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The value of retained Scots pines and their dead wood legacies for lichen diversity in clear-cut forests: The effects of retention level and prescribed burning

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A B S T R A C T

To mitigate the negative impact of intensive forest management on biodiversity, several silvicultural measures are used in production forests. We studied the effect of two such treatments, green-tree retention and prescribed burning on epiphytic lichens 11 years after timber harvests in Scots pine (*Pinus sylvestris*) dominated forests of middle boreal Finland. Lichen assemblages were sampled from retained Scots pines and their dead wood legacies (snags and logs) below a height of 2 m. Twelve study sites were harvested with two retention levels (10 and 50 m^3/ha) and six uncut old-growth sites were used as controls. Prescribed burning was applied to half of the sites immediately after harvest. A total of 42 and 65 lichen species was found in the burned (B) and unburned (UB) sites with 10 m^3/ha retention, 41 and 74 in the B and UB sites with 50 m^3/ha retention, and 63 and 80 in the B and UB controls respectively. Mean species richness per tree did not differ between the control and harvested sites at either retention level on any of the substratum types. However, microlichen species richness was significantly lower on harvested sites on both living trees and snags. The species richness of both micro- and macrolichens was lower on burned sites. In addition, the composition of lichen assemblages differed between burned and unburned sites. The negative effect of fire was significant for lichens growing on living trees and snags, but not on logs. We conclude that the retention of Scots pines is a successful measure for maintaining epiphytic lichen richness over a timescale of 11 years after harvest, and that roughly equal numbers of living trees and their dead wood legacies are needed to support the lichen species richness on harvested sites. Given that the negative effect of prescribed burning is relatively longer for lichens than for other wood-dependent organisms, burning should not be applied to stands with particularly rich lichen communities. However, these conclusions may have limitations as our inventory was limited to the lowermost part of tree trunks only. Hence inventories of whole trees are useful to assess the full effect of prescribed burning. Our results also imply that the specific impact of prescribed burning for specialist lichen species should be evaluated over a longer timescale.

1. Introduction

Intensive forestry has caused drastic changes in northern boreal forests through the creation of even-aged, structurally simplified forest stands, which contain a low amount of dead wood and old trees and have largely replaced structurally diverse late-successional forests (e.g. Esseen et al., 1997; Siitonen, 2001; Cyr et al., 2009). In addition, efficient fire suppression has shifted natural forest dynamics, resulting in a reduction of biological legacies created by fire (Esseen et al., 1997; Granström, 2001). These changes in habitat availability and quality have led to a rapid decrease of biodiversity in production forests. As the proportion of protected forests is relatively low in both Europe and North America (Schmitt et al., 2009; Anonymous, 2010), measures to maintain species diversity in production forests are also required for efficient conservation of biodiversity.

One such measure to benefit biota that has become common in various regions of the world is tree retention at clear-cutting (Franklin et al., 1997; Gustafsson et al., 2010, 2012). Another relevant tool, though not so widely applied, is prescribed burning of harvested stands to imitate the effects of forest fire, for instance enhancing the regeneration of deciduous trees and creating certain

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structural elements, such as charred and decaying wood, which are valuable for various organisms (e.g. Weber and Taylor, 1992; Esseen et al., 1997; Siitonen, 2001; Vanha-Majamaa et al., 2007). Creation of these fire-related habitats is particularly important in regions, such as Fennoscandia (Kouki et al., 2012) where naturally occurring fires, which previously have been frequent especially in pine-dominated forests, have virtually disappeared (Zackrisson, 1977; Linder and Östlund, 1998; Granström, 2001).

Lichens are a valuable component of the forest biota (Ellis, 2012) inhabiting various types of substrata. Species-rich and unique lichen assemblages have, in particular, been documented on substrata characteristic of old-growth forests, such as old trees and coarse woody debris (e.g. Kuusinen, 1996a; Kuusinen and Siitonen, 1998; Lie et al., 2009; Moning et al., 2009; Lõhmus and Lõhmus, 2011). However, logging effectively removes such substrata (Siitonen, 2001; Lõhmus and Kraut, 2010). In addition, given that their dispersal capacity is low, increased distances between suitable habitats may limit the occurrence of several old-growth-related lichens in younger and managed stands (Sillett et al., 2000b; Hilmo and Såstad, 2001). Therefore, commercial forestry has been proposed as the most important threat causing the decline of many lichen species (Thor, 1997). For example, in Finland forest management is considered the main threat to 42% of all threatened lichen species (Jääskeläinen et al., 2010). To mitigate this negative development, it is important to evaluate the effectiveness of different conservation techniques in production forests. In addition, as lichens represent sessile organisms that often inhabit substrata with slow natural dynamics, such as old living trees, they complement the results from studies of polypore fungi and beetles, which in contrast inhabit mainly dead wood.

Several studies have shown that retention of biological legacies, like living and dead trees in harvested stands can provide habitats for specialized epiphytic lichens (Lõhmus et al., 2006; Rosenvald and Lõhmus, 2008; Caruso et al., 2011; Gustafsson et al., 2013; Lundström et al., 2013; Runnel et al., 2013), maintain small lichen populations to function as dispersal sources in the regenerating stand (Sillett and Goslin, 1999; Hedenås and Hedström, 2007; Lõhmus and Lõhmus, 2010) and provide habitats for species adapted to sun-exposed conditions (Lundström et al., 2013). In the European boreal region, these results are based mainly on data from the common aspen (*Populus tremula* L.) that has rather distinct lichen assemblages (Kuusinen, 1996a) and occurs mainly in mesic forests. Retention effects of living and dead Scots pines (*Pinus sylvestris* L.) on lichen biota are poorly documented, even though dry pine-dominated forests are widespread in Fennoscandia. We found only one study from adjacent Estonia, which showed that wood-dwelling lichens prefer pine snags (i.e. standing dead trees) rather than aspen and birch (*Betula* spp.) snags in a regenerating stand, and that their species richness increased with bark loss and time since tree death (Runnel et al., 2013).

So far, the impacts of prescribed burning on maintaining biodiversity on harvested sites in commercial boreal forests have been mainly assessed for polypores and beetles over the short-term. Among these groups are certain species, such as pioneer decayers and pyrophilous species, that are favoured by burning after harvest. However, among Red-Listed and rare species, only saproxylic beetles receive an immediate benefit from burning (Hyvärinen et al., 2005; Junninen et al., 2008; Hjältén et al., 2010a; Berglund et al., 2011). For epiphytic lichens, the impact of prescribed burning after harvest has not yet been evaluated. Naturally, the direct short-term effect on species richness and abundance is negative, as epiphytes are destroyed in the fire (Vanha-Majamaa et al., 2007). Still, they may survive in unburned patches, bark crevices or on tree crowns above the flames (Wolseley and Aguirre-Hudson, 1997). However, their survival is greatly affected by the intensity of the fire (Hauck, 2011) and also by the population size of a given species (Johansson et al., 2006). As an indirect effect, fire creates unoccupied space for colonization, such as snags and logs for epixylic species (Johansson et al., 2006) and burned wood and ground for many others (Lõhmus and Kruustük, 2010).

Our aim was to examine the long-term (over 10 years) effects of tree retention and prescribed burning on the species richness and assemblage composition of epiphytic lichens. The effects of the treatments were assessed separately for micro- (crustose species) and macrolichens (foliose and fruticose species), as these two species groups may have different habitat requirements and responses to disturbance (e.g. Ellis and Coppins, 2006; Johansson et al., 2006; Hedenås and Hedström, 2007). We used a large-scale and long-term replicated field experiment to investigate lichens on retained living pines and their dead wood legacies in middle-boreal Finland. Our study sites were harvested with two levels of tree retention, and half of the sites were burned after harvest. Unburned and burned old-growth stands were used as a reference. Based on the results, we evaluated whether retention trees and their dead wood legacies, alone or combined with prescribed burning, could be used to promote epiphytic lichen diversity in boreal pine-dominated commercial forests.

2. Materials and methods

2.1. Study area and experimental design

Field experiments were carried out near Patvinsuo National Park in the community of Lieksa, in eastern Finland (approximately 63°N, 30°E, with an elevation above sea level ranging between 150 and 200 m). The study area is situated on the border of the southern and middle boreal zones (Ahti et al., 1968). The study sites are a part of a large-scale field experiment that was initiated in 2000 (e.g. Hyvärinen et al., 2005; Junninen et al., 2008). A total of 18 study sites, each 3–5 ha in size, were included in this study (Appendix A). Prior to treatments, all sites were approximately 150 year old, sub-xeric forest stands dominated by Scots pine. The sites had not been intensively managed, but some selective harvesting has taken place during the late 1800s or early 1900s. Forest structure and tree volume did not differ significantly between sites before application of the treatments (Hyvärinen et al., 2005). The distance between the different study sites varied, but with the exception of a few sites, it was always greater than 1 km (Appendix A).

We made the assumption that the distance between sites was far enough to maintain spatial independency within the factorial design layout. Although there is no way to prove this statistically (due to the rather low number of sites), our assumption is backed by the fact that we have not been able to find any indication of spatial dependency of the sites, based on the initial forest structure or assemblages of several taxa that have been explored previously (e.g. Hyvärinen et al., 2005, 2009; Junninen et al., 2008; S. Johnson, J. Strengbom, J. Kouki, unpubl. data on ground and field layer vegetation). However, the study landscape may have natural features that do not occur in other regions and that may limit the generality of the results (see Kouki et al., 2012), but this is a problem shared with almost all experimental field studies.

Experimental treatments (harvests and post-harvest burnings) were randomly assigned to the study sites. Treatments were combinations of two factors: green-tree retention and burning. Two different green-tree retention levels, 10 m^3/ha and 50 m^3/ha were included in this study. Both of these were replicated on six study sites. Prescribed burning took place after harvest on half of the sites of each retention level. Six uncut old-growth forest sites, of which half were also burned, were used as controls. The sites were harvested in winter 2000–2001. During the harvest standing dead