctionality was not resistant to intensive forestry. When the management focus was switched from timber production to multifunctionality, forest multifunctionality increased, that is, began to recover. However, multifunctionality did not reach values as high as under consistent multifunctionality-focused management in any of the paths where the forest had been first managed with a timber production focus (Figure 3). That is, there was no full recovery of multifunctionality in the 100-year timeframe of the study. In addition, the longer the timber-focused management had continued the lower were the values of multifunctionality achieved in all of the remaining planning periods and thus the longer it took for multifunctionality to recover (Figure 5). For example, when management focus was changed to multifunctionality after two planning periods of timber-focused management, multifunctionality was 0.57 in the third planning period since the change (path T2MF3; Figure 5). When the management change was preceded by only one period of timber-focused management, multifunctionality was higher, 0.61, already in the second period since the change (path T1MF4; Figure 5). The decrease in the rate of recovery caused by timber-focused management suggests a loss of resilience.

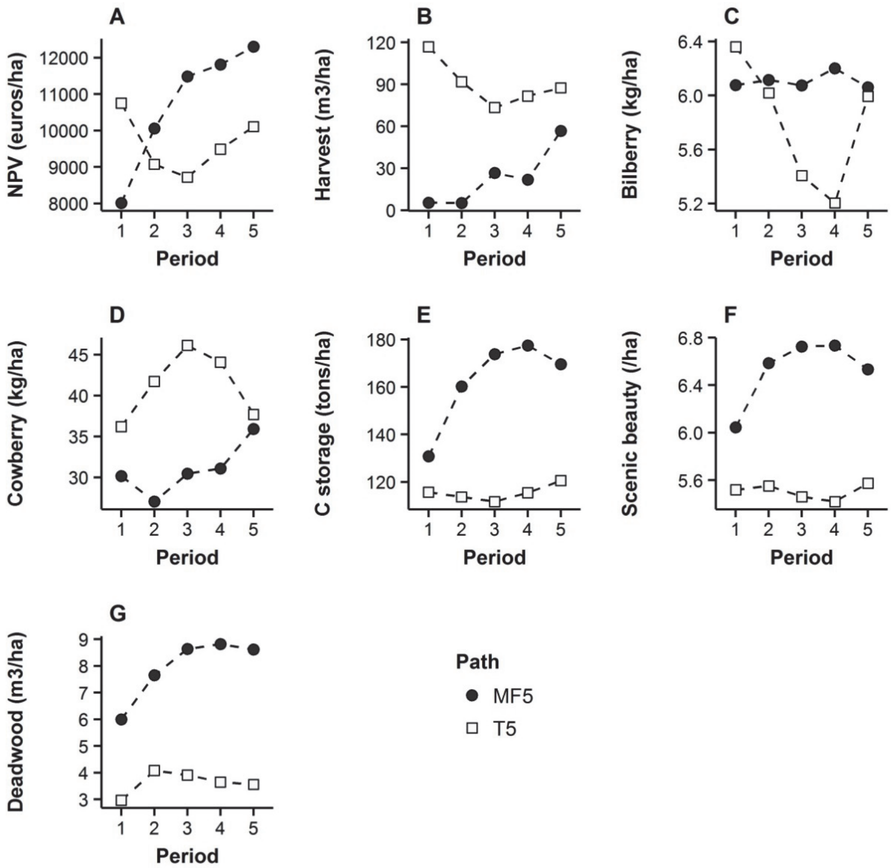

FIGURE 4 Values of the six objectives included in the analyses as achieved under five consecutive planning periods of multifunctionality-focused management (path 'MF5') or timber-focused management (path 'T5'). Net present value (NPV; panel A) and harvested wood (Harvest, panel B) are both measures of timber production. Shown are the per-hectare values averaged across the study area. NPV and harvested wood are calculated over each 20-year period. For the other objectives, yearly averages across the 20-year periods are shown. The dashed lines connect the points in order by time for graphical comparison.

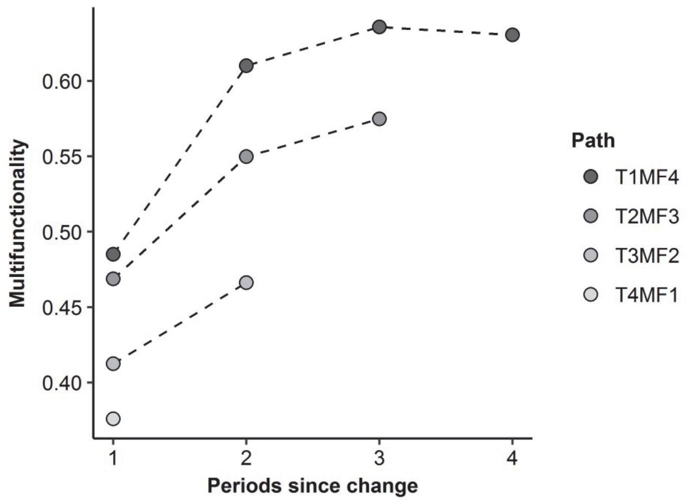

FIGURE 5 Forest multifunctionality across time under four alternative management paths in which the focus of management planning was changed from timber to multifunctionality. Shown are the multifunctionality values that were achieved after the management change, and the x-axis shows the number of planning periods that has passed since the change. In the abbreviations in the legend, 'MF' refers to multifunctionality, 'T' to timber production, and the numbers following the letters to the number of planning periods in which multifunctionality or timber production was targeted. For example, 'T1MF4' means timber production was targeted in the first planning period and multifunctionality in the remaining four periods, and the multifunctionality values achieved under these four periods are shown on the plot. The dashed lines have been added to connect the points to visualize the progression of the alternative paths.

Development of the forest landscape

In all planning periods the average stand age and basal area were higher under multifunctionality-focused management than under timber-focused management (Figure 6). The average stand age as well as its standard deviation increased over time in all of the alternative paths (Figure 6A-C). The same was true for average basal area (Figure 6D-F) except for the last planning period of multifunctionality-focused management, when the average basal area decreased slightly from the previous period (Figure 6D), and for the last planning period of timber-focused management, when the standard deviation of basal area decreased from the previous period (Figure 6F). A change of management focus from timber production to multifunctionality led to a sharp increase in average stand age and basal area (Figure 6B, E), suggesting that the recovery of multifunctionality was achieved by management that left more timber in the forest than timber-focused management.

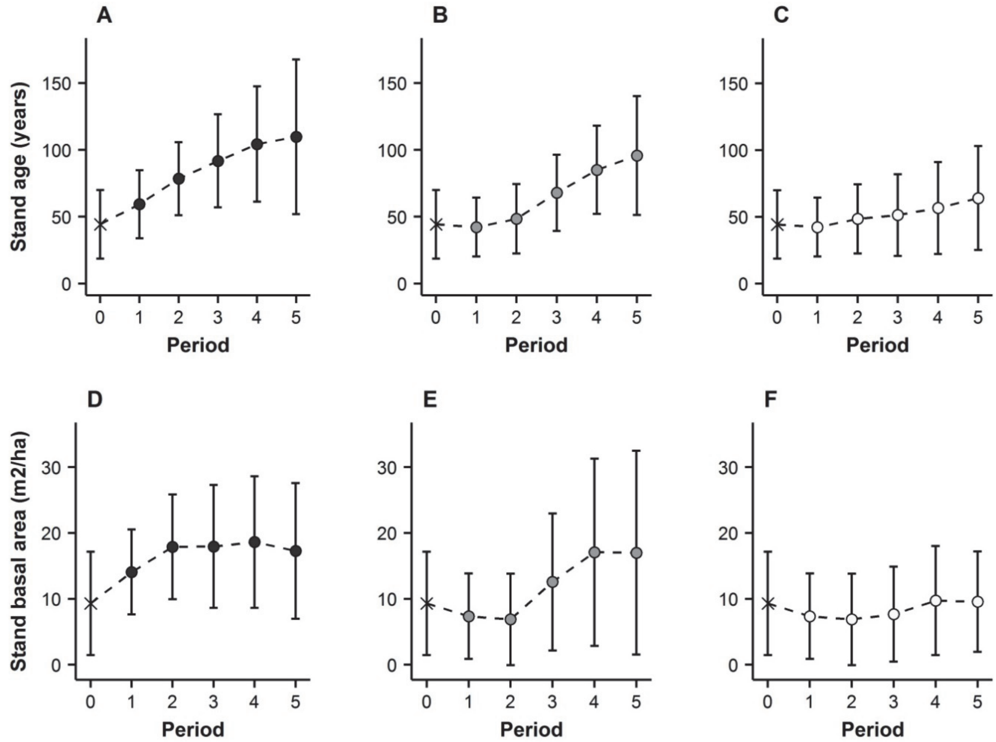

FIGURE 6 Top row: mean stand age ± standard deviation over the five planning periods under alternative management paths: A) MF5, B) T2MF3, C) T5. Bottom row: mean stand basal area ± standard deviation over the five planning periods under alternative management paths: D) MF5, E) T2MF3, F) T5. In path MF5, multifunctionality was targeted in every planning period, and in path T5 timber production was targeted in every planning period. In path T2MF3, timber production was targeted in the first two planning periods and multifunctionality in the following three periods. In all plots, the cross and error bar at period '0' shows the initial condition of the study area for comparison.

Discussion

As the various ecosystem services provided by production landscapes become recognized, the management of these landscapes is recommended to target multifunctionality and stability: the sustained provision of high levels of multiple benefits and biodiversity. However, demand for the resources produced in these systems is also increasing, for example for forest biomass. The ecological concepts of resistance and resilience offer a way to examine the ability of production landscapes to maintain the provision of diverse benefits under intensive management and exploitation. In this study, we used forest growth simulations and multi-objective optimization to create alternative future paths for a forest landscape where the forest was managed for maximal timber production, for forest multifunctionality, or first timber production and then forest multifunctionality. By comparing these paths we were able to measure the resistance and resilience of forest multifunctionality to intensive forestry. Our results showed the study landscape to have high potential for multifunctionality. However, this potential was diminished under intensive forestry, did not fully recover within the 100-year timeframe of the study, and recovered the slower the longer intensive forestry was continued.

We considered five ecosystem services (timber production, bilberry production, cowberry production, carbon storage and scenic beauty) and one biodiversity feature (availability of deadwood resources) as components of forest multifunctionality. We measured forest multifunctionality by first calculating the potential maximal supply of every objective in the landscape and then comparing their simultaneously achievable levels on their respective maximums. When forest management consistently targeted forest multifunctionality, all objectives could reach up to over 60 % of their potential maximums at the same time. Under timber-focused management, the corresponding number was 20–30 %. The clearly lower multifunctionality under timber-focused management than under multifunctionality-focused management shows that multifunctionality is not resistant to intensive forestry. Multifunctionality-focused management was more favorable than timber-focused management to all individual non-timber objectives except for cowberry production. The results indicate a strong conflict between production forestry and non-timber forest ecosystem services and biodiversity, as has been found in earlier work (Duncker et al. 2012, Schwenk et al. 2012, Mönkkönen et al. 2014, Peura et al. 2016, Triviño et al. 2017, Eggers et al. 2017, Pohjanmies et al. 2017b).

We used two measures of timber production: the economic value of the forest (net present value, NPV) and amount of harvested wood. NPV was used as the objective in the forest management optimizations, and the amount of harvested wood was examined subsequently to illustrate the outcome of management determined to be optimal. Under multifunctionality-focused management the economic value of the forest was higher than under timber-focused management, but the amount of harvested timber was lower. It appears that under multifunctionality-focused management NPV consisted for a large part of standing timber that was expected to be harvested later but then never was. This was likely influenced by the 1 % discount rate used – a higher discount rate would have put even more weight on actual income as compared with expected income. We speculate that with a higher discount rate or if the amount of harvested timber had been used as the measure of timber production in the optimization analyses, the achievable multifunctionality would have been lower than measured now in all scenarios because the simultaneously achievable levels of the different objectives would have been lower. The limited timber harvests under multifunctionality-focused management are also illustrated by the structure of the forest associated with multifunctionality-focused management as compared with timber-focused management: multifunctionality-focused management led to a forest with a clearly higher average stand age and basal area. A change of management focus from timber to multifunctionality, in particular, led to a sharp increase in stand age and basal area, implying a sharp decrease in timber harvests. This further demonstrates the conflict between forestry, specifically timber harvesting, and non-timber benefits.

In all scenarios, multifunctionality increased in time after the focus of management was changed from timber to multifunctionality, indicating that forest multifunctionality was recovering. However, it always remained lower than in the reference scenario of constant multifunctionality targeting. Even with one planning period, or 20 years, of timber-focused management followed by four planning periods, or 80 years, of multifunctionality-focused management, multifunctionality did not fully recover. The result that achievable multifunctionality was and remained the lower the longer the forest had been managed with a timber production focus indicates a loss of resilience that became increasingly consequential as intensive forestry was continued. A loss in the resilience of forest multifunctionality caused by intensive forestry adds another dimension to the previously shown conflict between timber harvesting and non-timber forest benefits and raises questions about the long-term sustainability of intensive forestry.

The results of our study also raise a point about the selection of the reference state used in studies comparing ecosystem management alternatives. For example, indices designed to quantitatively measure a system's resistance or resilience to disturbances are commonly based on the difference in the value of a response variable between a disturbed system and a control system (several such indices presented, e.g., by Orwin and Wardle 2004). In our study design, the forest managed with a timber production focus corresponds to the 'disturbed' system and the forest managed with a multifunctionality focus to the 'control' system. In the paths where the management focus is changed, the intensively managed, 'disturbed' forest becomes the 'control' after the change and could be compared with the intensively managed forest for the remaining planning periods. However, the multifunctionality value of the 'control' is negatively affected by the intensive forestry that preceded the change in management and, as described above, recovers increasingly slowly. The longer timber-focused management is continued, the lower the multifunctionality that is achievable in the following planning period is, and the smaller the difference between timber-focused management and multifunctionality-focused management becomes. If resistance or resilience were evaluated by examining the difference between timber-focused management and multifunctionality-focused management at a given point in time but not taking into account the management history of the forest used as control, the results could indicate a misleadingly small loss in multifunctionality caused by intensive forestry. In studies targeting ecosystems such as Finnish forests with long histories of human impact and slow natural succession rates, choices regarding the reference state and the timeframe of the study may become crucially important to the conclusions drawn from the results (Kuuluvainen 2002, Gossner et al. 2014, Ghazoul et al. 2015).

In this study, we examined the resistance and resilience of forest multifunctionality to intensive forestry. Besides intensifying resource use, production landscapes are faced with the uncertainties caused by climate change, pollution, and loss of biodiversity. These issues raise concerns over the long-term maintenance of ecosystem services, the resistance and resilience of the ecosystems to natural disturbances, and the stability of the resource production itself (Millar et al. 2007, Lindner et al. 2010, Isbell et al. 2015, Seidl et al. 2016). In particular, regulating services interact with other types of ecosystem services, and maintaining the former may increase the stability of the latter (Bennett et al. 2009). In this study, regulating services contributing to forest growth or maintaining forest structures (for example, maintenance of soil fertility, pest control, or resistance to abiotic disturbances) were not considered; however, they may also be affected by forestry activities (Pohjanmies et al. 2017a). The resistance and resilience of the forest were analyzed only in terms of the response of forest multifunctionality to intensive forest exploitation and not, for example, to natural disturbances. Including natural disturbances at varying intensities in the forest growth simulations and exploring the responses of the system under the influence of both natural and anthropogenic pressures would add further depth to the analysis and perhaps more realistically reflect a future of growing uncertainties (Bennet and Balvanera 2007, Millar et al. 2007).

Overall, our results show that intensive forestry not only reduces the supply of non-timber benefits but hinders their recovery once intensive forestry is terminated. Demand for diverse ecosystem services is growing globally (MEA 2005) as well as from Finnish forests specifically (Ministry of Agriculture and Forestry 2015). Ecosystem management for landscape multifunctionality may entail a switch of management focus from intensive exploitation to a more diverse set of approaches (Fischer et al. 2006), and our results suggest that, if high forest multifunctionality is desired, such a change should not be delayed. The supply of non-timber benefits should be considered as an equally important objective as timber production in forest management planning already now in order to secure it also in the future. Because of the uncertainties related to future climatic, ecological, and socio-economic conditions combined with the long timeframes with which forests develop and respond to management activities, it is of great importance to understand and promote the resilience of these ecosystems (Chapin et al. 2007, Reyer et al. 2015). Our approach of using step-wise simulation and management optimization could be developed further as well as repeated in different areas where such tools are available in order to improve the understanding of forest multifunctionality and stability.

Acknowledgements

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