

## **Chapter 4**

### **Short-term effects of forest clearfelling on soil N<sub>2</sub>O and CH<sub>4</sub> fluxes**

#### **4.1 Introduction**

##### **4.1.1. The contribution of forests to nitrous oxide (N<sub>2</sub>O) flux**

Nitrous oxide (N<sub>2</sub>O) is a powerful greenhouse gas, which destroys ozone in the stratosphere. The concentration of N<sub>2</sub>O in the atmosphere has increased from about 0.275 μmol mol<sup>-1</sup> in pre-industrial times to about 0.320 μmol mol<sup>-1</sup> at present times (Smith and Conen, 2004). The N<sub>2</sub>O concentration is increasing at a rate of 0.3% y<sup>-1</sup> (Schlesinger, 1997) and its current mean concentration indicates a global pool of 2400 Tg N<sub>2</sub>O (1500 Tg N) in the atmosphere (Schlesinger, 1997). Each molecule of N<sub>2</sub>O has the potential to contribute about 300 times as much to the greenhouse effect as each molecule of CO<sub>2</sub> (Houghton *et al.*, 1997), so the current increase in the atmosphere may contribute to climate change.

Soils contribute about 57% (9 Tg) of the total annual emissions of N<sub>2</sub>O (Houghton *et al.*, 1997). Although it is known that tropical soils are the most important natural source of N<sub>2</sub>O, there is an incomplete understanding of the magnitude and spatial pattern of N<sub>2</sub>O emissions across the globe. Conversion of forests to pasture and agricultural land in general results in large emissions of soil N<sub>2</sub>O (Mosier *et al.*, 1991; Keller *et al.*, 1993), although the emissions decrease in comparison with the emissions from forest soils, in older pastures (Verchot *et al.*, 1999; Melillo *et al.*, 2001). It is known that forest soils also exhibit large spatial variability in soil N<sub>2</sub>O emissions. Sitaula and Bakken (1993) estimated a coefficient of variation of 152% in a Norway spruce stand in Norway. Bouwman *et al.* (1995) reported a mean 0.2 kg N

$\text{ha}^{-1} \text{y}^{-1}$  for temperate forests, while a later study by Brumme *et al.* (1999) gives a mean  $0.39 \pm 0.27 \text{ kg N ha}^{-1} \text{y}^{-1}$ , although the rates range from 0.010 to  $7 \text{ kg N ha}^{-1} \text{y}^{-1}$  (Bowden *et al.*, 1990; Brumme and Beese, 1992; Papen and Butterbach-Bahl, 1999).

The processes leading to  $\text{N}_2\text{O}$  production have been reviewed in Chapter 1 and will not be repeated here.

The rate of  $\text{N}_2\text{O}$  production and emission depends primarily on the availability of mineral N in the soil (Bouwman, 1990). Thus the application of mineral N fertilisers in agricultural systems increases  $\text{N}_2\text{O}$  emissions greatly, provided that there is a population of active nitrifying or denitrifying microorganisms. Freeze-thaw cycles also stimulate mineral N production and thereafter  $\text{N}_2\text{O}$  production, due to the stimulation of microbial activity as a result of additional available C and N released from microbial cells killed by the freezing (Christensen and Tiedje, 1990; Röver *et al.*, 1998). In regions with distinct dry and wet season, rainfall can have a similar effect on  $\text{N}_2\text{O}$  emissions with the re-wetting of the soil reactivating the water-stressed bacteria, which then metabolise the pool of inorganic N that has accumulated in the dry soil (Davidson *et al.*, 1993).

The rates of nitrification and denitrification, and consequently the production of  $\text{N}_2\text{O}$ , increase with increasing temperature (Sitaula and Bakken, 1993); however the establishment of strong predictive relationships between fluxes and field-scale parameters such as temperature and soil water content is difficult (Groffman *et al.*, 2000). Papen and Butterbach-Bahl (1999) observed that  $\text{N}_2\text{O}$  fluxes from Norway spruce forest in Germany increased from winter to summer months, as the temperature increased, and then declined from late summer to autumn as temperature decreased again. Brumme *et al.* (1999) found  $Q_{10}$  values up to 14 for a beech forest in Germany. However, Bowden *et al.* (1990) found no relationship between soil  $\text{N}_2\text{O}$  fluxes and soil temperature and water content in a red pine and a hardwood forest in Massachusetts, USA.

Tietema *et al.* (1992) studied the effect of soil water content on nitrification rates in incubation experiments on soil samples from two conifer forests in the Netherlands (Douglas fir and mixed stand of Douglas fir and Scots pine). They found that nitrification rate was linearly related to water content ( $R^2=0.99$ ). Duxbury *et al.*, (1982), made field measurements of soil N<sub>2</sub>O emissions from cultivated soils in New York and Florida. They concluded that although higher fluxes of soil N<sub>2</sub>O were always associated with wet soil conditions, which promote denitrification, the converse was not always true.

Soil organic content and the availability of labile organic compounds also influence the emissions of N<sub>2</sub>O. Brumme *et al.* (1992), in laboratory experiments on soils from forest ecosystems in Germany, found that the removal of the upper organic horizon decreased the soil N<sub>2</sub>O emissions. Papen and Butterbach-Bahl (1999) measured soil N<sub>2</sub>O fluxes after removing the organic layer from soil samples from a Norway spruce forest in Germany and concluded that about 70% of the N<sub>2</sub>O fluxes originated from the organic layer. However, Brumme *et al.* (1999) combined N<sub>2</sub>O flux and soil organic carbon data from several temperate forest studies and showed that soil organic carbon is not a strong predictor of annual N<sub>2</sub>O fluxes. This was also reported by Groffman *et al.* (2000).

Soil acidity has also an effect on soil N<sub>2</sub>O fluxes. An increase in the pH of forest soil can often increase the rate of nitrification (Tietema *et al.*, 1992; Smolander *et al.*, 1998). Ambus and Christensen (1995) found a significant correlation between soil N<sub>2</sub>O fluxes and pH ( $R=0.85$ ) in forest-grassland and agricultural ecosystems in Denmark.

Also the type of forest has an indirect influence on soil N<sub>2</sub>O fluxes, with deciduous forest ecosystems having higher N<sub>2</sub>O fluxes than coniferous forests in the temperate regions. This is probably due to direct and indirect impacts on soil structure and soil physical properties, as well as to differences in the litter quality. Litter produced in

deciduous forests has a generally higher quality (e.g. low lignin) and higher rates of N cycling. Papen and Butterbach-Bahl (1999) measured 4-5 times higher N<sub>2</sub>O fluxes in a beech forest compared to a Norway spruce forest in Germany.

Forest ecosystem disturbances, such as clearfelling, may reduce plant uptake of nitrogen and enhance nitrification and nitrogen losses. After clearfelling, although the production of litter has ceased, large amounts of logging residues and roots remain to be mineralised. In these situations, soil microbes have an even more important role than usually in nitrogen transformations, not only in nitrogen losses, but also in retaining nitrogen in the ecosystem (Smolander *et al.*, 1998). Clearfelling of upland forests may increase the soil water content levels as a result of the cessation of transpiration by the trees and reductions in interception losses (Rosen, 1984), leading to a greater tendency for anaerobic conditions to develop in the soils and potential for denitrification (Ineson *et al.*, 1991). Dutch and Ineson (1990) found that denitrification rates increased 10-fold after clearfelling, from 1-3 kg N ha<sup>-1</sup> y<sup>-1</sup> in the standing forest to 10-40 kg N ha<sup>-1</sup> y<sup>-1</sup> for the 2 years after clearfelling. Furthermore residual logging adds labile organic matter to the soil in slash and roots, favouring increased microbial activity (Frazer *et al.*, 1990). Also, soil compaction due to operation by harvesting machines can change soil properties and influence soil N<sub>2</sub>O fluxes. Hansen *et al.* (1993) studied the effect of soil compaction on soil N<sub>2</sub>O fluxes on agricultural soils in Norway due to tractor traffic and measured higher fluxes of N<sub>2</sub>O under compacted soil.

#### **4.1.2 The role of soils in methane (CH<sub>4</sub>) flux**

Methane (CH<sub>4</sub>) is an important greenhouse gas -the most important after CO<sub>2</sub> -with a warming effect 23 times that of CO<sub>2</sub> in the 100-year time horizon (Houghton *et al.*, 2001).

The largest sink for atmospheric CH<sub>4</sub> is its reaction with hydroxyl radicals (OH) in the troposphere, estimated to be  $420 \pm 80 \text{ Tg y}^{-1}$  (Crutzen, 1991). The largest biological sink for CH<sub>4</sub> in terrestrial ecosystems is microorganisms in aerobic soils (Stuedler *et al.*, 1989). By this process 1-10% equivalent of the total global emissions of CH<sub>4</sub> is consumed (Hütsch *et al.*, 1994). The global terrestrial sink of CH<sub>4</sub> has been estimated at  $29 \text{ Tg CH}_4 \text{ y}^{-1}$ , with a wide uncertainty range (Smith *et al.*, 2000), while Ridgwell *et al.* (1999) give a modelled value of  $38 \text{ Tg CH}_4 \text{ y}^{-1}$ . Soil CH<sub>4</sub> can be oxidised by both methanotrophic and nitrifying bacteria (Castro *et al.*, 1994). Oxidation is most rapid in coarse-textured forest soils with well-developed structure and a surface organic layer through which gases can readily diffuse (Smith *et al.*, 2000). CH<sub>4</sub> production by methanogenic bacteria is limited to strictly anaerobic environments, and thus the process is normally assumed to play only a minor role for CH<sub>4</sub> flux in well-drained soils. However, CH<sub>4</sub> production may be significant in anaerobic microsites within such soils during periods with high soil water content (Sitaula *et al.*, 1995).

CH<sub>4</sub> fluxes are influenced by soil temperature, soil water content, soil pH and nitrogen additions (Stuedler *et al.*, 1989). Maljanen *et al.* (2003) found a positive correlation between CH<sub>4</sub> emissions and soil temperature on afforested boreal organic soils in Finland ( $R=0.30$ ). Castro *et al.* (1995) found that CH<sub>4</sub> consumption increased when the temperature increased from  $-5$  to  $10 \text{ }^\circ\text{C}$  and remained relatively constant at temperatures between  $10$  and  $20 \text{ }^\circ\text{C}$  in red pine and hardwood forests in USA. Stuedler *et al.* (1989), measured methane uptake in two temperate forests, a 62-year-old red pine (*Pinus resinosa* Ait.) plantation and a 80-years-old mixed black oak/red maple (*Quercus velutina* Lam./*Acer rubrum* L.) stand in USA and found that CH<sub>4</sub> uptake was reduced by 14% by increased water content and by approximately 33% after six months of fertilisation.

A decrease in soil pH affects CH<sub>4</sub> fluxes positively but the effect is strongly dependent on the soil water content, according to Sitaula *et al.* (1995), who carried

out a lysimeter experiment on soil collected from a Scots pine (*Pinus sylvestris*) forest in Norway. However, Bradford *et al.* (2000) suggested that the drop in pH in a clearfelled beech forest in the UK contributed to a decrease in CH<sub>4</sub> uptake. Part of the effect may be an indirect one; Brumme and Borken (1999) have shown that the inhibition of litter decomposition in acid conditions in some forest soils can reduce substantially the entry of atmospheric CH<sub>4</sub> into the soil.

Fertilisation with N can also reduce the oxidation of soil CH<sub>4</sub>. Castro *et al.* (1994) studied the effect of fertiliser addition on N<sub>2</sub>O and CH<sub>4</sub> fluxes in a 26-years-old slash pine (*Pinus elliottii* var. *elliottii* Englem.) plantation in Florida. They measured 8 to 600 times higher soil N<sub>2</sub>O fluxes from the fertilised plots compared to the control plots and five to twenty times lower CH<sub>4</sub> uptake by the fertilised plots compared to the control plots. They measured the lowest CH<sub>4</sub> uptake in the plots with the highest NH<sub>4</sub><sup>+</sup> and NO<sub>3</sub><sup>-</sup> concentrations. Mosier *et al.* (1991) measured the effect of fertilisation in grasslands in Colorado and suggested that high N turnover, whether native or due to fertilisation, suppresses CH<sub>4</sub> uptake, and even a single application of N fertiliser can cause long-term decrease in CH<sub>4</sub> oxidation.

Temperate and boreal forest soils are considered to be a major sink for atmospheric CH<sub>4</sub>, with a consumption of up to 9.3 Tg CH<sub>4</sub>-C y<sup>-1</sup> (Steudler *et al.*, 1989). Yavitt *et al.* (1990) estimated that the temperate region may contribute 6.67 Tg CH<sub>4</sub> y<sup>-1</sup> to the atmosphere, with 9.1 Tg CH<sub>4</sub> y<sup>-1</sup> emitted from wetlands and 2.4 Tg CH<sub>4</sub> y<sup>-1</sup> consumed by forest soils. They also observed that although individual measurements of CH<sub>4</sub> from forest soils in the Appalachians were dominated by CH<sub>4</sub> uptake (70% of the individual measurements), the forest soils emitted CH<sub>4</sub> periodically and under all seasonal conditions. The net soil consumption of CH<sub>4</sub> can be reduced by conversion to other land uses (e.g. pasture) (Dobbie and Smith, 1996; Hütsch *et al.*, 2001; Maljanen *et al.*, 2003). Dobbie and Smith (1996) measured CH<sub>4</sub> oxidation rates in a mixed deciduous forest soil that were three times higher (2.19 to 2.97 mg CH<sub>4</sub> m<sup>-2</sup> d<sup>-1</sup>) than those in an adjacent agricultural land (average: 0.82 mg CH<sub>4</sub> m<sup>-2</sup> d<sup>-1</sup>) in

Scotland. Maljanen *et al.* (2003) studied CH<sub>4</sub> uptake in an organic soil in eastern Finland, originally drained and planted with birch (*Betula pendula*) and later cultivated with barley and grass. The agricultural soil showed some CH<sub>4</sub> emission (0.2 mg CH<sub>4</sub>-C m<sup>-2</sup> d<sup>-1</sup>), while at the same time the forest soil was a sink for CH<sub>4</sub> (1.2 mg CH<sub>4</sub>-C m<sup>-2</sup> d<sup>-1</sup>). Maljanen *et al.* (2001) studied the effect of afforestation on previous agricultural peatlands on soil CH<sub>4</sub> fluxes. They measured CH<sub>4</sub> fluxes in a 1-year old birch stand and two pine stands, 6 and 32 years old. They found that the oldest stand was a small sink or source of CH<sub>4</sub>, while the younger stands were sources of CH<sub>4</sub>, indicating that when the land use changes from agriculture to forest again CH<sub>4</sub> oxidation might increase but very slowly. The same trend was shown by Priemé *et al.* (1997) who reported that oxidation rates in temperate forests and woodlands in Denmark and Scotland, established on abandoned agricultural land, increased for 100 to 200 years after cultivation was abandoned.

The effect of clearfelling on soil CH<sub>4</sub> fluxes has not been extensively studied. Clearfelling alters many factors that may influence CH<sub>4</sub> fluxes, such as soil N dynamics (Vitousek and Matson, 1985; Smolander *et al.*, 1998), decomposition of organic matter (Binkley, 1986; Hendrickson *et al.*, 1989) and availability of available substrate, soil water content and soil water table depth (Smethurst and Nambiar, 1990; Keenan and Kimmins, 1993) and soil temperature (Chen *et al.*, 1993). It has been generally reported that clearfelling of temperate forests causes a reduction in CH<sub>4</sub> uptake. Bradford *et al.* (2000) observed reduced oxidation of CH<sub>4</sub> in the soil of clearfelled temperate forests (beech, Japanese larch and oak) in the UK. Keller *et al.* (1986) reported a slight decrease in the CH<sub>4</sub> consumption rates in a clearfelled plot relative to an adjacent uncut *terra firme* forest in Amazonia (20.1 and 26 kg CH<sub>4</sub> ha<sup>-1</sup> y<sup>-1</sup>, respectively). Steudler *et al.* (1989) reported inhibitions of the CH<sub>4</sub> oxidation by elevated soil N, and increases in soil mineral N concentrations have been observed after clearfelling (Vitousek and Matson, 1985; Steudler *et al.*, 1991). However, Castro *et al.* (2000) were the first to report a change in CH<sub>4</sub> fluxes after clearfelling. They measured CH<sub>4</sub> fluxes in two slash pine plantations in Florida before and after

clearfelling. Before clearfelling the soil were CH<sub>4</sub> sinks with average consumption rates ranging from -0.5 to -2.6 mg CH<sub>4</sub> m<sup>-2</sup> d<sup>-1</sup>, while after clearfelling they became CH<sub>4</sub> sources with emissions ranging from 3 to 11 mg CH<sub>4</sub> m<sup>-2</sup> d<sup>-1</sup>. Soil compaction due to the operation of harvesting machines may also have an effect on soil CH<sub>4</sub> fluxes. Hansen *et al.* (1993) studied the effect of soil compaction on soil CH<sub>4</sub> fluxes on agricultural soils in Norway due to tractor traffic and they measured lower CH<sub>4</sub> uptake rates under compacted soil.

Sitka spruce plantations on peaty gley soils are the dominant type of forest in Britain under management and it is important to evaluate the impacts of clearfelling on the emissions of important trace gases, such as N<sub>2</sub>O and CH<sub>4</sub>.

The specific objectives of this study were:

- a) to quantify the short-term effects of clearfelling on soil fluxes of N<sub>2</sub>O and CH<sub>4</sub> and to compare them with the fluxes from the mature stand before clearfelling and from an intact mature stand (control);
- b) to measure the environmental factors (soil temperature, soil water content and depth to water table) affecting the emissions of the above gases;
- c) to investigate the relationships between the trace gas fluxes and these environmental factors.

## **4.2 Materials and Methods**

### **4.2.1 Site description**

The study sites are fully described in Chapter 1 and in Chapter 3.

### 4.2.2 Measurements of soil N<sub>2</sub>O and CH<sub>4</sub>

The static closed chamber (CC) technique used for CO<sub>2</sub> was also used for the determination of soil CH<sub>4</sub> and N<sub>2</sub>O effluxes (Smith *et al.*, 1995). The chambers, consisting of PVC cylinders (inside diameter 40 cm and height 20 cm) open at the top and the bottom, were inserted into the soil in a depth of about 5 cm to make a gas-tight seal. An aluminium sheet with a compressive rubber seal acted as a lid at the top of each chamber. The lid was fixed in position to seal the chamber one hour prior to the time of the measurement. The chambers were left permanently in position in order to minimise the effects of disturbance caused by their insertion in the soil.

At each site twelve chambers were inserted in a stratified design (4 sets of 3 strata: original soil surface, ridge and furrow) in both mature stands. After clearfelling of one of the stands, the chambers were replaced as close as possible to their old positions.

The chambers were sealed with the lids for an hour and then air samples from inside the chambers were taken with 60 ml syringes for the analysis of CH<sub>4</sub>, while for the analysis of N<sub>2</sub>O samples were transferred to 20 ml glass vials which had been evacuated with a hand-pump. Linear accumulation of both gases was checked occasionally.

The samples were transferred to the laboratory and analysed by gas chromatography. CH<sub>4</sub> samples were analysed within 24 hours of their collection in a Hewlett Packard gas chromatograph (Series II 5890) equipped with a flame ionisation detector (FID) and a digital integrator. The N<sub>2</sub>O samples were analysed in an Agilent Model 6890 gas chromatograph equipped with an electron capture detector (ECD) and an auto sampler for 50 samples, within 1 week of their collection (samples in the vials can be kept up to 4 weeks). Peak integration (and autosampler control) were done with a

PeakSimple system. Before the analysis, the gas chromatographs were calibrated using known standards (1 and 10  $\mu\text{mol mol}^{-1}$   $\text{N}_2\text{O}$  and 3 and 10  $\mu\text{mol mol}^{-1}$   $\text{CH}_4$ ).

The fluxes of  $\text{CH}_4$  and  $\text{N}_2\text{O}$  measured with the closed chambers were calculated with the following equation:

$$F = d \times \frac{V}{A} \times \frac{(C_t - C_0)}{t} \quad (1)$$

where:  $F$  is the gas flux ( $\text{mg m}^{-2} \text{d}^{-1}$ ),  $d$  ( $\text{mg m}^{-3}$ ) is the gas density calculated on the assumption that 1 mole of gas (44g of  $\text{N}_2\text{O}$  and 16g of  $\text{CH}_4$ ) occupies  $22.4 \times 10^{-3} \text{ m}^3$  of volume at 273 K,  $V$  is the volume of the chamber ( $\text{m}^3$ ),  $C_t$  is the concentration of the gas ( $\mu\text{mol mol}^{-1}$ ) in the chamber after closure time  $t$  (d),  $C_0$  is the initial concentration of the gas ( $\mu\text{mol mol}^{-1}$ ),  $A$  is the area of the chamber ( $\text{m}^2$ ) and  $t$  is the time of chamber closure (min).

### 4.2.3 Statistical Analysis

Since, the fluxes of  $\text{N}_2\text{O}$  in the  $\text{CF}_{\text{before}}$  and 40-yr stands followed a normal distribution, but the fluxes at the CF site were not normally distributed, the test of significant differences was performed with a non-parametric ANOVA (Kruskal-Wallis test).  $\text{CH}_4$  fluxes were not normally distributed at any site and transforming the data (log, square root etc.) did not result in normal distributions of the fluxes, so that non-parametric ANOVA was used for these data too (Kruskal-Wallis test). All the statistical analyses were performed using SAS and the significance level was set at 0.05.

## 4.3 Results

### 4.3.1 Effect of clearfelling on environmental factors

As was mentioned in Chapter 3, clearfelling had an effect on soil temperature, water content and water table depth. Briefly, clearfelling caused increases in soil temperature and soil water content in comparison with the control stand. The temperature at the soil surface (1 cm depth) averaged 4.8 °C higher than the surface temperature in the mature stand throughout the growing season. At 5 cm depth it was about 3.6 °C higher and at 10 cm depth was 2.8 °C higher. Soil water content at the CF site was significantly higher than in the 40-yr stand. During the same period soil water content averaged 0.308-0.468 cm<sup>3</sup> cm<sup>-3</sup> in the clearfelled site compared to 0.114–0.310 cm<sup>3</sup> cm<sup>-3</sup> in the uncut 40-yr stand. Clearfelling also caused an increase in the soil pH, by 1.4 pH-units.

### 4.3.2 Short-term effect of clearfelling on soil N<sub>2</sub>O fluxes

Table 1 shows the monthly averages for the N<sub>2</sub>O fluxes at all sites, and the P values. The two mature stands (CF<sub>before</sub> and 40-yr) exhibited different patterns in the N<sub>2</sub>O fluxes prior to clearfelling. From October 2001 to January 2002 (the period for which measurements for both stands are available) soil N<sub>2</sub>O fluxes appeared to be significantly different between CF<sub>before</sub> and 40-yr stands for all months except October, with N<sub>2</sub>O fluxes in the CF<sub>before</sub> stand being significantly higher than the fluxes in the 40-yr stand. Also, comparing the CF<sub>before</sub> stand with the CF site (for August to September 2001 and the same period during 2002 after clearfelling) showed differences in N<sub>2</sub>O fluxes. For August N<sub>2</sub>O fluxes before clearfelling were significantly higher than after clearfelling (P=0.004). However, the fluxes for September 2002 after clearfelling were significantly higher than before clearfelling,

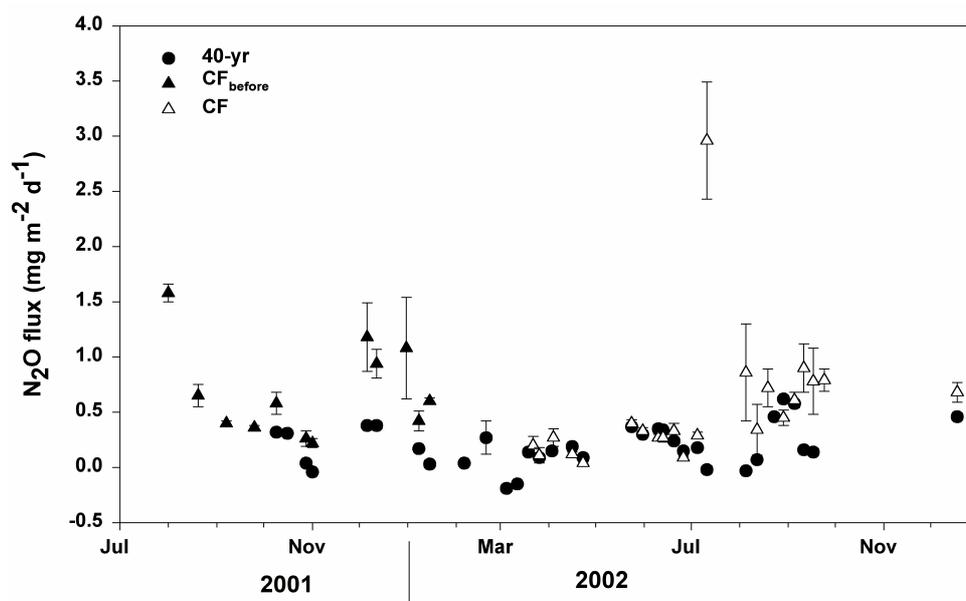
while fluxes of soil N<sub>2</sub>O for December 2001 were not significantly different from the fluxes after clearfelling in December 2002 ( $P < 0.001$  and  $P = 0.6$  for September and December, respectively).

From March to December 2002 total soil N<sub>2</sub>O fluxes from the CF site were  $1.80 \pm 0.36$  kg N<sub>2</sub>O ha<sup>-1</sup> ( $1.15 \pm 0.23$  kg N<sub>2</sub>O-N ha<sup>-1</sup>), while during the same period the N<sub>2</sub>O fluxes from the 40-yr stand were  $0.62 \pm 0.03$  kg N<sub>2</sub>O ha<sup>-1</sup> ( $0.40 \pm 0.02$  kg N<sub>2</sub>O-N ha<sup>-1</sup>).

**Table 1:** Monthly soil N<sub>2</sub>O fluxes (mg m<sup>-2</sup> d<sup>-1</sup>) and standard errors (in brackets) before and after clearfelling, compared with the fluxes in the uncut control 40-yr stand. Negative values denote soil N<sub>2</sub>O uptake. Statistical analysis was performed at the 0.05 significance level (Kruskal-Wallis test).

Month	CF <sub>before</sub> mg N <sub>2</sub> O m <sup>-2</sup> d <sup>-1</sup>	CF mg N <sub>2</sub> O m <sup>-2</sup> d <sup>-1</sup>	40-yr mg N <sub>2</sub> O m <sup>-2</sup> d <sup>-1</sup>	P value
<b>2001</b>				
August	0.95 (0.19)	-	-	-
September	0.38 (0.01)	-	-	-
October	0.46 (0.11)	-	0.21 (0.02)	0.06
November	0.22 (0.04)	-	-0.04 (0.01)	<0.001
December	1.05 (0.36)	-	0.38 (0.01)	0.009
<b>2002</b>				
January	0.48 (0.06)	-	0.10 (0.03)	<0.001
February	-	-	0.15 (0.03)	-
March	-	0.15 (0.05)	0.03 (0.02)	0.02
April	-	0.13 (0.03)	0.15 (0.01)	0.37
May	-	0.36 (0.02)	0.33 (0.01)	0.19
June	-	0.25 (0.03)	0.27 (0.01)	0.05
July	-	1.63 (0.38)	0.08 (0.02)	<0.001
August	-	0.55 (0.17)	0.25 (0.04)	0.79
September	-	0.77 (0.10)	0.29 (0.04)	<0.001
October	-	-	-	-
December	-	0.68 (0.22)	0.46 (0.05)	0.07

Figure 1 shows the weekly measurements during the period of study. N<sub>2</sub>O emissions in the 40-yr stand ranged from 0.03 to 0.62 mg m<sup>-2</sup> d<sup>-1</sup>, while occasionally N<sub>2</sub>O uptake took place with values ranging from -0.02 to -0.19 mg m<sup>-2</sup> d<sup>-1</sup>. In the CF<sub>before</sub> stand N<sub>2</sub>O emissions ranged from 0.22 to 1.58 mg m<sup>-2</sup> d<sup>-1</sup>, while after clearfelling the N<sub>2</sub>O emissions ranged from 0.04 to 2.96 mg m<sup>-2</sup> d<sup>-1</sup>. The highest emission of N<sub>2</sub>O (2.96 mg N<sub>2</sub>O m<sup>-2</sup> d<sup>-1</sup>) in the CF occurred in July, after heavy rain. Soil N<sub>2</sub>O fluxes did not follow a seasonal trend at any of the sites. Very high emissions of N<sub>2</sub>O were also observed in the CF<sub>before</sub> stand during December 2001, ranging from 0.94 to 1.18 mg m<sup>-2</sup> d<sup>-1</sup> when soil surface temperatures ranged from 0.5 to 4.3 °C (T<sub>1</sub>).



**Figure 1:** N<sub>2</sub>O fluxes before and after clearfelling and in the uncut control stand. The vertical bars indicate the standard error of the mean.

N<sub>2</sub>O fluxes were linearly correlated with soil temperature in the 40-yr stand with weak but significant relationships ( $R^2$ : 0.17, 0.19 and 0.20 for T<sub>1</sub>, T<sub>5</sub> and T<sub>10</sub>

respectively, all  $P < 0.05$ ). A weak and not significant ( $P > 0.05$ ) linear relationship with soil temperature was also observed in the  $CF_{\text{before}}$  stand ( $R^2 = 0.25$ , for  $T_5$ , not applicable for  $T_1$  and  $T_{10}$ ). After clearfelling  $N_2O$  exhibited a higher and significant dependence on soil temperature with a linear relationship ( $R^2$ : 0.33, 0.39 and 0.30 for  $T_1$ ,  $T_5$  and  $T_{10}$  respectively, all  $P < 0.05$ ). Figure 2 shows the linear relationship between soil  $N_2O$  emissions and  $T_5$  in the CF site.

No relationship was observed between  $N_2O$  fluxes and soil water content or soil water table depth in either of the sites.

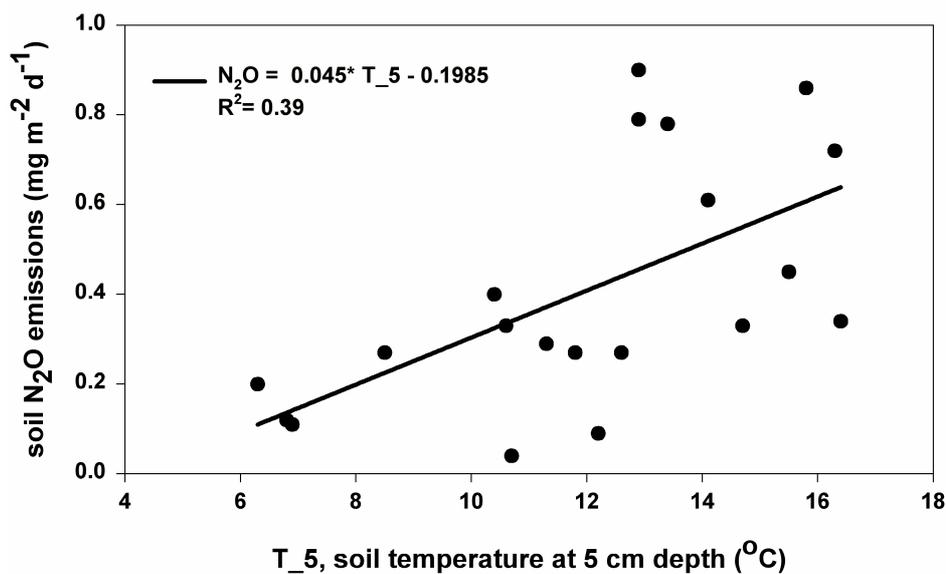
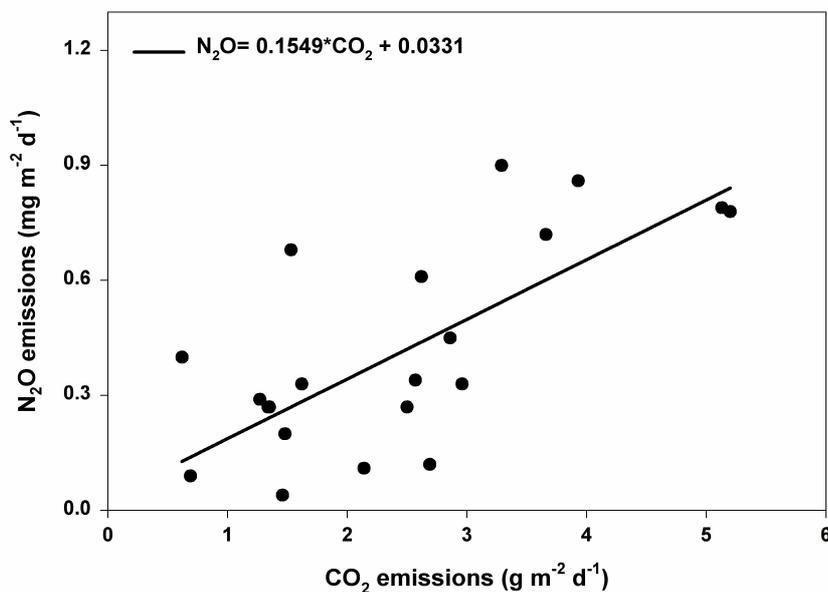


Figure 2: The dependence of  $N_2O$  fluxes on soil temperature in the CF site ( $R^2 = 0.39$ ,  $P = 0.005$ ).



**Figure 3:** The linear relationship ( $R^2=0.50$ ,  $P=0.0002$ ) between N<sub>2</sub>O and CO<sub>2</sub> emissions from the CF site. The points represent weekly measured values. The CO<sub>2</sub> values are the ones measured with the static closed chamber method.

A significant ( $P=0.0002$ ) linear relationship between N<sub>2</sub>O and CO<sub>2</sub> emissions was also found for the CF site (Figure 3) and the variability in CO<sub>2</sub> efflux explained 50% of the variability in N<sub>2</sub>O emissions. However, no relationship between N<sub>2</sub>O and CO<sub>2</sub> was observed for the 40-yr stand or the CF<sub>before</sub> stand.

### 4.3.3 Short-term effect of clearfelling on soil CH<sub>4</sub> fluxes

Monthly averages of soil CH<sub>4</sub> fluxes in all sites and their P values are shown in Table 2. Similar to soil N<sub>2</sub>O fluxes, CF<sub>before</sub> and 40-yr stands exhibited significant differences in CH<sub>4</sub> fluxes prior to clearfelling. CF<sub>before</sub> was a CH<sub>4</sub> sink from August 2001 till January 2002 while the 40-yr stand was a CH<sub>4</sub> sink from October to December 2001. However, the strength of the sink was much smaller compared to the CF<sub>before</sub> and resulted in significant differences.

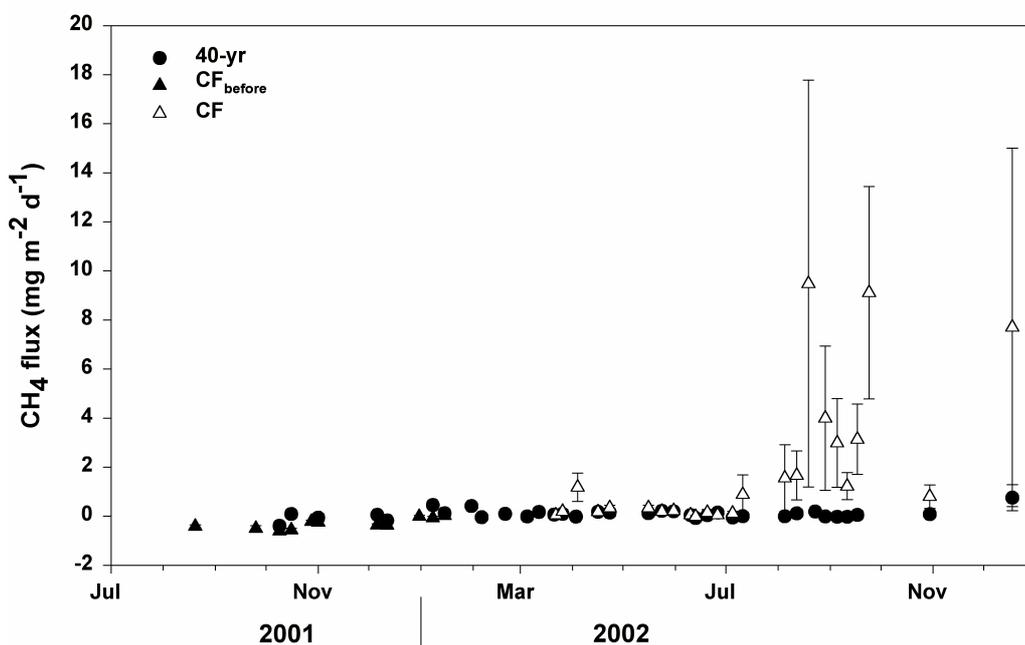
Also, when comparing the months before and after clearfelling for the same stand (August–December 2001 with August–December 2002), the soil turned from a sink for CH<sub>4</sub> to a source ( $P < 0.0001$ ), demonstrating an effect of clearfelling on the soil CH<sub>4</sub> fluxes.

From March to December 2002, total CH<sub>4</sub> fluxes from CF were  $6.6 \pm 4.7$  kg CH<sub>4</sub> ha<sup>-1</sup> ( $4.8 \pm 3.5$  kg CH<sub>4</sub>-C ha<sup>-1</sup>), while the 40-yr stand emissions during the same period were  $0.40 \pm 0.2$  kg CH<sub>4</sub> ha<sup>-1</sup> ( $0.3 \pm 0.15$  kg CH<sub>4</sub>-C ha<sup>-1</sup>).

**Table 2:** Monthly soil CH<sub>4</sub> fluxes (mg m<sup>-2</sup> d<sup>-1</sup>) and standard errors (in brackets) before and after clearfelling, compared with the fluxes in the uncut control 40-yr stand. Negative values denote soil CH<sub>4</sub> uptake. Statistical analysis was performed at the 0.05 significance level (Kruskal-Wallis test).

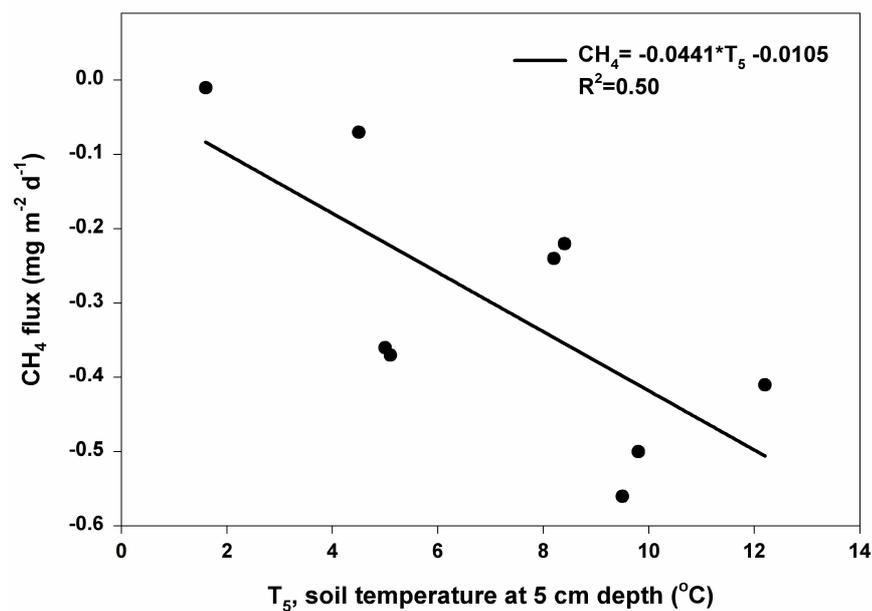
Month	CF <sub>before</sub> mg CH <sub>4</sub> m <sup>-2</sup> d <sup>-1</sup>	CF mg CH <sub>4</sub> m <sup>-2</sup> d <sup>-1</sup>	40-yr mg CH <sub>4</sub> m <sup>-2</sup> d <sup>-1</sup>	P value
<b>2001</b>				
August	-0.41 (0.06)		-	-
September	-0.50 (0.11)		-	-
October	-0.45 (0.04)		-0.1 (0.06)	<0.001
November	-0.24 (0.07)		-0.05 (0.06)	0.001
December	-0.24 (0.04)		-0.05 (0.06)	0.006
<b>2002</b>				
January	-0.02 (0.04)		0.30 (0.11)	0.005
February	-		0.11 (0.09)	-
March		0.12 (0.05)	0.12 (0.02)	0.26
April		0.49 (0.13)	0.19 (0.03)	0.002
May		0.26 (0.04)	0.18 (0.02)	0.02
June		0.06 (0.04)	0.04 (0.02)	0.29
July		0.50 (0.40)	-0.02 (0.01)	<0.001
August		4.07 (2.17)	0.07 (0.03)	<0.001
September		4.12 (1.27)	0.01 (0.001)	<0.001
October 2002		0.78 (0.48)	0.09 (0.02)	0.12
December 2002		7.72 (7.4)	0.76 (0.53)	0.63

Figure 4 shows the weekly fluxes of CH<sub>4</sub> before and after clearfelling as well in the uncut 40-yr control stand. In the 40-yr stand CH<sub>4</sub> emissions ranged from 0.004 to 0.76 mg m<sup>-2</sup> d<sup>-1</sup>, while occasionally consumption of CH<sub>4</sub> took place, at rates ranging from -0.002 to -0.386 mg m<sup>-2</sup> d<sup>-1</sup>. In the CF<sub>before</sub> stand, there was consumption of CH<sub>4</sub> from the period October to December 2001, ranging from -0.01 to -0.61 mg m<sup>-2</sup> d<sup>-1</sup>, whereas in January 2002 it was a source of CH<sub>4</sub> with emissions of 0.02 mg m<sup>-2</sup> d<sup>-1</sup>. The CF site was mainly a source of CH<sub>4</sub>, with emission ranging from 0.05 to 9.48 mg m<sup>-2</sup> d<sup>-1</sup>, but it was a sink for CH<sub>4</sub> during one sampling day in June, with an uptake of -0.01 mg CH<sub>4</sub> m<sup>-2</sup> d<sup>-1</sup>. CH<sub>4</sub> fluxes in the CF exhibited large spatial variability, as can be seen from the large standard errors of the mean in Figure 4. This is compatible with the main production of CH<sub>4</sub> occurring in microsites where anaerobic conditions dominated.

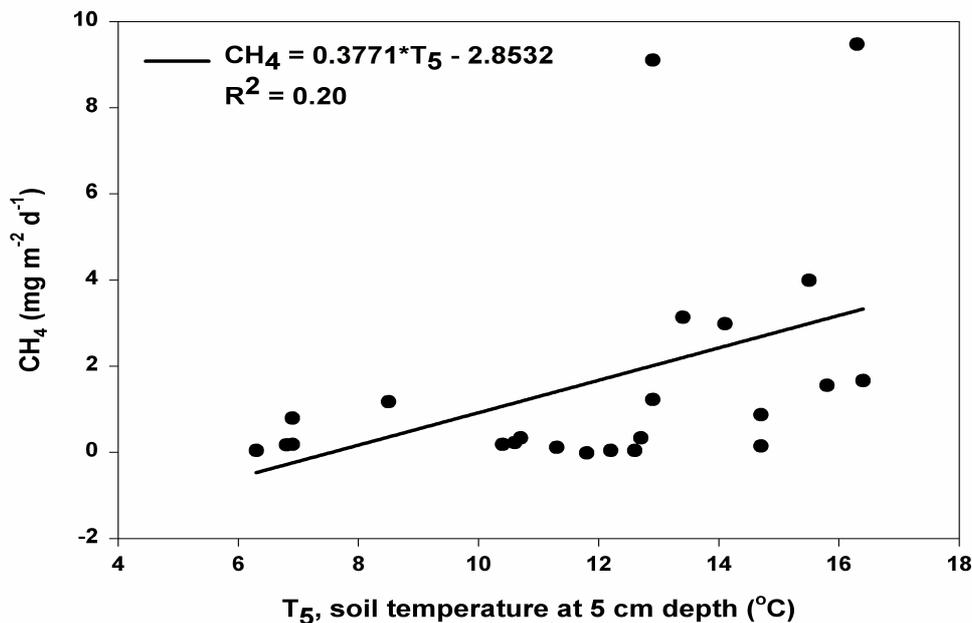


**Figure 4:** Soil CH<sub>4</sub> fluxes before and after clearfelling and in the uncut control stand. The vertical bars indicate the standard error of the mean.

CH<sub>4</sub> emissions were weakly but significantly correlated with soil temperature at 1 and 5 cm depth in the 40-yr stand (linear relationships,  $R^2=0.20, 0.18$ , for  $T_1$  and  $T_5$  respectively, all  $P<0.05$ ) and weakly but not significantly related to soil temperature at 10 cm depth (linear relationship,  $R^2=0.14$ ,  $P=0.06$ ). In the CF<sub>before</sub> stand soil CH<sub>4</sub> uptake was moderately correlated with soil temperature ( $R^2=0.50$ ,  $P=0.03$  Figure 5). After clearfelling, weak but significant linear relationships of CH<sub>4</sub> emissions with soil temperature were also observed ( $R^2: 0.19, 0.20, 0.22$  for  $T_1, T_5$  and  $T_{10}$  respectively, all  $P<0.05$ ). Figure 6 depicts the linear relationship between CH<sub>4</sub> emissions and  $T_5$  in the CF site.

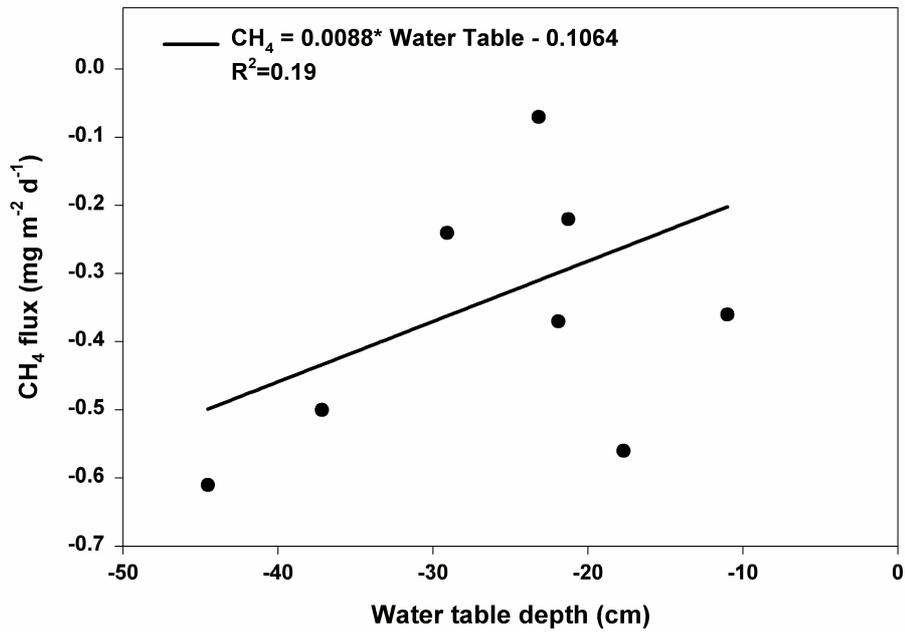


**Figure 5:** Linear relationship ( $R^2=0.50$ ,  $P=0.03$ ) between CH<sub>4</sub> fluxes and soil temperature at 5 cm depth in the CF<sub>before</sub> stand. The negative values indicate CH<sub>4</sub> uptake.

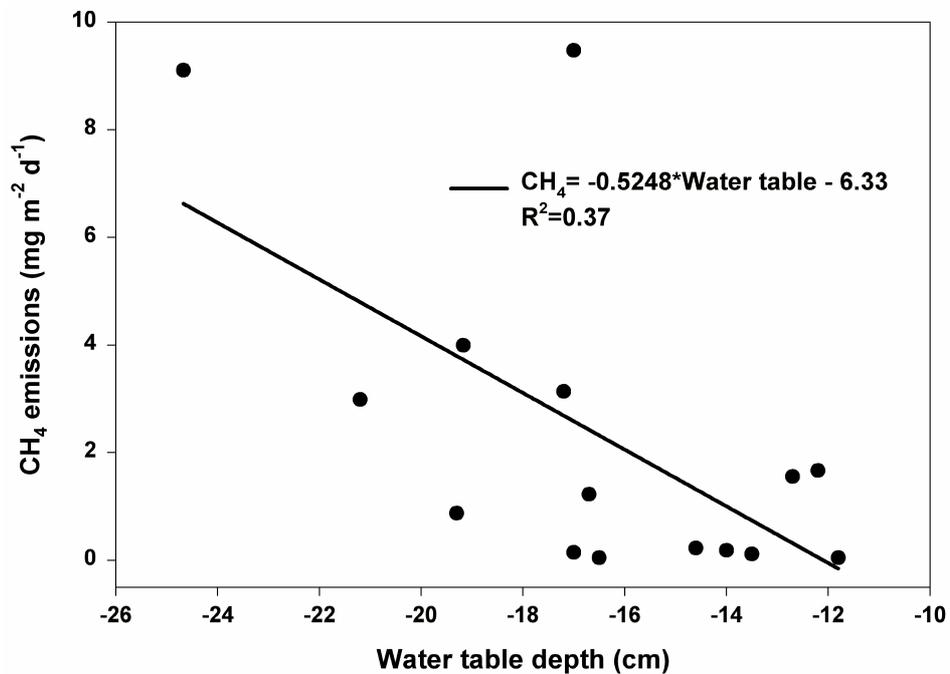


**Figure 6:** The dependence of CH<sub>4</sub> fluxes on soil temperature in the CF site ( $R^2=0.20$ ,  $P=0.03$ ).

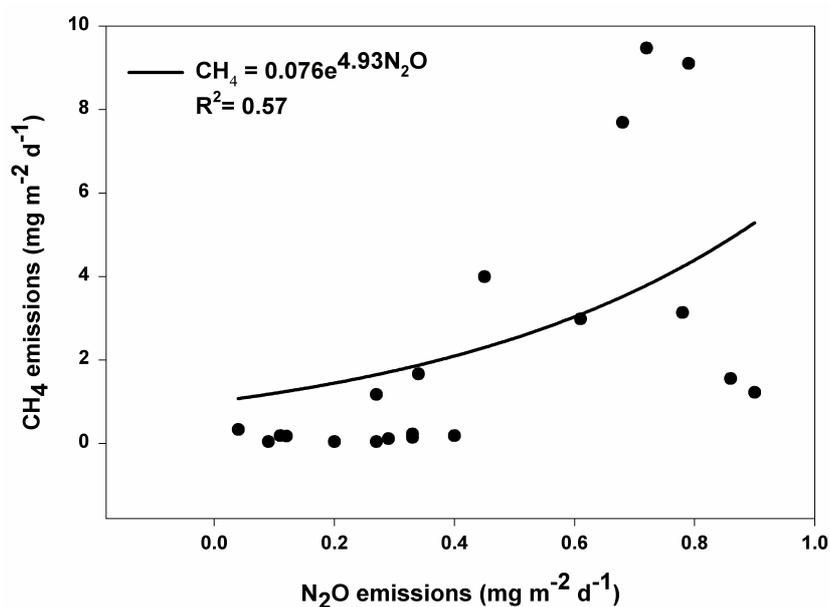
In the CF<sub>before</sub> stand, CH<sub>4</sub> uptake was weakly correlated with the depth of the water table ( $R^2= 0.20$ ,  $P=0.2$ , Figure 7), with uptake increasing as the depth to the water table increasing. After clearfelling, CH<sub>4</sub> emissions were correlated ( $R^2= 0.37$ ,  $P=0.01$ ) with water table depth during the summer period only (end of May 2002 to September 2002), with CH<sub>4</sub> emissions increasing when the water table was lowered from 12 to 25 cm below soil surface (Figure 8) and this is the period when the largest fluxes occurred. No relationship between CH<sub>4</sub> fluxes and the depth of the water table was observed in the 40-yr stand. Also, no relationship between CH<sub>4</sub> fluxes and soil water content was found at any of the sites.



**Figure 7:** The linear relationship ( $R^2=0.19$ ,  $P=0.2$ ) between soil CH<sub>4</sub> fluxes and the water table depth in the CF<sub>before</sub> stand. The negative values of CH<sub>4</sub> indicate uptake.



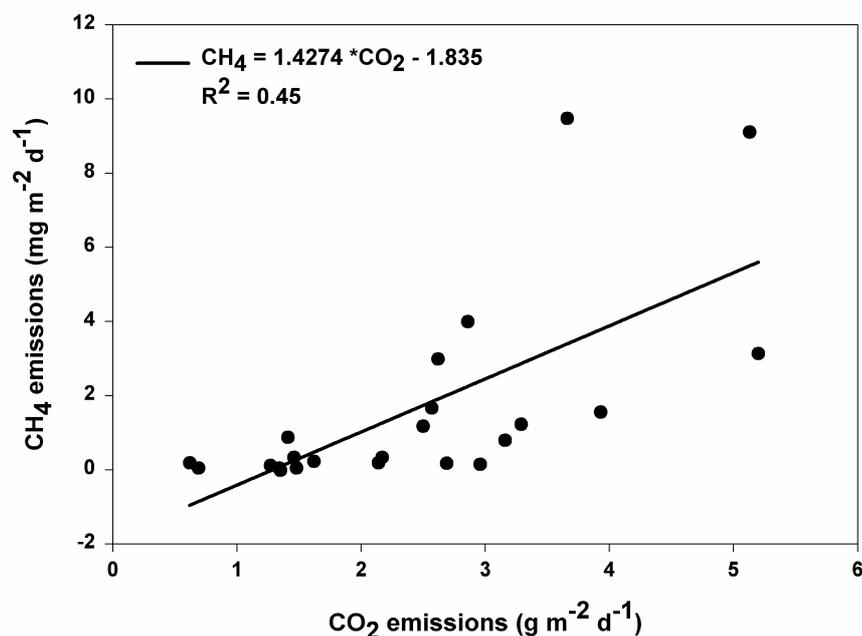
**Figure 8:** The linear relationship ( $R^2 = 0.37$ ,  $P=0.01$ ) between the CH<sub>4</sub> emissions and the water table depth, in the CF site for the summer period (end of May- September 2002).



**Figure 9:** The exponential relationship ( $R^2=0.57$ ,  $P=0.002$ ) between soil  $N_2O$  and  $CH_4$  emissions after clearfelling. The highest  $N_2O$  flux of  $2.96 \text{ mg m}^{-2} \text{ d}^{-1}$ , on 11/07/02 was excluded from the relationship. If this value was included the relationship would be power function ( $R^2=0.35$ ,  $P=0.2$ ).

Soil  $CH_4$  fluxes were correlated with soil  $N_2O$  fluxes, with an exponential relationship in the CF site ( $R^2=0.57$ ,  $P=0.002$  if the high flux of  $2.96 \text{ mg m}^{-2} \text{ d}^{-1}$  that occurred on the 11/07/02 was excluded, Figure 9). If this value was included the relationship would result in a weaker power ( $R^2=0.35$ ,  $P=0.2$ ). No relationship was found between the  $CH_4$  and  $N_2O$  fluxes in the  $CF_{\text{before}}$  stand or in the 40-yr stand.

Also,  $CH_4$  fluxes in the CF site were linearly correlated with the  $CO_2$  fluxes ( $R^2=0.45$ ,  $P=0.0004$ ), with  $CH_4$  flux increasing with increasing  $CO_2$  flux (Figure 10). No relationship was found in the 40-yr stand or the  $CF_{\text{before}}$  stand.



**Figure 10:** The linear relationship ( $R^2 = 0.45$ ,  $P=0.0004$ ) between CH<sub>4</sub> and CO<sub>2</sub> emissions from the CF site. The points are weekly measured values. The CO<sub>2</sub> fluxes are the ones measured with the static closed chamber method.

## 4.4 Discussion

### 4.4.1 Short-term effect of clearfelling on soil N<sub>2</sub>O fluxes

Over a period of ten months after clearfelling the CF stand was a significantly higher source of soil N<sub>2</sub>O to the atmosphere, emitting  $1.15 \pm 0.23$  kg N<sub>2</sub>O-N ha<sup>-1</sup>, whereas the 40-yr stand emitted only  $0.4 \pm 0.02$  kg N<sub>2</sub>O-N ha<sup>-1</sup> over the same period ( $P < 0.001$ ). However, it cannot be inferred with certainty that clearfelling had a significant effect on soil N<sub>2</sub>O fluxes, as before clearfelling the CF<sub>before</sub> stand showed generally significantly higher fluxes than the 40-yr control stand. The same trend was followed after clearfelling, with the CF site having generally higher fluxes than the 40-yr stand, although during May and August 2002 they were not significantly

different and during March 2002 N<sub>2</sub>O fluxes from the 40-yr stand were slightly higher but not significantly different from the CF N<sub>2</sub>O fluxes. Comparing the months for which there were measurements before and after clearfelling for the same site, there was not a consistent trend. N<sub>2</sub>O fluxes were significantly higher in the CF<sub>before</sub> stand in August 2001 than at the CF site in August 2002 (P=0.004), while N<sub>2</sub>O in the CF<sub>before</sub> stand in September 2001 were significantly lower than the CF N<sub>2</sub>O fluxes in September 2002 (P<0.001), and fluxes were not significantly different in December 2001 and December 2002 (P=0.6).

It is very well established that soil N<sub>2</sub>O fluxes exhibit large variability both in space and time (Sitaula and Bakken, 1993; Maljanen *et al.*, 2001). Ambus and Christensen (1995) found a mean CV of 285% for N<sub>2</sub>O fluxes in beech forests in Denmark. Maljanen *et al.* (2001) measured N<sub>2</sub>O fluxes on a previously cultivated peatland afforested with 6- and 23- years old pine stands and a 1-year-old birch stand. There was no distinct correlation between the age of the stand and the N<sub>2</sub>O fluxes. The highest mean N<sub>2</sub>O flux occurred at the oldest stand but the highest peak was found in the youngest stand. Papen and Butterbach-Bahl (1998) measured soil N<sub>2</sub>O fluxes in a 89-year-old Norway spruce forest (*Picea abies*) in Germany for 3 years and found high inter-annual variability (CV values > 210%) and demonstrated the need for long-term measurements in order to come up with more reliable estimates of the N<sub>2</sub>O source strength of an ecosystem.

Generally, clearfelling is considered to increase soil N<sub>2</sub>O fluxes. Smolander *et al.*, (1998) conducted incubation experiments with soil samples from a 60-years old Norway spruce (*Picea abies*) in Finland, before and after clearfelling. They observed that clearfelling initiated nitrification in the soil, which did not occur earlier unless the soil had been fertilised and limed. Keller *et al.* (1986) measured fluxes of soil N<sub>2</sub>O in a *terra firme* forest and an adjacent clearfelled site in the Amazon region of Brazil. They measured higher fluxes in the clearfelled site (5.69 kg N<sub>2</sub>O ha<sup>-1</sup> y<sup>-1</sup>) than in the adjacent forest (1.80 kg N<sub>2</sub>O ha<sup>-1</sup> y<sup>-1</sup>). In the clearfelled site no burning of the

residual logs had taken place and the ground was shaded by slash left after clearfelling and by re-growth. Steudler *et al.* (1991) reported that clearfelling a wet tropical forest in Puerto Rico increased soil N<sub>2</sub>O fluxes by 14-fold compared to the reference intact site. The soil N<sub>2</sub>O fluxes in the clearfelled site were 6.7 kg N<sub>2</sub>O-N ha<sup>-1</sup> y<sup>-1</sup> and 0.48 kg N<sub>2</sub>O-N ha<sup>-1</sup> y<sup>-1</sup> in the reference site. Bradford *et al.* (2000) measured soil N<sub>2</sub>O fluxes in intact soil cores from clearfelled and adjacent uncut stands of beech and larch in UK. The fresh soil cores from the clear-cut areas had higher N<sub>2</sub>O fluxes than those from the forest stands; however, they were not significantly different. Matson and Vitousek (1981) measured nitrification rates in incubated soil samples from clearfelled hardwood forests in Indiana, U.S.A. up to 9 years after clearfelling as well from control uncut stands. They found that net NO<sub>3</sub> production was always greater in the clearfelled soils than in the matched control soils, regardless of the time since tree felling or conditions of incubation (i.e. both constant laboratory conditions and *in situ* temperature and moisture conditions - buried bags experiment). They attributed this increase to the improved substrate quality in the clearfelled plots as well as to the higher initial population of autotrophic bacteria in the clearfelled soils. An initial higher population of autotrophic bacteria could explain the differences in soil N<sub>2</sub>O emissions in the two mature stands prior to clearfelling in the present study. This possibility can also be supported by the different CH<sub>4</sub> fluxes before clearfelling. CF<sub>before</sub> was a stronger CH<sub>4</sub> sink than the 40-yr stand, although they had almost the same soil physical properties. CH<sub>4</sub> can be oxidised both by methanotrophs and by nitrifying bacteria (Castro *et al.*, 1994), thus the higher CH<sub>4</sub> consumption in the CF<sub>before</sub> stand could have been caused by a larger nitrifying bacterial population as well by the presence of methanotrophs. Castro *et al.* (1994) measured CH<sub>4</sub> uptake in fertilised and unfertilised plots in a slash pine plantation in Florida and found that soils with the highest concentrations of NH<sub>4</sub><sup>+</sup> and NO<sub>3</sub><sup>-</sup> concentrations had the lowest CH<sub>4</sub> uptake and nitrifying bacteria dominated the CH<sub>4</sub> oxidation in the soil surface.

The N<sub>2</sub>O values reported here are higher than the mean 0.2 kg N ha<sup>-1</sup> y<sup>-1</sup> reported by Bouwman *et al.* (1995) and 0.39 ± 0.27 kg N ha<sup>-1</sup> y<sup>-1</sup> by Brumme *et al.* (1999), for temperate forests. However, Chen *et al.* (2000) measured soil N<sub>2</sub>O emissions in temperate coniferous forests in China and they found emissions ranging from 0.28 to 1.28 kg N ha<sup>-1</sup> y<sup>-1</sup>. Skiba *et al.* (1998) estimated a mean annual N<sub>2</sub>O flux from conifer forests (Sitka spruce and Scots pine-*Pinus sylvestris*) in Scotland of 0.4 kg N<sub>2</sub>O-N ha<sup>-1</sup> y<sup>-1</sup>, which is in agreement with the annual flux from the 40-yr stand. Henrich and Haselwandter (1997) estimated a N<sub>2</sub>O flux of 81 ± 8.1 g N ha<sup>-1</sup> y<sup>-1</sup> in a 80 year old Norway spruce stand in Austria. Butterbach-Bahl *et al.* (1997) measured emissions between 0.5 and 5.5 kg N<sub>2</sub>O ha<sup>-1</sup> y<sup>-1</sup> (0.3 to 3.5 kg N<sub>2</sub>O-N ha<sup>-1</sup> y<sup>-1</sup>) in a spruce forest (*Picea abies* L. Karst.) in Germany while, Butterbach-Bahl *et al.* (1998) measured soil N<sub>2</sub>O fluxes of between 0.07 and 0.29 kg N<sub>2</sub>O ha<sup>-1</sup> y<sup>-1</sup> (0.04 and 0.18 kg N<sub>2</sub>O-N ha<sup>-1</sup> y<sup>-1</sup>) in a 56-year-old spruce forest in Ireland. However, Bowden *et al.* (1990) measured soil N<sub>2</sub>O fluxes of 0.010 kg N ha<sup>-1</sup> y<sup>-1</sup> in a red pine (*Pinus resinosa* Ait.) forest in Massachusetts, USA. Goodroad and Keeney, (1984) reported soil N<sub>2</sub>O fluxes from a red pine forest (*Pinus strobus*) in Wisconsin (USA) of 0.80 kg N<sub>2</sub>O-N ha<sup>-1</sup> for a period extending from March to December, which is double the flux from the 40-yr stand during the same period.

Soil temperature had a significant effect (P<0.05) on soil N<sub>2</sub>O fluxes in the 40-yr stand, although the relationships were weak, however it had no significant effect on soil N<sub>2</sub>O at the CF<sub>before</sub> stand (P>0.05 for T<sub>5</sub> only, as it was not applicable for T<sub>1</sub> and T<sub>10</sub>). The effect of clearfelling on the soil N<sub>2</sub>O fluxes can be detected in the better and significant relationship between soil N<sub>2</sub>O fluxes and soil temperature in the CF site (P<0.05, for 1, 5 and 10 cm depth). Tree harvesting has resulted in large residues inputs into the soil and decomposition of roots, providing available substrate. Umarov (1990) concluded that denitrification is mainly dependent on the presence of sufficient organic matter, and other ecological factors, such as temperature, pH etc, have secondary significance. Thus the changes in the temperature status coupled with the available substrate have probably favoured increased production of N<sub>2</sub>O. Also,

diffusion rates increase with temperature (Maljanen *et al.*, 2002). Furthermore, higher temperatures in the CF site have probably enhanced microbial activity and this is reflected in the significant increase of N<sub>2</sub>O with temperature in the clearfelled site. Smolander *et al.* (1998) found that clearfelling slightly increased summer mean values for microbial biomass N in the first summer after clearfelling in a Norway spruce forest in Finland. Papen and Butterbach-Bahl (1999) also found a weak but significant relationship between soil temperature in the organic layer and N<sub>2</sub>O fluxes ( $R^2=0.23$ ,  $P<0.001$ ) at a WFPS > 65% in a 89-years-old Norway spruce forest in Germany, while at lower WFPS they found even weaker relationships. Henrich and Haselwandter (1997) found no relationship between N<sub>2</sub>O fluxes and soil temperature or water content in a 80-years-old Norway spruce stand in Austria.

No relationship was observed between N<sub>2</sub>O fluxes and the soil water content or the water table depth in any of the sites. The lack of dependence of soil N<sub>2</sub>O fluxes on soil water content or water table depth and the poor relationships with soil temperature in the CF<sub>before</sub> and 40-yr stands indicate that the emissions were probably dependent on the substrate available, rather than on abiotic factors. Although water content is an important regulator of N<sub>2</sub>O flux, its importance is more apparent in ecosystems in which N is cycling rapidly (Verchot *et al.*, 1999). Henrich and Haselwandter (1997) also found no relationship between N<sub>2</sub>O fluxes and soil water content in a 80-years-old Norway spruce stand in Austria. No relationship between N<sub>2</sub>O fluxes and soil water content was observed from Bowden *et al.* (1990) for a 62 years old red pine and a hardwood forest in the U.S.A. However, the highest flux of N<sub>2</sub>O (2.96 mg m<sup>-2</sup> d<sup>-1</sup>) was measured in the CF site in a warm day in July 2001 after heavy rain. Goodroad and Keeney (1984b) also observed high fluxes of N<sub>2</sub>O following periods of heavy rainfall during the summer, and Maljanen *et al.* (2002) measured the highest diurnal variation in soil N<sub>2</sub>O fluxes after the first rainfall of the summer in a birch forest in Finland, with a daytime maximum of 4.32 mg m<sup>-2</sup> d<sup>-1</sup>, which was about 3 times higher than 1.68 mg m<sup>-2</sup> d<sup>-1</sup> during the following night. Henrich and Haselwandter (1997) measured the highest N<sub>2</sub>O flux of  $6.5 \pm 6.6$  mg m<sup>-2</sup>

$\text{d}^{-1}$  in July, in an 80-years-old Norway spruce in Austria, but they could not attribute this high flux to either temperature or water content. Very high fluxes of  $\text{N}_2\text{O}$  fluxes were also measured in the  $\text{CF}_{\text{before}}$  stand in December 2001 during very cold days with soil surface temperature between 0.5 to 4.3 °C. High fluxes of  $\text{N}_2\text{O}$  from forest soil have been often reported with even soil temperatures even below the freezing point. Papen and Butterbach-Bahl (1999) reported that soil  $\text{N}_2\text{O}$  fluxes during the frost period in winter contributed more than 73% of the annual emissions in a 89-years-old Norway spruce forest in Germany. They linked these high fluxes to microbial activity. Furthermore they suggested that microbial biomass killed by very low temperatures offered available substrate to the surviving microbes and a much more easily degraded carbon source.

The increase in the pH at Harwood after clearfelling might have favoured nitrification, while the rise in the water table might have favoured denitrification in microsites. Smolander *et al.* (1998) found that clearfelling of a Norway spruce stand in Finland increased soil pH by 0.6-0.7 pH units; nitrification also increased and was weakly positively correlated with pH ( $R^2=0.39$ ,  $P=0.001$ ). Frazer *et al.* (1990) also measured higher rates of nitrification in clearfelled mixed conifer stands in northern California than in an adjacent uncut 100-years-old forest. Griffiths and Swanson (2001) found significantly higher denitrification rates in young stands in a chronosequence of harvested Douglas fir (*Pseudotsuga menziesii* (Mirb.) Franco) forests. Denitrification rates tended to decrease to values close to those for the 40-years-old and old growth stands with increasing time after harvest. Soil acidity has often been implicated as the factor inhibiting nitrification in coniferous forest soils (Tietema *et al.*, 1992); both of the 40-years-old stands in the present study had acid soil (pH ~ 3.4), and this fact coupled with the lack of available substrate  $\text{NH}_4$  necessary for nitrification, could explain the low  $\text{N}_2\text{O}$  fluxes in these stands.

Finally, soil compaction that was caused from the operation of harvesting machines could also have had an effect on soil  $\text{N}_2\text{O}$  fluxes in the CF. There is no evidence in

the literature of such an effect in a forest soil, but Hansen *et al.* (1993) reported higher N<sub>2</sub>O fluxes in an agricultural field in Norway after soil compaction due to tractor traffic, compared with an uncompacted site.

Soil N<sub>2</sub>O fluxes were significantly correlated to soil CO<sub>2</sub> fluxes in the CF site with a linear relationship ( $R^2=0.50$ ,  $P=0.0002$ ), but no relationship was observed in the 40-yr stand (or the CF<sub>before</sub> stand). The relationship between N<sub>2</sub>O and CO<sub>2</sub> in the CF site suggests that factors controlling the production of these gases, e.g. soil respiration and nitrification and denitrification depend on the availability of labile organic carbon. Furthermore, higher CO<sub>2</sub> emissions indicate higher decomposition rates that consume oxygen and create the low-oxygen or anaerobic conditions needed for N<sub>2</sub>O production (Garcia-Montiel *et al.*, 2002; Maljanen *et al.*, 2002). So it seems that clearfelling has enhanced denitrification, with the greater availability of organic substrate playing a role. Garcia-Montiel *et al.* (2002) found a strong empirical relationship between N<sub>2</sub>O and CO<sub>2</sub> emissions in forests in the Amazon, Brazil, and the CO<sub>2</sub> flux explained 62% of the variability in N<sub>2</sub>O emissions ( $P<0.0001$ ). Hansen *et al.* (1993) reported correlated or weakly correlated soil N<sub>2</sub>O emissions with soil CO<sub>2</sub> concentrations in an agricultural field in Norway in both compacted and uncompacted soil ( $R^2=0.46$ ,  $P<0.01$  and  $0.48$ ,  $P<0.01$  in the uncompacted and compacted soil, respectively, in plots receiving 140 kg fertilizer N ha<sup>-1</sup> ( $R^2=0.24$ ,  $P>0.05$  and  $0.56$ ,  $P<0.001$  in unfertilised uncompacted and compacted plots). Brumme *et al.* (1999) observed a close relationship between soil CO<sub>2</sub> and N<sub>2</sub>O fluxes, during periods of high emissions, in temperate and tropical forests. Soil respiration consumes oxygen and that, combined with the rise of the water table, generally creates anaerobic conditions that favour denitrification and N<sub>2</sub>O production in the CF site, although N<sub>2</sub>O fluxes were not correlated with soil water content or the depth to the soil water table. Not all studies show the linkage between N<sub>2</sub>O and CO<sub>2</sub>. For example, Dong *et al.* (1998) measured fluxes of CO<sub>2</sub>, CH<sub>4</sub> and N<sub>2</sub>O in an undisturbed deciduous forest (consisting mainly of beech and oak) in Germany and found no relationship between N<sub>2</sub>O and CO<sub>2</sub> fluxes.

#### 4.4.2 Short-term effect of clearfelling on the soil CH<sub>4</sub> fluxes

The two mature 40-years-old (CF<sub>before</sub> and 40-yr) stands exhibited different patterns in the CH<sub>4</sub> fluxes, with the 40-yr stand being a net source of CH<sub>4</sub> and the CF<sub>before</sub> stand being a net sink, although the soil temperature, soil water content and soil C were very similar. CF<sub>before</sub> showed emissions only during one sampling day (15/01/02) which were significantly lower than the 40-yr emissions ( $0.02 \pm 0.04 \text{ mg m}^{-2} \text{ d}^{-1}$  and  $0.13 \pm 0.07 \text{ mg m}^{-2} \text{ d}^{-1}$ , respectively,  $P=0.03$ ). On three sampling days (9/10/01, 1/11/01 and 12/12/01) CH<sub>4</sub> consumption was observed in the 40-yr stand, but it was 2 to 5 times lower than in the CF<sub>before</sub> stand ( $P=0.002$ ). This difference could be due to inherent site fertility. Castro *et al.* (1995) measured CH<sub>4</sub> consumption rates simultaneously in red pine (*Pinus resinosa* Ait.) forests in Massachusetts in two different sites with the same soil type, same temperature and identical soil water content. One of the two sites had 3 times higher CH<sub>4</sub> consumption rates than the other.

Clearfelling turned the stand from a net sink of CH<sub>4</sub> to a net source. The same effect was only reported by Castro *et al.* (2000) for two slash pine plantations in Florida. Before clearfelling they were CH<sub>4</sub> sinks with average consumption rates ranging from  $-0.5$  to  $-2.6 \text{ mg CH}_4 \text{ m}^{-2} \text{ d}^{-1}$ , while after clearfelling they became a CH<sub>4</sub> source with fluxes ranging from  $3$  to  $11 \text{ mg CH}_4 \text{ m}^{-2} \text{ d}^{-1}$ . No other studies were found that reported this change of direction in the soil CH<sub>4</sub> flux, from a sink before clearfelling to a source after clearfelling; published studies on the effects of clearfelling on soil CH<sub>4</sub> have, however, generally shown that clearfelling reduces CH<sub>4</sub> consumption. Bradford *et al.* (2000) observed reduced oxidation of CH<sub>4</sub> in the soil of clearfelled temperate forests (beech, Japanese larch and oak) in the UK. Keller *et al.* (1986) reported a slight decrease in the CH<sub>4</sub> consumption rates in a clearfelled plot relative to an adjacent uncut *terra firme* forest in the Amazon ( $20.1$  and  $26 \text{ kg CH}_4 \text{ ha}^{-1} \text{ y}^{-1}$ , respectively). Steudler *et al.* (1991) reported that clearfelling a wet tropical forest in Puerto Rico reduced CH<sub>4</sub> uptake rate by 50% to  $0.9 \text{ kg CH}_4 \text{ ha}^{-1} \text{ y}^{-1}$  compared with

the  $1.8 \text{ kg ha}^{-1} \text{ y}^{-1}$  of the reference plot. The change of direction of  $\text{CH}_4$  fluxes after clearfelling in this study could be because of the increase of available substrate and N. Measurements at Harwood in a nearby clearfelled and replanted site showed that available soil nitrogen was significantly higher than in the 40-yr stand (Cullen, 2000). Castro *et al.* (2000) observed that clearfelling of slash pine plantations in Florida increased the  $\text{NH}_4$  and  $\text{NO}_3$  pools; however, they found no correlation between soil N pools and  $\text{CH}_4$  fluxes.

During the first ten months after clearfelling, the uncut control stand (40-yr) emitted  $0.40 \pm 0.02 \text{ kg CH}_4\text{-C ha}^{-1}$ , while during the same period the emissions from the clearfelled site were  $4.8 \pm 3.5 \text{ kg CH}_4\text{-C ha}^{-1}$ . These values contrast with the generally observed net uptake in temperate forests. For example, Ambus and Christensen (1995) reported an annual  $\text{CH}_4$  uptake of  $0.91 \text{ kg CH}_4\text{-C ha}^{-1}$  in spruce forests (*Picea abies* L.) in Denmark, and Dong *et al.* (1998) reported an uptake of  $4.8 \text{ kg CH}_4\text{-C ha}^{-1} \text{ y}^{-1}$  in a deciduous forest soil in Germany. However, all such reported uptake rates are derived from well drained and aerated soils with low soil organic matter, which is not the case in Harwood. Also, Yavitt *et al.* (1990) observed that although temperate forest soils in the Appalachians were sinks of  $\text{CH}_4$ , they were also sources seasonally. This trend in  $\text{CH}_4$  fluxes was also observed at Harwood, mainly in the  $\text{CF}_{\text{before}}$  stand. The 40-yr stand was mainly a low source of  $\text{CH}_4$  and occasionally a sink, although as was mentioned before, the sink strength was much lower than that of the  $\text{CF}_{\text{before}}$  stand.

$\text{CH}_4$  fluxes in the CF site exhibited large spatial variability, as is indicated by the large standard errors of the mean. This suggests that production of  $\text{CH}_4$  occurs in microsites where anaerobic conditions occur and methanogens are active (Schütz *et al.*, 1990).

Soil  $\text{CH}_4$  fluxes in the 40-yr stand were significantly, although weakly, correlated with soil temperature at 1 and 5 cm depth ( $R^2 = 0.18$  and  $0.14$ , respectively,  $P < 0.05$ )

but not significantly correlated with soil temperature at 10 cm depth ( $R^2=0.13$ ,  $P=0.6$ ), indicating that the emissions were probably dependent on the substrate available, rather than abiotic factors. Also, the lack of a strong relationship with temperature could also be because of the possibility of the presence of methanogens and methanotrophs at a greater depth in the soil profile than where the temperature was measured. Yavitt *et al.* (1990) found no significant correlation between  $\text{CH}_4$  fluxes and soil temperature at 2 and 10 cm depth ( $R=0.31$ ,  $P>0.05$ ) in forest soils in the Appalachians.

Before clearfelling, in the  $\text{CF}_{\text{before}}$  stand,  $\text{CH}_4$  uptake was significantly related to soil temperature at 5 cm depth with a linear relationship ( $R^2=0.50$ ,  $P=0.03$ ). Although soil  $\text{CH}_4$  oxidation is generally thought to be less temperature dependent than  $\text{CH}_4$  production (Dunfield *et al.*, 1993), here the opposite was observed. Castro *et al.* (1995) observed that temperature was a strong controller of soil  $\text{CH}_4$  consumption at low temperatures, with consumption increasing from  $0 \text{ mg CH}_4 \text{ m}^{-2} \text{ d}^{-1}$  at  $-5 \text{ }^\circ\text{C}$  to  $3.84 \text{ mg CH}_4 \text{ m}^{-2} \text{ d}^{-1}$  at  $10 \text{ }^\circ\text{C}$ , but then becoming independent of soil temperature between 10 and  $20 \text{ }^\circ\text{C}$ . Maljanen *et al.* (2003) measured  $\text{CH}_4$  flux in a birch (*Betula pendula*) forest soil in Finland and found that  $\text{CH}_4$  uptake was correlated positively with soil temperature ( $R=0.31$  at 5 cm depth). Dong *et al.* (1998) found higher  $\text{CH}_4$  uptake with increasing temperature ( $R=0.50$ ), in a deciduous forest (consisting mainly from beech and oak) in Germany.

After clearfelling, there was a significant linear relationship of soil  $\text{CH}_4$  emissions with soil temperature; however, the relationship was weak ( $R^2=0.19$  to  $0.22$ ,  $P<0.05$ ). Sjogersten and Wookey (2002) measured  $\text{CH}_4$  fluxes in forest (*Betula pubescens*)-tundra ecotones in Sweden and Norway and they found no significant relationship between  $\text{CH}_4$  fluxes and temperature. Also, the temperature range in Harwood is below the general optimum for  $\text{CH}_4$  production or consumption. Dunfield *et al.* (1993) studied the effect of temperature on peaty soils from Ontario and Québec in laboratory conditions. They observed that  $\text{CH}_4$  production rates showed a marked

dependence on temperature, with optima in the region of 25–30 °C and extremely low activities in the low temperature range 0-15 °C, which is similar to the range of soil temperatures in Harwood. They also observed that CH<sub>4</sub> consumption was optimal in the range 20-25 °C, although the temperature dependence was much lower than that of CH<sub>4</sub> production.

A weak linear relationship ( $R^2 = 0.20$ ,  $P > 0.05$ ) was observed between CH<sub>4</sub> uptake and the depth of the water table in the CF<sub>before</sub> stand, while no relationship was found either between CH<sub>4</sub> uptake or CH<sub>4</sub> fluxes and the depth of the water table in the 40-yr stand. This is similar to the results of Yavitt *et al.* (1990), who found no significant correlation between CH<sub>4</sub> fluxes and soil water table depth ( $R = 0.15$ ,  $P > 0.05$ ) in forest soils in the Appalachians.

CH<sub>4</sub> emissions were linearly related ( $R^2 = 0.37$ ,  $P = 0.01$ ) with the water table depth in the CF site, and increased when the water table fell from 12 cm to 25 cm below the soil surface, although the opposite generally occurs in wetland environments (Liblik *et al.*, 1997). One possible explanation is that the positive effect of temperature on CH<sub>4</sub> production and emissions was greater than any corresponding loss in CH<sub>4</sub> production resulting from previously waterlogged peat layers, causing to be a source of CH<sub>4</sub> as the water table went down. However, same trend of CH<sub>4</sub> emissions was observed by Bellisario *et al.* (1999) who found that CH<sub>4</sub> emissions decreased as the water table rose from 14 cm below the soil surface to 1 cm above the soil surface ( $R^2 = 0.22$ ,  $P = 0.02$ ) in peatlands in Canada. Maljanen *et al.* (2001) measured CH<sub>4</sub> fluxes in afforested peatlands (with 1 year old birch stand, 6 and 23 years old pine stands) in Finland and the highest emission of 154 mg m<sup>-2</sup> d<sup>-1</sup>, was measured in the birch stand and the 6-year old pine stand in the middle of August during a warm and dry period when the depth to the water table was at its greatest.

Soil CH<sub>4</sub> flux was not correlated with soil water content in either of the mature stands or after clearfelling. A possible explanation is that CH<sub>4</sub> fluxes were a result of

production in a greater depth than the top 5 cm, where the soil water content was measured. Castro *et al.* (2000) found relatively strong exponential relationships ( $R^2=0.66$  and  $0.71$ ) for two slash pine plantations in Florida that were clearfelled. The relationships described  $\text{CH}_4$  with soil water content before clearfelling, when the stands were a sink for  $\text{CH}_4$  and after clearfelling when they became a source. Maljanen *et al.* (2003) measured  $\text{CH}_4$  flux in a birch (*Betula pendula*) forest soil in Finland and found that  $\text{CH}_4$  did not correlate with soil water content ( $R=-0.01$ ). Sitaula *et al.* (1995) in a lysimeter experiment on soil from a Scots pine forest in Norway, found that  $\text{CH}_4$  uptake decreased when the soil water content increased from  $0.32$  to  $0.45 \text{ cm}^3 \text{ cm}^{-3}$  soil. Sjoergersten and Wookey (2002) measured  $\text{CH}_4$  fluxes in forest (*Betula pubescens*)-tundra ecotones in Sweden and Norway and they found no significant relationship between  $\text{CH}_4$  flux and water content. Their mean water content also ranged between  $0.20$  to  $0.30 \text{ cm}^3 \text{ cm}^{-3}$  soil, very similar to the water content in the sites in this study. They attributed this to the insensitivity of methanotrophic bacteria to these levels of soil water content, with small variations having no effect.

The relationship between  $\text{CH}_4$  and  $\text{N}_2\text{O}$  emissions in the CF site suggests that clearfelling caused changes that favoured the two gases in common. The rise of the water table created anaerobic conditions, which could increase denitrification rates and methanogenesis. Also, tree harvesting must have caused changes in the N cycle and increased the availability of labile organic material, which influences both gases. A relationship between  $\text{N}_2\text{O}$  emissions and  $\text{CH}_4$  has been often observed by other researchers and it is usually between  $\text{N}_2\text{O}$  emissions and  $\text{CH}_4$  uptake. Bradford *et al.* (2000) observed a significant negative correlation between soil  $\text{N}_2\text{O}$  emissions and  $\text{CH}_4$  uptake ( $R^2=-0.60$ ) after clearfelling. Sitaula and Bakken (1993) found a significant inverse relationship ( $P<0.01$ ) between  $\text{CH}_4$  uptake and  $\text{N}_2\text{O}$  emissions in a 63-years-old Norway spruce forest in Norway. They attributed that to the changes in the N cycle after clearfelling and the availability of  $\text{NH}_4$ . Keller *et al.* (1986) also found an inverse relationship between  $\text{CH}_4$  uptake and  $\text{N}_2\text{O}$  emissions in tropical

forest soils. However, Dong *et al.* (1998) found no relationship between CH<sub>4</sub> and N<sub>2</sub>O fluxes, in a mature (80-years-old) deciduous forest (consisting mainly of beech and oak) in Germany. In this study also, no relationship was observed between soil N<sub>2</sub>O and CH<sub>4</sub> fluxes in the two mature stands, probably to low N availability.

In the clearfelled site, a significant linear relationship ( $R^2=0.45$ ,  $P=0.0004$ ) between CH<sub>4</sub> and CO<sub>2</sub> fluxes was observed, indicating that both gases are partially controlled by common factors, such as the availability of organic substrate and microbial activity. Moore and Dalva (1997) incubated peatland soil and measured CH<sub>4</sub> and CO<sub>2</sub> production potentials. They found that CH<sub>4</sub> production rates were positively correlated with CO<sub>2</sub> production rates ( $R= 0.38$ ). They attributed the relationship to be a function of the overall microbial activity. Dong *et al.* (1998) observed a positive correlation ( $R= 0.49$ ) between CH<sub>4</sub> and CO<sub>2</sub> production, in a deciduous forest (consisting mainly from beech and oak) in Germany.

## 4.5 Conclusions

Though soil N<sub>2</sub>O fluxes for the first ten months after clearfelling were much greater in the CF site than in the uncut 40-yr stand, it cannot be said with certainty whether clearfelling was the main cause of these high fluxes, since the two stands prior to clearfelling exhibited different patterns in soil N<sub>2</sub>O fluxes. And since N<sub>2</sub>O can be produced under aerobic and anaerobic conditions the situation is even less clear. However, it seems that clear felling had some effect on soil N<sub>2</sub>O fluxes, through their dependence on soil temperature and the higher availability of organic substrate.

Measurements of N<sub>2</sub>O over several years would be required in order to get robust estimates of annual fluxes of N<sub>2</sub>O and to be able to distinguish differences between different sites and the impact of a disturbance effect such as clearfelling.

Clearfelling had a much more distinct effect on soil CH<sub>4</sub> fluxes. It caused an increase in soil CH<sub>4</sub> emissions and turned the site from a net sink to a net source of CH<sub>4</sub>. This change can be attributed to the rise of the water table, as CH<sub>4</sub> is produced under anaerobic conditions, and thus higher emissions will result from wetter soils, as well as from the availability of more labile organic matter from log residues and small brash left in the site and dying roots. The change in the soil temperature did not seem to have any important effect on the soil CH<sub>4</sub> fluxes.

Moderate relationships were observed between CO<sub>2</sub>, N<sub>2</sub>O and CH<sub>4</sub> fluxes in the CF site only, indicating that, overall, clearfelling had an effect on the emission of these trace gases. This happened by stimulating common factors influencing these trace gases, such as organic substrate availability and microbial activity, and their interactions with changes in the water table and temperature regimes.