

APPENDIX D

# Biodiversity of boreal forest ecosystems

- Effects of nitrogen fertilization, whole-tree harvest and stump harvest



A report from Belyazid  
Consulting & Communication



## **A report from Belyazid Consulting and Communication AB**

This report is an appendix to the report “Effects of climate change, nitrogen fertilization, whole-tree harvesting and stump harvesting on boreal forest ecosystems - A review of current knowledge and an evaluation of how these factors may influence the possibilities to reach the Swedish environmental objectives”. January 2013.

### **Authors**

Ulrika Jönsson Belyazid & Salim Belyazid, Belyazid Consulting and Communication AB, Fersens väg 9, 211 42 Malmö, Sweden  
Cecilia Akselsson, Physical Geography and Ecosystem Sciences, Lund University, Sölvegatan 12, 223 62 Lund, Sweden

### **Contact**

Ulrika Jönsson Belyazid, E-mail: [ulrika@belyazid.com](mailto:ulrika@belyazid.com), Tel: +46 735 244462

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# SUMMARY

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This report consists of a literature review about the effects of forest management on biodiversity of boreal forest ecosystems, with specific emphasis on Swedish conditions. The effects of nitrogen (N) fertilization, whole-tree harvesting (WTH) and stump harvesting (SH) have been evaluated. Results from empirical as well as modelling studies are included in the review.

## Effects of nitrogen

There is a large amount of evidence of damage to European terrestrial biodiversity due to elevated N deposition. Boreal ecosystems are among the most sensitive, partly because they are adapted to low nutrient levels and partly because many of them are poorly buffered against acidification. Direct foliar damage, eutrophication and acidification of ecosystems and increased susceptibility of vegetation to secondary stress are some of the mechanisms by which elevated N might influence boreal, and other, ecosystems.

In Scandinavian boreal forests, changes in understory plant community structure as a consequence of N additions have been reported since long. Like for many other ecosystems, the abundance and/or frequency of fast-growing grasses and herbs have generally increased as a consequence of N addition, while the abundance of for example dwarf shrubs has decreased. The influence of N on the interaction between a pathogen and its host has been identified as one of the major drivers of biodiversity change in these systems.

Research on the effects of increased N input on faunal diversity is sparse, and in boreal forest ecosystems practically non-existent. However, in other ecosystems, elevated levels of N have been linked to negative effects on both caterpillars and butterflies. Other species that have been reported as negatively affected by elevated N levels include the ground beetle assemblages of open coastal grasslands and the red-backed shrike which is a resident of the coastal dunes of Western Europe. It is generally believed that the negative effects on the fauna occur primarily through changes in vegetation diversity, composition and/or structure. With regard to the diversity of EM fungi, response to fertilization seems to vary among species with certain taxa declining in abundance and diversity, while others continue to flourish at higher deposition levels.

Several recent studies have reported substantial effects from lower chronic levels of N addition, indicating that even low rates of N addition may have significant impact on the diversity of forest ecosystems. It is very likely that N deposition acts synergistically with other stressors, such as climate change, acid deposition and ground-level ozone. However, these synergies are currently poorly understood. Generally, the rates of recovery of ecosystems after N application has ceased are slow.

## Effects of whole-tree harvesting

In boreal forests, dead wood hosts a variety of fungi, bryophytes, lichens and invertebrates and the importance of dead wood for the biodiversity of forest ecosystems has been acknowledged in many

studies, both in Scandinavia and elsewhere. In Finland, at least 4000 forest species have been estimated to be dependent on dead wood. In general, coarse woody debris has been regarded as the most important wood fraction for preserving biodiversity. However, several recent studies have emphasized finer woody debris and logging residues as important to many species, in particular in managed forests where dead trees are relatively rare.

Many different animal species have been reported to be negatively affected by WTH, including several types of beetles, nematodes, enchytraeids and spiders. In particular, animals at higher trophic positions and more mobile animals seem to be negatively affected, something that has been suggested to have important implications for forest nutrient cycling. In deciduous forests, forest fuel piles have been found to act as ecological traps for several uncommon and red-listed saproxylic species.

While diversity of wood substrates and the presence of decomposing wood is clearly important for preserving fungal communities and the number of fungal species, removal of logging residues have so far not been found to have any long-term effects on any of the characteristics of fungal communities. With regard to vascular plants and bryophytes, results are contradictory with studies showing no effects on species composition and richness as well as studies showing negative effects on typical forest species as a consequence of WTH.

## Effects of stump harvesting

Stumps represent a significant component of the dead-wood habitat of harvested forest sites and stumps have been identified as providing long-lasting dead-wood habitat for a wide range of toadstools, bracket fungi, mosses, lichens, saproxylic beetles, other invertebrates and species supported by the above. Apart from influencing the amount of dead wood, SH also result in reduced habitat heterogeneity and changes in the abiotic characteristics of sites, conditions that may favour pioneering species.

In general, there are very few studies of the effects of SH on biodiversity in forest ecosystems. The only studies available are those investigating the importance of stumps for various plant and animal species showing that many species – of both plant and animal origin – are dependent on stumps. Traditionally, high stumps have been regarded as most important for biodiversity. However, several recent studies have emphasized that low stumps may also host a wide variety of species, although the species composition may sometimes be complementary to that of high stumps.

## Conclusions

Most available studies present negative effects of N fertilization and potentially negative effects of forest fuel harvesting on forest biodiversity. While the effects of N are relatively well investigated, the number of studies on the effects of WTH and SH on biodiversity are very limited. Firm conclusions about the long-term impact of WTH and SH are thus difficult to draw.



# TABLE OF CONTENTS

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<b>SUMMARY</b> .....	<b>3</b>
<b>1. BRIEF</b> .....	<b>7</b>
<b>2. NITROGEN FERTILIZATION</b> .....	<b>8</b>
2.1 The impacts of N on terrestrial biodiversity .....	8
2.2 Evidence of changes in biodiversity as a consequence of elevated N .....	8
2.2.1 Vegetation changes .....	8
2.2.2 Fauna changes .....	11
2.2.3 Fungal changes .....	11
<b>3. WHOLE-TREE HARVEST</b> .....	<b>12</b>
3.1 The impacts of WTH on terrestrial biodiversity .....	12
3.2 Evidence of changes in biodiversity as a consequence of WTH .....	12
3.2.1 Vegetation changes .....	12
3.2.2 Fauna changes .....	13
3.2.3 Fungal changes .....	13
<b>4. STUMP HARVEST</b> .....	<b>14</b>
4.1 The impacts of SH on terrestrial biodiversity .....	14
4.2 Evidence of changes in biodiversity as a consequence of SH .....	14
4.2.1 Vegetation changes .....	14
4.2.2 Fauna changes .....	15
4.2.3 Fungal changes .....	16
<b>5. CONCLUSIONS</b> .....	<b>17</b>
<b>6. REFERENCES</b> .....	<b>18</b>







# 1. BRIEF

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This report consists of a literature review about the effects of forest management on terrestrial biodiversity of boreal forest ecosystems, with specific emphasis on Swedish conditions. The literature review is an appendix to the report “Effects of climate change, nitrogen fertilization, whole-tree harvesting and stump harvesting on boreal forest ecosystems - A review of current knowledge and an evaluation of how these factors may influence the possibilities to reach the Swedish environmental objectives” published by Belyazid Consulting & Communication AB in January 2013. The forest management practices investigated are some that are likely to become more common in Sweden in the future as a consequence of the increased interest in obtaining biofuel from forests, namely nitrogen (N) fertilization, whole-tree harvesting (WTH) and stump harvesting (SH).

Biodiversity, or biological diversity, may be defined in numerous ways. According to Dise *et al.* (2011), one of the simplest ways of describing biodiversity is as “the variety of life, in all its manifestations” (Gaston & Spicer, 2004), including the diversity of genes, populations, species, communities and ecosystems. A more comprehensive definition is included in the 1992 Convention on Biological Diversity: “Biological diversity means the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological

complexes of which they are part; this includes diversity within species, between species and of ecosystems” (CBD, 1992). Apart from the aesthetic value of biodiversity, the benefits, or services, provided by natural habitats have been increasingly recognized during the past decades. No single measure encompasses all of the elements of biodiversity and according to Dise *et al.* (2011), the measure used thus have to be appropriate to the scale of investigation and the purpose of the study.

From a nature conservation point of view, investigating the effects of forest management on diversity on a landscape level over longer periods of time is usually considered the most relevant approach. However, these kinds of studies are rare. Here, we have mainly looked at the effects on organismal level, primarily species richness (i.e. the number of species in a defined area) and species abundance (i.e. how common a species is in an area) at tree or stand level, which is the information most commonly presented in the literature with regard to both plant and animal species. Foremost, we have collected information from scientifically published articles concerning boreal forest ecosystems in Scandinavia. When the information available was scarce, we extended the literature search to other types of forest ecosystems and other regions of the world. Both empirical and modelled data have been included in the review.



# 2. NITROGEN FERTILIZATION

N is, together with habitat conversion and climate change, regarded as one of the major drivers of biodiversity loss in Europe (Dise *et al.*, 2011). The effects of deposition of reactive N on biodiversity were recently thoroughly reviewed in the European Nitrogen Assessment (Sutton *et al.*, 2011). The information below is therefore mainly a summary of the findings presented in the chapter about N as a threat to terrestrial biodiversity (Dise *et al.*, 2011). Although there are certain differences between N deposition and N fertilization (i.e. the rates of atmospheric N deposition are generally low compared with the rates of fertilizer application, and fertilizer applications are often one or a few-time applications whereas atmospheric deposition is chronic), we consider both of them as input processes with similar effects on terrestrial ecosystems.

## 2.1 The impacts of N on terrestrial biodiversity

N influences the diversity of terrestrial ecosystems through several different mechanisms:

1) It may damage the vegetation directly. At high concentrations, nitrogen gases, aerosols and dissolved compounds can be directly toxic to the above-ground parts of plants and leaf injury, changes in physiology as well as growth reductions have been reported for natural vegetation at high concentrations of air-borne N pollutants (Dise *et al.*, 2011). The vegetation group most sensitive to dry-deposited N, especially in the form of  $\text{NH}_3$ , is lichens. Both lichens and bryophytes are sensitive to wet deposited N. Direct foliar impacts on trees have been observed, but are nowadays relatively rare due to emission reductions.

2) It may eutrophy the ecosystem (see Appendices B & C). N is the limiting nutrient for plant growth in many natural and semi-natural ecosystems. Enhanced availability of N thus results in increased plant productivity in N-limited vegetation, both through increased growth of existing species and by invasion of new and more productive species (Dise *et al.*, 2011). Above a certain level of primary productivity, local species diversity may decline as the production of a few species that are able to exploit the available N greatly increases. Competitive exclusion of characteristic species by fast-growing nitrophilic species occurs, with rare species at low abundance especially at risk (Dise *et al.*, 2011). The increase in N concentration of plant tissue (see Appendix B) also influences the palatability of the vegetation for herbivores and its sensitivity to pathogens (see Appendix E), something that may also contribute to, or intensify, vegetation changes as a consequence of eutrophication.

3) It may acidify ecosystems (see Appendices B & C). According to Dise *et al.* (2011), soil acidification triggers many long-term changes. Among those are depletion of base cations, reduction of the ecosystem's capacity to remove N (through reduced nitrification and plant uptake rates) and reduced decomposition rate. As a result, species typical of intermediate and higher pH disappear, while those that are acid-resistant increase in frequency.

4) It may exacerbate the impact of stress factors such as pathogens (see Appendix E). Although evidence is still scarce, there are studies demonstrating that plant susceptibility to fungal pathogen attacks and insect attacks is enhanced by increased levels of N (Dise *et al.*, 2011; Appendix E). This is probably a result of a combination of factors, such as increased palatability of plant tissue when tissue concentrations of N are higher, reduced concentrations of defence compounds in plant tissue at elevated N and an overall lower vitality of individual plants exposed to high levels of N (Dise *et al.*, 2011; Appendix E).

According to Dise *et al.* (2011), the severity of the impacts on a species or a community depend on a number of different factors: 1) the duration and the total amount of the N input, 2) the chemical and physical form of N (a major uncertainty today is whether different wet-deposited forms of N, i.e.  $\text{NO}_3^-$  versus  $\text{NH}_4^+$ , have different impacts on biodiversity), 3) the intrinsic sensitivity of the plant and animal species present, 4) the local climate and other abiotic conditions (such as soil acid neutralising capacity and availability of other nutrients) and 5) the past and present land use and management. As a consequence, different ecosystems can show a wide variability in sensitivity and a variety of responses to N deposition.

Generally, boreal ecosystems are regarded to be among the more sensitive to elevated N (Dise *et al.*, 2011). Bryophytes and lichens, which often dominate ecosystems of cold climates, are particularly sensitive to direct foliar N deposition, while forests on nutrient-poor soils are vulnerable to the acidification and eutrophication induced changes caused by elevated N.

## 2.2 Evidence of changes in biodiversity as a consequence of elevated N

There is a large amount of evidence of damage to European terrestrial biodiversity due to elevated N deposition (Dise *et al.*, 2011). Most of the evidence is based on studies of plant communities.

### 2.2.1 Vegetation changes

The most well-known examples of N-induced vegetational changes comes from heathlands in the Netherlands and the UK, where grasses such as *Molinia caerulea* and *Deschampsia flexuosa* have replaced dwarf shrubs such as *Erica tetralix* and *Calluna vulgaris* (Heil & Diemont, 1983; Aerts & Berendse 1988; Aerts *et al.*, 1990; Aerts & Heil, 1993). However, a wide range of ecosystems are sensitive to adverse effects of N on biodiversity. In general, the species richness reduction has been associated to be greatest where the cation exchange capacity is low, conditions are nutrient poor, temperature is low and the increase in production in response to N is greatest. It is the cumulative, rather than the current, rates of N deposition that is related to plant biodiversity (Bobbink *et al.*,





2010; Dise *et al.*, 2011). The most impacted plant functional types are forbs, bryophytes, lichens and nutrient-poor shrubs. Graminoids, on the other hand, are the main beneficiaries of elevated N (Dise *et al.*, 2011).

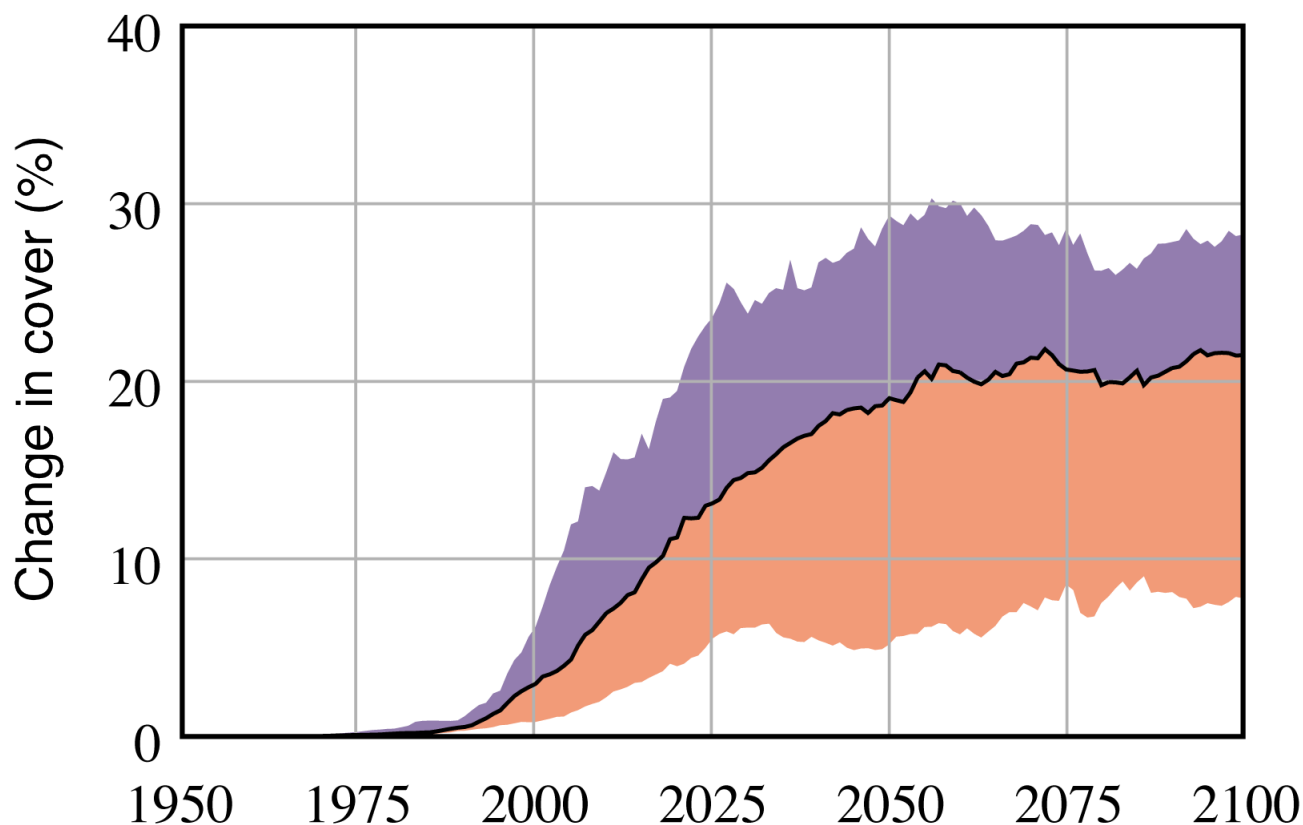
In Scandinavian boreal forests, changes in understory plant community structure as a consequence of N fertilization have been reported since long (Tamm, 1991; Mäkipää, 1994; Kellner & Redbo-Torstensson, 1995). Like for other ecosystems, the abundance or frequency of fast-growing grasses and herbs like *D. flexuosa*, *Festuca altaica*, *Epilobium angustifolium* and *Achillea millefolium* generally increased while the abundance of dwarf shrubs such as *Vaccinium myrtillus* decreased. Results similar to those of the fertilization experiments, i.e. a change towards a more nitrophilous flora, have also been reported as a consequence of N deposition in both temperate (Falkengren-Grerup & Eriksson, 1990; Falkengren-Grerup, 1995) and boreal (Økland, 1995) forests.

More recently, Strengbom *et al.* (2003) investigated the occurrence of understory species in relation to N deposition in 557 coniferous forests distributed across Sweden. In areas with high deposition, *V. myrtillus* was less frequent, less abundant and more susceptible to the leaf pathogen *Valdensia heterodoxa* than in areas with lower levels of deposition. The occurrence of *Vaccinium vitis-idaea* was also negatively correlated with increasing N deposition. For *D. flexuosa*, on the other hand, no such trend was found. In

accordance with the study by Strengbom *et al.* (2003), Strengbom *et al.* (2002) found that elevated levels of amino acids in leaves of *V. myrtillus*, as a consequence of experimental additions of N, caused increased attacks from the parasitic fungus *V. heterodoxa*. The infection, which caused premature leaf loss of *V. myrtillus*, explained four times as much of the variation in grass abundance as N did. The most prominent effect of N as a driver of biodiversity change was thus through its effects on the interaction between the pathogen and its host. The experimental N loads were similar to those deposited over large areas in Europe. That small-scale fertilization experiments might even underestimate the large-scale effects of N deposition on disease incidence, due to the scale-dependent dispersal of the fungus was shown by Strengbom *et al.* (2006). In this article, the authors also noted the need for taking into account the interaction between N and other abiotic factors, such as precipitation changes, for disease development.

Nordin *et al.* (2006) emphasized the importance of the form of N for the response of the vegetation. In a short-term experiment, they showed that both  $\text{NH}_4^+$  and  $\text{NO}_3^-$  was more efficiently used by *D. flexuosa* than by the *Vaccinium* species normally dominating the understory vegetation of boreal forests, but that the addition of  $\text{NO}_3^-$  was more favourable for grass proliferation than was  $\text{NH}_4^+$  addition. The addition of  $12,5 \text{ kg N ha}^{-1} \text{ yr}^{-1}$  of  $\text{NH}_4^+$  had little effect on *D. flexuosa*, while the addition of the same amount of  $\text{NO}_3^-$  favoured grass growth. In contrast, the addition of  $12,5 \text{ kg N}$





**Figure 1. Predicted change over time in vegetation cover of 49 forest stands in Sweden and Switzerland, comparing the scenarios of no controls enacted on N deposition in the 1980s with maximum feasible future reductions in N deposition (both under the IPCC climate scenario A2). The median (50%) estimate is shown with the 10<sup>th</sup> and 90<sup>th</sup> percentiles shaded below and above the median. Modified from Dise *et al.* (2011).**

ha<sup>-1</sup> yr<sup>-1</sup> of NH<sub>4</sub><sup>+</sup> tended to have an effect on bryophytes (as moss N concentrations increased), while adding the same dose of NO<sub>3</sub><sup>-</sup> had no effect. The pathogenic fungi attacking the plants showed no difference in their response to the various forms of N.

Significant vegetation changes in the second generation of trees 20 years after a two-time forest fertilization with 150 kg N ha<sup>-1</sup> was demonstrated by Strengbom & Nordin (2008). Ground vegetation in the fertilized stand was denser, showed decreased species evenness, lower biodiversity and higher Ellenberg N-values compared with the unfertilized control stand. The abundance of dwarf shrubs was reduced by more than 40%, while grasses and some nitrophilous herbs increased by more than 100%. For bottom-layer species, fertilization favoured litter-dwelling bryophytes normally inhabiting more nutrient-rich habitats (such as *Brachythecium species*), while the normally dominant *Hylocomium splendens* and ground-living lichens were disfavoured (Strengbom & Nordin, 2008).

According to Dise *et al.* (2011), the results currently seen probably provide a rather conservative estimate of the long-term impact of N on biodiversity, since much of the evidence originate from areas that have received elevated N deposition over the past 50 to 60 years. The changes we measure today thus probably do not encompass the full range of change. Substantial effects from lower chronic levels of deposition have been emphasized in several

recent publications (Nordin *et al.*, 2005; Emmett, 2007; Clark & Tilman, 2008) and some UK studies re-surveying sites with historical vegetation data showed that species richness was already significantly lower in areas with higher rates of N deposition several decades ago (Dise *et al.*, 2011).

In accordance with experimental field studies and monitoring studies, modelling studies also indicate significant changes as a consequence of elevated N. Using ForSAFE-Veg, Dise *et al.* (2011) investigated the influence of climate change and N deposition on 17 boreal and cool temperate coniferous forest stands in Sweden and 32 Swiss forests (both deciduous, coniferous and mixed, from the plains and the Alps). The results suggested that had the European legislation of the late 20th century to reduce the peak of N deposition (UNECE, 2010) not been enacted, 20% of the ground vegetation of these forests (by cover) would have shifted to a new vegetation type by 2100 (Figure 1; Dise *et al.*, 2011). Dise *et al.* (2011) concluded that although climate change alone will significantly impact the vegetation cover in these forests, the change would be even greater in the absence of N pollution control policies enacted since the 1980s.

Although there are examples of ecosystems recovering relatively rapidly from additions of N (Clark & Tilman, 2008), the rates of recovery from biodiversity losses are generally slow (Dise *et al.*,

2011). Strengbom *et al.* (2001) found no signs of recovery nine years after cessation of N application (34-108 kg ha<sup>-1</sup> yr<sup>-1</sup> for 18 years) to a boreal forest in northern Sweden, although biochemical recovery had started. In a second experiment, examined 47 years after termination of fertilization, N-favoured bryophytes were more abundant, and the most common bryophyte under normal N conditions was less abundant, in the formerly N-treated plots as compared with control plots. No difference in plant species composition was noticed for vascular plants (Strengbom *et al.*, 2001). Similar studies of other ecosystems have also reported recovery to be a slow process (Dise *et al.*, 2011). However, most of these experiments only remove additional experimental N inputs, i.e. the plots still receive ambient N deposition. Few studies have examined the effect of a reduction from current rates of N deposition to pristine deposition rates. One exception is the study by Boxman *et al.* (1998). After six years of replacing wet N deposition by artificial clean rainwater, they found improved growth of pine trees, an increased number of sporocarps of mycorrhizal fungi and a decline in the number of nitrophilic species. Jones (2005) removed vegetated cores from an acid grassland community in Wales to an artificial enclosure and irrigated the cores with deionized water only. In response, the cover of the sensitive moss *Racomitrium* increased, but there was little change in vascular plant cover. According to Dise *et al.* (2011), species that are impacted by direct deposition of N, such as lichens, fungi and bryophytes, may recover rather rapidly once N deposition has been reduced. However, recovery of vascular plant species may take substantially longer time (decades), and significant biogeochemical recovery may have to precede it. Based on the significant vegetation changes at low N doses in both field experiments and large-scale monitoring studies, and the slow recovery of the vegetation after ceasing N input, Nordin *et al.* (2005) suggested that the critical load for understorey vegetation of boreal forests should be lowered to 6 kg N ha<sup>-1</sup> yr<sup>-1</sup>.

### 2.2.2 Fauna changes

Research on the effects of increased N inputs on faunal diversity in semi-natural and natural ecosystems is sparse. However, changes in both nutrient content of the vegetation and the plant species composition can impact the fauna dependent on that vegetation (Haddad *et al.*, 2009; Dise *et al.*, 2011). There is evidence that the frequency of caterpillars, and therefore butterflies and moths, has declined in areas of high N deposition due to both intrinsic vegetation changes and community composition changes (Dise *et al.*, 2011). Dise *et al.* (2011) emphasized, however, that some butterfly or moth species might also profit from N deposition if the preferred plant species of their larval forms become more dominant. In accordance with the reasoning of Dise *et al.* (2011), Öckinger *et al.* (2006), surveying 13 grassland sites in southern Sweden at two separate occasions with an interval of 21 years (1980-1982 and 2002-2003), found that the local extinctions of butterflies were greater for those species whose larval host plants were adapted to low nutrient conditions. For butterflies whose larval host plants were adapted to nutrient-rich conditions, on the other hand, new colonisations were reported. Other studies of grassland ecosystems have also indicated changes in insect communities as a consequence of long-term N loading (Haddad *et al.*, 2009). In a Swedish Norway spruce forest, Lindberg & Persson (2004) reported clear shifts

in the community composition of soil fauna (*Oribatida*, *Collembola*) following N application in solid as well as liquid form. Species number and diversity was not significantly affected, probably as a consequence of an increase in the number of tolerant species that balanced the decrease in non-tolerant species.

Other species reported to be negatively affected by elevated N levels include the ground beetle (*Carabidae*) assemblages of dry open coastal grasslands and the red-backed shrike (*Lanius collurio*), a resident of the coastal dunes of Western Europe (Dise *et al.*, 2011). The reduced heterogeneity of vegetation as a consequence of elevated N (i.e. extensive grass intrusion) is the main factor responsible for the decline of both of these species. Landscape heterogeneity may affect the occurrence of animal species in several different ways; species may depend on specific conditions that are only present in the transition zone between habitats, many animal species require different part of the landscape for different activities (such as reproduction, resting and foraging) and/or heterogeneity creates the possibility of risk spreading (Dise *et al.*, 2011).

### 2.2.3 Fungal changes

With regard to the diversity of EM fungi, response to fertilization seems to vary among species with certain taxa declining in abundance and diversity, while others continue to flourish at higher deposition levels (Lilleskov, 2005; Cudlin *et al.*, 2007). For more information about the effects of N on mycorrhizal infection and colonization of trees, see Appendix A section 4.3.4.





# 3. WHOLE-TREE HARVEST

## 3.1 The impacts of WTH on terrestrial biodiversity

Stem-only harvest (CH) leaves large quantities of dead wood, in the form of residues, stumps and roots, which following final felling may constitute up to 80% of the dead-wood volume (Caruso *et al.*, 2008). In boreal forests, dead wood hosts a variety of fungi, bryophytes, lichens and invertebrates and the importance of dead wood for biodiversity in forest ecosystems has been acknowledged in Scandinavia (Berg *et al.*, 1994; 1995; Siitonen, 2001) and elsewhere (Brassard & Chen, 2006). In Finland, at least 4 000 forest species (but maybe 5 000 species or more – i.e. 20-25% of all species living in forests) are dependent on dead wood (Siitonen, 2001). These values are according to Siitonen (2001) probably valid also for other regions of the boreal forest.

In general, coarser dead wood has been regarded as the most important wood fraction for preserving biodiversity. However, fine woody debris may also host a significant number of species (Kruys & Jonsson, 1999; Nordén *et al.*, 2004) and logging residues have been shown to constitute both shelter and substrate to many species and to protect ground-dwelling organisms and ground vegetation to extreme microclimatic conditions (Gunnarsson *et al.*, 2004). Although it is unlikely that fine woody debris can substitute coarse woody debris when red-listed species are concerned, it is not unimportant for diversity, especially in managed forests where dead wood is relatively rare (Kruys & Jonsson, 1999; Nordén *et al.*, 2004). Furthermore, habitat diversity (a variety of logs, low and

high stumps, slash) has been shown to be of uttermost importance for biodiversity preservation (Heilmann-Clausen & Christensen, 2004; Nordén *et al.*, 2004; Nittérus & Gunnarsson, 2006; Jonsell *et al.*, 2007).

When applying WTH, around 70% of the material that would have been left on site if CH had been practiced is removed (65% of the volume, 77% of the surface area and 84% of the pieces of slash; Rudolphi & Gustafsson, 2005). Furthermore, the amount of old coarse woody debris (CWD) is reduced (after WTH in combination with mounding; Rabinowitsch-Jokinen & Vanha-Majamaa, 2010) and the composition of dead wood altered (Eräjää *et al.*, 2010).

## 3.2 Evidence of changes in biodiversity as a consequence of WTH

### 3.2.1 Vegetation changes

Bergquist *et al.* (1999), investigating 20 clear-cuts located in southern Sweden, found that slash removal had no impact on total plant biomass 1-8 years after harvest. However, there were more plant species present on areas where slash had been retained than on areas from which it had been removed. Generally, the biomass of trees, shrubs, herbs and ferns were higher where slash had been retained, while several dwarf shrubs, grasses and sedges were favoured by





slash-removal (although not statistically significant). That the presence of woody debris had an initial weak suppressive effect on the cover of certain graminoids, such as *Deschampsia flexuosa*, was also found by Olsson & Staaf (1995). Removal of slash, on the other hand, resulted in a lower cover of most vascular plant species and a higher cover of epigeic lichens and *V. myrtilus* 16 years after harvest (Olsson & Staaf, 1995).

In contrast to the above studies, Åström *et al.* (2005), investigating the effects of slash removal on bryophytes and vascular plants in 28 paired boreal forest stands in southern Sweden 5-10 years after clear-cuts, found no effect on vascular plant species composition or richness. However, they did find a significant effect on the species composition of mosses and liverworts. Drought-intolerant and wood-inhabiting species were particularly badly affected 5-10 years after WTH. Furthermore, the species richness of ecological groups (liverworts + mosses) of bryophytes was reduced. This is in contrast with Olsson & Staaf (1995), who found bryophytes to be mostly indifferent to logging residue treatment. However, the effect might depend on the position of the bryophytes in the clear-cut area. Dynesius *et al.* (2008) reported that the effect of branch cover was higher in the centre of clear-cut areas, while it was unimportant close to forest edges. Climatic measurements suggested that the additional sheltering provided by branches buffers during periods of extreme microclimatic conditions. That the community response of bryophytes is very sensitive to microclimate and ecological processes is supported by Åström *et al.* (2007), comparing the difference between clear-cuts on south- and north-facing slopes. North-facing slopes lost fewer bryophyte species, and among those fewer forest species and fewer species associated with wood and bark than south-facing slopes.

### 3.2.2 Fauna changes

With regard to insect species, Gunnarsson *et al.* (2004) found that beetle (*Coleoptera*) populations were negatively affected on the short-term (<1 year) by slash removal in coniferous forest stands in southern and central Sweden. The number of morphospecies per trap was significantly higher where slash had been left than where it had been removed. Furthermore, microhabitat structure (measured by above-ground height of slash heaps) affected both the abundance and the diversity of arthropods. The authors thus concluded that slash heaps left on site may provide important refuges for ground-active beetles and that extensive slash removal is likely to result in impoverished species richness of beetle populations at a local scale. Similar short-term effects were found by Nittérus & Gunnarsson (2006). In contrast to the study by Gunnarsson *et al.* (2004) and Nittérus & Gunnarsson (2006), Nittérus *et al.* (2007), investigating the effects of slash-removal on ground-active beetle populations 5-7 years after clear-cut, found that diversity increased with slash harvest. However, the proportional abundance of forest species in relation to generalist species declined (although certain forest species still existed in high numbers) and the community dominance was shifted towards more generalist species.

That forest fuel piles might act as ecological traps for several uncommon and red-listed saproxylic (wood-living) beetles was shown by Hedin *et al.* (2008). They recorded 39 different species of



beetles and 3528 individuals when investigating forest fuel piles in 12 oak forests in southern Sweden. The highest density of both individuals and species were in the top layer of the piles and 6 of the species were red-listed. Hedin *et al.* (2008) thus recommended that if piles cannot be removed before the insects colonize them, the top layer should be retained. That logging residues host many species important for conservation was also reported by Jonsell *et al.* (2007), who found 22 different red-listed species in logging residues of various tree species in southern Sweden. Most of them were present in deciduous logging residues, implying a greater risk of removing these as compared with coniferous logging residues.

Negative effects of removing logging residues have also been found for nematodes and some other microfaunal species (Sohlenius, 1982), enchytraeids (Lundkvist, 1983) and for springtails, gamasid mites, spiders, predatory insects and dipterous larvae (Bengtsson *et al.* (1998). In the study by Bengtsson *et al.* (1998), animals at higher trophic positions and more mobile animals were most negatively affected by WTH. Some predators actually decreased in abundance by as much as 30-50% after WTH. Enchytraeids and nematodes as well as the biomass of bacteria and fungi, on the other hand, were generally not affected by WTH. Bengtsson *et al.* (1998) thus concluded that the decreased input of plant litter as a consequence of WTH had significant and long-term (in the order of decades) effects on the soil food web, something that according to Walmsley & Godbold (2010) might affect long-term nutrient cycling and thus also productivity of WTH sites.

### 3.2.3 Fungal changes

While a diversity of wood substrates and the presence of decomposing wood is clearly important for preserving the fungal community and the number of fungal species, including red-listed ones (Heilmann-Clausen & Christensen, 2004; Nordén *et al.*, 2004; Josefsson *et al.*, 2010), removal of harvest residues have not yet been found to have any long-term effect on biomass, species richness or frequency of abundant species of saprotrophic fungi (Bengtsson *et al.*, 1998; Allmér *et al.*, 2009) or on biomass and species composition of EM fungi (Mahmood *et al.*, 1999; Hagerberg & Wallander, 2002).

# 4. STUMP HARVEST

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## 4.1 The impacts of SH on terrestrial biodiversity

Stumps represent a significant component of the dead-wood habitat of harvested forest sites. Following WTH, stumps may comprise up to 80% of the remaining dead wood (Egnell *et al.*, 2007). Even a partial SH (removing 66% of stump volume) may remove as much as one fourth of the potential CWD remaining after soil preparation (Rabinowitsch-Jokinen & Vanha-Majamaa, 2010). Furthermore, forest fuel harvesting could potentially have a negative effect on the volume of old dead wood, because more activity on clear-cuts means more damage to the remnant dead wood (Eräjää *et al.*, 2010). That some old dead wood is often destroyed during conventional clear-felling in combination with scarification was shown by Hautala *et al.* (2004) and, recently, a study by Rabinowitsch-Jokinen & Vanha-Majamaa (2010) showed that SH in combination with mounding might possibly augment this effect. Adding SH on top of WTH may thus lead to severe depletion of the dead-wood habitat in managed forests.

Today, the necessity of dead wood for the survival of a wide range of species, of both plant and animal origin, is well-known (see section 3). Stumps have been identified as providing long-lasting dead-wood habitat for a wide range of specialist toadstools, bracket fungi, mosses, lichens, saproxylic beetles, other invertebrates and species supported by the above (Walmsley & Godbold, 2010). The importance of standing dead-wood, high stumps and snags for saproxylic beetles (Jonsell *et al.*, 2004; Hedgren, 2007) have even resulted in guidelines regarding how much of this material that should be left at final harvest (Walmsley & Godbold, 2010). Apart from influencing the amount of dead wood, the removal of stumps also results in reduced habitat heterogeneity. As a consequence, the number of ecological niches and substrates are reduced, something that according to both ecological theory (see for example Crawley, 1997) and practical experiments (see for example Kaila *et al.*, 1997; Ecke *et al.*, 2002; Jonsell & Weslien, 2003) result in lower, rather than greater, diversity. Furthermore, the removal of stumps changes the abiotic conditions of a site, generally resulting in a more exposed and disturbed clear-fell site – conditions that usually favour pioneering species (Olsson & Staaf, 1995; Åström *et al.*, 2005).

## 4.2 Evidence of changes in biodiversity as a consequence of SH

### 4.2.1 Vegetation changes

There are basically no studies available examining changes in biodiversity of plants as a consequence of SH. One exception is the

study by Kaye *et al.* (2008), who examined the effect of SH on the understorey plant communities and on the cover and diversity of functional groups at five forest sites in the north-western part of the US 24-28 years after clear-felling. SH was found to have caused significant changes in the species composition at all sites and several plant species were associated with either control plots or SH plots. In general, the diversity of graminoids, forbs and introduced species increased in SH areas. SH also resulted in reduced cover and diversity of shrubs and increased cover of graminoids and forbs at some sites. The authors thus concluded that SH had lasting impacts on plant communities, and that it may make them more vulnerable to colonization by introduced species. Walmsley & Godbold (2010) emphasized the impacts of SH on changes in the composition, abundance and growth of various plant species as one of the key research questions for the future, since increases in the prevalence of invasive non-forest vegetation may lead to an increased necessity for chemical herbicide applications in managed forests.

Apart from Kaye *et al.* (2008), the only studies available are those investigating the importance of stumps as a substrate for the diversity of bryophytes and lichens. Generally, stumps seem to be more important to overall lichen diversity as compared with bryophyte diversity (Caruso & Rudolphi, 2009). When investigating the importance of substrate age and quality to lichen and bryophyte diversity on cut surfaces of stumps at 30 sites in south-central Sweden, Caruso & Rudolphi (2009) found that there were more lichen species per stump. Furthermore, while several uncommon lichens were found, only common bryophytes were registered.

Comparing stumps and forest residue as substrates for lichen diversity, Caruso *et al.* (2008) found that stumps hosted a greater number of lichen species when equal volumes of stumps and forest residue were compared. The majority of species found on both substrates were more frequent on stumps, and stumps also had a higher number of unique species and species identified in the literature as nationally rare or uncommon. The poorer lichen flora on slash as compared with stumps is probably due to stumps offering a greater diversity of microhabitats (Caruso *et al.*, 2008). Recently, Rudolphi *et al.* (2011) found that stumps in thinned stands were more species rich than those in clear-cuts. They thus suggested that SH in open, clear-cut stands will probably have a minor effect on bryophyte diversity, and that conservation efforts in thinned stands might be more important for promoting diversity of bryophytes in managed forests. However, the species composition on stumps in thinned stands differed from that of stumps in clear-cut stands, indicating the importance of a variety of substrates for bryophyte diversity (see also section 4.2.2).



#### 4.2.2 Fauna changes

As for vegetation, there are very few studies investigating the effects of SH on fauna diversity. However, several studies have demonstrated that retention of managed high stumps (created by cutting some trees at several meters height above ground during tree harvesting) can benefit insects as well as fungi (Lindhe & Lindelöw, 2004; Jonsell *et al.*, 2004; 2005). The importance of ordinary low stumps as habitats for saproxylic species is considerably less well-known. Several recent studies have indicated that low stumps may host a wide variety of species. Hedgren (2007), investigating fresh stumps of Norway spruce at five sites in central Sweden, found that most taxa of early-arriving bark- and wood-boring beetles and associated insect enemies occurred in both low and high stumps, even if high stumps had slightly better values than low stumps for parasitoids in terms of occupancy, density and Shannons index values (index value for diversity). The authors thus concluded that there is a positive effect on insect biodiversity also of low stumps.

In accordance with Hedgren (2007), Hjältén *et al.* (2010) also reported no significant differences in beetle abundance or species richness between low stumps, high stumps and logs 5-7 years after clear-cut at ten different localities in northern Sweden. However, there were clear differences in the assemblage composition between all substrate types. Although the saproxylic beetle species that were common in the low stumps in their study have also been reported from other types of habitats and substrates, Hjältén *et al.* (2010) concluded that man-made stumps are important (Table 1). This is because the natural coarse-woody

debris is rare in the managed forest landscape and, given the considerable volume of low stumps in relation to high stumps and logs on clear-cuts, substantially more individuals are expected to emerge from low stumps than from other dead wood substrates (Hjältén *et al.*, 2010). Furthermore, low stumps serve as an important substrate also for other saproxylics, i.e. lichens and bryophytes (see section 4.2.1). Hjältén *et al.* (2010) thus predicted that extensive harvesting of low stumps is likely to be detrimental to saproxylic species connected to early successional stages and species connected to more sun-exposed habitats – at least on stand level. Similar conclusions were drawn by Walmsley & Godbold (2010). The effects at landscape level are more difficult to predict as they depend on the proportion of clear-felled areas that are subjected to SH (Hjältén *et al.*, 2010).

In contrast to the above mentioned studies, Abrahamsson & Lindbladh (2006), investigating high and low stumps of Norway spruce at 16 sites in southern Sweden, found that the number of species of saproxylic beetles in high stumps at ground level were significantly higher than the numbers found in high stumps at breast height or in low stumps. Some beetle species clearly preferred the high stumps over the low stumps, since the high stumps provided new habitat types for beetles as compared with normally cut low stumps. However, in coherence with the results of Hjältén *et al.* (2010), the beetle assemblages in low stumps were to some extent complementary to the ones in high stumps (see also Lindbladh & Abrahamsson, 2008).





**Table 1.** Simper analysis performed on emergence trap data showing the most important species with regard to differences between low stumps and high stumps. The species are listed in decreasing order of importance. The percentage shows the percentage contribution to the dissimilarities in assemblages. LS=low stumps, HS=high stumps. Modified from Hjalten et al. (2010). For information on differences between stumps and logs, see Hjalten et al. (2010).

Species	%	Difference
<i>Scaphisoma agaricinum</i>	4.73	LS>HS
<i>Anaspis marginicollis</i>	4.07	HS>LS
<i>Crypturgus pusillus</i>	3.96	HS>LS
<i>Ampedus tristis</i>	3.15	LS>HS
<i>Enicmus rugosus</i>	3.14	LS>HS
<i>Dasytes niger</i>	2.67	HS>LS
<i>Cerylon histeroides</i>	2.54	HS>LS
<i>Cis punctulatus</i>	2.44	HS>LS
<i>Asemum striatum</i>	2.41	LS>HS
<i>Abdera triguttata</i>	2.37	HS>LS
<i>Euplectus punctatus</i>	2.31	LS>HS
<i>Hylastes cunicularius</i>	2.01	LS>HS

#### 4.2.3 Fungal changes

We have not found any studies on long-term effects of SH on fungal communities. Lindhe *et al.* (2004) compared the saproxylic fungi succession of cut logs and high stumps of several different species in central Sweden during nine years following clear-cut. They found that both logs and stumps hosted diverse communities of fungi, including red-listed species. However, logs hosted a higher number of species, and more red-listed species, than stumps (annually and cumulatively). Fungal communities on logs were also more diverse,

with a higher proportion of unique species. Stumps hosted more common, or rather common, species. The authors emphasized, however, that several of these common species were species contributing to decomposition and recycling of nutrients, functioning as precursors for rarer fungi that utilize wood in later stages of decay and/or being of importance for saproxylic insects. Allmér (2005) suggested that because stumps generally host a wide variety of wood-dwelling fungi, their removal is likely to influence the occurrence of fungi more severely than what WTH has been shown to do.





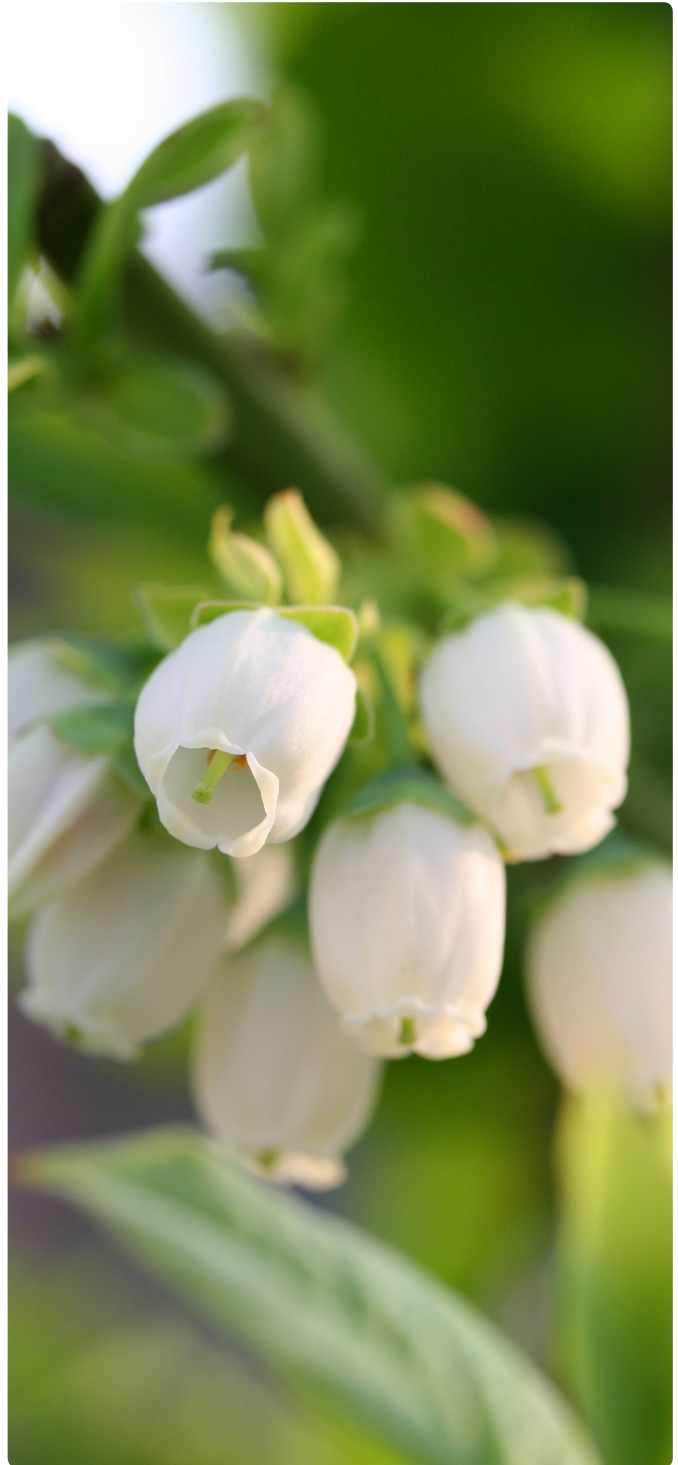
# 5. CONCLUSIONS

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Traditionally, the overall effects on ground vegetation of commercial forest fertilization have been considered to be of moderate magnitude (Högbom & Jacobson, 2002; Olsson & Kellner, 2006). However, the results presented by Dise *et al.* (2011) and in several of the studies by Joachim Strengbom and Annika Nordin (Strengbom *et al.*, 2002; 2003; Nordin *et al.*, 2005; Strengbom & Nordin, 2008) question this idea. Strengbom & Nordin (2008) writes in their conclusion that “Although we acknowledge that the fertilization may not result in a completely new type of vegetation, i.e. a total dominance of grasses and herbs instead of dominance by dwarf shrubs, the residual effects from fertilization were substantial, indicated by large changes in abundances of common species”. They continue: “...as our data show that these effects extend over more than one forest generation, the differences between unfertilized and fertilized stands will likely be reinforced if the upcoming forest generation is fertilized in the same way as the previous one. Hence, if forest fertilization is implanted as a standard silvicultural practice, it seems likely to assume that fertilized forests eventually will be of a completely different vegetation type than unfertilized forests. From this it is questionable whether forest fertilization as performed today in Sweden, as well as in other Nordic countries, is in accordance with principles of long-term sustainable forestry and preservation of biodiversity.” Strengbom & Nordin (2008) thus suggest that forest fertilization as a silvicultural practice must be developed and modified fertilization practices (such as lower N doses or less frequent fertilizations) tested before N fertilization can be launched as a standard silvicultural measure. From a biodiversity perspective, and based on the extensive negative effects reported in the studies included in this review, we cannot but agree.

The number of studies on the effects of WTH and SH on biodiversity are very limited and the information of their potential effects is thus mainly based on the importance of stumps and forest residues as substrates for various plant and animal species. Despite the indisputable importance of these substrates for many plant and animal species, the effects of forest fuel harvesting on forest biodiversity have often been concluded to be small, or even negligible, as compared with the effects of clear-felling. Furthermore, it has often been accentuated that short stumps is not a natural or preferred substrate, but a substrate that has been created by humans as a consequence of the intensified forestry during the past century and thus is of limited importance for biodiversity. The literature reviewed in this report do not support these conclusions. Also Walmsley & Godbold (2010) object to these kinds of conclusions, stating that although clear-felling alone lead to significant impacts on forest species, the removal of residues and stumps will substantially exacerbate these effects. Other arguments often presented in order to reduce the negative effects of forest fuel harvesting are that most results are based on stand level studies and that as long as there are enough stumps and residues on landscape level, WTH and SH do not pose a problem for biodiversity. Important to consider, however, is that it is the environmental conditions at the individual stands that shape the environmental qualities at the landscape level.

We conclude that most available studies present negative effects of N fertilization and potentially negative effects of forest fuel harvesting on forest biodiversity. These practices must thus be regarded as genuine threats to forest biodiversity. However, the extent of negative effects of these practices in relation to other threats are currently difficult to evaluate, especially considering the lack of information with regard to the effects of WTH and SH.



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