Forest Ecology and Management 304 (2013) 303–311

Contents lists available at SciVerse ScienceDirect

Forest Ecology and Management

journal homepage: www.elsevier.com/locate/foreco

Full length article

# Initial effects of restoring natural forest structures in Estonia

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ARTICLE INFO

*Article history:*
Received 30 October 2012
Received in revised form 8 May 2013
Accepted 10 May 2013
Available online 6 June 2013

*Keywords:*
Forest restoration
Gaps
Hemiboreal forest
Protected forests
Rehabilitation

ABSTRACT

The legacy of structural homogenization due to forest management for commercial products is a loss of biodiversity. A common policy in many European countries is to increase forest diversity by converting managed forests to more natural conditions. The aim of this study was to provide an early evaluation of the effectiveness of different restoration treatments to rehabilitate managed stands in order to increase their naturalness. Restoration treatments were imposed on 30–60 years old conifer plantations including gap creation with and without added deadwood, added deadwood without gaps, gaps plus overburning, and controls. We sampled stand structure, understory vegetation and beetles before and after treatments on 50 circular permanent plots. Diversity of different groups responded differently to treatments with understory vegetation diversity increasing the most in gaps with burning, lichens in gaps without burning and bryophytes with the addition of dead wood. Increased beetle abundance and greater species diversity was a direct effect of changed light conditions inside the canopy. Gaps with overburning had the greatest recruitment of tree seedlings. Stands that were homogeneous pre-treatment increased in heterogeneity in structural conditions and microclimatic conditions after treatments and therefore richness and abundance of different species groups increased.



## 1. Introduction

Ecological restoration aims to return degraded ecosystems to an idealized natural state as before anthropogenic intervention, with similar species diversity, composition and structure (SERI, 2004; Stanturf and Madsen, 2002). Rehabilitation of a degraded forest stand aims to restore naturalness in terms of stand structure, species composition or disturbance regimes (Bradshaw, 2002; Stanturf, 2005). Rehabilitation at the landscape scale can be used to complement conservation efforts in protected areas in order to enhance habitat quality and quantity, to improve connectivity between fragmented areas and to create buffer zones between reserved and managed forest areas (Kuuluvainen et al., 2002).

The ultimate goal of restoration is to create a self-maintaining ecosystem that is resilient to perturbation without further assistance (Urbanska et al., 1997). Integrated approaches are suggested to measure restoration success including examining vegetation characteristics, species diversity and ecosystem processes (Ruiz-Jaen and Mitchell Aide, 2005). The main aim of restoring forest naturalness is to initiate natural processes in forests that have been heavily influenced by human manipulation, to monitor these processes going forward, including monitoring of important inter-related processes of stand regeneration, small-scale disturbances and tree mortality (Kuuluvainen, 2002; Beatty and Owen, 2005). Intervention at the stand regeneration phase can be the basis for diversification and dynamics of forests (Vodde et al., 2011). Small and large disturbances generate different possibilities and scales for successional development. Tree mortality, for example is a natural process with many causes and high spatiotemporal variability (Laarmann et al., 2009). Gap formation in natural and semi-natural forests is dependent on mortality processes that also add deadwood structure to forests. Quality of coarse woody debris is a key structural component of unmanaged forests and plays an extremely important role in ecosystem function and biodiversity conservation (Lilja-Rothsten et al., 2008; Köster et al., 2009).

One common effect of forest management to produce commercial products is structural homogenization and compositional simplification over time (Halpern and Spies, 1995). The legacy of such landscape homogenization is a loss of biodiversity; forest policy in many European countries has been to increase forest diversity by converting managed forests to more natural states (Fries et al., 1997; Lõhmus et al., 2005). In Estonia, it can be seen as reversing the trend in forest management from cultivated, even-age coniferous forests of Scots pine (*Pinus sylvestris*) and Norway spruce (*Picea abies*) toward more complex structures that include special attention to spatial and quality properties of deadwood in forest stands (Laarmann et al., 2009; Lõhmus and Kraut, 2010; Paal et al., 2011; Liira et al., 2011).

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0378-1127/$ - see front matter 
http://dx.doi.org/10.1016/j.foreco.2013.05.022

Ensuring the sustainable dynamics processes for sustaining or steadily increasing forest biodiversity and structural complexity may be expected from either allowing natural disturbance processes to operate (Franklin et al., 1997; Kangur et al., 2005; Shorohova et al., 2009) or by attempting to emulate disturbance processes by management intervention (North and Keeton, 2008; Long, 2009) although, management actions may never fully mimic natural disturbance regimes (Lindenmayer and Franklin, 2002). Restoration involves rehabilitating stands using a set of silvicultural treatments to speed up the development of structural complexity including thinning, creating snags or cavities, enhancing recruitment of woody debris and where necessary under-planting with desired species. Forest stands with high spatial heterogeneity (indicated by a large number of gaps) are typically a result of continuous moderate-intensity canopy disturbances (Lerzman and Fall, 1998; Bradshaw et al., 2011). Many non-traditional approaches can be taken in thinning for designing multispecies and multi-storeyed stands that mimic such a moderate-intensity disturbance regime (Coates and Burton, 1997; Fulé et al., 2005; Keeton, 2006; Vanha-Majamaa et al., 2007; Felton et al., 2010).

The aim of this study was to examine the early effects of treatments that targeted restoring naturalness in Estonian hemiboreal protected forests. Study questions we addressed were: (1) what were the initial effects of restoration treatments on biological diversity and (2) were there significant differences on assemblages of understory and beetle diversity and abundance of deadwood between restoration treatments? To address these questions we focused on detection of small changes at an early stage after restoration treatments.

## 2. Materials and methods

### 2.1. Study design

The study was carried out in Estonia (lat. 58–259N, long. 26–209E), which is situated in the hemiboreal vegetation zone (Ahti et al., 1968). The climate varies from maritime to continental. Annual average precipitation ranges from 600 to 700 mm. Mean temperature ranges from 16.3 to 17.4 °C in July and from −2.0 to −7.4 °C in February. Forests cover 51% of the land area of the country and the terrain is flat. Forests under some form of protection are 26% of the total forest area; about half of them are situated in nature protection areas (Yearbook, 2010).

The study was connected to the LIFE-Nature project "Protection of priority forest habitat types in Estonia" where one of the purposes was naturalness restoration on recently designated protected areas that had a low-level of naturalness and diversity. Stands requiring restoration that were selected included plantations, middle-aged (30–60 years old), normally or densely stocked, pure coniferous (*P. sylvestris* or *P. abies*) stands growing on mineral soils. Restoration treatments were implemented altogether on 350 ha in seven nature protection areas in Estonia according to existing management plans. For monitoring the restoration process, in 2004 50 permanent sample plots (PSPs) were established in 23 forest stands with total area of 78.1 ha (Korjus, 2005).

The 50 restoration treatment plots were divided into 27 with interventions and 23 without interventions designated as control plots (Table 1). Plots were established before treatments, re-measured after treatments and then re-measured after 3 years. In the 27 forest stands studied there was at least one treated plot and one control plot. Four stands had two treated plots and one stand had three treated plots.

The primary treatment was to create gaps (72–1463 $m^2$) by removing overstory trees; gaps were defined as an opening in the forest canopy extending vertically through all layers down to 2 m above ground (Brokaw, 1982). Our treatments were a single gap (G) with four installations; a gap with added dead wood (GDW) with 17 installations; and four installations with low intensity fire from burning branches and needles at the end of summer within a gap (GB). Intentional burning in the forest is usually not allowed in Estonia, even for research purposes. Other treatments included two installations with added dead wood but no overstory manipulation (DW) and 23 controls with no manipulation (C).

The PSP were established as a circular layout with a radius ranging between 15 and 25 m. The PSP radius varied depending on forest density and age structure, following the rule that every plot needed to include at least 100 main canopy trees before treatment. On each plot before treatment the tree coordinates were determined by measuring azimuth and distance from plot cente. Diameter at breast height (DBH) of each tree larger than 4 cm was measured. For every fifth tree the total height and height to crown base were measured.

Mortality was calculated for the 3 year period after treatments. The cause of mortality of each dead tree was categorized into (a) density dependent mortality; (b) wind damage; (c) game damage; (d) insect attacks; (e) fungi and diseases; (f) others (Laarmann et al., 2009).

Regeneration establishment was recorded in newly established measurement plots on each treatment plot in 2008. Five 25 $m^2$ subplots were established on each treatment plot. Subplots included one in the center of the treatment plot and the other four subplots were each located 10 m from the plot center in cardinal directions. All seedlings in each subplot were counted by tree species and the two tallest seedlings of each species were selected for height measurement.

We used a crown shape model (Lang and Kurvits, 2007) to reconstruct crowns for gap size estimation. Based on the methods of Green (1996) we used sixteen distance/direction coordinates as a polygon to estimate the gap area, which is more accurate (Zhu et al., 2009) than the widely used method of Brokaw (1982).

Biodiversity in a given area is usually evaluated through surveys of species richness in different taxonomic groups (Terradas et al., 2003; Liira and Sepp, 2009). Data on understory vegetation were collected before, immediately after and 3 years after treatments. Herbaceous species and mosses were surveyed using a step-line intercept method (Jõgiste et al., 2008). On each PSP a permanent quadrate ($5 \times 5$ m) was located 4 m from the center of the PSP in the north direction. Within the quadrate, species were recorded on step-line, where after each step a $10 \times 10$ cm square was described, resulting in total 100 squares. Lichens were inventoried on selected host material before treatment and measured 1 and 3 year after treatments (Jõgiste et al., 2008). Lichens sampling was done on: (1) 5 randomly selected main canopy trees (dominating tree species), (2) all trees from co-dominating tree species on the plot, (3) five standing dead trees or/and snags (diameter > 10 cm), (4) from three different fallen logs and (5) from three different decaying stumps and root mound.

Beetle diversity was inventoried with flight-intercept traps on the treatment areas. Beetle diversity was not monitored on control plots as forest stands are quite small (1–2 ha) in Estonia and control plots are close to treatment plots. Therefore any treatment in a stand influences beetle fauna also on control plot and control plot does not represent an area without treatment. In total 22 traps were set out and beetles were collected six times (every 2 weeks) during the summer of the pre-treatment year and in years one and three after treatment. 82% of the all beetles collected were identified to the species level.

### 2.2. Data analysis

Biodiversity was calculated using the coverage data per species and the total coverage by species group using Shannon–Wiener

**Table 1**
Distribution of plots by treatment. DW – dead wood input, G – gap cutting, GB – gap cutting and over burning; GDW – gap cutting with dead wood inclusion.

| Treatment | No of plots | Deadwood input ($m^3\ ha^{-1}$) | | Gap size ($m^2$) | | Stand age (year) | | Basal area ($m^2\ ha^{-1}$) | |
|---|---|---|---|---|---|---|---|---|---|
| | | Min | Max | Min | Max | Min | Max | Min | Max |
| DW | 2 | 7 | 12 | – | – | 50 | 100 | 31 | 41 |
| G | 4 | – | – | 119 | 225 | 30 | 45 | 19 | 28 |
| GDW | 17 | 1 | 67 | 72 | 404 | 30 | 60 | 22 | 38 |
| GB | 4 | – | – | 933 | 1463 | 50 | 50 | 1 | 5 |
| C | 23 | – | – | – | – | 30 | 100 | 20 | 47 |

index. Species richness was set to the average number of different species per plot sampled for the herbs, mosses, lichens and beetles, this variable is expected to have the Poisson distribution.

Thus, we used two different procedures for analysis of variance (ANOVA). With unbalanced data, as our data were, it is more appropriate to use the SAS procedure GLM for analysis instead of the procedure ANOVA (Littell et al., 2002). The procedure GLM was used for species diversity. A repeated measures ANOVA was conducted to determine the effects of treatment and time (2004–2008) on Shannon indexes of the herbs, mosses, lichens and beetles. Every species group was analyzed separately.

For species richness analysis we used the SAS procedure GLIMMIX, because it allows having data with the Poisson distribution. For analyzing significance of year effect for every treatment a statement CONTRAST in the procedure GLIMMIX was used.

We used Multiple Response Permutation Procedure (MRPP) to test for differences in species composition among treatments in 2008 (3 years post-treatment) with PC-ORD ver. 6 (McCune and Mefford, 2010). MRPP is a nonparametric procedure that tests the hypothesis of no difference in compositional similarity among two or more groups. MRPP gives a $p$-value based on the probability that the observed within group distance is smaller than could have occurred by chance. Measure of effect size is provided by the value $A$ which describes the within group homogeneity.

We used Indicator Species Analysis (ISA) for the 2008 measurement to test for differences of species by treatments. Indicator species are species that are used as ecological indicators of environmental conditions or environmental changes (De Caceres et al., 2010). ISA produces indicator values for each species in each treatment, based on the standards of a perfect indicator, which based on relative frequency and abundance of a given species. Indicator values were tested for statistical significance using Monte Carlo simulation tests (1000 runs). Indicator species were identified from additional species that appeared between 2004 and 2008.

For tracking changes in forest composition over time in the treatment plots we used principal components analysis (PCA in PC-ORD v.6.0, McCune and Mefford, 2010). Stand parameters used were basal area ($m^2\ ha^{-1}$), mean height of trees (m), species richness and diversity indices for each species group; gap size ($m^2$), diameter of gap (m), ratio between diameter of gap and stand mean height, coarse woody debris ($m^3\ ha^{-1}$), number of dead trees and cause of mortality, which were logarithmically transformed.

## 3. Results

Among the 50 plots, we identified a total of 138 vascular plant and bryophytes species in 2004–2008; of these, there were 94 herbs, 7 shrubs and 37 mosses. Moss species *Hylocomium splendens* and *Pleurozium schreberi* were present in 91–100% of plots and vascular plants *Deschampsia flexuosa* and *Melampyrum pratense* were present in all treatments in 2008. Over all treatment plots there were 79 lichen species, including rare and threatened species in Estonia *Biatora botryosa*, *Calicium pinastri, Cladonia incrassata*, *Cladonia norvegica*, *Micarea hedlundii*, *Placynthiella dasaea*, and *Trapelia coarctata* (Lilleleht, 1998; Randlane and Saag, 1999; Randlane et al., 2008). A total of 512 beetle species were found during 2004–2008: 145 species in 2004, 340 species in 2005, and 365 species in 2008.

Herb species richness (Fig. 1) was affected significantly by changes in light conditions with gap formation increasing richness (G + GB + GDW treatments together, $F = 6.94$, $p < 0.001$; GDW treatment alone, $F = 4.76$, $p = 0.01$). However, we did not find significant changes between the years in single treatments except GDW (G, $p = 0.37$; DW, $p = 0.73$; GB, $p = 0.11$) or on control plots ($p = 0.63$), but the biggest change was in the G treatment where an average of 4.25 species was added. Richness of mosses and lichens was not significantly affected in any treatment. The average richness of mosses was 4.74 in 2004 and 5.46 three years later and the average richness of lichens was 5.6 and 6.2 respectively. Richness of beetles increased over time, since a response in any treatment affects other treatments due to the small stand size ($p < 0.001$).

Shannon indexes of herbs, mosses and lichens were significantly dependent upon treatments ($p < 0.003$) (Table 2). Shannon index of herbs increased the most in the gap with burning treatment (GB), and decreased in the added deadwood treatment (DW). The smallest change was on the control plots. The deadwood and gap with burning treatments positively affected Shannon index of mosses, while the gap alone (G) treatment affected moss that negatively. Changes in Shannon index of lichens were higher in the gap alone and gap with added deadwood treatments but decreased in the gap with burning treatment. Shannon index of beetles was different among years ($p = 0.01$, $f = 8.16$), but did not show any differences among the treatments. Interaction between time and treatment was not significant for any variable.

Differences in effects of composition 3 years after treatments are shown by MRPP analysis (Table 3). We found significant differences among treatments for herbs ($p < 0.001$, $A = 0.07$), mosses ($p = 0.04$, $A = 0.03$), lichens ($p < 0.001$, $A = 0.03$) and beetles ($p < 0.001$, $A = 0.14$). In pairwise comparisons, herb species in the gap with burning treatment (GB) was significantly different from control ($T = -7.87$, $p < 0.001$) and the rest of the treatments. Moss composition in the GB treatment was different from the other treatments ($p < 0.05$) except from the DW treatment ($p = 0.12$). Lichens were similar in control and GDW treatment ($p = 0.44$); in all the other treatments lichens were different. Beetles differed between deadwood and gap, deadwood and gap with added deadwood; and between gap with burning and gap with added deadwood.

Indicator species analysis (ISA) for the year 2008 presented seven herbaceous species, 1 moss, 8 lichen species and 28 insect species as indicators (Table 4). Indicator species in 2008 were not the same as in the pre-treatment measurement in 2004.

The ISA revealed some beetle species specific to some treatments; species such as *Cardiophorus ruficollis*, *Mycetochara flavipes*, *Ochthebius minimus*, *Tomoxia bucephala*, *Cis comptus, Sericoderus lateralis,* which were present on burned sites (Süda and Voolma, 2007). *Hylobius abietis*, *Hylastes brunneus*, *Scolytus ratzeburgi* were present close to fresh cutting (DW) sites (Voolma et al., 2003). A new indicator species in Estonia was recorded for treatment DW, *Malthinus facialis* (Süda, 2009).

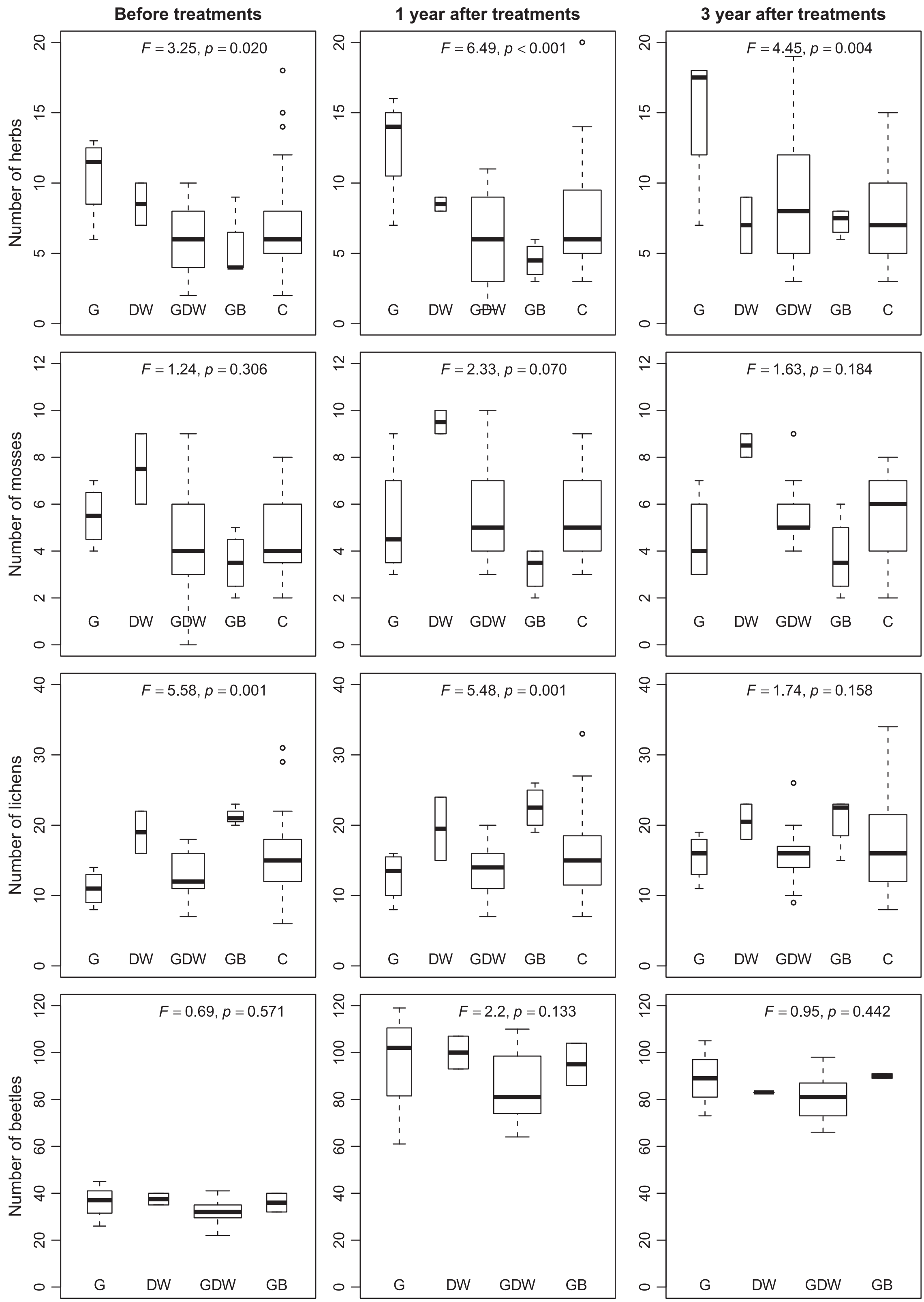


**Fig. 1.** The average species richness of vascular plants, mosses, lichens and beetles per plot in the treatment categories before the treatments, 1 year after the treatments and 3 years after the treatments. Treatment categories: G, gap cutting; DW, deadwood input; GDW, gap cutting with deadwood input; GB, gap cutting and overburning; C, control plots. Values of ANOVA test are presented. The upper and lower boundaries of the boxes are the 75th and 25th percentiles, the horizontal line within the box is the median and the error bars show the 10th to 90th percentiles. Width of box depends on number of plots.

**Table 2**
Dynamics of mean Shannon diversity index in species groups by treatments (G – gap cutting, DW – deadwood input, GDW – gap cutting with deadwood input, GB – gap cutting and overburning, C – control plots). Bold values indicate p-values lower the threshold value 0.05 (significant).

| Attribute and year | Treatment | | | | | ANOVA results | | | |
|---|---|---|---|---|---|---|---|---|---|
| | G | DW | GDW | GB | C | Treatment | | Time | |
| | | | | | | *F* | *P* | *F* | *P* |
| *Herbs* | | | | | | 5.31 | **<0.001** | 2.829 | 0.06 |
| 2004 | 1.852 | 1.616 | 1.179 | 0.987 | 1.365 | | | | |
| 2005 | 2.060 | 1.610 | 1.210 | 1.025 | 1.476 | | | | |
| 2008 | 2.199 | 1.368 | 1.521 | 1.535 | 1.521 | | | | |
| *Mosses* | | | | | | 4.29 | **<0.003** | 3.05 | 0.05 |
| 2004 | 1.189 | 1.398 | 0.952 | 0.789 | 0.984 | | | | |
| 2005 | 1.059 | 1.574 | 1.126 | 0.731 | 1.140 | | | | |
| 2008 | 1.105 | 1.615 | 1.149 | 0.998 | 1.144 | | | | |
| *Lichens* | | | | | | 5.59 | **<0.001** | 1.22 | 0.30 |
| 2004 | 1.167 | 1.527 | 1.356 | 1.947 | 1.475 | | | | |
| 2005 | 1.233 | 1.502 | 1.419 | 2.048 | 1.522 | | | | |
| 2008 | 1.486 | 1.648 | 1.571 | 1.859 | 1.564 | | | | |
| *Beetles* | | | | | | 2.64 | 0.14 | 8.16 | **0.01** |
| 2004 | 0.966 | 0.499 | 0.487 | 0.867 | – | | | | |
| 2005 | 2.486 | 2.933 | 2.688 | 3.399 | – | | | | |
| 2008 | 3.010 | 3.438 | 3.066 | 3.247 | – | | | | |

**Table 3**
Multi-response permutation procedure analysis between treatments (G – gap cutting, DW – deadwood input, GDW – gap cutting with deadwood input, GB – gap cutting and overburning, C – control plots) in species groups 3 years after treatment (H-herbs; M-mosses; L-lichens; B-beetles).

| | G | DW | GDW | GB | C |
|---|---|---|---|---|---|
| G | | | | | |
| DW | L** ; B*** | | | | |
| GDW | L* | L***; B* | | | |
| GB | H**; M*; L*** | H*; L*** | H***; M**; L***; B** | | |
| C | L** | L** | | H***; M**; L*** | |

* Significance code 0.05.
** Significance code 0.01.
*** Significance code 0.001.

The treatments reduced the number of living trees on all plots and increased the amount of dead wood on plots receiving added deadwood (DW and GWD). We found differences in the distribution of seedling species and their abundances between treatments ($X^2$; $p < 0.05$). The abundance of *Pinus* seedlings increased on the gap and burned plots (GB) (Fig. 2). Treatments GB and GDW increased the number of *Betula* seedlings and G increased number of *Picea* seedlings. The number of other trees, mainly *Populus tremula* and *Quercus robur* did not differ among treatments.

Comparing stand structure from before and 3 years after treatment (2004 vs. 2008) shows different patterns on treatment plots (Fig. 3). Control plots were excluded in this analysis because we did not find any significant changes between years. The PCA results indicated that five statistically significant ($p = 0.001$) axes were available for interpretation, which explained 79% of the variation in the data. The first principal component (PC1) was strongly related to density of stand while PC2 was related to attributes of dead wood; the others components described together 30% of structural variation.

We found significant changes between pre-treatment and subsequent years ($F = 6.18$; $p < 0.001$). Pre-treatment stands were structurally quite even and similar to each other; after treatment stand structure changed and this initiated changes in herb, moss, lichens and beetle composition as well as in tree regeneration.

The variables with the strongest correlation with the first axis were gap size ($r = -0.92$; $p < 0.001$), diameter of gap ($r = -0.92$; $p < 0.001$), removed basal area ($r = -0.91$; $p < 0.001$), ratio of gap and stand mean height ($r = -0.91$; $p < 0.001$), number of seedlings ($r = -0.65$; $p = 0.001$), and basal area ($r = 0.58$; $p < 0.04$). The second axis correlated best with number of dead trees per hectare ($r = -0.64$; $p < 0.001$), number of reasons for tree mortality ($r = -0.62$; $p < 0.001$), amount of coarse woody debris ($r = -0.55$; $p < 0.001$), richness of mosses ($r = -0.61$; $p = 0.03$) and richness of herbs ($r = -0.52$; $p = 0.02$).

Three years after treatment there was no clear effect on mortality. Tree mortality during the 3 years after treatment was highest on control plots, 6.7% (over the 3 year period). Mortality on the gap alone (3.9%) and gap with added deadwood treatments (4.7%) was higher than the treatment of added deadwood (1.6). Plots receiving the gap plus burning treatment had significantly larger gaps than were produced for the gap or gap with added deadwood treatments (G or GDW) but had no dead trees during this 3-year period.

## 4. Discussion

Managing disturbances by manipulative treatments is part of the process of restoring natural disturbance regimes that have been disrupted by human intervention (Stanturf, 2004). A challenge for forest managers is to develop and implement management practices that restore stand structural complexity and compositional diversity (Lindenmayer and Franklin, 2002). It is possible to imitate gap dynamics, which is one of the typical disturbance regimes in hemi-boreal forests (Shorohova et al., 2009). Canopy gaps are created by death of one or more trees naturally or by management (thinning, selective cutting); gaps are critical in community dynamics, species coexistence and regeneration of many types of forests (Liu and Hytteborn, 1991; Gray and Spies, 1996). Deadwood contributes to stand structural complexity and creating different sizes of slowly dying and dead wood is another important tool for restoring the natural characteristics of managed forests (Siitonen, 2001). To maintain biodiversity and ecosystem processes in managed forests different individual structures are needed such as standing dead trees, logs and coarse woody debris on forest floor, and large dimension deadwood in different decay stages (Lindenmayer and Franklin, 2002). Spatial variability of deadwood distribution in a stand is caused by different mechanisms of individual tree death (e.g., insect attack, fungi, competition); spatial variability of deadwood is one of the characteristics of natural/semi-natural stands (Laarmann et al., 2009).

**Table 4**
Indicator species analysis of vascular plant (V) and mosses (M), lichens (L) and beetles (B) for 3 years after treatment (G – gap cutting, DW – deadwood input, GDW – gap cutting with deadwood input, GB – gap cutting and overburning) based on relative frequency and abundance of a given species.

| Species | Group | Treatment | *P* | Indicator value | % Of plots in given group where given species is present | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | | | | G | DW | GDW | GB | C |
| *Miarus* sp. | B | G | 0.001 | 100 | 100 | – | – | – | n/a |
| *Alosterna tabacicolor* | B | G | 0.029 | 80 | 100 | – | 82 | – | n/a |
| *Anthribus nebulosus* | B | G | 0.010 | 79 | 100 | – | 27 | – | n/a |
| *Liodopria serricornis* | B | G | 0.022 | 73 | 100 | – | 45 | – | n/a |
| *Leptura melanura* | B | G | 0.035 | 73 | 100 | 100 | 91 | 100 | n/a |
| *Meligethes* sp. | B | G | 0.009 | 70 | 100 | – | 91 | 50 | n/a |
| *Amara brunnea* | B | G | 0.029 | 67 | 67 | – | – | – | n/a |
| *Phyllobius argentatus* | B | G | 0.029 | 67 | 67 | – | – | – | n/a |
| *Caenocara affinis* | B | G | 0.033 | 67 | 67 | – | – | – | n/a |
| *Anemone nemorosa* | V | G | 0.022 | 55 | 75 | – | 6 | – | 17 |
| *Fragaria vesca* | V | G | 0.014 | 48 | 75 | – | – | 25 | 9 |
| *Deschampsia cespitosa* | V | G | 0.044 | 45 | 50 | – | 6 | – | – |
| *Cladonia coniocraea* | L | G | 0.015 | 17 | 50 | 19 | 25 | 32 | 15 |
| *Hypogymnia tubulosa* | L | G | 0.030 | 9 | 18 | – | 6 | 8 | 3 |
| *Hylobius abietis* | B | DW | 0.012 | 100 | – | 100 | – | – | n/a |
| *Malthinus facialis* | B | DW | 0.012 | 100 | – | 100 | – | – | n/a |
| *Myzia oblongoguttata* | B | DW | 0.012 | 100 | – | 100 | – | – | n/a |
| *Hylastes brunneus* | B | DW | 0.020 | 89 | – | 100 | 18 | – | n/a |
| *Scolytus ratzeburgi* | B | DW | 0.043 | 67 | – | 100 | – | 50 | n/a |
| *Athous subfuscus* | B | DW | 0.009 | 50 | 100 | 100 | 100 | 100 | n/a |
| *Melampyrum* sp. | V | DW | 0.042 | 50 | – | 50 | – | – | – |
| *Lycopodium annotinum* | V | DW | 0.042 | 50 | – | 50 | – | – | – |
| *Trapeliopsis flexuosa* | L | DW | 0.035 | 7 | 4 | 14 | 2 | 4 | 2 |
| *Agathidium* sp. | B | GDW | 0.031 | 52 | 100 | 50 | 100 | 100 | n/a |
| *Dimerella pineti* | L | GDW | 0.026 | 15 | 7 | 5 | 28 | – | 21 |
| *Carex ericetorum* | V | GB | 0.001 | 100 | – | – | – | 100 | – |
| *Cardiophorus ruficollis* | B | GB | 0.014 | 100 | – | – | – | 100 | n/a |
| *Cryptolestes corticinus* | B | GB | 0.014 | 100 | – | – | – | 100 | n/a |
| *Mycetochara flavipes* | B | GB | 0.014 | 100 | – | – | – | 100 | n/a |
| *Ochthebius minimus* | B | GB | 0.014 | 100 | – | – | – | 100 | n/a |
| *Orthotomicus suturalis* | B | GB | 0.014 | 100 | – | – | – | 100 | n/a |
| *Prosternon tessellatum* | B | GB | 0.014 | 100 | – | – | – | 100 | n/a |
| *Rhagonycha elongata* | B | GB | 0.014 | 100 | – | – | – | 100 | n/a |
| *Tomoxia bucephala* | B | GB | 0.014 | 97 | 33 | – | – | 100 | n/a |
| *Hydroporinae* | B | GB | 0.007 | 93 | – | – | 27 | 100 | n/a |
| *Anidorus nigrinus* | B | GB | 0.040 | 92 | – | 50 | 18 | 100 | n/a |
| *Cis comptus* | B | GB | 0.039 | 75 | 33 | – | – | 100 | n/a |
| *Sericoderus lateralis* | B | GB | 0.049 | 73 | 33 | 50 | 9 | 100 | n/a |
| *Epilobium angustifolium* | V | GB | 0.011 | 70 | 75 | – | 29 | 100 | 9 |
| *Funaria hygrometrica* | M | GB | 0.011 | 49 | – | – | 6 | 50 | – |
| *Dicranum polysetum* | M | GB | 0.034 | 45 | 50 | 100 | 76 | 100 | 74 |
| *Lecanora phaeostigma* | L | GB | 0.001 | 27 | – | – | 2 | 32 | 6 |
| *Evernia prunastri* | L | GB | 0.003 | 11 | – | – | – | 12 | 1 |
| *Hypocenomyce scalaris* | L | GB | 0.027 | 9 | 4 | 5 | 4 | 20 | 5 |

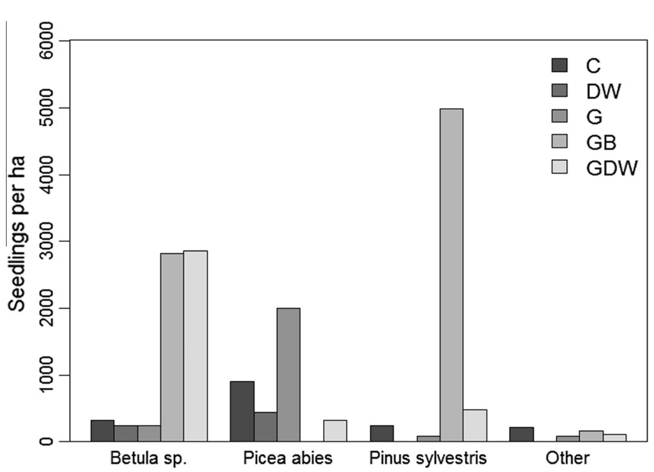


**Fig. 2.** Number of tree seedlings in 2008, 3 years after the treatments on gap cutting (G), deadwood input (DW), gap cutting with deadwood input (GDW), gap cutting and overburning (GB), control plots (C).

This study was initiated to provide an early evaluation of the effectiveness of different restoration treatments to rehabilitate managed stands in order to increase their naturalness. The study was conducted in previously managed forest that had recently been brought under nature protection status. Since this protection status was only developed during the last decade, managers are interested to know the possible suitable actions for conservation management purposes. The studied areas were located in the buffer zones around core natural areas and thus were not under strict protection. Although these are managed areas, the management actions allowed are to enhance biodiversity development and as such are different from management of commercial forests. Several earlier studies (Fries et al., 1997; Carey and Curtis, 1996; Kuuluvainen et al., 2002) guided the choice of restoration treatments used in this study and therefore the results should be of value not only for buffer zone management but also for increasing naturalness in commercial forests.

The treatments in this study intended to mimic natural wind disturbance that is typical of boreonemoral forests (Shorohova et al., 2009). The effects on understory diversity should follow a similar pattern as after windthrow. An increase in variation of light intensity after cutting treatments and intensive dead wood inclusion generates suitable habitats for shade-dependent species under

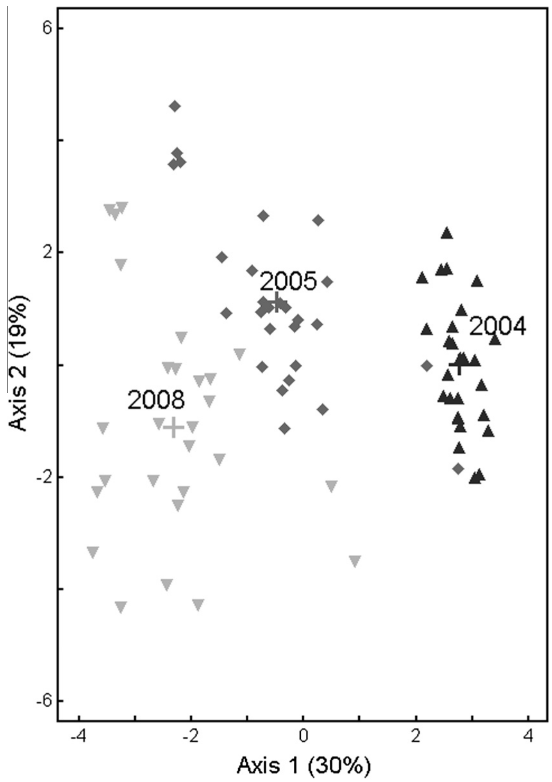


**Fig. 3.** Ordination of structural characteristics according to the Principal Component Analysis. Plots described by symbols: plots before treatments are indicated with triangles, 1 year after treatments with diamonds, 3 year after treatment with reverse triangles. Centroids of each year are indicated by crosses.

the shade of fallen tree trunks (Lilja-Rothsten et al., 2008). The relationship between the gap diameter and the height of the surrounding stand determines the light availability inside the gap. If the relationship is close to one or larger (i.e., the gap is as wide as the height of surrounding trees or wider) there is greater recruitment and successful establishment of light demanding species. On the other hand, understory vegetation establishment is favoured by the variability of diffuse light (Moora et al., 2007).

Several authors reported higher understory species diversity in recently cut areas as compared with that in old-growth stands (Zobel, 1993; Pykäla, 2004; Aavik et al., 2009; Moora et al., 2007) but some studies indicated that felling methods may result in a decrease in species richness in the short-term (Jalonen and Vanha-Majamaa, 2001). Our results indicate that gap and deadwood treatments (G, GDW, and DW) did not influence herbaceous species richness in comparison to control plots. But at the same time the change in species composition after treatments is illustrated by the results of indicator species analysis. Stands with gap cuttings were characterized by common pioneer species such as *Epilobium angustifolium* and light-demanding species such as *Deschampsia cespitosa*, *Fragaria vesca*, and *Carex ericetorum*. Similar results were found among bryophytes.

The gap plus burn treatment (GB), however, differed significantly from control plots for all species groups. Other studies with burning treatments showed similar results, for example Vanha-Majamaa et al. (2007) and Glasgow and Matlack (2007) where there were clear differences in understory vegetation between the biotype classes after burning treatments.

Compared to species groups on control areas, only the lichen group differed among G and DW treatments. Surprisingly the species groups were fairly similar on control plots and on treatment GDW, although this may be due to the higher variation on sample plots in the GDW treatment. Previous studies (Schimmel and Granström, 1996; Wikars, 2002; Junninen et al., 2008; Parro et al., 2009; Ruokolainen and Salo, 2009) have also shown that wildfire provides an opportunity for fire-dependent and fire-adapted species to develop in previously closed-canopy stands.

The current study results present a preliminary response to the restorative treatments, but give some confidence that creating gaps and increasing dead wood along with over burning can direct and restore rapid development of structural diversity in formerly managed forests. A conceptual scheme of the temporal trajectory of treatment effects is presented in Fig. 4. The more intensive treatments (gap cutting and burning) show an initial decline in complexity but more move rapidly towards greater complexity than the less manipulative treatments (control and added deadwood alone). This conceptual response should be interpreted in light of the differential response of the several taxonomic groups to different treatments (i.e., lichens respond to one treatment, bryophytes to another, vascular plant to another, etc.). This suggests multiple treatments and an emphasis on creating stand heterogeneity can increase biodiversity more than one homogenous application of a single treatment. Nevertheless, the conceptual diagram can be used as hypotheses to be tested in subsequent monitoring of the PSPs.

Our results indicate that deadwood input and heterogeneity increases after treatments in a stand. Similar results are reported from an on-going experimental research project (EVO) in Finland (Lilja et al., 2005; Vanha-Majamaa et al., 2007) and from simulation modelling for restoration of old-growth structural features in hardwood forests in Northern America (Choi et al., 2007). Although, we did not analyze spatial patterns and arrangement effects of deadwood inclusion in this study, the Laarmann et al. (2009) results indicated that in stand structural complexity the dead wood attributes are able to describe 40% of the variation. It is crucial to follow the enlargement of the gap size, since there is clear evidence of gap enlargement due to trees dying or being disturbed by wind on the gap edges. For example, Köster et al. (2009) reported that in Norway spruce stands the highest deadwood creation along forest edges appeared 3 years after the disturbance.

Natural regeneration of stands dominated by Scots pine is often difficult. On dry sandy soils the problem is the thick litter and raw humus layers that develop; on moderately wet or wet sandy soils the difficulty is due to abundant understory vegetation cover or

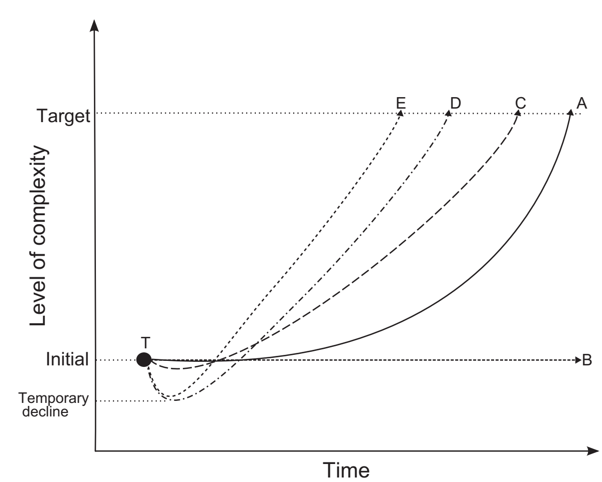


**Fig. 4.** Conceptual scheme of temporal trajectory of treatment effects in conifer even-aged stands: different pathways from initial level to targeted level of forest stand complexity. Some destructive treatments have temporary decline effect after that stand complexity increases more rapidly than with constructive treatments. Treatments: (A) no treatment (natural development of stand); (B) continuation of previous management regime for timber production; (C) deadwood input; (D) gap cutting; and (E) gap cutting with overburning.

thick raw humus (or peat) layers. Gap cutting and overburning treatments create conditions for successful Scots pine natural regeneration by partial soil scarification, reduced competition and increased light availability. Occurrence of successful natural regeneration allows rapid development of treated stands towards a multispecies uneven- aged stand (Hanewinkel and Pretzsch, 2000). Our results support the positive effect for Scots pine regeneration from gap cutting treatment with over burning. In comparison, the birch species abundance was higher on treatments with over burning and gaps with deadwood input. Norway spruce natural regeneration was most abundant on the treatment of gap alone. We were not able to distinguish between the individual impacts of over burning or gap creation and this needs further study. There was no notable browsing by herbivores on newly established tree regeneration on sample plots. Still, this must be monitored further because browsing may increase in the future and affect the course of stand development and understory development.

Several studies have reported increased richness of beetle species in moderately managed stands, where storm damage has been partly salvaged (Duelli et al., 2002; Ehnström, 2001). The increased abundance was due to freshly cut stumps and added deadwood that released terpenes which acted as a strong attractant for some of the beetle species. On the other hand, the insect abundance and greater species diversity was a direct effect of changed light and temperature conditions inside the canopy. Beetle monitoring is especially valuable and proper to evaluate changes on restoration areas because of beetle species and habitat richness, fast life cycle, distribution patterns and variety of monitoring methods available. Even as the species dependent on freshly cut materials moved away, there was an increase in the species dependent on fungi and dead wood. On the treatment with the added dead wood (stem parts, stumps, snags) the debris had decomposed enough to create a habitat for some new species (e.g. *Hylastes cunicularius*, *Monochamus urussovii*, and *Pityogenes chalcographus*) although their abundance was not as high as the abundance of species moving in right after treatment.

Gap cuttings in the current study presented a clear positive effect on the abundance and diversity of insects. Although older stands are expected to provide more habitat variability and therefore host more rare species, rare or threatened species were found in the current study after only 3 years post-treatment. Studied forest stands were rather small (1–2 ha) and, most likely, treatments influenced insect fauna on larger areas than these stands. Treatment areas and control areas are not physically isolated from each other and therefore it was not possible to distinguish beetle diversity between them. In future, some new control plots in similar stands should be established outside the treatment influence area. Continued study will be necessary to determine if there will be changes of insect fauna in the future.

Presence and abundance of insect species depends not only on forest management activities but rises and falls according to favourability of weather and habitat conditions. The year-to-year effect is sometimes a stronger signal than treatment effects. Our study results nevertheless indicate significantly high species diversity and species abundance shortly after the treatments. It is important to continuously monitor insect species abundance and diversity in the future to determine the long-term effect of restoration treatments on insect diversity..

## 5. Management implications

Forest management does not mean a direct threat to species diversity; the effect of management interventions can be positive or negative, depending upon the nature and intensity of treatments (Voolma and Õunap, 2006). Knowledge from ecological studies of natural and managed forests has helped to assess and redesign forest management policies, both to increase diversity in commercially managed forests and also in restoring naturalness to such stands now managed for nature protection.

Our study covered only the first 3 years of development after restoration treatments, which obviously limits our interpretations for management of protected areas. Nevertheless the early indications from our study are that the rehabilitation techniques we used (single gap cutting, gap cutting with dead wood input, gap cutting with over burning, and dead wood input alone) positively affected general structural heterogeneity but also species diversity. These changes were evident also in the dynamics of different species groups (vascular plant, bryophytes and beetles) in comparison to pre-treatment conditions.

Recruitment processes for coarse woody debris and tree seedling regeneration have been successfully reinitiated at a higher level and hopefully will increase the general level of biodiversity to an even higher level than it was before treatments. In selecting restorative measures it is important to base actions on the natural disturbance regime for a particular forest site. The gap creation treatment with over burning resulted in high site-specific species diversity and also produced good results for seedling establishment. Prescribed burning, however, is not allowed in Estonia; our results suggest this ban should be re-considered for restoration measures. These preliminary results underscore the importance of continuous systematic monitoring for evaluating the restoration actions as an investment but even more as an assessment of forest naturalness dynamics. This study underscores the importance of systematic monitoring for conservation management.

## Acknowledgements

The research reported here was supported by the Estonian Ministry of Education and Research (Project No. SF0170014s08) and by the Estonian Science Foundation (Grant No. 8496) and Estonian Environmental Investment Centre. We thank the Environmental Board, Estonia, and fieldwork staff for their input and two anonymous reviewers for valuable comments to the manuscript.

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