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DOC and N_2O dynamics in upland and peatland forest soils after clear-cutting and soil preparation

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Abstract Forest clear-cutting followed by soil preparation means disturbance for soil microorganisms and disruption of N and C cycles. We measured fluxes of N_2O and dissolved organic carbon (DOC) in upland soil (podzol) and adjacent peat within a clear-cut forest catchment. Both soil types behaved in a similar way, showing net uptake of N_2O in the first year after the clear-cutting, and turning to net release in the second. The N_2O flux dynamics were similar to those of N content in logging residues, as reported from a nearby site. As organic matter is used in the food web of the decomposers, we attempted to explain the dynamics of N_2O uptake and release by measuring the concurrent dynamics of the low molecular weight (LMW) fraction and the aromaticity of DOC in a soil solution. The labile and most readily available LMW fractions of DOC were nearly absent in the year following the clear-cutting, but rose after two years. The more refractory high molecular weight (HMW) fraction of DOC decreased two years after the clear-cutting. The first year's net uptake of N_2O could be accounted for by the growth of decomposer biomass in the logging residues and detritus from the degenerating ground vegetation, resulting in immobilization of nitrogen. Simultaneously, the labile, LMW fraction of DOC became almost completely exhausted. The low availability of the LMW fraction could retard the growth and cause the accumulated decomposer biomass to collapse. During the following winter and summer the fraction of LMW clearly increased, followed by increased N_2O emissions. The presence of LMW DOC fractions, not the concentration of DOC, seems to be an important controller for N_2O liberation after a major disturbance such as clear-cutting and site preparation. The complex connection between DOC characteristics, nitrification or denitrification merits further studies.

Keywords Dissolved organic carbon · DOC · Nitrous oxide · N_2O · Forestry · Clear-cutting

Abbreviations

BOD	Biological oxygen demand
C	Carbon
CH_4	Methane
DOC	Dissolved organic carbon
N	Nitrogen
N_2	Nitrogen

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N_2O	Nitrous oxide
NH_4^+	Ammonium
NO_3^-	Nitrate
SAR	Specific absorption ratio
sUVa	Specific UV absorbancy

Introduction

Soil decomposer microorganisms face drastic changes in the substrate supply and soil conditions after a clear-cut and the associated soil surface manipulation. Increased rate of decomposition of soil organic matter in oxic conditions and the absence of ground vegetation may result in a release of extra nitrogen (N) in soil solution. Clear-cutting and soil preparation are found to increase nitrogen leaching from boreal peatland forest (Nieminen 1998) and upland forest sites (Jacks and Norrström 2004; Mannerkoski et al. 2005; Piirainen et al. 2007), but the duration of N loss seems variable in runoff. From peatland forests after clear-cutting, organic N originating from the disturbed peat and logging residues starts to leach immediately (Nieminen 1998), but only after the first growing season does the leaching take place in inorganic form as ammonium (NH_4^+) and nitrate (NO_3^-). In upland forest, the release of organic N continues for at least 5 years, while inorganic N leaches for only 1–2 years after clear-cutting (Piirainen et al. 2007). However, Kubin (1998) found that the concentration of NO_3^- rises for 3.5–5 years after clear-cutting in upland forests and may remain above the pre-cut level for as long as ten years. The maximum may be reached even later (Mannerkoski et al. 2005). By contrast, Nohrstedt et al. (1994) reported that neither clear-cutting in mineral soil nor N fertilization affects the N load on water or the atmosphere. Similarly, Piirainen et al. (2002) found negligible N leaching after clear-cutting in mineral soil.

Clear-cuttings also increase the concentrations of dissolved organic carbon (DOC) in a soil solution in both peatland and upland forest sites, where outflow concentrations of DOC increased for several years after clear-cutting (e.g., Nieminen 2004; Piirainen et al. 2007; Morris 2009). However, possible changes in the quality of DOC after clear-cutting are poorly known, although these characters of DOC may be significant for biogeochemical cycles. The linkages between DOC and gaseous N are also relatively little known.

Clear-cutting causes a collapse of ground vegetation biomass, which lowers the N binding potential and may cause N leaching from the litter (Palviainen et al. 2005). The source of initial N leaching in clear-cuts is probably not the logging residues of conifers, despite their high N content: extra N accumulates in the residues during the first year after clear-cutting, but the N content is lowered only in the second year (Palviainen et al. 2004). In the third year after cutting, the decomposing branches and pine roots again absorb N. However, logging residues have been observed (Kubin 1998) to release NO_3^-–N even years after the clear-cut. Soil disturbance by mixing increases the release of NO_3^- and keeps it high longer than does clear-cutting alone (Booth et al. 2006). In clear-cuts of peatland forests, leaching of organic N originates from deeper peat layers compared with leaching from mineral soils (Nieminen 1998).

The conditions for nitrification and denitrification in the soil profile can be observed through the fluxes of dinitrogen monoxide (nitrous oxide, N_2O). In these processes inorganic N is changed from ammonia to nitrate and further to NO, N_2O and N_2. The emissions of N_2O have been reported to increase after clear-cut and soil disturbance (Robertson et al. 1987; Booth et al. 2006; Neill et al. 2006). Increased N_2O emissions may be due to a release of the N_2O retained in the soil pore space through soil mixing, or due to N_2O formation through denitrification, which may be accelerated by the greater NO_3^- reserves and carbon availability in the disturbed soils (Booth et al. 2006). Under post-harvesting conditions, increased denitrification may have an impact on N losses similar to that of leaching (Robertson et al. 1987). However, Nieminen (1998) did not find any changes in N_2O fluxes after clear-cutting.

We measured N_2O fluxes from a clear-cut followed by soil mounding on both peat and mineral soil. We also measured the fluxes of CH_4 to illustrate the oxic topsoil dynamics. In the case of net consumption of CH_4, the conditions of the topsoil must be oxic, which might influence the emissions of N_2O (Wrage et al. 2004). In addition, the dynamics of biological oxygen demand (BOD) and the concentrations and characteristics of DOC (molecular size fractions and aromaticity) in vadose water retrieved from zero-tension soil lysimeters and ground water retrieved from piezometers were compared in order to find temporal changes in the availability of carbon (C) for the decomposers. We hypothesized that, after clear-cutting and soil preparation, the more labile DOC fractions would become available for the decomposers from logging residues and dying ground vegetation. These dynamics should be reflected in the soil-atmosphere N_2O exchange through nitrification and denitrification, if the inorganic N is not consumed by re-vegetation.

Materials and methods

Study site

A clear-cut was performed in January 2005 in a catchment of 5 ha consisting both of upland soil and peat in eastern Finland, Nurmes (63°39′26N, 29°29′34E, 204 m a.s.l., Fig. 1). After clear-cutting, the ditches were cleaned and the site prepared for regeneration in June 2005 using mounding (peat) and scarification (upland). The annual mean temperature is +3.0°C and the annual precipitation 533 mm in the Nurmes area. The upper part of the study site (about 3 ha) was on mineral soil (podsol) on the hill, and the lower parts, 2 ha, on peat soil below the slope, with an average peat depth of 1.1 ± 0.5 m. Boardwalks were constructed on both subsites in order to prevent soil disturbance during gas measurements.

Water in the soil is considered to be either vadose when it infiltrates through the ca. 1 m deep topsoil, especially within the peat layer, or ground water when running over the bedrock. In order to collect vadose or ground water samples, water wells were drilled in November 2005. The zero-tension soil lysimeters, used for capturing the vadose water extended to a depth of 0.91–1.1 m, and the piezometers used for the groundwater to 1.17–5.85 m, respectively. The lysimeters with an inner diameter of 52 mm differed from the piezometers as follows: The lysimeters consisted of a 1 m long sieve tube with a 0.3 mm sieve extending from the surface to a depth of one meter attached to a collector tube with a closed bottom. In the piezometers the sieve was located at the bottom of the tube, drilled into the bedrock. The vadose water accumulated in the collector tube between the samplings, while the samples taken from the piezometers represented the conditions at the sampling time. In mineral soil sites the average pH of the vadose water was 6.2, and in the peat sites 5.1, respectively. In peat, the ground water level was so high that the ground water and the vadose water were mixed. During summer 2005, before the deep piezometers were established, the water table could be measured no deeper than 60 cm below the soil surface, using perforated 32 mm diameter plastic tubes, allowing the water to be collected only in situations when the water table level was high.

Prior to clear-cutting, the forest was dominated by spruce (*Picea abies*) and pine (*Pinus sylvestris*),

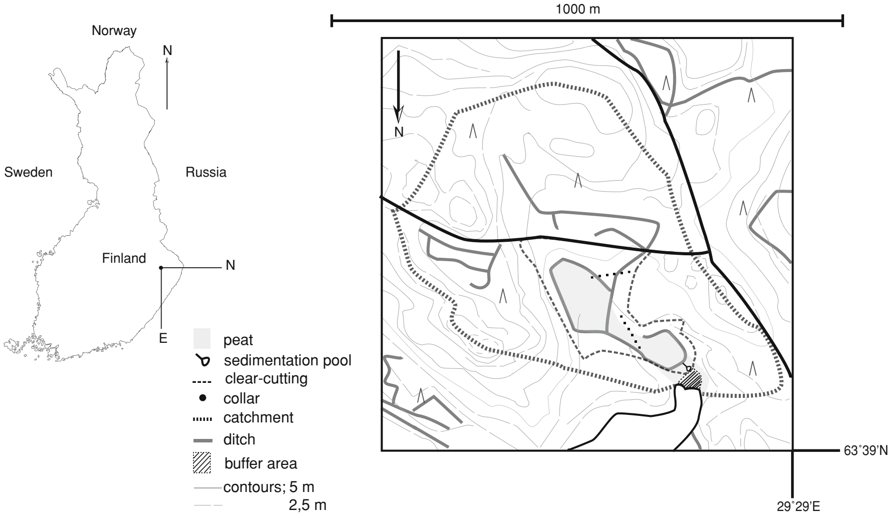


Fig. 1 Map of the study area

shrubs such as *Vaccinium myrtillus* and *Vaccinium vitis-idaea,* and mosses (e.g., *Pleurozium schreberi*, *Aulacomnium palustre*). After the clear-cut, the forest shrubs and mosses died, and during summers 2006 and 2007 the area was characterized by pioneer herbs *Epilobium angustifolium* and grasses *Deschampsia fluxuosa*, and in some spots by mosses *Polytrichum commune*, *P. juniperinum*, *P. strictum*, and *Pleurozium schreberi*. The vegetation coverage was estimated in 2006 and 2007 as a percentage of the projection of vascular plant species and the moss layer. The proportions of litter and dead plants were also estimated on seven 7 dm^2 plots in the mineral soil area and on five plots in the peat area, respectively. The complete vegetation data are not reported here, and only the essential changes are described.

N_2O and CH_4 measurements

The soil-atmosphere exchange of N_2O and CH_4 was measured from June 2005 to September 2007 along the boardwalks at permanently installed 12 steel collars (Ø 30 cm). The edge of the collar extended about 15 cm deep into the soil. Seven of the collars were located in mineral soil and five in peat. A chamber was locked onto the collar and sealed airtight with a soft rubber gasket attached to the edges of both the collar and the 30 cm high chamber. The vented, static chamber had a battery-operated fan mixing the 21.2 dm^3 head-space. Four 30 ml air samples were taken every 5 or 10 min from the top of the chamber through a thin plastic line into 50 ml polypropylene syringes equipped with three-way stop cocks. The headspace was closed altogether for 20 min in the summertime, but in the autumn and before the spring thaw, it was closed for 40 min. The temperature of the headspace was measured using a thermometer (multi-thermometer or digital-thermometer, Suomen lämpömittari Oy). The temperatures in the soil profile (soil surface, −2, −5, −10, −15, −20, −25, −30, −40 and −50 cm) were measured using a k-type thermocouple in a piercing probe and a Fluke 52 K/J Thermometer. The moisture of the surface soil layer (−5 cm) was also measured (moisture meter HH2, Delta-T Devices Ltd, detector ML2X, Delta-T Devices Ltd) in summers 2005 and 2006.

Gas concentrations were determined in the laboratory within two days by injecting a 2 ml sample into a gas chromatograph (Shimadzu GC-14-A) equipped with a 1 m packed column, a flame ionization detector (FID) and an electron capture detector (ECD). The accuracy of the detector was 30 ppb for CH_4 (FID) and 20 ppb for N_2O (ECD). The gas flux rates were calculated from the change in concentration in the combined airspace of the chamber and collar.

In wintertime, the gas fluxes were determined by sampling the gas concentrations below and above the snowpack (Sommerfeld et al. 1993; Alm et al. 1999). Air samples were taken into 50 ml polypropylene syringes using a metal pipe 4 mm in diameter equipped with three-way stop cocks. Ambient air samples were taken above the snow pack. Samples were also taken from different depths between the ice layers in the snow pack to check the linearity of the gas concentration gradient in the snow profile. The concentration of N_2O and CH_4 in the sample air was analysed in the laboratory using gas chromatography.

Volumetric snow samples from top to bottom of the snowpack were weighted, and the average porosity of the snow profile was calculated for each gas sampling session, assuming the snow to consist of ice (density 0.9168 g cm^{-3}) and air. The gas fluxes were then calculated using Fick's first law of diffusion through a porous medium

$$J_g = D_g(dC_g/dz)f, \tag{1}$$

where J_g is the diffusive gas (g) flux, D_g is the diffusion coefficient for the gas in air, dC_g/dz is the measured vertical concentration gradient through the snow pack (z cm), and f is the snow porosity. The values used for D_g were 0.139 $cm^2\ s^{-1}$ for N_2O and 0.22 for CH_4 (Sommerfeld et al. 1993).

Seasonal means of N_2O–N and CH_4 for the snow free period and winter and annual averages were calculated using daily means of the gas fluxes and the number of days in the respective period. The snow free period was estimated at 240 days and the winter period at 120 days in the Nurmes area.

N measurements

Water samples were taken from the sedimentation pool, which collected waters flowing from the clear-cut and from which the water flooded to a buffer zone. Samples for total N (N_{TOT}), NH_4^+, and NO_3^- analysis were taken 1–3 times per month, from May to September 2007. Values for pH (PHM 92 Radiometer) and conductivity (CDM 92 Radiometer,) were measured after filtering. Concentrations of N_{TOT}, NO_3^- and NH_4^+ were analyzed from filtered (Schleicher & Schuell GF 52, glass fibre, 1–1.2 µm, Ø 47 mm and Schleicher & Schuell membrane filter ME 25, 0.45 µm, Ø 47 mm) and deep-frozen water samples using the standards ISO 13395, SFS-EN ISO 10304 and ISO 1173, respectively. Of these, N_{TOT} and NH_4^+ were measured by spectrophotometry (FIA-Star 5000 Analyzer FOSS TECATOR) and $NO_2 + NO_3$ by ion chromatography (DIONEX 500).

Quantification and characterization of DOC

Dissolved organic carbon (DOC) is operationally defined as the fraction of organic carbon that passes through a 0.45 µm filter (Thurman 1985, p. 2; McDonald et al. 2004). The absolute content of DOC (mg l^{-1}) was measured by TOC 5000A (Total Organic Carbon Analyzer, Shimadzu) from filtered water samples taken at least once a month. The quality of DOC was characterized by measuring the specific UV and visible absorbency by UV-1601PC (UV–visible spectrophotometer, Shimadzu) in quartz cuvettes and by counting specific UV absorbency ($sUVa_{\lambda 254\ nm}$), specific visible absorbency ($sVISa_{\lambda 400\ nm}$) and specific absorption ratios (SAR_{UV} and SAR_{VIS}). The values of indexes were calculated as ratios: $sUVa_{\lambda 254\ nm} = A_{\lambda 254\ nm}/mg\ C\ l^{-1}$, $sVISa_{\lambda 400\ nm} = A_{\lambda 400\ nm}/mg\ C\ l^{-1}$, $SAR_{UV} = A_{\lambda 254\ nm}/A_{\lambda 400\ nm}$ and $SAR_{VIS} = A_{\lambda 400\ nm}/A_{\lambda 600\ nm}$. The sUVa and sVISa values can be considered to reflect the degree of aromaticity (Traina et al. 1990; Akkanen et al. 2004): If the index value is high, the DOC consists of more aromatic molecules than does DOC with a low value. SAR has been shown to correlate with the size of molecules in DOM (Peuravuori and Pihlaja 1997), a lower SAR indicating a higher molecular weight.

If the concentration of DOC in a water sample was over 80 mg l^{-1}, the samples were diluted 1:10 with 0.1 mM artificial surface water (1 l ionized water + 0.2 ml $MgSO_4 \times 7H_2O$-solution (24.65 g l^{-1} l ionized water) + 0.2 ml $CaCl_2 \times 2H_2O$-solution (58.80 g l^{-1} l) + KCl-solution (1.15 g l^{-1} l) +$NaHCO_3$-solution (12.95 g l^{-1} l)), and both the DOC concentration and absorbency were measured again. If the DOC concentration in the water sample was over 35 mg l^{-1}, the samples were diluted to DOC 35 mg l^{-1} for absorbency measurements in order to meet the analysis capacity of the spectrophotometer.

Size exclusion chromatography (HPLC:Agilent) was used to separate the four relative molecular size fractions of DOC with similar retention times in all the different water samples (Fig. 2). HPLC equipped with diode array detector (DAD:Agilent) and fluorescence detector (FLD:Agilent) was used. The columns were the LC guard column TSK-GEL F0474 (7.5 mm ID, 7.5 cm, 10 µm) and TSK-GEL G30000SW, F0188-16C G30000SW (7.5 ID, 30 cm, 10 µm), manufactured by Agilent. The wavelength used was 254 nm and the injected sample volumes were 5, 10, 20 or 30 µl, depending on the amount of DOC (>100, 51–100, 10–50 and <10 mg l^{-1}, respectively).

BOD

The water samples for biological oxygen demand (BOD) were taken from March to September in 2007, 1 to 3 times per month. BOD (7 days) was measured using Consort C932 with calvanic oxygen electrode SZ10T, otherwise following the SFS-EN 1899-2 standard, but the samples were incubated at 22°C instead of the standard 20°C.

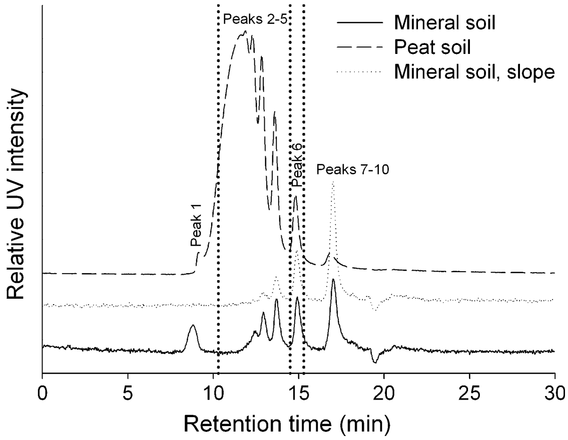


Fig. 2 Size fractions (*peaks 1–10*) of dissolved organic carbon (DOC) in water samples from mineral soil on the hill and on the intermediate slope and from the study site on peat below the hill

Statistical tests

Pearson correlations were computed in order to find links between soil profile temperature (surface, −5, −10, −15, −20, −25, −30, −40, −50), moisture, DOC, CH_4 and N_2O. A linear mixed model was used for testing the spatial and temporal differences in N_2O and CH_4 fluxes, water table level and the fractions of DOC (y) between different parts of the study site, and the differences in DOC molecular size fractions in the ground water retrieved from the piezometers and the vadose water retrieved from the lysimeters in both soil types. The model used was

$$y_{ijk} = m + \text{soiltype}_i + \text{year}_j + \text{soiltype} \times \text{year}_{ij} + e_{ijk}, \qquad (2)$$

where m is a constant, soil type (i = mineral soil or peat) and year are fixed effects, k refers to the kth measurement within a soil type × year combination, and e_{ijk} is the error term. It was assumed that the distribution of the error vectors $e_{ijk} = (e_{ij1}, e_{ij2}, \ldots, e_{ijk})$, where the measurements in the year j follow independent multivariate normal distributions with homogeneous variance and first order autoregressive covariance structure. The marginal distributions of the residuals were graphically checked for normality and homogeneity of the variance. The selection of the covariance structure was based on Akaike's information criteria.

Results

N_2O and CH_4

Average N_2O fluxes during the whole study period did not significantly differ between peat and mineral soil either during the snow free period or in winter (between December and April) (Table 1; Fig. 3), but they differed significantly between the years in summer time ($p = 0.025$). Especially, the second summer after the clear-cutting, 2006, differed from the third ($p = 0.028$). In the first snow free period after the clear-cutting, N_2O emissions were on average 0.036 mg m^{-2} day^{-1}. The fluxes did not correlate significantly with soil temperature. N_2O was net consumed during the second snow free period in 2006 at an average rate of −0.316 mg m^{-2} day^{-1}

Table 1 N_2O and CH_4 fluxes with ± SE (in parenthesis) in summer and winter from mineral soil and peat subsites in the clear-cut

Snow free period (and year) after the clear-cut	Soil type	N_2O–N (μg m^{-2} day^{-1}) (±SE; n)		CH_4 (μg m^{-2} day^{-1}) (±SE; n)	
		Summer	Winter	Summer	Winter
1. (2005)	Mineral soil	57	11	-1416^{c}	−781
		(±49; 107)	(±24; 72)	(±593; 96)	(±62; 72)
	Peat	−31	13	132^{c}	−261
		(±101; 75)	(±39; 38)	(±737; 67)	(±93; 38)
2. (2006)	Mineral soil	-284^{a}	25	-1795^{d}	−655
		(±176; 93)	(±10; 60)	(±267; 102)	(±65; 60)
	Peat	-85^{b}	60	-409^{d}	−172
		(±196; 66)	(±6; 40)	(±134; 77)	(±48; 40)
3. (2007)	Mineral soil	240^{a}	21	-1123^{e}	944
		(±335; 76)	(±75; 63)	(±184; 92)	(±826; 71)
	Peat	307^{b}	−64	301^{e}	4553
		(±412; 59)	(±68; 44)	(±230; 67)	(±2750; 48)
Total	Mineral soil	−8	18	−1456	−140
		(±124; 276)	(±21; 195)	(±228; 290)	(±232; 203)
	Peat	51	1	−011	1601
		(±144; 200)	(±28; 122)	(±275; 211)	(±730; 126)

The significantly different means are indicated by the letters a–e

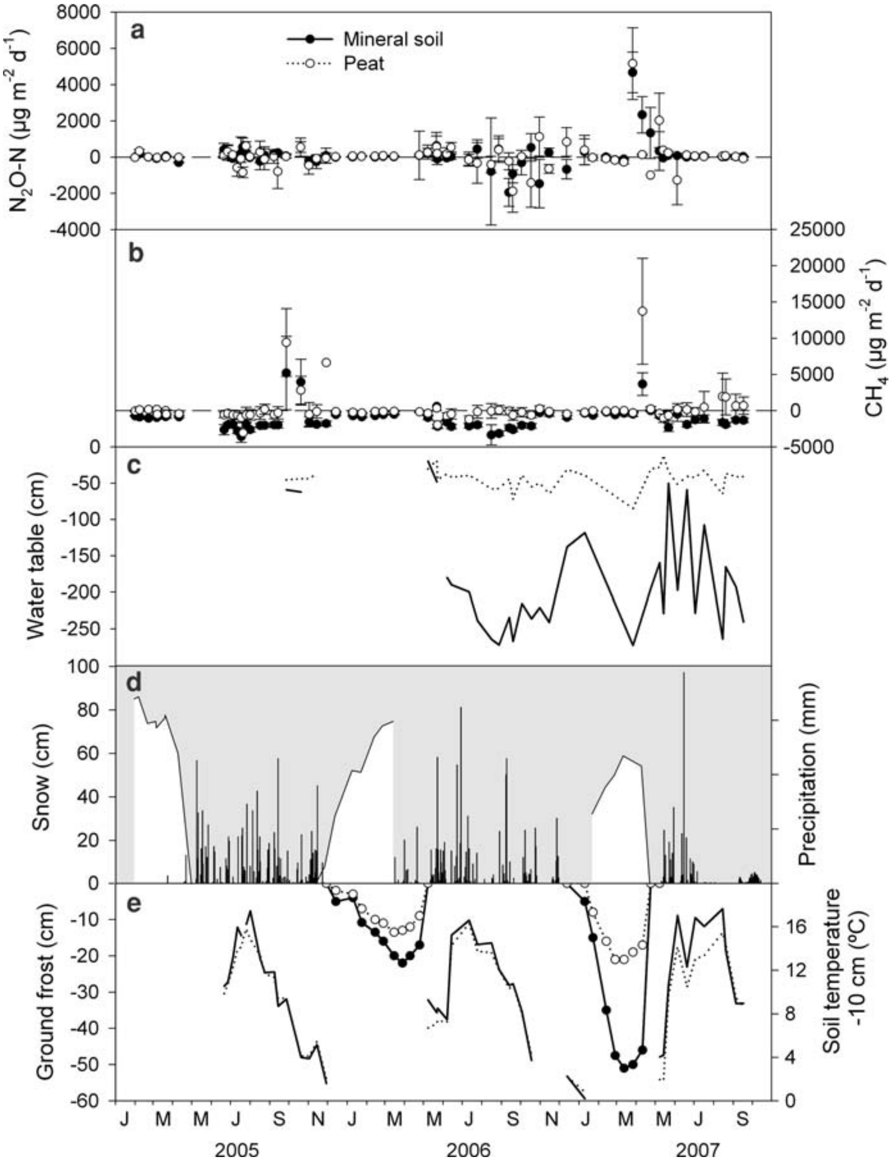


Fig. 3 The dynamics of **a** daily fluxes of nitrous oxide (N_2O), **b** methane (CH_4) with SE from the mineral soil and peat in the clear-cut area, **c** water table levels (cm), **d** daily precipitation (mm) and depth of snow cover (cm), **e** ground frost (cm) and soil temperatures (°C) from a depth of −10 cm in frost-free periods in the clear-cut area

and net emitted in the next year at a rate of 0.423 mg m^{-2} day^{-1}. The highest N_2O emissions were measured during the spring thaw (Fig. 3), when the ground frost was melting and the soil surface was water-saturated.

CH_4 flux rates differed significantly between mineral soil and peat during summers ($p = 0.003$), but not during winters (Table 1; Fig. 3). The consumption of CH_4 was higher at the clear-cut mineral soil site, but within the two soil types, the fluxes did not differ significantly between years. Further, CH_4 fluxes did not correlate with water table level, showing a positive correlation only with the temperature at soil surface ($r = 0.797$, $p = 0.032$). Similarly to N_2O, the highest CH_4 emissions were measured during the spring thaw, when ground frost was melting on the water-saturated soil surface (Fig. 3).

Water table and N leaching

Water table levels in mineral soil and peat did not differ significantly in 2006 and 2007 (Table 2). There was no correlation with N_2O fluxes and water table or

Table 2 Average depth of water table and pH with ranges in different parts of the clear-cut area during 3 years

Growth period (with year) after the clear-cutting	Soil type	Water table (cm)	Min-max (cm); *n*	pH	Min-max; *n*
1. (2005)	Mineral			6.01	5.95–6.48; 7
	Peat			5.35	4.23–6.44; 18
2. (2006)	Mineral	217	396–15; 94	5.89	5.13–7.42; 85
	Peat	47	142–3; 74	5.30	4.11–6.36; 62
3. (2007)	Mineral	188	455–13; 83	5.86	5.08–6.41; 61
	Peat	42	146–8; 64	5.21	4.14–6.39; 41

soil moisture near the soil surface (−5 cm) during the unfrozen season. In the third summer after clear-cutting, the only year when dissolved N was measured at our site, the concentration of total N in the water outlet increased toward the autumn. Of the increase, most was in the form of organic N, while the concentration of inorganic N decreased (Fig. 4).

Total DOC, its molecular size and aromaticity, and BOD

DOC concentrations in vadose water (lysimeters) were higher in peat than those in the mineral soil (Fig. 5). The occurrence of neither rain nor drought seemed to affect the DOC concentrations (Fig. 3). On the other hand, the molecular weight of DOC was lowest after a dry spell or in winter, as indicated by the highest values of SAR (Fig. 5). The aromaticity of DOC, measured by sUVa, was higher in peat than in mineral soil (Fig. 5), where the molecular weight also varied more over the seasons. The combined fractions 2–5 represented the highest absolute amount of DOC in the vadose water samples (Fig. 6). The content of DOC in both fractions 1 and 2–5 in vadose water were higher in the second (2006) than in the third year (2007) at both peat and mineral soil sites (Table 3). At the same time, the fractions 7–10, however, were raised only in peat. Distinctively, in the ground water (piezometers), only the fractions 2–5 were higher in 2006 than in 2007 in mineral soil, but the differences were not statistically significant. The only significant difference was in fraction six ($p = 0.030$), where the concentration of DOC was higher in ground water in 2007 when compared with that measured in 2006. According to the BOD–DOC correlations for the samples from “Ditch”, “Margin ditch” and “Sedimentation pool” (Figs. 1, 7), the most degradable DOC appeared to be in fractions 2–5, 6 and 7–10, while fraction 1 was the least degradable in all environments (Fig. 8).

Vegetation cover

There was no live vegetation left in the first summer after the clear-cut, since the ground vegetation and mosses died when the soil was prepared, turned and cleared of canopy shade. Vegetation cover was still negligible in the second growth season after the operation (Table 4), but the coverage of mosses and ground vegetation increased vigorously during the third growth season 2007.

Discussion

Fluxes of N_2O after harvesting, ditch cleaning and soil preparation were very low in 2005 during the snow free period when mounding and scarification were carried out. This was in line with other observations from peatland forests showing that liberation and leaching of inorganic N (NH_4^+, NO_3^-) only starts one year after the clear-cut (Nieminen 1998; Piirainen et al. 2007). At both site types the low net emissions measured in 2005 turned into a clear net N_2O consumption of −0.47 and −0.04 kg N ha^{-1} a^{-1} in 2006 in mineral soil and peat, respectively, and turned again into a net N_2O emission of over 0.4 kg N ha^{-1} a^{-1} in 2007. Our results thus show N_2O dynamics parallel to those reported by Palviainen et al. (2005) in litter N contents under similar conditions: the nitrogen content of the pine and spruce logging residues increase in the first year after the logging (+10.3 kg N ha^{-1}), and start to decrease in the

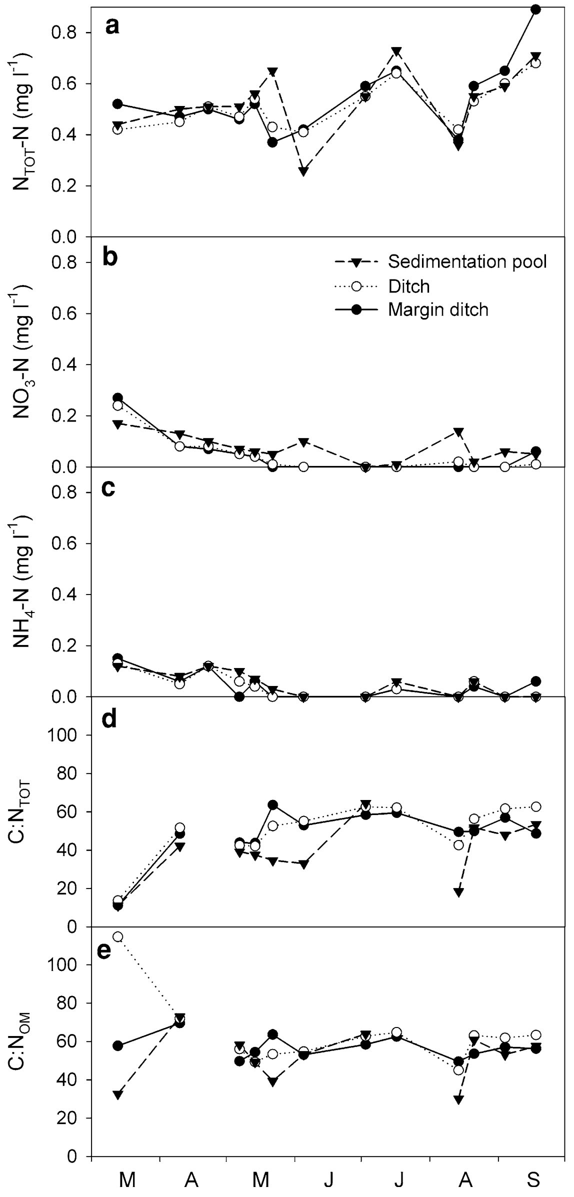


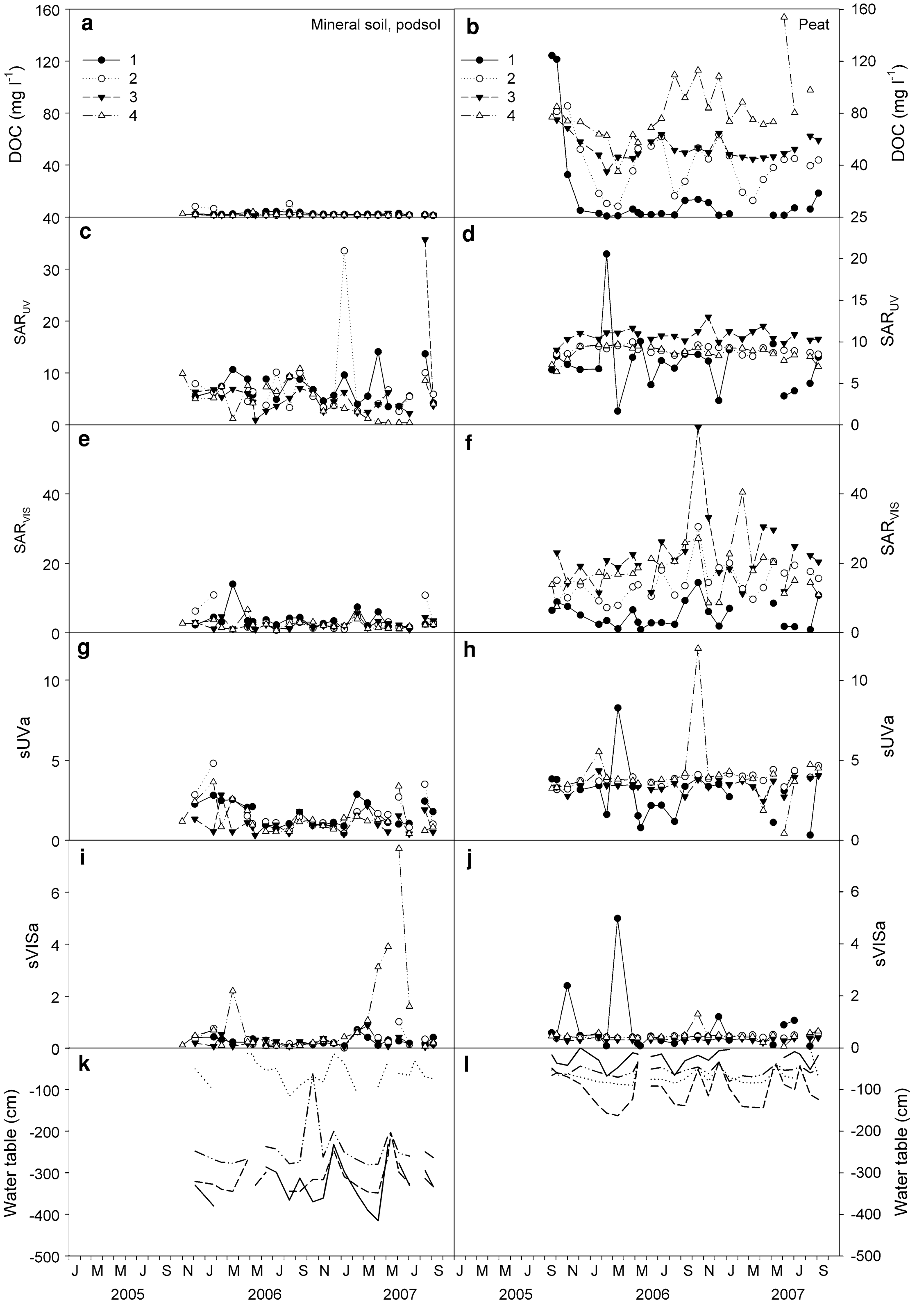
Fig. 4 Concentrations of **a** total N (N_{TOT}), **b** nitrate (NO_3–N), and **c** ammonium (NH_4–N) from the water outlet of the clear-cut area in summer 2007. The C:N ratios were calculated using DOC both **d** N_{TOT} and **e** organic N (N_{OM})

second year (−20.4 kg N ha^{-1}). Our data on N_2O dynamics could be explained by changes in the decomposition of the logging residues, when the rate of decomposition increases after clear-cutting, consuming inorganic and organic N liberated from the slash.

The dynamics of the consumption and release of N_2O depend on soil conditions and the state of microbial populations in a complex way and combine the scopes of numerous studies. Consumption of N_2O can take place through denitrifiers, but probably also through nitrifiers (Robertson and Tiedje 1987; Chapuis-Lardy et al. 2007). One explanation for the N_2O net consumption may be the latter process, since conditions of low water level would favor nitrification and sustain mainly CH_4 oxidation (Fig. 3) 1 year after clear-cutting and soil preparation. Denitrification could have been feasible only during the freezing and thawing times and rainy days. Denitrification is found mainly under anoxic conditions (see e.g., Klemedtsson et al. 2005), which probably occurred during the spring thaw in the lower part of our peaty clear-cut (Fig. 3). Observations on net consumption of N_2O and its possible controls hint that different nitrification and denitrification processes are involved (Chapuis-Lardy et al. 2007). Of the various types of nitrification and denitrification as reviewed by Anderson et al. (1993); Rudebeck (2000); Zumft (1997); Schimel et al. (1984); Paul and Clark (1996); Islam et al. (2007), autotrophic nitrification by bacteria is thought to be a more significant source of N_2O than heterotrophic nitrification by bacteria and fungi, but in aerated soils, similar to those of our clear-cut, heterotrophic nitrification may be an important source of N_2O (Payne 1981, p. 27; Anderson et al. 1993). N_2O may also be released through the pathway of nitrifier denitrification, which can, however, continue the pathway from N_2O to N_2, potentially leading to net N_2O consumption (Wrage et al. 2001).

Before clear-cutting and ditch cleaning, the amount of available organic C was probably lower at the forest site than after clear-cutting and soil preparation (see e.g., Nieminen 2004). Soil mounding together with ditch cleaning lowered the water table, thus drying the peat surface layers. The observed very low emissions or net consumption of CH_4 (Fig. 3) indicated that at least some oxygen was present in the soil surface layers. The measured net consumption of N_2O from the atmosphere means that an increased biomass of decomposers might be present in the soil, evidently fed by logging residues and litter from the destroyed ground vegetation.

According to Saetre and Stark (2005), high N release may be associated with the degradation of microbial substrates of high N concentration. At our study site, the decomposition of the microbial


a
Mineral soil, podsol
b
Peat
1
2
3
4
DOC (mg l^{-1})
c
d
SAR$_{UV}$
e
f
SAR$_{VIS}$
g
h
sUVa
i
j
sVISa
k
l
Water table (cm)
J M M J S N J M M J S N J M M J S
2005
2006
2007

◀ **Fig. 5** Sample concentrations of DOC and its characterization at four different water sampling points in mineral soil and peat, respectively (for explanations to abbreviations, see text)

biomass formed during the first year after the clear-cutting would serve as a source of N during the second and third years. Soil mixing can also reduce the microbial activity and lower the rate of N mineralization at least in mineral soils (Paul et al. 2001). This could mean marked N immobilization and an increased N_2O uptake from the atmosphere due to lack of inorganic N for nitrification during the first year after the clear-cutting.

In the course of decomposition, the C:N ratio of the residues probably changed to become more favorable for nitrification. Peatland drainage and subsequent decomposition reduces the C:N ratio over time and potentially increases N_2O emissions (Klemedtsson et al. 2005). Emissions of N_2O occur mainly from sites with a soil C:N ratio lower than 25 (Klemedtsson et al. 2005; Ernfors et al. 2007). According to Palviainen et al. (2004) the C:N ratio of foliage decreased to below 25 during the second year after the clear-cutting in the area close to our site. Because the nutrients and C are directly available for microbes in water, and the dynamics are more visible than in bulk peat, we calculated the $C{:}N_{TOT}$ ratio from a soil solution using DOC and N_{TOT}. Furthermore, the ratio ($C{:}N_{OM}$) was also calculated in order to view the proportion of organically bound N in the total N. The resulting dynamics of $C{:}N_{TOT}$ or $C{:}N_{OM}$ in the sedimentation pool and ditches showed low enough values only in March (Fig. 4). The same conditions were also observed for those of high N_2O emissions during the frost melting period (Fig. 3). On the other hand, during the snow free period the concentration of inorganic N decreased and the N_2O fluxes became relatively low. It seems that except during the snowmelt, the $C{:}N_{OM}$ ratio remains nearly constant despite the evident changes in N_{OM} concentration in the water (Fig. 4). Hence the conditions of low $C{:}N_{TOT}$ and $C{:}N_{OM}$ appear to be favorable for N_2O production only during the thaw period.

Carbon is lost from logging residues at a rate faster than that of N loss (Palviainen et al. 2004) and is mainly released into the atmosphere as CO_2 (Piirainen et al. 2002). As indicated by our BOD results on O_2 consumption activity, LMW DOC is more easily decomposable than the HMW DOC fraction. Rather high amounts of the labile DOC from fractions 2–5 and 6 were measured in the water outlet (Fig. 7) in spring 1 year after clear-cutting (2006). The proportions of labile fractions were low during summer 2006, but again in 2007 the sizes of fractions 6 and 7–10 were high in the sedimentation pool and ditches and at mineral soil and peat sites (Fig. 6), containing labile compounds according to the BOD response (Fig. 8). This shows that in the first spring following the clear-cutting and soil preparation, the DOC available for the microorganisms in a soil solution is high, but is exhausted during the summer, when the

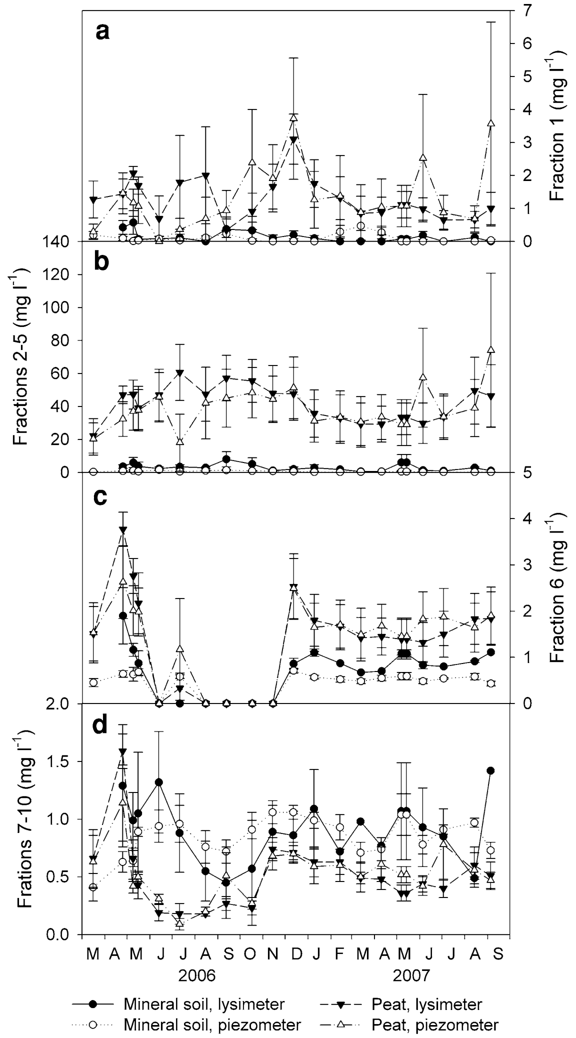


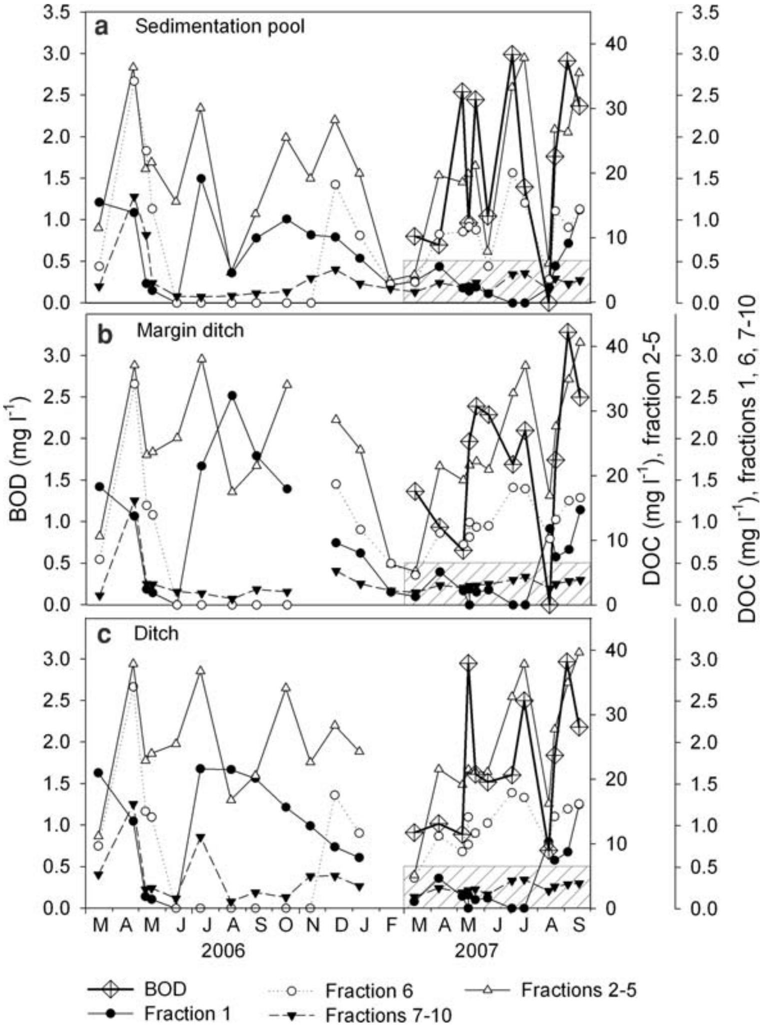
Fig. 6 The fractions of dissolved organic carbon (DOC) at mineral soil and peat sites, from lysimeters and piezometers within the clear-cut

Table 3 Average concentrations of separate DOC fractions (1, 2–5, 6, and 7–10) in vadose water (lysimeters) and ground water (piezometers), with SE and number of samples (*n*)

Year	Soil type	Water sample	DOC fractions (mg l^{-1}) ± SE (*n*)			
			1	2–5	6	7–10
2006	Mineral	Vadose	0.26 ± 0.06 (35)	3.98 ± 0.64 (35)	0.62 ± 0.21 (35)	0.94 ± 0.09 (35)
		Ground	0.07 ± 0.002 (86)	0.86 ± 0.11 (86)	0.33 ± 0.10 (86)	0.82 ± 0.06 (86)
	Peat	Vadose	1.56 ± 0.23 (44)	46.34 ± 3.08 (44)	1.22 ± 0.42 (44)	0.54 ± 0.13 (44)
		Ground	1.27 ± 0.32 (44)	38.36 ± 3.29 (44)	1.07 ± 0.33 (44)	0.50 ± 0.09 (44)
2007	Mineral	Vadose	0.08 ± 0.02 (19)	3.08 ± 0.66 (19)	0.96 ± 0.05 (19)	0.96 ± 0.08 (19)
		Ground	0.10 ± 0.05 (68)	0.31 ± 0.03 (68)	0.54 ± 0.02 (68)	0.89 ± 0.04 (68)
	Peat	Vadose	1.03 ± 0.10 (50)	35.34 ± 2.23 (50)	1.56 ± 0.06 (50)	0.49 ± 0.03 (50)
		Ground	1.43 ± 0.29 (46)	39.14 ± 4.69 (46)	1.66 ± 0.05 (46)	0.56 ± 0.03 (46)

See text and Fig. 2 for determination of the fractions from the absorption spectra. The samples were taken at mineral soil and peat subsites in 2006 and 2007

Fig. 7 Fractions 1, 2–5, 6, and 7–10 of dissolved organic carbon, DOC (mg l^{-1}) and biological oxygen demand, BOD (mg l^{-1}) in the water outlet of the clear-cut area collected from **a** the sedimentation pool, **b** in the margin ditch flowing into the sedimentation pool, and **c** the ditch flowing into the margin ditch during 2 years after clear-cutting. The *shaded band* indicates the detection limit for BOD



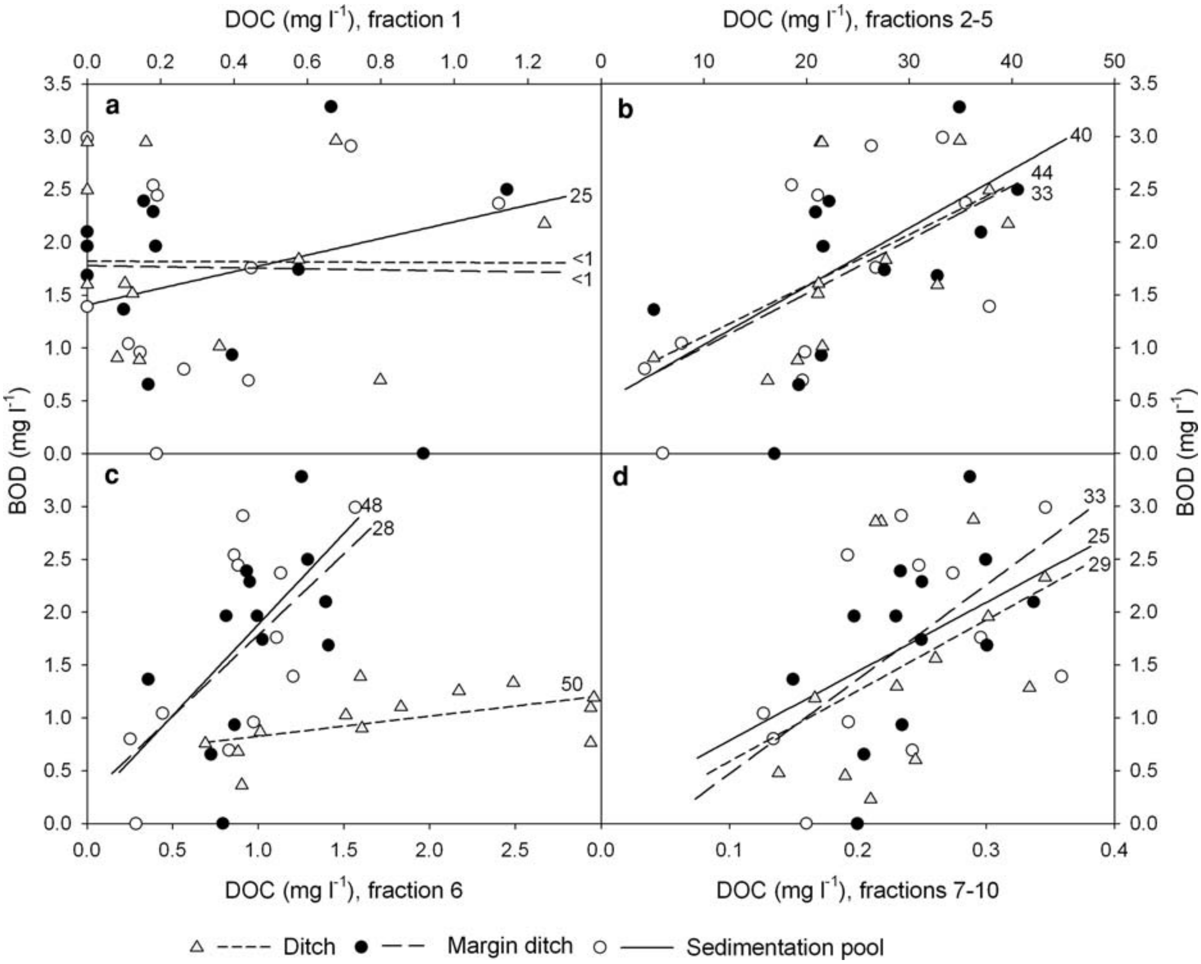


Fig. 8 Biological oxygen demand, BOD (mg l^{-1}) versus concentration of molecular size fractions of dissolved organic carbon, DOC (mg l^{-1}). The symbols denote the observed data. Regression lines, with coefficient of determination R^2 (%), are shown in order to illustrate the character of dependency of BOD at different locations along the water flow path of the concentration in the particular DOC fraction. Labels **a**–**d** denote the different fractions of DOC

Table 4 Total vegetation cover (%) with ± SE at the mineral soil and peat subsites in 2006 and 2007

Soil type		Year ± SE	
		2006	2007
Mineral soil	Ground vegetation	12.7 ± 7.0	26.5 ± 12.4
	Mosses	4.3 ± 1.5	21.5 ± 6.6
Peat	Ground vegetation	8.7 ± 4.5	28.4 ± 8.1
	Mosses	14.7 ± 7.0	47.6 ± 12.1

easily mobilized C is consumed. Again during the next winter, the availability of DOC increases, remaining high during the next summer. The reason for this supply of available DOC may lie in the more slowly decomposable litter and logging residues that were broken up later (compare e.g., Palviainen et al. 2004). Some labile fractions of DOC were present during both years, even at the mineral soil sites (Fig. 6), which could mean that the availability of C did not constrain the biomass of the microorganisms there. The relationship between the labile fraction six and N_2O is far from simple, however. There appears to be a positive correlation of N_2O with increasing DOC6 (Fig. 9) when the freeze-thaw period data are excluded, but more data are needed to verify this relationship.

The results on aromaticity were contradictory to our expectations in that the degree of aromaticity did not covariate with net N_2O consumption and

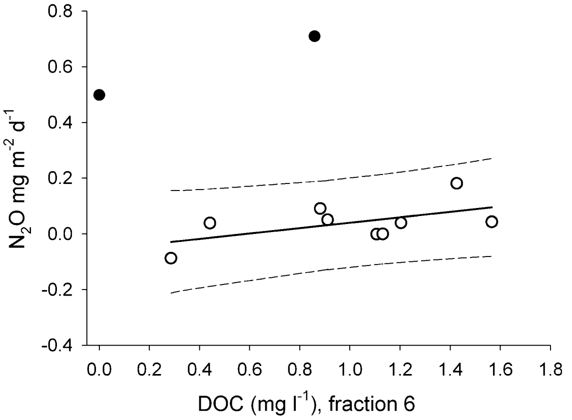


Fig. 9 Linear relationship of N_2O and DOC fraction 6 with 95% confidence limits in the sedimentation pool. The freeze-thaw period data (*closed symbols*) that significantly fall outside those of the rest of the population are excluded

emission. The aromatic compounds in DOC are considered relatively stable chemically, which is why they should be slowly mineralized and older than the less aromatic ones (Kalbitz et al. 2003a, b). At our sites, aromaticity was overall higher in peat (Fig. 5), implying that the quality of DOC should be more decomposable and available for microorganisms in mineral soil than in peat. However, the rate of N_2O uptake in mineral soil was higher than that in peat, and the net N_2O emissions were lower from upland than those from peat in the second year. The significance of less degradable aromatic compounds in our study may be negligible, since the amount of C is much higher in peat than in mineral soil with a thin organic horizon. Peat also contains more organic nitrogen relative to mineral soils, and thereby the effect of aromaticity is masked in mineralization.

Emissions of N_2O increased in the second year although the recovered ground vegetation cover consumed more nutrients, indicating an adequate supply of N. Plant cover and probably also NH_4^+ uptake increased only during the third growth period (Table 4), 2 years after the clear-cutting and soil preparation. According to Booth et al. (2006), gross nitrification may not increase after soil preparation, because heterotrophic NH_4^+ assimilation competes with the nitrifiers for NH_4^+. Since organic and inorganic N leached out from the clear-cut area (Fig. 4), there must have been an excess of NH_4^+ and/or NO_3^- available. This would mean that the inorganic uptake of N in vegetation did not limit the production of N_2O in the second year.

Conclusions

The fluxes of N_2O change in consecutive summers after clear-cutting from net consumption to net production is similar in both mineral soil and peat. We do not fully know the reasons for the uptake of N_2O during the first year after clear-cutting or for the second year N_2O emissions. The quality of DOC and especially the dynamics of labile C compounds seem important to N_2O production and consumption, and this requires further investigation. Labile C in a soil solution may affect the biomass of decomposers and thus connect the dynamics of N in logging residues with N_2O fluxes.

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References

Akkanen J, Vogt RD, Kukkonen JVK (2004) Essential characteristics of natural dissolved organic matter affecting the sorption of hydrophobic organic contaminants. Aquat Sci 66(2):171–177. doi:10.1007/s00027-004-0705-x

Alm J, Saarnio S, Nykänen H et al (1999) Winter CO_2, CH_4 and N_2O fluxes on some natural and drained boreal peatlands. Biogeochemistry 44(2):163–186

Anderson IC, Poth M, Homstead J et al (1993) A comparison of NO and N_2O production by the autotrophic nitrifier *Nitrosomonas europaea* and the heterotrophic nitrifier *Alcaligenes faecalis*. Appl Environ Microbiol 59(11):3525–3533

Booth MS, Stark JM, Hart SC (2006) Soil-mixing effects on inorganic nitrogen production and consumption in forest and shrubland soils. Plant Soil 289:5–15. doi:10.1007/s11104-006-9083-6

Chapuis-Lardy L, Wrage N, Metay A et al (2007) Soils, a sink for N_2O? A review. Glob Change Biol 13(1):1–17. doi:10.1111/j.1365-2486.2006.01280.x

Ernfors M, von Arnold K, Stendahl J et al (2007) Nitrous oxide emissions from drained organic forest soils—an up-scaling based on C:N ratios. Biogeochemistry 84(2):219–231. doi:10.1007/s10533-007-9123-1

Islam A, Chen D, White RE (2007) Heterotrophic and autotrophic nitrification in two acid pasture soils. Soil Biol Biochem 39:972–975. doi:10.1016/j.soilbio.2006.11.003

Jacks G, Norrström A-C (2004) Hydrochemistry and hydrology of forest riparian wetlands. For Ecol Manage 196:187–197. doi:10.1016/j.foreco.2004.01.055

Kalbitz K, Schmerwitz J, Schwesig D et al (2003a) Biodegradation of soil-derived dissolved organic matter as related to its properties. Geoderma 113:273–291. doi:10.1016/S0016-7061(02)00365-8

Kalbitz K, Schwesig D, Schmerwitz J et al (2003b) Changes in properties of soil-derived dissolved organic matter induced by biodegradation. Soil Biol Biochem 35:1129–1142. doi:10.1016/S0038-0717(03)00165-2

Klemedtsson L, von Arnold K, Weslien P et al (2005) Soil CN ratio as a scalar parameter to predict nitrous oxide emissions. Glob Change Biol 11(7):1142–1147. doi:10.1111/j.1365-2486.2005.00973.x

Kubin E (1998) Leaching of nitrate nitrogen into the groundwater after clear felling and site preparation. Boreal Environ Res 3:3–8

Mannerkoski H, Finér L, Piirainen S et al (2005) Effect of clear-cutting and site preparation on the level and quality of groundwater in some headwater catchments in eastern Finland. For Ecol Manage 220:107–117. doi:10.1016/j.foreco.2005.08.008

McDonald S, Bishop AG, Prenzler PD et al (2004) Analytical chemistry of freshwater humic substances. Anal Chim Acta 527(2):105–124. doi:10.1016/j.aca.2004.10.011

Morris DM (2009) Changes in DOC and DON fluxes in response to harvest intensity of black-spruce dominated forest ecosystems in northwestern Ontario. Can J Soil Sci 89(1):67–79

Neill C, Piccolo MC, Cerri CC et al (2006) Soil solution nitrogen losses during clearing of lowland Amazon forest for pasture. Plant Soil 281:233–245. doi:10.1007/s11104-005-4435-1

Nieminen M (1998) Changes in nitrogen cycling following the clearcutting of drained peatland forests in southern Finland. Boreal Environ Res 3:9–21

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 Fenn 32:123–132

Nohrstedt HÖ, Ring E, Klemedtsson L et al (1994) Nitrogen losses and soil water acidity after clear-felling of fertilized experimental plots in a Pinus sylvestris stand. For Ecol Manage 66:69–86. doi:10.1016/0378-1127(94)90149-X

Palviainen M, Finér L, Kurka AM (2004) Decomposition and nutrient release from logging residues after clear-cutting of mixed boreal forest. Plant Soil 263:53–67. doi:10.1023/B:PLSO.0000047718.34805.fb

Palviainen M, Finér L, Mannerkoski H et al (2005) Changes in the above- and below-ground biomass and nutrient pools of ground vegetation after clear-cutting of a mixed boreal forest. Plant Soil 275:157–167. doi:10.1007/s11104-005-1256-1

Paul EA, Clark FE (1996) Soil microbiology and biochemistry. Academic Press, San Diego

Paul K, Black S, Conyers M (2001) Development of nitrogen mineralization gradients through surface soil depth and their influence on surface soil pH. Plant Soil 234:239–246. doi:10.1023/A:1017904613797

Payne WJ (1981) Denitrification. Wiley, New York

Peuravuori J, Pihlaja K (1997) Molecular size distribution and spectroscopic properties of aquatic humic substances. Anal Chim Acta 337:133–149. doi:10.1016/S0003-2670(96)00412-6

Piirainen S, Finér L, Mannerkoski H et al (2002) Effects of forest clear-cutting on the carbon and nitrogen fluxes through podzolic soil horizons. Plant Soil 239:301–311. doi:10.1023/A:1015031718162

Piirainen S, Finér L, Mannerkoski H et al (2007) Carbon, nitrogen and phosphorus leaching after site preparation at a boreal forest clear-cut area. For Ecol Manage 243:10–18. doi:10.1016/j.foreco.2007.01.053

Robertson GP, Tiedje JM (1987) Nitrous oxide sources in aerobic soils: nitrification, denitrification and other biological processes. Soil Biol Biochem 19(2):187–193. doi:10.1016/0038-0717(87)90080-0

Robertson GP, Vitousek PM, Matson PA et al (1987) Denitrification in a clearcut Loblolly pine (*Pinus taeda* L.) plantation in the southeastern US. Plant Soil 97:119–129. doi:10.1007/BF02149830

Rudebeck A (2000) Factors affecting nitrification in European forest soils. Acta Universitatis Agriculturae Sueciae, Silvestria 145. Dissertation, Swedish University of Agricultural Sciences

Saetre P, Stark JM (2005) Microbial dynamics and carbon and nitrogen cycling following re-wetting of soils beneath two semi-arid plant species. Oecologia 142(2):247–260. doi:10.1007/s00442-004-1718-9

Schimel JP, Firestone MK, Killham KS (1984) Identification of heterotrophic nitrification in a Sierran forest soil. Appl Environ Microbiol 48(4):802–806

Sommerfeld RA, Mosier AR, Musselman RC (1993) CO_2, CH_4 and N_2O flux through a Wyoming snowpack and implications for global budgets. Nature 361(6408):140–142. doi:10.1038/361140a0

Thurman EM (1985) Organic geochemistry of natural waters. Martinus Nijhoff/Dr W. Junk Publishers, Netherlands

Traina SJ, Novak J, Smeck NE (1990) An ultraviolet absorbance method of estimating the percent aromatic carbon content of humic acids. J Environ Qual 19:151–153

Wrage N, Velthof GL, van Beusichem et al (2001) Role of nitrifier denitrification in the production of nitrous oxide. Soil Biol Biochem 33(12–13):1723–1732. doi:10.1016/S0038-0717(01)00096-7

Wrage N, Velthof GL, Laanbroek HJ et al (2004) Nitrous oxide production in grassland soils: assessing the contribution of nitrifier denitrification. Soil Biol Biochem 36:229–236. doi:10.1016/j.soilbio.2003.09.009

Zumft WG (1997) Cell biology and molecular basis of denitrification. Microbiol Mol Biol Rev 61(4):533–616