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Bioenergy from forests: Impact of logging residues on the carbon and nitrogen cycles

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Academic dissertation

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ABSTRACT

Sustainable forest management practices are crucial for minimizing environmental impacts, and to keep forests and the underlying soils healthy to maintain productivity, and to improve adaptability to climate change and mitigate it in the long-term. There, carbon and nitrogen cycling play crucial roles in proper functioning of forest ecosystems.

The overall aim of this thesis was to explore the effects of logging residues on the early-stage dynamics of the main nitrogen and carbon cycling processes in upland forest soils after final felling. A specific aim was to compare the effects of logging residue piles consisting of different tree species, i.e., Scots pine (Pinus sylvestris L.), Norway spruce (Picea abies (L.) Karst.), and silver birch (Betula pendula Roth.). A special attention was given to the response of soil to different amounts of residues.

The effect of logging residues on soil chemical properties and processes was stronger in the organic layer than in the mineral soil layer. Logging residue piles stimulated carbon and nitrogen cycling, especially net nitrification within the first year after the residue treatment. Subsequently, nitrogen was lost via leaching as nitrate and gaseous emission as nitrous oxide. Nitrous oxide fluxes were generally low, although higher in the plots containing logging residues. Spruce residues tended to stimulate nitrous oxide emissions for longer than the residues of the other tree species. Depending on the tree species, nitrous oxide production originated from both autotrophic nitrification and denitrification. Nitrogen concentrations in the soil percolate water already accelerated one year after the establishment of the residue piles. Logging residues increased nitrate and ammonium concentrations, which were highest under birch residues. The effect of logging residue piles on soil was stronger when compared to milder tree species effects. Biological nitrogen fixation activity tended to be higher in branches than in needles or leaves, and higher in coniferous residues.

The results of this study raise the importance of more even distribution of logging residues on the forest floor instead of piling them to avoid harmful environmental effects and to maintain soil nutrient balance. The ability of soils to retain nitrogen needs additional attention in forest management practices, especially in boreal areas where nitrogen limits the growth of forests.

Keywords: logging residues, tree species, nitrogen cycling, carbon cycling, clear cutting

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Tuusula, September 2022

Tiina Törmänen

LIST OF ORIGINAL ARTICLES

The thesis is based on the following research articles, which are referred to in the text by the Roman numerals I–IV.

I. **Törmänen, T**., Kitunen, V., Lindroos, A.-J., Heikkinen, J. and Smolander, A. 2018. How do logging residues of different tree species affect soil N cycling after final felling? For Ecol Manage 427, 182-189. https://doi.org/10.1016/j.foreco.2018.06.005.

II. **Törmänen, T**., Lindroos, A.-J., Kitunen, V. and Smolander, A. 2020. Logging residue piles of Norway spruce, Scots pine and silver birch in a clear-cut: Effects on nitrous oxide emissions and soil percolate water nitrogen. Sci Total Environ 738, 139743. https://doi.org/10.1016/j.scitotenv.2020.139743.

III. Smolander, A., **Törmänen, T**., Kitunen, V. and Lindroos, A.-J. 2019. Dynamics of soil nitrogen cycling and losses under Norway spruce logging residues on a clear-cut. For Ecol Manage 449, 117444. https://doi.org/10.1016/j.foreco.2019.06.041.

IV. **Törmänen, T**. and Smolander, A. 2022. Biological nitrogen fixation in logging residue piles of different tree species after final felling. J Environ Manage 303, 113942. https://doi.org/10.1016/j.jenvman.2021.113942.

AUTHOR'S CONTRIBUTION

Tiina Törmänen performed most of the experimental (field and laboratory) work, calculated and interpreted the results and was the main author in papers I, II and IV. In paper III she participated to the experimental work and manuscript preparation.

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ABBREVIATIONS

C	carbon
CO_2	carbon dioxide
DOC	dissolved organic carbon
GHG	greenhouse gas
LR	logging residue
N	nitrogen
N_2	dinitrogen, nitrogen gas
N_2O	nitrous oxide
NH_4-N	ammonium-nitrogen
NO_3-N	nitrate-nitrogen
SOH	stem-only harvesting
WTH	whole-tree harvesting

1 INTRODUCTION

1.1. Forest-based biomass harvesting

Globally, and especially in Scandinavia, forest biomasses are receiving increased political attention as a source of bioenergy to substitute fossil fuels, and also other materials such as plastics and composites (CCFM 2018) for biobased products. The use of forest biomasses has potential to limit the growth of atmospheric greenhouse gas (GHG) concentrations, although the presumed carbon (C) neutrality is often questioned (Laganiere et al. 2017; Mather-Gratton et al. 2021), as are other environmental aspects too (Ranius et al. 2018; Repo et al. 2020). Especially, ongoing debate is questioning the trade-off between forest-based biomasses utilized in the circular bioeconomy and fossil-based materials (Achat et al. 2015; Titus et al. 2021), as the actual long-term impacts on health of the whole forest ecosystem, C sequestration, and site fertility remain unclear.

By the year 2020, the goal of The Renewable Energy Directive of the European Union (EU) was to reduce GHG emissions by 20 %, and at the same time increase the consumption of renewable energy to 20 % of the total consumption (European Commission 2009). In Finland, the share of wood-based fuels for energy usage has increased considerably, and already accounts for almost one-third of the total energy consumption (Ylitalo 2019). Small-sized trees, logging residues (LR; tree tops, branches, needles, etc.), and in some cases stumps, are the main raw material for wood chips, and together with sawdust and bark residues from the wood industry are used in large power plants for heat and power production.

The aim of the Finnish energy and climate strategy is to increase the use of wood chips as an energy source in heating and power plants from around 7.2 (in 2017) to 14–18 million m^3 per year by 2030 (Ylitalo and Torvelainen 2018). For instance, in 2018, the consumption of forest wood chips slightly increased to 7.4 million m^3. In addition, the use of LRs increased from 2.3 (2017) to 2.7 million m^3 (Ylitalo 2019).

Because of the increasing need for forest-based biomasses, harvesting is sometimes executed with whole-tree harvesting (WTH), contrary to the more traditional stem-only harvesting (SOH). In SOH, nutrient-rich LRs, i.e., branches and tree tops, are left on-site (Vanguelova et al. 2010). In contrast, in WTH, both tree stems and LRs are harvested, and WTH thus increases the removal of nutrients from the forest (Palviainen and Finer 2012). Removal of LRs and the effects on C and nutrient cycling (especially N, P, Ca, Mg, K) has been widely studied all over the world and in different forest ecosystems (Achat et al. 2015; Thiffault et al. 2011). In Finland, LRs are mainly harvested from sites after final felling, and currently less from thinning forest stands (Helmisaari et al. 2011, 2014).

1.2. C and N cycling in forest soils

Forests and forest soils are major contributors to global C sequestration and storage, and are thus largely responsible for the consistency of the global C sink. The boreal upland forest soils are typically podzolized (Figure 1) (Blake et al. 2008) where C is stored in the surface organic layer (O horizon) and in the mineral soil (A, E, B, and C horizons). In podzolized soils, the vertical distribution of C content per volume decreases with depth being highest in the organic layer (De Luca and Boisvenue 2012).

Figure 1. A typical podzolized soil profile, where most of the C is stored in the upper soil horizons.

Both belowground and aboveground litter production affects soil C stocks. Aboveground litterfall from trees and ground vegetation is estimated to be the most important source of C inputs to the forest floor (Nordén 1994). In addition to the aboveground litterfall, the roots and root-associated fungi in boreal forest soils play a crucial role in long-term C sequestration (Clemmensen et al. 2013; Högberg et al. 2008). C is accumulated into boreal forest soils because of its slow decomposition (Lupi et al. 2013). Factors affecting these slow decomposition rates are low temperatures, recalcitrant litter rich of lignin and polyphenols, and the limited availability of nutrients (Prescott et al. 2000).

In forest soils, C and N cycling are tightly coupled together (Gruber and Galloway 2008). In boreal forests, biological N fixation adds N to the system by root nodule symbiosis between N-fixing bacteria and alder trees or legumes, by N-fixing bacteria associated with mosses, and by free-living bacteria or archae in soil and dead wood (De Luca et al. 2002; Jurgenssen et al. 1984, 1992; Leppänen et al. 2013; Rinne et al. 2017; Smolander 1990). Biological N fixation in boreal areas is estimated to be 0.3–4 kg ha^{-1} y^{-1} (Granhall and Lindgren 1980). In addition, it is possible for N to be added to the system via lightning (Draphco et al. 1983), dry and wet deposition (Korhonen et al. 2013), or commercial fertilization (Saarsalmi & Mälkönen 2001). Annual N deposition is in southern Finland about 4 kg/ha and in the northern Finland about 1 kg/ha or less (Salemaa et al. 2019). A typical dose of N in forest fertilization is 150 kg ha^{-1} (Nohrstedt 2001; Pettersson and Högbom 2004). In Finland, forest fertilization has been active from the 1960's and activity peaked out in the

1970's when the annual fertilization areas could reach 150 000 ha. In the 1980's areas decreased and by the 1990's, annual fertilization areas were very low. Interest towards forest fertilization is increasing again since fertilization is considered as an efficient tool to enhance the production of renewable resources and increase C sequestration (Hedwall et al. 2014; Saarsalmi and Mälkönen 2001; Smolander et al. 2022).

Boreal forests soils have high amounts of N, but N is a limiting nutrient for forest growth (Näsholm et al. 1998). This is because most of soil N is bound in complex proteinaceous structures and only a small portion of soil N is in plant-available forms, inorganic (NH_4-N and NO_3-N) or in small-molecular-size organic N in soil solution (Näsholm et al. 2009). Depolymerization of large organic N compounds into smaller organic N compounds, and further net N mineralization, determines the amount of labile inorganic N available for plant uptake. Net N mineralization is determined as the difference between gross N mineralization and N immobilization in the microbial biomass.

N mineralization is estimated to cover 0.5–3 % of the total amount of N in boreal forest soils annually (Persson and Wiren 1995). In boreal forests, soils are the greatest pool of N (1000–2500 kg N ha^{-1}), thus 80–90 % of N is stored in soils (Korhonen et al. 2012; Mälkönen 1974). Annually, 1–2 kg N ha^{-1} is accumulated in these soils (Berg and Dise 2004).

Another important N transformation is nitrification, an undesirable aerobic process, where ammonia (NH_3-N) is converted via microbial processes to highly leachable nitrate-nitrogen (NO_3-N), a process that simultaneously acidifies soils. Normally, nitrification activity is low, so NO_3-N occurs in low concentrations in boreal forest soils (Ambus et al. 2006), and in boreal areas NH_4-N is most commonly utilized by plants. This causes immobilization of inorganic N and decreases net N mineralization (Scott & Binkley 1997). As a cation, it is possible for NH_4-N to bind to negatively charged mineral and humus surfaces. In contrast, NO_3-N occurs as an anion, and is highly leachable from forest ecosystems to surrounding water systems, thus causing a potential risk to water quality (Lepistö et al. 1995; Vitousek and Melillo 1979) depending, for instance, on weather conditions, site and catchment properties and forest management history.

Gaseous N_2O is possible to produce in nitrification and simultaneously, the anaerobic process denitrification may occur. Then NO_3-N can be reduced to N_2O and N_2, allowing gaseous N losses from terrestrial systems to the atmosphere. In boreal areas, N_2O fluxes are normally low (Baggs and Philippot 2010; Pihlatie et al. 2007; Rennenberg et al. 2009; Vitousek and Melillo 1979). Closed, natural N cycling can be turned to an open process as a result of human activities, including harvesting of forests, especially clear-cutting (Smolander et al. 2000), which enhances N losses (Gundersen and Rasmussen 1990; Rósen and Lundmark-Thelin 1987; Vitousek et al. 1997).

1.3. Influence of logging residue harvesting on soils

Influence of LR harvesting on soils differs somewhat between thinning and final felling. Thinning sites may be rather sensitive to nutrient removal with LRs (Jacobson et al. 1996), since growing tree stocks demand and consume more nutrients, such as N. In some thinning stand experiments, soil fertility, volume increment, amounts of exchangeable base cations, and microbial activities in C and N cycling decreased after LR harvesting (Dighton et al. 2012; Helmisaari et al. 2011; Jacobson et al. 2000; Rosenberg and Jacobson 2004; Smolander et al. 2008, 2010, 2013; Tamminen et al. 2012; Thiffault et al. 2011).

However, results have not always been consistent. Changes in soil organic matter composition have been observed in coniferous thinning stands after WTH, such as long-term decreases in concentrations of terpenes in the organic layer (Smolander et al. 2010, 2013). In current practices, trees in thinning stands are mostly harvested as limbed trees, to avoid removal of nutrients from the site.

In clear-cut sites, some long-term changes are noticed in soil properties after the harvest of LRs; for instance, decreases in the concentrations of exchangeable base cations and in net N mineralization (Brais et al. 2002; Kaarakka et al. 2014; Olsson 1996a, 1996b; Piatek and Lee Allen 1999; Saarsalmi et al. 2010; Smolander et al. 2015; Thiffault et al. 2011). Thus, WTH in clear-cutting may lower nutrient availability in the soil (Johnson & Todd 1998; Saarsalmi et al. 2010), although no observable impact on soil nutrients or site productivity is also possible (Slesak et al. 2017; Voldseth et al. 2011). With regard to soil microbial activities and nutrient status, WTH and SOH were similar in four of five coniferous study sites observed 10 years after clear-cutting (Smolander et al. 2015).

In addition, five years after clear cutting, the concentrations of terpenes (except of monoterpenes) and tannins were noticeably higher in soil under Norway spruce LRs than in control soils with no LRs (Adamczyk et al. 2016). The few clear-cutting studies available that focused on the effects of LRs per area unit reported increases in the concentrations of N, including NO_3-N, in soil percolate water under coniferous LRs (Clarke et al. 2018; Lindroos et al. 2016; Ring et al. 2015; Rosén and Lundmark-Thelin 1987; Wall 2008).

As described above, modern LR harvesting accomplished with heavy machines is not completely problem-free. In some cases, profitability may limit LR harvesting, as may environmental issues. In modern harvesting techniques, LR piles are created on the forest floor when stems are mechanically limbed on site, and also when LRs are spread on the forest floor to improve soil-bearing capacity and to prevent the creation of deep wheel tracks by off-road traffic (Ring et al. 2019). Thus, LRs may be unevenly distributed on the forest floor, with piles covering 10–60 % of the soil surface (Nurmi 1994; Rosen and Lundmark-Thelin 1987).

To avoid too intense removal of LRs and to ensure sufficient soil conditions and nutrient balance, in Finland it is recommended to leave 30 % of the green LRs on site (Äijälä et al. 2019) (Figure 2). This value is based on The Best Practice Guidelines for Sustainable Forest Management. For example, approximately 20–40 % of the LR dry mass of Norway spruce is left on site (Nurmi 2007; Peltola et al. 2011), although the amount can vary from 4 to 89 %, with the highest values found in Nordic countries (Thiffault et al. 2014). Foliage forms the largest pool of N when compared to woody components of LRs (Hyvönen et al. 2000), thus the drying of foliage and shedding of needles to the forest floor can decrease the removal of nutrients, especially N and P, from forest ecosystems (Olsson et al. 2000; Thiffault et al. 2011).

Figure 2. Fresh LR piles of spruce left on the forest floor in practical forestry.

1.4. Interactions between tree species and soil processes

The soil responses to LR of different tree species are poorly known. When the effects of tree species on soils are discussed more widely than the LR perspective alone, vegetation is one of the Jenny's soil formation factors (Jenny 1941), and tree species be a possible surrogate for it. Tree species affect the microclimate of forests differently, by changing the shading intensity, precipitation and interception through canopy structure, and the depth of soil freezing by snow cover thickness (Ylivakkuri 1960; Hyvärinen 1990; Augusto et al. 2015; Huuskonen et al., 2021). There are also species-specific differences between trees in the below-ground rooting system and root exudates (Augusto et al. 2015). A close relationship can be noticed between tree species and soil physical and chemical properties.

Tree species also affect soil processes and properties through the composition of litter and litter fall mass (Augusto et al. 2015; Huuskonen et al. 2021). These effects are also relevant when considering the effects of LR on soils, although LRs predominantly contain green leaves instead of the usually brown leaves found in litter fall, and more woody material than litter.

For boreal and temperate tree species, the N content is typically ca. 0.8–2.5 % of the dry mass, being generally higher in broadleaved species than evergreen coniferous species (Berg and McClaugherty 2003). As a result of the decomposition of foliage material and LRs, tree species affect the chemical composition of soil solutions and percolated soil water through substances leached from the litter (Johansson 1995; Kiikkilä et al. 2012; Prescott et al. 2000).

Differences in the water permeability of foliage material have been suggested to be one reason for greater DOC (dissolved organic carbon) leaching from birch leaves compared to coniferous needles (Kuiters 1993). The effect of tree species on the overall composition of soil DOC, described as the molecule-size distribution and chemical nature, was minimal (Smolander and Kitunen 2002, 2011), but there are large differences in the concentrations of several individual compounds or groups of compounds in litter, soil organic matter, and in dissolved organic matter (Kanerva et al. 2008; Kiikkilä et al. 2012).

Needle litter, which has a surface layer consisting of waxes, lignin and other polyphenolic compounds, decomposes slower compared to leaf litter. In comparison, leaf litter that is enriched with nutrients and more easily leachable and decomposable water-soluble compounds, decomposes faster (Hongve 1999; Nykvist 1963), simultaneously releasing nutrients faster compared to needle litter.

Differences in the degradation rate of litter between tree species are particularly evident during the initial stages of decomposition. Foliage plays a role in rapid tree species effects on soils, since woody components decompose more slowly (Palviainen et al. 2004; Prescott et al. 2000; Smolander et al. 2008). For instance, Palviainen et al. (2004) found that the mass of foliage after two years decreased in the order birch > pine > spruce. In branches, differences between tree species were negligible, with pine having the largest mass loss. In addition, the mass loss of spruce needles was reported to be more than 30 % during the first two years, while that of the main branches was about 10 % (Smolander et al. 2008). Thus, the type of material being decomposed matters in decomposition processes.

In many studies, Norway spruce increases the soil acidity and decreases the concentrations of exchangeable nutrients when compared to birch (Mikola 1985; Nihlgård 1971; Priha and Smolander 2000). Birch is generally seen as a positive component that improves multiple soil properties, thus improving growing conditions (Gardiner 1968) via increased soil pH and base saturation, as well as nutrient concentrations and microbial activity (e.g., Côte et al. 2000; Kanerva and Smolander 2007; Mikola 1985; Priha and Smolander 1999; Smolander et al. 2005; Smolander and Kitunen 2002). Further, the effect of tree species may differ to some extent when comparing the weathering rates of minerals (Augusto et al., 2000), and even soil pedogenesis detected as podsolization (Nihlgård 1971) in boreal forest soils (Lindroos et al. 2011). This is related to differences in the soil pH and production of chelating organic acidic compounds that form complexes with metals (Raulund-Rasmussen et al. 1998) that eventually have an impact on soil C storage. Changes in soil processes related to nutrient and C cycling can lead to alter C sequestration and net primary production (Augusto et al. 2002; Vesterdal et al. 2013). In general, detectable differences in soil processes among tree species may demand rather long-term investigation. However, as emphasized above, it is difficult to distinguish litter effects.

In addition to the tree species differences discussed above, tree species greatly influence the potential amount of LRs harvested from a site. When comparing similarly-sized treetops of coniferous species, Norway spruce has double the dry mass compared to Scots pine (Hakkila 1991). On clear-cut stands in Finland, the average amount of LRs consisting of Norway spruce and Scots pine were 20–60 and 15–30 Mg ha^{-1}, respectively, and the corresponding N values were 150–300 and 75–150 kg ha^{-1} (Hakkila et al. 1998).

2 AIM OF THE STUDY

The overall aim of this thesis was to explore the effect of LRs on the early-stage dynamics of the main N and C cycling processes in upland forest soils after final felling. A specific aim was to compare the effects of LRs from three different tree species, i.e., Scots pine (*Pinus sylvestris* L.), Norway spruce (*Picea abies* (L.) Karst.), and silver birch (*Betula pendula* Roth.).

Forest soils differ vertically and horizontally; hence, the humus and mineral soil layers (articles I and III) were studied separately. This division allowed to determine the depth of the influence of LR on forest soils. To get a more general picture of the response of soil to LRs, the study sites also differed in parent materials and the time elapsed since harvest (articles I, II, and IV in comparison to III).

An important aim was to monitor how LR piles consisting of different tree species (article II) affect C and N losses after final felling via leaching and N_2O emissions, and how these losses are related to C and N cycling processes. Leaching and gaseous losses were studied in the field, and the processes contributing to N_2O emissions were determined in the laboratory (articles II and III). The composition of C and N in percolation water was also characterized. To identify the potential ability of forest soil (article III) and LRs of different tree species (article IV) to add external N to the forest ecosystem, the biological N fixation was identified.

Hypotheses:

H1. LR piles strongly stimulate soil processes associated with C and N cycling in boreal forests, particularly net nitrification.

H2. Intensified net nitrification increases N mobilization and the risk for N losses via leaching and gaseous emissions after clear-cutting.

H3. The intensity of the effects and dynamics of C and N cycling processes differs depending on the tree species.

H4. Biological N fixation occurs in LRs and in forest soils under LRs.

H5. Biological N fixation differs depending on the tree species, and also on the compositional material (soil, leaves, needles, and/or branches).

3 MATERIALS AND METHODS

3.1. Study sites

To study C and N cycling, four forest stands located in Lapinjärvi (Mysmalm 1 and 2 and Latokartano) and in Evo, all in southern Finland, were used in this study. The Mysmalm 1 site was further divided into Experiments I and II. The basic information of the sites is presented in Table 1.

Table 1. Basic information of the study sites.

Experiment	Lat.	Long.	Soil texture	Site type	Soil type	Tree species treatment	Publication
Mysmalm 1	60	26	sorted sand	V. myrtillus	Podzol	pine, spruce, birch	I, II, IV
Mysmalm 2	60	26	sorted sand	V. myrtillus	Podzol	spruce	III
Latokartano	60	26	till	V. myrtillus	Cambisol	spruce	III
Evo	61	25	sorted sand	V. myrtillus	Podzol	spruce	III

Figure 3. General view of the Mysmalm 1 Experiment I study site (articles I, II, IV) three years after establishment of LR piles.

The *Mysmalm 1* site (Figure 3) was used for articles I, II, and IV. Originally, this site was an 80-year-old spruce-dominated mixed stand. The stand was clear-cut in September 2014. Experiment I (16 plots, A = 1.5 m x 2 m = 3 m^2, four replicates each) was established three days after clear-cutting to study basic soil characteristics (pH, total C, total N, DOC), C and N transformations in the soil (net N mineralization, net nitrification, C mineralization, microbial biomass C and N, N_2O fluxes and production), and biological N fixation in the LR.

Experiment II (article II) (Figure 4) was established in the same clear-cut forest to study C and N losses. LR piles (12 plots, A = 0.6 m x 0.6 m = 0.36 m^2, three replicates) were established on top of the soil column lysimeters in September 2015.

Figure 4. General view of the Mysmalm 1, Experiment II study sites (article II) two years after establishment of LR piles.

To study the effects of LR piles, and especially the effects of different tree species at study site 1, the study plots (articles I, II, IV) were established on the same geological formation (sorted sand), and on the same clear-cut area. Therefore, we can assume that similar soil formation factors including human intervention, climate, parent material, time, and topography, were relatively constant, and affected the soil properties of a site in a similar way. The LRs of pine, spruce, and birch were used as a surrogate for biological soil formation factors affecting soil properties, thereby allowing a direct comparison of the tree species.

Article III presents the results from the *Latokartano site* (Figure 5), Evo, and Mysmalm 2 sites. The Latokartano study site had originally been a spruce-dominated mixed forest. The site was clear-cut in July 2012, and in August 2012, 0, 10, and 40 kg/m^2 of LRs were added above lysimeters to monitor pH, NH_4-N, NO_3-N, and DON (dissolved organic nitrogen) in soil percolation water. To study the basic soil characteristics and C and N transformations in the soil, LR piles consisting of spruce LRs (0 and 40 kg/m^2 of fresh LRs) were placed in plots (A = 3 m^2, 4 replicates) that were established one year later, in June 2013.

Evo and Mysmalm 2 sites were originally Norway spruce-dominated stands. Sites were clear-cut in 2008 and LR piles (6 study plots, A = 1 m2, 2 replicates) in each site were established in 2009. LR piles had 0, 10, and 40 kg m-2 of fresh LRs. Soil basic characteristics and N cycling processes were studied six years after that.

All the study sites were natural site types for all three tree species used in this study. To create the piles, the fresh LRs were harvested from the study site or from the adjacent forest. The LR piles used in this study were built to be of the same order of magnitude (40 kg m-2) as the piles left on the forest floor after clear-cutting, or evenly distributed on the forest floor (10 kg m-2), comparable to when harvesting of LRs is executed with modern machinery-harvesting techniques.

Figure 5. General view of the Latokartano study site (article III) three years after establishment of LR piles.

3.2. Soil sampling and determinations of C and N transformations

To study C and N transformations (articles I and III), soil sampling was carried out as follows: In the Mysmalm 1 site, soil samples were taken just before the establishment of the LR piles in 2014 (September), and twice in both 2015 and 2016 (May and November). At the Latokartano site, sampling was carried out just before establishment of the LR piles in June 2013, and thereafter in spring and/or in autumn every year for four years. In the Evo and Mysmalm 2 experiments, sampling was carried out once, six years after the establishment of the LR piles, in June 2014.

In each plot, soil samples were systematically taken from the organic layer (Of and Oh) and from the top mineral soil (uppermost 5 cm). The litter layer was carefully separated from the organic layer in the field. Five replicates of soil cores were taken per plot and combined to form one composite humus sample and one composite mineral soil sample.

Soil samples (I, III) were sieved, and dry and organic matter content, pH, and total C and N were measured. The dry matter content was determined by drying the samples overnight and weighing them, after which soil organic matter content was determined as loss on ignition. Soil pH was measured in a soil–water suspension. Total C and N were measured from air-dried samples using an automated CHN analyzer.

Net N mineralization and net nitrification (I, III) were measured in aerobic laboratory incubation experiments. Before and after incubation, NH_4-N and NO_2+NO_3-N were extracted with KCl, and the filtrates analyzed with a flow injection analyzer. In order to determine net N mineralization, the difference in mineral N (the sum of NH_4-N and (NO_2 + NO_3)-N) in the final (post-incubated) samples and the corresponding initial values were calculated. To determine the net nitrification, values of (NO_2 + NO_3)-N were handled in a similar way.

In Mysmalm 1 experiment (article I), the same incubation bottles were used for aerobic C mineralization measurements. The evolution of carbon dioxide (CO_2) was detected in the headspace of the bottles after about 24 hours using gas chromatography (articles I and III). In order to estimate aerobic C mineralization, detected soil respiration was calculated on an organic matter basis.

Microbial biomass C and N of the fresh humus samples were determined by using chloroform fumigation-extraction (FE) (articles I and III) under constant moisture (60 % WHC) and temperature (28 °C) conditions. C and N flushes from the microbial biomass were determined as the difference between the fumigated and unfumigated samples, and converted to microbial biomass C (C_{mic}) and microbial biomass N (N_{mic}) using the formulas of Martikainen and Palojärvi (1990).

3.3. Nitrogenase activity and quality of logging residues

Biological N fixation was determined as nitrogenase activity using the acetylene reduction assay (ARA). Reduction of acetylene (C_2H_2) to ethylene (C_2H_4) by nitrogenase mimics the N-fixing enzyme, which converts N_2 to ammonia (NH_3).

Biological N fixation activities were determined in humus layer samples (article III) and in LRs (article IV). Humus soil samples were taken as described above. LR samples were taken from the upper and lower parts of the piles. In the laboratory, samples were divided into two classes (branches and leaves/needles). The ethylene evolved was measured with a gas chromatograph. Nitrogenase activity was reported as the rate of C_2H_4 production. There was no endogenous ethylene production. Dry mass was determined from the same LR samples, and total C and N were determined with a CHN analyzer to calculate the C-to-N ratio (IV).

3.4. Determination of carbon and nitrogen losses

In the field, N_2O fluxes were measured several times, mostly during growing seasons. In Mysmalm, measurements were done in 2016–2019 (Experiment 1) and in 2016–2017 (Experiment 2) (article II), and in Latokartano measurements were done in 2016 (article III).

The N_2O samples were collected using the closed chamber method. Stainless steel chambers with a temperature recorder were inserted tightly to the soil and gas samples that were collected. In Experiment 1 (article II), piles were lifted with wooden sticks before the measurements (Figure 6), and in Experiment 2 (II) and in Latokartano (III) LR piles were enclosed inside the chambers. In Experiment 2, the volume of the chamber with LRs of all tree species was identified. The samples were analyzed with a gas chromatograph, and the N_2O flux rates were calculated from the linear change in the gas concentrations.

In the laboratory, the origin of N_2O production in the humus layer was determined using the acetylene (C_2H_2) inhibition method (articles II and III). This method is based on the addition of different quantities of C_2H_2, which regulates microbial activity in the soil. This allows for the determination of whether N_2O is produced via nitrification or denitrification. Lower quantities (0.003 %) of C_2H_2 prevent autotrophic NH_4-N oxidation during nitrification. Addition of higher quantities (2.5 %) of C_2H_2 prevents both autotrophic nitrification and reduction of N_2O to N_2 during denitrification.

Samples were incubated and different quantities of C_2H_2 were added to bottles. Gas sampling was done from the headspace of the bottle after 3–24 h, and N_2O analyzed with a gas chromatograph. All parameters were calculated on a soil organic matter basis.

Figure 6. The closed chamber technique used to collect N_2O samples from the soil under the LR piles (articles II and III).

To monitor C and N in soil percolation water, water sampling was carried out in Mysmalm 2 site in 2016–2017, i.e., two to three years after clear-cutting (article II). In Latokartano, monitoring began in the same year as clear-cutting, and continued for four years during 2012–2016 (article III). Water sampling was done at 4–6-week intervals during the snow-free period using zero-tension lysimeters. Soil-water samples were collected from a depth of 40 cm.

The soil solution pH was analyzed in the laboratory. NH_4-N and NO_3-N concentrations, DOC, and dissolved N (DN) concentrations were measured. To calculate DON, the mineral N (NH_4-N + NO_3-N) concentrations were subtracted from the concentrations of DN. Annual area-based leaching fluxes (kg ha^{-1} y^{-1}) were further detected based on field measurements of the volume of the water leaching and the concentrations of NH_4-N, NO_3-N and DON (mg L^{-1}) (article II).

The molecular size distribution of the DOC in the soil percolate water was also determined (article II), as described by Smolander and Kitunen (2002). Determination was performed for percolate waters, sampled after rain events in June 2016 and in October 2017. Samples were filtered through 0.45-µm membranes and molecular size fractionation was performed by ultrafiltration using Amicon's diaflow membranes with nominal weight cutoffs at 100, 10 and 1 kD.

3.5. Soil temperature measurements

Soil temperatures were recorded in articles I (in 2016) and II (in 2016 and 2017) with data loggers. The temperature sensors of the data loggers were placed 2 cm belowground, i.e., at the interface between the O-horizon and mineral soil layers.

3.6. Statistical analyses

Linear mixed models (SPSS Statistics 22 software) were used to analyze the effects of LR piles and tree species on soil properties and processes over the study period (article I). Differences in N losses affected by LR treatments were analyzed with parametric ANOVA, parametric repeated measures analysis of variance (ANOVA), or Friedman's nonparametric ANOVA followed by pairwise testing (articles II and IV). Bonferroni and Tukey's tests were used to determine pairwise differences between treatments for multiple comparisons.

A threshold *P*-value (α) of 0.05 and 0.1 were used to define significance. The normality assumptions of residuals were assessed using Kolmogorov- Smirnov and Shapiro-Wilk tests, and the homogeneity of variances using Levene's test. Ln-transformations were used when needed. When assumptions of normality or equality of variances were not met, the non-parametric Wilcoxon test, Kruskal-Wallis test, and Mann-Whitney U-test were used to identify differences for variables describing N transformations (articles I and III); soil temperatures, leaching fluxes, and N_2O fluxes and production (articles II and III); molecular size distribution of the soil percolation waters (article II), and nitrogenase activity rates of the LRs and total C, total N, and C-to-N ratio (article IV) among LR piles, control treatments, and tree species treatments. The relationships between the variables included in the study were assessed using Spearman's correlation coefficients. All statistics were performed with SPSS Statistics 22 and 25.

4 RESULTS AND DISCUSSION

The overall effects of LR retention and harvesting after clear-cutting on boreal forest N and C cycling is not fully understood. When this thesis was started, a few individual studies existed on the effects on LRs, but the majority had only included one tree species, and the comparison of piled LRs of different tree species was missing. Therefore, this thesis focuses on the effects of LR piles of birch, spruce and pine on soil N and C cycling dynamics and both gaseous and leaching losses on a clear-cut as well as possible gains of N via biological N fixation.

4.1. Effects of logging residues on soil properties and soil C and N transformations

To describe the effects of LRs on the quality of organic matter, the properties of organic matter were studied with sieved material from the humus layer, and the results are expressed on the basis of organic matter. Differences between treatments were more intense in the organic layer than in the uppermost mineral soil layer (articles I, III). In general, forest management practices affect the mineral soil layers less (Piirainen et al. 2015), thus their influence is weakened in comparison with the organic soil layers (Adamczyk et al. 2016).

In the organic layer, LRs started to increase OM concentration (%) from the following growing season, significantly in Latokartano (article III) two years after placing the LR piles, and was highest under LRs (40 kg m^{-2}) six years after their establishment at the Evo and Mysmalm II sites. Also, DOC concentrations increased significantly under residues, most notably in spruce plots.

The soil pH was already significantly higher from the following growing season after establishing the LR piles (articles I and III). These observations are comparable to Adamczyk et al. (2016), who found a similar effect of spruce LR piles five years after clear-cutting. Birch is known to decrease soil acidity, particularly when compared to spruce (Smolander and Kitunen 2011). In the second growing season after clear-cutting, soil pH was significantly higher under birch and spruce residues compared to that under pine residues, but prior to that the pH was similar under all piles (article I).

For the uppermost 5 cm of mineral soil, LRs significantly increased soil pH in Latokartano already from the first growing season onwards (article III), whereas in Mysmalm significant increase occurred later on the second growing season (article I). In Latokartano, LRs significantly increased OM one to two years after the establishment of the piles (article III).

In agreement with the proposed H1, LR piles clearly stimulated soil N cycling processes during the study period irrespective of the sampling time, i.e., from the beginning to six years after establishment of the LR piles. In Mysmalm I, the rate of net N mineralization and net nitrification was measured in the second and third growing seasons, when these processes were stimulated, as expected. Net nitrification tended to trigger later (article I). LRs significantly increased the rate of net N mineralization in comparison to the control. Concentrations of NH_4-N and NO_3-N developed unevenly, and there were also sudden drops in concentrations. Overall, mineral N concentrations were higher under residues two to three years after the establishment of the LR piles. Accelerated net N mineralization and net nitrification under LR piles of Norway spruce is consistent with the results of a study performed five years after clear-cutting (Adamczyk et al. 2016). In addition, elevated concentrations of NH_4-N and NO_3-N were found under LRs in a coniferous plantation in Spain (Perez-Batallon et al. 2001).

There was a clear overall influence of LR piles, regardless of the tree species. Contrasting with H3, the tree species effect was minor, i.e., the differences between the effects of birch, spruce, and pine LR were small (article I). In growing forest stands, several studies have reported effects of different tree species on the soil microbial processes related to C and N cycling (Blagodatskaya and Anderson 1998; Menyailo et al. 2002; Priha and Smolander 1999; Smolander and Kitunen 2002; Vesterdal et al. 2008). However, those situations are not comparable to LRs, since in growing forests there are multiple other interactions influencing the soil than litter alone. Moreover, LRs contain much larger quantities and larger-sized woody material than litter. LRs also differ from litter in the quality of foliage, which is green and contains more nutrients than fallen litter.

Within the entire study period, the most remarkable effect was noticed for net nitrification. In Latokartano (article III), the first time a significant increase in nitrification was already seen in the same growing season following establishment of the piles, although this trend as not continuous. In contrast, in Mysmalm (article I) significant differences were not detected earlier than two years after clear-cutting, and also occurred less intensively. This could be due to the different time intervals that elapsed between clear-cutting and the establishing of the LR piles at these two sites. In Latokartano (article III), LR piles were created one year after clear-cutting, whereas in Mysmalm 1 (article I) it was done after only a few days. The Latokartano (article III) site had a lower C-to-N ratio and a higher soil pH.

In the uppermost 5 cm mineral soil layer, LR also tended to accelerate N transformations. In Latokartano (article III), NH_4-N and NO_3-N concentrations were already higher in the first autumn (2013), and then declined. In Mysmalm (article I), LR significantly increased net N mineralization. Also, nitrification was triggered, and the concentrations of NO_3-N consequently increased.

In Mysmalm (article I), C mineralization in the organic layer tended to accelerate under the LR piles. A similar response was noticed five years after clear-cutting (Adamczyk et al. 2016), whereas ten years after LR treatment, the stimulating effect was only noticed at one pine site (Smolander et al. 2015). In a drained peatland after clear-cutting, LR piles more clearly increased the release of CO_2 (Mäkiranta et al. 2021). C mineralization intensified in the first spring after clear-cutting. Later, in autumn 2016, spruce residues re-stimulated C mineralization, maybe due to the slower decomposition and release of easily degradable C substances of spruce needles than from the foliage of birch or pine (article I).

In all the current study sites (articles I, III), LRs tended to decrease the amount of N in the soil microbial biomass, and at Mysmalm (article I) the microbial biomass C was also negatively affected. Therefore, at Mysmalm, the existing soil microbial biomass was suggested to be more active under LRs. Negligible or opposite effects of LRs on soil microbial biomass were found in some earlier studies in thinning stands, or after longer periods of time had elapsed since final felling (Smolander et al. 2013, 2015).

In mineral soil layers, the effect of LRs on C mineralization were minor (articles I, III), which is consistent with the results of Adamczyk et al. (2016), who also found a less intensive effect of LR piles on the mineral soil layer 5 years after clear cutting. Thus, the limited effects of LR piles and tree species on C mineralization in the upper mineral soil layer, in contrast with the larger effects in the organic horizon, are not surprising.

4.2. Effects of logging residues on soil C and N losses

The concentrations of different forms of N and C in soil percolate water at a depth of 40 cm were monitored after clear-cutting and establishment of the LR piles (articles II and III). Overall, the LR piles acidified the soil percolation water, but did not affect the DOC concentrations significantly. However, in several samplings the mean concentrations of DOC were higher under the LRs, with the range still being similar to that of undisturbed forest soil (Lindroos et al. 2008; Mustajärvi et al. 2008). In addition to LRs, clear-cutting can increase the leaching of DOC from the organic horizon, but C is effectively retained in lower mineral soil horizons (Piirainen et al. 2002).

In comparing the effect of tree species (article II), the percolation water pH varied from 4.8 to 6.7. Interestingly, the lowest values were detected under the LR piles of birch two years after their establishment. The DOC concentrations (article II) differed among the LR of the different tree species. The concentrations tended to increase under the birch LRs one year after creating the LR piles, while under the spruce LRs this happened later, after two years.

In agreement with the H2, LR piles increased N leaching. Especially in Mysmalm (article II), most of the N in soil percolation waters was dominated by NO_3-N, whereas the proportions of the other N forms were clearly smaller. The concentrations of all N forms in the control plots remained relatively low during the monitoring period. Net nitrification can be assumed to be negligible before the clear-cutting, because of the low concentration of NO_3-N in percolation waters almost two years after clear-cutting in summer 2016. Instead, both the NH_4-N and NO_3-N concentrations clearly already peaked one year after establishment of the piles at the Latokartano site (article III). Thus, NO_3-N concentrations increased one to two years after establishment of the piles, depending on the site. The highest concentrations of NH_4-N were detected during the summer months. When comparing the effect of tree species (article II), the highest concentrations of NH_4-N and NO_3-N were detected under the LRs of birch, two years after pile establishment.

These results are in accordance with our hypothesis and with previous studies, in which elevated concentrations of N in the soil percolation waters under the LR piles were found soon after clear-cutting (Clarke et al. 2018; Hedwall et al. 2013; Lindroos et al. 2016; Ring et al. 2015; Rosen and Lundmark-Thelin 1987; Wall 2008). In these studies, only the LRs of coniferous species were used. Instead, no statistically significant differences in the concentrations of NH_4-N and NO_3-N in centrifuged soil solution were detected 7 years after the clear-cut when plots with and without LRs were compared (Lindroos et al. 2018).

The N uptake of the developing ground vegetation probably contributed to the later-occurring decreases in mineral N concentrations in Latokartano (article III). Over time, the N uptake by ground vegetation and tree seedlings can bring the concentrations down, and decrease the risk of N losses, although N cycling processes may still be stimulated. Accordingly, in a 4-year study in Sweden, the cumulative N leaching was inversely correlated with plant N uptake (Bergholm et al. 2015). To avoid triggered N cycling and N losses, it is of utmost importance to get an evenly distributed and well-growing seedling stand on the site as soon as possible.

Interestingly, the DON concentrations tended to decrease under the LR piles in Mysmalm (article II) between years 2016 and 2017, i.e., approximately two to three years after clear cutting, and increase in Latokartano (article III) from 2014 onwards, i.e., two to four years after clear-cutting. The magnitude was still similar to that of undisturbed forest soil (Lindroos et al. 2008; Mustajärvi et al. 2008). DON in the percolate waters may be mobilized partly from LR piles (Rósen and Lundmark-Thelin 1987), probably from degrading foliage and partly from decomposition of SOM (soil organic matter) as a microbial waste product (Guggenberger et al. 1994), or stimulated depolymerisation of soil organic N.

In the present study, DON concentrations (article II) were very low, and they were less influenced by LRs than NH_4-N and NO_3-N, although they already peaked earlier (in March 2016) under the pine LRs. For instance, Casals et al. (1995) observed that most of the organic N in the leachate in pine forests originated from the freshly fallen litter and the partially decomposed litter layers of the forest floor (O horizon). Results of our studies probably point to the degradation of N in the soil going all the way to mineral N forms, but it is also possible that the dissolved organic N was retained in the 40 cm soil layer above the lysimeters. In addition, the degradation of DON in lysimeter collectors between water samplings cannot be excluded.

The percentage distribution of DOC into different fractions based on molecular size was relatively similar in all treatments (article II). The smallest size fraction (< 1 kD) accounted for 34–64 % of the DOC. The corresponding molecular size distribution of DON was not performed here. On a clear-cut, small molecular-sized DON was shown to be formed during soil incubation, together with high amounts of NO_3-N (Smolander et al. 2001). In comparison, a molecular size fraction of 10–100 kDa was always found to be the most abundant in spruce, pine, and birch stands (Smolander and Kitunen 2011).

In Mysmalm (article II), total annual leaching fluxes (mg m^{-2}) were calculated for each individual treatment. From one to two years after piling the LRs, fluxes distinctly increased. Then, both N-cycling processes were assumed to be stimulated, and larger amounts of water leached through the lysimeters. In 2017, i.e., two years after piling LRs, leaching fluxes of NH_4-N and NO_3-N were greater under LR piles compared to control treatments. Leaching fluxes of DON and DOC were rather similar among all tree species.

In 2016, i.e., one year after the establishment of LR piles, DOC fluxes were higher under pine and birch LRs compared to spruce (article II). In 2017, leaching fluxes of all fractions (NH_4-N, NO_3-N, DON, and DOC) clearly increased. Then, the highest fluxes of NH_4-N and NO_3-N were found under birch LRs. NO_3-N (6800 mg m^{-2}) clearly dominated the N leaching fluxes (7200 mg m^{-2}), in comparison with the lower fluxes of NH_4-N (715 mg m^{-2}). The DON fluxes were much lower (< 100 mg m^{-2}). The DOC fluxes were rather similar (approximately 2000 mg m^{-2}) under all the tree species treatments.

Our total annual leaching fluxes (mg m^{-2}), especially under birch residues, were higher than those in a study of leaching fluxes of Norway spruce by Lindroos et al. (2016). That study assumed LR piles of 40 kg m^{-2} to cover approximately 10 % of the clear-cut area and LR piles consisting of 10 kg m^{-2} and 0 kg m^{-2} to cover 30 % and 60 %, respectively (according to Rosen and Lundmark-Thelin 1987).

N_2O fluxes varied between the sites and growing seasons, and among years. However, LR piles increased N_2O fluxes in both Mysmalm (article II) and Latokartano (article III) compared to control plots without LR, as proposed in H1 and H2.

The coniferous LRs contributed most to N_2O emissions, and those emissions occurred most intensively in late summers or early autumns during the third growing season in Mysmalm (article II). Interestingly, spruce residues showed continually increasing N_2O emissions in every sampling occasion in 2018 and 2019, i.e., four to five years after creating the LR piles (article II). Similarly, N_2O emissions were still significantly increased in the Latokartano site (article III) in autumn of the fourth growing season following establishment of the piles. A similar phenomenon, where N_2O peaked among otherwise low N_2O fluxes after harvesting, was described by Tate et al. (2006), and occurred under LRs in fertile soil in New Zealand. The N_2O fluxes detected in this thesis are of a similar magnitude recorded in other studies of Finnish forest mineral soils (Pihlatie et al. 2007; Schindlbacher et al. 2004). Thus, these fluxes can be considered low when compared to levels occurring on managed peatland forests, or cultivated, afforested, drained, or abandoned fields (Liimatainen et al. 2014, 2018).

In general, the low concentrations of N limit nitrification and denitrification rates in boreal upland forests. These processes produce N_2O fluxes, thus the order of magnitude and inconsistency in N_2O data is not surprising (Pihlatie 2007; Pilegaard et al. 2006; Schiller and Hastie, 1996; Tupek et al. 2015).

In this study, N_2O was produced via both nitrification and denitrification (articles II and III), depending on the tree species and site properties. In Mysmalm, N_2O consumption was most often noticed in study plots without LR piles, and when N_2O was produced in these plots, it was mainly via denitrification (article II). Under birch residues, denitrification contributed most to the N_2O fluxes, and denitrification seemed to stop N_2O. Under coniferous residues, N_2O continued to harmless N_2 gas to a larger extent. N_2O originated mainly (89 %) through autotrophic nitrification in pine LR plots, and under spruce residues by both processes (article II). In contrast, at the Latokartano site (article III), under spruce residues on till soils, N_2O was mainly produced (70 %) through denitrification and stopped to N_2O. Grain size analysis was not performed, but both study sites are porous, and water can penetrate the soil relatively easily.

LR piles affect the soil physical properties and soil conditions (Rósen and Lundmark-Thelin 1987). The mean soil temperatures (°C) in the boundary between the organic and mineral soil layers were monitored in Mysmalm Experiment 1 (articles I and II) and Experiment 2 (article II). The daily temperature variation was larger in the control plots without LRs than under LR piles. Hence, LR piles provided insulation and stabilized variation in soil temperatures, and effectively provided cooler conditions. This is consistent with the studies concerning LRs in mineral and organic soils (Harrington et al. 2013; Ojanen et al. 2017; Proe et al. 2001; Ring et al. 2015; Trottier-Picard et al. 2014). No differences among tree species were found when soil temperature changes were compared. It can be concluded that factors other than soil temperature, such as chemical-biological effects, were influencing the soil processes (Ojanen et al. 2017) and tree species differences of N and C cycling under the piles more strongly.

4.3. Nitrogenase activity and quality of logging residues

Contrary to H4, no consistent nitrogenase activity was detected in the organic layer under LR piles (article III). Non-symbiotic biological N fixation consumes lots of energy (Bernhard 2010), and therefore the process may be reduced or inhibited if N is easily gained elsewhere. Here, stimulated N transformations in the soil provide easier access to inorganic N, and increased availability of inorganic N may have diminished biological N fixation.

Instead, as proposed in H4, nitrogenase activity was detected in LRs (article IV) of all tree species. These results are consistent with other studies related to coarse woody debris, where nitrogenase activity has been found to vary on average from 0.4 to 19.5 nmol C_2H_4 d^{-1} (g dry mass, equal to 0.02-0.8 nmol C_2H_4 g h^{-1}) (Brunner and Kimmins 2003; Granhall and Lindberg 1980; Jurgensen et al. 1987, 1992).

N fixation may decrease in the LR itself, if it is in close contact with soil enriched with inorganic N, as found in the organic layer (article III). That, and increased amounts of oxygen, inhibit biological N fixation rates (Bernhard 2010) in woody material (Jurgensen et al. 1984), and in mineral soil (Burgoyne and DeLuca 2009). In contrast, increased moisture, temperature, external C sources, and pH generally increase non-symbiotic N fixation (Gundale et al., 2009, 2012; Rinne et al., 2017; Son, 2001; Weber and Sundman, 1986).

In addition to the previously mentioned variables, differences in degradation rates should also be noticed, especially when comparing N fixation between fine and coarse parts of LRs. For instance, in Norway spruce needles, the mass loss after one year increased from 25 % to 60 % in five years, but for main branches the corresponding mass loss only increased from 10 % to 20 % (Smolander et al. 2008). Accordingly, we can expect that in the current study branches were in an earlier stage of decomposition. However, N fixation may also increase during decomposition (Jurgenssen et al. 1984; Ronkoski 1980).

C-to-N ratios of needles/leaves is lower in comparison with woody branches (article IV); thus, the fine parts have more N compared to branches, and decompose faster than thicker branches. Granhall and Lindberg (1978) suggested higher N fixation when C-to-N ratios were higher in woody litter of coniferous trees, and Leppänen et al. (2013) proposed a similar pattern in boreal forest mosses. In the current study, N fixation increased in LRs of spruce (needles and branches), with an increasing C-to-N ratio.

In addition, two years after clear-cutting and the establishment of the LR piles, biological N fixation tended to be more intense in conifer LRs than in birch LRs (article IV), as proposed in H5. Conifer litter contains more recalcitrant constituents compared to broadleaf litter (Berg 2000). However, birch residues contained more N compared to conifers. Leaf litter decomposition is faster than coniferous litter during the first year, with spruce litter decomposing slower than pine (Palviainen et al. 2004). Later, the differences between these litter materials diminishes (Prescott et al. 2000, 2004).

LRs of Scots pine, Norway spruce, and silver birch are, by means of biological N fixation, a source of additional N to the forest ecosystem, although the amounts of fixed N are small. Therefore, possible N losses may partly be compensated for through biological N fixation occurring in LRs. The mentioned N losses may be caused by harvesting, clear-cutting, LR retention, or combinations thereof.

4.4. Relationship between logging residues and soil carbon and nitrogen cycling

Some general observations could be made concerning the relationships between quality of LRs (article IV), soil N and C transformations (articles I and III), soil N and C losses (articles II and III), and N gains (article IV).

LRs consist of fresh foliage and branches and, especially in the early stage of decomposition, release a sudden of pulse of leachates and volatiles to the soil. These compounds can either stimulate or inhibit soil microbial processes (Adamczyk et al. 2015, 2016; Smolander and Kitunen 2008). For instance, in the first stages of decomposition of Norway spruce needles, NH_4-N and easily degradable DOM, including N-containing compounds, are leached (Kiikkilä et al. 2012; Wall 2008). Leaching of these substances and the subsequently stimulated net N mineralization in the organic layer may intensify net nitrification. Fine parts of LR material (needles and leaves) are probably most responsible for releasing compounds to the soil soon after the establishment of LR piles.

In the organic and mineral soil layers, positive and significant correlations between variables describing N transformation processes (net N mineralization, net nitrification) and NH_4-N and NO_3-N concentrations is not surprising. In Mysmalm (articles I and II), a rational pattern was detected: net N mineralization increased simultaneously with the concentrations of NH_4-N, until nitrification was strongly intensified and the NH_4-N concentrations went down. The concentrations of NO_3-N remained high in LR plots, since the vegetation had not yet recovered. Uptake of N by recovering vegetation and tree seedlings decreased both NH_4-N and NO_3-N levels later (Piirainen et al. 1998; Titus et al. 1997).

In addition, positive and significant correlations were found in comparisons of net N mineralization and net nitrification to pH and DOC concentrations. An increase in soil pH is reported to stimulate soil microbial activity and nitrification (Paavolainen and Smolander 1998). Similar positive correlations between soil DOC concentrations and net N mineralization as an indicator of microbial activity in the organic layer has been detected before (Michel and Matzner 1999; Smolander and Kitunen 2011). On the other hand, negative correlations between soil DOC concentrations and net N mineralization have also been detected (Zeller et al. 2000). N transformations, specifically net N mineralization, tended to intensify more by LR compared to C mineralization.

Tree species differ with regard to the chemical composition of litter leachates (Johansson 1995; Kiikkilä et al. 2012), to which microbial processes and soil properties may respond quickly. In a two-stage laboratory decomposition experiment by Kiikkilä et al. (2012), the total flux of easily degradable DOM was 40 % higher from birch leaf litter than from spruce needle litter. However, the differences in decomposition of LRs and litter should be considered when comparing these. Leaves and needles in LRs decompose faster, since LRs contain more nutrients compared to litter. This is due to nutrient translocation from the litter to growing tissues, before it is shed from the trees (Helmisaari 1992; Lundmark-Thelin and Johansson 1997).

When net mineralization and net nitrification are increased, and the uptake of N by vegetation is simultaneously low, the risk for N losses due to leaching or gaseous fluxes increases. The risk for N losses, particularly as NO_3-N, can also result from a continuous decrease in C-to-N ratios (Gundersen et al. 1998), a change that did not emerge clearly in the current study. At our study sites, leaching of N intensified one to two years after establishment of the piles, pointing to an excess of N in the soil for heterotrophic microbes. Simultaneously, N_2O fluxes increased, thus also being less limited by mineral N. N-transformation processes provide mineral N substrates to soil, further stimulating N losses as N_2O fluxes (Chapuis-Lardy et al. 2007; Ullah et al. 2008) if N availability is the main restricting factor for N_2O production and not the availability of C. Significant correlations among the variables describing the organic layer N transformations (net N mineralization, net nitrification, NH_4-N and NO_3-N concentrations) and N_2O fluxes indicate similar phenomena.

As indicated by Smolander and Kitunen (2011), the positive correlation of C mineralization with soil pH and DOC concentrations is not surprising. C mineralization was positively correlated with soil pH in the mineral soil layer as well. In the current study, C and N in the microbial biomass were negatively correlated with the concentration of NO_3-N.

During the period of the study in 2016 and 2017 (Mysmalm I, Experiment 2, article II), the pH of the soil percolation water and the DON were significantly and negatively correlated with NO_3-N. In contrast, the soil N_2O fluxes were significantly and positively correlated with NO_3-N and NH_4-N levels of the soil percolation waters.

4.5. Significance of logging residues to soil carbon and nitrogen

In our study, larger LR piles (article III) increased the net N mineralization, net nitrification and leaching of N more as compared to smaller piles. DOM dynamics in soils are dependent on the amount of fresh OM input and closely related to N cycling. According to Hongve (1999), soil water DOC is mostly derived from litter, so an increase in soil percolate water DOC under LR piles of 40 kg m^{-2} (II) is expected, irrespective of whether it originates directly from litter leachates or is a microbial product. Gundersen et al. (1998) demonstrated that an increase in the quantity of litter caused an increased flux of DOC into the soil. Tipping et al. (1999) observed in a field manipulation experiment on three different soils that the soil with the highest organic matter content exported more DOC than the other two soils.

In practical harvesting executed with modern harvesting techniques, LRs are spread unevenly. As a result, part of the forest floor has differently sized LR piles, with part of the area being without LRs. Then, LR piles act as “hotspots” on the forest floor, stimulating the N-cycling processes underneath. With the results gained in this study, we can build estimations of N losses and gains per hectare, although these values are based on simplified situations. These calculations include various uncertainties, due to the low number of annual sampling occasions, the high variability of the soil affecting the results, and the site-specific nature of the soils.

If we assume that after harvesting, 40 kg piles of LRs are created, covering 10 % of the forest floor, and the remaining surface area is uncovered, we will end up with 5.8, 4.5, and 10.5 kg total N ha^{-1} annual leaching for Scots pine, Norway spruce and silver birch, respectively. These results are based on the leaching fluxes from 2017 in Mysmalm I Experiment II (article II), i.e., the second growing season after placing the piles on the study site. After vegetation has developed, the leaching losses of N will very probably decrease considerably.

When a similar ha-based calculation is performed for the N_2O fluxes, we will end up with considerably lower values, and the corresponding values for the total annual N_2O fluxes being 0.003, 0.002, and 0.002 kg ha^{-1} year^{-1} for Scots pine, Norway spruce, and silver birch, respectively, with the average annual losses for all treatments being −0.001 kg N_2O ha^{-1}. Thus, according to the results of this thesis, N losses occur mostly through the leaching of soil percolation waters and not as N_2O emissions.

In general, total N outputs from the boreal Scots pine forest are estimated to be much lower (0.3 kg N ha^{-1} yr^{-1}) when compared to both the storages and inputs (Portillo-Estrada et al. 2013). Also, the average annual N deposition in southern Finland is estimated to be low, being 2.9 and 2.8 kg N ha^{-1} for Scots pine and Norway spruce, respectively (Merilä et al. 2014). In contrast, the average export of N over a typical rotation period in whole-tree harvesting with 70 % removal of LRs is much greater; 158 and 306 kg N ha^{-1} for Scots pine and Norway spruce, respectively (Merilä et al. 2014).

For N gains via biological N fixation we used the conversion factor 4 and assumed the same activity for 4 months annually. Then the magnitude of biological N fixation is 0.17–0.52 kg ha^{-1} year^{-1}. Here we estimated N fixation rates for branches only, since the portion of branches in logging residues is larger, their degradation is slower, and N limitation is greater compared to needles or leaves. In general, biological N fixation has been reported to vary between 0.3–3.8 kg ha^{-1} year^{-1} in Fennoscandian boreal forests (Granhall and Lindberg 1980, Rosén and Lindberg 1980).

As a summary, LR piles act as N cycling “hotspots” and affect to the early-stage dynamics of N cycling processes (Figure 7). In the clear-cut sites LRs create a large sudden pulse of

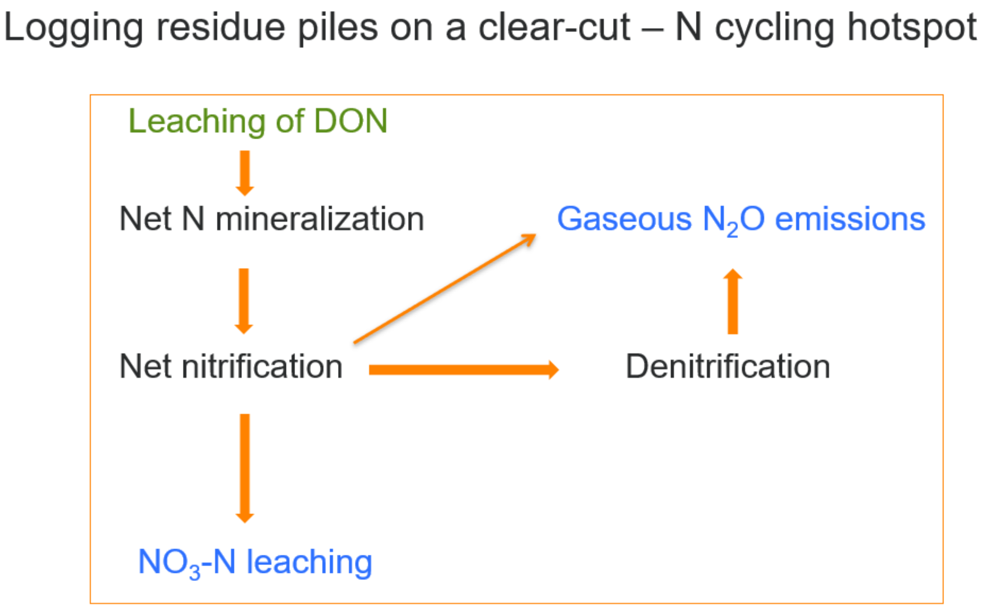

Figure 7. Logging residue piles act as N cycling "hotspots" on a clear-cut site, proposing a risk for N losses.

organic matter, including a significant source of DON (Wall 2008), that enters the soil surface underneath. This stimulates net N mineralization, net nitrification and denitrification and further can lead to N losses from the forest ecosystem via leaching of NO_3-N and the creation of gaseous N_2O.

4.6. Criticisms and application of findings

The effects of forest management practices must aim at achieving overall sustainability in all aspects—economical, ecological, social, and cultural—and their interaction with climatic sustainability. We understand that ensuring sustainability in harvesting practices can benefit forest owners' economics, as well as climate actions. We especially need more information on the ecological aspects.

The results of this thesis can be utilized in the sustainable LR harvesting guideline framework. This study clarifies the role of LRs affecting soil C and N cycling, processes that should be more deeply included in ongoing discussions in the field of forestry when aiming towards overall sustainability. Although C sequestration of forest soils is an important function, the ability of soils to retain N needs additional attention, especially in boreal areas where N limits the growth of forests.

Simultaneously, these thesis findings are applicable to the aims to achieve more climatically sustainable forestry. For example, Finnish forestry aims to increase the share of mixed forests and deciduous tree species. This thesis reveals differences between the LRs of different tree species in terms of their soil and environmental effects after final felling. However, mixed LR piles composed of all three tree species in the same pile were not included. Thus, future research should focus on studies of the effect of LRs of mixed tree species on C and N transformations.

This study supports earlier studies that conclude that LRs should be more evenly distributed on forest soil and big piles avoided, as more even distribution of LRs may diminish N leaching. Then, less N may originate from a LR pile itself when the pile acts as a N cycling hotspot, and from the organic layer of the soil underneath the piles. Also, the vegetation can develop through the piles, and N uptake may be enhanced. It should be noted that this conclusion requires comparison of several amounts of logging residues. In a thinning stand, some soil properties responded linearly to logging residue amounts but others not (Smolander et al. 2013). When comparing 10 and 40 kg m^{-2}, the highest concentrations of NO_3-N were associated with the highest amount of LRs left on the site (article III; Lindroos et al. 2016). Despite this pattern, more studies are needed to draw clear conclusions on this effect. The retention of LRs adds N to the system in the long-term. Particularly, Scots pine and Norway spruce retain N rather effectively, and form a steady N addition with a slow release rate (Palviainen et al. 2008). Thus, retention of LR itself secures nutrients for the growth of the next tree generation, and OM to better sustain soil productivity.

LR piles have a clear influence on soil processes during the first years after clear-cutting. Instead the tree species had relatively minor effects on some of the studied soil properties and processes, especially in the mineral soil layers, which might be due to the short time that had elapsed after clear-cutting. Therefore, it is possible that the study period was not long enough to reveal all the studied changes. Instead, some changes might only develop later. The development of the effects of LR retention and tree species over time remain unclear. Thus, long-term field measurements are needed to reveal the overall influencing factors, as rotation times in boreal forests are long. Also, the practical impact of forestry, for instance on soil structural damages, should be included when designing studies.

Additionally, the effect of LRs on consumption and production of GHG (CO_2, N_2O and CH_4) needs more long-term measurements. This study was focused only on N_2O. A deeper understanding of the variables, particularly influencing N_2O consumption and production, is crucial. To achieve this, N_2O fluxes should be monitored continuously or at shorter sampling intervals, since fluxes are highly variable. Fluxes may vary within a day, but also within seasons. Hence, in Finland, sufficient whole winter-lasting sampling should be included in N_2O studies. The strength of this thesis is that N_2O fluxes were monitored *in situ* in two different mineral soil study sites, with balanced replicates of treatments within a site, and the main responsible process for the production was determined.

It is possible that N_2O may have leaked during the measurements either after the chambers were installed, from the syringes, or during the measurements, although the field work was executed carefully. Based on the results of this thesis, it seems that fluxes of N_2O probably only increased in the short-term, but it should be kept in mind that if a forest ecosystem gains external N elsewhere, the influence may change during the long rotation periods of forests.

In current study, installation of lysimeters may have contributed to water infiltration, leaching fluxes, and decomposition rates of OM, but we can assume that it affected the results in a similar way for all treatments. The possible installation impact was most probably quite small, because the measured DOC concentrations were very comparable to those determined earlier for Finnish forest soils using funnel-type lysimeters (Lindroos et al. 2008). It should be noted that the flux calculations using the volume of soil water solution in the zero-tension lysimeter are only assumptions and very uncertain. This is because the zero-tension lysimeter itself may introduces an artefact into the soil which then may divert the flow paths of water through the soil.

The potential uptake of N by ground vegetation probably differs among treatments, as is the case for LR harvesting or retaining LRs in normal forestry practices. In this study, the effect of ground vegetation was not investigated. Hardly any vegetation was developed on the LR plots during the study period in Mysmalm 1 site (articles I and II) or in Latokartano (article III), while Evo, the Mysmalm 2 site (article III), and the control plots did show vegetation cover. Accordingly, the presence of the LR piles can have negative effects for a few years on both ground vegetation plant species numbers and their cover (Økland et al. 2016) but the length of the effect remains unclear. In addition to the LR piles, the decomposition of dying ground vegetation contributed to the LR plots.

Soil moisture was not determined in this study, although it is recognized as a major variable affecting N transformations (reviewed by Sahrawat 2008). LR piles intercept precipitation (Wall 2008), and prevent the soil under the piles from getting wet during minor rain events. However, once the soil under the LR piles is moist, the piles prevent evaporation and thus the loss of soil moisture, as the soil surface is simultaneously kept cooler.

Future studies should provide more detailed and long-term information on LR retention and harvesting, when variability of the soil is taken into account, i.e., studies performed across different site types (different soil types from mineral to organic soils, contrasting soil textures, etc.). Several different amounts of LRs are needed to better understand the effects of LR retention. In any case, this study offers knowledge of common site types and mineral soil types in southern Finland, such as Myrtillus-type forests, podzolized soils, and Podzols developed on till and sorted sandy soils, which are well represented in Finland.

5 CONCLUSIONS

The results presented in this thesis advance our current understanding of the early-stage effects of LR harvesting and retention of three different tree species after clear-cutting. The focus of the study was on the C and N cycling processes in boreal forest sites and soils. To only study the effects of tree species, but exclude the effect of the size of LR piles, the same amount of LRs consisting of different tree species were intentionally used. To study the effect of differently sized LR piles, the same tree species were used with different amount of LRs.

LR piles stabilized the variation and lowered the soil temperatures. The results showed that the effect of LRs was more prominent in the organic layer than in the mineral soil layer. Soil pH was clearly increased under LR piles, and simultaneously, C and N cycling processes were strongly stimulated as was hypothesized. N transformations, specifically net N mineralization, already intensified one to two years after the clear-cut, and to a greater degree than C mineralization. Measurements proved that net nitrification was triggered under LRs, as was already suggested in previous studies without actual measurements of the process.

According to hypothesis, as a result of accelerated nitrification and denitrification under LR piles, N_2O fluxes were increased, although fluxes remained low and highly variable. In contrast, N losses dominated by NO_3-N via soil percolation waters were rapidly and clearly elevated already one year after LRs were piled. The positive correlation between the concentration of water-extractable DOC and mineralization of C and N in the organic layer indicates close interactions between microbial processes and DOM. The most intensive effects occurred under the largest amount of LRs.

Contrasting with hypothesis, the effect of tree species was mild when compared to the overall effect of LRs. However, the response of some microbial activities differed between tree species. It is suggested that tree species-related differences in soil processes under LRs were driven by chemical factors and differences in decomposition rates of LR of the tree species, and not by LR's effects on physical conditions. C mineralization was stimulated and concentrations of DOC in both the organic layer and soil percolation waters increased most under LRs of spruce. N transformations (net N mineralization and net nitrification) triggered later and N_2O fluxes peaked later under LR piles of spruce. In contrast, N losses detected as increased concentrations of mineral N forms (NH_4-N, NO_3-N) in soil percolation waters accelerated most under the LRs of birch, as did increased soil pH of the mineral soil layers. An additional disadvantage of increased N concentrations in soil percolation waters for birch LRs was that denitrification stopped at N_2O, while under coniferous LRs denitrification continued to harmless N_2.

Additionally, no consistent nitrogenase activity was detected in the organic layer under LR piles, as hypothesized. There was a trend that two years after clear-cutting and establishment of the piles, biological N fixation intensified more in coniferous LRs than in birch LRs, although birch residues contained more N compared to conifers. According to the hypothesis, nitrogenase activity tended to be higher in branches than in needles or leaves. Although biological N fixation rates remained low in the LRs of all three tree species, it is suggested that LRs may provide considerable N inputs through biological N fixation through the whole rotation period. In addition, LRs themselves are a source of N.

In conclusion, LR retention in piles can accelerate C and N transformations in boreal forest soils. Stimulation under LR piles is seen already one year after clear-cutting, which should be considered when optimizing the harvesting time of LRs. Due to the lack of knowledge concerning N_2O fluxes and the proved effectivity of N_2O as GHG, monitoring N_2O dynamics under LR in different site types, regions, and changing climate conditions will be crucial to gaining a comprehensive understanding of methods to manipulate N_2O uptake and emissions. The duration of the effects on soils remains unclear. This needs further investigation, as does the effect of LRs from mixed tree species.

When LRs are left on the forest floor, it is possible to mitigate the negative effects of retention by spreading the LRs more evenly on the forest floor instead of piling them. However, in harvesting operations LRs are spread on the forest floor, and are used to improve the soil-bearing capacity and prevent the creation of deep wheel tracks by off-road traffic. These are important actions to minimize soil compaction and its possible consequences. Also, the evenly distributed regeneration of new tree seedlings is important, since growing vegetation retains N.

The information produced in this thesis is valuable for developing future LR harvesting guideline frameworks that need to consider all aspects of overall sustainability, including soil effects. It should be noted that although C sequestration of forest soils is an important function, the ability of soils to retain N needs additional attention in practical forestry, especially in boreal areas where N limits the growth of forests. Piling of LRs should be avoided when possible and LRs should be more evenly distributed on the forest floor on sites with good bearing-capacity of soil to avoid harmful environmental effects and to maintain soil nutrient balance. This should be clearly indicated also when educating forest biomass harvesting operators.

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