Agronomy Research **15**(3), 767–787, 2017

Impact of clearfelling on dissolved nitrogen content in soil-, ground-, and surface waters: initial results from a study in Latvia

Z. Libiete[1,2,*], A. Bardule[1], S. Murniece[1] and A. Lupikis[1]

[1]Latvian State Forest Research Institute 'Silava', Rigas str. 111, LV-2169 Salaspils, Latvia
[2]MNKC, Dzerbenes Str. 27, LV-1006 Riga, Latvia
[*]Correspondence: zane.libiete@silava.lv

Abstract. Conventional forest management has traditionally been targeted to enhance provisioning ecosystem services. Recently, however, awareness about the effect of forest management on other groups of ecosystem goods and services has been raised at the European and global levels. A number of initiatives addressing the evaluation and mitigation of the impact of forest management operations on biodiversity, soil quality, nutrient cycling, and water quality have been reported. In 2011, the development of a monitoring system to assess the impact of forest management on biodiversity and environment in the state forests of Latvia was initiated in the Latvian State Forest Research Institute 'Silava'. A number of studies to obtain experimental data and to test potential monitoring methods were implemented during this project. Among other activities, three research objects related to the quantification of changes in nutrient cycling after clearcut with whole-tree harvesting and stem-only harvesting were established. Data on changes in nutrient concentrations in soil solution, ground water, and surface waters, and on nutrient input through precipitation, are presently available for one year before and two years after clearfelling. Significant increase of dissolved nitrogen concentration in soil solution, as well as differences between stem-only and whole-tree harvested plots emerged only in the second year after harvesting. No significant increase of the dissolved N in the streams was observed, compared to the reference period. Ground vegetation recovery, amount of slash, soil properties and processes in the buffer zone are among those factors influencing the N leaching most, and these will be investigated further.

Key words: forest management, water, dissolved nitrogen content.

INTRODUCTION

In recent years, there has been a growing scientific and political awareness of various goods and services provided by the world's ecosystems. Forests, among other ecosystem types, deliver a wide range of ecosystem goods and services, with timber, energy-wood, non-wood forest products, biodiversity, carbon sequestration, and clean water being among those mentioned most often (e.g., Krieger, 2001; Powell et al., 2002; Fisher et al., 2009; EUSTAFOR & Patterson, 2011; Wunder & Thorsen, 2014).

Each intervention in the ecosystem processes simultaneously alters several ecosystem functions and consequently affects the delivery of various ecosystem goods and services. Conventional forest management (FM) has traditionally been targeted to use and enhance provisioning services. Recently, however, awareness about the effect of FM on other groups of ecosystem goods and services has been raised on the European and global levels (Nasi et al., 2002; FAO, 2010; EUSTAFOR & Patterson, 2011; Miura et al., 2015). At the same time, the worlds' forests are confronted with growing pressure due to the need to reduce fossil fuel usage. Also in Latvia, wood biomass utilization for energy purposes is predicted to increase substantially in the future. According to the National Renewable Energy Action Plan, Latvia's overall objective is to increase the share of energy produced from renewable energy sources in gross final energy consumption from 32.6% in 2005 to 40% in 2020 (Latvian Ministry of Economics, 2010). There are several potential methods to meet the increasing demand for energy wood; one of those is increased utilization of forest biomass. Intensified biomass harvesting potentially includes removal of branches, treetops, and stumps during clearcutting, thinning, drainage system renovation, and other silvicultural activities.

Clearfelling, especially whole-tree harvesting, may have adverse effects on several ecosystem services. A number of studies suggest that forest harvesting may cause a decline of water quality both in groundwater and surface waters (e.g., Ahtiainen, 1992; Kubin, 1998; Ahtiainen & Huttunen, 1999; Gundersen et al., 2006; Laudon, 2009; Miettinen et al., 2012). Mechanical disturbance of the forest floor may increase the potential for nitrate and potassium leaching to ground- and surface waters, as well as that of other pollutants, for example, mercury (Olsson & Staaf, 1995; Porvari et al., 2003, Nieminen, 2004; Munthe & Hultberg, 2004; Laurén et al., 2005; Gundersen et al., 2006; Bishop et al., 2009). Leaching of nutrients from harvested sites can lead to particular problems of eutrophication and acidification, causing major ecosystem damage to streams, rivers and lakes (Carpenter et al., 1998; Ahtiainen & Huttunen, 1999; Gundersen et al., 2006; Kreutzweiser et al., 2008). Concern has also been expressed about long-term nutrient depletion and loss of soil fertility, especially in N-limited ecosystems (Bengtsson & Wikström, 1993; Rolff & Ågren, 1999; Merganičová et al., 2005; Thiffault et al., 2011). Forest harvesting, particularly the removal of logging residues and stumps, affects ground vegetation and fauna (e.g., Olsson & Staaf, 1995; Bengtsson et al., 1998; Gunnarson et al., 2004; Åstrom et al., 2005). Thus, forest utilization clearly leads to changes in ecosystem processes and subsequently may alter provision of practically all ES groups, as these are directly dependent on ecosystem processes (Maes et al., 2013). Results related to the impact of forest harvesting on future forest productivity and leaching of nutrients and pollutants to adjacent water ecosystems are, however, site- and scale-dependent and rather contradictory (Futter et al., 2010; Wall, 2012; De Wit, 2014).

Most available data on the effects of forest management on water quality originate from Finland and Sweden, but due to dissimilar soil and hydrological conditions, results and conclusions obtained there may not necessarily be applicable in Latvian conditions. Soils of Latvia, compared to other soil regions, provinces and states have a wide range of distinct specific traits, mainly determined by diverse parent material (Nikodemus et al., 2009). Latvia, together with Lithuania, Estonia, parts of Poland, Russia and Belarus, as well as large area of the Baltic Sea, including island of Gotland, is a part of Baltic artesian basin, multilayered and complex hydrogeological system. Intense confined aquifer discharge is an important factor influencing nutrient cycling in Latvian forests (Dzilna, 1970; Virbulis et al., 2013). 86% of forests on wet and drained peat soils and 60% of forests on wet and drained mineral soils in Latvia are located in confined aquifer discharge areas, this situation being essentially different from that in Fennoscandia (Indriksons & Zalitis, 2000; Zālītis, 2006; Indriksons, 2010; Zālītis, 2012). The continuous nutrient supply by confined aquifer discharge waters explains high and stable long-term productivity of forest stands established on drained organic soils. Evidence exists that if horizontal water flow and soil aeration is maintained at these drained areas, forest productivity is enhanced also at adjacent nutrient-poor dry mineral sites.

In 2011, the development of a monitoring system to assess the impact of forest management on biodiversity and the environment in the state forests of Latvia was initiated in the Latvian State Forest Research Institute ‘Silava’. A number of studies to obtain experimental data relevant for Latvian conditions and to test potential monitoring methods were implemented during this project. Among other activities, research objects for the quantification of changes in nutrient cycling after clearcut with whole-tree harvesting and stem-only harvesting were established in the Kalsnava forest district research forests. Aim of this study was to test whether different types of clearfelling have significant and different impact on soil and water chemistry, forest regeneration and development of ground vegetation. This particular paper summarizes the first results on nitrate, ammonium, organic and total nitrogen concentration changes in soil solution, ground water, and surface waters, and nitrate, ammonium, organic and total nitrogen input through precipitation one year before and two years after clearfelling in pine and spruce forests in Latvia. We hypothesized that:

- clearfelling will result in significant increase of dissolved N-compounds in soil-, ground and streamwater;
- the concentration of dissolved N-compounds in the soil water will differ significantly between plots with stem-only and whole-tree biomass removal.

MATERIALS AND METHODS

The study area is located in experimental forests of Kalsnava Forest district, eastern part of Latvia (56°40–44′N, 25°50–54′E) (Fig. 1). Climate is continental; according to meteorological data from Jaunkalsnava meteorological station 10 km distant, the annual precipitation amount was 1,023 mm in 2012, 590 mm in 2013 and 823 mm in 2014, with largest share (61–74%) falling as rain from April to October. Mean annual air temperature was 4.4 °C in 2012, 5.1 °C in 2013 and 5.0 °C in 2014.

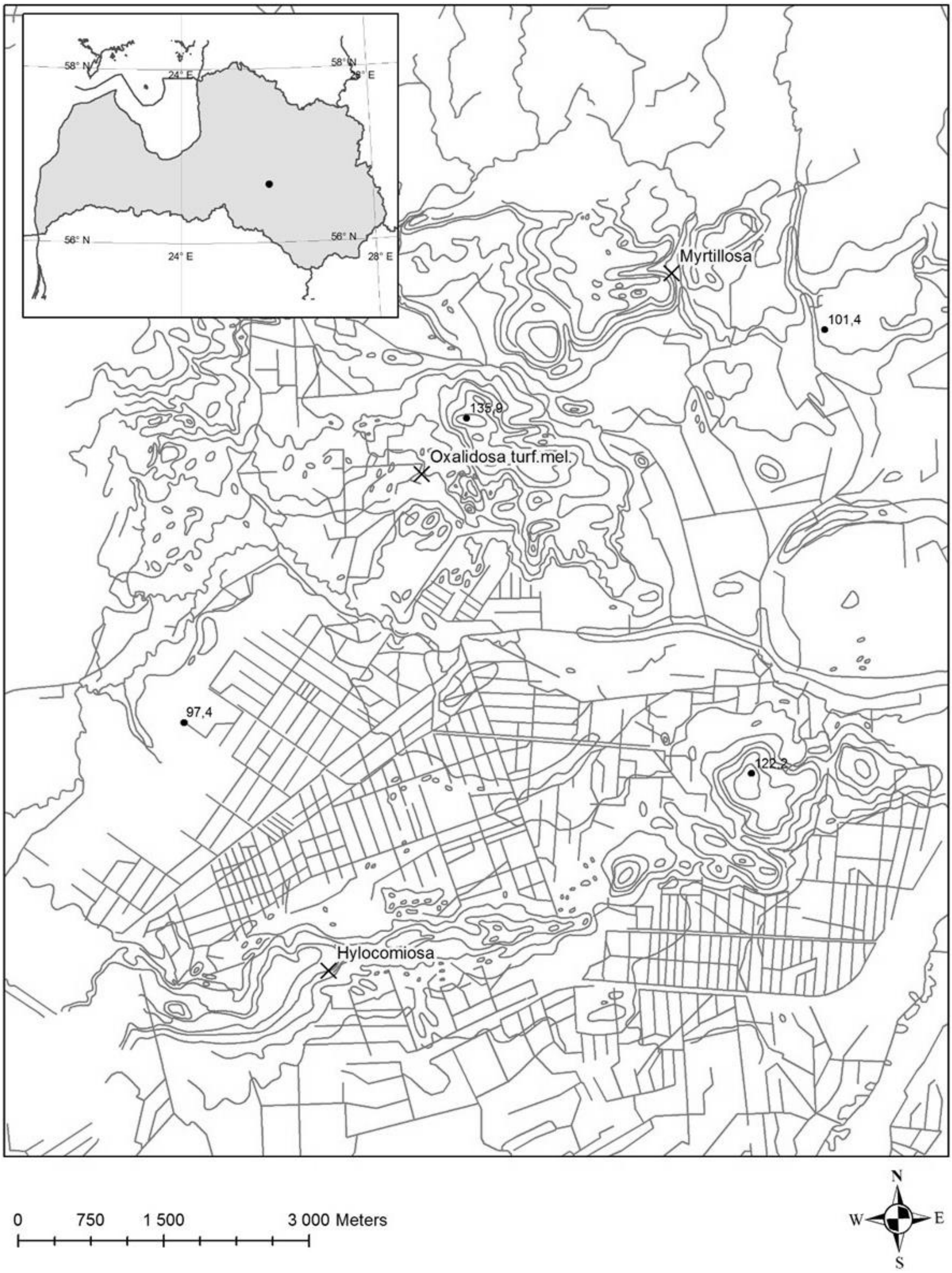

Figure 1. Location of the study sites.

Research was carried out at three sites: two were located on mineral soils (*Myrtillosa* and *Hylocomiosa* site type, dominant tree species *Pinus sylvestris* L.) and one on drained peat soil (*Oxalidosa turf. mel.* site type, dominant tree species *Picea abies* (L.) Karst.). Drainage was carried out in 1960. The sites were located on slopes (5° in *Oxalidosa turf.mel.*, 15° in *Hylocomiosa* and *Myrtillosa*), with bufferzone in the lower part (Fig. 1). Site description is presented in Table 1 and Table 2.

Table 1. Description of the study sites

Site	Dominant tree species	Mean diameter, cm	Mean height, m	Basal area, m^2 ha^{-1}	Standing volume before felling, m^3 ha^{-1}
Hylocomiosa	*Pinus sylvestris* L.	34	31	35.3	541.3
Oxalidosa turf.mel.	*Picea abies* L. (Karst.)	31	25	17.4	315.0
Myrtillosa	*Pinus sylvestris* L.	31	26	21.2	270.9

Table 2. Soil description of the study sites

Site	Soil type (WRB)	Soil texture (FAO)	Depth of O horizon, cm	Depth of H horizon, cm	Total C content, $g\ kg^{-1}$ (O horizon/0–40 cm/40–80 cm)	Total N content, $g\ kg^{-1}$ (O horizon/0–40 cm/40–80 cm)
Hylocomiosa	*Folic Umbrisols* (*Albic, Hyperdystric, Arenic*)	Sand	0–10	n.a.	545.4/7.8/3.9	15.5/0.2/0.2
Oxalidosa turf.mel.	*Rheic Histosols* (*Eutric, Drainic*)	Sand	0–3	3–95	555.4/104.6/46.1	22.1/5.6/2.4
Myrtillosa	*Albic Arenosols* (*Dystric*)	Sandy loam	0–5	n.a.	422.1/7.2/2.9	11.3/0.3/0.1

At each site, three sampling plots were established: whole tree harvesting (WTH, only above-ground biomass harvested), stem-only harvesting (SOH) and control (C). Size of the plot varied from 3.00 to 3.75 ha. Suction tube lysimeters (lysimeter cup made of porous ceramic – 92% pure Al_2O_3 and body of trace metal-free PVC) at 2 depths (30 and 60 cm), open precipitation collectors, and groundwater wells were installed at all sampling plots to collect soil water, wet precipitation, and groundwater samples. Three pairs of lysimeters (30 and 60 cm), and one precipitation collector per sample plot were installed in autumn 2011. Already existing groundwater wells (established in the sites in 2006) were used for groundwater sampling but some of those were dry or did not correspond to site layout (slope) therefore groundwater sampling was possible only in the SOH and WTH plots of the *Hylocomiosa* site, and in the C and WTH plots of the *Myrtillosa* site. Groundwater table level was 10.1 m in *Hylocomiosa* site and 20.9 m in *Myrtillosa* site. Water samples were also taken from small streams bordering *Hylocomiosa* and *Oxalidosa turf. mel*. sites. Water samples were collected twice per month during the vegetation season in 2012 (reference period), 2013, and 2014 (first and second years following clearcutting). In the summer some of the lysimeters were sometimes dry but there was always at least one sample per plot per sampling time. Precipitation samples taken in the clearcut after felling are referred to as bulk precipitation, while those sampled under tree canopy are referred to as throughfall samples.No bulk precipitation samples were taken in the reference period, as the closest open area was located too far from the sites. Clearfelling was performed in early spring 2013 with harvester, timber was extracted and logging residues were removed with forwarder, following ‘business as usual’ principle. During harvest the soil was frozen, and no damage to the soil due to the movement of machinery was observed. At the whole-tree harvested plots all above-ground part of the tree was harvested (in practice this means that approximately 70% of tree tops and branches were removed). At the stem-only harvested plots only the stemwood was removed and logging residues were evenly scattered throughout the plot.

The following chemical parameters were measured in the water samples: pH determined according to LVS ISO 10523:2012; ammonium nitrogen (NH_4-N) determined using manual spectrometric method according to LVS ISO 7150/1:1984; nitrate nitrogen (NO_3-N), nitrite nitrogen (NO_2-N) and dissolved total nitrogen (DTN) determined using FORMACS^{HT} TOC/TN Analyzer (ND25 nitrogen detector) according to LVS EN 12260:2004. Dissolved organic nitrogen (DON) was calculated by subtracting NO_3-N, NO_2-N and NH_4-N concentrations from DTNconcentration. Preservation and handling of water samples were done according to ISO 5667-3:2012. Water samples were filtered using borosilicate glass fiber filters without a binder before determination of NH_4-N, NO_3-N and DTN. Limitof detection (LOD) forNH_4-N is 0.008 mg L^{-1} (9.4% of results below LOD), LOD for NO_3-N is 0.018 mg L^{-1} (4.4% of results below LOD) andLOD for DTN is 0.06 mg L^{-1} (0.4% of results below LOD).

Statistical differences in the monitored chemical parameters of precipitation, groundwater and surface water between study sites as well as significance of changes in monitored chemical parameters due to harvestingwere analyzed with Wilcoxon rank sum test with continuity correction. Statistical differences in monitored chemical parameters of precipitation, groundwater and surface water between different years (significant differences from reference period before harvesting) within each site were analyzed with Wilcoxon signed rank test with continuity correction. We used results of repeated-measures analysis of variance and Tukey’s honestly significant difference (HSD) test to assess the significance of treatment means (within each site) of monitored chemical parameters in soil solution. There were no statistically significant differences in pH, nitrate, ammonium, organic nitrogen and total nitrogen concentration in soil solution between 30 cm and 60 cm depth within each study site and year; consequently, we combined data from both soil solution sampling depths in a single statistical analysis. We used a 95% confidence level in all analyses. Data analysis was conducted in program R (R Core Team, 2015) for Linux.

RESULTS AND DISCUSSION

Precipitation

There were no statistically significant differences in pH, concentration of NO_3-N, NH_4-N, DON and DTN in throughfall and bulk precipitation between the study sites, but we found significant impact of harvesting on mean pH and ammonium concentration in precipitation reaching the ground if the data from all three sites were combined. Bulk precipitation pH values at the clearcut areas were significantly ($p = 0.015$) more alkaline compared with mean throughfall pH values at the control plots at all study sites (Table 3).

Table 3. Mean precipitation pH before (2012) and after (2013–2014) clearfelling in study period (April-October) at the study sites

Study site	Before harvesting	After harvesting	
	Forest	Clearcut	Control
Hylocomiosa	6.6 ± 0.2	6.8 ± 0.2	6.4 ± 0.1
Oxalidosa turf.mel.	6.5 ± 0.2	6.8 ± 0.2	6.5 ± 0.1
Myrtillosa	6.8 ± 0.3	6.9 ± 0.2	6.4 ± 0.2
Mean	6.6 ± 0.1	6.8 ± 0.1*	6.5 ± 0.1

*Significant differences between treatment and control after harvesting.

We detected significantly higher ($p = 0.023$) mean ammonium concentration in bulk precipitation at clearcut areas compared to throughfall precipitation at control plots after felling. Also a higher nitrate concentration in bulk precipitation at clearcut areas compared to throughfall precipitation at control plots was observed, but no significant differences between nitrate as well as DTN concentration in bulk and throughfall precipitation (in clearcut and control) were detected after felling. It should be noted that nitrate, ammonium, and total nitrogen concentration also decreased in throughfall of the control plots compared with the mean values in the study period before harvesting. After harvesting, DON concentration in throughfall was significantly higher than that in the bulk precipitation at *Hylocomiosa* and *Oxalidosa turf.mel.* sites ($p = 0.001$ in both cases). The same was true if precipitation data from all three sites were combined ($p = 0.000$) (Table 4).

At all plots, the N deposition was higher in the first year of the study due to higher throughfall amounts. In Latvia, 2012 was the second wettest year of the 21st century. The adsorption of N (both ammonium and nitrate) from wet deposition by tree canopies in the boreal zone has been demonstrated in several studies, and, at low levels of N deposition, N concentration in the throughfall is smaller than in the bulk precipitation (e.g., Hyvärinen, 1990; Nieminen, 1998; Kristensen et al., 2004). Opposite results are reported for areas with high N deposition (e.g., Kopáček et al., 2009; Drápelová, 2012). Also Nieminen (1998) observed increased nitrate concentration in throughfall compared to bulk precipitation in an area where N deposition was higher than the average for Finland. At our plots annual N deposition may be considered low to moderate (below or slightly above 5 kg ha^{-1}), and our results show slightly higher nitrate and ammonium concentrations in throughfall (but no significant differences). According to Cornell et al (2003), the contribution of the organic component to dissolved total nitrogen in precipitation in Europe is on average 23 ± 8%. At our sites, the DON proportion in the dissolved total nitrogen concentration ranged from 12.5 ± 3.22% to 16.2 ± 7.90% at clearcut plots and from 20.2 ± 7.75% to 54.3 ± 6.53% at control plots. Canopy functions as the source of DON, consequently DON concentration in the bulk precipitation was significantly lower at all study sites. No significant DON concentration differences from the reference period were observed, as these are not correlated with the mean annual precipitation (Michalzik et al., 2001). Despite the fact that DON contribution is highly variable and site specific, it is an important component of atmospheric nitrogen deposition.

Table 4. Mean concentration and annual input of nitrate, ammonium, dissolved total nitrogen and dissolved organic nitrogen with precipitation before (2012) and after (2013–2014) clearfelling in study period (April–October) at the study sites

Study site	Before harvesting		After harvesting			
	Forest		Clearcut		Control	
	Content, mg L^{-1}	Input, kg ha^{-1} yr^{-1}	Content, mg L^{-1}	Input, kg ha^{-1} yr^{-1}	Content, mg L^{-1}	Input, kg ha^{-1} yr^{-1}
NO_3-N						
Hylocomiosa	0.52 ± 0.17	1.83	0.27 ± 0.08	1.61	0.18 ± 0.07*	0.80
Oxalidosa turf.mel.	0.59 ± 0.13	2.46	0.23 ± 0.05*	1.15	0.18 ± 0.04*	0.75
Myrtillosa	0.71 ± 0.23	2.32	0.27 ± 0.08*	1.00	0.24 ± 0.06*	1.00
Mean	0.60 ± 0.10	2.20 ± 0.19	0.26 ± 0.04*	1.25 ± 0.18	0.20 ± 0.03*	0.85 ± 0.08
NH_4-N						
Hylocomiosa	0.29 ± 0.12	1.00	0.27 ± 0.10	1.67	0.13 ± 0.06	0.43
Oxalidosa turf.mel.	0.42 ± 0.13	1.60	0.23 ± 0.06	1.24	0.16 ± 0.11	0.55
Myrtillosa	0.30 ± 0.17	0.77	0.31 ± 0.13	1.46	0.22 ± 0.10	0.81
Mean	0.34 ± 0.08	1.12 ± 0.25	0.27 ± 0.06**	1.46 ± 0.12	0.17 ± 0.05*	0.60 ± 0.11
DTN						
Hylocomiosa	1.18 ± 0.53	3.74	0.59 ± 0.18	3.58	0.53 ± 0.13	2.24
Oxalidosa turf.mel.	1.71 ± 0.68	5.93	0.51 ± 0.11*	2.63	0.64 ± 0.14*	2.82
Myrtillosa	1.87 ± 1.05	6.14	0.62 ± 0.17	3.10	0.56 ± 0.17*	2.37
Mean	1.59 ± 0.43	5.27 ± 0.77	0.58 ± 0.08*	3.10 ± 0.27	0.58 ± 0.08*	2.48 ± 0.18
DON						
Hylocomiosa	0.36 ± 0.26	0.91	0.05 ± 0.02**	0.31	0.22 ± 0.03	1.01
Oxalidosa turf.mel.	0.70 ± 0.50	1.87	0.05 ± 0.01**	0.24	0.30 ± 0.03	1.52
Myrtillosa	1.01 ± 0.70	3.05	0.04 ± 0.02	0.24	0.11 ± 0.03	0.57
Mean	0.69 ± 0.29	1.94 ± 0.62	0.05 ± 0.01**	0.26 ± 0.02	0.21 ± 0.02	1.03 ± 0.27

*Significant differences from reference period/ **Significant differences between treatment and control.

Changes in pH and nitrogen content in soil water

Mean annual soil solution pH at C, SOH and WTH plots of all three sites before and after harvesting, as well as significant differences, are shown in Table 5. Gradual pH value decrease in the soil water after felling was observed nearly at all harvested plots, reaching significant levels at *Myrtillosa* SOH plot (p = 0.000) and *Oxalidosa turf.mel*. WTH plot (p = 0.001) in the second year after treatment.

Table 5. Mean soil solution pH at the study sites

Site	Treatment	pH 2012	pH 2013	pH 2014
Hylocomiosa	C	7.7 ± 0.1	7.4 ± 0.1	7.4 ± 0.1
	SOH	6.8 ± 0.3	6.8 ± 0.2*	6.2 ± 0.2*
	WTH	6.5 ± 0.3	6.7 ± 0.1*	6.1 ± 0.2*
Myrtillosa	C	6.3 ± 0.2	6.7 ± 0.1	6.7 ± 0.1
	SOH	6.8 ± 0.2	6.1 ± 0.1*	*5.2 ± 0.1**
	WTH	7.2 ± 0.3	6.7 ± 0.1**	6.7 ± 0.1**
Oxalidosa turf.mel.	C	7.3 ± 0.1	*7.5 ± 0.1*	7.4 ± 0.1
	SOH	7.5 ± 0.1*	7.7 ± 0.1	7.5 ± 0.1
	WTH	6.8 ± 0.1$^{*/**}$	6.1 ± 0.2$^{*/**}$	*5.4 ± 0.2*$^{*/**}$

*Significant differences between treatment and control/ **Significant differences between treatments/ Significant differences from reference period are indicated in italics.

The mean soil water pH value differences between treatments and control varied considerably depending on the year, plot and site (Fig. 2).

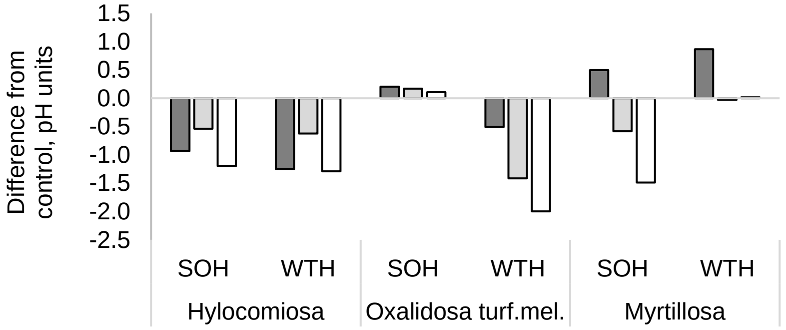

Figure 2. Mean pH value difference from control at the study sites. WTH – whole-tree harvesting; SOH – stem-only harvesting. Error bars show combined standard errors. Dark grey – 2012, light grey – 2013, white – 2014.

These were significant in 2013 and 2014 at SOH ($p = 0.034$ and $p = 0.000$, respectively) and WTH plots ($p = 0.014$ and $p = 0.000$, respectively) of *Hylocomiosa* and at SOH plot of *Myrtillosa* ($p = 0.000$ in both years after harvesting), in 2012 at SOH plot of *Oxalidosa turf. mel.*($p = 0.015$) but in all study years – at WTH plot of *Oxalidosa turf.mel.* ($p = 0.000$ in all three years). Significant soil water pH differences between whole-tree harvested and stem-only harvested plots were observed at *Myrtillosa* and *Oxalidosa turf.mel.* sites, both in 2013 and 2014 ($p = 0.000$ in all cases).

Mean annual plot NO_3-N and NH_4-N concentration at C, SOH and WTH plots of all three sites before and after harvesting, as well as significant differences, are shown in Table 6. Already prior to clearfelling, significant mean NO_3-N concentration differences between treatment and control plots were observed at *Myrtillosa* ($p = 0.000$ at WTH) and *Oxalidosa turf.mel.* ($p = 0.037$ at SOH and $p = 0.026$ at WTH) sites (Table 6). In 2013, the first year after felling, the soil water nitrate nitrogen concentration actually decreased at most plots, except SOH and WTH of *Myrtillosa* and WTH of *Oxalidosa turf.mel.* In the second year after harvesting, the soil water nitrate nitrogen concentration further decreased at all control plots (significantly at both sites on mineral soils; $p = 0.000$) but started to increase at both harvested plots of *Hylocomiosa* ($p = 0.016$ at SOH and $p = 0.002$ at WTH), SOH plot of *Myrtillosa* ($p = 0.000$) and WTH plot of *Oxalidosa turf.mel.*($p = 0.037$). At these plots also the difference from the control plot was most pronounced (Fig. 3). On the contrary, significant nitrate nitrogen decrease compared to the reference period was observed at SOH plot of *Oxalidosa turf.mel.*in 2014 ($p = 0.000$). No significant nitrate concentration differences between clearcut with whole-tree harvesting and clearcut with stem-only harvesting were observed at *Hylocomiosa* site but such differences were found at *Myrtillosa* site in 2012 ($p = 0.014$) and 2014 ($p = 0.000$) and at *Oxalidosa turf.mel.*site in 2013 and 2014 ($p = 0.000$) (Table 6).

Table 6. Mean nitrate and ammonium concentration in soil solution at the study sites

Site	Treatment	NO_3-N, mg L^{-1}			NH_4-N, mg L^{-1}		
		2012	2013	2014	2012	2013	2014
Hylocomiosa	C	1.76 ± 0.15	*0.38 ± 0.14*	*0.11 ± 0.04*	0.05 ± 0.02	0.11 ± 0.07	0.65 ± 0.58
	SOH	1.81 ± 0.30	1.66 ± 0.38	*5.49 ± 0.61**	0.05 ± 0.01	0.23 ± 0.10	0.39 ± 0.11
	WTH	2.07 ± 0.36	1.51 ± 0.47	*6.46 ± 0.61**	0.04 ± 0.01	0.08 ± 0.03	0.43 ± 0.18
Myrtillosa	C	0.58 ± 0.08	*0.13 ± 0.02*	*0.10 ± 0.03*	0.04 ± 0.01	0.02 ± 0.004	0.03 ± 0.01
	SOH	1.01 ± 0.09	2.98 ± 0.47*	*5.82 ± 0.61**	0.05 ± 0.01	0.02 ± 0.01	0.23 ± 0.07
	WTH	1.82 ± 0.36$^{*/**}$	1.99 ± 0.33*	1.10 ± 0.26$^{*/**}$	0.05 ± 0.01	0.02 ± 0.01	0.02 ± 0.01**
Oxalidosa turf.mel.	C	1.06 ± 0.23	0.84 ± 0.18	0.80 ± 0.19	0.06 ± 0.01	*0.02 ± 0.01*	*0.02 ± 0.01*
	SOH	3.10 ± 0.50*	*1.41 ± 0.36*	*0.96 ± 0.31*	0.05 ± 0.01	0.03 ± 0.01	*0.02 ± 0.004*
	WTH	3.87 ± 1.69*	7.62 ± 1.24$^{*/**}$	*10.72 ± 1.59*$^{*/**}$	0.12 ± 0.04**	0.15 ± 0.04$^{*/**}$	0.31 ± 0.08$^{*/**}$

*Significant differences between treatment and control/ **Significant differences between treatments/ Significant differences from reference period are indicated in italics.

NH_4-N concentration in the soil water was similar at all plots during the reference period (2012). Significant difference between treatment and control was observed only at WTH plot of *Oxalidosa turf.mel.* in 2013 and 2014 ($p = 0.000$) (Table 6, Fig. 3). Significant difference between plot with whole-tree harvesting and plot with stem-only harvesting was observed only at *Oxalidosa turf.mel.* site ($p = 0.000$ both in 2013 and 2014), but in this case ammonium concentration in the soil water at WTH plot was significantly higher already before treatment in 2012 ($p = 0.016$). No significant differences from the reference period were observed at both sites on mineral soils but significant decrease of ammonium concentration was detected at *Oxalidosa turf.mel.* site, at the control plot in 2013 and 2014 ($p = 0.000$) and at the SOH plot in 2014 ($p = 0.005$).

DTN concentration in the soil water tended to increase with the time but significant differences from the reference period were observed only at SOH ($p = 0.007$) and WTH

(p = 0.006) plots of *Hylocomiosa* in 2014. Quite contrary, at the SOH plot of *Oxalidosa turf.mel.* the total dissolved N concentration in the soil water in 2013 and 2014 was significantly lower than in 2012 (p = 0.011 and p = 0.003, respectively) (Table 7). Significant differences between treatment and control plots were observed at SOH and WTH plots of both sites on mineral soils (p = 0.021), as well as at the WTH plot of *Oxalidosa turf.mel.* (p = 0.000 in 2013 and 2014). It has to be noted, however, that significant differences between WTH and control plots of *Myrtillosa* and *Oxalidosa turf.mel.* existed already in the reference period (p = 0.001 and p = 0.007, respectively) (Fig. 3). In 2014, at the *Myrtillosa* WTH plot the DTN concentration in the soil water was significantly lower than at the SOH plot of the same site (p = 0.000). The opposite was true for the same plots in the reference period (p = 0.025) and for the *Oxalidosa turf.mel.* site in 2013 and 2014 (p = 0.000).

Table 7. Mean dissolved total and organic nitrogen concentration in soil solution at the study sites

Site	Treatment	DTN, mg L^{-1}			DON, mg L^{-1}		
		2012	2013	2014	2012	2013	2014
Hylocomiosa	C	1.86 ± 0.22	1.03 ± 0.30	1.43 ± 0.70	0.03 ± 0.01	0.41 ± 0.12	*0.46* *± 0.10*
	SOH	3.23 ± 0.68	3.17 ± 0.53*	*9.36* *± 1.12**	0.93 ± 0.87	1.35 ± 0.39	3.38 ± 0.76
	WTH	2.20 ± 0.36	2.17 ± 0.45	*11.09* *± 1.64**	0.09 ± 0.03	0.86 ± 0.17	*4.59* *± 1.32**
Myrtillosa	C	0.68 ± 0.07	0.38 ± 0.04	0.89 ± 0.50	0.06 ± 0.03	0.20 ± 0.04	0.26 ± 0.05
	SOH	1.10 ± 0.10	3.73 ± 0.52*	*7.84* *± 0.73**	0.03 ± 0.01	0.72 ± 0.18*	*1.70* *± 0.27*
	WTH	1.90 ± 0.37$^{*/**}$	2.35 ± 0.34*	1.74 ± 0.33**	0.04 ± 0.01	0.33 ± 0.07	*0.50* *± 0.11***
Oxalidosa turf.mel.	C	2.96 ± 0.49	2.42 ± 0.29	2.56 ± 0.27	2.24 ± 0.40	2.01 ± 0.19	1.78 ± 0.16
	SOH	5.47 ± 0.99	*2.87* *± 0.41*	*2.57* *± 0.46*	2.32 ± 0.58	1.63 ± 0.22	1.59 ± 0.20
	WTH	8.37 ± 2.51*	12.27 ± 2.23$^{*/**}$	15.16 ± 1.62$^{*/**}$	5.42 ± 2.30*	6.37 ± 1.50$^{*/**}$	4.38 ± 0.50$^{*/**}$

*Significant differences between treatment and control/ **Significant differences between treatments/ Significant differences from reference period are indicated in italics.

In the reference period, DON concentration in the soil water was highest at WTH plot of *Oxalidosa turf.mel.*site, and the difference from control was significant (p = 0.040). Dissolved organic N concentration at the WTH plot of *Oxalidosa turf.mel.* remained significantly higher than at the control site also in 2013 and 2014 (p = 0.000). Also at SOH plot of Myrtillosa in 2013 and WTH plot of Hylocomiosa in 2014 DON concentration was significantly higher than at the control plots (p = 0.043 and p = 0.027, respectively). Dissolved organic N concentration in the soil water at both sites on mineral soil tended to increase after the harvesting; significant increase compared to the reference period were observed in the second year after harvesting at SOH and WTH plots of Myrtillosa (p = 0.000 and p = 0.009, respectively), WTH plot of *Hylocomiosa* (*p* = 0.049), but also at control plot of *Hylocomiosa*. In 2014, soil water DON concentration at SOH plot of *Myrtillosa* was significantly higher than that at WTH plot of the same site (*p* = 0.023). The opposite was true for harvested plots of *Oxalidosa turf.mel.*, both in 2013 and 2014 (*p* = 0.000) (Table 7, Fig. 3).

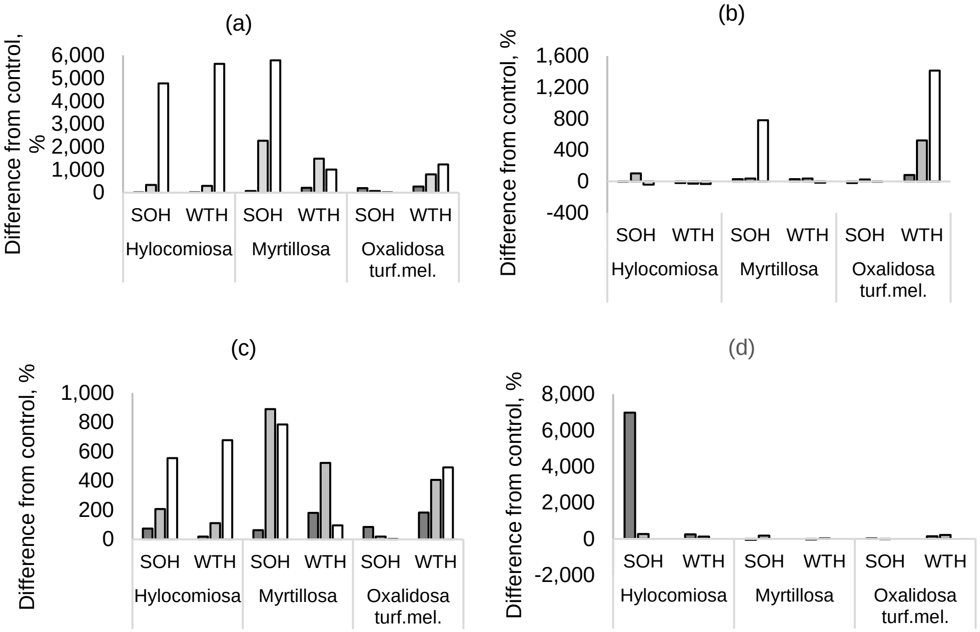

Figure 3. Nitrate (a), ammonium (b), dissolved total nitrogen (c) and dissolved organic nitrogen (d) concentration difference (%) from control in soil solution at the study sites. Dark grey – 2012, light grey – 2013, white – 2014.

High nitrate concentrations in the soil water may suggest high potential for leaching losses of nitrogen. Nieminen (1998) reported increase of the ammonium concentration in soil water in the first year after clearfelling but no statistically significant DON or nitrate differences. At our sites no significant changes of nitrate, ammonia or DON concentration in the soil water at the harvested plots were observed in 2013, except significant decrease of nitrate at the SOH plot of *Oxalidosa turf.mel*. In 2014, elevated N concentrations in the soil solution were detected at several but not all plots. Differences between harvested and control plots were more explicit in the second year after harvesting. Ring (1996, 2004) explains elevated nitrate N content with increased nitrification. Also our results suggest this, since nitrate concentrations were elevated at those plots where pH was lowered. Previous results suggest that there may be a considerable time lag following harvest before significant changes of N content in the soil water are demonstrated (Ring, 1996; Löfgren et al., 2009; Futter et al., 2010). Concentration increase of inorganic N compounds in the soil water after felling may be caused by higher mineralization rate of organic matter due to soil disturbance, changed microclimate and availability of logging residues, increased runoff and disrupted nitrogen uptake by vegetation (Gundersen et al., 2006; Löfgren et al., 2009). Data on this are, however, not consentient, e.g., Palviainen et al. (2004) concluded that logging residues may cause significant N immobilization with no net release during the first years after harvesting. Bergholm et al. (2015) also suggested that immobilization of N by stump and root necromass may be of importance. Evidence on positive relationship between the site productivity or N deposition and nitrogen leaching is repeatedly reported (e.g., Wiklander et al., 1983; Bredemeier et al., 1998; Rothe & Mellert, 2004). A few studies on richer Norway spruce sites found increasing nitrate concentrations in soil water immediately after clear-cut (Berdén et al., 1997; Hedwall et al., 2013). Leaching of inorganic N after clear-cutting normally lasts for 5-8 years, with a peak after 1–2 years (Huber et al., 2004; Ring, 2004; Futter et al., 2010; Hedwall et al., 2013). The peak concentration of NO_3-N in the soil solution can vary from below 0.5 mg L^{-1} (Berdén et al., 1997) to 30 mg L^{-1} (Huber et al., 2004). As our sites represent medium fertility conditions and the N deposition is low, it can be expected that concentration of N compounds in the soil water may still increase in the following years. We hypothesized that soil solution N concentration would differ in clearcuts with all above-ground biomass and stem-only biomass removed. Indeed, we found significant differences between SOH and WTH plots at both our sites that differed most in site productivity, but the pattern of difference was opposite. At the *Oxalidosa* (more productive) site concentration of all N compounds was higher at the WTH plot, both in 2013 and 2014. At the *Myrtillosa* (less productive) site concentration of all N compounds was higher at the SOH plot, and significant differences were demonstrated only in 2014. Ring et al (2001) found no significant effect of brash removal on soil solution N concentration 5 years after felling. At our plots, the differences between treatments may still increase in the following years. Model simulations performed by Laurén et al. (2005) suggested that most important sinks of N after clearfelling are immobilization by the soil microbes, uptake by ground vegetation and sorption to soil. Stem-only biomass removal with brash left on site may suppress the ground vegetation, and this is most likely an important factor influencing soil water N content at the *Myrtillosa* site on poor sandy soil where soil microbial activity and soil sorption capacity is low.

Changes in pH and nitrogen content in groundwater

Due to the reasons explained in the Material and Method, groundwater sampling was possible only at the SOH and WTH plots of the *Hylocomiosa* site and at the C and WTH plots of the *Myrtillosa* site. In the reference period, mean NO_3-N concentration in the groundwater was the highest at SOH plot of *Hylocomiosa* and lowest at control plot of *Myrtillosa*. Significant difference was observed only between SOH and WTH plots of *Hylocomiosa* in 2012 ($p = 0.027$). In 2013 and 2014, the nitrate nitrogen concentration in groundwater decreased at all plots, and the difference from the reference period was significant in all cases ($p < 0.022$) (Table 8, Fig. 4).

Groundwater NH_4-N concentration in the reference period was highest at the SOH plot of *Myrtillosa* and lowest at the WTH plots at both sites, but significant differences were observed only between the SOH and WTH plots of *Hylocomiosa* ($p = 0.043$). Also, the ammonium concentration in ground water decreased after felling; the difference from the reference period was significant at the WTH plot of *Hylocomiosa* ($p = 0.035$) and at the C plot of *Myrtillosa* ($p = 0.028$) in 2013, and at the SOH plot of *Hylocomiosa* ($p = 0.014$), as well as at the C and WTH plot of *Myrtillosa* in 2014 ($p = 0.010$ and $p = 0.005$, respectively) (Table 8, Fig. 4).

DTN concentration at all plots was highest in the reference period. Significant differences were detected between the SOH and WTH plots of *Hylocomiosa* in 2013 ($p = 0.038$), but difference was not significant in 2014. At all plots, there was a significant total nitrogen concentration difference from the reference period in both 2013 and 2014 ($p < 0.002$).

DON concentration in groundwater tended to decrease after harvesting, significant decrease compared to the reference period was observed at WTH plot of *Hylocomiosa* in 2013 ($p = 0.024$) and at all sampled plots in 2014 – SOH and WTH of *Hylocomiosa* ($p = 0.013$ and $p = 0.042$, respectively) and C and WTH plots of *Myrtillosa* ($p = 0.005$ and $p = 0.017$, respectively). Significant differences between plots of the same site were detected only at *Myrtillosa* site in 2014, with groundwater DON concentration significantly higher at the harvested than at the control plot ($p = 0.001$) (Table 8, Fig. 4).

Table 8. Mean pH, nitrate, ammonium, dissolved total nitrogen and dissolved organic nitrogen concentration in groundwater at the study sites

Site	Treatment	2012	2013	2014
pH				
Hylocomiosa	SOH	8.0 ± 0.1	7.9 ± 0.1	8.0 ± 0.1
	WTH	8.0 ± 0.1	7.8 ± 0.2	8.1 ± 0.1
Myrtillosa	C	7.9 ± 0.1	8.1 ± 0.1**	8.0 ± 0.1
	WTH	7.7 ± 0.1	8.0 ± 0.1**	8.0 ± 0.1**
NO_3-N, mg L^{-1}				
Hylocomiosa	SOH	1.07 ± 0.018	0.20 ± 0.04**	0.08 ± 0.03**
	WTH	0.59 ± 0.20*	0.12 ± 0.02**	0.05 ± 0.01**
Myrtillosa	C	0.47 ± 0.10	0.07 ± 0.02**	0.04 ± 0.01**
	WTH	0.57 ± 0.10	0.09 ± 0.01**	0.04 ± 0.01**
NH_4-N, mg L^{-1}				
Hylocomiosa	SOH	0.06 ± 0.02	0.03 ± 0.01	0.02 ± 0.01**
	WTH	0.02 ± 0.01*	0.01 ± 0.003*/**	0.01 ± 0.004
Myrtillosa	C	0.03 ±0.003	0.01 ± 0.01**	0.01 ± 0.002**
	WTH	0.02 ± 0.01	0.03 ± 0.003	0.01 ± 0.002**
DTN, mg L^{-1}				
Hylocomiosa	SOH	1.67 ± 0.32	0.86 ± 0.52**	0.20 ± 0.07**
	WTH	1.06 ± 0.26	0.19 ± 0.02*/**	0.11 ± 0.01**
Myrtillosa	C	1.09 ± 0.40	0.16 ± 0.03**	0.08 ± 0.01**
	WTH	1.81 ± 0.72	0.19 ± 0.02**	0.13 ± 0.01*/**
DON, mg L^{-1}				
Hylocomiosa	SOH	0.57 ± 0.25	0.63 ± 0.53	0.10 ± 0.05**
	WTH	0.45 ± 0.22	0.06 ± 0.02**	0.03 ± 0.01**
Myrtillosa	C	0.60 ± 0.43	0.08 ± 0.03	0.03 ± 0.01**
	WTH	1.22 ± 0.76	0.08 ± 0.02	0.07 ± 0.01*/**

*Significant differences between treatments within year/ **Significant differences from reference period within treatment.

Part of the N from the soil solution is leached to groundwater and further exported to streams. Kubin (1998) reported elevated N concentration in the groundwater as long as 10 years after clearfelling and slash removal at the middle boreal conifer forest zone. In our study, nitrate, ammonium, dissolved organic nitrogen and dissolved total nitrogen concentration in groundwater actually decreased after felling. It has to be noted, however, that groundwater level was very low at all plots; therefore, it is possible that the concentrations of N-compounds in groundwater were not at all affected by felling.

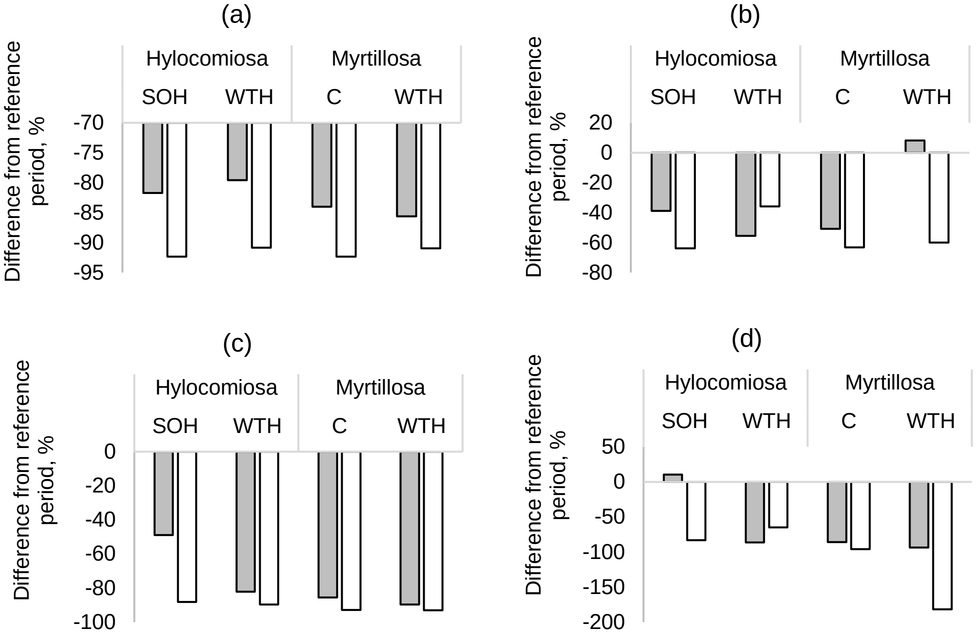

Figure 4. Nitrate (a), ammonium (b), dissolved total nitrogen (c) and dissolved organic nitrogen (d) concentration difference from reference period (2012) in groundwater at the study sites WTH – whole-tree harvesting; SOH – stem-only harvesting; C – control. Grey – 2013, white – 2014.

Changes in pH and nitrogen content in surface water

In 2012, the mean NO_3-N concentration in the streams bordering the *Hylocomiosa* and *Oxalidosa turf. mel.* sites was similar. Concentrations decreased significantly in 2013 (p = 0.013 in *Hylocomiosa* and p = 0.000 in *Myrtillosa*) and increased again in 2014, but did not reach the reference period level (Table 9).

Mean NH_4-N concentration in both streams was the same in 2012, and it decreased in the following years, with the differences from the reference period being significant (in *Hylocomiosa*, p = 0.003 and p = 0.004 in 2013 and 2014; in *Oxalidosa turf.mel.*, p = 0.000 in both 2013 and 2014).

DTN concentration was initially (in 2012) higher in the stream bordering the *Hylocomiosa* site. Also in this case the pattern of change was rather similar: a decrease was observed in 2013, and an increase occurred again in 2014. Differences from the reference period were significant in both streams in both years after harvesting (p = 0.000), and the reference period concentration of total nitrogen after the clearfelling was not reached.

DON concentration followed the same pattern of change – decrease in 2013, slight increase in 2014 but below the level of reference period. Significant differences from the reference period were observed in the stream bordering *Hylocomiosa* both in 2013 (p = 0.001) and 2014 (p = 0.004) and in the stream bordering *Oxalidosa turf.mel.* in 2013 (p = 0.000) (Table 9, Fig. 5).

Table 9. Mean pH, nitrate, ammonium, dissolved total nitrogen and dissolved organic nitrogen concentration in surface water at the study sites

Site	2012	2013	2014
pH			
Hylocomiosa	7.9 ± 0.1	8.1 ± 0.1*	8.0 ± 0.1
Oxalidosa urf.mel.	7.9 ± 0.1	8.1 ± 0.1*	8.1 ± 0.1*
NO_3-N, mg L^{-1}			
Hylocomiosa	0.67 ± 0.12	0.24 ± 0.06*	0.29 ± 0.04*
Oxalidosa urf.mel.	0.69 ± 0.13	0.20 ± 0.06*	0.66 ± 0.22
NH_4-N, mg L^{-1}			
Hylocomiosa	0.03 ± 0.003	0.01 ± 0.003*	0.02 ± 0.004*
Oxalidosa urf.mel.	0.03 ± 0.003	0.01 ± 0.003*	0.01 ± 0.001*
DTN, mg L^{-1}			
Hylocomiosa	2.01 ± 0.27	0.80 ± 0.08*	0.90 ± 0.09*
Oxalidosa urf.mel.	1.38 ± 0.21	0.36 ± 0.08*	1.08 ± 0.35
DON, mg L^{-1}			
Hylocomiosa	1.51 ± 0.29	0.55 ± 0.05*	0.59 ± 0.07*
Oxalidosa urf.mel.	0.69 ± 0.16	0.15 ± 0.03*	0.41 ± 0.14

*Significant differences from reference period within each site.

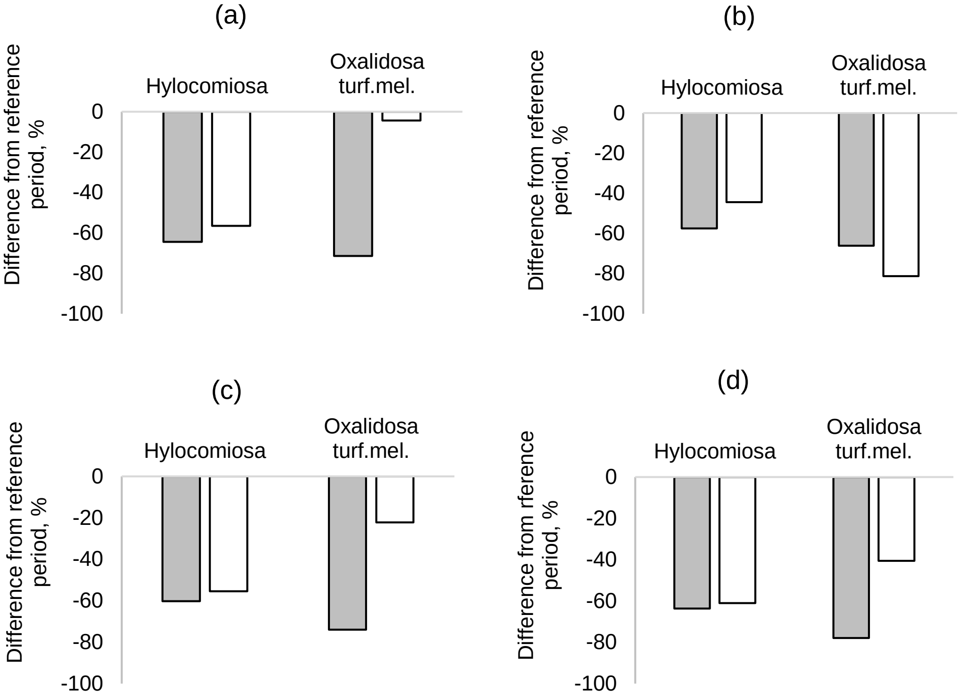

Figure 5. Nitrate (a), ammonium (b), dissolved total nitrogen (c) and dissolved organic nitrogen (d) concentration difference from reference period (2012) in surface water at the study sites. Grey – 2013, white – 2014.

Forest management activities have the potential to adversely affect downstream water quality (Futter et al., 2010). Studies in Sweden have demonstrated that while all forest management activities can impact surface water quality, the effects of final felling and subsequent site preparation are the most dramatic (Löfgren et al., 2014). In the study by Nieminen (2004) on Norway spruce forests growing on drained peatlands in southern Finland, outflow concentrations of NH_4-N and NO_3-N increased at clear-cut areas, but large differences were observed between sites. The average annual increase in NH_4-N concentrations during the first four years after clear-cutting ranged from 0.04 mg L^{-1} to 0.31 mg L^{-1}; increases for NO_3^--N ranged from 0.05 to 0.22 mg L^{-1}. Doubled total N and nitrate concentration in brooks after clearfelling was reported by Ahtiainen and Huttunen (1999). According to study by Nieminen (2003), leaching of dissolved N was most pronounced in the second and third year after treatment, and it was favored by clearcutting with ditching and mounding while clearcutting alone and clearcutting with mounding did not cause significant changes. Results from streams bordering our study objects demonstrated a decrease of nitrate, ammonium, and dissolved organic nitrogen concentration in the first year after clearfelling. Concentrations started to increase in the second year, but the concentration of N-compounds did not reach the level of the reference period. Not all dissolved N is exported to streams, large differences between soil solution and streamwater nitrate concentrations may be observed (Futter et al., 2010). Typically streamwater N concentration is lower than that of the soil solution, as it was also in our case. Riparian buffer zone may attenuate N export to streams by denitrification, immobilization and plat uptake (Gundersen et al., 2010). The processes at the riparian zones are, however, complex and not yet fully clear, and N attenuation may strongly depend on site hydrology and other specific local conditions of the area that should be considered. Design of site-specific buffer zones may provide for reduced leaching of dissolved N to surface waters, as well as for reduction of forest management costs (Ågren et al., 2014; Kuglerová et al., 2014). These topics are presently of large interest in the Nordic-Baltic forest research community and are closely followed also by the authors of current study.

Our results are the first preliminary contribution to the quantification of inorganic nitrogen in soil-, ground-, and surface waters following two types of clearfelling (whole tree harvesting and stem-only harvesting) in Latvia. Sampling in the experimental sites is being continued, with soil preparation carried out in autumn 2014 and planting in spring 2015. As the impact of harvesting on N leaching usually lasts at least 5 years after felling further monitoring of the plots will be carried out to determine middle-term effects of forest management. Longer study period, calculation of N fluxes and inclusion of additional factors in the analysis (e.g., growth of the young stand, ground vegetation dynamics, amount of slash, nitrogen content changes in the buffer zone), will provide us with the results that will further contribute to better understanding of nutrient cycling processes in the forest ecosystems after forest management operations and the nature of possible differences between Nordic and Baltic countries.

CONCLUSIONS

Soil solution nitrate nitrogen concentration at the harvested plots was elevated in 2014 while pH values were lowered, suggesting enhanced nitrification. Generally, increase of the concentration of N-compounds after clearfelling, as compared with the reference period, was observed only in the second year after clearfelling. The differences between WTH and SOH varied depending on the site. Vegetation cover and soil properties are likely the most important influencing factors but quantification of this impact requires additional data and analyses.

Nitrogen concentration in the groundwater decreased after the clearfelling. Forests on dry mineral soils with very low groundwater levels are probably not subject to the risk of groundwater pollution after clearfelling.

No elevated nitrogen concentration in streamwater was observed during the first and second year after harvesting. This is most likely related to N attenuation by the forested buffer between the clearcut and stream, and effect of the bufferzone will be further investigated, as the sampling continues.

ACKNOWLEDGEMENTS. This study was conducted with the support of project Nr. L-KC-11-0004, ‘Methods and Technologies to Increase the Capital Value of Forest’, in collaboration with the *Forest Sector Competence Centre* and the Joint Stock Company *Latvian State Forests*. We gratefully acknowledge support from *Forest Research Station* in Kalsnava Forest district in performing forestry operations at the study sites and valuable advice and help from our colleagues at *Silava* – Roberts Matisons and Martins Lukins.

REFERENCES

Ågren, A., Lidberg, W., Strömgren, M., Ogilvie, J. & Arp, P. 2014. Evaluating digital terrain indices for soil wetness mapping – a Swedish case study. *Hydrology and Earth System Sciences* **18**, 3623–3634.

Ahtiainen, M. 1992. The effects of forest clear-cutting and scarification on the water quality of small brooks. *Hydrobiologia* **243/244**, 465–473.

Ahtiainen, M. & Huttunen, P. 1999. Long-term effects of forestry management on water quality and loading in brooks. *Boreal Environment Research* **4**, 101–114.

Åström, M., Dynesius, M., Hylander, K. & Nilsson, C. 2005. Effects of slash harvest on bryophytes and vascular plants in southern boreal forest clear-cuts. *Journal of Applied Ecology* **42**, 1194–1202.

Bengtsson, J., Lundkvist, H., Saetre, P., Sohlenius, B. & Solbreck, B. 1998. Effects of organic matter removal on the soil food web: Forestry practices meets ecological theory. *Applied Soil Ecology* **9**, 137–143.

Bengtsson, J. & Wikström, F. 1993. Effects of whole-tree harvesting on the amount of soil carbon: model results. *New Zealand Journal of Forest Science* **23**, 380–389.

Berdén, M., Nilsson, S.I. & Nyman, P. 1997. Ion leaching before and after clear-cutting in a Norway spruce stand – effects of long-term application of ammonium nitrate and superphosphate. *Water, Air and Soil Pollution* **93**, 1–26.

Bergholm, J., Olsson, B.A., Vegerfors, B. & Persson, T. 2015. Nitrogen fluxes after clear-cutting. Ground vegetation uptake and stump/root immobilisation reduce N leaching after experimental liming, acidification and N fertilisation. *Forest Ecology and Management* **342**, 64–75.

Bishop, K., Allan, C., Bringmark, L., Garcia, E., Hellsten, S., Hogbom, L., Johansson, K., Lomander, A., Meili, M., Munthe, J., Nilsson, M., Porvari, P., Skyllberg, U., Sorensen, R., Zetterberg, T. & Akerblom, S. 2009. The effects of forestry on Hg bioaccumulation in nemoral/boreal waters and recommendations for good silvicultural practice. *Ambio* **38**, 373–380.

Bredemeier, M., Blanck, K., Xu, Y.J., Tietema, A., Boxman, A.V., Emmett, B., Moldan, F., Gundersen, P., Schleppi, P. & Wright, R.F. 1998. Input-output budgets at the NITREX sites. *Forest Ecology and Management* **101**, 57–64.

Carpenter, S.R., Caraco, N.F., Correll, D.L., Howarth, R.W., Sharpley, A.N. & Smith, V.H. 1998. Nonpoint pollution of surface waters with phosphorus and nitrogen. *Ecological Applications* **8**, 559–568.

Cornell, S.E., Jickells, T.D., Cape, J.N., Rowland, A.P. & Duce, R.A. 2003 Organic nitrogen deposition on land and coastal environments: a review of methods and data. *Atmospheric Environment* **37**, 2173–2191.

De Wit, H.A., Granhus, A., Lindholm, M., Kainz, M.J., Lin, Y., Veiteberg Braaten, H.F. & Blaszczak, J. 2014. Forest harvest effects on mercury in streams and biota in Norwegian boreal catchments. *Forest Ecology and Management* **324**, 52–63.

Drápelová, I. 2012. Organic and inorganic nitrogen in precipitation and in forest throughfall at the Bílý Kříž site (Beskydy Mts., Czech Republic) *Journal of Forest Science* 58(2), 88–100.

Dzilna, I. 1970. Resources, composition and dynamics of groundwater in the Baltics. Riga, 185 pp. (in Russian).

EUSTAFOR & Patterson, T. (2011). *Ecosystem Services in European State Forests*. European State Forest Association, Brussels, 40 pp.

FAO 2010. *Global Forest Resources Assessment 2010*. FAO Forestry Research Paper 163, Rome, 387 pp.

Fisher, B., Turner, R.K. & Morling, P. 2009. Defining and classifying ecosystem services for decision making. *Ecological Economics* **68**, 643–653.

Futter, M.N., Ring, E., Högbom, L., Entenmann, S. & Bishop, K.H. 2010. Consequences of nitrate leaching following stem-only harvesting of Swedish forests are dependent on spatial scale. *Environmental Pollution* **158**, 3552–3559.

Gundersen, P., Laurén, A., Finér, L., Ring, E., Koivusalo, H., Saetersdal, M., Weslien, J.O., Sigurdsson, B.D., Högbom, L., Laine, J. & Hansen, K. 2010. Environmental services provided from riparian forests in the nordic countries. *Ambio* **39**, 555–566.

Gundersen, P., Schmidt, I.K. & Raulund-Rasmussen, K. 2006. Leaching of nitrate from temperate forests e effects of air pollution and forest management. *Environmental Reviews* **14**, 1–57.

Gunnarsson, B., Nittérus, K. & Wirdenäs, P. 2004. Effects of logging residue removal on ground-active beetles in temperate forests. *Forest Ecology and Management* **201**, 229–239.

Hedwall, P.O., Grip, H., Linder, S., Lövdahl, L., Nilsson, U. & Bergh, J. 2013. Effects of clear-cutting and slash removal on soil water chemistry and forest-floor vegetation in a nutrient optimised Norway spruce stand. *Silva Fennica* **47**, 2, 16 pp. [online 16 June 2015] URL: http://dx.doi.org/10.14214/sf.933

Huber, C., Weis, W., Baumgarten, M. & Göttlein, A. 2004. Spatial and temporal variation of seepage water chemistry after femel and small scale clear-cutting in a N-saturated Norway spruce stand. *Plant and Soil* **267**, 23–40.

Hyvärinen, A. 1990. Deposition on forest soil – effect of tree canopy on throughfall. In: *Acidification in Finland*, Kauppi et al. (Eds.). Springer-Verlag Berlin, Heidelberg, pp. 199–213.

Indriksons, A. 2010. Cycle of biogenous elements in drained forests. *Resume of the PhD thesis*, Jelgava, 64 pp.

Indriksons, A. & Zālītis, P. 2000. The impact of hydrotechnical drainage on cycle of some biogenous elements in forest. *Baltic Forestry* **6**(1), 18–24.

Kopáček,J., Turek,J., Hejzlar, J. &Šantrůčková, H. 2009. Canopy leaching of nutrients and metals in a mountain spruce forest. *Atmospheric Environment* **43**, 5443–5453.

Kreutzweiser, D.P., Haylett, P.W. & Gunn, J.M. 2008. Logging impacts on the biogeochemistry of boreal forest soils and nutrient export to aquatic systems: a review. *Environmental Reviews* **16**, 157–179.

Krieger, D.J. 2001. *The Economic value of forest ecosystem services: a review*. The Wilderness Society, Washington D.C., 40 pp.

Kristensen, H.L., Gundersen, P., Callesen, I.& Reinds, G.J. 2004. Throughfall nitrogen deposition has different impacts on soil solution nitrate concentration in European coniferous and deciduous forests. *Ecosystems* **7**, 180–192.

Kubin, E. 1998. Leaching of nitrate nitrogen into the groundwater after clear felling and site preparation. *Boreal Environment Research* **3**, 3–8.

Kuglerová, L., Ågren, A., Jansson, R. & Laudon, H. 2014. Towards optimizing riparian buffer zones: Ecological and biogeochemical implications for forest management. *Forest Ecology and Management* **334**, 74–84.

Latvian Ministry of Economics. 2010. Republic of Latvia National Renewable Energy Action Plan for implementing Directive 2009/28/EC of the European Parliament and of the Council of 23 April 2009 on the promotion of the use of energy from renewable sources and amending and subsequently repealing Directives 2001/77/EC and 2003/30/EC by 2020, *Latvian Ministry of Economics*, Riga, pp. 103. [online 16 June 2015] URL: http://ec.europa.eu/energy/sites/ener/files/documents/dir_2009_0028_action_plan_latvia.zip

Laudon, H., Hedtjärn, J., Schelker, J., Bishop, K., Sørensen, R. & Ågren, A. 2009. Response of Dissolved Organic Carbon following Forest Harvesting in a Boreal Forest. AMBIO: A *Journal of the Human Environment* **38**(7), 381–386.

Laurén, A., Finér, L., Koivusalo, H., Kokkonen, T., Karvonen, T., Kellomäki, S., Mannerkoski, H. & Ahtiainen, M. 2005. Water and nitrogen processes along a typical water flowpath and streamwater exports from a forested catchment and changes after clear-cutting: a modelling study. *Hydrology & Earth System Sciences* **9**(6):657–674.

Löfgren, S., Fröberg, M., Yu, J., Nisell, J. & Ranneby, B. 2014. Water chemistry in 179 randomly selected Swedish headwater streams related to forest production, clear-felling and climate. *Environmental Monitoring and Assessment* **186**(12), 8907–8928.

Löfgren, S., Ring, E., von Brömssen, C., Sørensen, R. & Högbom, L. 2009. Short-term effects of clear-cutting on the water chemistry of two boreal streams in northern Sweden: A paired catchment study. *Ambio* **38**, 347–356.

Maes, J., Teller, A., Erhard, M., Liquete, C., Braat, L., Berry, P., Egoh, B., Puydarrieux, P., Fiorina, C., Santos, F., Paracchini, M.L., Keune, H., Wittmer, H. & Hauck, J. 2013. *Mapping and Assessment of Ecosystems and their Services. An analytical framework for ecosystem assessments under action 5 of the EU biodiversity strategy to 2020*. Publications office of the European Union, Luxembourg. 60 pp.

Merganičová, K., Pietsch, S.A. & Hasenauer, H. 2005. Testing mechanistic modelling to assess impacts of biomass removal. *Forest Ecology and Management* **207**, 37–57.

Michalzik, B., Kalbitz, K., Park, J.H., Solinger, S. & Matzner, E. 2001. Fluxes and concentrations of dissolved organic carbon and nitrogen – a synthesis for temperate forests. *Biogeochemistry* **52**, 173–205.

Miettinen, J., Ollikainen, M., Finér, L., Koivusalo, H., Laurén, A. & Valsta, L. 2012. Diffuse load abatement with biodiversity co-benefits: the optimal rotation age and buffer zone size. *Forest Science* **58**(4), 342–352.

Miura, S., Amacher, M., Hofer, T., San-Miguel-Ayanz, J., Ernawati & Thackway, R. 2015. Protective functions and ecosystem services of global forests in the past quarter-century. *Forest Ecology and Management* **352**, 35–46.

Munthe, J. & Hultberg, H. 2004. Mercury and methylmercury in runoff from a forested catchment – concentrations, fluxes, and their response to manipulations. *Water, Air and Soil Pollution: Focus* **4**, 607–618.

Nasi, R., Wunder, S. & Campos, J.J. 2002. *Forest Ecosystem services: can they pay our way out of deforestation?* A discussion paper for the Forestry Roundtable to be held in conjunction with the UNFF II, Costa Rica on March 11, 2002. 38 pp.

Nieminen, M. 1998. Changes in nitrogen cycling following the clearcutting of drained peatland forests in southern Finland. *Boreal Environment Research* **3**, 9–21.

Nieminen, M. 2003. Effects of clear-cutting and site preparation on water quality from a drained Scots pine mire in southern Finland.*Boreal Environment Research* **8**,5 3–59.

Nieminen, M. 2004. Export of dissolved organic carbon, nitrogen and phosphorus following clear-cutting of three Norway spruce forests growing on drained peatlands in southern Finland. *Silva Fennica* **38**(2), 123–132.

Nikodemus, O., Kārkliņš, A., Kļaviņš, M. & Melecis, V. 2009. *Sustainable use and protection of soil.* Rīga, LU Akadēmiskais Apgāds, 256 pp. (in Latvian).

Olsson, B.A. & Staaf, H. 1995. Influence of harvesting intensity of logging residues on ground vegetation in coniferous forests. *Journal of Applied Ecology* **32**, 640–654.

Palviainen, M., Finér, L., Kurka, A.M., Mannerkoski, H., Piirainen, S. & Starr, M. 2004. Decomposition and nutrient release from logging residues after clear-cutting of mixed boreal forest. *Plant and Soil* **263**(1), 53–67.

Porvari, P., Verta, M., Munthe, J. & Haapanen, M. 2003. Forestry practices increase mercury and methyl mercury output from boreal forest catchments. *Environmental Science & Technology* **37**, 2389–2393.

Powell, I., White, A. & Landell-Mills, N. 2002. *Developing markets for the ecosystem services of forests*. Forest Trends Washington D.C., 12 pp.

Ring, E. 1996. Effects of previous N fertilizations on soil-water pH and N concentrations after clear-felling and soil scarification at a Pinus sylvestris site. *Scandinavian Journal of Forest Research* **11**, 7–16.

Ring, E. 2004. Experimental N fertilization of Scots pine: effects on soil-solution chemistry 8 years after final felling. *Forest Ecology and Management* **188**, 91–99.

Ring, E., Högbom, L. & Nohrstedt, H.Ö. 2001. Effects of brash removal after clear felling on soil and soil solution chemistry and field layer biomass in an experimental nitrogen gradient. *The Scientific World* **1**(S2), 457–466.

Rolff, C. & Ågren, G.I. 1999. Predicting effects of different harvesting intensities with a model of nitrogen limited forest growth. *Ecological Modelling* **118**, 193–211.

Rothe, A. & Mellert, K.H. 2004. Effects of Forest Management on Nitrate Concentrations in Seepage Water of Forests in Southern Bavaria, Germany**.** *Water, Air, and Soil Pollution* **156**(1), 337–355.

Thiffault, E., Hannam, K.D., Pare, D., Titus, B.D., Hazlett, P.W., Maynard, D.G. & Brais, S. 2011. Effects of forest biomass harvesting on soil productivity in boreal and temperate forests – a review. *Environmental Reviews* **19**, 278–309.

Virbulis, A., Bethers, U., Saks, T., Sennikovs, J. & Timuhins, A. 2013. Hydrogeological model of the Baltic artesian basin. *Hydrogeology Journal* **21**(4), 845–862.

Wall, A. 2012. Risk analysis of effects of whole-tree harvesting on site productivity. *Forest Ecology and Management* **282**, 175–184.

Wiklander, G. 1983. Nitrogen leaching from fertile forest land in southern Sweden. Stockholm. *Journal of the Royal Forestry and Agricultural Academy* **1122**, 311–317 (in Swedish).

Wunder, S. & Jellesmark-Thorsen, B. 2014. Quantifying water externalities from forests. In: *The provision of forest ecosystem services,* Thorsen et al. (Eds.). European Forest Institute, Joensuu, Finland, 21–25.

Zālītis, P. 2006. *Preconditions of forest management*. Riga, et cetera, 217 pp. (in Latvian).

Zālītis, P. 2012. *Forest and water*. Salaspils, Silava, 356 pp. (in Latvian).