Rehabilitating boreal forest structure and species composition in Finland through logging, dead wood creation and fire: The EVO experiment

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Abstract

This paper reviews an ongoing, large-scale multidisciplinary experiment designed to study the possibilities of rehabilitating forest structure and species composition through logging, dead wood creation and fire in managed Norway spruce (Picea abies) forests in southern Finland. These forests have been utilized for several centuries with intensive management and clear-cut harvesting, which has been the dominant practice in Finland since World War II. During this era, the forest structure has become relatively even-aged, and the amount of dead wood has been reduced considerably. Simultaneously, due to an effective fire suppression policy, the role of fire in Finnish nature has been almost completely eliminated. One of the key species in biodiversity, aspen (Populus tremula), has also been actively removed from the forests in the past. Forest restoration activities, such as the creation of dead wood and the reintroduction of fire to forest management, have been suggested in conservation and restoration programmes. So far we have studied the immediate effects of restorative actions on forest structure, regeneration, soil nutrient status, understorey and epixylic vegetation, lichens and beetles. In the larger Evo research area we have also studied the population structure of aspen in both protected and managed forests. Our early results show that it is possible, through active forest restoration, i.e. the creation of dead wood and prescribed burning, to rehabilitate boreal forest diversity, even when a significant part of the wood volume is harvested for commercial use. Despite the fact that the immediate effects of fire on many species groups were negative, the long-term effects are expected to be predominantly positive. There is currently a decline in aspen populations in Finnish forests. The absence of large aspens in managed forests and the absence of younger trees/cohorts in conservation areas, combined with high mortality, is a significant threat to aspen-dwelling species. We conclude that studies on active restoration treatments, together with long-term inventories of several species groups, are necessary in order to assess the impacts of varying restoration practices for cost-efficient large-scale applications.

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1. Introduction

In Fennoscandia, forestry-related degradation of natural forest habitats, and a subsequent high number of red-listed species, are well-documented phenomena (e.g. Linder and Östlund, 1998; Kouki et al., 2001; Rassi et al., 2001). In the forest environment, intensive forest management and the resulting changed forest structure, a lack of deciduous tree species, dead wood and fire are the key elements and main reasons for the decline in biodiversity (Ohlson et al., 1997; Granström, 2001; Kouki et al., 2001; Siitonen, 2001; Kuuluvainen et al., 2002; Gandhi et al., 2004; Kouki et al., 2004; Hyvärinen et al., 2005; Jonsson et al., 2005).

Dead wood microsites are characteristic to natural stands (Kuuluvainen and Laiho, 2004), and serve as habitats for several species and species groups (Kruys et al., 1999; Uotila, 2004; Laaka-Lindberg et al., 2005). The amount of coarse woody debris (CWD, dead trees larger than 10 cm at DBH) in

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the natural forests of southern Finland averages 60–90 m$^3$ ha$^{-1}$ (Siitonen, 2001), but in managed mature $Picea$ abies stands only 6.8–14.4 m$^3$ ha$^{-1}$ (Siitonen et al., 2000; Jalonen and Vanha-Majamaa, 2001), and in southern Finland in general as little as 1.2–2.9 m$^3$ ha$^{-1}$ (Tomppo et al., 1999). The serious discrepancy between the amount of dead wood in natural and in managed forests has been recognized. When new silvicultural policies were introduced in Finland during the 1990s, one of the goals was to increase the amount of CWD by increasing tree retention in loggings. Leaving retention trees soon became an accepted practice with a level of 5–10 retention trees per ha in managed stands (Vanha-Majamaa and Jalonen, 2001). However, these levels are too low to bring about any substantial benefit for biodiversity (Vanha-Majamaa and Jalonen, 2001; Penttilä et al., 2004; Halpern et al., 2005; Jonsson et al., 2005). A new target was recently set for protection of an average of 30 m$^3$ ha$^{-1}$ during the next 20 years (Anon., 2004; Hokkanen et al., 2005). However, research on the effects of dead wood increment on biodiversity is still scarce.

Fire plays an important role in boreal forest ecology, the effects of fire depending on fire severity (Viro, 1974; Lynham et al., 1998). Very light fires may have no major effects, fires of medium severity may have beneficial effects, i.e. decreased acidity and more plant-available nutrients, and severe fires may additionally have harmful effects, i.e. high loss of soil organic matter and nitrogen. Fire changes the succession of the tree stand by favouring pioneer tree species, especially deciduous trees (Weber et al., 1995; Vanha-Majamaa et al., 1996; Lampainen et al., 2004; Lilja et al., in preparation). Prescribed burning with partial cutting has been shown to enhance regeneration after disturbance (Vanha-Majamaa et al., 1996) and, in general, controlled fire can be used to create structural elements, such as charred and decaying wood, that are important for biodiversity (Esseen et al., 1997; Granström, 2001; Bergeron et al., 2002). Forest fires are generally regarded as detrimental and a threat to the environment in boreal ecosystems, where millions of hectares may burn annually, especially in Canada and Russia (FAO, 2001). However, in some regions, such as in Fennoscandia, the situation is rather different because nowadays forest ecosystems are almost totally excluded from the effects of forest fires.

Before there was any significant human impact in Fennoscandia, forest fires were relatively rare but usually affected larger areas than is currently the case (Niklasson and Granström, 2000). Slash-and-burn cultivation was widely practiced for centuries, thereby increasing the fire frequency (Niklasson and Granström, 2000; Pitkänen et al., 2003; Huttunen, 1980; Tasanen, 2004). After slash-and-burn cultivation ceased around the beginning of the 1900s, prescribed burning became increasingly used as a management tool. This activity reached its culmination in Finland during the 1950s and 1960s, when the annual prescribed burning area varied between 15,000 and 35,000 ha (Metla, 2003). Since then, mainly due to the development of alternative soil preparation methods, there has been a steady decline and the annual prescribed burning area is nowadays around 500–2000 ha. At the same time, wild fire prevention and control became more effective and the area burned annually decreased considerably (Vanha-Majamaa et al., 2004). The reduction in the annually burned area has recently led to suggestions for increased use of prescribed burning in forest certification criteria and forest conservation programmes (Lindberg and Vanha-Majamaa, 2004). However, the anticipated increase has not occurred (Anon., 2003), and in fact the use of prescribed burning has further declined during the last few years (Metla, 2005). The effects of fire are currently very restricted in Finnish forests, as only 500–600 ha are burned annually in wildfires, and the average fire size is less than 1 ha (FAO, 2006).

The aim of forest restoration is to rehabilitate natural structures, processes, and species composition in ecosystems altered by human actions (Bradshaw, 1997). A number of different approaches, based on the exploitation of natural disturbances and which aim at the development of sustainable forestry practices (Fries et al., 1997; Angelstam, 1998; Keeley and Stephenson, 2000; Bergeron et al., 2002; Nordlind and Östlund, 2003), can be regarded as restoration in its wide sense.

In general, natural disturbance dynamics can be used as a guideline to define forest restoration goals and methods that enhance natural habitat variability in forest structures (Haila, 1994; Vanha-Majamaa and Jalonen, 2001; Franklin et al., 2002; Kuuluvainen et al., 2002; Lampainen et al., 2004).

One aim of forest restoration in Finland is to introduce conditions favouring deciduous trees, which in Finland are mainly pioneer species. Special interest in restoration activities is focused on European aspen ($Populus$ tremula). Aspen hosts several hundred different lichen, bryophyte, polypore, and beetle species (Siitonen, 1999). Old, decaying aspens especially are known to form numerous niches during the dying and decaying process both as standing and fallen trees in different microclimatic conditions (Kuusinen, 1994; Siitonen, 1999; Siitonen and Martikainen, 1994; Martikainen, 2001). Recent studies have shown a significant change and decline in aspen population structures, not only in managed forests, where the long-lasting removal of aspen from the stands has occurred, but also in conservation areas as a consequence of increased moose browsing and lack of natural disturbances (Kouki et al., 2004; Latva-Karjanmaa et al., 2007).

Besides retaining natural forest structures, the restoration and conservation management of threatened species is one key topic in current nature protection. The causes of rarity may be numerous (Kotiaho et al., 2005), and the restoration and maintenance of rare species requires extensive knowledge of the autecology of the target species and their habitat requirements (Kalamees et al., 2005).

Examples of important indicator species groups are beetles – one of the best-known insect groups in Fennoscandia (Siitonen, 2001) – and polypores (Penttilä et al., 2004). Beetles are ecologically an especially diverse group and include a large number of highly specialized species, thus making them valuable tools for assessing the biological value of boreal forests and the consequences of management and restoration methods (Hammond et al., 2004; Hyvärinen et al., 2005). As a consequence of intensive forest utilization, the loss and fragmentation of old-growth forests and subsequently reduced
amounts of CWD, and the absence of fire, the populations of saproxylic, i.e. dead wood dependent species especially have greatly declined (Rassi et al., 2001; Grove, 2002). The short-term effects of forest fires are naturally deleterious to many species groups, such as wood-rotting fungi, especially polypores (Penttilä and Kotiranta, 1996) and epixylic (i.e. growing on dead wood) bryophytes and lichens (Ryömä and Laaka-Lindberg, 2005), but the short-term effects of forest fires can also be positive (e.g. Hyvärinen et al., 2005). However, practically nothing is known about, for example, the long-term effects of dead wood creation by fire on epixylic bryophytes and lichens.

In fact, research on forest restoration in general is still rather limited, and there is a clear need for forest restoration research in the boreal zone (Esseen, 1994; Rydgren et al., 1998; Vanha-Majamaa and Jalonen, 2001; Kuuluvainen et al., 2002), especially simultaneous studies with several species groups. Some large-scale multidisciplinary experiments to fulfil these research needs are, however, already being carried out, such as EMEND, SAFE, DEMO, etc. (Aubry et al., 1999; Harvey et al., 2002; Gandhi et al., 2004; Hyvärinen et al., 2005), and common databases, such as NOLTFOX (Nordic and Baltic database for long-term forest experiments, http://noltfox.metsa.fi) are being developed.

The most widely applicable restoration actions combine multiple, both restoration and silvicultural objectives. Thus, in economically feasible forest restoration part of the wood may be harvested to cover the costs of restoration actions without abandoning the initial objectives (Vanha-Majamaa et al., 1996; Lilja et al., 2005). Following this approach, we aimed at determining whether it is possible to restore natural post-disturbance structural characteristics in managed stands by partial cuttings, combined with dead tree creation and fire in an experimental research project EVO. The project is divided into several sub-studies, and the study areas have been reserved for research purposes for the next 20–30 years.

The general aim of this article is to introduce the study approach and to discuss some of the early findings of our project in relation to restoration ecology. We have so far concentrated on studies on stand structure, regeneration of the tree stand and the effects of different microhabitats on regeneration, succession of ground and field layer vegetation, changes in organic layer and mineral soil, effects of logging and fire on epixylic species, their colonization after disturbance, effects of fire on beetles, dead wood dynamics and polypores.

One special aim was to study aspen regeneration and population structure.

2. Experimental setup

2.1. Study area

All the studies were performed in the Evo-Vesijako area, southern Finland, which is a large, mostly publicly owned, relatively uninhabited forest area. The Evo area has a long tradition in ecological research and, in recent years, biodiversity and restoration actions have been one of the key interests in the area (e.g. EU EvoLife project).

The study area (61°N, 25°E) is located in the southern boreal zone (Ahti et al., 1968). The mean annual temperature is +3.1 °C and the duration of the growing period is 160 days. The annual average precipitation is about 670 mm. The bedrock consists of orogenic granitoids and is covered with a thick layer of till (Anon., 1995).

The restoration experimental sites were located in mature Norway spruce-dominated managed forest stands (area 1–3 ha) of the mesic site type. Most of the stands were of the Vaccinium myrtillus site type (MT), but five stands also had characteristics of the Oxalis myrtillus site type (OMT) (Cajander, 1926; Pählsson, 1995). The stands were of a mixed tree species composition, including Betula pendula, B. pubescens Roth., P. tremula L., and Pinus sylvestris L. In addition, Sorbus aucuparia L. and Juniperus communis L. occurred in the sapling layer. Each stand consisted of an upland and a paludified upland biotope. The upland biotopes belonged to the MT site type. The vegetation and moisture conditions of the paludified upland biotopes varied considerably, and consisted of patches of paludified MT site type and spruce mire (Laine and Vasander, 2005).

Altogether 24 stands were selected for the restoration experiment and the treatments were randomized among the stands. The treatments consisted of three levels of down wood retention (DWR) and a partial cutting with a constant volume of 50 m³ ha⁻¹ of standing dispersed retention trees, and a fire treatment applied in half of the stands (Table 1). In addition, burned and unburned stands without cutting treatments were used as reference stands. Each treatment was replicated three times. Inside each treatment, and both in the upland and paludified upland biotope, sample plots 20 m × 40 m in size were placed randomly for further sampling. The total number of sample plots in the experiment was 48 (Lilja et al., 2002) (Table 1).

The restorative cuttings were conducted in spring 2002, and all the burnings during summer 2002 (Fig. 1). The burnings were carried out using the traditional Finnish prescribed burning technique (Lemberg and Puttones, 2003). With this method, the ignition lines form a circle around the stand and the burning front advances partly against the wind, thus decreasing the risk of fire escape.

Table 1  

<table>
<thead>
<tr>
<th>Treatment</th>
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<th>Replicates</th>
<th>Unburn</th>
<th>Replicates</th>
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<td>3</td>
<td>5 m³/ha DR</td>
<td>3</td>
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<td></td>
<td>50 m³/ha SR</td>
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<td>50 m³/ha SR</td>
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<tr>
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<td>30 m³/ha DR</td>
<td>3</td>
<td>30 m³/ha DR</td>
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</tr>
<tr>
<td>Reference</td>
<td>No cuttings</td>
<td>3</td>
<td>No cuttings</td>
<td>3</td>
</tr>
</tbody>
</table>

Total 24 forest stands. DR: down retention; SR: standing retention (Lilja et al., 2005).
2.2. Inventories and preliminary results

2.2.1. Stand structure

Pre-treatment inventories of the stand structure were made during the field season of 2001, and post-treatment inventories in 2002–2005. A detailed description of stand structure was made for all tree layers, including both living and dead trees. The diameter, height, crown characteristics and the location of each tree (height >2 m) were measured (Lilja et al., 2005). Tree seedlings by species (<2 m), their height, location, and microhabitat, were measured as well (Lilja et al., in preparation).

The pre-treatment volume of living trees did not differ between the upland biotopes and the paludified ones. The pre-treatment volume of *Picea* was significantly higher on the upland biotopes, and the volume of *Betula* was higher on the paludified biotopes (Lilja et al., 2005).

The treatments naturally reduced the number of living trees and increased the amount of CWD in relation to the planned CWD levels (Table 1, Lilja et al., 2005). The most effective burning result was achieved with the highest level of down wood retention. Without down wood retention, the fuel load was not high enough to create a severe fire and cause high mortality among the standing trees (Lilja et al., 2005; Fig. 1). The cutting treatments increased dead wood-related and sheltered microsites, while fire greatly increased the cover of even ground. The treatments increased the number of *Betula* seedlings in both biotopes, but especially in the paludified biotopes with an intermediate CWD level immediately after restoration (Lilja et al., in preparation). On the reference plots, 4 years after the treatment the number of *Betula* seedlings was much higher in the upland biotopes after fire than without fire (Fig. 2). *Picea* seedlings occurred more abundantly in the unburnt upland biotopes than in the burnt ones, whereas in the paludified biotopes their number was approximately equal in both treatments. The number of other deciduous trees, mainly...
Sorbus aucuparia, was higher in the unburnt treatments than in the burnt ones (Fig. 2; Nijp, 2006).

2.2.2. Understorey vegetation

Understorey vegetation was inventoried in 24 quadrates (size 50 cm × 50 cm) in every forest stand before and after the treatments (total 576 quadrates) (Järvinen, 2004). The post-treatment inventories were carried out on a yearly basis.

In the understorey vegetation after fire, there were clear differences in the fire damage between the biotopes. The dry biotopes burned relatively well, whereas unburned patches remained in the wet biotopes (Kujala and Toivonen, 2004). Even in the dry biotopes the plant species composition, e.g. in the moss layer, affected the burning of the organic layer in similar (40–70%) moisture conditions (Fig. 3), creating a mosaic of burned and unburned patches. The decrease in organic layer thickness due to fire was the highest with a Pleurozium schreberi cover, and differed significantly compared with Dicranum sp. and Hylocomium splendens cover, indicating, e.g. morphological differences between the species (Järvinen, 2004; Järvinen et al., in preparation).

After fire there were notable differences in the understorey vegetation between the biotope types and fire severity classes. Succession has been fast, and approximately half of the species present prior to the treatments reappeared already 2–3 years after the treatments. In addition to these species, several species typical to early successional stages after fire have appeared in the vegetation community. Ceratodon purpureus and Funaria hygrometrica are among the most common bryophyte species colonising burnt and exposed soil, and Marchantia polymorpha is one of the first hepatic species colonising moist, burnt soil. Common forest ground layer species, such as Hylocomium splendens and Pleurozium schreberi, are very slow in recolonising, and were mainly absent during the first years after the fire (Järvinen, 2004; Ryömä and Laaka-Lindberg, 2005; Järvinen et al., in preparation).

2.2.3. Soil

The effects of fire on the organic layer were studied by measuring the changes in the organic layer thickness and weight, in acidity and in total and extractable nutrient concentrations. The organic layer was sampled in August 2001 before experimental fire and in August 2002 after burning treatment.

The organic layer was sampled by taking two composite samples at 16 sampling points with a cylinder (d = 58 mm). During the sampling after burning, fire intensity was estimated at every sampling spot using a scale: (0) not affected by fire, (1) field layer has burned, (2) bottom layer (mosses, lichens) has burned, (3) 1–25% of organic layer has burned, (4) 25–90% and (5) over 90% of the organic layer has burned.

The samples were air-dried at 40 °C and ground in a mill with a 2 mm bottom sieve. Samples were analysed for pH in water (10 ml of sample:25 ml of water), acid ammonium acetate (pH 4.65; 15 ml of sample to 150 ml of extractant) extractable and total (dry combustion + HCl extraction) nutrient concentrations, and for total carbon and nitrogen (dry combustion with the Leco 1000 CHN). Element concentrations in the total and extractable extracts were determined by ICP/AES.

Because of weak correlations between the burning class and fire-induced changes in the chemical variables, statistical analyses were performed only the values before and after fire, irrespective of the fire intensity. The change was calculated as...
the relative difference, \(100 \times (\text{value after fire} - \text{value before fire})/\text{value before fire}\) \((\text{value after fire})/\text{value before fire}\) \((\%)\) (Alamo Carrasco, 2003).

After fire the observed decrease in the organic layer thickness was generally small (on average 11 mm), but there was a significant decrease in the amount of organic matter and carbon. The amount of organic matter and carbon decreased by 13\%. Burning increased the pH by 0.4 pH units. There were clear relative changes in the element concentrations (Fig. 4).

The calcium concentrations seemed to increase the most due to fire. The increase was statistically significant also for other elements, except for the concentration of extractable phosphorus. The concentrations and amounts of total nitrogen decreased. The variation in fire-induced changes between the plots was high (Alamo Carrasco, 2003; Tamminen et al., in preparation).

The increase in nutrient concentrations and weights must be due to the concentrating effect of the ash of the burned ground vegetation, litter and needles and twigs of trees.

2.2.4. Epixylic bryophytes

The pre-treatment inventories of epixylic communities were carried out in autumn 2001 and spring 2002. Parameters depicting the CWD quality, including tree species, log length and diameter, decay stage and bark coverage, were recorded on the sample plots. The species composition, cover of colonies of individual species and total epixylic vegetation, also including lichens on logs, were recorded. Post-treatment inventories were made during the field seasons 2002–2005. Approximately 800 permanent plots of 10 cm \(\times\) 20 cm in size were established on the logs in order to follow the development of the epixylic vegetation after the restoration treatments. In order to obtain a rough estimate of the effects of cuttings and burning on the establishment probabilities, the colonization of epixylic bryophytes on dead wood was studied using spore traps (micro slides coated with Vaseline, see e.g. Miles and Longton, 1992).

Spore trapping after the restoration treatments was conducted during 16 trapping periods in summers 2002–2004. Each spore-trapping period lasted 24 h, and the numbers of spores/1 cm\(^2\) were counted and identified.

Fire completely destroyed the vegetation cover on the logs. Especially treatment 3 (Table 1) was very destructive for epixylic bryophytes in terms of total coverages of the species. The total species richness observed on the study plots prior to fire was relatively low, only 40 species, including 16 hepatics and 24 mosses (Fig. 5). The total species richness declined by 22.5\% after the fire treatments. The reduction in species richness was higher in the upland biotopes (31\%) than in the paludified biotopes (22\%, Fig. 5). Establishment and growth of bryophyte protonema was observed on two burnt logs already in 2003, 1 year after prescribed burning in the study sites. The protonema could not be identified to the species level at that time but, in summer 2004, the species was identified as the moss *Ceratodon purpureus*. Five bryophyte species were observed on charred wood 3 years after the fire. Furthermore, the preliminary results of the spore trapping study show great temporal variation in the numbers of bryophyte spores. The effects of the restoration treatments on the amount and quality of spore deposition also seem to be clear (Ryömä and Laakala-Lindberg, personal observation).

2.2.5. Beetles

No pre-treatment inventories were conducted on beetles, but the post-treatment inventory was conducted in 2002–2004. Beetle diversity was inventoried by sampling beetles with flight-intercept traps. The traps consisted of two crosswise-located transparent plastic panes with a funnel and container below them. Saline water with detergent in the containers was used to preserve the beetles. To ensure random samples, the traps were set hanging on a string between two trees or poles. Five traps were set at random locations on each study plot, giving a total number of 120 traps.

The majority of the beetles caught (99.9\%) were identified to the species level. The beetles were classified into saproxylic and non-saproxylic species. In addition, we formed a group consisting of rare and red-listed species. A species was classified as rare if it had 25 or less occurrences in Finland according to the frequency score list of Finnish beetles (Rassi, 1993). The beetle species considered threatened (IUCN
categories CR, EN and VU) or near threatened (NT) in Finland (Rassi et al., 2001) were included in the group as red-listed species.

The total species richness of beetles was higher after both burning, logging and dead wood creation, as well as logging and dead wood creation alone, compared to the control. However, the effect of the latter was due to the fact that, in general, the harvested plots have more species than unharvested plots, while the level of volume of dead wood created had no effect on species richness (Toivanen and Kotiaho, 2004, 2007). Most importantly, rare and red-listed species also benefitted from burning and logging with dead wood creation (Fig. 6). This was due to the responses of saproxylic species in this group: the number of rare and red-listed non-saproxylic beetles did not differ between the treatments (Toivanen and Kotiaho, 2007). Species assemblages were strongly modified by burning and logging, and there were also additional differences according to the volume of dead wood created (Toivanen and Kotiaho, 2007).

2.2.6. Aspen

The study on the population structure of aspen was partly carried outside the restoration treatment in the larger Evo research area (Ahola, 2005; Frances Palacin, 2006; Lindberg et al., in preparation). In order to study the population structure of aspen in managed stands, all aspen trees (n = 4079) were surveyed over an area of 1500 ha in Evo during 2001–2006. The stand structure changes and the mortality of aspen during a 10-year period were studied in the Kotinen Old Growth Nature Reserve, where a 25 ha-sized permanent sample plot established in 1995–1996 was re-inventoried in summer 2005. Burned restoration sites were used in the aspen sowing experiment, in which aspen seeds were sown in different substrates and microsites (de Chantal et al., 2005).

The average density of aspen (dbh > 7 cm) in managed forests in the Evo area was 2.7 individuals/ha (Ahola, 2005). The stem frequency series show the absence of larger trees (Fig. 7) and the abundance of smaller trees in managed stands. In spatial analysis, larger aspens were found to have aggregated in certain locations, e.g. in damp depressions, under cliffs and near forest compartment borders (Ahola, 2005; Lindberg et al., in preparation).

In the Kotinen Old Growth Nature Reserve the density was 50.2 aspens/ha (Ahola, 2005). The stem frequency series differed from that of managed forests, and was normally distributed and showed the absence of younger aspen trees. During the 10-year period, the mortality of aspen trees in Kotinen was 13% (Ahola, 2005; Frances Palacin, 2006).

In the aspen sowing experiment, seedling establishment was highest on exposed mineral soil, but seedling establishment was also noted on burned moist substrate. No seedling establishment was found on burned organic substrate on drier upland sites, nor on any unburned substrates (de Chantal et al., 2005). On all types of substrate early establishment was, however, noted to be unpredictable and sensitive to weather conditions.

Inventories were also carried out in the research area of wood-rotting fungi (polypores and a few corticioid fungi), CWD, tree mortality (Sidoff et al., 2007), the threatened vascular plant *Pulsatilla patens* (Kalamees et al., 2005), soil microbes (Jaatinen et al., 2004) and bark beetle dispersion risk (Eriksson et al., 2006), but the results are not included in this review.

3. Discussion

3.1. Effects on trees and soil

The results presented here are very preliminary ones concerning the responses to the restorative treatments. Only long-term monitoring will reveal the overall effects of the treatments. However, our early results suggest that it is possible, through the use of logging, dead wood creation and prescribed fire, to rapidly restore structural diversity in ecologically impoverished managed stands, even when a significant proportion of the wood volume is harvested for economic use (Lilja et al., 2005).
The amount of downwood retention can be used to regulate the severity of fire according to ecological restoration objectives (Lilja et al., 2005). Our experiment also demonstrates the important role of small, within-stand paludified biotopes in creating a small-scale mosaic of burned and unburned patches, which is often the case after natural wildfires, and is known to be important as species refugia and for the subsequent colonization of burned areas (Van Wagner, 1983; Vasander and Lindholm, 1985; Granström and Schimmel, 1993; Hörnberg et al., 1998; Vanha-Majamaa and Jalonen, 2001; Wallenius et al., 2004). Regeneration of the tree stand was clearly different in these paludified patches (Lilja et al., in preparation), probably leading to relatively different stand structure development, thereby affecting also the species composition of many species groups. Fire alone also clearly changed the tree species composition by increasing the regeneration of deciduous trees.

It should be noted, however, that the experimental burnings did not completely burn all of the upland sites either. Therefore, heterogeneity or patchiness of the burning in some cases made it difficult to comparatively sample each treatment stand (increased sampling error, not shown here). For example, systematic soil sampling levelled down the plot-wise values, because the weights of non- or lightly-burned sub-samples were systematically larger than those of the severely burned sub-samples. Therefore a stratified sampling for the soil and understorey vegetation might have produced more accurate and more detailed information about the effects of fire.

The decrease in the thickness and organic matter content in the organic layer was low compared to the results of some other studies after fire (Dyrness and Norum, 1983; Lynham et al., 1998). This may have been due to the relatively un-windy conditions during burning. The rather small changes in the acidity and nutrient concentrations and amounts may also be explained by the relatively lighter overall burning. The rise in pH (0.4 pH units) was low compared to that reported for many other prescribed burning sites (Viro, 1974).

The nutrient concentrations and amounts increased due to fire, except for nitrogen, and the amounts of both total and extractable nutrients increased. It was expected that easily soluble nutrient concentrations would be increased due to fire more than the total concentrations, but the increase in the extractable concentrations and amounts was not, on the average, higher than that in the total concentrations. The weight of organic matter decreased due to fire (Alamo Carrasco, 2003; Tamminen et al., in preparation).

3.2. Effects on flora and fauna

The immediate effects of the restoration treatments, especially fire, were lethal for most of the studied species groups, especially epiphytic bryophytes, lichens and polypores, thus strongly reducing the diversity most probably for many years (Ryömmä and Laakka-Lindberg, 2005; Ryömmä et al., in preparation; Penttilä et al., unpublished). The decline is in accordance with the findings of earlier studies, which show that the short-term effect of forest fire is deleterious to many species groups, for example polypores (Penttilä and Kotiranta, 1996). The reason for this decline in species number and species abundances is simply that they are destroyed by the fire, and partly due to the destruction of the substrate (dead trees) by fire and mechanical harvesting. In addition, both fire and harvesting create more open, sunny habitats, which in some cases is a more hostile environment for those threatened species requiring closed, old-growth forest habitats.

Beetles, however, responded immediately and strongly positively to the restorative treatments. The beetle diversity showed a substantial increase after burning as compared to the unburnt sites, and also rare and red-listed beetle species were the most abundant on the burnt plots (Toivanen and Kotiaho, 2007; see also Hyvärinen et al., 2005). There are many species of saproxylic invertebrates, known to favour open disturbance areas due to the increased amount of resources and sun-exposed conditions, and these species also include some rare or threatened species (Jonsell et al., 1998; Siitonen, 2001; Similä et al., 2002; Selonen et al., 2005).

The speed of re-colonisation of bryophytes on burnt logs, and the long-term effects of fire on the bryophyte species diversity, are not well known. Although the spores of most bryophytes have the potential for long-distance dispersal, a large proportion of spores are often deposited in the immediate vicinity of the parent colonies (Miles and Longton, 1992; Sundberg, 2005). Thus, local negative effects on the destroyed species colonies may last for several decades, and recovery may even be completely hindered if there are no dispersal source close enough (Zechmeister et al., 2007). Such negative effects on epiphytic species are especially serious for rare species. Bryophyte colonisation on charred wood begins rather rapidly, but the first colonizing species seem to be the same as those on the burnt soil, such as Ceratodon purpureus, but not the specialist species dependent on CWD (Ryömmä and Laakka-Lindberg, 2005). The long-term suitability of charred wood for epiphytic bryophytes is still largely unknown, and only very few species are known to grow on this substrate (Ryömmä and Laakka-Lindberg, 2005). However, the initiative for a long-term monitoring study on epiphytic species colonization is available within the research framework of the EVO project.

The positive effect of fire on germination and establishment of many plant species has been demonstrated earlier (e.g. Granström and Schimmel, 1993). This may be caused, for example, by changes in the soil, reduced competition (incl. removal of ericaceous dwarf shrubs that exclude secondary metabolites retarding the germination and establishment of other plants) (Zackrisson and Nilsson, 1992; Zackrisson et al., 1996), and increased light availability after fire caused by increased tree mortality (Sidoroff et al., 2007). The germination study with a threatened plant species, Pulsatilla patens, in pine-dominated stands in the Evo research area, supported the positive effect of fire on the establishment and thus on the dispersal success of this species (Kalamees et al., 2005).

Assessment of the long-term effects of the treatments on biodiversity can only be made after careful and continuous monitoring. Positive long-term effects may, however, be expected on the basis of the increase in the amount of
CWD, including charred wood, which is a substrate known to be important for many endangered and threatened species (Wikars, 2004). Therefore, more fungi species, for example, are likely to be found in the research area in the future. During the first years of decay succession in the newly created dead wood, only pioneer fungal species with a mainly low conservation value are expected to be found but, later on as the decay proceeds, more fungal species with a higher conservation value are expected to appear.

The results of our aspen study in the larger Evo area confirmed the structural change and decline of aspen populations noted in recent studies in Eastern Finland (Kouki et al., 2004; Latva-Karjanmaa et al., 2007). The combined effect of the absence of large aspens in managed forests and the absence of younger cohorts and considerably high mortality in conservation areas indicate a significant threat to aspen-dwelling species. In conservation areas the suppression of natural disturbances, especially fires, has prevented the establishment of younger aspen cohorts (Romme et al., 1995; Kouki et al., 2004). Large aggregations of dead wood after fire can prevent browsing (de Chantal and Granström, 2007). However, the rapidly increasing moose population is hindering aspen regeneration, and makes aspen restoration problematic (Kouki et al., 2004). Our aspen results also support the view that biotope variation and paludified patches are important, and partly also explain the patchy occurrence of aspen in forests (Vanha-Majamaa and Jalonen, 2001; de Chantal et al., 2005).

Aspen research in the Evo experimental area is currently focusing on aspen regeneration in several experiments. A replicated set \( n = 5 \) of 0.1 ha sized canopy gaps has been established to investigate the suitability of gap cuttings as a restoration method. Each set consists of three gaps with different browsing protection treatments (no protection, wooden fence, iron fence). The gap studies will be continued with different burning and scarification treatments. The effect of browsing on aspen sapling development is being monitored on permanent sample plots, established in recent clear-cut areas (Lindberg et al., in preparation).

3.3. Early conclusions from the multidisciplinary experimental study

One clear benefit from a large-scale, multidisciplinary ecosystem level study, such as EVO experiment, is that the effects of the treatments on biodiversity can be more comprehensively studied. Our early results have already shown that focusing only on a few species group can give a very biased picture of the overall effects of forest management and restoration on species diversity.

One difficulty that was encountered is how to arrange funding and labour for a large number of sub-studies focusing on a single experiment. Also, long-term monitoring of the effects on species and ecological succession is necessary to evaluate the overall efficiency of the various restoration treatments in maintaining and restoring biodiversity. Long-term monitoring also requires well-organized data management, which is also critical for efficient data sharing and collaborative research focussing on multiple aspects of responses both nationally and internationally. Besides data management, communication between researchers, research groups and stakeholders (land-owners, funding bodies, government ministries, forestry organizations, the general public) is important, especially to help in disseminating the results.

3.4. Implications for forest restoration and management

Two main topics with potential implications for forest restoration and management have arisen from the early results of the EVO experiment. The first is related to fire behaviour in a heterogeneous site mosaic, and the second concerns the overall applicability of our approach in practical restoration.

In our experiment, upland and paludified upland patches within the same stands were initially different and also burned differently, the latter being left partly unburned. It is also known that paludified patches are often left unburnt in wild fires. These patches can act as refugia and dispersal centres for the species, thus helping in the re-colonization of the sites (Hörnberg et al., 1997; Vanha-Majamaa and Jalonen, 2001). However, this ecologically significant within-stand variability, mainly due to the soil moisture conditions, is generally neglected in stand descriptions. We suggest that this small-scale, within-stand patchiness, in combination with the disturbances, their type and subsequent stand- and landscape-level colonization-extinction dynamics, is important for understanding natural forest and species dynamics and in developing guidelines for management and restoration activities (Vanha-Majamaa and Jalonen, 2001; de Chantal et al., 2005; Lilja et al., 2005).

In Scandinavia, simplification of the structural complexity of forests and the high number of threatened forest-dwelling species call for restorative actions in both managed and protected forests (Angelstam and Andersson, 2001; Kuuluvainen et al., 2002). Early results from our study suggest that early successional structure, which is a characteristic typical to natural stands, can be successfully and rapidly restored in mature managed spruce forests even when a significant proportion of the wood volume is harvested (Lilja et al., 2005). This is encouraging because actions that aim at a balanced consideration of economical, ecological and social values, could be those that are the most beneficial in most cases (Kuuluvainen et al., 2002). In the conditions such as those in Southern Finland, where less than 1% of the forests is protected, restorative treatments, such as our experiment, can be of high value for biodiversity conservation.

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References


