

Effects of elevated CO₂ and N deposition on CH₄ emissions from European mires

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[1] Methane fluxes were measured at five sites representing oligotrophic peatlands along a European transect. Five study plots were subjected to elevated CO₂ concentration (560 ppm), and five plots to NH₄NO₃ (3 or 5 g N yr⁻¹). The CH₄ emissions from the control plots correlated in most cases with the soil temperatures. The depth of the water table, the pH, and the DOC, N and SO₄ concentrations were only weakly correlated with the CH₄ emissions. The elevated CO₂ treatment gave nonsignificantly higher CH₄ emissions at three sites and lower at two sites. The N treatment resulted in higher methane emissions at three sites (nonsignificant). At one site, the CH₄ fluxes of the N-treatment plots were significantly lower than those of the control plots. These results were not in agreement with our hypotheses, nor with the results obtained in some earlier studies. However, the results are consistent with the results of the vegetation analyses, which showed no significant treatment effects on species relationships or biomass production.

INDEX TERMS: 1055 Geochemistry: Organic geochemistry; 1040 Geochemistry: Isotopic composition/chemistry; 0330 Atmospheric Composition and Structure: Geochemical cycles; 0322 Atmospheric Composition and Structure: Constituent sources and sinks; **KEYWORDS:** elevated CO₂, N deposition, methane, European peatlands

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1. Introduction

[2] Peatlands are generally considered to be sinks for CO₂ and sources for CH₄ [Gorham, 1991; Bartlett and Harriss, 1993]. The fluxes and balances of these two biogenic greenhouse gases are tightly interconnected in the biospheric carbon cycle [Mosier, 1998]. The climatic net effect of the fluxes from an individual mire or peatland area may vary from warming to cooling depending on the mire type, weather conditions and timescale under consideration

[Korhola *et al.*, 1996; Laine *et al.*, 1996]. During the industrial period, the greatest increase in biogenic greenhouse gases has been in the concentration of methane, ~145%. In the anticipated climatic change, the contribution of methane has been estimated to be ~25% [Houghton *et al.*, 1996]. The level of global methane emissions to the atmosphere is now about 540 Tg yr⁻¹, of which ~40% originates from soils, and half of this is emitted from natural wetlands [Mosier, 1998]. The emission estimate for northern peatlands is about 40 Tg CH₄ yr⁻¹ [Bartlett and Harriss, 1993].

[3] The characteristics of peatlands vary due to regional and local environmental factors [Overbeck, 1975; Gore, 1983; Succow and Jeschke, 1990], and these differences are also reflected in their methane emissions. Important regulating factors are temperature, the water level and nutrient status [Moore and Dalva, 1993; Nykänen *et al.*, 1998; Bellisario *et al.*, 1999]. The rate of primary production and CH₄ fluxes have been shown to correlate across different types of wetland [Whiting and Chanton, 1993]. In recent decades the human impact on wetlands has been very strong, especially in densely populated areas [Bragg *et al.*, 1992]. In western and central Europe, for instance, only a small proportion of the peatlands (originally millions of hectares) are still in a natural state [Succow and Jeschke, 1990]. Various human activities have either increased soil CH₄ emissions (e.g., rice paddies [Cole *et al.*, 1996]) or decreased them (e.g., as a result of peatland drainage

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[Roulet and Moore, 1995; Nykänen et al. [1998]]. Some drained wetlands have been restored, and the development of a mire vegetation and peat accumulation have started on top of the old dry peat [Grosvernier et al., 1995, 1997; Tuittila et al., 2000b]. However, the reestablishment of biogeochemical soil processes takes a considerable period of time, and restored drained areas may therefore have lower CH₄ emissions even though the water table is at the same depth as that in corresponding virgin sites [Nykänen et al., 1998; Tuittila et al., 2000a].

[4] Global change can affect methane emissions in several ways. Climate warming may increase methanogenesis, but the reduction in soil moisture predicted by some scenarios would nevertheless depress emissions [Manabe and Wetherald, 1986; Mitchell, 1989; Roulet et al., 1992]. Increased nitrogen deposition may cause a shift toward more fertile mire types, with higher CH₄ emissions, while in some sites it may lead to a decrease in CH₄ production or oxidation [Crill et al., 1994; Aerts and de Caluwe, 1999].

[5] Elevated atmospheric CO₂ has been shown to increase photosynthesis and plant biomass production in the short term [Eamus and Jarvis, 1989], but less so in the longer term [Tissue and Oechel, 1987; Jauhiainen and Silvola, 1999]. In addition to the peaty organic matter itself, fresh plant litter and root exudates are important substrates for methane production in peatlands [Schütz et al., 1991]. Thus the link between the atmospheric CO₂ concentration and methanogenesis via photosynthesis is theoretically obvious. In some recent experiments a doubling of the CO₂ concentration increased methane emissions by as much as 145% [Hutchinson et al., 1995]. A decrease in methane oxidation has also been observed at elevated CO₂ concentration [Ineson et al., 1998]. As concluded by Magonigal and Schlesinger [1997], this would mean a dramatic increase in the CH₄ load, since soils account for ~40% of globally emitted CH₄, and soil methane oxidation plays an important role in the global CH₄ balance [Whalen and Reeburgh, 1990; Mosier, 1998]. Saarnio et al. [1998], in contrast, found only a small effect of doubled CO₂ concentrations on methane emission from mire microcosms kept at temperatures prevailing in boreal mires.

[6] In order to test the effects of elevated CO₂ concentration and nitrogen deposition on methane emissions from mires in real field conditions, methane fluxes were measured in five experimental sites representing mires along a European transect.

[7] Our hypotheses were that (1) an elevated CO₂ concentration increases methane emissions; (2) the CO₂ effect increases with increasing temperature; and (3) the effect of N application on CH₄ emissions depends on the nutrient status and level of N deposition on peatlands. We tested these hypotheses in the BERI project [Hoosbeek et al., 1996], in which we studied the responses of mire vegetation and biogeochemical soil processes to elevated CO₂ and N during three successive growing seasons.

2. Material and Methods

2.1. Study Sites

[8] Experimental sites were established in Switzerland (CH), Finland (FIN), the Netherlands (NL), Sweden (SE)

and the United Kingdom (UK) (Figure 1). The aim was to find as similar sites as possible in all the countries: an ombrotrophic to slightly minerotrophic mire habitat with a vegetation cover of *Sphagnum magellanicum*/*S. papillosum* and *Eriophorum vaginatum*/*E. angustifolium*. For practical reasons (electricity and CO₂ supply), the Dutch group had to use peat + vegetation sods, transported from the mire site and installed outdoors close to the laboratory. The locations and some physical and chemical characteristics of the sites are shown in Table 1.

2.1.1. Swiss Site (CH)

[9] The Swiss site (La Chau-des-Breuleux) is situated in the Jura in the subalpine zone. The mire has developed on impermeable marl deposits in the bottom of a shallow valley. The site is mostly open with some scattered trees (*Pinus uncinata*, *Picea abies*, *Betula carpatica*). The mire is mainly ombrotrophic, with some minerotrophic influences from the surrounding pasture. The mire has been drained and peat was mined up until the end of World War II. Since then, vegetation has reestablished itself through natural regeneration [Grosvernier et al., 1995], and a layer of young, poorly decomposed peat about 30 cm thick has formed. A mosaic of lawn, hummocks and hollows has now developed well with *Eriophorum vaginatum*, *Carex nigra*, *Calluna vulgaris* and *Vaccinium oxycoccus*. The dominant mosses are *Sphagnum fallax* and *Polytrichum strictum*, with *P. commune* and *Aulacomnium palustre* in places.

2.1.2. Finnish Site (FIN)

[10] The site is situated in the Salmisuo mire, near Ilomantsi, within the transition region of the southern and middle boreal zone. The Salmisuo mire complex is an eccentric bog with some minerotrophic strips, and the site is located in the northernmost, slightly minerotrophic part of the mire. According to the Finnish mire classification system, the site is a low-sedge, *Sphagnum papillosum* pine fen consisting of hummock, lawn and hollow microsites [Ruuhijärvi, 1983]. The site is almost open, with only a few small scattered pines (*Pinus sylvestris*) on hummocks. All the study plots are situated in the lawn surface, which represents intermediate moisture conditions. Dominating moss species are *Sphagnum papillosum*, *S. balticum*, *S. angustifolium* and *S. magellanicum*, and the most important vascular plants *Eriophorum vaginatum*, *Carex pauciflora*, *C. lasiocarpa*, *Andromeda polifolia* and *Vaccinium oxycoccus*. The site has been described in more detail by Saarnio et al. [1997].

2.1.3. Dutch Site (NL)

[11] The peat sods that were used in Wageningen originated from the Dwingeloo State Forestry, in the province of Drenthe in the north of the Netherlands. The peat in this small mire (<1 ha) has developed in a depression formed by a late glacial brook system. Up to 1955 the site was used for peat cutting and, for this reason, the vegetation nowadays consists of a mosaic of secondary successional stages. The peat sods consist of *Sphagnum magellanicum* rich carpets, the other mosses being *S. papillosum*, *S. fallax*, *Aulacomnium palustre* and *Calliergon stramineum*. The dominant vascular plants are *Erica tetralix*, *Vaccinium oxycoccus*, *Eriophorum angustifolium* and *Drosera rotundifolia*. The



Figure 1. The location of the experimental sites: 1 = SE, 2 = FIN, 3 = UK, 4 = NL, 5 = CH.

mire and experimental plots have been described in more detail by Heijmans *et al.* [2001].

2.1.4. Swedish Site (SE)

[12] The Swedish site (Kopparås mire) is situated in the boreonemoral zone in the middle of the province of Småland in southern Sweden. The underlying acidic mineral soil material is derived mainly from gneiss. The mire is largely open, with scattered *Pinus sylvestris* trees. The mire is mostly minerotrophic, with sporadic smaller ombrotrophic areas, and its surface slopes slightly toward the SSW. All the study plots are situated in a lawn habitat dominated by *Eriophorum angustifolium*, *Calluna vulgaris*, *Andromeda polifolia*, *Narthecium ossifragum* and *Scirpus caespitosus*. The predominant peat mosses are *Sphagnum magellanicum*, *S. papillosum* and *S. rubellum*.

2.1.5. British Site (U.K.)

[13] The British site is situated in the Roudsea Wood National Nature Reserve, North West of England, on the northern edge of Morecambe Bay. The reserve contains an extensive area of open ombrotrophic raised bog that has been partially drained in the past. The drainage ditches have been blocked for over a decade and the mire surface is gradually recovering. The vegetation corresponds to an M18 *Erica tetralix* - *Sphagnum papillosum* raised mire of the UK National Vegetation Classification [Rodwell *et al.*, 1991]. *Sphagnum papillosum* is the predominant *Sphagnum* species in the study area. The lawn also contains *Eriophorum vaginatum*, *Calluna vulgaris*, *Andromeda polifolia* and *Erica tetralix* as major species, with small amounts of

Table 1. Location, Climatic Data and Physical and Chemical Characteristics of the Study Sites

Site	Location	Altitude, m Above Sea Level	Precipitation Annual, mm	Temperature Daily		Snow Cover, Days	Peat Depth, m	pH	DOC, mg L ⁻¹	NO ₃ , mg L ⁻¹	NH ₄ , mg L ⁻¹	SO ₄ , mg L ⁻¹	Cl, mg L ⁻¹
				Mean of Warmest and Coldest Month, °C	Coldest Month, °C								
FIN	62°47'N, 30°56'E	150	650	16, -10	16, -10	150-200	1.5	4.43	18.08	0.38	0.00	0.03	0.00
SE	57°08'N, 14°30'E	225	800	16, -2	16, -2	80-120	2	4.73	21.55	1.47	0.03	21.51	8.43
UK	54°14'N, 03°01'W	5	1800	13, 1	13, 1	<20	>2	4.47	41.34	1.55	0.46	16.6	11.64
NL	52°49'N, 06°26'E	13	750	18, 3	18, 3	<10	1	4.72	39.09	0.13	0.92	0.00	7.98
CH	47°13'N, 07°03'E	1000	1390	15, -5	15, -5	80-120	1.5	4.8	15.38	0.18	0.00	0.01	1.21

Scirpus caespitosus, *Rhynchospora alba*, *Eriophorum angustifolium* and *Narthecium ossifragum*.

2.2. Experimental Design

[14] Twenty circular plots (diameter 1 m) were selected to represent lawn surface with homogeneous vegetation. The plots for the four different treatments with five replicates were selected randomly in blocks, apart from the NL site where 20 monoliths were randomly assigned to the different treatments without blocking. The treatments were CO₂ enrichment (CO₂), an ambient air control for the CO₂ treatment (AIR), nitrogen addition (NITROGEN) and a control for the nitrogen treatment (CONTROL). At the FIN and SE sites every block included one plot of each treatment, the distance between the blocks being 5–20 m. At the UK and CH sites, the plots for the CO₂ and NITROGEN treatments with their controls were located in two separate areas with minor ecological differences (e.g., in the water table). At the NL site the peat + vegetation monoliths were placed in plastic containers (1.1 m diameter, 60 cm deep) inserted into the soil. At the FIN, SE and UK sites the water table of the measuring plots followed that of the surrounding mire. In the peat monoliths of the NL site, the water levels were allowed to fluctuate between 5 and 20 cm below the moss surface. The CH₄ measuring points of the CH site had 80 cm deep collars that blocked lateral drainage, resulting in a somewhat higher water level at the CH₄ measuring points compared to the surrounding mire.

[15] For the CO₂ and AIR treatments, the plots were enclosed with miniFACE rings (5-cm-diameter tube) with 70 vertical venting pipes (18 cm long). Ambient air was vented by means of fans to these rings via pipes. CO₂ from tanks was added to the air flow in the five rings in order to enhance the CO₂ concentration to 560 ppm. The CO₂ concentration was controlled by sucking sample air from the middle of the rings to an IRGA, and regulating the CO₂ supply by PC-controlled mass-flowmeters (for more detail see *Miglietta et al.* [2001]). The nitrogen plots were not enclosed within a miniFACE ring. Nitrogen was added by spraying NH₄NO₃ onto the plots every 3 weeks (snow-free period) to give a total addition of 3 g N (5 g in the NL site) m⁻² yr⁻¹. These N additions resulted in at least a doubling of the annual atmospheric N deposition, which varies from 0.4 g N m⁻² yr⁻¹ (Finland) to 3.9 g N m⁻² yr⁻¹ (the Netherlands) [*Berendse et al.*, 2001]. The CONTROL plots were sprayed with the same amount of deionized water (2000 mL) at the same time as the nitrogen treatment.

2.3. Methane Measurements

[16] The study sites were equipped with boardwalks to prevent disturbance of the soil and vegetation during sampling. At the CH, NL, SE and UK sites, permanent plastic collars (diameter 19 cm) with a groove for a water seal were installed on every plot. For methane sampling, water was added to the groove and cylindrical plexiglass chambers of the same diameter (height 28 cm) were fitted onto the collars. Four 10-mL samples were taken at intervals of 5–8 min with a plastic syringe after careful pumping to mix the air in the chamber and tubing (inner diameter 1 mm, length 1.5 m) between the chamber and syringe. The gas in

the syringe was immediately transferred to a preevacuated glass flask (9 mL).

[17] At the FIN site, 60 × 60 cm aluminium collars with a groove for a water seal were used because the other measuring systems needed collars of this shape and size. Aluminium chambers (60 × 60 × 20 cm) were fitted onto the collars and four gas samples (40 mL) were taken with syringes during the 20-min measuring period. In order to compare the performance of the different chambers, the FIN site also had small chambers similar to those in the other sites on the CONTROL plots. Glass flasks were not used since the methane concentrations in the syringe samples were analyzed within 24 hours. The samples from all the sites were analyzed with a gas chromatograph using a FI detector. The gas chromatographs used were Shimadzu 14A (FIN), Packard 428 (SE), ATI UNICAM 610 (UK), HP 5710A (NL) and Perkin-Elmer F22 (CH).

[18] The depth of the water level and the air and peat temperatures were measured at the time of sampling. Methane fluxes were measured during the snow-free period (1996–1998 at the FIN, NL and SE sites, 1997–1998 at the UK site, and only during summer 1998 at the CH site) weekly at the FIN site and approximately every 3 weeks at the other sites. Methane fluxes were calculated by linear regression of the CH₄ concentrations over time. Only rates having r² ≥ 0.9 were accepted.

2.4. Vegetation Analyses

[19] Plant species composition and abundance were measured by the point-quadrat method [*Buttler*, 1992]. A homogeneous subplot (25 × 37.5 cm) was chosen on every plot. For counting, a Plexiglas plate with 150 holes was placed over the subplot. A thin pin was pushed down through the holes and contacts with the plant species were counted. At the end of the experiment (between late July and early September 1998 depending on the site), the vascular plants on the point quadrat areas were harvested. The species were separated, dried (70°C) and weighed. The amount of living belowground biomass was determined from core samples taken from the peat. The vegetation studies have been described in more detail by *Mitchell et al.* [2000], *Berendse et al.* [2001], *Heijmans et al.* [2001] and *Hoosbeek et al.* [2001]. The cover of the most abundant plant species on the different sites is shown in Table 2. The number of shoots of *Carex* and *Eriophorum* species growing in the methane measurement collars were counted on the CH, NL and UK sites.

2.5. Nutrient Analyses

[20] For the analysis of soil water chemistry, samples were taken at the beginning of the growing season in 1996 and in spring, summer and autumn in 1997. DOC, pH and major cations and anions were analyzed at the Wageningen Agricultural University according to *Buurman et al.* [1996]. The sampling method and soil water chemistry are described in more detail by *Mitchell et al.* [2000].

2.6. Statistical Treatments

[21] Factors causing differences in CH₄ emissions among different sites were evaluated using data from the CON-

Table 2. Cover of the Most Important Vascular Plant and Moss Species and the Total Covers of Those Plant Groups in the Different Sites^a

	FIN	SE	UK	NL	CH
	<i>Vascular Plants</i>				
Total cover,%	23	40	40	13	8
Most abundant species (cover% in parentheses)	<i>Eriophorum vaginatum</i> (9)	<i>Eriophorum angustifolium</i> (18)	<i>Eriophorum vaginatum</i> (14)	<i>Erica tetralix</i> (5)	<i>Carex nigra</i> (3)
	<i>Andromeda polifolia</i> (7)	<i>Drosera rotundifolia</i> (6)	<i>Erica tetralix</i> (13)	<i>Vaccinium oxycoccos</i> (4)	<i>Eriophorum vaginatum</i> (3)
	<i>Mosses</i>				
Total cover,%	100	95	79	98	147
Most abundant species (cover% in parentheses)	<i>Sphagnum balticum</i> (63)	<i>Sphagnum magellanicum</i> (81)	<i>Sphagnum papillosum</i> (79)	<i>Sphagnum magellanicum</i> (96)	<i>Sphagnum fallax</i> (89)
	<i>Sphagnum papillosum</i> (35)	<i>Sphagnum papillosum</i> (10)		<i>Sphagnum papillosum</i> (1)	<i>Polytrichum strictum</i> (58)

^aCover percentages have been estimated by the point-intercept and classical vegetation relevés methods [Mitchell *et al.*, 2000]. At the Swiss site, *Polytrichum* overtops *Sphagnum*, resulting in a total cover of over 100%.

TROL plots. Dependence of the CH₄ flux on the water table (WT) and on temperature at a depth of 15 cm (T) was calculated by multiple regression. For the UK site, the peat temperature at a depth of 15 cm was calculated as an average of measurements made at depths of 10 and 20 cm. Q₁₀ values (10–20°C) of the log-transformed CH₄ emission were calculated in order to compare the temperature dependence at the different sites. CH₄ emissions corrected to a common temperature were used when testing the effect of certain physical and chemical characteristics at the different sites.

[22] On the basis of previous studies, elevated CO₂ was expected to influence methane emission particularly in warm climatic conditions [cf. Saarnio and Silvola, 1999; Saarnio *et al.*, 2000]. Therefore the summer time data (weeks 23–32 in 1997 and 1998) were used when testing the effects of the CO₂ and N treatments on CH₄ emission. Three covariates, soil temperature at a depth of 15 cm (T), depth of the water table (WT), and the number (CH, NL and UK sites) or biomass (FIN and SE sites) of shoots of *Carex* and *Eriophorum* species, were tested in the analysis of covariance. If any of the covariates were statistically significant, the analysis was repeated using only this (these) covariate(s). If none of the covariates were statistically significant, the effect of the treatment (CO₂ enrichment or nitrogen addition) on the CH₄ flux was tested using the t-test. In all analyses, $p < 0.05$ was used as the criterion for statistically significant effects.

3. Results

3.1. Methane Emissions in the CONTROL Plots

[23] The average CH₄ emissions (mg CH₄ m⁻² h⁻¹) of all the CONTROL plot measurements were 6.5 (NL), 4.3 (CH), 2.8 (SE), 2.5 (FIN, large chambers) and 0.45 (UK). In all the sites the CH₄ emissions were highest in the middle of summer, although especially in the UK site the annual variation was very small (Figure 2). Typically the summer time fluxes varied between 6 and 15 mg CH₄ m⁻² h⁻¹ in the NL site, 2 and 7 in the FIN, SE and CH sites and 0.5 and 1 in the UK site.

[24] The dependence of the CH₄ fluxes on soil temperature and moisture varied considerably among the sites. In the analysis of regression, WT and T (at a depth of 15 cm) were significant variables ($p < 0.001$) only in the NL and FIN sites. r^2 was 53% for WT and 57% for T in the NL site, and 4% and 41% in the FIN site, respectively. In the multiple regression, the combined r^2 of WT and T was 68% and 50% for the NL and FIN sites, respectively. The Q₁₀ values (peat temperature 10–20°C at a depth of 15 cm) calculated from linear regressions of the log-transformed CH₄ efflux of the CONTROL plots were 6.5, 3.4, 1.5 and 1.1 for the NL, FIN, SE and UK sites, respectively. In the SE and UK sites, however, temperature was not a statistically significant variable. Q₁₀ was not calculated for the CH site because of the limited number of measurements with a small temperature range (~2°C).

[25] The differences in the CH₄ flux between the sites could only be partly explained by the physical and chemical characteristics of the sites. The temperature explained 68% of the variation in the flux rates among the different sites (Figure 3a). A rise in the pH tended to increase emissions and a rise in the DOC, total N and SO₄ concentrations to decrease it. In all cases, however, r^2 was low (Figures 3b–3f).

[26] The small and large chambers gave similar results with low and medium flux rates, except that variability was much higher with small chambers (Figure 4). However, at higher flux rates smaller chambers yielded higher flux rates.

3.2. Effects of the Treatments on Methane Emissions

[27] The CH₄ emissions, temperatures and water tables with their standard errors are shown in Figure 5, and the results of the analysis of covariance in Table 3. At the CH site, CH₄ fluxes from the enhanced CO₂ treatment plots ($\bar{x} = 6.6$ mg CH₄ m⁻² h⁻¹) were 23% higher and those from the NITROGEN treatment ($\bar{x} = 4.1$ mg CH₄ m⁻² h⁻¹) 43% lower than from the corresponding control plots. None of the covariates or treatment effects were statistically significant.

[28] At the FIN site, CH₄ emission from the enhanced CO₂ plots ($\bar{x} = 4.6$ mg CH₄ m⁻² h⁻¹), as well as from the

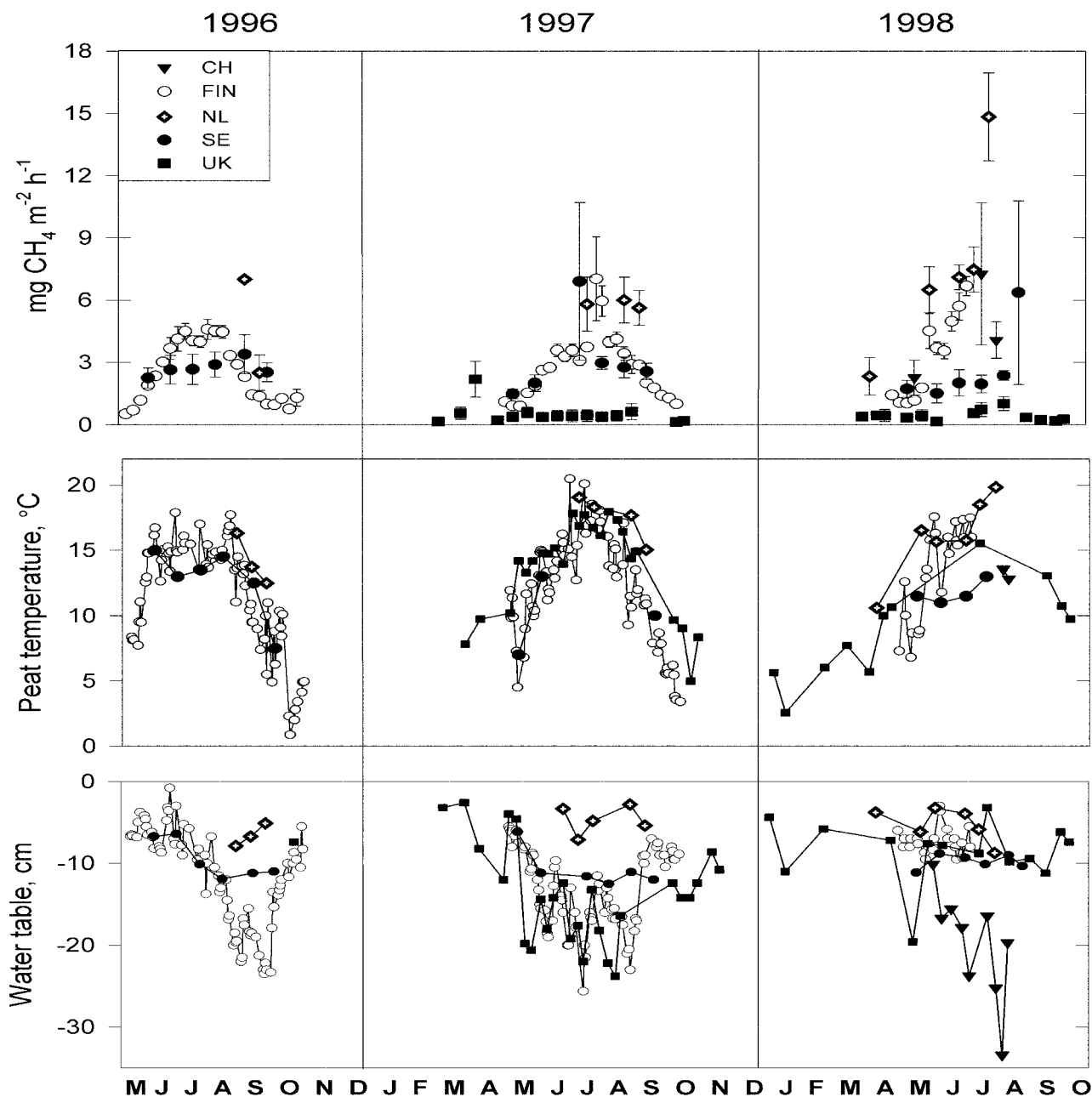


Figure 2. CH₄ fluxes, peat temperature (at a depth of 15 cm in the NL and CH sites, 10 cm in the other sites) and depth of the water table in the CONTROL plots at the different sites, calculated as weekly averages (S.E.).

NITROGEN plots ($\bar{x} = 3.5 \text{ mg CH}_4 \text{ m}^{-2} \text{ h}^{-1}$), were 14% (elevated CO₂) and 7% (N addition) higher than from the corresponding control plots. In the case of the CO₂ treatment, WT and T were significant covariates and the treatment effect was slightly above the 0.05 significance level ($p = 0.085$). In the case of the N treatment, none of the covariates and treatment effects were statistically significant.

[29] At the NL site, CH₄ emissions from the CO₂ plots ($\bar{x} = 11.0 \text{ mg CH}_4 \text{ m}^{-2} \text{ h}^{-1}$) were 4% lower and those from

the NITROGEN plots ($\bar{x} = 8.7 \text{ mg CH}_4 \text{ m}^{-2} \text{ h}^{-1}$) 28% higher than those from the corresponding controls. In the elevated CO₂ plots, the number of *Eriophorum* shoots was a significant covariate, but the treatment effect was statistically nonsignificant. In the NITROGEN plots, the treatment effect and covariates were nonsignificant.

[30] At the SE site, CH₄ emission from the CO₂ plots ($\bar{x} = 3.0 \text{ mg CH}_4 \text{ m}^{-2} \text{ h}^{-1}$) was 4% higher and that from the NITROGEN plots ($\bar{x} = 3.4 \text{ mg CH}_4 \text{ m}^{-2} \text{ h}^{-1}$) 7% higher than that from the corresponding control plots. None of the

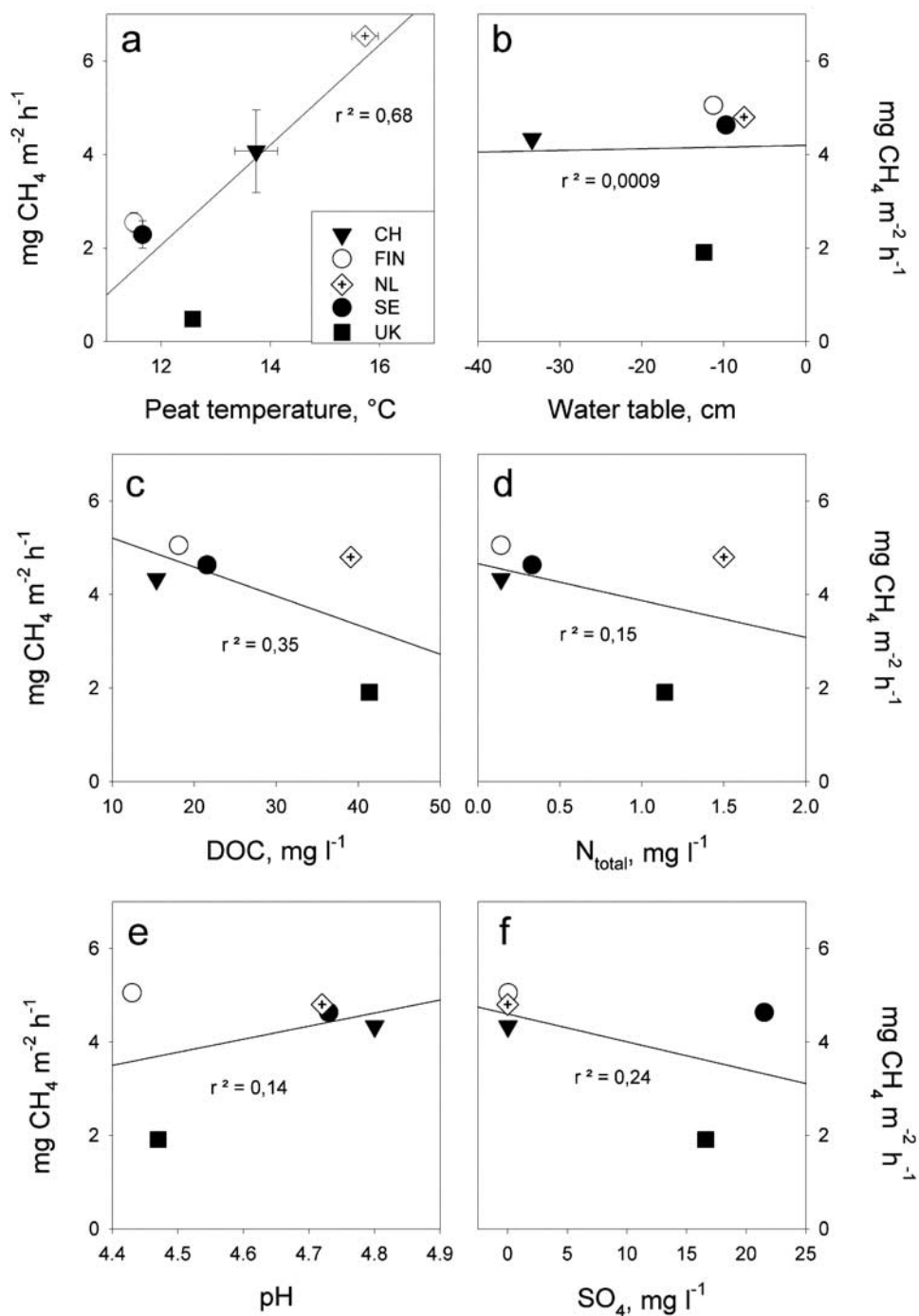


Figure 3. CH₄ fluxes of the CONTROL plots (averages of all measurements) compared with the different site characteristics. Water table depth and temperature are averages of the same data as in Figure 2. The pH, and the DOC, total N and SO₄ concentrations were measured on the soil water [Mitchell *et al.*, 2000]. CH₄ fluxes are standardized to a common reference temperature of 14°C in Figures 3b–3f.

covariates (T was not included in the test because it was not measured on a plot-specific level) or treatment effects were significant.

[31] At the UK site, CH₄ emission from the CO₂ plots ($\bar{x} = 0.99$ mg CH₄ m⁻² h⁻¹) was 33% lower and that from

the NITROGEN plots ($\bar{x} = 0.47$ mg CH₄ m⁻² h⁻¹) 2% lower than that from the corresponding control plots. The number of *Eriophorum* shoots was a significant covariate in the N experiment (T was not included in the test because it was not measured on a plot specific level). The CO₂ effect

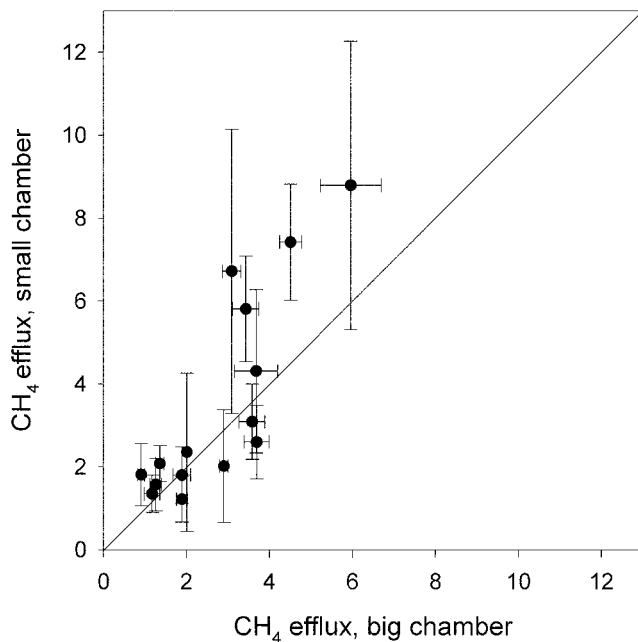


Figure 4. Comparison of the CH₄ fluxes measured simultaneously with small and large chambers in the FIN CONTROL plots.

was nonsignificant, but after the covariate correction, the treatment effect was statistically significant in the NITROGEN plots.

4. Discussion

4.1. CONTROL Plots

[32] In the FIN and SE sites, which were located in natural peatlands, the CH₄ fluxes were typical for slightly minerotrophic mires in the boreal and nemoral regions [Bubier *et al.*, 1993; Saarnio *et al.*, 1997; Nykänen *et al.*, 1998]. Because the other sites were former drained or peat-cutting areas, their CH₄ emissions are probably not fully comparable with those from natural peatlands [cf. Nykänen *et al.*, 1998; Tuittila *et al.*, 2000a]. However, they do represent the prevailing conditions in Central Europe, where most of the peatlands are more or less disturbed. Therefore our experimental design can be expected to give representative results, since both treatments were compared to control plots in corresponding conditions at every site.

[33] The Q₁₀ values of the methane fluxes varied considerably among the sites. The large variation is understandable, because methane is the end product of a series of processes. Temperature and other limiting factors can exert an effect on the individual phases. Furthermore, during the growing season when the temperature varies, other changes also occur that affect CH₄ fluxes, for example, changes in the water table and the number of vascular plant shoots. In this study the other Q₁₀ values fell within the range (1.3–28) presented in the review of Segers [1998], apart from the Q₁₀ of the UK site which was exceptionally low (1.1). The UK site seemed to be very insensitive to changes in environmental conditions (Figure 2). One possible reason

for this may be the history of the site. Also Nykänen *et al.* [1998] reported that the effects of variations in the water table or temperature on CH₄ fluxes were very small in some drainage areas. Another possibility could be a high SO₄ concentration [Bartlett *et al.*, 1987]. The UK site is located very close to the coast, and most of the marine-derived sulphur deposition in coastal areas takes place during the summer and autumn [Stuedler and Peterson, 1984]. The SE site also had a high SO₄ concentration, and the Q₁₀ value at this site was rather low (1.5). The CH₄ emissions, however, were markedly higher than at the UK site.

[34] The average CH₄ flux rates of the different sites correlated rather well with the corresponding peat temperatures (Figure 3a). In contrast, differences in water table depths did not explain the differences in the CH₄ fluxes among the study sites (Figure 3b). The water table has been shown to explain quite well the CH₄ fluxes from different habitats or from individual habitats during the seasons [Bubier *et al.*, 1993; Nykänen *et al.*, 1998]. One probable explanation for the lack of correlation between water table and emission in our study is that, apart from the CH site, the range in average water table was very small (which was deliberate, and due to the criterion applied for site selection). The explanatory power of the other physical and chemical factors was also rather low. Increasing methane emissions with increasing pH have been reported by some researchers [e.g., Dunfield *et al.*, 1993; Valentine *et al.*, 1994], but not in all cases [Moore and Knowles, 1990].

[35] Comparison of the small and large chambers shows that the sampling method has some effect on the methane flux results. There are at least two possible reasons for the differences. If bubbling occurs during sampling as a result of movements of the collar in the peat, this will have a greater effect on the CH₄ concentration in small chambers. Another reason could also be the use of transparent (small) and opaque (large) chambers. In the transparent chambers the temperature increases more during sunshine. On the other hand, if the plants close their stomata in the opaque chambers, this may decrease gas transport through the plants. However, the role played by stomata in CH₄ emissions is not clear [cf. Kelker and Chanton, 1997; Bellisario *et al.*, 1999]. Anyway, the sampling method had only a small effect on the Q₁₀ value. At the FIN site, Q₁₀ was 3.4 with large chambers and 3.7 with small chambers.

4.2. Effects of the Treatments

[36] Three of the sites had higher, and two lower CH₄ fluxes in the CO₂ plots than in their controls (AIR), but none of the differences were statistically significant. This was somewhat surprising and not in agreement with our expectations, which were based on two earlier studies showing a strong stimulation of CH₄ fluxes by elevated CO₂. Doubling of the CO₂ concentration increased methane fluxes by 80% in brackish marsh soils [Dacey *et al.*, 1994], and by 145% in cores from an ombrotrophic mire [Hutchin *et al.*, 1995]. Such effects of elevated CO₂ on methane fluxes are most probably related to the response of plants to elevated CO₂ concentrations [cf. Hutchin *et al.*, 1995; Magonigal and Schlesinger, 1997]. The CO₂ concentration in the soil is usually so high that an increase in the

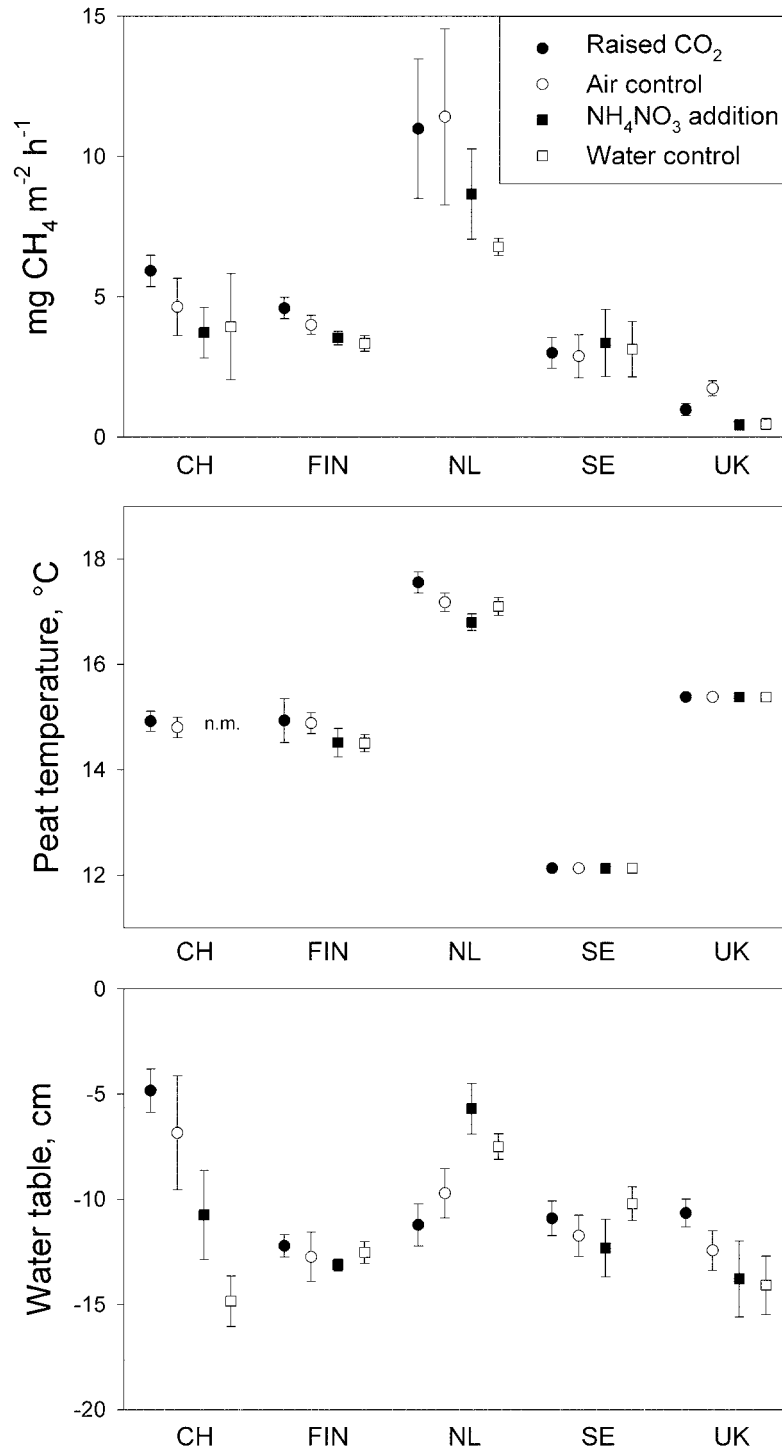


Figure 5. Average CH₄ flux, peat temperature and water table depth (mean, S.E.) in the plots with different treatments at the different sites during 15 June to 15 August in 1997 and 1998. All acceptable CH₄ results are included (compare Table 3). The temperature was measured at a depth of 10 cm, except at 15 cm in the NL and CH sites. At the SE and UK sites the temperature was measured at one point in the middle of the sites, but at the other sites in the separate plots.

atmospheric concentration from 360 to 560 is hardly likely to have any direct effects on methanogenic bacteria [cf. Wang and Adachi, 1999]. In our study, elevated CO₂ concentration had only a small effect on the plants. Under

elevated CO₂ the *Sphagnum* biomass production increased slightly in the FIN and NL sites, but it decreased in the CH and SE sites. None of these differences were statistically significant [Berendse et al., 2001]. Similarly, the above-

Table 3. Results of the Analysis of Covariance^a

	CO ₂	Air	Water Table	Peat Temperature	Shoots/Biomass	Treatment (p-Value)
CH	6.58	5.36	n.s.	n.i.	n.s.	n.s. (0.32)
FIN	4.52 ^b	4.05 ^b	0.017	0.029	n.s.	n.s. (0.09)
NL	11.15 ^b	11.25 ^b	n.s.	n.s.	0.021	n.s. (0.97)
SE	3.00	2.88	n.s.	n.i.	n.s.	n.s. (0.65)
UK	0.99	1.48	n.s.	n.i.	n.s.	n.s. (0.12)
	Nitrogen	Control	Water Table	Peat Temperature	Shoots/Biomass	Treatment (p-Value)
CH	4.12	7.27	n.s.	n.i.	n.s.	n.s. (0.45)
FIN	3.53	3.31	n.s.	n.s.	n.s.	n.s. (0.12)
NL	8.67	6.78	n.s.	n.s.	n.s.	n.s. (0.32)
SE	3.35	3.13	n.s.	n.i.	n.s.	n.s. (0.77)
UK	0.35 ^b	0.60 ^b	n.s.	n.i.	<0.0005	0.04

^aThe analysis includes only those CH₄ results for which concurrent WT and T data (if included) were available (cf. Figure 5). Abbreviations: n.s. = nonsignificant, n.i. = not included.

^bCovariance-corrected mean of the CH₄ flux.

ground biomass of the vascular plants on the CO₂ plots was higher than that on the AIR plots in the CH, FIN and NL sites, but lower in the other sites (nonsignificant [Berendse *et al.*, 2001]). The belowground biomass of the plants was, however, significantly higher (10–20%) in the CO₂ plots than in the AIR plots at the end of the experiment [Berendse *et al.*, 2001]. An increased ratio of belowground to aboveground biomass is a typical phenomenon under elevated CO₂ concentrations [cf. Eamus and Jarvis, 1989]. Theoretically this should be reflected in the CH₄ emission, but a treatment period of three growing seasons may have been too short to cause any significant effects.

[37] When Saarnio *et al.* [1998] studied peat cores in growth chambers at an elevated CO₂ concentration in conditions typical of boreal mires, the increase in methane emission was also smaller than that found in previous studies. The reason for the different results was assumed to be the difference in temperatures [Saarnio *et al.*, 1998]. In the experiment of Saarnio *et al.* [1998], the effect of CO₂ increased when the temperature was kept unnaturally high at the end of the growth experiment. In our study the different sites were therefore compared using the summer data of 1997 and 1998 (Figure 5), when any effect would have been the most obvious. The CH₄ emissions in the FIN site support the assumption concerning the role of temperature; the increase in CH₄ emission caused by elevated CO₂ was highest in the middle of summer [Saarnio *et al.*, 2000]. However, the results of this experiment do not in general support this assumption. A small increase occurred at the cooler sites (CH, FIN, SE), while at the warmest site (NL) the CO₂ plots had lower CH₄ emissions than their controls (Figure 5).

[38] Like the CO₂ treatment, the N treatment had no clear effects on CH₄ emissions. In the UK site the NITROGEN and CONTROL plots had almost the same CH₄ emission rates, but the covariate-corrected emissions were significantly lower in the NITROGEN plots. Theoretically, increased nitrogen supply can have various effects on CH₄ emissions. A possible direct negative effect is decreased methane production due to suppression by nitrate. Another possibility is inhibition of methane oxidation by the nitrogen compounds [Stuedler *et al.*, 1989; Adamsen and King, 1993; Crill *et al.*, 1994], leading to higher emissions.

Indirect effects would most likely occur via the vegetation. For example, increased biomass production may result in more substrate being available for methanogens. On the other hand, fertilization may also decrease the root/shoot ratio of the plants [Eamus and Jarvis, 1989], thereby suppressing emissions. In line with this, Granberg *et al.* [2001] found that the effect of nitrogen addition on methane emission from a similar peatland to those used in this study, was negative at high sedge cover but insignificant at low sedge cover. Interestingly, the UK site that showed a negative effect of N addition had a comparatively high abundance of sedges (Table 2). However, the SE site also had a lot of sedges but no significant N effect.

[39] In our study the N treatments decreased *Sphagnum* production significantly in the CH and NL sites and nonsignificantly in the SE and UK sites, but increased it nonsignificantly in the FIN site [Berendse *et al.*, 2001]. The aboveground biomass of the vascular plants on the NITROGEN plots was higher in the FIN, UK and NL sites but lower in the other sites, but none of these differences was significant [Heijmans *et al.*, 2001; Berendse *et al.*, 2001]. However, at the end of the experiment the standing dead and litter biomass were higher in the NITROGEN plots than in the CONTROL plots at the FIN, CH, NL and SE sites [Berendse *et al.*, 2001]. In the experiment of Aerts and de Caluwe [1999], the addition of N increased CH₄ emission on a low-fertility peat soil, but there were no effects in the high-N, eutrophic peat. In the 6-year fertilization experiment carried out by Nykänen *et al.* [2002], the addition of nitrogen increased CH₄ emission only in the poorest *Sphagnum fuscum* site. This was concluded to be due to the secondary effect of increased coverage of *Eriophorum vaginatum* in the N-treated plots. Taken together, the results of previous studies suggest that an increased N load may result in decreased methane emissions at high sedge cover [Granberg *et al.*, 2001], in increased emissions in very nutrient poor peatlands [Aerts and de Caluwe, 1999; Nykänen *et al.*, 2002], but often in small and insignificant effects [Saarnio and Silvola, 1999; Granberg *et al.*, 2001]. On the basis of this, it is understandable that the effects of increased N on methane flux were small at most of our sites.

[40] CO₂ exchange (NEE and respiration) was measured only in the FIN site, where only small treatment effects were

found. [Saarnio *et al.*, 2003]. No treatment effects were found when the decomposition rates of peat samples from the FIN site were measured in the laboratory after the experiment (data not presented). However, indirect evidence for a CO₂ effect on belowground processes came from the observed increase in bacterial biomass in the living *Sphagnum* at all five sites [Mitchell *et al.*, 2003].

[41] In conclusion, an increased CO₂ and N supply for 3 years had little or no effects on the abundance of the plant species, biomass production or on the biological and biogeochemical processes of the plants and soil. This minor response of the vegetation probably explains why the treatments had no clear effects on the methane emissions. Thus it appears that elevated CO₂ concentrations do not cause fast, dramatic increases in methane emissions in boreal and nemoral peatlands. However, the methane emissions clearly depend on the temperature, moisture, nutrient status and vegetation of peatlands [e.g., Segers, 1998]. An increase in nutrient deposition and greenhouse gases would very probably cause changes in these regulatory factors in the long term. Longer time periods and larger changes in the vegetation are probably needed to bring about significant changes in CH₄ fluxes [cf. Verville *et al.*, 1998]. At the NL site, for instance, changes in the biomass ratio of vascular plants and mosses occurred on the NITROGEN plots in the third year especially [Heijmans *et al.*, 2001]. Thus experiments of longer duration are needed.

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