Methane emissions from grasslands

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Methane emissions from grasslands

Agnes van den Pol-van Dasselaar

Proefschrift

ter verkrijging van de graad van doctor
op gezag van de rector magnificus
van de Landbouwuniversiteit Wageningen,
dr. C.M. Karssen,
in het openbaar te verdedigen
op woensdag 16 september 1998
des namiddags te vier uur in de Aula.

This study was carried out at the Department of Environmental Sciences (Subdepartment Soil Science and Plant Nutrition) of Wageningen Agricultural University. It was one of the four projects of an integrated "CH_4 grassland project". This integrated project aimed to understand and quantify CH_4 formation and consumption in grasslands on peat soils, and the resulting net CH_4 emissions between grassland and atmosphere (Segers & van Dasselaar, 1995). Two of the four projects focused on the processes CH_4 production (Kengen & Stams, 1995; Wageningen Agricultural University, former Department of Microbiology) and CH_4 consumption (Heipieper & de Bont, 1995; Wageningen Agricultural University, former Department of Industrial Microbiology). In a fourth project, CH_4 production and CH_4 consumption were related to net CH_4 emissions via process based models (Segers & Leffelaar, 1996; Wageningen Agricultural University, former Department of Theoretical Production Ecology).

The present study was also one of three projects of an integrated "N_2O and CH_4 grassland project" (Langeveld et al., 1997). The other two projects were carried out at the former department of Theoretical Production Ecology of Wageningen Agricultural University. In these two projects, simulation models of N_2O and CH_4 emissions were developed (Langeveld & Leffelaar, 1996; Segers & Leffelaar, 1996).

Furthermore, the present study contributed to the EU-project "GEFOS": "Greenhouse gas emissions from farmed organic soils", which aimed to estimate the emissions of CO_2, CH_4 and N_2O from farmed organic soils and to develop options to mitigate these emissions (Klemedtsson et al., 1998).

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Stellingen

1. Als intensief beheerde, ontwaterde veengraslanden worden omgezet in extensief beheerde, natte graslanden zal de methaanemissie uit deze graslanden aanzienlijk toenemen.
   Dit proefschrift.

2. Bij onderzoek naar broeikasgasemissies is de toepassing van geostatistische technieken op perceelsniveau niet zinvol, omdat variogrammen van broeikasgasemissies binnen een dag sterk kunnen veranderen.
   Dit proefschrift.

3. In Nederland is het effect van stikstofbemesting en het verschil tussen beweiden en maaien op methaanemissie uit graslanden te verwaarlozen.
   Dit proefschrift.

4. Ontwatering is het enige instrument op het gebied van graslandbeheer waarmee methaanemissie uit graslanden in Nederland verminderd kan worden.
   Dit proefschrift.

5. Recente schattingen van methaanemissies (Van Amstel et al., 1993) veronderstellen dat ontwaterde veengraslanden een belangrijke bron van methaan zijn; deze graslanden nemen echter methaan op.
   Van Amstel AR, Swart RJ, Krol MS, Beck JP, Bouwman AF & van der Hoek KW (1993) Methane, the other greenhouse gas. Research and policy in the Netherlands. National institute of public health and environmental protection, Bilthoven.
   Dit proefschrift.

6. Een deel van het energieverbruik bij dierlijke produktie moet als energiewaarde aan dierlijke mest toegekend worden, omdat dierlijke mest een waardevolle bron van nutriënten en organische stof voor de bemesting van bouw- en grasland is.

7. De huidige schadedrempels voor bestrijding van emelten in grasland zijn te laag. Bij deze aantallen emelten zijn de kosten van bestrijding gemiddeld *ƒ*50,- per ha hoger dan het verlies door schade.

8. Indien onze maatschappij van het landbouwbedrijfsleven natuurontwikkeling verlangt, zal zij ook bereid moeten zijn om de landbouwers financieel te compenseren.

9. Een *uitkering op basis van de Wet Arbeidsongeschiktheidsverzekering* Zelfstandigen (WAZ) zou gebaseerd moeten zijn op kosten voor vervanging in plaats van op inkomensderving.

10. Agenda 2000, het voorstel van de Europese Commissie voor overgang van prijsondersteuning naar inkomenssteun, leidt tot een vermindering van bedrijfsontwikkeling in de landbouw.

11. Meer geld voor de gezondheidszorg is van levensbelang.

12. Ecologische melkveehouderij en weidevogelbeheer vormen geen optimale combinatie.

13. Door de invoering van MINAS is optimalisering van bemesting van grasland en maïs op gewasniveau niet langer voldoende, maar dient optimalisering op bedrijfsniveau plaats te vinden.

14. Het geloof nu is de zekerheid der dingen, die men hoopt, en het bewijs der dingen, die men niet ziet.
   Hebreeën 11:1.

Stellingen behorende bij het proefschrift "Methane emissions from grasslands" van Agnes van den Pol-van Dasselaar, Wageningen, 16 september 1998.

Voor pa en ma

Voorwoord

Velen hebben op de een of andere wijze bijgedragen aan het werk, dat beschreven is in dit proefschrift, en daar ben ik hen allen zeer dankbaar voor! Enkele mensen wil ik toch even met name noemen.

Als eerste mijn promotor prof. Oene Oenema. Beste Oene, jij wist mij door jouw adviezen, suggesties en ideeën altijd te inspireren. Ik heb het erg prettig gevonden dat je mij volop de vrijheid gaf en tegelijkertijd snel en adequaat reageerde bij vragen van mijn kant. Een betere begeleiding kon ik mij niet wensen!

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Na een dag veldwerk kwam ik altijd met een hele serie spuiten gevuld met lucht terug op de vakgroep. Om deze te analyseren heb ik heel wat uurtjes achter de gaschromatograaf doorgebracht bij Jaap Nelemans en Willeke van Tintelen in het lab. Jaap en Willeke, bedankt voor de gezelligheid en hulp in het lab, en ook voor de koffie op Duivendaal!

I gratefully acknowledge the work done by students H. de Wit (NL), K. McCallum and Michael Dent (UK), Rowan Sturgess and Mike Brooks (UK), and A. Hernàndez Villaró (S). Dear Heleen, Kevin and Michael, Rowan and Mike, and Anna, you contributed to the work presented in Chapter 7 and 8. But also, we had lots of fun, and I really hope to meet you again some day!

Mijn project was in de periode 1993-1995 onderdeel van het "geïntegreerde grasland-methaan-project" en in de periode 1995-1997 van het "geïntegreerde grasland-lachgas-methaan-project". Dr. Peter Leffelaar was projectleider van het laatste project. Beste Peter, bedankt voor alle energie die je in het project gestopt hebt om de voortgang ervan te verzekeren. Zonder jou was het project mogelijk al gestopt in 1995 en had dit proefschrift er niet gelegen.

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Agnes

ABSTRACT

Van den Pol-van Dasselaar A (1998) Methane emissions from grasslands.
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179 pages.

This study aims to provide insight into the major factors that contribute to net methane (CH_4) emissions from grasslands, and to provide quantitative data on net CH_4 emissions from typical grasslands with a range of soil wetness and N input in the Netherlands. CH_4 emissions from grasslands were measured with vented closed flux chambers at a number of sites in the period 1994-1997. Furthermore, several incubation experiments were carried out.

Wet grasslands with low N input on peat soil were considerable sources of CH_4. They emitted 80-200 kg CH_4 ha^{-1} yr^{-1}. Main determining factors for temporal variability of CH_4 emissions from these grasslands were ground water level and soil temperature. Main determining factors for spatial variability were CH_4 production capacity and aboveground biomass of sedges. Fractionation of wet peat soils into different size and density fractions indicated that recently died plant material is a major substrate for methanogens.

Intensively managed grasslands with a range of N input on drained peat soils consumed 0.1 to 0.3 kg atmospheric CH_4 ha^{-1} yr^{-1}. Spatial dependence of greenhouse gas emissions from drained peat soils showed differences between sites and also between succeeding days. Extensively managed grasslands with low N input on relatively dry sandy soil consumed 1.1 kg CH_4 ha^{-1} yr^{-1}. Temporal variability of CH_4 uptake by these grasslands was related to differences in soil temperature and soil moisture content.

It is concluded that grassland management, other than drainage, is not an option to mitigate net CH_4 emissions from grasslands in the Netherlands. The effects of N fertilisation, withholding N fertilisation, grazing versus mowing and stocking density on net CH_4 emissions were negligible or small. Thus far, CH_4 emissions from grasslands in the Netherlands were not well-documented. The present study estimates that national net CH_4 uptake by grasslands (excluding wet grasslands) is 0.5 Gg CH_4 yr^{-1}. Wet soils, which occupy only 0.5% of the total surface area, emit 5-10 Gg CH_4 yr^{-1}. Estimates of CH_4 emissions in the Netherlands should be adjusted to put straight the role of grasslands in the national CH_4 budget.

Key words:
carbon mineralisation, drainage, geostatistics, grassland, grassland management, grazing, greenhouse effect, greenhouse gas, ground water level, methane, methane consumption, methane production, mowing, nitrogen fertilisation, nitrogen input, organic matter, peat, sand, soil fractionation, soil moisture content, soil wetness, spatial dependence, spatial variability, temperature, temporal variability, vegetation, wetlands

Contents

Chapter 1

General introduction

GENERAL INTRODUCTION

The greenhouse effect

Radiatively active gases, the so-called greenhouse gases, are transparent for incoming short-wave radiation from the sun. They however trap part of the outgoing infra-red radiation from the earth's surface, thereby causing a heating of the atmosphere. This is the so-called greenhouse effect. Without a greenhouse effect, temperature on earth would be about 33 °C lower than the present temperature (IPCC, 1995a), which would make life in its present form impossible.

The concentration of greenhouse gases in the atmosphere has been increasing since pre-industrial times, mainly due to human activities. This increase gives concern, because it may cause additional global warming due to an enhanced greenhouse effect. Climate change may be a serious threat. Some characteristics of the important greenhouse gases carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) are shown in Table 1.1. Halocarbons, methylchloroform and carbon tetrachloride also contribute to greenhouse gas radiative forcing (IPCC, 1995a).

Table 1.1. Characteristics of key greenhouse gases (IPCC, 1995a).

	CO_2	CH_4	N_2O
Pre-industrial concentration, ppmv[1]	280	0.70	0.28
Concentration in 1992, ppmv	355	1.71	0.31
Concentration increase over 1980s, % yr^{-1}	0.4	0.8	0.25
Atmospheric life-time, yr	50-200	12-17	120
Global Warming Potential[2]	1	24.5	320
Contribution to greenhouse gas radiative forcing, %	62	20	4

[1] ppmv = parts per million by volume.

[2] Global Warming Potential: cumulative radiative forcing over 100 years, caused by a unit mass of gas emitted now (expressed relative to CO_2).

Since the first reports of possible global warming (late 1980s), greenhouse gases have been the subject of many research topics. However, the uncertainties regarding the enhanced greenhouse effect are still large. The temperature of the earth has been increasing in the last decades, but it is very difficult to distinguish between temperature increase due to an enhanced greenhouse effect and temperature increase due to climate variability. The general consensus has been summarised in the second assessment report of the Intergovernmental Panel on Climate Change: "the balance of evidence suggests a discernible human influence on global climate" (IPCC, 1995b). Further on, estimates of the magnitude of sources and sinks of greenhouse gases are still highly variable (e.g. Table 1.2). Finally, controlling factors and the effects of these factors on sources and sinks of greenhouse gases are not fully understood. It is especially important to understand the impact of human activities, as these may be adjusted to reduce the hazards of enhanced global warming.

Global CH_4 emissions

CH_4 is an important greenhouse gas. The estimated contribution of CH_4 to the anticipated enhanced greenhouse effect is 20% (IPCC, 1995a). According to IPCC (1995a), global CH_4 source strength equals 535 Tg CH_4 yr^{-1} (Table 1.2). Natural wetlands are the biggest CH_4 source. Global CH_4 sink strength is estimated to be 515 Tg CH_4 yr^{-1}. The main removal process for CH_4 is the reaction with hydroxyl radicals (OH) in the atmosphere. Atmospheric CH_4 increase is estimated to be 37 Tg CH_4 yr^{-1}. This implicates that there is a difference between estimated total identified sources and estimated implied total sources (Table 1.2). The increase in atmospheric CH_4 concentration is thought to be related to a rise in the human population and the accompanying activities, e.g. oil and gas production and distribution, coal production, animal husbandry, wetland rice production, biomass burning and landfills.

Table 1.2. Estimated sources and sinks of CH_4 in Tg CH_4 yr^{-1} (1 Tg = 10^{12} g) (IPCC, 1995a).

	Estimate	Range
Sources		
Wetlands	115	55-150
Fossil fuel related	100	70-120
Enteric fermentation	85	65-100
Rice paddies	60	20-100
Biomass burning	40	20-80
Landfills	40	20-70
Animal waste	25	20-30
Domestic sewage	25	15-80
Termites	20	10-50
Oceans	10	5-50
Other natural	15	10-40
Total identified sources	*535*	*410-660*
Sinks		
Troposphere	445	360-530
Stratosphere	40	32-48
Soils	30	15-45
Total sinks	*515*	*430-600*
Atmospheric increase	*37*	*35-40*
Implied total sources (atmospheric increase + total sinks)	*552*	*465-640*

CH_4 dynamics in soils

CH_4 production

Methanogenesis may occur in soils when organic matter is degraded under anaerobic conditions, in the absence of electron acceptors other than CO_2 (Oremland, 1988). It is a microbial process, in which methanogens produce CH_4, either in anaerobic soil layers below the ground water level or in anaerobic microsites above the ground water level. CH_4 production is thought to occur primarily via two pathways: aceticlastic methanogenesis ($CH_3COO^- + H^+ \rightarrow CH_4 + CO_2$), which is the predominant pathway, and CO_2 reduction or hydrogenotrophic methanogenesis ($CO_2 + 4H_2 \rightarrow CH_4 + 2H_2O$). Main factors influencing the rate of CH_4 production are aeration of the soil, presence of alternative electron acceptors, like NO_3^-, Mn^{4+}, Fe^{3+}, and SO_4^{2-}, type and amount of available organic matter and size and activity of the methanogenic population (Conrad, 1989; Oremland, 1988; Segers, 1998).

CH_4 consumption

Under aerobic conditions both CH_4 that has been produced in anaerobic parts of the soil and atmospheric CH_4 can be oxidised. Two kinds of CH_4 oxidising activity can be distinguished: low affinity (at high CH_4 concentrations) and high affinity (at low, atmospheric CH_4 concentrations). High affinity CH_4 oxidisers are poorly identified. Next to methanotrophs also nitrifiers may oxidise CH_4 (Bédard & Knowles, 1989). Probably, the contribution of nitrifiers to atmospheric CH_4 consumption is small (Hütsch et al., 1993, Schnell & King, 1994). In this thesis, all CH_4 oxidisers are generally referred to as methanotrophs. Main factors influencing the rate of CH_4 oxidation are O_2 and CH_4 concentration in the soil and size and activity of the methanotrophic population (Segers, 1998).

CH_4 emissions

The contribution of soils to the global CH_4 balance is significant: 14-47% of the total source (i.e. wetlands and rice paddies) and 3-9% of the total sink (IPCC, 1995a). Net CH_4 emissions, i.e. the resultant of CH_4 exchanges between soil and atmosphere, encompasses the processes CH_4 production, CH_4 oxidation, and CH_4 transport. CH_4

transport in the soil can take place via diffusion, ebullition, and via the vascular system of plants (e.g. Schimel, 1995).

Determining factors for CH_4 emissions from soils

Determining factors for CH_4 emissions from soils can be split up into environmental factors and management factors. Important environmental factors are ground water level and soil temperature (Bartlett & Harriss, 1993). Soil temperature affects CH_4 production and CH_4 consumption. Ground water level roughly indicates the transition zone between aerobiosis and anaerobiosis in the soil. Soil moisture content of the unsaturated soil and precipitation may also be important as they influence the degree of anaerobiosis in the unsaturated soil. Further on, vegetation characteristics may be important (Bubier et al., 1995b; Schimel, 1995; Whiting & Chanton, 1993), since plant material may serve as substrate for methanogens and CH_4 may be transported via the vascular system of plants. Important management factors may be drainage (e.g. Martikainen et al., 1992; Roulet et al., 1993), since drainage affects the ground water level, and nitrogen (N) fertilisation (Hütsch et al., 1994; Mosier et al., 1991; Steudler et al., 1989; Willison et al., 1995). N fertilisation may decrease CH_4 consumption, either caused by an immediate inhibition of methanotrophy (short-term effect) or by a change in the composition and size of the microbial community due to repeated fertiliser N application (long-term effect). It is not known whether there is an effect of grazing versus mowing.

CH_4 emissions from soils in the Netherlands

Estimates of CH_4 emissions in the Netherlands have been first compiled by Van den Born et al. (1991), and have been updated by Van Amstel et al. (1993). CH_4 emissions from wet organic soils and drained organic soils were estimated to be 80-200 and 10-200 mg CH_4 m^{-2} d^{-1}, respectively. These estimates were based on research at organic soils in other countries (e.g. Aselmann & Crutzen, 1989; Moore & Knowles, 1989). In general, drained soils are considered to be a net sink for atmospheric CH_4 (e.g. Hütsch et al., 1994; Minami et al., 1994; Mosier et al., 1991). Total national CH_4 source strength was estimated at 1227 Gg CH_4 yr^{-1} (1 Gg = 10^9 g) with a contribution of 10-27 Gg CH_4 yr^{-1} from wet organic soils and 32-89 Gg CH_4 yr^{-1} from drained organic soils (Van Amstel et al., 1993). Van Amstel et al. (1993)

estimated net CH_4 uptake by forests at 2.0 Gg CH_4 yr^{-1}. Net CH_4 uptake by grasslands and arable land were not considered separately. Net CH_4 uptake by arable land is usually much smaller than net CH_4 uptake by grasslands (e.g. Ambus & Christensen, 1995; Glenn et al., 1993; Hütsch et al., 1994).

About one-third of the soil surface in the Netherlands is occupied by grasslands. Thus far, CH_4 emissions from these grasslands are poorly quantified. Grasslands in the Netherlands have certain aspects, which distinguish them from grasslands in other countries. More than 95% of the grasslands is intensively managed and drained, but they have a relatively shallow ground water table. Further more, grassland management in the Netherlands is characterised by large N inputs (300-500 kg N ha^{-1} yr^{-1}) via N fertilisers, animal excreta and atmospheric N deposition, which may reduce CH_4 consumption (e.g. Hütsch et al., 1994; Steudler et al., 1989). Unfertilised grasslands still have a total N input of 30-50 kg N ha^{-1} yr^{-1} via atmospheric deposition. Furthermore, about 30% of the grassland area is situated on peat soils. Peat soils have a high CH_4 emitting potential (Bartlett & Harriss, 1993), because they may be anoxic at shallow depth and have high organic matter contents. Thus far, it is not known whether intensively managed grasslands on drained peat soil are a source or a sink of CH_4. Insight into possible CH_4 uptake by soils is relevant. Although the uptake of CH_4 by aerobic soils is thought to represent only 3 to 9 % of total global CH_4 sinks, it is a CH_4 sink that is directly under the influence of human activity, and as such a CH_4 sink that can possibly be managed.

Next to grasslands on drained soils, a small part of the grasslands in the Netherlands is classified as wet grasslands. They may emit considerable amounts of CH_4 (e.g. Bartlett & Harriss, 1993). Although the area of wet grasslands is small, insight into CH_4 emissions from these grasslands is relevant, since (i) wetlands contribute significantly to national and global CH_4 budgets and (ii) the government in the Netherlands intends to convert part of the agricultural used, drained grasslands on peat soil into wet grasslands.

Aims of this thesis

The major aims of this thesis are:

(i) To provide insight into the major factors that contribute to net CH_4 emissions from grasslands. More information on determining environmental and management factors is needed to investigate the possibilities of reducing CH_4 emissions.

Furthermore, such information can be used in simulation models, that aim at gaining quantitative insight in the underlying processes of CH_4 emissions, which may help to judge the reliability of extrapolations (e.g. Segers & Leffelaar, 1996). Special emphasis is put on the effect of soil wetness and N fertilisation, since they can both be managed. In the Netherlands, soil wetness can be influenced via the water level in ditches. Since large variability of net CH_4 emissions is a common phenomenon, special attention is paid to temporal and spatial variability, both in long-term and short-term field experiments.

(ii) To provide quantitative data on net CH_4 emissions from typical grasslands with a range of soil wetness and N input in the Netherlands. Special emphasis is put on defining the limits of net CH_4 emissions.

Set-up of the study

Field experiments

Field experiments form the main part of this study. Field monitoring provides the most realistic estimates of CH_4 emissions, since the variability in both environmental and management factors is incorporated. Furthermore, the impact of possible determining factors for CH_4 emissions can most realistically be measured in the field. To provide realistic estimates of annual CH_4 emissions, monitoring should be done for a year at least, and for several years in situations where large interannual differences are expected. Net CH_4 emissions and possible determining factors of CH_4 emissions were measured in the period 1994-1997 at grasslands with a range of soil wetness and N input:

(i) "Nieuwkoopse Plassen", the Netherlands, wet peat soils with low N input.

To explore maximum CH_4 emissions from grasslands in the Netherlands, measurements were carried out in the nature preserve "Nieuwkoopse Plassen". "Nieuwkoopse Plassen" includes good representatives of wet grasslands. It is a former peat mining and agricultural area with narrow grassland and reed fields, surrounded by ditches. Ground water level is kept near the surface. The sites can be classified as minerotrophic peat soils (fens).

(ii) Zegveld, the Netherlands, drained peat soils with both low and high N input.

About 30% of the grassland area in the Netherlands is situated on drained peat soils and intensively managed. To examine whether these grasslands are a source or a sink of CH_4, measurements were carried out at the experimental farm Zegveld. Zegveld is a good representative of intensively managed grasslands on peat soil. There were several treatments available: relatively high and relatively low ground water level, grazing and mowing, N fertilisation and withholding of N fertilisation.

(iii) Fallköping, Sweden, drained peat soils with moderate N input.

Large variability of CH_4 emissions is a common phenomenon. Spatial variability of greenhouse gases and possible determinants were studied in co-operation with researchers from several EU-countries during a field campaign at Fallköping. Fallköping is a good representative of agriculturally used grasslands on drained peat soil in north-western Europe.

(iv) Gelderland, the Netherlands, relatively dry sandy soils with low N input.

Grasslands on relatively dry soils may consume significant amounts of CH_4. However, uptake of atmospheric CH_4 may also be restricted due to human influences like N input. To explore the maximum uptake of atmospheric CH_4 by grasslands in the Netherlands, measurements were carried out at Wolfheze and Bovenbuurtse Weilanden, relatively dry sandy soils with a low soil fertility status in the province Gelderland.

Incubation experiments

To unravel some confounding effects from the field experiments, several experiments were carried out under controlled conditions. Potential CH_4 production and CH_4 consumption were determined in incubation experiments with soil samples from different depths. To improve our understanding of CH_4 emissions from wet peat soils, CH_4 production capacity was studied for individual soil fractions obtained by a physical soil fractionation method. The main determining environmental factors of CH_4 consumption capacity, i.e. soil moisture content and soil temperature, often are interrelated in the field. The separate effect of these factors was assessed in incubation experiments.

Methodology of flux measurements

In the field experiments, CH_4 emissions between soil and atmosphere were measured with vented closed flux chambers (Hutchinson & Mosier, 1981; Mosier, 1989). Flux gradient techniques were not considered as plots were small. The use of flux chambers in measuring CH_4 emissions has several advantages and disadvantages (e.g. Mosier, 1989; Velthof, 1997). Main advantages of the use of flux chambers are (i) small-scale differences can be studied, (ii) small fluxes can be measured, (iii) flux chambers are easy in use, and (iv) construction of flux chambers is relatively simple and inexpensive. There are however also disadvantages. Firstly, disturbances by inserting the chamber into the soil may release CH_4 trapped by the soil. This can be minimised by using flux chambers with sharp edges, by carefully inserting them into the soil and/or using boardwalks and permanently installed frames at sites with a soft top soil. Furthermore, build-up of high CH_4 concentrations in flux chambers may decrease CH_4 emissions from the soil; this problem can be minimised by adapting the time of flux chamber closure to the anticipated CH_4 emission. Differences in pressure and/or temperature between the flux chamber and the atmosphere may also be a problem; this can be minimised by using vented chambers and covering these chambers with an insulating sheet.

Outline of this thesis

In Chapters 2, 3, and 4, CH_4 emissions from wet grasslands on peat soil in a nature preserve are studied. In Chapter 2, a simple regression model of CH_4 emissions is presented, with ground water level and soil temperature as independent variables. In Chapter 3, the determinants of spatial variability of CH_4 emissions are evaluated. In Chapter 4, CH_4 production and C mineralisation capacities are presented for individual soil fractions obtained by a physical fractionation method.

In Chapters 5 and 6, CH_4 emissions from intensively managed grasslands on drained peat soil are studied. In Chapter 5, the effect of grassland management on CH_4 emissions is presented. In Chapter 6, results from a measurement campaign, investigating spatial variability of greenhouse gas emissions and determining factors, are presented and analysed using geostatistics.

In Chapter 7, CH_4 emissions from grasslands on sandy soils are studied. Furthermore, the effect of soil moisture content and temperature on CH_4 oxidation is investigated. In Chapter 8, the effects of N input and grazing on CH_4 uptake by extensively and intensively managed grasslands on peat and sandy soils in the Netherlands are presented. Finally, Chapter 9 gives an overview and general discussion of all results.

Chapter 2

Methane emissions from wet grasslands on peat soil in a nature preserve

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METHANE EMISSIONS FROM WET GRASSLANDS ON PEAT SOIL IN A NATURE PRESERVE

Abstract

The area of wet grasslands on peat soil in the Netherlands is slowly increasing at the expense of drained, agriculturally used grasslands. This study aimed (i) to assess the contribution of wet grasslands on peat soil to CH_4 emissions, and (ii) to explain differences among sites and between years in order to improve our understanding of controlling factors. For these purposes, a field study was conducted in the period 1994-1996 in the nature preserve "Nieuwkoopse Plassen", which is a former peat mining and agricultural area. Net CH_4 emissions were measured weekly to monthly with vented closed flux chambers at three representative sites, and at ditches near these sites. Three-years average of CH_4 emissions was 79 kg CH_4 ha^{-1} yr^{-1} for Drie Berken Zudde, 133 for Koole, and 204 for Brampjesgat. Ditches near the sites emitted 42-225 kg CH_4 ha^{-1} yr^{-1}. The time course of CH_4 emissions for all experimental sites and years was fit with a multiple linear regression model with ground water level and soil temperature as independent variables. Lowering or raising the ground water level by 5 cm could decrease or increase CH_4 emissions by 30-50%. Therefore, ground water level management of these grasslands should be done with care.

Introduction

CH_4 is one of the most important greenhouse gases. Methanogenesis occurs wherever organic matter is decomposed under anaerobic conditions, in the absence of electron acceptors other than CO_2. Under aerobic conditions CH_4 can be oxidised by methanotrophs. Net CH_4 exchanges between the soil and the atmosphere are the result of the biogeochemical processes: CH_4 production and CH_4 consumption, and CH_4 transport. Transport can take place via diffusion, ebullition, and via the vascular system of plants. Important environmental factors determining CH_4 emissions from soils are vegetation, temperature and ground water level (e.g. Bartlett & Harriss, 1993; Bubier et al., 1995b; Kettunen et al., 1996; Whiting & Chanton, 1993). Vegetation may serve as a conduit for CH_4 and O_2 transport, and as a substrate for methanogens. Soil temperature affects the kinetics of both microbial CH_4 production and consumption. Ground water level acts as a border between anaerobic and aerobic layers in the soil.

Wetlands are mainly situated on peat soil (Bartlett & Harriss, 1993). They have the potential of emitting large amounts of CH_4, since they are mainly anaerobic. On a global scale, wetlands contribute between 10 and 28% to total CH_4 sources (IPCC, 1995a). Approximately half of the total surface area of the Netherlands has been wetland for some time during the last 3000 years. Especially during the last few centuries the area of wetlands has decreased drastically. Most of it has been transformed into agricultural land. At present, less than 1% of the total surface area is wetland. However, this area is increasing, due to the policy of the government of the Netherlands to withdraw intensively managed, drained grasslands from agriculture and turn them into wet grasslands. Thus far, CH_4 emissions from wet grasslands in the Netherlands had not been quantified.

The aim of our study was (i) to assess the contribution of wet grasslands on peat soil to CH_4 emissions, and (ii) to explain differences among sites and between years in order to improve our understanding of possible controlling factors. For these purposes, a three-year field study was conducted at wet grasslands in the nature preserve "Nieuwkoopse Plassen". Net CH_4 emissions, CH_4 concentrations in the soil profile, ground water levels and soil temperatures were measured. We hypothesised (i) that the wet grasslands would emit considerable amounts of CH_4, and (ii) that differences in CH_4 emissions among sites and between years would mainly depend on differences in ground water level and soil temperature.

Materials and methods

Site description

"Nieuwkoopse Plassen" is a nature preserve located in the major peat area of the western part of the Netherlands (52°08'N, 4°48'E). During several centuries, the "Nieuwkoopse Plassen" area has been used for peat mining and agriculture. Since some decades, it is a nature preserve with narrow (30-80 m wide) grassland and reed fields, surrounded by ditches. Ground water level is kept near the surface via water level of the ditches. The area can be characterised as a fen, i.e. a minerotrophic peat soil, due to the influence of surface water. Measurements were done at three grassland sites spread over the area, Drie Berken Zudde, Koole and

Brampjesgat (Table 2.1), which could only be reached by boat, and at ditches near these sites. The sites form a good representative of the wet, nutrient-poor grassland sites in the area. They were chosen to cover a range of ground water levels and vegetation types. Drie Berken Zudde and Koole have not been fertilised for more than 20 years. Brampjesgat still receives every second year farm yard manure (about 5 ton dry matter ha^{-1}). The sites are high in organic matter and low in nitrogen and phosphorus. The pH is low, especially at Drie Berken Zudde. The vegetation of the sites is dominated by grasses (mainly *Agrostis canina* L., *Anthoxanthum odoratum* L. and *Molinia caerulea* (L.) Moench), mosses (*Sphagnum* spp., *Polytrichum* spp.), rushes (*Juncus* spp.), sedges (*Carex* spp.), and reed (*Phragmites australis* (Cav.) Trin. ex Steudel). Rushes, sedges and reed may transport CH_4 directly from anaerobic layers to the atmosphere via their aerenchymatous tissues. The sites are mown once or twice every year in the period July to September. Typical yields are 3-5 ton dry matter ha^{-1} yr^{-1}.

Table 2.1. Mean ground water level (GWL) and ranges in the years 1994-1996, soil characteristics of the 0-20 cm layer, and vegetation types of the sites Drie Berken Zudde (DBZ), Koole and Brampjesgat.

	DBZ	Koole	Brampjesgat
GWL, cm, mean	-18	-9	-11
ranges	-36 to -7	-35 to 3	-25 to -1
Loss-on-ignition (%)	94	78	69
Total nitrogen (g kg^{-1})	15	16	16
Total phosphorus (g kg^{-1})	0.6	0.9	1.5
pH-H_2O	3.5	4.9	5.3
Vegetation type, in % of total dry matter above 5 cm, average of harvest 1994 and 1995			
Grasses	55	8	23
Mosses	31	57	21
Rushes	4	17	17
Sedges	2	16	11
Reed	3	0	24
Remainder	5	2	4

Monitoring net CH_4 emissions, ground water levels and soil temperatures

Net CH_4 emissions were measured weekly to monthly from January 1994 to October 1996 with six vented closed flux chambers (Hutchinson & Mosier, 1981; Mosier, 1989) per site. To prevent artificially induced fluxes due to the very soft top soil, wooden boardwalks were installed. Circular, steel frames (I.D. 25 cm, height 30 cm) were permanently installed into the soil to a depth of about 25 cm. During measurements, circular, PVC flux chambers (I.D. 25 cm, height 25 to 77 cm depending on the height of the vegetation) were placed on top of the frames. Preliminary measurements showed that size of the flux chambers did not affect calculated CH_4 emissions. Flux chambers were closed by a PVC lid and covered with insulating sheets to prevent temperature changes within the chamber.

Net CH_4 emissions from ditches near the sites were measured biweekly to monthly from October 1994 to November 1995 with three to six flux chambers, i.e. one or two per ditch. These circular, PVC flux chambers were connected to a floating tray and carefully placed on the water. Measurements could only be carried out when there was not much wind.

To examine diurnal variability, CH_4 emissions were measured several times during 31 October 1994 and 24 July 1995. On 28 September and 3 October 1996, CH_4 emissions were measured in the coldest and warmest period of the day (just before sunrise between 4.30 and 6.30 h, and between 13.00 and 15.00 h).

At each site and at each measurement, ground water level was recorded from water level readings in perforated pipes (I.D. 4 cm) with the peat surface as reference point. Ambient temperature and soil temperatures at 0, 2, 5, 10, 20, 30, 40, and 50 cm depth were also recorded.

CH_4 concentrations in flux chambers are expected to follow a linear increase or decrease and finally level off (Mosier, 1989). We measured in the linear phase. Four gas samples were taken with glass syringes at regular time intervals (10 to 20 minutes) from the headspace of the chambers. Gas samples were analysed for CH_4 within 24 h by gas chromatography using a flame ionisation detector (coefficient of variation: 0.08%). A standard CH_4 concentration of 2.0 $\mu L\ L^{-1}$ (± 5%) was used for calibration. Net CH_4 emissions were calculated from linear regression of the time course of CH_4 concentration in the headspace of the chambers. Mean annual CH_4 emissions, ground water levels and soil temperatures were estimated by trapezoidal integration over time. Data for November and December 1996 were taken from the average of the corresponding periods in 1994 and 1995. CH_4 emissions followed a skewed distribution and were transformed to a near-normal distribution by ln-transforming the data. Simple and multiple linear regression analyses were performed with ln-transformed CH_4 emissions as dependent variable, and ground water level and soil temperatures at several depths as independent variables.

Monitoring CH_4 concentrations in the soil profile

CH_4 concentrations in the soil profile were measured weekly to monthly at Drie Berken Zudde from January to July 1995 and at Koole from September 1994 to November 1995. Perforated PVC pipes (I.D. 5 cm) were permanently installed in the soil in an area of about 0.1 m^2 about three months before the start of the measurements. The end of each pipe consisted of a small perforated compartment of 10 cm height. Pipes of different lengths with compartments at 0-10, 20-30, 30-40, and 40-50 cm at Drie Berken Zudde, and at 0-10, 10-20, 20-30, 30-40, and 95-105 cm at Koole were used. The holes in the compartments allowed water and air from the surrounding soil to enter. CH_4 concentrations in the compartments were assumed to be in equilibrium with CH_4 concentrations in the surrounding soil. Each compartment had two polythene tubes going from the compartment to the soil surface. Gas samples were taken via the tubes with glass syringes, and analysed for CH_4 by gas chromatography. Whenever ground water level reached a certain compartment of the pipe, water samples were taken and injected into incubation bottles. The bottles were shaken vigorously to degas the water. Subsamples from the headspace of the bottles were taken through rubber septa and analysed for CH_4. As the solubility of CH_4 in water is low, amount of dissolved CH_4 in the water was neglected. CH_4 concentrations in the soil profile were converted to mg m^{-3} using an average peat porosity of 0.95, as found by Liblik et al. (1997). CH_4 storage (mg m^{-2}) was determined by integrating CH_4 concentrations of the upper 40 cm of the peat profile.

Results

Weather and ground water level

Average air temperature was 10.6 °C in 1994, 10.4 in 1995 and 8.6 in 1996 (long-term average is 9.4 °C). Annual precipitation was 903 mm in 1994, 739 in 1995 and

597 in 1996 (long-term average is 792 mm). Fig. 2.1 shows the time course of soil temperatures at 5, 20, and 50 cm depth at the site Koole. The time course of soil temperatures at Drie Berken Zudde and Brampjesgat was similar (not shown). Drie Berken Zudde had the lowest ground water level (Fig. 2.2). Ground water levels generally decreased in summer.

Net CH_4 emissions

All three sites were sources of CH_4 (Fig. 2.3). Differences among sites and between years were quite large. Mean net CH_4 emissions from ditches near the sites were highest in summer (Fig. 2.4).

Results on diurnal variability are based on few observations. On 28 September and 3 October 1996, day-time net CH_4 emissions were 52 and 59% of night-time net CH_4 emissions. On 31 October 1994 and 24 July 1995, no clear pattern of CH_4 emissions was found during the day.

Annual net CH_4 emissions

At Drie Berken Zudde and Koole, mean annual net CH_4 emissions were highest in 1994, and at Brampjesgat in 1995 (Table 2.2). At all three sites, CH_4 emissions were low in 1996. The relatively high standard deviations indicate a high spatial variability of CH_4 emissions within sites. Drie Berken Zudde showed the lowest mean annual net CH_4 emissions and Brampjesgat the highest. CH_4 emissions from ditches however were lowest near Brampjesgat and highest near Drie Berken Zudde (Table 2.2). Emissions from ditches were never measured in situations of disturbances like heavy wind or passing boats, in which mixing of ditch water and release of CH_4 from CH_4-rich bottom water may occur. Emissions were thus underestimated. Underestimation may have been smallest at the ditch near Drie Berken Zudde, which was best protected against wind and is located in a rather remote area where no boats pass.

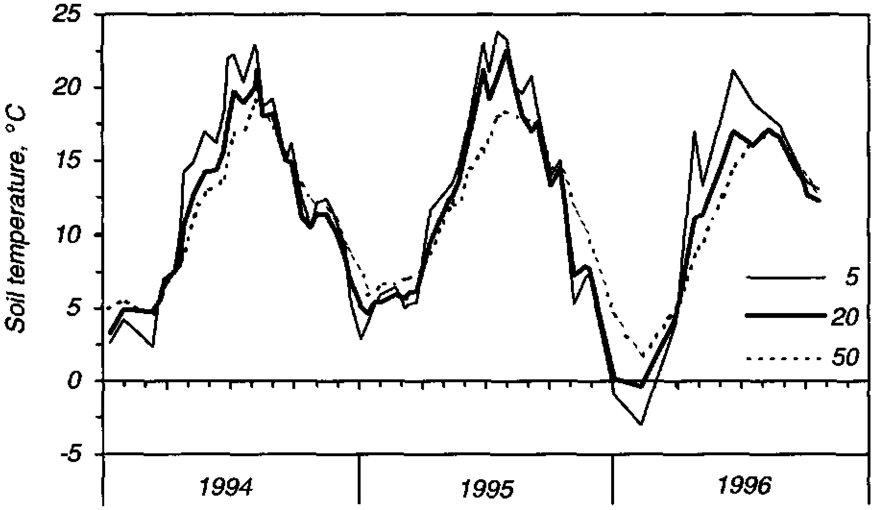


Fig. 2.1. Time course of soil temperatures (°C) at 5, 20, and 50 cm depth at the site Koole.

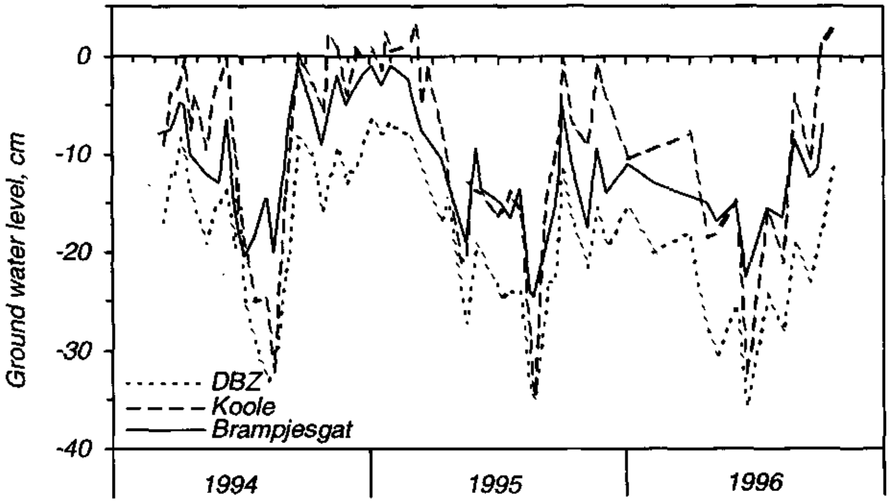
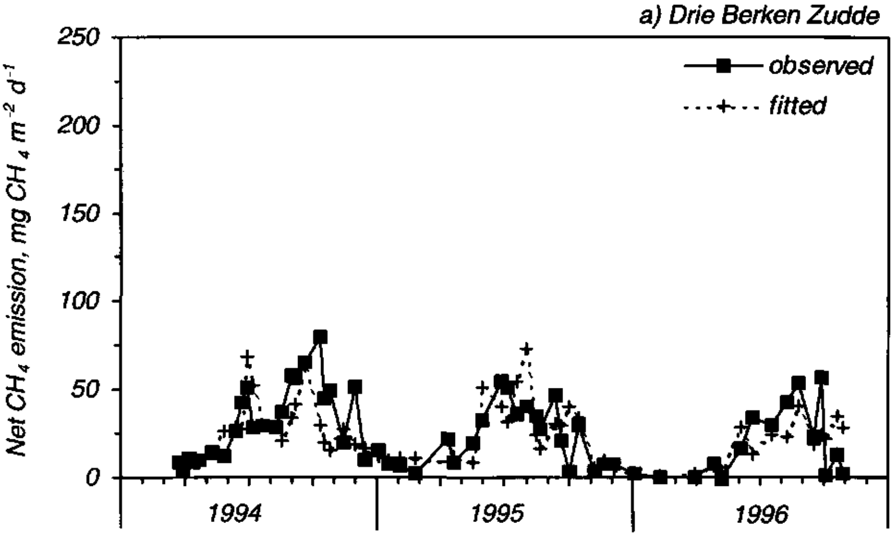


Fig. 2.2. Time course of ground water level (cm below the surface) at the sites Drie Berken Zudde (DBZ), Koole, and Brampjesgat.



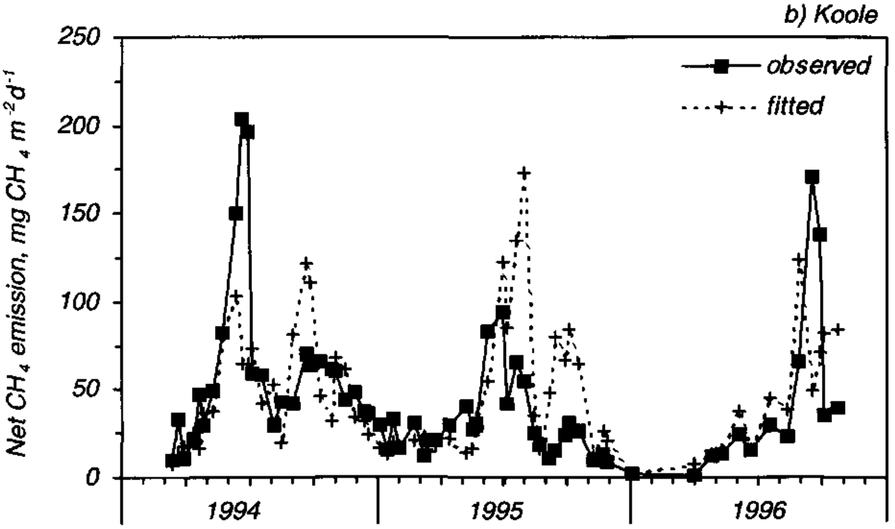


Fig. 2.3. Time course of observed and fitted (Equation 2.1) mean net CH_4 emissions (mg CH_4 m^{-2} d^{-1}) from the sites **a)** Drie Berken Zudde, **b)** Koole, and **c)** Brampjesgat (see next page). Each data point represents the average of six measurements.

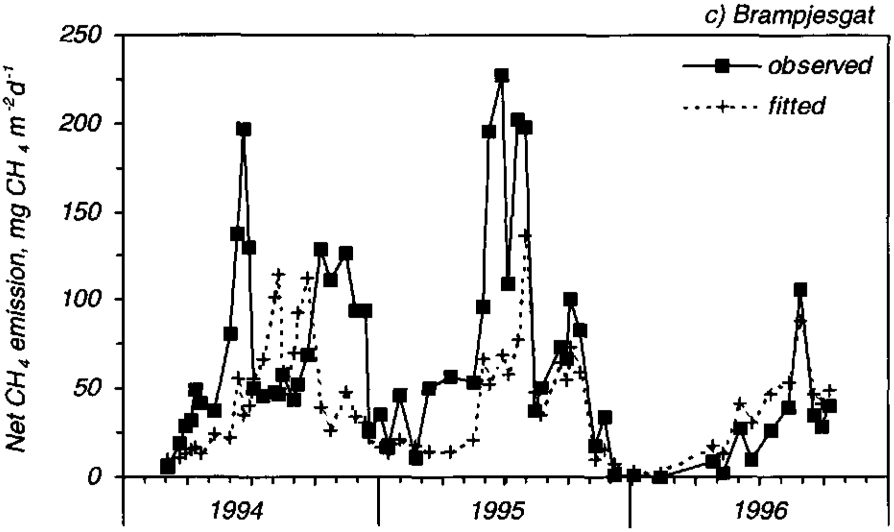


Fig. 2.3. c) Time course of observed and fitted mean net CH$_4$ emissions (mg CH$_4$ m^{-2} d^{-1}) from the site Brampjesgat. Each data point represents the average of six measurements. See also previous page.

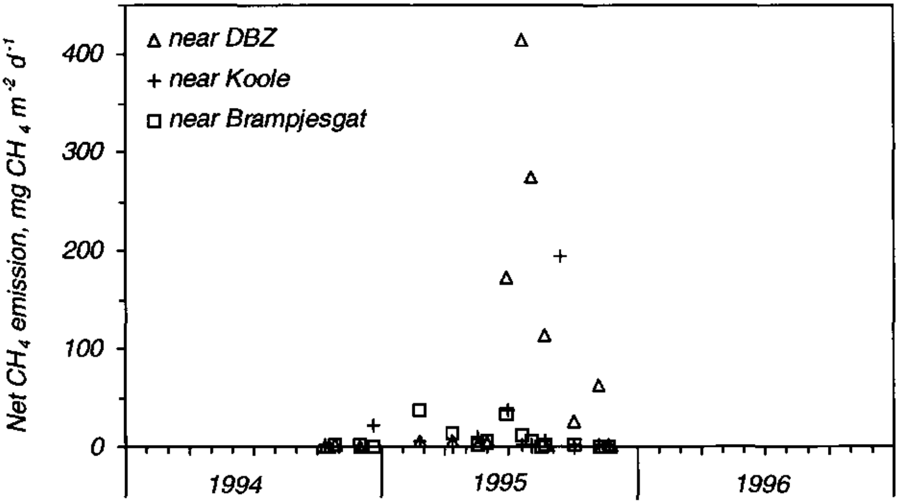


Fig. 2.4. Time course of mean net CH$_4$ emissions (mg CH$_4$ m^{-2} d^{-1}) from ditches near Drie Berken Zudde (DBZ), Koole and Brampjesgat. Each data point represents one or two measurements.

Table 2.2. Mean annual net CH_4 emissions (kg CH_4 ha^{-1} yr^{-1} ± SD) of six flux chambers at Drie Berken Zudde (DBZ), Koole and Brampjesgat, and for ditches near the sites. Mean annual ground water level (cm below the surface) is given in brackets.

	DBZ		Koole		Brampjesgat	
1994	96±91	(16 cm)	182±122	(7 cm)	224±107	(9 cm)
1995	83±98	(18 cm)	111±111	(10 cm)	278±220	(12 cm)
1996	60±68	(21 cm)	107±110	(11 cm)	111±51	(13 cm)
1994-1996	79±88	(18 cm)	133±120	(9 cm)	204±160	(11 cm)
Ditches	225		73		42	

CH_4 emissions in relation to ground water level and soil temperatures

Simple linear regression analyses of each of the individual data from the three sites showed that ln-transformed mean net CH_4 emissions were related to soil temperatures (r^2 of 0.27-0.33; n=183). The relation with only ground water level was poor (r^2 of 0.01; n=183). The best model obtained with multiple linear regression analyses included both soil temperature and ground water level (r^2 of 0.54; n=183) (see Fig. 2.3):

$$\ln(Y_t) = 1.75 + 0.20 * T_{20,t} - 0.075 * GWL_t \qquad (2.1)$$

t = time, d;
Y_t = net CH_4 emissions at time t, mg CH_4 m^{-2} d^{-1};
$T_{20,t}$ = soil temperature at 20 cm depth at time t, °C;
GWL_t = ground water level at time t, cm below the surface.

Linear regression analyses were also carried out with ln-transformed mean annual net CH_4 emissions as dependent variable, i.e. for each year and for each site, and mean annual ground water level and soil temperatures as independent variables. The best model included ground water level (r^2 of 0.57; n=9):

$$\ln(Y_t) = 5.87 - 0.08 * GWL_t \qquad (2.2)$$

t = time, year;

Y_t = annual net CH_4 emissions at time t, kg CH_4 ha^{-1} yr^{-1};

GWL_t = annual ground water level at time t, cm below the surface.

CH_4 concentrations in the soil profile

CH_4 concentrations in the soil (Fig. 2.5) increased from ambient at the surface to about 3000 mg m^{-3} in deeper layers. Ground water level influenced the CH_4 concentration profile. CH_4 concentrations in soil layers above the ground water table were near ambient. A considerable increase was found immediately below the ground water level. This indicates a net flux of CH_4 from the saturated peat into the unsaturated zone. Calculated CH_4 storage is dependent on the depth to which CH_4 concentrations are integrated. CH_4 storage (±SD) in the upper 40 cm of the soil profile was 120 ± 155 mg m^{-2} at Drie Berken Zudde and 209 ± 124 mg m^{-2} at Koole. At Drie Berken Zudde, CH_4 storage was on average 258 mg m^{-2} in winter and 11 in summer. At Koole, however, there was no clear seasonal pattern, probably due to the relatively high ground water level throughout the year (Fig. 2.2). Turnover time of CH_4, i.e. storage to emission ratio, was on average 5-6 days for both sites.

Discussion

CH_4 emissions

In a review, Bartlett & Harriss (1993) arrive for wetlands in the latitudes 45-60°N at a mean estimate of 87 mg CH_4 m^{-2} d^{-1} during the emission period (standard error of mean: 18, range: 0-664), this is equivalent to 131 kg CH_4 ha^{-1} yr^{-1} (winter fluxes were assumed to be zero). More recent studies (e.g. Bubier et al., 1993; Martikainen et al., 1995; Suyker et al., 1996) yielded similar CH_4 emissions. CH_4 emissions from our grasslands were in the same range (60-280 kg CH_4 ha^{-1} yr^{-1}, Table 2.2). Differences among the three sites could largely be attributed to ground water level differences.

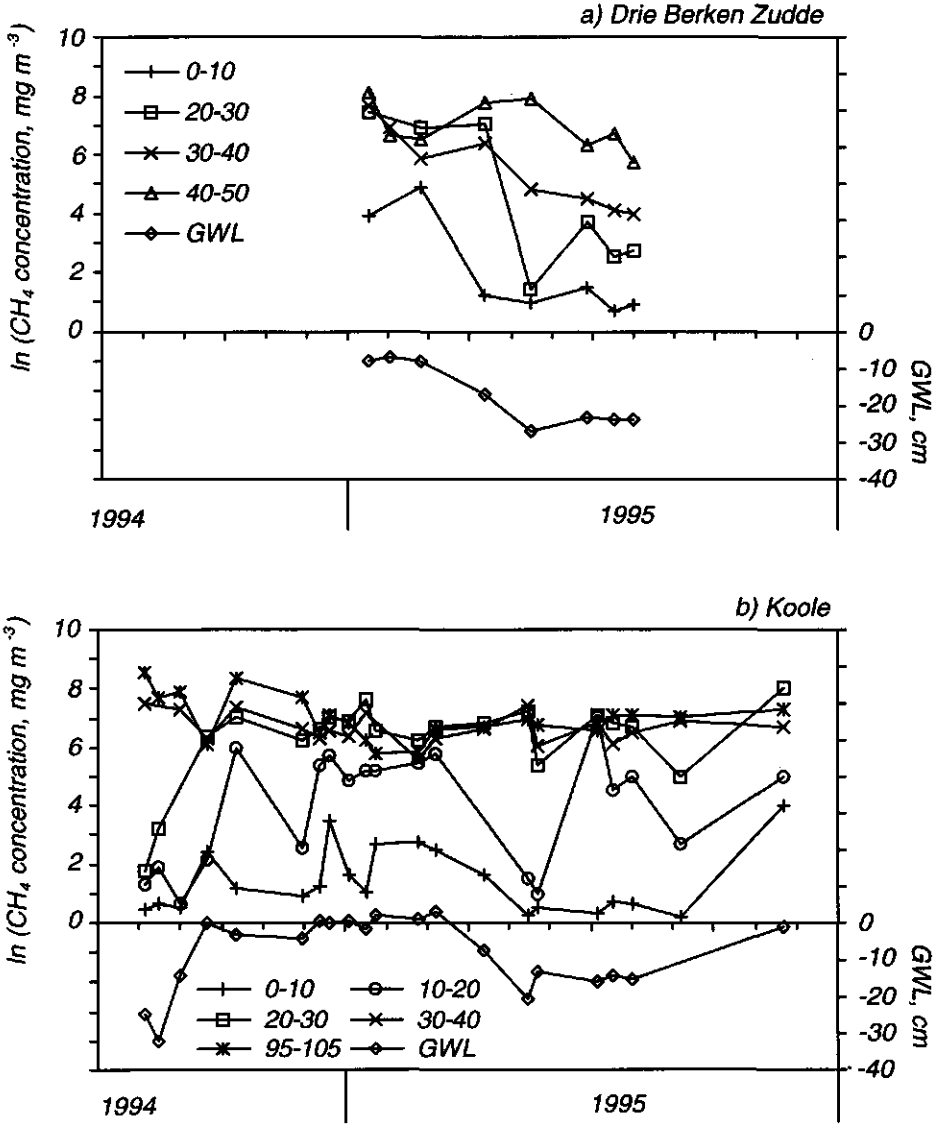


Fig. 2.5. Time course of ln-transformed CH_4 concentration (mg m^{-3}) in different soil layers (cm below the surface) at **a**) Drie Berken Zudde, and **b**) Koole, and ground water level (GWL) (cm below the surface).

However, Brampjesgat showed higher CH_4 emissions than Koole whilst the annual mean ground water level was almost similar. This might be explained by other differences among the sites, for example in vegetation and in pH (Table 2.1) (Van den Pol-van Dasselaar et al., 1998c).

In the "Nieuwkoopse Plassen" area, fields are surrounded by ditches. CH_4 emissions from these ditches ranged from 0-425 mg CH_4 m^{-2} d^{-1}, which is similar to the CH_4 emissions from ditches of drained bogs and fens in Ontario, Canada (5-400 mg CH_4 m^{-2} d^{-1}) as found by Roulet & Moore (1995). Emissions from ditches seemed to be related to temperature with the highest emissions occurring in summer (Fig. 2.4). We expect that the major CH_4 transport mechanism in ditches was diffusion and convection for low emissions and ebullition for high emissions. This was supported by an often occurring non-linear increase in CH_4 concentration in the headspace of the chamber in summer, indicating the escape of CH_4 bubbles to the headspace between two measurements. Further on, bubbles were frequently seen in summer.

Temporal variability of CH_4 emissions

Interannual variability

Interannual variability of CH_4 emissions (Table 2.2) was related to differences in ground water level (Equation 2.2). Moore & Roulet (1993) and Liblik et al. (1997) found a strong relationship between the logarithm of the average seasonal CH_4 flux and average seasonal water table depth for wetland sites in Canada. Our results support this relationship. Furthermore, our results confirm that the relationship between CH_4 emissions and ground water level is not only present in summer, but during the whole year. The latter is important in estimating annual emissions. The existence of large differences between years implies that monitoring of CH_4 emissions should be done for at least several years to obtain a reliable estimate of emissions from a particular area. If extrapolation of measurements is needed, special attention should be paid to ground water level and soil temperature. This is especially important during the growth period, as the highest CH_4 emission rates may occur in that period.

Seasonal variability

We found a strong seasonal variability with high emissions in summer and low emissions in winter (Fig. 2.3). We hypothesised that differences in CH_4 emissions would mainly depend on differences in ground water level and soil temperature. This was true (Equation 2.1, r^2 of 0.54; n=183), but still a large part of the temporal variability of CH_4 emissions could not be explained. Although this is found often (e.g. Frolking & Crill, 1994; Rouse et al., 1995), in some studies a larger part of the variability could be explained (e.g. Dise et al., 1993; Suyker et al., 1996). However, our regression model fitted the time course of CH_4 emissions for all three sites and all three years (Fig. 2.3), even though there were relatively large differences among sites and between years. Therefore, we conclude that both ground water level and soil temperature have a large influence on CH_4 emissions.

Temperature may affect the microbial CH_4 producing and consuming processes instantaneously, while ground water level may have both short- and long-term effects controlling methanogenic and methanotrophic populations within a site. A decrease of CH_4 production due to a decrease of ground water level may be caused by an increase of electron acceptors, such as oxygen, nitrate and sulphate (Freeman et al., 1994), by a decrease of available methanogenic substrates, as well as by a reduction in the population of methanogenic bacteria (Shannon & White, 1994). Often, there appears to be a hysteresis effect, i.e. CH_4 emissions are greater on the falling than rising water table limbs, for a set water table depth (e.g. Moore & Dalva, 1993). For our sites, a hysteresis effect could not be detected. According to Shannon & White (1994), a substantial drop of the water table may cause lower CH_4 emissions for at least a year following the return of saturated conditions. This would suggest that the relatively dry conditions of 1995 and 1996 might affect CH_4 emissions of coming years.

Diurnal variability

Our CH_4 emission estimates (Fig. 2.3 and 2.4, Table 2.2) are based on measurements during the day. However, CH_4 emissions may fluctuate within a time scale of hours. This may be caused by changes in e.g. temperature, which may affect both methanogenesis and methanotrophy and radiation, which may affect plant-mediated CH_4 transport and substrate availability for methanogens through affecting plant photosynthesis and subsequent carbon translocation to roots (Mikkelä et al., 1995). The magnitude of diurnal variability of CH_4 emissions from wetlands is inconsistent. For example, Suyker et al. (1996) showed that CH_4 emissions from a fen were higher during day-time than during night-time. Klinger et al. (1994) could not detect diurnal variability of CH_4 emissions from peatlands. Mikkelä et al. (1995) found no significant differences between CH_4 emissions during day and night for sites with high ground water levels. For sites with low ground water levels however (5-30 cm below the surface), they found higher emissions during the night than during the day. They speculate that this may be caused by diurnal variability of CH_4 oxidation due to temperature changes in the top layer, and hardly any diurnal variability of CH_4 production, occurring in the deeper layers. Our few data on variability of CH_4 emissions during the day suggest that we might have underestimated CH_4 emissions, as CH_4 emissions were higher during night-time than during day-time. However, this is based on too little information to draw any firm conclusions. Possible effects of measurement time during the day may be excluded, as the measurement chronology was changed at every measurement date.

Effect of climate and land use changes on CH_4 emissions

In the Netherlands, about 3% of the peat area is similar to the "Nieuwkoopse Plassen" area. The greater part of peat soils is drained and agriculturally utilised. Intensively managed and drained grasslands on these peat soils are a small sink of CH_4 with an annual consumption of 0.1 to 0.3 kg atmospheric CH_4 ha^{-1} yr^{-1} (Van den Pol-van Dasselaar et al., 1997). The government of the Netherlands intends to convert part of the intensively managed grasslands on peat soil from agriculture into more natural ecosystems. The ground water level of these grasslands will than be raised again, and both fertilisation and dry matter yield will be reduced. Eventually, these grasslands will be comparable to grasslands in the "Nieuwkoopse Plassen" area with estimated mean annual net CH_4 emissions of 60-280 kg CH_4 ha^{-1} yr^{-1}. Therefore, this policy may lead to increased CH_4 emissions.

Both soil temperature and ground water level have a large influence on CH_4 emissions. If, as a consequence of increased greenhouse gas emissions, temperatures on earth will increase, then CH_4 emissions will also increase, unless, as a consequence of a warmer climate, ground water level will drop simultaneously. According to Equation 2.1, the effect of changes in soil temperature might be considerable: an increase of soil temperature at 20 cm depth by 2 °C would cause CH_4 emissions to increase by 50%, assuming that all other factors remain constant.

In the nature preserve "Nieuwkoopse Plassen", ground water level is maintained on a higher level than in the surrounding agriculturally used areas. It is possible to manage the ground water level via the water level in the ditches. Changes in ground water level management will affect CH_4 emissions. If, for example, ground water level is maintained 5 cm above the present level, then CH_4

emissions may increase by 45-50% (according to Equation 2.1 and 2.2). If however, ground water level drops 5 cm, CH_4 emissions may decrease by 30%. The largest impact will be achieved in summer, when CH_4 emissions are high. As the impact of ground water level on CH_4 emissions may be considerable, ground water level management of wet grasslands should be done with care.

Acknowledgements

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Chapter 3

Determinants of spatial variability of methane emissions from wet grasslands on peat soil

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DETERMINANTS OF SPATIAL VARIABILITY OF METHANE EMISSIONS FROM WET GRASSLANDS ON PEAT SOIL

Abstract

Methane (CH_4) emissions from soils, representing the consequence of CH_4 production, CH_4 consumption and CH_4 transport, are poorly characterised and show a large spatial variability. This study aimed to assess the determinants of field-scale spatial variability of CH_4 emissions from wet grasslands on peat soil. Mean CH_4 emission rates of a three-year experiment at 18 plots distributed over three sites in the nature preserve "Nieuwkoopse Plassen" on peat soil in the Netherlands were related to CH_4 production and CH_4 consumption capacities of soil layers, and to soil and vegetation characteristics. Spatial variability of CH_4 emissions and possible determining factors was high. Annual CH_4 emissions ranged from 30 to 370 kg CH_4 ha^{-1} yr^{-1}. Coefficients of variation (CV) of CH_4 emissions were on average 37% among sites and 83% within sites. Most important determinants of spatial variability were CH_4 production capacity (average: 211 ng CH_4 g^{-1} dry soil h^{-1}; CV: 131%) and aboveground biomass of sedges (*Carex* spp.) (average: 0.45 g dm^{-2}; CV: 127%) ($P<0.01$). Sedges may affect CH_4 emissions by stimulating CH_4 transport from anaerobic layers to the surface via their vascular system and/or by serving as substrate for methanogens. For extrapolation of CH_4 emissions to larger areas, best results will be obtained by using factors that are easy to determine, like vegetation.

Introduction

Methane (CH_4) emissions from soils have been measured extensively, both from large areas using micrometeorological measurement techniques and small areas using flux chamber measurements. All these studies show high spatial variability of CH_4 emissions, both among and within sites (e.g. Bartlett & Harriss, 1993; Bubier et al., 1993; Shurpali & Verma, 1998; Van den Pol-van Dasselaar et al., 1998b; Waddington & Roulet, 1996). CH_4 emissions may vary an order of magnitude within several metres.

CH_4 emissions from the soil to the atmosphere are the result of the biogeochemical processes: CH_4 production and CH_4 consumption, and CH_4

transport. Each of these processes is influenced by a multitude of factors (e.g. Segers, 1998). CH_4 production is a strict anaerobic microbial process, in which methanogenic micro-organisms reduce organic matter in the absence of other electron acceptors. CH_4 production is influenced (i) by the aeration of the soil, as methanogenic micro-organisms require anoxic conditions to produce CH_4, (ii) by the presence of alternative electron acceptors, like nitrate and sulphate, (iii) by type and amount of available organic matter, and (iv) by the size of the methanogenic population. CH_4 consumption is an aerobic microbial process, in which CH_4 is oxidised by methanotrophs. It is influenced by the CH_4 and O_2 concentration in the soil, and by the size of the methanotrophic population. CH_4 transport in the soil can take place via diffusion, ebullition and plants. If CH_4, produced in anaerobic layers of the soil, is transported via diffusion, a considerable part of the CH_4 can be oxidised again in aerobic layers of the soil before it reaches the atmosphere. Transport via ebullition, i.e. via bubbles, and transport via plants limits the possibility of CH_4 oxidation in aerobic layers. Therefore, these types of CH_4 transport greatly facilitate CH_4 emissions.

Even though the mechanisms of CH_4 production, CH_4 consumption, and CH_4 transport are qualitatively reasonably well understood, quantification of determinants of spatial variability of CH_4 emissions is poor. For extrapolation of results from a particular plot or field to a larger area, we need quantitative insight of the dependence of CH_4 emissions on environmental factors under a wide range of conditions. Possible determinants of spatial variability are related to the scale at which these determinants dominate (Klinger et al., 1994). For example, on microscale, important determinants of CH_4 emissions may be soil aeration, methanogenesis, and methanotrophy. On the scale of an individual field, important determinants may be plant growth and fluctuations in ground water table and soil temperature.

Our study aimed to assess the determinants of spatial variability of CH_4 emissions on field-scale. We studied CH_4 production and CH_4 consumption capacities of soil layers, and soil and vegetation characteristics of 18 plots at wet grasslands in the nature preserve "Nieuwkoopse Plassen" on peat soil in the Netherlands. CH_4 emissions from these 18 plots had been measured for three years (Van den Pol-van Dasselaar et al., 1998b). We also included additional plots to investigate the relation between CH_4 emissions and the occurrence of individual plant species, as vegetation composition is thought to be one of the main factors influencing CH_4 emissions.

Materials and methods

Site description and main plots

The experimental site has been described in detail elsewhere (Van den Pol-van Dasselaar et al., 1998b). In brief, in the period 1994 - 1996, we have measured CH_4 emissions from 18 main plots spread over three representative wet grasslands in the "Nieuwkoopse Plassen" area in the Netherlands with vented closed flux chambers. "Nieuwkoopse Plassen" is a nature preserve with narrow (30-80 m wide) grassland and reed fields, surrounded by ditches. Ground water level is kept near the surface via the water level in the ditches. Distances between the three sites, i.e. Drie Berken Zudde, Koole and Brampjesgat, were approximately two kilometres. Distances between plots within a site were about two metres.

Detailed study of main plots

In October 1996, soil of each of the main plots was sampled. Ground water levels were then 11.5 cm below the surface at Drie Berken Zudde, 2.5 cm above the surface at Koole and 7 cm below the surface at Brampjesgat. Soil pH-H_2O was measured in-situ in the field. Soil samples were stored at 4 °C and processed the next day. We studied the top layer of recently died plant material, and the layers 0-5 cm, 5-10 cm, 10-20 cm, 20-30 cm, 30-40 cm, and 40-50 cm of the soil.

We determined total aboveground biomass, and aboveground biomass per vegetation type, i.e. grasses, sedges, rushes, reed, mosses, and remainder. For soil layers up to 30 cm, we determined dry bulk density, and for Drie Berken Zudde and Brampjesgat also biomass of roots, after wet-sieving with a mesh size of 2 mm.

Each soil layer was analysed (according to Houba et al., 1995) for soil moisture content (gravimetrically after drying at 105 °C for 24 h), loss-on-ignition (550 °C for 2 h), total carbon content (850 °C for 2 h), total nitrogen and phosphorus content (both spectrophotometrically measured in digests obtained after treating a soil sample with H_2SO_4-salicylic acid-H_2O_2-Se). For Brampjesgat, we determined dissolved organic carbon by extraction with 0.1 *M* $CaCl_2$.

Per soil layer, 25 g field-moist soil was incubated in a 580 ml glass bottle at 20 °C both anaerobically (N_2) and aerobically (air + 100 μL L^{-1} CH_4) to determine respective CH_4 production and CH_4 consumption capacity. For Drie Berken Zudde and Koole, also both anaerobic and aerobic CO_2 production capacities were determined. For each bottle, a time series of CH_4 and CO_2 concentration in the headspace was obtained by sampling the headspaces of the bottles daily for three to five days. CH_4 concentrations were determined either by a gas chromatograph (PU 4400), using a flame ionisation detector, or by a photo-acoustic infra-red absorption gas analyser (Brüel & Kjær 1300). There were no significant differences between results obtained with the gas chromatograph and the gas analyser, at the high concentrations found. CO_2 concentrations were determined by the gas analyser.

Additional plots

In July 1995, 18 additional plots were selected at Koole to assess the effect of individual plant species on spatial variability of CH_4 emissions. We measured CH_4 emissions from each plot with PVC flux chambers (I.D. 20 cm, height 16 cm). Four gas samples were taken with glass syringes at regular time intervals (10 to 20 minutes) from the headspace of the chambers. They were analysed for CH_4 within 24 h after collection by gas chromatography using a flame ionisation detector. Furthermore, total aboveground biomass, and aboveground biomass per individual species were assessed for each plot.

Data acquisition

Annual net CH_4 emissions from the main plots were estimated by trapezoidal integration of CH_4 emissions over time (Van den Pol-van Dasselaar et al., 1998b). CH_4 production capacity ($\mu g\ CH_4\ g^{-1}$ dry soil h^{-1}) was calculated from linear regression of the time course of CH_4 concentration in the headspace of the bottles. CH_4 consumption capacity ($\mu g\ CH_4\ g^{-1}$ dry soil h^{-1}) was calculated using the first-order rate constant and an initial CH_4 concentration of 100 $\mu L\ L^{-1}$. Net CH_4 emissions from the additional plots were calculated from linear regression of the time course of CH_4 concentration in the headspace of the chambers.

Simple linear regression analyses were carried out with CH_4 emissions as dependent variable, and soil and vegetation characteristics as independent variables. Regression analyses were performed for each individual soil layer and for combinations of soil layers. Best results were obtained by using the average values of the layer 0-20 cm depth (including top layer) as independent variables. These results are presented here. For each individual site (n=6), the effect of a certain variable was considered significant for $r^2>0.53$ ($P<0.05$) and highly significant for

$r^2>0.78$ ($P<0.01$). For all sites and in the experiment with additional plots ($n=18$), the effect was considered significant for $r^2>0.16$ ($P<0.05$) and highly significant for $r^2>0.29$ ($P<0.01$).

Results

Spatial variability of CH_4 emissions, soil and vegetation characteristics

We found high spatial variability of CH_4 emissions from the main plots, both among and within sites (Fig. 3.1). Coefficient of variation among sites was on average 37% (32% in 1994, 55% in 1995 and 25% in 1996; calculated using mean annual net CH_4 emissions from each individual site). Coefficient of variation within sites was on average 83% (107% for Drie Berken Zudde, 84% for Koole and 58% for Brampjesgat; calculated using annual net CH_4 emissions from individual plots within a site). High-emission-plots remained high-emission-plots and low-emission-plots remained low-emission-plots throughout the measurement period (not shown). The time course of coefficients of variation within a site (not shown) indicated that spatial variability of CH_4 emissions was rather constant throughout the year.

Results of vegetation and soil analyses (means of the upper 20 cm of the profile) of the main plots showed a high variability (Table 3.1). Highest CH_4 production and CH_4 consumption capacities were found in the upper layers (Fig. 3.2).

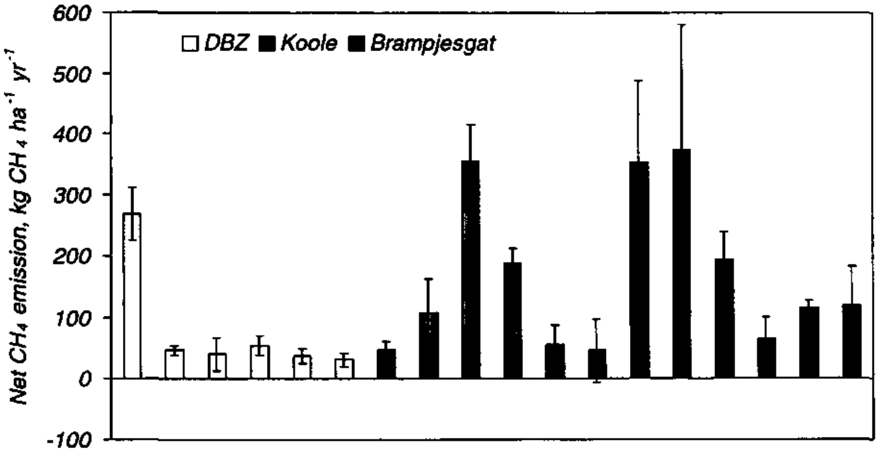

Fig. 3.1. Mean annual net CH_4 emissions (kg CH_4 ha^{-1} yr^{-1} ± SD) (avg. of 1994, 1995 and 1996) from plots at the sites Drie Berken Zudde (DBZ), Koole and Brampjesgat.

Table 3.1. Averages and coefficients of variation of variables determined at Drie Berken Zudde (DBZ) (*n*=6), Koole (*n*=6), Brampjesgat (*n*=6) and at all three sites (*n*=18). All soil characteristics are means of the upper 20 cm of the soil profile.

	DBZ		Koole		Brampjesgat		All	
	avg	CV,%	avg	CV,%	avg	CV,%	avg	CV,%
Annual CH_4 emission, kg CH_4 ha^{-1} yr^{-1}	79	107	133	84	204	58	139	85
CH_4 production capacity, ng CH_4 g^{-1} dry soil h^{-1}	5	98	162	108	466	65	211	131
CH_4 consumption capacity, ng CH_4 g^{-1} dry soil h^{-1}	484	27	537	44	442	28	487	36
Anaerobic CO_2 production capacity, µg CO_2 g^{-1} dry soil h^{-1}	44	13	54	8	n.d.[1]		49	15
Aerobic CO_2 production capacity, µg CO_2 g^{-1} dry soil h^{-1}	57	15	53	18	n.d.[1]		55	17
Loss-on-ignition, % of dry matter	92	4	76	14	64	5	77	17
Dissolved organic carbon, mg kg^{-1} dry soil	n.d.[1]		n.d.[1]		1540	15	n.d.[1]	
pH	3.5	3	4.9	8	5.4	3	4.6	19
N-total, g kg^{-1}	15	7	16	17	16	4	15	12
P-total, g kg^{-1}	0.6	8	0.9	20	1.5	5	1.0	38
Dry bulk density, mg cm^{-3}	140	21	147	45	153	16	147	30
Aboveground biomass, g dm^{-2}	6.0	14	3.9	19	4.8	24	4.9	26
Grass, g dm^{-2}	0.97	45	0.93	69	1.62	42	1.17	58
Sedges (*Carex* spp.), g dm^{-2}	0.11	142	0.63	116	0.63	82	0.45	127
Rushes (*Juncus* spp.), g dm^{-2}	0.05	224	0.08	75	0.69	110	0.27	195
Reed (*Phragmites* australis), g dm^{-2}	0		0		0.31	171	0.1	329
Mosses, g dm^{-2}	4.89	11	2.15	52	1.29	74	2.77	64
Roots, mg cm^{-3}	67	10	n.d.[1]		30	24	48	41

[1] n.d. = not determined

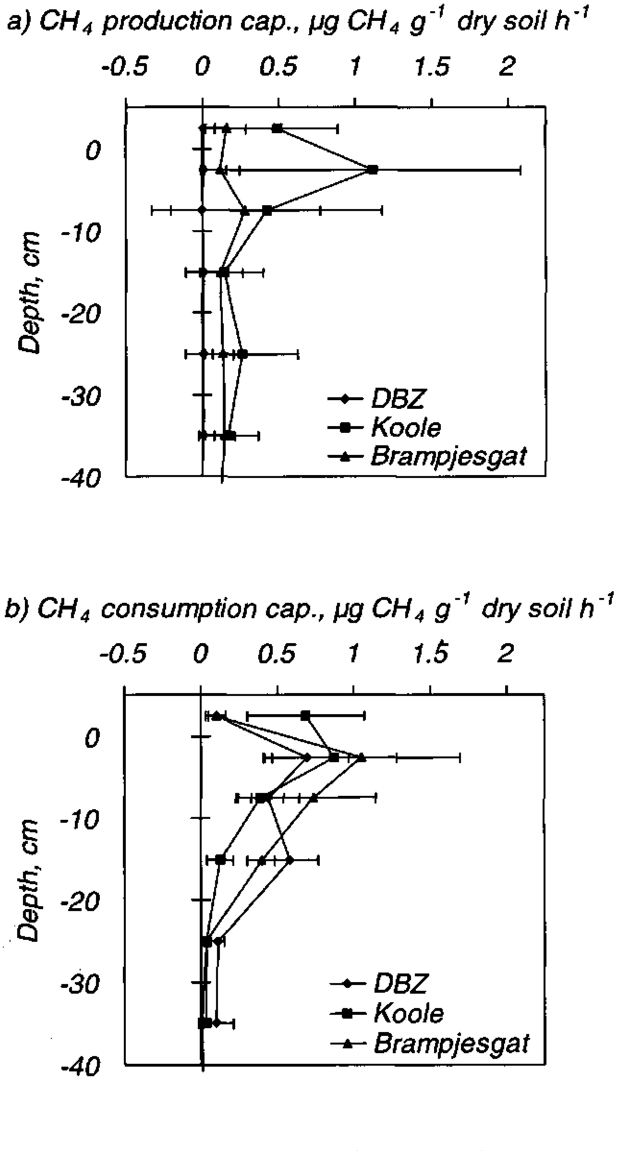

Fig. 3.2. **a**) CH_4 production capacity (µg CH_4 g^{-1} dry soil h^{-1}) and **b**) CH_4 consumption capacity (µg CH_4 g^{-1} dry soil h^{-1}, initial CH_4 concentration of 100 µL L^{-1}) in different soil layers of the sites Drie Berken Zudde (DBZ), Koole and Brampjesgat (layer above 0 cm is top layer of recently died plant material) at incubation at 20 °C.

Determinants of spatial variability of CH_4 emissions

In the experiment with additional plots, studying the effect of individual plant species on spatial variability of CH_4 emissions, CH_4 emissions were best related to *Juncus* spp. ($P<0.01$) (Fig. 3.3). Correlations with other aerenchymatous plant species, like *Carex* spp. were not significant, but this may have been due to the limited amount of *Carex* present. Total aboveground biomass also showed a significant relation with CH_4 emissions ($P<0.05$). Since relations between individual plant species and CH_4 emissions were in general poor, we decided to determine biomass per vegetation type in our detailed study of the main plots, instead of per individual species.

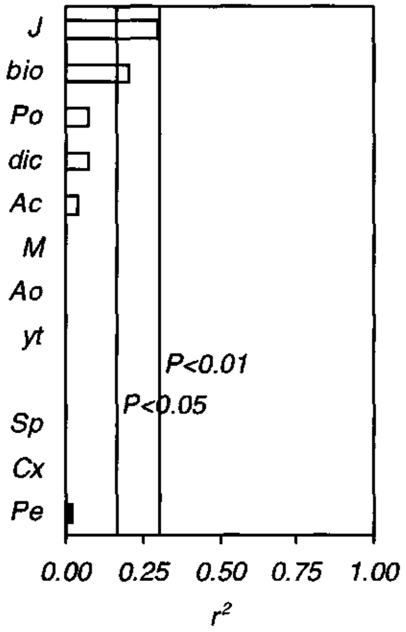

Fig. 3.3. Coefficients of determination (r^2) of simple linear regression analyses between CH_4 emissions (mg CH_4 m^{-2} d^{-1}) at the site Koole and aboveground biomass of several plant species (g dm^{-2}). Light bars indicate a positive and dark bars a negative relationship.
J = *Juncus* spp.; *bio* = total aboveground biomass; *Po* = *Polytrichum* spp.; *dic* = Dicotyledones; *Ac* = *Agrostis canina* L.; *M* = *Molinia caerulea* (L.) Moench; *Ao* = *Anthoxanthum odoratum* L.; *yt* = young tree species; *Sp* = *Sphagnum* spp.; *Cx* = *Carex* spp.; *Pe* = *Potentilla erecta* (L.) Räuschel.

For the main plots, mean annual CH_4 emissions were positively related to CH_4 production and CH_4 consumption capacities ($P<0.05$), except for CH_4 consumption capacity at Drie Berken Zudde. CH_4 production capacity was positively related to CH_4 consumption capacity ($P<0.05$) (not shown).

Main determinants of CH_4 emissions differed among sites (Fig. 3.4). At Drie Berken Zudde, CH_4 emissions were best correlated with the aboveground biomass of rushes and with CH_4 production capacity ($P<0.01$). At Koole, CH_4 emissions were best correlated with CH_4 consumption and CH_4 production capacities ($P<0.01$). Furthermore, both aerobic CO_2 production capacity and aboveground biomass of sedges were positively correlated with CH_4 emissions ($P<0.05$). The significant relation between CH_4 emissions and aerobic CO_2 production capacity suggests the importance of metabolisable organic matter in the top soil. At Brampjesgat, CH_4 emissions were best correlated with CH_4 consumption capacity and aboveground biomass of sedges ($P<0.01$). Other important factors were CH_4 production capacity, total aboveground biomass and aboveground biomass of reed ($P<0.05$).

Combination of all sites (Fig. 3.5) shows that CH_4 emissions were positively related to CH_4 production capacity, aboveground biomass of sedges ($P<0.01$), CH_4 consumption capacity and pH ($P<0.05$), and negatively to aboveground biomass of mosses ($P<0.05$). The main determinant of spatial variability of CH_4 emissions, other than CH_4 production or CH_4 consumption capacity, was aboveground biomass of sedges (r^2 of 0.60, $n=18$):

$$Y = 67 + 32 * biomass_{sedges} \tag{3.1}$$

Y = mean annual CH_4 emission, kg CH_4 ha^{-1} yr^{-1};
$biomass_{sedges}$ = aboveground biomass of sedges, g dm^{-2}.

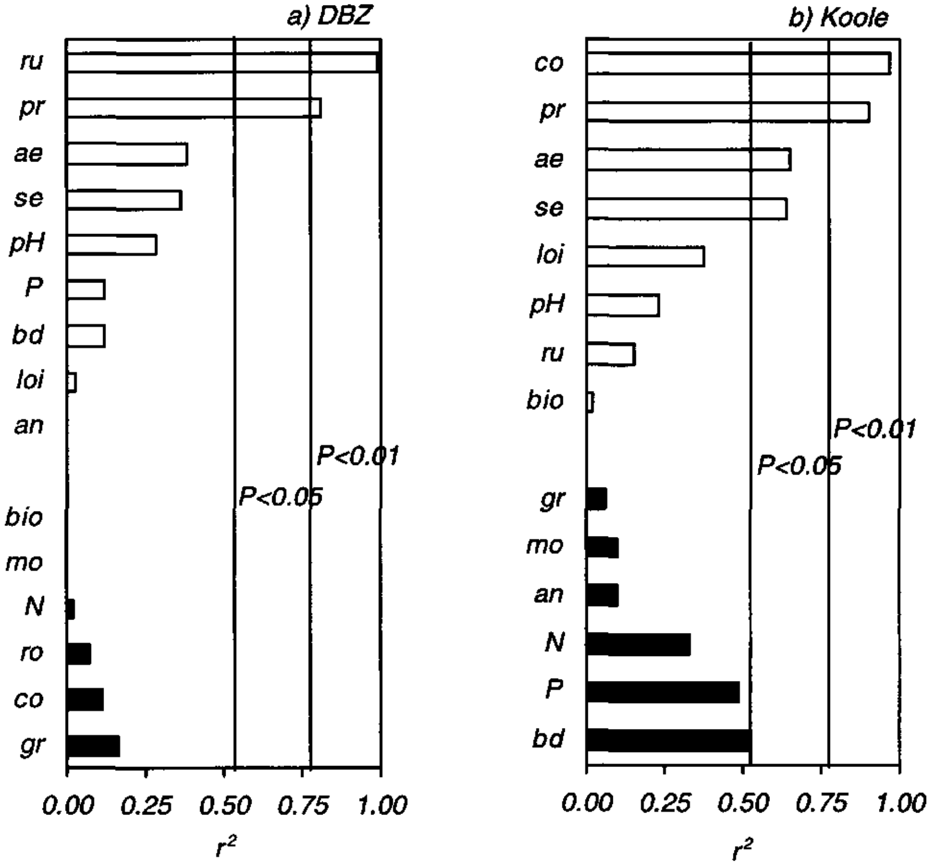

Fig. 3.4. Coefficients of determination (r^2) of simple linear regression analyses between mean annual net CH_4 emissions (kg CH_4 ha^{-1} yr^{-1}) and several variables at the sites **a**) Drie Berken Zudde (DBZ) (*n*=6), **b**) Koole (*n*=6), and **c**) Brampjesgat (*n*=6) (see next page). All soil characteristics are means of the upper 20 cm of the soil profile. Light bars indicate a positive and dark bars a negative relationship.

ae = aerobic CO_2 production capacity, mg CO_2 g^{-1} dry soil h^{-1}; *an* = anaerobic CO_2 production capacity, mg CO_2 g^{-1} dry soil h^{-1}; *bd* = dry bulk density, mg cm^{-3}; *bio* = total aboveground biomass, g dm^{-2}; *co* = CH_4 consumption capacity, µg CH_4 g^{-1} dry soil h^{-1}; *doc* = dissolved organic carbon, mg kg^{-1} dry soil; *gr* = grasses, g dm^{-2}; *loi* = loss-on-ignition, % of dry matter; *mo* = mosses, g dm^{-2}; *N* = total nitrogen, g kg^{-1}; *P* = total phosphorus, g kg^{-1}; *pH* = pH; *pr* = CH_4 production capacity, µg CH_4 g^{-1} dry soil h^{-1}; *re* = reed, g dm^{-2}; *ro* = roots, mg cm^{-3}; *ru* = rushes, g dm^{-2}; *se* = sedges, g dm^{-2}.

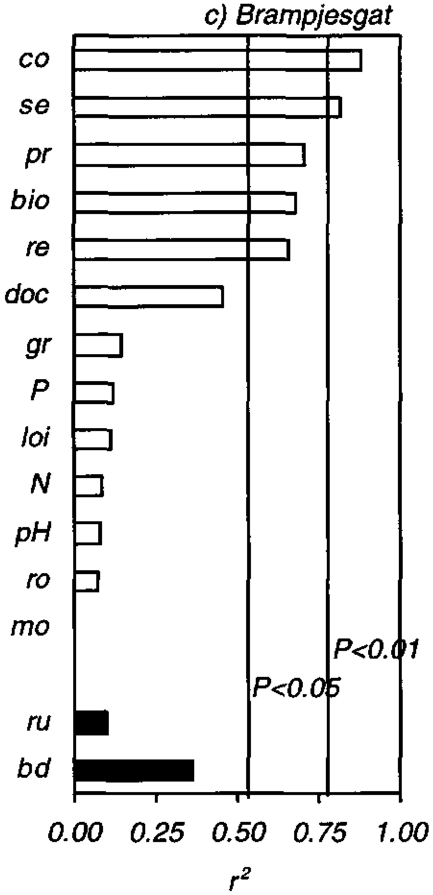

Fig. 3.4. c) Coefficients of determination (r^2) of simple linear regression analyses between mean annual net CH_4 emissions (kg CH_4 ha^{-1} yr^{-1}) and several variables at the site Brampjesgat (n=6) (see also previous page).

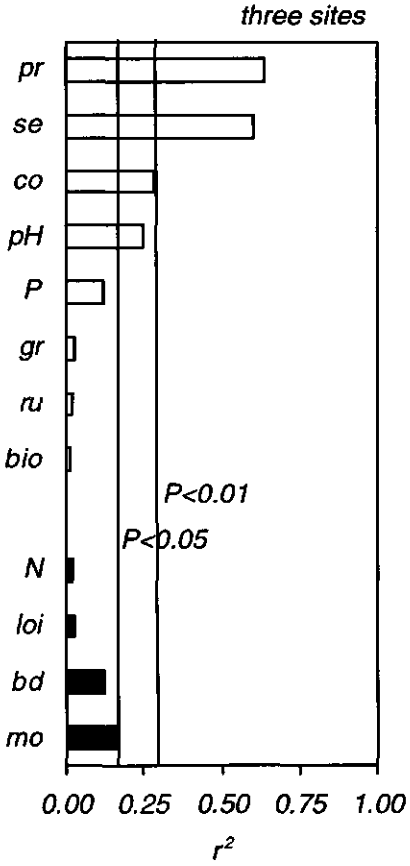

Fig. 3.5. Coefficients of determination (r^2) of simple linear regression analyses between mean annual net CH_4 emissions (kg CH_4 ha^{-1} yr^{-1}) from the three sites (n=18) and several variables. All soil characteristics are means of the upper 20 cm of the soil profile. Light bars indicate a positive and dark bars a negative relationship. Abbreviations are explained in Fig. 3.4.

Discussion

Spatial variability of CH_4 emissions

Large spatial variability of CH_4 emissions from wetland systems is a common phenomenon (e.g. Bartlett & Harriss, 1993; Bubier et al., 1993; Waddington & Roulet, 1996; Fig. 3.1). However, quantification of determinants of spatial variability of CH_4 emissions is poor. So far, CH_4 emission measurements over several years had not been combined with a detailed study of the plots including incubation experiments, soil and vegetation analyses. Three years of measuring CH_4 emissions from 18 main plots showed that the impact of variables may be rather stable throughout the year as plots with a relative high emission always showed a relative high emission and plots with a relative low emission always showed a relative low emission.

CH_4 production and CH_4 consumption capacity

CH_4 production and CH_4 consumption capacity were highly variable (Table 3.1, Fig. 3.2). We found that CH_4 emissions were significantly related to both CH_4 production capacity and CH_4 consumption capacity (Fig. 3.4 and 3.5). Therefore, CH_4 production and consumption capacities may serve as predictors of CH_4 emissions. The significant relation between CH_4 emissions and CH_4 consumption capacity may be explained by the significant relation between CH_4 production capacity and CH_4 consumption capacity. The latter may be explained by the dependence of CH_4 consumption capacity on CH_4 concentration in the soil.

Methanogenesis contributes to C mineralisation, thus CH_4 production capacity may be positively related to CO_2 production capacity (Moore & Dalva, 1997; Schimel, 1995; Yavitt & Lang, 1990). CO_2 production capacity is an estimate of total microbial activity, while CH_4 production capacity is an estimate of the activity of methanogens. We found a significant positive relation between CH_4 production capacity and CO_2 production capacity at Koole. We also found that the variability of CH_4 production capacity was larger than the variability of CO_2 production capacity (Table 3.1). This indicates that methanogens make higher demands on quality of their substrates than the combined total microbial biomass.

If we assume that methanogenesis produces 1 mole of CH_4 and 1 mole of CO_2, which is true when methanogenesis proceeds via acetate fermentation and no accumulation of fermentation products occur, then carbon mineralisation via methanogenesis was only about 1% of total anaerobic carbon mineralisation in the top layers and 5 to 10% in the deeper layers. The increase in the contribution of methanogenesis to carbon mineralisation with depth may have been caused by increasing anaerobiosis in the field with depth, and an accompanying relative increase in the size of the methanogenic population. In a review of CH_4 production, Segers (1998) found that for peat samples the ratio anaerobic C mineralisation to CH_4 production may vary as much as two or three orders of magnitude.

Oxygen accelerates decomposition of dead plant material. The increase in CO_2 production rates under aerobic compared with anaerobic conditions (Table 3.1) indicates the importance of water table and anaerobism in slowing decomposition rates. Peats which are more decomposed contain less relatively fresh plant material for methanogens and may yield lower CH_4 emissions.

Soil characteristics

Organic matter has been reported to be a good predictor of CH_4 production capacity in peatlands, with highest CH_4 emissions where organic matter is relatively labile and lowest CH_4 emissions where organic matter is relatively recalcitrant (Yavitt & Lang, 1990). Crozier et al., (1995) and Yavitt & Lang (1990) found a significant relation between loss-on-ignition and CH_4 emissions. In our sites, loss-on-ignition was high, and therefore it may not have been a limiting factor for CH_4 production. Schimel (1995) believes that soil organic matter is the main substrate for methanogens, while others believe that recently died plant material is the main substrate (e.g. Chanton et al., 1995; Whiting & Chanton, 1993). Chanton et al. (1995) suggest that the main source of organic matter for methanogens is recently fixed organic compounds, most likely dissolved organic compounds produced from the decay of recently produced litter, roots and root exudation products. In correspondence with this, we found a better relationship between dissolved organic carbon and CH_4 emissions than between loss-on-ignition and CH_4 emissions at Brampjesgat. Unfortunately, dissolved organic carbon was not measured at the other two sites.

In incubation experiments, Wang et al. (1993) showed that CH_4 emissions were highest around neutral pH. Dunfield et al. (1993) found that both CH_4 production and CH_4 consumption capacities were optimal at about 2 pH units higher than the native peat pH in acidic peats (pH<5) and 0-1 pH units higher in the more alkaline peats. They suggested that methanogens and methanotrophs are only partially adapted to acidic conditions. Jugsujinda et al. (1996) showed that soil pH was the dominant variable which influenced organic matter decomposition, low soil Eh conditions and subsequent CH_4 production in flooded acid sulfate soils. In their experiments, soils with pH below 6.1 produced no significant quantities of CH_4. Even though the pH of our plots was much lower, especially at Drie Berken Zudde (Table 3.1), the plots showed significant CH_4 production and CH_4 emission. We found a significant positive relation between CH_4 emissions and pH, but only at the combination of the three sites (Fig. 3.5). Ranges of pH at individual sites were probably too low to detect any effects.

Vegetation

Correlation between CH_4 emissions and plant biomass has been found in several studies (e.g. Chanton et al., 1993; Whiting & Chanton, 1993). Plant biomass may affect CH_4 emissions, as organic material from plants may serve as substrate for methanogens. Especially roots may influence CH_4 production through supply of organic matter at depth via root decay and root exudation. Furthermore, plant species with aerenchymatous tissues may directly transport CH_4 from the anaerobic zone of the soil to the atmosphere (e.g. Schimel, 1995), thereby protecting it against oxidation. The effect of plants may however be ambiguous, as plants may also transport O_2 to deeper layers, leading to increased CH_4 oxidation. Plant communities may also control CH_4 fluxes in peatland ecosystems through indirect effects on the water table (e.g. Bubier et al., 1995a; Bubier et al., 1995b).

In our study, vegetation analysis yielded good results in describing spatial variability of CH_4 emissions. CH_4 emissions from the additional plots were best related to *Juncus* spp. (Fig. 3.3). CH_4 emissions from the main plots were significantly related ($P<0.01$) to aboveground biomass of sedges (Equation 3.1; Fig. 3.5). Strong correlations between sedge biomass and CH_4 emissions have been observed before (e.g. Klinger et al., 1994; Whiting & Chanton, 1992). Torn & Chapin (1993) suggested that sedges are highly effective in CH_4 transport not only because they provide a conduit to the soil surface, but also because their large root surface and air-filled aerenchyma provide both an effective collection system and rapid diffusion path. Further on, due to the large root system of sedges, which may penetrate to more than 2 m depth (Saarinen, 1996), sedges may stimulate CH_4 production by serving as a substrate for methanogens.

Vegetation is affected by management and environmental factors like ground water level, nutrient status and climate. These factors each have their influence on CH_4 emissions. Therefore, it has been suggested (e.g. Bubier et al., 1995b) that vegetation may have predictive value for CH_4 emissions. The use of vegetation in predicting CH_4 emissions has many advantages: (i) it is much easier to determine the vegetation type of a certain area than to measure all the individual factors; (ii) vegetation composition is rather stable throughout the year, while environmental factors may show considerable temporal variability; and (iii) vegetation may be remotely sensed.

Extrapolation

Our results show that spatial variability of CH_4 emissions can be described by vegetation and soil characteristics. The effect of these variables on CH_4 emissions was mainly according to what we expected, although relations were often weak (Fig. 3.4 and 3.5). This may be explained by the relatively small number of measurements per site (n=6), which may be too small to reveal possible relationships. The relative importance of influencing factors may vary independently and on different spatial scales. Probably factors, which influence CH_4 production, are the primary factors associated with spatial variability of CH_4 emissions. If conditions are favourable for CH_4 production, then secondary factors like temperature and CH_4 transport ways become important.

We found that determinants of spatial variability of CH_4 emissions differed among sites. Most important influencing factors were CH_4 production capacity and aboveground biomass of sedges (Fig. 3.5). For extrapolation to larger areas, best results will be obtained by using factors which are easy to determine, like vegetation. When vegetation is used to predict CH_4 emissions, the most important factors influencing CH_4 emissions are considered to be also the most important ones influencing species distribution and biomass density. At the same time, vegetation also affects determinants of CH_4 emissions like ground water level. However, since vegetation is affected by several hydrological and physiological parameters, which also affect the processes CH_4 production, CH_4 consumption and CH_4 transport, *vegetation can be a good predictor of CH_4 emissions.*

Acknowledgements

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Chapter 4

Methane production and carbon mineralisation of size and density fractions of peat soils

Van den Pol-van Dasselaar A & Oenema O (1998)
Submitted to *Soil Biology & Biochemistry*

METHANE PRODUCTION AND CARBON MINERALISATION OF SIZE AND DENSITY FRACTIONS OF PEAT SOILS

Abstract

For the purpose of characterisation of soil organic matter breakdown, soil organic matter is often divided into different fractions, each with its own decomposition rate. Thus far, no attempts had been made to quantify methane (CH_4) production capacity of individual soil fractions. This study aimed to improve our understanding of CH_4 emissions from peat soils by studying CH_4 production capacity of individual soil fractions in coherence with carbon (C) mineralisation capacity. Samples from two wet grasslands on peat soil (0-60 cm) were fractionated into different size and density fractions using sieves and Ludox, an aqueous colloidal dispersion of silica particles, respectively. The individual fractions were rather similar with respect to C mineralisation capacity and C/N ratio, but not with respect to CH_4 production capacity. We found that significant CH_4 production only occurred for fractions with a large particle size. Furthermore, CH_4 production capacity strongly decreased with depth. This indicates that in the wet peat soils recently-died plant material is a major substrate for methanogens.

Introduction

Methane (CH_4) is a greenhouse gas, which may contribute to the enhanced greenhouse effect. CH_4 emissions from wetlands contribute between 10 and 28% of total global CH_4 sources (IPCC, 1995a). Estimates of CH_4 emissions from soils and estimates of the effect of management and climatic factors on CH_4 emissions from soils have a high uncertainty, since the controls of the underlying processes CH_4 production, CH_4 consumption and CH_4 transport are not well-known. While flux measurements are clearly required to obtain reliable estimates of CH_4 emissions, it may be equally important to study the mechanisms of the subsequent processes production, consumption and transport. This study aimed to improve our understanding of CH_4 emissions from soils by studying CH_4 production capacity in coherence with carbon (C) mineralisation capacity.

CH_4 production may occur wherever organic matter is degraded under anaerobic conditions. It is generally accepted that CH_4 production, which is part of total C mineralisation, is influenced by both quality and quantity of organic matter present (e.g. Crozier et al., 1995; Schimel, 1995; Whiting & Chanton, 1993; Yavitt & Lang, 1990). However, the relationship between CH_4 production and total C mineralisation is highly variable. For example, the ratio of anaerobic C mineralisation to CH_4 production in peat samples may vary as much as two or three orders of magnitude (Segers, 1998). Furthermore, the source of methanogenic substrates is uncertain. Some believe that soil organic matter is the main substrate for methanogens (e.g. Schimel, 1995), while others believe that recently died or fresh plant material (either the current or the previous year's production) is the main substrate (e.g. Whiting & Chanton, 1993).

In simulation models, soil organic matter is often divided into several fractions each with its own decomposition rate, for example easily decomposable material, which merely consists of recently died plant material, and material more resistant to decomposition (e.g. Jenkinson & Rayner, 1977; Van Veen & Paul, 1981). However, this division is often arbitrary and not based on direct determinations. During the last decades, several attempts have been made to find a useful physical way to divide soil organic matter (e.g. Cambardella & Elliott, 1993; Meijboom et al., 1995; Skjemstad et al., 1986; Tiessen & Stewart, 1983).

In this study, we used a modified version of the size and density fractionation method of Meijboom et al. (1995) to divide peat soils into different fractions, because it is, unlike the majority of other physical fractionation methods, a fast, cheap and non-toxic method, that does not alter the characteristics of the organic matter fractions. Methane production capacity and C mineralisation capacity were assessed for each individual fraction. C mineralisation capacities of individual size and density fractions have been measured before (e.g. Hassink, 1995b), but thus far no attempts have been made to quantify CH_4 production capacity of individual fractions.

The concept behind size and density fractionation assumes that soil organic matter can be divided into pools differing in structure and function. The coarse, light fractions will consist of relatively young material, which is not yet associated with soil material, while the fine, heavy fractions will consist of relatively old material, which is associated with soil. The coarse, light fractions will have the highest decomposition rate. We hypothesise that CH_4 production capacity is linearly related to C mineralisation capacity and that fractions with relatively young material have the highest CH_4 production capacity. We selected two wet peat soils to fractionate, because they have a potentially high CH_4 production capacity, and thus far the method of Meijboom et al. (1995) had not been used for peat soils. The method has been proven useful for clay, sandy and loamy soils.

Materials and methods

Site description and soil sampling

Soil samples were taken from two wet grasslands, Koole and Drie Berken Zudde, on peat soil in the "Nieuwkoopse Plassen" area in the Netherlands. During several centuries this area has been used for peat mining and agriculture. At present, it is a nature preserve with narrow grassland and reed fields, surrounded by ditches. Our sites can be classified as fens, i.e. minerotrophic peat soils. Vegetation is dominated by grasses, mosses, sedges, rushes and reed, and is mown once every year. Mean annual ground water level is 9 cm below the surface at Koole and 18 cm below the surface at Drie Berken Zudde, and pH-H_2O in the layer 0-20 cm is 4.9 at Koole and 3.5 at Drie Berken Zudde. In 1994-1996, mean annual net CH_4 emissions were 133±120 kg CH_4 ha^{-1} yr^{-1} at Koole and 79±88 kg CH_4 ha^{-1} yr^{-1} at Drie Berken Zudde (Van den Pol-van Dasselaar et al., 1998b).

From each site, three cores were taken along a transect perpendicular to the ditch on 30 November 1995. Immediately after coring a profile of redox potential was determined (Fig. 4.1). Soil temperature was 5°C at 10 cm depth. Ground water level was 3.5 cm below the surface at Koole, and 17.5 cm below the surface at Drie Berken Zudde. Vegetation of the soil cores was removed. The soil cores were subsampled in the layers 0-5 cm, 5-10 cm, 10-20 cm, 20-30 cm, 30-40 cm, and 40-60 cm, and stored for about a month at 4°C in the dark, until fractionation.

Size and density fractionation

Size and density fractionation of soil samples was carried out according to the method of Meijboom et al. (1995) with some modifications: we used an additional mesh sieve of 2 mm, and a different density of Ludox. In brief, carefully homogenised field-moist soil samples (500 g) were wet-sieved over a 2 mm, a

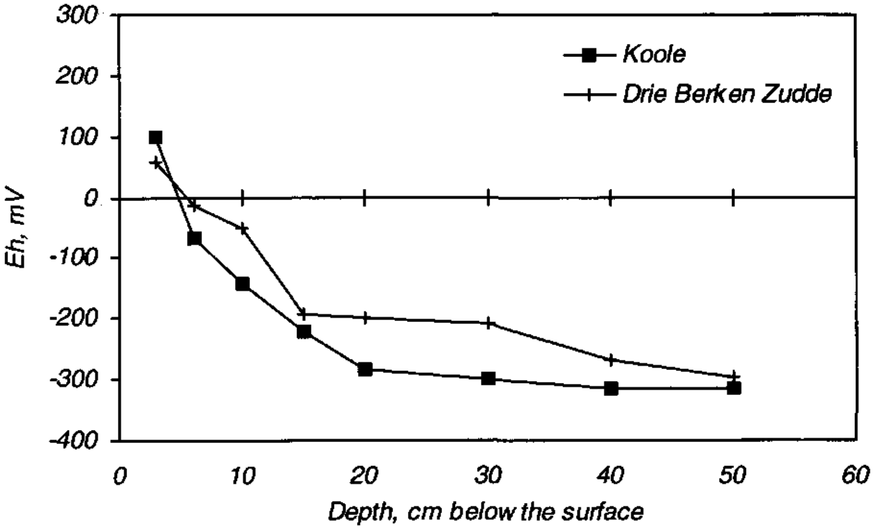


Fig. 4.1. Redox potential (mV) at several depths (cm below the surface) of Koole and Drie Berken Zudde.

250 µm and a 150 µm mesh sieve. The material present on the 250 and 150 µm sieve was fractionated in Ludox TM, which is an aqueous colloidal dispersion of silica particles. The material was placed in Ludox with a density of 1.20 g cm^{-3} and was stirred. In this way, the material was separated into a floatable or light fraction, and into a sinking or heavy fraction. The density of 1.20 g cm^{-3} was chosen as an intermediate between 1.0 (approximately the density of fresh plant material; Meijboom et al., 1995) and 1.37 (approximately the maximum density of Ludox). Finally, six fractions were obtained:

>2.0 mm;
0.25-2.0 mm-L(ight);
0.25-2.0 mm-H(eavy);
150-250 µm-L(ight);
150-250 µm-H(eavy);
<150 µm.

For one soil core per site, the fraction <150 µm was further subdivided into 20-150 µm and <20 µm. Each fraction was analysed for loss-on-ignition at 550°C. C and N contents were determined by total C and N analysers (Carlo Erba NA1500) after combustion at 1020°C.

C mineralisation capacity

C mineralisation capacity was measured by incubating moist samples aerobically at 20°C in 1.5-l airtight jars containing a vial of 10 ml 0.5 *M* NaOH. At the sampling dates, the trapped CO_2 was measured after precipitation of the carbonate with excess $BaCl_2$. A flush in CO_2 production was observed during the first 10 days, probably caused by the disturbance of the soil. The period between 10 and 20 days after the start of the incubation was used to calculate the C mineralisation rate.

We believe that for our peat soils aerobic C mineralisation capacity is a rather good estimate of anaerobic C mineralisation capacity, since previous incubation experiments showed that the ratio aerobic C mineralisation capacity to anaerobic C mineralisation capacity was about 1.30 at the site Drie Berken Zudde and about 1.0 at Koole (Van den Pol-van Dasselaar et al., 1998c).

CH_4 production capacity

Anaerobic incubation experiments (N_2-atmosphere, 20°C) were performed to determine CH_4 production capacity. After a pre-incubation period of several days, the individual fractions were incubated in 580 ml glass bottles. Gas samples from the headspaces of the bottles were taken on average once a week for four weeks, and analysed for CH_4 by gas chromatography using a flame ionisation detector (coefficient of variation: 0.08%). CH_4 production capacity was calculated by linear regression on the time course of CH_4 concentration in the headspace of the bottles.

Statistics

Statistical differences were tested with an analysis of variance procedure (ANOVA) (Genstat, 1993) with factors of site (Koole, Drie Berken Zudde), layer (0-5, 5-10, 10-20, 20-30, 30-40, and 40-60 cm), and fraction (>2.0 mm, 0.25-2.0 mm-L, 0.25-2.0 mm-H, 150-250 μm-L, 150-250 μm-H, and <150 μm) (P=0.05).

Results

Size and density fractionation

Both sites were high in organic matter (Table 4.1). Total amount of C was higher at Drie Berken Zudde than at Koole (*P*<0.05). The amount of C decreased with depth to about 10-30 cm, and increased again in deeper soil layers. However, differences

Table 4.1. Amount of C in individual soil layers (g C kg^{-1} dry soil), and amount of C in individual size and density fractions (g C kg^{-1} dry soil ± SD) (L = light fraction, H = heavy fraction) (*n*=3) at Koole and Drie Berken Zudde (DBZ).

	0-5	5-10	10-20	20-30	30-40	40-60	0-60
Individual soil layers							
Koole	299±96	276±29	275±65	306±97	328±62	384±44	327±54
DBZ	415±7	413±8	397±33	360±66	379±48	397±4	389±14
Individual fractions							
Koole							
>2.0 mm	145±132	108±51	53±42	83±87	109±65	181±40	122±48
0.25-2.0 mm-L	23±4	26±11	35±26	49±32	54±18	51±17	44±8
0.25-2.0 mm-H	19±27	17±12	23±15	27±18	30±27	29±23	26±17
150-250 µm-L	8±2	5±1	4±2	8±7	10±10	11±6	8±4
150-250 µm-H	8±4	8±6	16±4	16±4	29±8	24±15	20±4
<150 µm	96±27	111±12	144±31	124±42	95±23	87±30	107
DBZ							
>2.0 mm	276±89	242±27	217±32	230±89	210±108	191±54	220±55
0.25-2.0 mm-L	21±13	33±16	21±6	24±9	37±20	62±34	34±19
0.25-2.0 mm-H	4±5	4±2	10±9	7±10	6±4	30±20	12±7
150-250 µm-L	8±7	8±3	5±1	10±4	14±3	9±4	9±1
150-250 µm-H	12±10	11±4	16±12	14±2	19±8	26±8	17±7
<150 µm	94±55	115±17	129±26	76±17	94±29	79±1	96±16

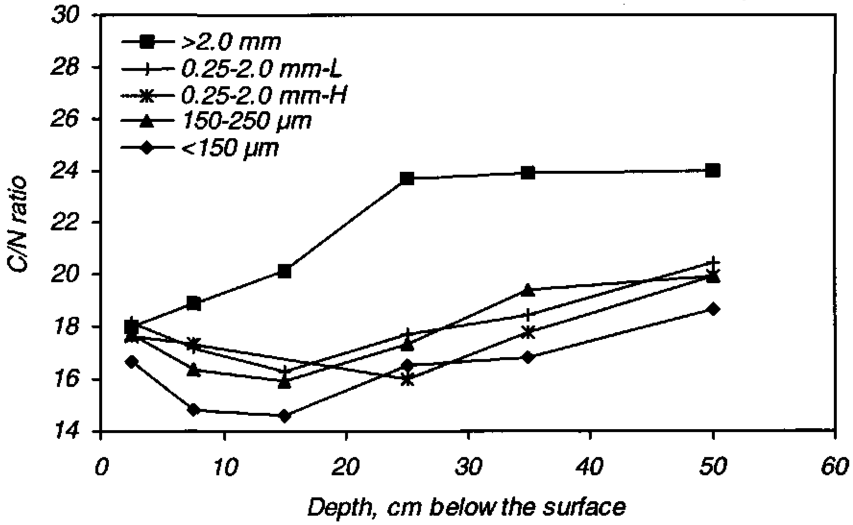
between layers were not significant. The fraction >2.0 mm was the major contributor to total C ($P<0.05$). It consists mainly of dead plant material, in which plant parts are still recognisable. The fraction <150 µm was the second most important contributor to total C ($P<0.05$). In soil samples where the fraction <150 µm was subdivided, we found that the contribution of the fraction 20-150 µm to total C was higher than the contribution of the fraction <20 µm (at Koole two times higher, and at Drie Berken Zudde five times higher) (not shown). The two size fractions which had been divided in a light and a heavy part showed contrasting results. In the size fraction 0.25-2.0 mm most C was found in the light part ($P<0.05$). In the size fraction 150-250 µm most C was found in the heavy part, but differences between the light and the heavy part were not significant ($P<0.05$). Ash content was rather low for all fractions (<40%). C/N ratio (Fig. 4.2) was higher at Drie Berken Zudde than at Koole, due to a higher C content in the samples of Drie Berken Zudde ($P<0.05$). N contents of all fractions at both sites were in the range of 1.7-1.9%. C/N ratio was highest for the fraction >2.0 mm ($P<0.05$). C/N ratios of light fractions were slightly higher than C/N ratios of heavy fractions ($P<0.05$).

C mineralisation capacity

All fractions showed C mineralisation capacity (Fig. 4.3). In the top layers, C mineralisation capacity was highest for the fraction >2.0 mm. In the deeper layers, there was no difference in C mineralisation capacity between individual fractions ($P<0.05$).

The contribution of individual fractions to total C mineralisation capacity is presented in Table 4.2. The layer 0-5 cm showed the largest contribution to total C mineralisation capacity ($P<0.05$). For the individual fractions, the fraction >2.0 mm contributed most to C mineralisation capacity ($P<0.05$). The light part of the size fraction 0.25-2.0 mm contributed more to total C mineralisation capacity than the heavy part of the size fraction 0.25-2.0 mm ($P<0.05$). In contrast, for the size fraction 150-250 µm the heavy part contributed more than the light part, but differences between the contribution of the heavy part and the light part were not significant ($P<0.05$).

a) Koole



b) Drie Berken Zudde

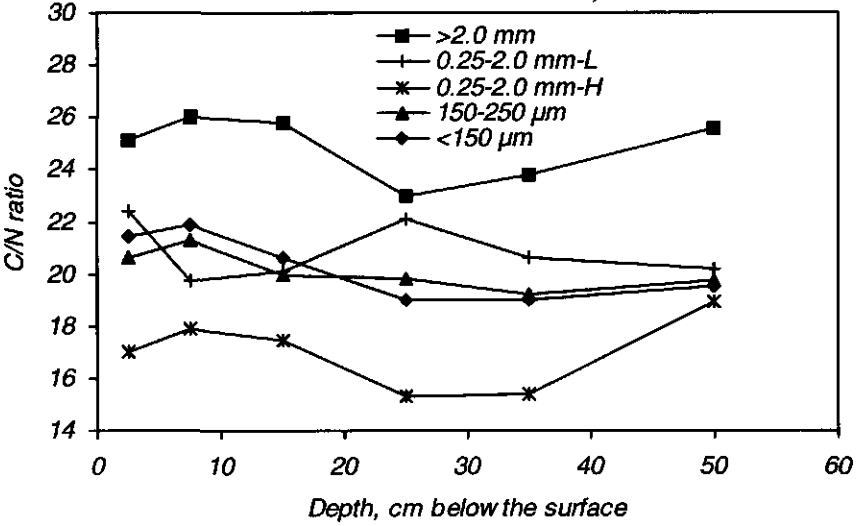


Fig. 4.2. C/N ratio of size and density fractions at several depths (cm below the surface) of **a**) Koole and **b**) Drie Berken Zudde (L = light fraction, H = heavy fraction).

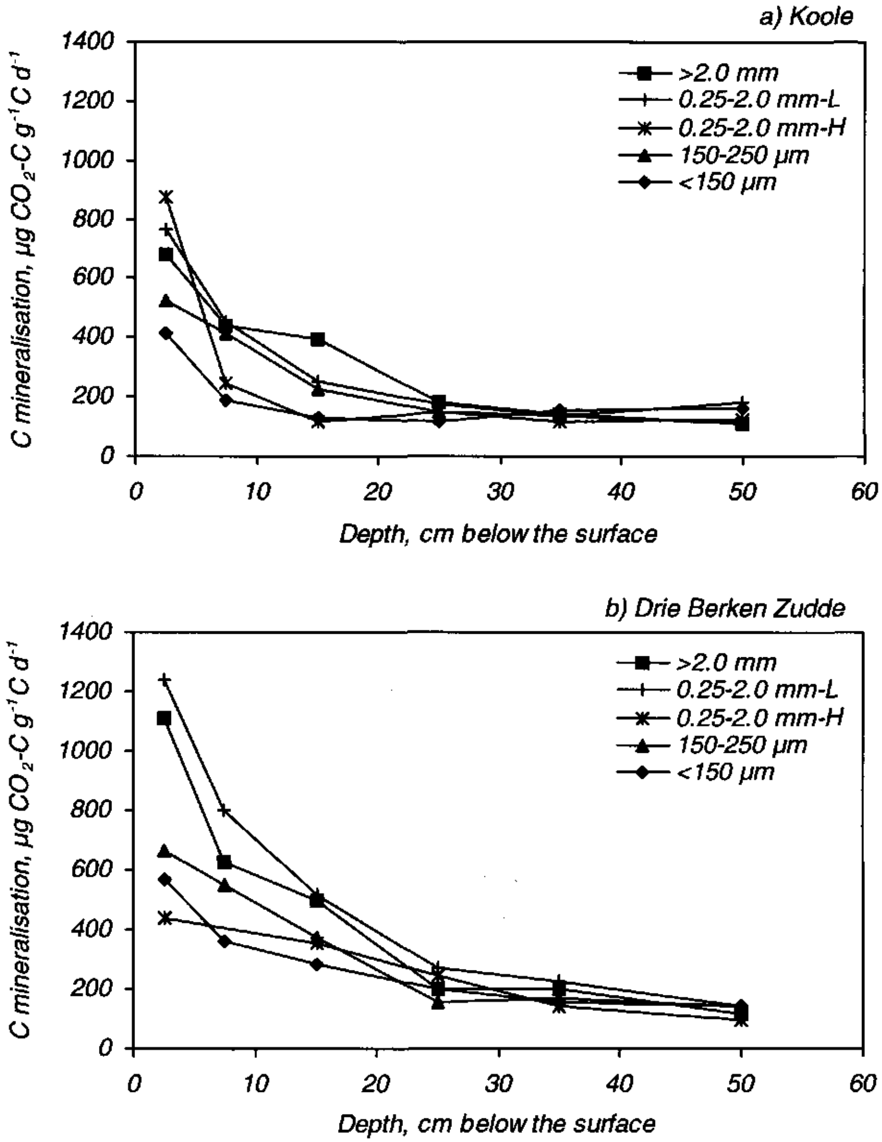


Fig. 4.3. C mineralisation capacity (μg CO_2-C g^{-1} C d^{-1}) of size and density fractions at several depths (cm below the surface) of **a**) Koole and **b**) Drie Berken Zudde (L = light fraction, H = heavy fraction).

Table 4.2. C mineralisation capacity of individual soil layers (% of total C mineralisation capacity of the layer 0-60 cm depth), and C mineralisation capacity of individual size and density fractions (% of total C mineralisation capacity of individual soil layers) (L = light fraction, H = heavy fraction) (*n*=3) at Koole and Drie Berken Zudde (DBZ).

	0-5	5-10	10-20	20-30	30-40	40-60	0-60
Individual soil layers							
Koole	39	19	12	10	10	11	100
DBZ	40	23	17	7	7	5	100
Individual fractions							
Koole							
>2.0 mm	55	53	38	33	33	39	43
0.25-2.0 mm-L	10	13	16	18	16	18	15
0.25-2.0 mm-H	9	5	5	9	8	7	7
150-250 µm-L	2	2	1	2	3	2	2
150-250 µm-H	3	5	7	6	9	6	5
<150 µm	22	23	34	32	32	28	28
DBZ							
>2.0 mm	76	66	65	63	59	45	65
0.25-2.0 mm-L	6	11	6	9	12	17	10
0.25-2.0 mm-H	0	0	2	2	1	6	2
150-250 µm-L	1	2	1	2	4	2	2
150-250 µm-H	2	3	4	3	4	8	4
<150 µm	13	18	22	20	21	22	19

CH_4 production capacity

Both between fractions and between layers, differences in CH_4 production capacity were larger than differences in C mineralisation capacity. The fraction >2.0 mm showed the highest CH_4 production capacity ($P<0.05$). CH_4 production capacity strongly decreased with depth (Fig. 4.4, note the log-scale). Unfortunately, there was

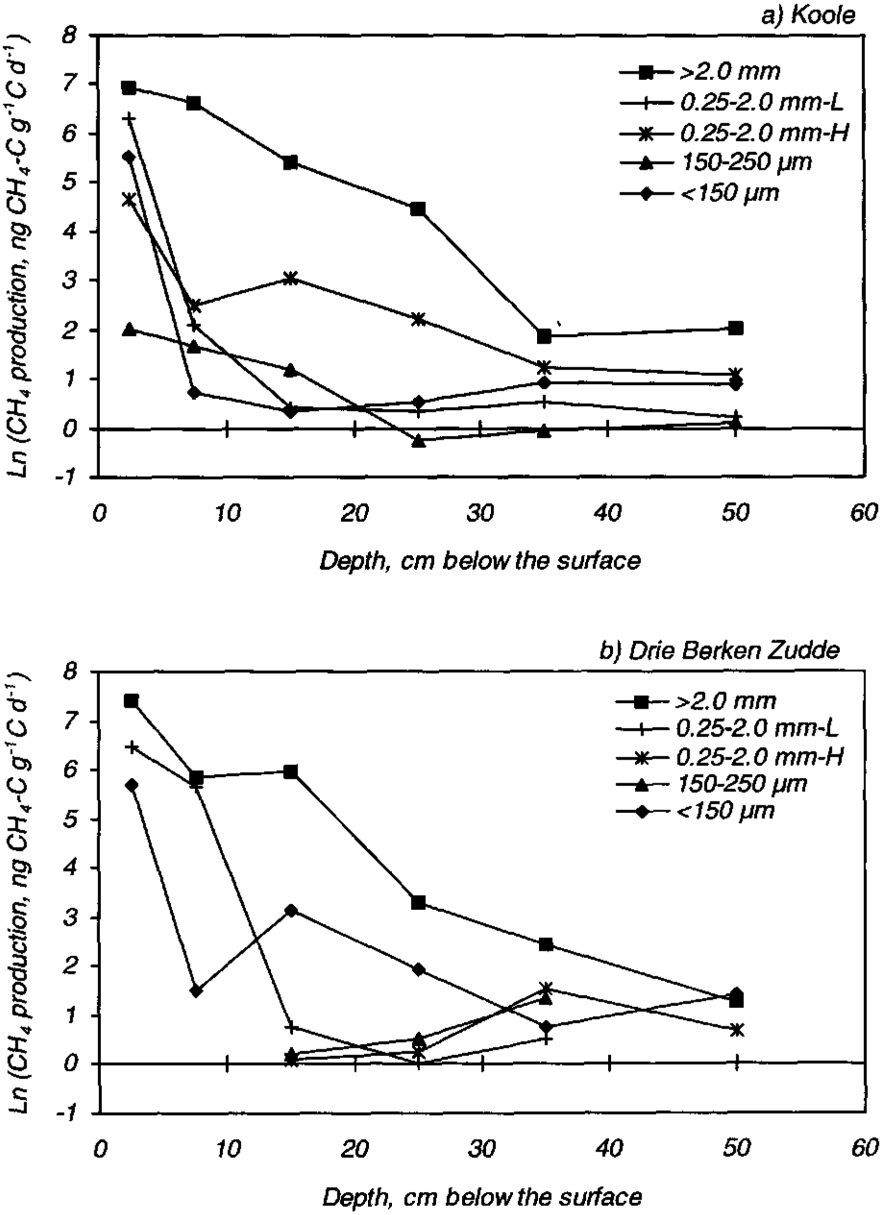


Fig. 4.4. Ln-transformed CH_4 production capacity (ng CH_4-C g^{-1} C d^{-1}) of size and density fractions at several depths (cm below the surface) of **a)** Koole and **b)** Drie Berken Zudde (L = light fraction, H = heavy fraction).

not enough material available to assess CH_4 production capacity of some fractions of the layers 0-5 and 5-10 cm of Drie Berken Zudde: 0.25-2.0 mm-H, 150-250 µm-L and 150-250 µm-H.

The layer 0-5 cm showed the largest contribution to total CH_4 production capacity ($P<0.05$) (Table 4.3). For the individual fractions, the fraction >2.0 mm contributed most to CH_4 production capacity ($P<0.05$). The light part of the size fraction 0.25-2.0 mm contributed more to total CH_4 production capacity than the

Table 4.3. CH_4 production capacity of individual soil layers (% of total CH_4 production capacity of the layer 0-60 cm depth), and CH_4 production capacity of individual size and density fractions (% of total CH_4 production capacity of individual soil layers) (L = light fraction, H = heavy fraction) (n=3) at Koole and Drie Berken Zudde (DBZ).

	0-5	5-10	10-20	20-30	30-40	40-60	0-60
Individual soil layers							
Koole	64	28	4	3	0	1	100
DBZ	72	14	13	1	0	0	100
Individual fractions							
Koole							
>2.0 mm	80	99	93	93	60	77	86
0.25-2.0 mm-L	7	0	0	1	8	4	4
0.25-2.0 mm-H	1	0	4	3	9	5	1
150-250 µm-L	0	0	0	0	0	0	0
150-250 µm-H	0	0	0	0	3	2	0
<150 µm	13	0	2	3	20	12	8
DBZ							
>2.0 mm	92	90	97	91	85	29	92
0.25-2.0 mm-L	3	10	0	0	2	46	3
0.25-2.0 mm-H	n.d.[1]	n.d.	0	0	1	2	0
150-250 µm-L	n.d.	n.d.	0	0	2	1	0
150-250 µm-H	n.d.	n.d.	0	0	2	8	0
<150 µm	6	1	3	8	7	14	5

[1] n.d. = not determined

heavy part ($P<0.05$). For the size fraction 150-250 µm, differences between the light and the heavy part were not significant ($P<0.05$).

Discussion

C content of size and density fractions

The peat soils were, as expected, high in organic matter and low in ash content. Ash contents of light fractions were lower than ash contents of heavy fractions ($P<0.05$) (not shown), indicating a separation of mineral-free plant remainders and more humified complexed soil organic matter. Differences were however smaller than found by Meijboom et al. (1995) for sandy, loamy and clayey soils. Furthermore, Hassink (1995a) found that C/N ratios of light fractions were higher than C/N ratios of heavy fractions for sandy, loamy and clayey soils. Although at our peat soils, C/N ratios of light fractions were significantly higher than C/N ratios of heavy fractions (Fig. 4.2), differences were smaller than found by Hassink (1995a). The relatively small differences in ash content and C/N ratios compared to sandy, loamy and clayey soils are probably a characteristic feature of peat soils. Either the decomposition process in peat soils did not lead to organic matter fractions differing in C/N ratio, or the fractionation method cannot discriminate between the possible fractions, or both. This suggests that the fractionation method of Meijboom et al. (1995) has less functional value for peat soils than for sandy, loamy and clayey soils, since the different fractions are more alike.

Variability of C/N ratio (Fig. 4.2) and amount of C in different size and density fractions (Table 4.1) with depth was surprisingly small. The absence of clear patterns with depth in amounts of C in the various size and density fractions again may suggest that the fractionation method has little functional value for peat soils, because visual differences and also C mineralisation capacities (Fig. 4.3) showed clear depth-related patterns. Alternatively, the diagenetically introduced variations in organic matter fractionation with depth may be masked by the variation introduced by the accretion of different vegetation types. Further studies are needed to single this out.

We expect that the vegetation of our wet peat soils has always been dominated by C_3 plants. This was confirmed by results of relative C isotope composition

($^{13}C/^{12}C$, expressed as $\delta^{13}C$) of individual fractions of one soil core (Dörsch, pers. com.). C_3 and C_4 plants have significantly different $\delta^{13}C$ values, with mean values of -26 and -12‰, respectively (Bender, 1968). The $\delta^{13}C$ value of our fractions ranged from -29 to -25‰. Differences in $\delta^{13}C$ between size and density fractions and between soil layers were small. Fractions 0.25-2.0 mm-H and >2.0 mm showed a somewhat higher $\delta^{13}C$ than the other fractions (Dörsch, pers. com.). These small differences may be attributed to differences in the composition of individual fractions. Carbohydrate, lignin, cellulose, lipids, etc. can have different $\delta^{13}C$ values. Therefore, differences in accumulation of these components in size and density fractions can influence the isotopic composition of the fractions (Golchin et al., 1995). Furthermore, microbial degradation may affect $\delta^{13}C$ values. It has been demonstrated that micro-organisms utilise ^{12}C in preference to ^{13}C, leading to a slight ^{13}C enrichment of the residual substrate. In many soils under C_3 vegetation, increasing values of $\delta^{13}C$ with depth have been observed (e.g. Balesdent et al., 1993). In contrast, $\delta^{13}C$ values of our peat soils did not change or even slightly decreased with depth. This may be a typical feature of C accumulating soils (Balesdent et al., 1993).

C mineralisation capacity

As expected, the fraction >2.0 mm, i.e. the fraction with recently died plant material, showed the highest C mineralisation capacity (Fig. 4.3, Table 4.2). Hassink (1995b) found for sandy, loamy and clayey soils, that C mineralisation capacity decreased in the order light fractions, heavy fractions, other fractions. Hassink (1995b) suggested that decomposition rate constants of individual fractions are not dependent on soil type. Our results show that this hypothesis does not hold for peat soils, since we found no significant differences between C mineralisation capacities of light and heavy parts of the particle size fraction 150-250 µm.

CH_4 production capacity

We hypothesised that the fractions with relatively young material would have the highest CH_4 production capacity. Indeed, highest CH_4 production capacity was found in the fraction >2.0 mm (Fig. 4.4, note the log-scale). Furthermore, this fraction was by far the biggest contributor to total CH_4 production capacity (Table 4.3). This is caused by both the relatively large amount of C in this fraction, and the high CH_4 production capacity per unit of C, especially in the top layers.

The main substrate for methanogens is thought to be either recently died plant material (e.g. Chanton et al., 1995; Whiting & Chanton, 1993) or soil organic matter (e.g. Schimel, 1995). Charman et al. (1994) suggested that at depth methanogens use an additional C source other than the adjacent peat because CH_4 at depth was younger of age than the surrounding peat. This additional C source may be dissolved organic C, either from root exudates or transported from the surface by water movement. Our results show that CH_4 production capacity is highest for fractions with a large particle size, and that CH_4 production capacity decreases with depth. This indicates that in our peat soils recently died plant material is a major substrate for methanogens. Plant material may be transported to deep layers via the root system. For example, Saarinen (1996) found living roots of *Carex*, one of the dominant species at our experimental sites, down to a depth of 230 cm. The identification of type and location of substrates for CH_4 production can improve explanatory models of CH_4 emissions and reduce the uncertainties of CH_4 emission estimates.

CH_4 production in relation to C mineralisation

CH_4 production capacity and C mineralisation capacity decreased with depth (Fig. 4.3 and 4.4), while the amount of organic matter did not (Table 4.1), indicating that the characteristics of organic matter change with depth. In the top layers, organic matter consists of recently died plant material that is relatively easy decomposable, while in the deeper layers organic matter will consist of plant parts that are rather resistant to decomposition, like stems of rushes and reed. In the field, these plant parts were still recognisable.

C mineralisation capacity is a measure of total microbial activity, while CH_4 production capacity is an estimate of the activity of methanogens. We hypothesised that CH_4 production capacity is linearly related to C mineralisation capacity. We found, however, that differences between CH_4 production capacities of individual fractions were much larger than differences between C mineralisation capacity of individual fractions (Fig. 4.3 and 4.4). Furthermore, the decrease with depth was much sharper for CH_4 production capacity than for C mineralisation capacity. This suggests that methanogens make higher demands on quality of their substrates than the combined total microbial biomass. The relatively small differences in C mineralisation capacities of individual fractions suggest that the fractionation method of Meijboom et al. (1995) has little functional value for C mineralisation of peat soils. However, the differences in CH_4 production capacity of individual fractions suggest that the fractionation method may have functional value for CH_4 production of peat soils.

Acknowledgements

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Chapter 5

Effects of grassland management on the emission of methane from intensively managed grasslands on peat soil

Van den Pol-van Dasselaar A, van Beusichem ML & Oenema O (1997)
Plant and Soil 189: 1-9

EFFECTS OF GRASSLAND MANAGEMENT ON THE EMISSION OF METHANE FROM INTENSIVELY MANAGED GRASSLANDS ON PEAT SOIL

Abstract

Methane (CH_4) is the most important greenhouse gas next to CO_2 and as such it contributes to the enhanced greenhouse effect. Peat soils are often considered as sources of CH_4. Grasslands on the other hand are generally considered to be a net sink for atmospheric CH_4. The aim of this study was twofold: (i) to quantify the net CH_4 emission of intensively managed grasslands on peat soil in the Netherlands; and (ii) to assess the effects of grassland management, i.e. drainage, nitrogen (N) fertilisation, and grazing versus mowing, on CH_4 emission rates. Net CH_4 emissions were measured weekly or biweekly for one year with vented closed flux chambers at two sites, one with a mean ground water level of 22 cm below surface and one with a mean ground water level of 42 cm. On each site there were three treatments: mowing without N application, mowing with N application, and grazing with N application. The dominating species was perennial ryegrass (*Lolium perenne* L.). Net CH_4 emissions were low, in general in the range of -0.2 to 0.2 mg CH_4 m^{-2} d^{-1}. In the relatively warm summer of 1994, consumption of atmospheric CH_4 peaked at 0.4 mg m^{-2} d^{-1}. On an annual basis, the sites were net consumers of atmospheric CH_4. However, the consumption was small: 0.31 to 0.08 kg CH_4 ha^{-1} yr^{-1}. Effect of mean ground water level was significant, but small. There were no significant effects of withholding N fertilisation for some years and grazing versus mowing on net CH_4 emissions. We conclude that grassland management of intensively managed grasslands on peat soil is not a suitable tool for reducing net CH_4 emissions.

Introduction

Methane (CH_4) is the most important greenhouse gas next to CO_2. It has been reported to contribute about 20% to the enhanced greenhouse effect (IPCC, 1995a). The concentration in the atmosphere is about 1.8 ppmv (parts per million by volume). It has been increasing at a rate of about 1% per year, but recently this increase has slowed down somewhat (Khalil & Rasmussen, 1993; Rudolph, 1994).

Methanogenesis occurs whenever organic matter is degraded under anaerobic conditions, in the absence of electron acceptors other than CO_2. Under aerobic conditions CH_4 can be oxidised by methanotrophs. Net CH_4 emissions, that are CH_4 exchanges between soil and atmosphere, are the result of CH_4 production, CH_4 consumption, and CH_4 transport. Transport can take place via diffusion, ebullition, and via the vascular system of plants. Wetlands are an important source of CH_4 and well-drained soils are an important sink of CH_4 (Van Amstel et al., 1993). The contribution of soils to the global CH_4 balance is significant (14-47% of the total source and 3-9% of the total sink) (IPCC, 1995a).

Grasslands are generally considered to be a net sink for atmospheric CH_4 (e.g. Hütsch et al., 1994; Minami et al., 1994; Mosier et al., 1991). It is suggested that the CH_4 consumption rate depends on environmental conditions and grassland management. Most important environmental factors are soil temperature and ground water level (Bartlett & Harriss, 1993). Soil temperature affects the kinetics of the microbial CH_4 production and consumption, while ground water level indicates the border between aerobic and anaerobic zones in the soil. Important management factors may be drainage, nitrogen (N) fertilisation, and grazing versus mowing. Drainage may decrease CH_4 emissions (e.g. Martikainen et al., 1992; Roulet et al., 1993). As the ground water table drops, the aerobic top layer of the soil becomes thicker, thereby possibly decreasing CH_4 production and increasing CH_4 consumption. Several authors have suggested that N fertilisation may decrease CH_4 consumption (Hütsch et al., 1994; Mosier et al., 1991; Steudler et al., 1989; Willison et al., 1995). This is either caused by an immediate inhibition of methanotrophs or by a change in the microbial community due to repeated fertiliser N application. Grazing versus mowing may affect CH_4 emissions through differences in annual supply of organic material and N to the soil (Van den Pol-van Dasselaar & Lantinga, 1995). So far, there are no experimental data on the effect of grazing versus mowing on CH_4 emissions.

Peat soils often show net CH_4 emission (Bartlett & Harriss, 1993), as they are anoxic at shallow depth and have a high organic matter content. In the Netherlands, about 30% of the grasslands are situated on peat soils. They are mainly agriculturally utilised and intensively managed. They are partially drained, so the oxic top layer is much thicker than in undrained peat soils. As a consequence, the sink-source balance of these soils for CH_4 may be much different than of undrained peat soils.

We hypothesise that intensively managed grasslands on peat soil in the Netherlands produce CH_4 at depth, but this CH_4 is in part consumed again in the upper layers. The overall-balance would show net CH_4 consumption in summer and net production in winter, as the result of seasonal changes in soil temperature, ground water level, and net precipitation excess. We also hypothesise that (i) drainage would decrease CH_4 production and increase CH_4 consumption; (ii) N fertilisation would decrease CH_4 consumption; and (iii) mowing would decrease CH_4 production compared to grazing as the annual supply of organic material to the soil is lower.

The aim of this study was twofold: (i) to quantify the net CH_4 emission of intensively managed grasslands on peat soil in the Netherlands; and (ii) to assess the effects of grassland management, i.e. drainage, N fertilisation, and grazing versus mowing, on CH_4 emission rates. For these purposes, a field study was conducted at the Regional Research Centre in Zegveld during 1994. Net CH_4 emissions and CH_4 concentrations in the soil profile were measured. CH_4 production and consumption rates in the soil profile were determined in incubation experiments in the laboratory. The grasslands in Zegveld are considered to be representative for intensively managed grasslands on peat soil in the Netherlands.

Materials and methods

Site description

Zegveld is located in the major peat area of the western part of the Netherlands (52°08'N, 4°48'E) (FAO classification: Terric Histosol). Monitoring was done on two sites, one with a relatively low mean ground water level (site Bos 6/low) and one with a relatively high mean ground water level (site 8B/high). Ground water levels vary greatly during the year. The top soil of both sites consists of clayey peat. The vegetation consists of only grasses. Perennial ryegrass (*Lolium perenne* L.) is the dominating species (25-55%). Soil characteristics of the sites are given in Table 5.1.

Table 5.1. Soil characteristics of the sites Bos 6/low and 8B/high (Van den Pol-van Dasselaar & Oenema, 1996; Velthof & Oenema, 1995).

	Bos 6/low	8B/high
Ground water level in 1994, cm, mean	42	22
range	7-73	0-61
Loss on ignition, %, 0-20 cm	45	38
20-40 cm	70	57
Clay, %, 0-20 cm	29	28
pH-KCl, 0-20 cm	4.7	5.0
Total S content, mg kg^{-1}, 0-20 cm	55	60
20-40 cm	150	90

Treatments

The sites at Zegveld were part of the SANS project: System for Adjusted N Supply (Vellinga et al., 1996). At both sites we used three different grassland management treatments:

M- : mowing, no N application;
M+ : mowing, N application;
G+ : grazing, N application.

Plot size was 2.8 x 15 m. The treatments were established in spring 1992. Before 1992, all plots were intensively managed and treated similarly. For both M+ and G+, fertiliser N was applied as calcium ammonium nitrate in six dressings, before each cut of M+. In 1994, cumulative application rates were 426 kg N ha^{-1} for site 8B/high and 378 kg N ha^{-1} for site Bos 6/low. The stocking density of the grazed plots was adjusted in such a way that the dairy cattle were able to graze the sward in about one day. Nitrate and ammonium concentrations in the layers 0-20 cm and 20-40 cm of the soil were measured by extraction with 0.01 M $CaCl_2$ on a weekly to three-weekly basis throughout 1994 for the treatments M- and M+ at both sites and G+ at the site 8B/high.

Monitoring net CH_4 emissions

Net CH_4 emissions were monitored weekly or biweekly from December 1993 until January 1995 with vented closed flux chambers (Hutchinson & Mosier, 1981; Mosier, 1989). Each combination of site and treatment was monitored with two to four flux chambers on one or two plots. Circular, stainless steel flux chambers (I.D. 20 cm, height 16 cm) were randomly placed on the plots and carefully inserted in the soil to a depth of 2 to 4 cm. They were air-tight closed by a stainless steel lid and covered with insulating sheets to prevent temperature changes within the chambers. At regular time intervals (15 to 30 minutes) four gas samples were taken from the headspace of the chambers with glass syringes.

Monitoring CH_4 concentrations in the soil profile

For the treatment M+ (mowing, N application) at site 8B/high, CH_4 concentrations in the soil profile were measured every month from March 1994 until December 1994. For that purpose, a perforated PVC pipe (I.D. 7 cm, height 50 cm) was installed in the soil more than a year before the start of the measurements. Possible effects of disturbances due to installation could be excluded at the time of measurements. The pipe was divided into five compartments of 10 cm each. The holes in the pipe were large enough to allow water and air from the surrounding soil to come into the compartments. The soil did not enter the compartments. Each compartment had two polythene tubes going from the compartment to the soil surface. CH_4 concentrations in the compartments were assumed to be in equilibrium with CH_4 concentrations in the surrounding soil. Gas samples were taken via the tubes with glass syringes. Whenever ground water level reached a certain compartment of the pipe, water samples were taken and injected into incubation bottles with ambient air. The bottles were shaken vigorously to degas the water. Subsamples from the headspace of the bottles were taken through rubber septa.

Incubation experiments

CH_4 production and consumption rates in the soil at various depths were studied in incubation experiments. For this purpose, 25 g homogenised field moist soil of the layers 0-5, 5-10, 10-20, 20-30, 30-40, and 40-50 cm of treatment M+ of both the sites 8B/high and Bos 6/low were placed in incubation bottles of 580 ml. They were incubated at 20 °C, either anaerobically (N_2-atmosphere), or aerobically (air-atmosphere) with concentrations of both 2 and 100 $\mu L\ L^{-1}$ CH_4 in the headspace. There were three replicates for each combination of soil layer and treatment. Control bottles containing no soil were included. The anaerobic incubation experiment lasted 10 weeks. Gas samples from the headspace of the bottles were taken on average twice a week through rubber septa. The aerobic incubation experiment lasted six days with a time interval of two days between sampling.

Soil data collection

Soil temperatures were recorded at 0, 2, 5, 10, 20, 30, 40, and 50 cm depth at each measurement. Moisture content of the upper 10 cm soil at each site was determined gravimetrically after drying at 105 °C for 24 h. Ground water levels were recorded from water level readings in 12 perforated pipes (I.D. 4 cm) per site, monthly in winter and on a weekly basis during the growing season.

Analytical procedures

Gas samples from the field experiments were analysed for CH_4 within 24 h by gas chromatography using a flame ionisation detector (coefficient of variation: 0.08%). Gas samples of the incubation experiments were analysed immediately after sampling. A standard concentration of 2.01 $\mu L\ L^{-1}$ CH_4 (± 5%) was used for calibration.

Data acquisition

CH_4 concentrations in flux chambers are expected to follow initially a linear increase or decrease and finally level off (Mosier, 1989). We aimed to measure in the linear phase. CH_4 concentrations in the flux chambers never exceeded 3.0 $\mu L\ L^{-1}$. Net CH_4 emissions were calculated by linear regression on the time course of CH_4 concentration in the headspace of the chambers. The coefficient of determination (r^2) was related to the amount of net CH_4 emission, which means that the precision of the calculated flux increased with increasing flux. The frequency distribution of all the individual data of net CH_4 emissions could best be described by a normal distribution, so statistical differences could be tested with an analysis of variance procedure. The ANOVA procedure of Statgraphics Plus 7.1 was used with factors of date of measurement, site, and treatment (P=0.05). Annual CH_4 emissions were estimated by trapezoidal integration of the mean CH_4 emissions over time.

In the aerobic incubation experiments, the decrease in CH_4 concentration in the headspace of the incubation bottles showed typical first-order-kinetics:

$$\ln Y_t = \ln Y_0 - kt \tag{5.1}$$

Y = CH_4 concentration in headspace, $\mu L\ L^{-1}$;
t = time, d;
k = rate constant, d^{-1}.
The rate constant k was used as a characteristic value for CH_4 consumption.

Results

Weather

The spring of 1994 was relatively cold and wet. The summer was warm and dry with air temperatures above 20 °C. The autumn of 1994 was relatively wet. Total annual rainfall in Zegveld (1000 mm) was 200 mm more than average. The time course of ground water levels is shown in Fig. 5.1.

Nitrate and ammonium concentrations

Nitrate concentrations in the treatments M+ (mowing, N application) and G+ (grazing, N application) increased during the growing season. Nitrate concentrations in M- (mowing, no N application) and ammonium concentrations in all treatments showed no clear pattern throughout the year. Average nitrate and ammonium concentrations are given in Table 5.2. Note that the N concentrations in M- are still quite high, especially for ammonium, although no N had been applied since 1992.

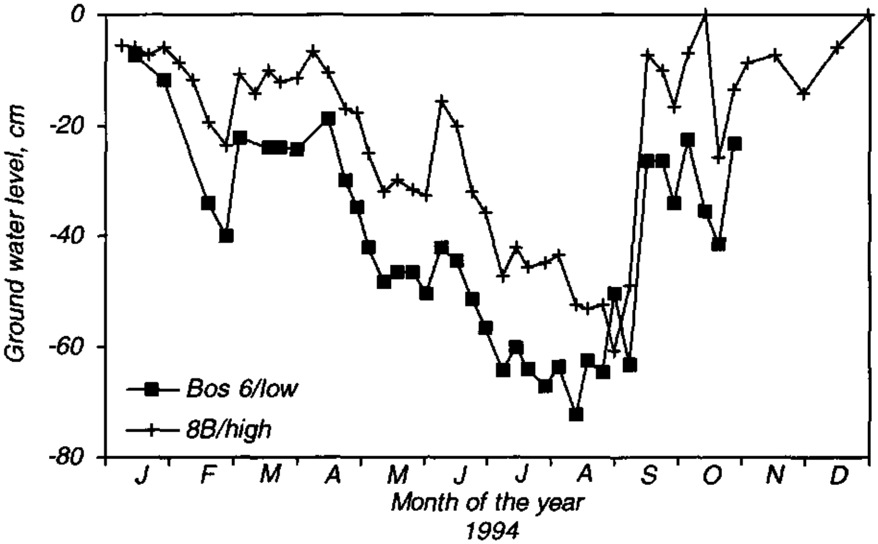

Fig. 5.1. Time course of ground water level (cm below the surface) at sites Bos 6/low and 8B/high.

Table 5.2. Average nitrate and ammonium concentrations (mg N kg^{-1} dry soil) in the layers 0-20 and 20-40 cm of the sites Bos 6/low and 8B/high throughout 1994 (n=28, ranges are given in brackets) (M- : mowing, no N application; M+ : mowing, N application; G+ : grazing, N application).

Site	Treatment	Nitrate		Ammonium	
		0-20 cm	20-40 cm	0-20 cm	20-40 cm
Bos 6/low	M-	2.9 (0-12)	1.1 (1-3)	14.9 (2-25)	14.6 (7-28)
	M+	13.9 (1-45)	4.4 (1-35)	16.8 (8-24)	15.0 (5-28)
8B/high	M-	2.3 (0-6)	1.2 (1-4)	15.1 (11-22)	13.1 (7-24)
	M+	16.4 (1-50)	2.9 (1-19)	19.4 (8-27)	11.5 (7-22)
	G+	20.8 (2-47)	6.2 (1-33)	22.7 (14-34)	14.4 (10-21)

Net CH_4 emissions

Throughout 1994, net CH_4 emissions at both sites and all treatments were generally in the range of -0.2 to 0.2 mg CH_4 m^{-2} d^{-1} (negative numbers indicate consumption of atmospheric CH_4, positive numbers indicate CH_4 emission from the soil to the atmosphere). In the relatively warm summer of 1994, consumption of atmospheric CH_4 peaked at about 0.4 mg CH_4 m^{-2} d^{-1}. This may be the result of a low ground water level (Fig. 5.1) in combination with high temperatures.

Fig. 5.2 shows the time course of mean net CH_4 emissions at all treatments and both sites. It has to be emphasised that CH_4 production by cattle is not included in these estimates. Standard deviations were generally in the range of 0 to 0.2 mg CH_4 m^{-2} d^{-1} for all treatments. Note that in winter 1994/1995 at site Bos 6/low consistent differences between the treatments occurred. Treatment M- (mowing, no N application) showed atmospheric CH_4 consumption, while the other treatments showed CH_4 emission from the soil to the atmosphere. Throughout the whole year and for both sites, net CH_4 emission was on average -0.06 mg CH_4 m^{-2} d^{-1} for treatment M- (mowing, no N application), -0.07 for treatment M+ (mowing, N application), and -0.03 for treatment G+ (grazing, N application) (standard error: 0.014). There were no significant effects of N fertilisation and grazing versus mowing on net CH_4 emissions ($P<0.05$).

Fig. 5.3 shows the time course of mean net CH_4 emissions at the two sites. The differences between site Bos 6/low and site 8B/high were small, but statistically significant ($P<0.05$). Net CH_4 emission was on average -0.09 mg CH_4 m^{-2} d^{-1} for site Bos 6/low and -0.02 mg CH_4 m^{-2} d^{-1} for site 8B/high (standard error: 0.012).

Annual net CH_4 emissions

In Table 5.3 annual net CH_4 emissions are presented. The two sites were net consumers of atmospheric CH_4, but the consumption was small. CH_4 consumption was significantly higher at a relatively low mean ground water level than at a relatively high mean ground water level. CH_4 consumption was also higher for mowing than for grazing, but this difference was not significant.

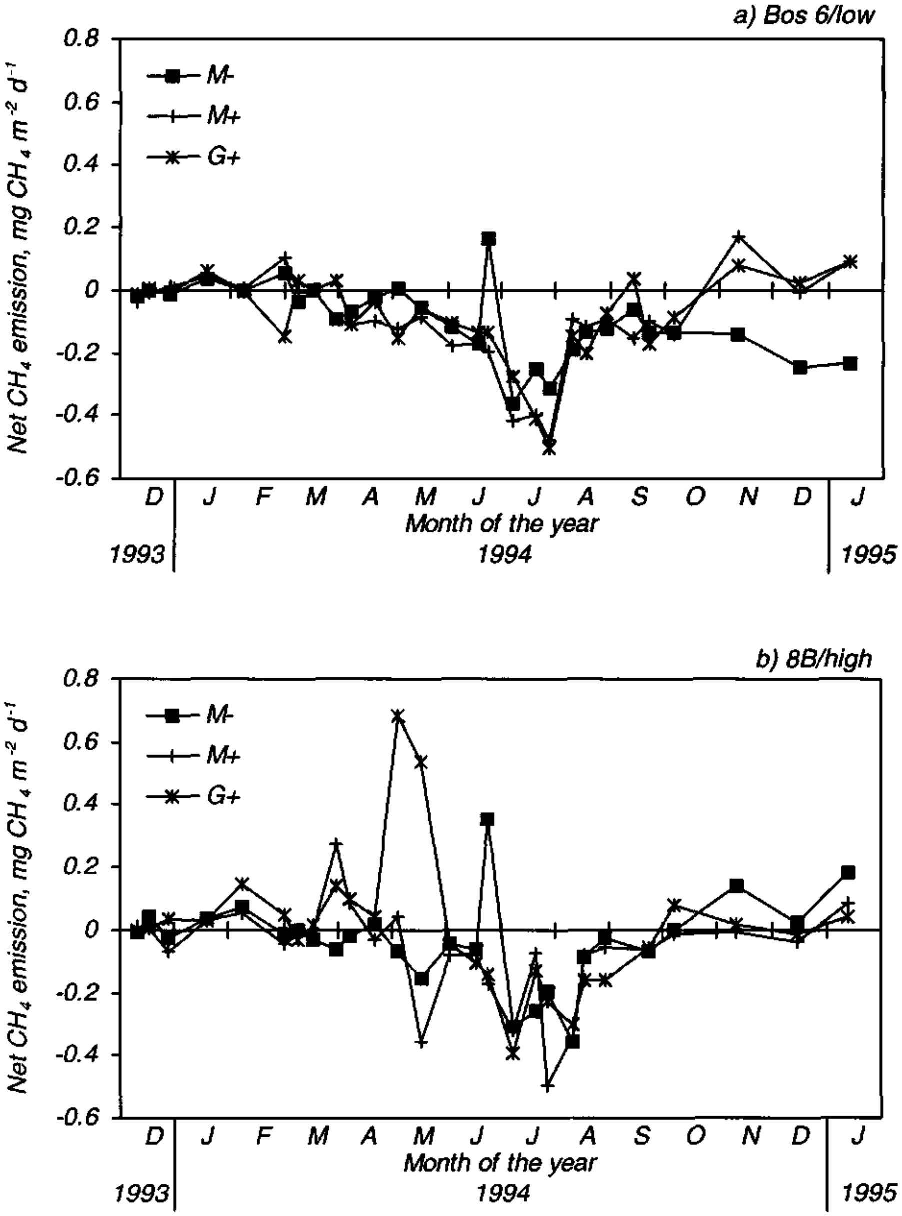

Fig. 5.2. Time course of mean net CH_4 emissions (mg CH_4 m^{-2} d^{-1}) at three different treatments (M- : mowing, no N application; M+ : mowing, N application; G+ : grazing, N application) and two sites: **a**) Bos 6/low and **b**) 8B/high. Each data point is the average of two to four measurements.

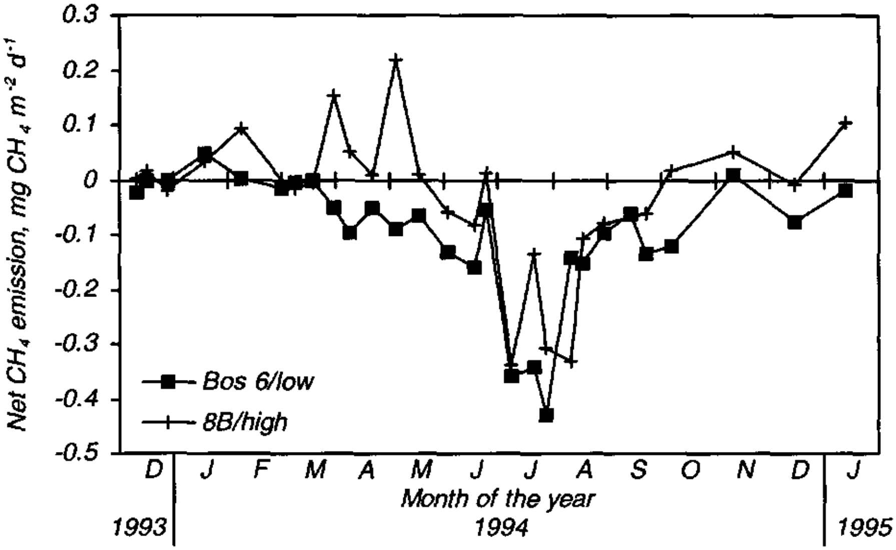

Fig. 5.3. Time course of mean net CH_4 emissions (mg CH_4 m^{-2} d^{-1}) at two sites: Bos 6/low with a mean ground water level (GWL) of 42 cm, and 8B/high with a mean GWL of 22 cm. Each data point is the average of 6 to 12 measurements.

Table 5.3. Annual net CH_4 emissions (kg CH_4 ha^{-1} yr^{-1}) at the sites Bos 6/low and 8B/high for three different treatments (M- : mowing, no N application; M+ : mowing, N application; G+ : grazing, N application) in 1994.

Treatment	Bos 6/low	8B/high
M-	-0.38	-0.08
M+	-0.29	-0.22
G+	-0.27	0.07
Average	-0.31	-0.08

CH_4 emissions in relation to environmental factors

CH_4 emissions are often related to soil temperature, soil moisture content, or ground water level (Bartlett & Harriss, 1993; Svensson & Sundh, 1992). In Zegveld, the highest correlation was found between ground water level and net CH_4 emission (r^2 = 0.52) (Fig. 5.4). Soil temperatures were negatively correlated with net CH_4 emission (r^2 = 0.44 to 0.49). Soil moisture content was positively correlated with net CH_4 emission (r^2 = 0.42). Multiple linear regression analyses with a combination of two or more factors did not improve r^2. Ground water level, soil moisture content, and soil temperatures are interrelated. The separate effect of each of these environmental factors can not be unravelled.

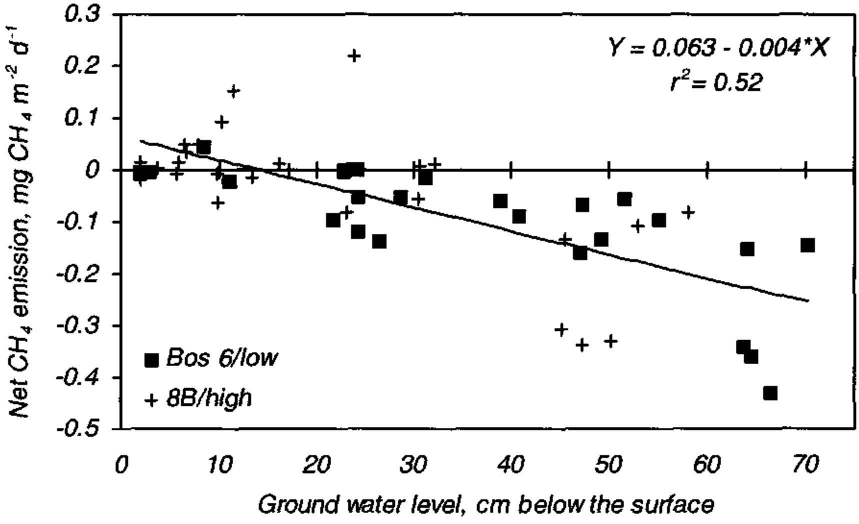

Fig. 5.4. Relationship between ground water level (cm below the surface) and mean net CH_4 emission (mg CH_4 m^{-2} d^{-1}) at Bos 6/low and 8B/high. Each data point is the average of 6 to 12 measurements.

CH_4 concentrations in the soil

Fig. 5.5 presents the time course of CH_4 concentrations in the soil of M+ (mowing, N application) at site 8B/high. From April until July, CH_4 concentrations decreased from ambient at the surface to about 1 µL L^{-1} in the layer 10-30 cm, indicating consumption of atmospheric CH_4. In the deeper soil layers CH_4 concentrations increased. During the rest of the year, CH_4 concentrations increased with depth, but remained rather low in all layers. Concentrations above ambient in deep soil layers indicate the presence of a CH_4 source.

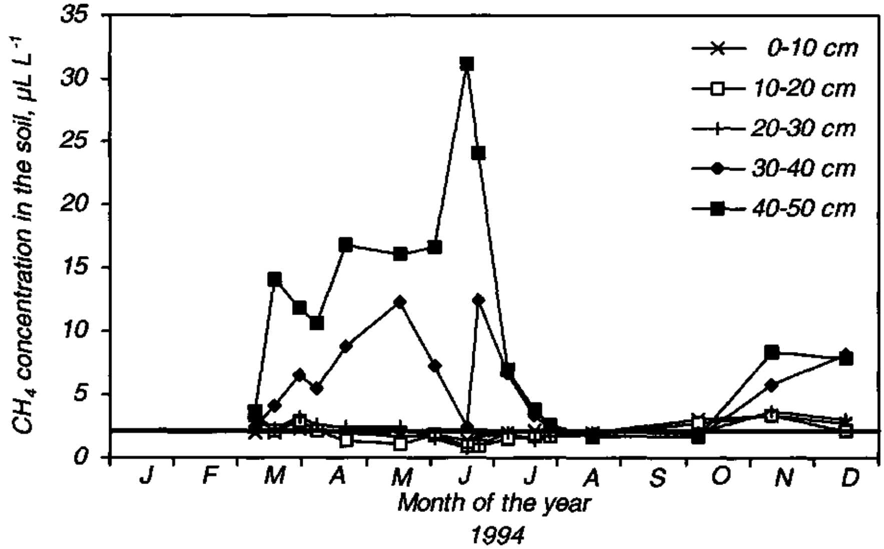

Fig. 5.5. Time course of CH_4 concentration (µL L^{-1}) in different soil layers of treatment M+ (mowing, N application) at site 8B/high. The horizontal line indicates ambient CH_4 concentration.

Incubation experiments

During anaerobic incubation, significant CH_4 production was observed in the upper 10 cm of site 8B/high, with the upper 5 cm being most active, and in the upper 5 cm of site Bos 6/low (Fig. 5.6). Production of CH_4 started after an initial lag phase of 20 to 40 days. Soil from depths of more than 10 cm did not yield increases in CH_4

concentration in the headspace during an incubation time of 70 days.

During aerobic incubation, decrease in CH_4 concentration appeared to obey first-order kinetics. The rate constant *k* of Equation 5.1 can be used as a characteristic value for CH_4 consumption. Table 5.4 shows these rate constants normalised to dry weight of incubated soil for aerobic incubation with 100 µL L^{-1} CH_4. Results of incubation with 2 µL L^{-1} (not shown) followed a similar pattern. CH_4 consumption was observed in all layers of the soil, with the highest activity in the layers 5-20 cm for Bos 6/low and 10-30 cm for 8B/high.

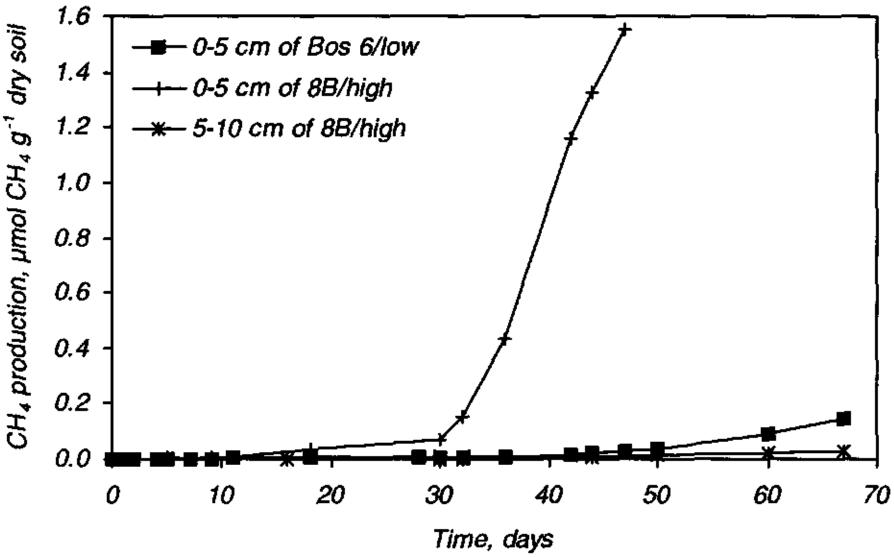

Fig. 5.6. CH_4 production (µmol CH_4 g^{-1} dry soil) in time for the layers 0-5 cm of treatment M+ (mowing, N application) of sites Bos 6/low and 8B/high, and 5-10 cm of M+ of site 8B/high during anaerobic incubation at 20 °C.

Discussion

So far, it was unknown whether intensively managed grasslands on peat soil in the Netherlands emitted CH_4 to the atmosphere or consumed atmospheric CH_4. Our first aim was therefore to quantify net CH_4 emissions from these peat soils. Table 5.3

Table 5.4. CH_4 consumption (d^{-1} g^{-1} dry soil) for different soil layers (cm below the surface) of the treatment M+ (mowing, N application) at sites Bos 6/low and 8B/high during aerobic incubation at 20 °C.

Layer	CH_4 consumption	
	Bos 6/low	8B/high
0-5	0.0084	0.0016
5-10	0.0121	0.0029
10-20	0.0094	0.0075
20-30	0.0007	0.0080
30-40	0.0009	0.0056
40-50	n.d.[1)]	0.0018

[1)] n.d. = not determined

shows that the sites in Zegveld were net consumers of atmospheric CH_4. However, the consumption was small: 0.31 to 0.08 kg CH_4 ha^{-1} yr^{-1}. Table 5.5 provides an overview of literature data on CH_4 emissions from drained soils. Our results fall within the range of these CH_4 emissions.

The anaerobic incubation experiments showed that only the top layer of the soil was capable of producing significant amounts of CH_4 after prolonged periods of anaerobiosis (Fig. 5.6). In comparable incubation experiments, Kengen & Stams (1995) studied CH_4 production at Bos 6/low to a depth of 120 cm and at 8B/high to a depth of 70 cm throughout 1994. They also found no CH_4 formation in the deeper layers, even after an incubation period of 120 days. They concluded that the methanogenic potential of drained grasslands on peat soil is low. The deeper layers are anoxic, but probably restricted in their supply of methanogenic substrates, whereas the top soil has a potential methanogenic capacity, but requires a prolonged wet period to generate methanogenic conditions. In the field, complete anaerobiosis of the top soil for several weeks may occur only in winter and high CH_4 production is thus not to be expected. Although CH_4 production in the deeper layers is negligibly small, Fig. 5.5 shows increasing CH_4 concentrations at depth. We believe that this CH_4 originates from an 'old source', from which it is slowly diffusing to the surface. The upward diffusing CH_4 is nearly completely oxidised in the top soil before it can reach the surface. We speculate that this 'old source' might be organic layers which are still capable of producing small quantities of CH_4, or CH_4 dissolved in the pore waters, produced by the peat a long time ago.

The aerobic incubation experiments showed a subsurface maximum in CH_4 consumption at a depth of 5-20 cm for Bos 6/low and 10-30 cm for 8B/high. This phenomenon has been found earlier (e.g. Dunfield et al., 1995; Koschorreck & Conrad, 1993; Roulet et al., 1993). It was also found in comparable aerobic incubation experiments of Heipieper & de Bont (1995) with soil from Zegveld taken throughout 1994. Fig. 5.5 shows that from April until July CH_4 concentrations in the soil decreased with increasing depth up to 30 cm, indicating atmospheric CH_4 consumption. A subsurface minimum of CH_4 concentration is consistent with a subsurface maximum of CH_4 consumption. Apparently, the oxidisers switch from

Table 5.5. Net CH_4 emissions (mg CH_4 m^{-2} d^{-1}) from drained soils and soils with a natural low ground water level (negative numbers indicate consumption of atmospheric CH_4, positive numbers indicate CH_4 emission from the soil to the atmosphere).

Site description	Net CH_4 emissions	Reference
Drained peat soil, grass, Canada	-0.45	Glenn et al., 1993
Loam grassland soil in winter, UK, poorly drained	-0.1	
well drained	-0.7	Jarvis et al., 1994
Drained peat soils, Finland	-0.7 to 30	Martikainen et al., 1995; Martikainen et al., 1992
Temperate grassland, Japan	-1.0 to 0	Minami et al.,1994
Shortgrass prairie soils, USA	-0.8 to -0.1	Mosier et al., 1991
Drained peatlands, Canada	-0.4 to 0.1	Roulet et al., 1993
Intensively managed grasslands on peat soil, the Netherlands	-0.4 to 0.2	This paper, Fig. 5.3

atmospheric CH_4 to CH_4 diffusing from the deeper soil layers at some depth in the soil. Careful evaluation of our dataset suggested this to occur between 20 and 30 cm depth. For drained peat soils in Canada, this depth varied between 25 and 35 cm (Roulet et al., 1993).

The second aim of this study was to assess the effects of grassland management on CH_4 emission rates. We studied the effects of drainage, N fertilisation, and grazing versus mowing. Drainage generally decreases CH_4 emissions from peat soils. As the ground water table drops, the aerobic top layer of the soil becomes thicker. Conditions become less favourable for CH_4 production and more favourable for CH_4 consumption. Indeed, temporal variation in net CH_4 emissions in Zegveld could best be explained by ground water level (Fig. 5.4) rather than temperature or soil moisture content. Differences between drained and undrained sites are most distinct if the undrained sites have a ground water level close to the surface (e.g. Martikainen et al., 1992; Roulet et al., 1993). Site 8B/high with a mean ground water level of 22 cm showed a higher consumption of atmospheric CH_4 than site Bos 6/low with a mean ground water level of 42 cm. Part of the differences between the two sites was related to differences in CH_4 production and consumption rates as observed in the laboratory. The incubation experiments showed that on treatment M+ of 8B/high CH_4 production was higher and consumption lower compared to M+ of Bos 6/low, especially in the top layer (Fig. 5.6 and Table 5.4). Although the differences in CH_4 emissions between Bos 6/low and 8B/high were significant, they were small (Table 5.3). Therefore, we conclude that on intensively managed grasslands on peat soil the effect of mean ground water level in the range of 20 to 40 cm below surface on CH_4 emissions is small.

Fertiliser N application, especially ammonium, may decrease CH_4 consumption (Dunfield & Knowles, 1995; Hütsch et al., 1994; King & Schnell, 1994; Mosier et al., 1991; Steudler et al., 1989; Willison et al., 1995). This is either caused by an immediate inhibition of methanotrophs, e.g. by ammonium or nitrite, or by a change in the microbial community due to repeated fertiliser N application. In our study, there were no significant differences in net CH_4 emission between the fertilised and unfertilised treatments at the two sites. This could be due to the relatively short period without N fertilisation, since the treatments started only in 1992. Ammonium concentrations in the treatments without N fertilisation were still high compared to the fertilised treatments (Table 5.2). Interesting to notice is that in winter 1994/1995, treatment M- (mowing, no N application) at site Bos 6/low showed consumption of atmospheric CH_4, while the other treatments showed CH_4 emission from the soil to the atmosphere (Fig. 5.2). Possibly the methanotrophs slowly started to recover. We conclude that on intensively managed grasslands on peat soil the effect of withholding N fertilisation for some years on CH_4 emissions is negligible.

We hypothesised that mowing would decrease CH_4 production compared to grazing as the annual supply of organic material to the soil is lower on mown grassland than on grazed grassland (Van den Pol-van Dasselaar & Lantinga, 1995). Table 5.3 shows that on the mown treatments net CH_4 emission was slightly lower than on the grazed treatments. However, there were no significant differences between the mown and grazed treatments at the two sites. We conclude that on intensively managed grasslands on peat soil the effect of grazing versus mowing on CH_4 emissions is negligibly small. It has to be emphasised that CH_4 production by cattle is not included in these estimates.

Our results indicate that the effect of variations in management of intensively managed grasslands on peat soil on CH_4 emissions is negligible or small. Moreover, net CH_4 emissions from these grasslands are low (Table 5.3). Therefore, we conclude that grassland management of intensively managed grasslands on peat soil is not a suitable tool for reducing net CH_4 emissions.

Acknowledgements

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Chapter 6

Spatial variability of methane, nitrous oxide, and carbon dioxide emissions from drained grasslands

Van den Pol-van Dasselaar A, Corré WJ, Priemé A, Klemedtsson ÅK, Weslien P,
Stein A, Klemedtsson L & Oenema O (1998)
Soil Science Society of America Journal, in press

SPATIAL VARIABILITY OF METHANE, NITROUS OXIDE, AND CARBON DIOXIDE EMISSIONS FROM DRAINED GRASSLANDS

Abstract

Emissions of CH_4, N_2O, and CO_2 from soils are the result of a number of biological and physical processes, each influenced by several environmental and management factors exhibiting spatial variability. This study aimed to assess the spatial variability and spatial dependence of CH_4, N_2O, and CO_2 emissions and their underlying soil processes and properties from grasslands on drained peat soil. Emissions and possible controlling factors were measured at a field location in Sweden. Measurements were done on two adjacent sites on peat soil on two successive days for each site. Spatial variability was analysed with trend analysis and variograms. Both sites consumed small amounts of atmospheric CH_4, i.e. 0.03 and 0.05 mg CH_4 m^{-2} d^{-1}, and emitted N_2O and CO_2, i.e. 5 to 19 mg N_2O m^{-2} d^{-1} and 4 to 6 g CO_2 m^{-2} d^{-1}. Spatial variability of emissions was high with coefficients of variation of 50 to 1400%. Emissions either showed a spatial trend or were spatially dependent with ranges of spatial dependence of 50 to >200 m. However, spatial dependence of emissions showed differences between sites and short-term temporal variability. Variograms of emissions and soil processes, which are partly biological in nature and have a high degree of inherent variability, should be interpreted with care.

Introduction

Emissions of CH_4, N_2O, and CO_2 from peat soils result from various biological and physical processes, which in turn are influenced by several environmental and management factors. For example, CH_4 emissions result from CH_4 production, CH_4 oxidation and CH_4 transport, which are influenced by factors like ground water level, soil temperature, diffusion rates of CH_4 and O_2 in the soil, drainage, and fertilisation (Bartlett & Harriss, 1993; King, 1992; Willison et al., 1995). Each of these processes and factors has its own temporal and spatial variability. Some are distributed uniformly, while others are not. Some will interact in a multiplicative manner to create skewed frequency distributions. Spatial variability of soil variables will directly influence CH_4, N_2O, and CO_2 emissions from soils, and occurs at scales of centimetres to kilometres. Description of spatial variability of soil variables should improve the description of the spatial distribution of emissions.

The most important factor regulating CH_4 emissions from peat soil is the distribution of O_2 in the top layer of the soil (King, 1992). Aerobic soil conditions suppress methanogenic processes in favour of methanotrophic processes. Differences in water level may therefore determine to a large extent spatial variability of CH_4 emissions. The most important factors regulating N_2O emissions include the distributions of O_2, N and organic matter in the soil (Martikainen et al., 1993). The most important factors regulating CO_2 emissions include the distributions of O_2 and organic matter (Bouwman, 1990). As the distribution of O_2 in the top layer of the soil is one of the main regulating factors for CH_4, N_2O, and CO_2 emissions, and drained peat soils have a non-uniform distribution of O_2 due to the presence of drainage pipes, spatial variability of emissions should be high on drained peat soils.

Spatial variability contains systematic and random components. In standard statistics, the variability around the mean is assumed to be random and spatially independent. However, variability of soil variables often contains a spatially dependent component, i.e. the variability can be described as a function of spatial separation. We studied spatial variability using both a statistical and a geostatistical approach. First, trend surfaces were fitted through the data to check for any non-stationarity. Next, for data without a trend, geostatistics was applied. Geostatistics, i.e. the theory of regionalised variables, models the spatial dependence of spatially distributed variables (e.g. Trangmar et al., 1985). Geostatistical techniques are currently widely applied by soil scientists, e.g. in analysing spatial variability of nutrient contents (Borges & Mallarino, 1997; Boyer et al., 1996) and hydraulic properties (Mallants et al., 1996). Priemé et al. (1996) used geostatistics in studying spatial variability of CH_4 emissions and Velthof et al. (1996) in studying spatial variability of N_2O emissions. So far, analysis of spatial variability of CH_4, N_2O, and CO_2 emissions has not been combined with analysis of spatial variability of controlling factors in one study. As such it is unclear whether these controlling factors display spatial variability at scales similar to emissions.

Our study aimed to assess the spatial variability and spatial dependence of CH_4, N_2O, and CO_2 emissions and their underlying soil processes and properties from grasslands on drained peat soil. Emissions and possible controlling factors were measured at a field location in Sweden. Possible controlling factors included several soil processes (CH_4 production, CH_4 oxidation, N_2O production, denitrification, and CO_2 production) and soil properties (soil moisture content, loss-on-ignition, NO_3 and NH_4 content, and ground water level).

We hypothesised that both the magnitude of spatial variability of emissions and the range of spatial dependence of emissions are related to the magnitude of spatial variability and the range of spatial dependence of governing soil processes and soil properties. Measurements were done on two adjacent sites, one fertilised with 90 kg N ha^{-1} yr^{-1} and one with 115 kg N ha^{-1} yr^{-1}, and on two successive days for each site.

Materials and methods

Site description

The field location was on permanent grassland on peat soil (FAO classification: Terric Histosol) near Fallköping, Sweden (58°20'N, 13°30'E). We studied two sites: Field 1 (45 m wide and 265 m long) and Field 2 (110 m wide and 110 m long). For the last 7 yr, Field 1 has received annually 115 kg N, 17.5 kg P and 62.5 kg K ha^{-1}, and Field 2 has received 90 kg N, 17.5 kg P and 62.5 kg K ha^{-1}. Both fields have been grazed by cattle every summer. Cattle manure collected in the winter period has been spread on the fields during the growing season. Soil pH varied between 7.0 and 7.5. In Field 1, two drainage pipes in length were present. In Field 2, two drainage pipes in length and three drainage pipes in width were present. Average drainage depth was 1 m, average drainage distance was 26 and 43 m for drainage pipes in length and width respectively. Our field measurements were conducted after a very cold winter and started just after the soil had thawed. Field 2 had not been fertilised yet and Field 1 had received 30 kg N ha^{-1}.

Experimental set-up

Field 1 was studied on 13 and 14 May, 1996. It was divided into 102 (6 × 17) grids of 7.5 by 15 m. Field 2 was studied on 15 and 16 May. This field was divided into 100 (10 × 10) grids of 10 by 10 m. Each second measurement day, all grids were moved: 7.5 m in *y* direction on Field 1 and 5 m in both *x* and *y* direction on Field 2. All measurements were done in the centre of the grids on an area of about 0.03 m^2, between 8.00 and 16.00 h.

CH_4, N_2O, and CO_2 emissions

Net CH_4, N_2O, and CO_2 emissions were measured with vented closed flux chambers (I.D. 20 cm, height 16 cm) (Mosier, 1989). For each grid a separate flux chamber was used. Flux chambers were constructed either of stainless steel or polyvinyl chloride, implying that for CO_2 we measured emission from the soil and dark respiration of the grass. Flux chambers were closed for approximately 1 h. Previous experiments at the site showed that closure of flux chambers for this period would ensure detectable fluxes with minimal numbers of gas samples taken. Two gas samples per flux chamber were taken, at t_0 and t_{end}, in vials and analysed for CH_4, N_2O, and CO_2 by gas chromatography (Varian 3400, Varian Associates, Sunnyvale, CA) using an electron capture detector and a flame ionisation detector. By use of a Ni catalyst, CO_2 was converted to CH_4 before detection. By taking only two gas samples per flux chamber, a maximum number of flux chambers per day could be measured.

Soil processes

After flux measurements two soil cores (I.D. 4.7 cm, depth 20 cm) were taken in the centre of 102, 100 and 40 flux chambers on 13, 15, and 16 May, respectively, and incubated in the field in 1-L glass jars, closed with a polyacetale lid with rubber septa. One soil core was incubated under air for measurements of N_2O and CO_2 production. The other soil core was incubated under air with 5% acetylene for denitrification measurements (Ryden et al., 1987). N_2O and CO_2 accumulation were measured after about 24 h with a photoacoustic infrared absorption gas analyser (Brüel & Kjær 1300).

The soil cores from the incubation under air were thoroughly mixed in the field immediately after incubation and subsampled for soil analyses and determination of CH_4 oxidation and CH_4 production. After transport, soil samples were stored at 4 °C. Aerobic and anaerobic incubation experiments started respectively 1 day and 1 week after soil sampling. CH_4 oxidation was measured under ambient CH_4 concentrations in aerobic incubation experiments. CH_4 production was measured in anaerobic incubation experiments (N_2-atmosphere). Gas samples were analysed for CH_4 by gas chromatography (Varian 3400 and PU 4400, Unicam Ltd., Cambridge, UK) using a flame ionisation detector.

Soil properties

The ground water level of Field 2 was recorded in the flux chambers of 15 May. Soil was analysed for soil moisture content (percentage of field-moist soil, gravimetrically after drying at 105 °C for 24 h), loss-on-ignition (at 550 °C), NO_3 and NH_4 contents (by extraction with 0.01 M $CaCl_2$).

Data acquisition, statistics and geostatistics

Net CH_4, N_2O, and CO_2 emissions were calculated using the concentration in the headspace of the chambers at t_0 and t_{end}. CH_4 oxidation was calculated assuming first-order kinetics. N_2O production, denitrification and CO_2 production rates were calculated using the measured concentrations and the incubated area. Prior to statistical analyses, variables that followed a skewed distribution were transformed to a near-normal distribution by lognormally transforming the data. Simple and multiple linear regression analyses with (ln-transformed) emissions as well as soil processes as dependent variables and ln-transformed soil processes and properties as independent variables were carried out to assess possible relations between emissions, soil processes and properties. Differences between the different sites and measurement dates were determined by an analysis of variance with factors of site and measurement date (P=0.05). Spatial variability of emissions, soil processes and properties was analysed with statistics and geostatistics (e.g. Trangmar et al., 1985). First, for CH_4, N_2O, and CO_2 emissions the relation with the x and y coordinates of the measurement locations was determined with the trend surface model:

$$Emission = \beta_0 + \beta_1 x + \beta_2 y + \beta_3 x^2 + \beta_4 xy + \beta_5 y^2 \qquad (6.1)$$

with regression coefficients β_0 to β_5. The presence of a trend is elucidated by significance of any of the parameters β_1 to β_5, whereas β_0 models the intercept. In the absence of a trend, the degree of spatial dependence was modelled with variograms. A variogram $\gamma(h)$ describes the variance of the squared difference of a spatial variable between pairs of points at distance h. Variograms were constructed, assuming direction independent (isotropic) spatial variation, by:

$$\gamma(h) = \frac{1}{2N(h)} \sum_{i=1}^{N(h)} \{Z(x_i) - Z(x_i + h)\}^2 \tag{6.2}$$

in which for each distance approximately equal to *h*, *N(h)* is the number of pairs of observations at this distance and $Z(x_i)$ and $Z(x_i+h)$ represent the values of the spatial variable *Z* at two positions separated by *h*. To the values thus obtained a variogram model was fit. We distinguish between transitive and intransitive models. A transitive variogram model increases with distance between sample points to a more or less constant value (the sill, which equals the sample variance as determined in standard statistics) at a separation distance, called the range of spatial dependence. An intransitive variogram model keeps increasing with the distance of sample separation. Many variograms show a nugget variance, i.e. a non-zero value as the distance tends to zero. It represents variability not accounted for, either because of microscale variability, which can not be detected at the scale of sampling, or because of measurement errors. In the absence of spatial dependence, i.e. in cases of large point-to-point variation at short distances, the variogram will show a pure nugget effect. Variograms were fit to a linear or a spherical model using weighted least squares approximation (McBratney & Webster, 1986).

Results

Weather conditions

Weather data were obtained from a weather station about 200 m from the sites. It was relatively warm on 13 and 14 May, while 15 and 16 May were relatively cold. It was mainly dry, except for some heavy rainfall during a thunderstorm on 14 May. Air temperature, photosynthetic active radiation and rainfall are shown in Fig. 6.1.

CH_4, N_2O, and CO_2 emissions

Both sites were small sinks of CH_4 and sources of N_2O and CO_2 (Table 6.1). Consumption of atmospheric CH_4 was larger at Field 2 than at Field 1. N_2O and CO_2 emissions were larger at Field 1. Differences between emissions from Fields 1 and 2 were significant ($P<0.05$). Emissions were not significantly affected by measurement time during the day, probably due to rather constant weather conditions on each day.

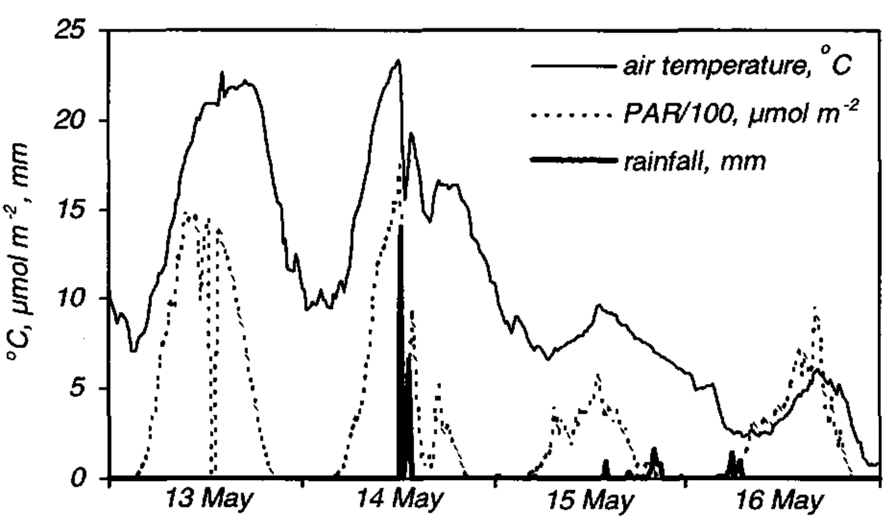

Fig. 6.1. Time course of air temperature (°C), photosynthetic active radiation (PAR) (µmol m^{-2}) and rainfall (mm).

This is very important, as it allowed us to combine all data from a particular day for statistics and geostatistics. The standard deviation of emissions was high compared with the mean, indicating a high spatial variability, especially for CH_4 and N_2O emissions (Table 6.1). Frequency distributions of CO_2 and CH_4 emissions could be described by a normal distribution, frequency distributions of N_2O emissions by a lognormal distribution.

Trend surface analysis revealed trends in CH_4 emissions at 13 and 15 May, in ln-transformed N_2O emissions at 13, 15, and 16 May and in CO_2 emissions at 14 and 15 May (Table 6.2). Trends could be totally different on succeeding days on the same site, such as those for ln-transformed N_2O emissions on 15 and 16 May. Also notice the low coefficients of determination, which ranged from 0.16 to 0.32. For emissions without a trend, geostatistics was applied to analyse spatially dependent components of spatial variability. CH_4 emissions were spatially dependent for distances smaller than about 50 m on 14 May. On 16 May, they were spatially dependent throughout the whole site (Fig. 6.2a and 6.2b). Lognormally transformed N_2O emissions were spatially dependent throughout the whole site on 14 May

(Fig. 6.2c). CO_2 emissions were spatially dependent for distances smaller than about 55 and 75 m on 13 and 16 May, respectively (Fig. 6.2d and 6.2e). All variograms showed a positive nugget effect. Two results were remarkable: the change from a stationary emission on one day to a non-stationary emission on the succeeding day on the same site, and secondly the difference in variograms between sites. Differences in both variance and shape of the variogram occurred.

Table 6.1. Summary statistics of CH_4, N_2O, and CO_2 emissions: average, median, standard deviation, minimum, maximum, and coefficient of variation.

	Avg.	Median	SD	Min.	Max.	CV, %
CH_4, mg CH_4 m^{-2} d^{-1}						
13 May	-0.072	-0.060	0.209	-0.814	0.830	-291
14 May	0.022	-0.024	0.316	-0.754	1.728	1424
15 May	-0.065	-0.048	0.134	-0.529	0.504	-206
16 May	-0.042	-0.052	0.113	-0.379	0.472	-267
Field 1	-0.025					
Field 2	-0.054					
N_2O, mg N_2O m^{-2} d^{-1}						
13 May	21.7	6.4	36.3	-2.5	181	167
14 May	16.1	5.2	34.2	-2.2	266	212
15 May	4.9	0.2	16.7	-0.2	148	341
16 May	5.3	0.1	26.0	-0.4	247	493
Field 1	18.9					
Field 2	5.1					
CO_2, g CO_2 m^{-2} d^{-1}						
13 May	5.2	5.0	2.6	-0.4	11.6	50
14 May	6.1	5.8	3.1	-1.5	16.8	51
15 May	3.8	3.8	2.2	-0.0	10.3	58
16 May	3.2	3.2	1.5	-0.0	8.6	48
Field 1	5.6					
Field 2	3.5					

Table 6.2. Significant regression coefficients of Equation (6.1), and coefficients of determination (r^2), for CH_4 emissions, ln-transformed N_2O emissions, and CO_2 emissions.

	β_0	β_1	β_2	β_3	β_4	β_5	r^2
CH_4 emissions, mg CH_4 m^{-2} d^{-1}							
13 May	1.5E-1		-3.2E-3**			9.0E-6*	0.21
15 May	1.4E-2	-5.7E-3**		6.0E-5**	3.2E-5*		0.24
Ln-transformed (N_2O emissions + 5), mg N_2O m^{-2} d^{-1}							
13 May	2.2			-1.7E-3**	2.3E-4*		0.32
15 May	1.4	2.7E-2**		-2.5E-4**	-2.2E-4**		0.24
16 May	1.8				-1.7E-4*		0.23
CO_2 emissions, g CO_2 m^{-2} d^{-1}							
14 May	8.1		-5.8E-2**			1.4E-4*	0.22
15 May	4.2				-5.8E-4*		0.16

*, ** Significant at the 0.05 and 0.01 levels of probability, respectively.

Soil processes and properties

Soil characteristics of both sites are given in Table 6.3. Mean CH_4 oxidation capacity was larger and mean CH_4 production capacity was smaller at Field 1 than at Field 2. Mean N_2O production, denitrification, and CO_2 production were higher at Field 1. Differences between soil processes and soil properties of Fields 1 and 2 were significant ($P<0.05$), except for NH_4 content. Simple and multiple linear regression analyses showed that CH_4, N_2O, and CO_2 emissions were essentially unrelated to soil processes determined in the various incubation experiments ($r^2 < 0.01$-0.05). Furthermore, emissions were very poorly related to soil properties ($r^2 = 0.03$-0.35). The best model with CH_4 emissions as dependent variable, for example, included

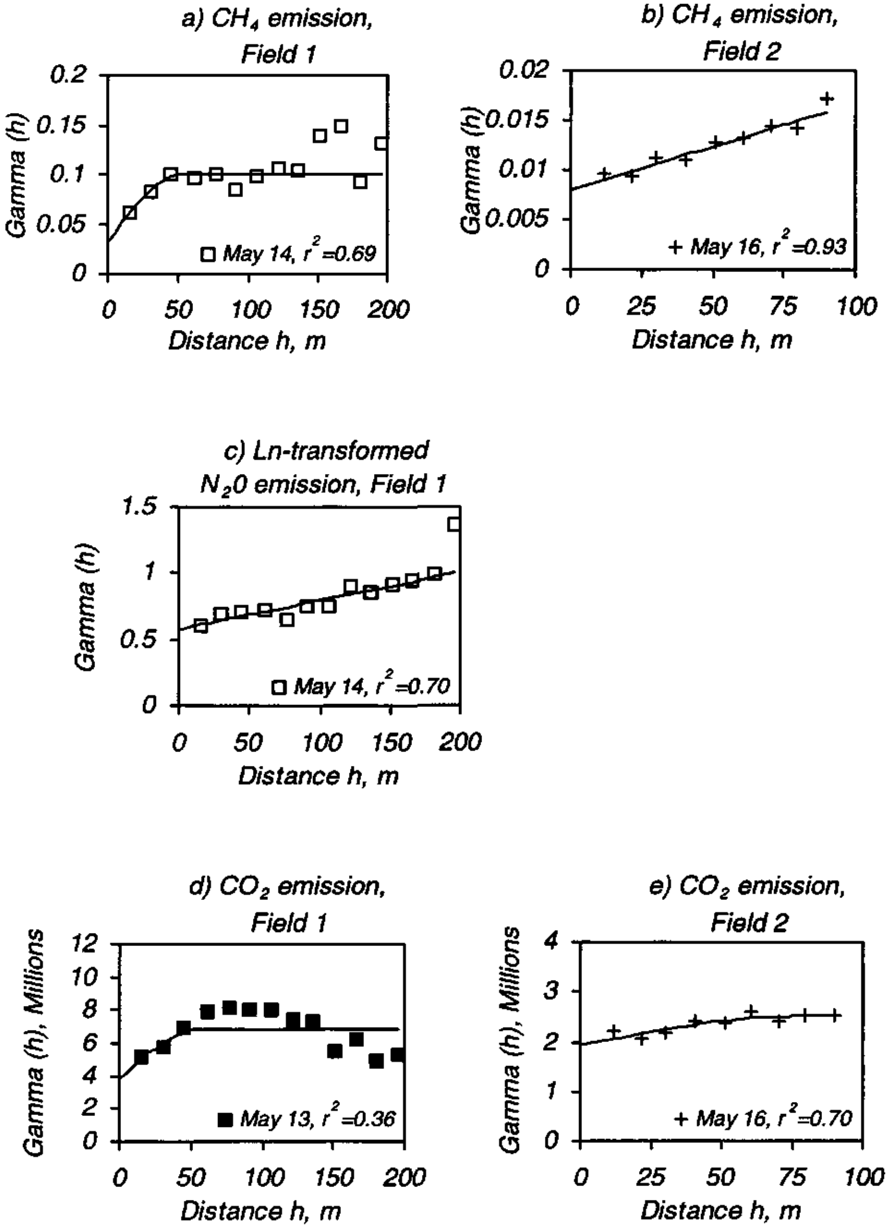

Fig. 6.2. Variogram of **a)** CH_4 emissions from Field 1, **b)** CH_4 emissions from Field 2, **c)** ln-transformed N_2O emissions from Field 1, **d)** CO_2 emissions from Field 1, and **e)** CO_2 emissions from Field 2: variance $\gamma(h)$ vs. distance h (m).

only ground water level ($r^2 = 0.19$). However, soil processes, as determined in the various incubation experiments, were better related to soil properties ($r^2 = 0.20$-0.61). The best model with CH_4 oxidation as a dependent variable included both loss-on-ignition and soil moisture content ($r^2 = 0.50$). The best model with ln-transformed CH_4 production as dependent variable included only loss-on-ignition ($r^2 = 0.20$). The best model with CO_2 production as dependent variable included both ground water level and soil moisture content ($r^2 = 0.61$). Soil processes and properties showed spatial dependence (Fig. 6.3 and 6.4).

Table 6.3. Averages and coefficients of variation of soil processes and properties as determined in soil cores of the 0 to 20-cm layer, and ground water level.

	Field 1		Field 2	
	Avg.	CV, %	Avg.	CV, %
Soil processes				
CH_4 oxidation, μg CH_4 kg^{-1} dry soil d^{-1}	15	42	8	68
CH_4 production, μg CH_4 kg^{-1} dry soil d^{-1}	10	388	74	160
N_2O production, mg N m^{-2} d^{-1}	28	146	8	58
Denitrification, mg N m^{-2} d^{-1}	45	113	9	58
CO_2 production, g CO_2 m^{-2} d^{-1}	23	35	18	35
Soil properties				
Soil moisture content, %	57	10	59	9
Loss-on-ignition, %	37	37	38	33
NO_3, mg kg^{-1} dry soil	75	52	44	34
NH_4, mg kg^{-1} dry soil	20	28	18	33
Ground water level, cm below surface	n.d.(1)	n.d.	28	48

(1)n.d. = not determined.

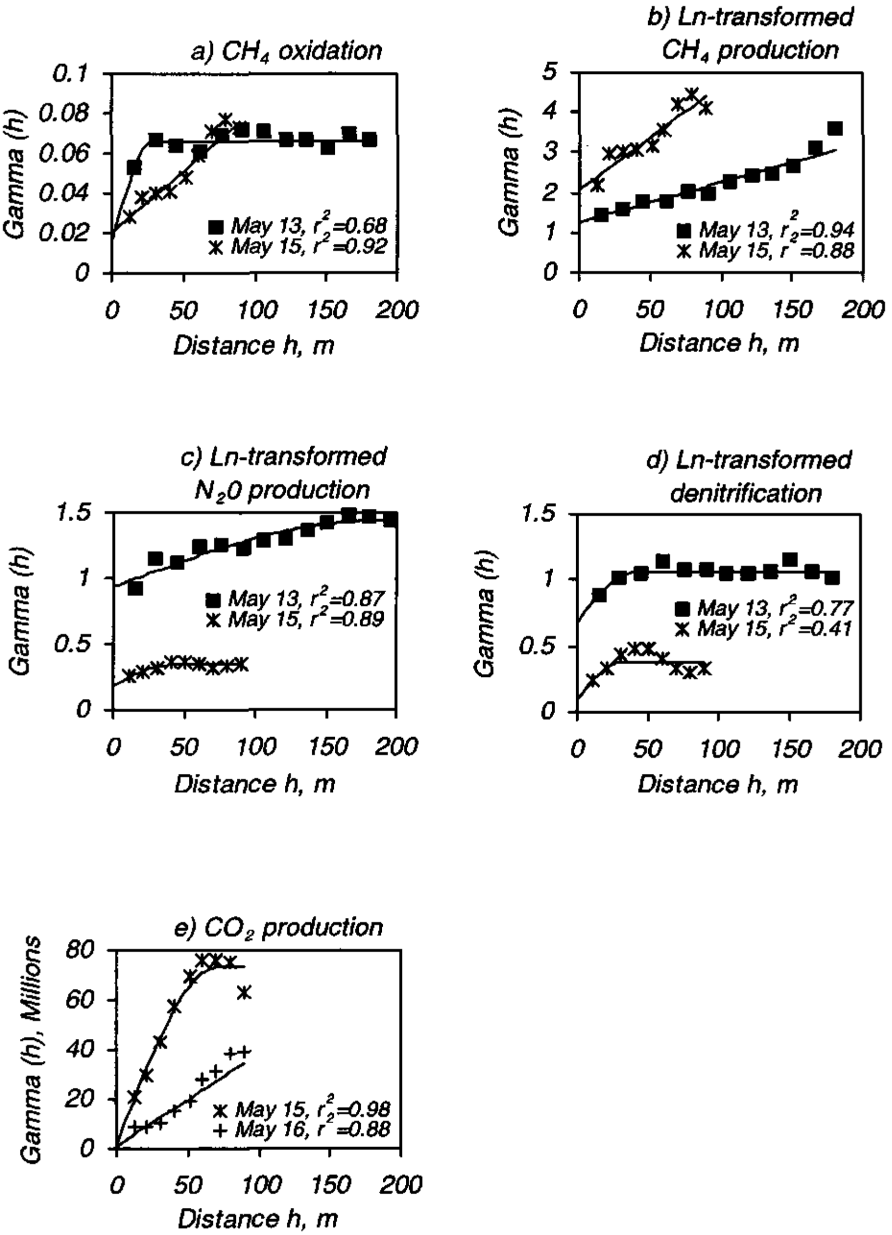

Fig. 6.3. Variogram of **a)** CH_4 oxidation, **b)** ln-transformed CH_4 production, **c)** ln-transformed N_2O production, **d)** ln-transformed denitrification, and **e)** CO_2 production: variance $\gamma(h)$ vs. distance h (m).

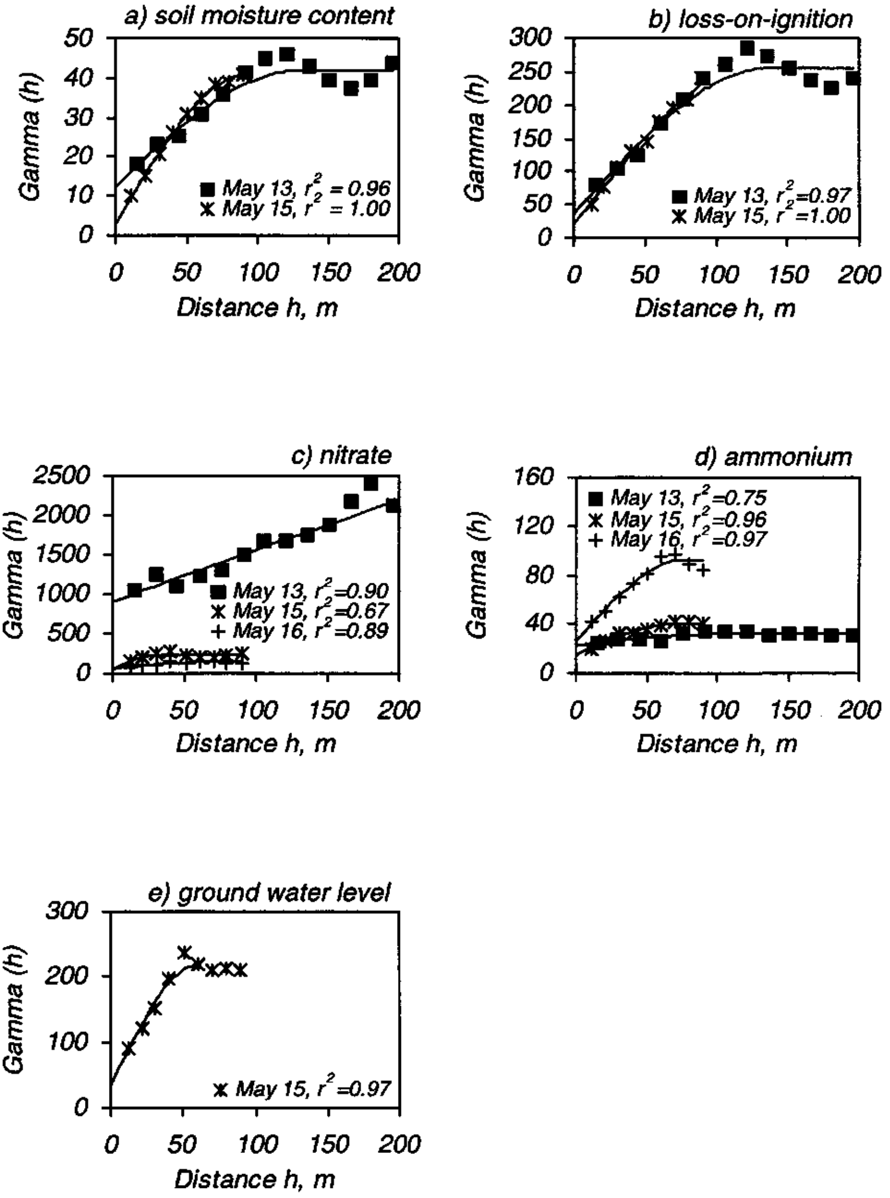

Fig. 6.4. Variogram of **a**) soil moisture content, **b**) loss-on-ignition, **c**) NO_3, **d**) NH_4, and **e**) ground water level: variance $\gamma(h)$ vs. distance h (m).

Discussion

Spatial variability

Spatial variability of CH_4, N_2O, and CO_2 emissions is generally high. For example, Ambus & Christensen (1995) found coefficients of variation of 166 to 1787% for CH_4 and 106 to 617% for N_2O. Coefficients of variation were also high in our experiments, especially for CH_4 and N_2O (Table 6.1). Emissions, soil processes, and properties (Tables 6.1 and 6.3) differed significantly between the two sites ($P<0.05$), but a real comparison is difficult to make as measurements on Fields 1 and 2 were carried out on different days.

We hypothesised that spatial variability of CH_4, N_2O, and CO_2 emissions would be related to spatial variability of soil processes and properties. Even though a large number of measurements were available and anticipated main controlling factors were included, relations between emissions and soil processes or soil properties were generally poor. A reason for this might be that the emissions were measured on an area of about 300 cm^2 while the soil processes and properties were determined using soil cores with a surface area of approximately 17.5 cm^2, which were taken from the centre of the flux chambers. The volume of the soil cores may not have fully represented the spatial and temporal distribution of the soil processes and properties that control emissions. This is to some extent supported by the relatively better relation of soil processes, as determined in soil cores, with soil properties (r^2 = 0.20-0.61). Furthermore, soil processes and properties were measured in the upper 20 cm of the soil while emission rates integrate the activity of the whole soil profile. Christensen et al. (1996) found the main N_2O producing layers just above the ground water at a depth of 1 m. If the main producing layers are found at depth, then properties in the upper 20 cm will not explain emission rates. Another reason for the poor relationships might be that determination of soil variables in bulk samples does not accurately represent the integrated effect of multiplicate interactions of factors in soil microsites controlling emissions. A poor relation between emissions and soil variables is often found (e.g. Ambus & Christensen, 1995; Velthof et al., 1996). Due to the poor relations, it was not possible to describe spatial variability of emissions as a function of spatial variability of soil processes or properties.

Spatial dependence

Little is known about spatial dependence of greenhouse gas emissions. Priemé et al. (1996) found spatial dependence of CH_4 emissions from two forest soils to occur only at distances smaller than about 10 to 12 m. In our study, however, the range of spatial dependence of CH_4 emissions was larger (Fig. 6.2a and 6.2b). This may be due to the much more heterogeneous distribution of controlling soil variables in the forest soil, due to the presence of trees, compared with our drained grassland soil. Velthof et al. (1996) found N_2O emissions from a slightly sloping, poorly drained clay soil to be spatially dependent throughout their whole measurement area, i.e. 10 m^2, on mown grassland. However, on intensively grazed grassland they found no spatial dependence. In our study the range of spatial dependence of N_2O emissions was larger (Fig. 6.2c). Our sites had been under grazing for several years, but as the growing season had just started, no cattle had grazed the sites yet. Grazing might decrease spatial dependence of N_2O emissions, as it will probably increase random variation due to dung and urine causing an uneven distribution of N in the soil. The work of White et al. (1987) also gave some evidence for this. They showed that variograms of NO_3 were best fitted to nugget models under grazing and to linear models when there had been no grazing for a period of 14 months. For our sites, this may imply that N_2O emissions may become less spatially dependent as soon as grazing starts.

We hypothesised that spatial dependence of CH_4, N_2O, and CO_2 emissions would be related to spatial dependence of soil processes and properties. Situations where the range of spatial dependence of emissions and the range of spatial dependence of soil variables is approximately the same indicate that several processes controlling spatial variability work at the same scale. Surface trend analysis revealed trends in several variables (Table 6.2). These variables may be analysed with more advanced methods (e.g. Kitanidis, 1983; Kitanidis & Lane, 1985), but that would fall beyond the scope of this study. However, we did compare the variograms of the stationary emissions with the variograms of soil processes and properties. We found that it was not possible to describe spatial dependence of emissions as a function of spatial dependence of soil processes or properties. Our results show that good relations between spatial dependence of soil variables may exist. For example, on 15 May, ground water level and NO_3 content, the main controlling factors for N_2O emissions, were spatially dependent for distances <30-55 m, the same as N_2O production and denitrification (Fig. 6.3 and 6.4). Where emissions were spatially dependent throughout the whole site (i.e. showed a regional trend), and anticipated controlling factors showed a range of spatial dependence, emissions must be affected by large-scale factors. Soil type may very well have been one of these factors. Although the whole site was classified as peat soil, it was not uniform, as we observed differences in clay and peat contents of the soil (data not shown).

Many of the variables exhibited a relatively high nugget variance, indicating that non-spatial variability was high compared with spatial variability. If the variables would have been studied in greater detail, heterogeneity that was defined as non-spatial variability may have been found to contain a spatial component. We expect that in our study the nugget variance represents both variance due to measurement error and due to microvariability of emissions.

Bramley & White (1991) suggested that in the case of soil physical properties, which are not expected to vary markedly with time, the variogram is not only readily reproducible, but can also be transferred between sampling areas of similar soil type and topography. We also found some evidence for this. Variograms of the physical soil properties soil moisture content and loss-on-ignition were almost identical for the two sites (Fig. 6.4a and 6.4b).

Variograms of emissions

Emissions were mainly spatially dependent. This implies that a certain precision of an emission estimate for a site may be reached with minimal sampling by the use of kriging. Kriging is a technique of making optimal, unbiased estimates of regionalised variables at unsampled locations using variograms and initial sets of data values (Trangmar et al., 1985). However, there are at least two serious limitations to this technique in the case of emissions. First, variograms from one site cannot always be applied to another, as the shape of the variogram may differ between sites. Second, spatial dependence of emissions may show short-term temporal variability, which is either shown by different variograms on succeeding days or by a change from a stationary emission on one day to a non-stationary emission on the succeeding day (Fig. 6.2, Table 6.2). The differences in spatial dependence between two successive days of measurement on the same site could not be ascribed to measurement order during a day as emissions were not significantly affected by time of measurement. Large changes in spatial variability of a variable in short periods suggest that main controlling factors of this variable are highly dynamic or the variable is highly sensitive to slight changes of the controlling factors. Water level of the soil might be one of those controlling factors for emissions. It may display day-to-day variability, which might have a relatively large effect on emissions due to a change in the anaerobic/aerobic ratio in the soil. Bramley & White (1991) already suggested that geostatistical analysis of biological properties may cause problems, due to changes of variance with time. However, they used a time period of several months to a year between their observations. Ryel et al. (1996) discovered differences in variograms of NO_3, NH_4, and soil moisture contents of a soil during a growing season. Our results show that spatial dependence of emissions may change during even much smaller time periods of a day. It appears that variograms of emissions and soil processes, which are partly biological in nature and have a high degree of inherent variability, should be interpreted with care.

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Chapter 7

Effects of soil moisture content and temperature on methane uptake by grasslands on sandy soils

Van den Pol-van Dasselaar A, van Beusichem ML & Oenema O (1998)
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EFFECTS OF SOIL MOISTURE CONTENT AND TEMPERATURE ON METHANE UPTAKE BY GRASSLANDS ON SANDY SOILS

Abstract

Aerobic grasslands may consume significant amounts of atmospheric methane (CH_4). We aimed (i) to assess the spatial and temporal variability of CH_4 uptake by grasslands on aerobic sandy soils, and (ii) to explain the variability in CH_4 uptake by differences in soil moisture content and temperature. Net CH_4 emissions were measured with vented closed flux chambers at two sites with low N input on sandy soils in the Netherlands: (i) Wolfheze, a heather grassland, and (ii) Bovenbuurtse Weilanden, a grassland which is mown twice a year. Spatial variability of net CH_4 emissions was analysed using geostatistics. In incubation experiments, the effects of soil moisture content and temperature on CH_4 consumption capacity were assessed.

Temporal variability of net CH_4 emissions at Wolfheze was related to differences in soil temperature (r^2 of 0.57) and soil moisture content (r^2 of 0.73). CH_4 uptake was highest at high soil temperatures and intermediate soil moisture contents. Spatial variability of net CH_4 emissions was high, both at Wolfheze and at Bovenbuurtse Weilanden. Incubation experiments showed that, at soil moisture contents lower than 5% (w/w), CH_4 consumption was completely inhibited, probably due to physiological water stress of methanotrophs. At soil moisture contents higher than 50% (w/w), CH_4 consumption was greatly reduced, probably due to the slow down of diffusive CH_4 and O_2 transport in the soil, which may have resulted in decreased CH_4 oxidation and possibly some CH_4 production. Optimum soil moisture contents for CH_4 consumption were in the range of 20 to 35% (w/w), as prevailing in the field. The sensitivity of CH_4 consumption to soil moisture content may result in short-term variability of CH_4 uptake in response to precipitation and evapotranspiration, as well as in long-term variability due to changing precipitation patterns as a result of climate change.

Introduction

Increased concentrations of methane (CH_4) and other greenhouse gases in the atmosphere may contribute to climate change (IPCC, 1995a). In soils, CH_4 can be formed under anaerobic conditions by methanogens. Under aerobic conditions, both CH_4 that has been produced in anaerobic parts of the soil and atmospheric CH_4 can be oxidised by methanotrophs (Segers, 1998). Aerobic grasslands may consume significant amounts of atmospheric CH_4. CH_4 uptake generally ranges from 0 to 1 mg CH_4 m^{-2} d^{-1} (e.g. Dobbie et al., 1996; Jarvis et al., 1994; Mosier et al., 1991; Van den Pol-van Dasselaar et al., 1997).

Major determining factors for both temporal and spatial variability of CH_4 uptake are grassland management and environmental conditions (e.g. Czepiel et al., 1995; Hütsch et al., 1994; Mosier et al., 1991). The most important environmental factors are ground water level and soil moisture content as they determine the proportion of anaerobic/aerobic sites in the soil. Soil moisture also acts as a diffusion barrier for methanotrophic micro-organisms. Furthermore, temperature is important, as CH_4 oxidation is a microbial process (Czepiel et al., 1995; Dobbie et al., 1996; Van den Pol-van Dasselaar et al., 1997).

Within a field, spatial variability of CH_4 uptake is most likely related to small-scale heterogeneity in soil conditions, e.g. in soil moisture content. Variability between fields is probably related to differences in grassland management, soil characteristics and climate (Czepiel et al., 1995; Dobbie et al., 1996; Hütsch et al., 1994; Mosier et al., 1991). Climate change, for example changes in rainfall distribution and temperature, may affect CH_4 uptake capacity of grasslands. A decrease in wetness and an increase in temperature will probably increase CH_4 uptake rates. However, the relationships between soil moisture, soil temperature and CH_4 uptake rates are complex (Dobbie et al., 1996; Van den Pol-van Dasselaar et al., 1997).

We aimed (i) to assess the spatial and temporal variability of CH_4 uptake by grasslands on aerobic sandy soils, and (ii) to explain the variability in CH_4 uptake by differences in soil moisture content and temperature, in order to improve our understanding of determining factors. We studied the relationships between soil moisture content, soil temperature and CH_4 uptake rates using two approaches. Firstly, CH_4 uptake rates measured in the field were related to soil moisture content and temperature, using both statistical and geostatistical methods. Secondly, relationships between CH_4 consumption capacity and soil moisture and temperature were studied in incubation experiments. We hypothesised (i) that at both low and high soil moisture contents, CH_4 consumption is small, due to microbial stress, diffusion limitation and/or CH_4 production, and (ii) that CH_4 consumption is larger at high temperatures than at low temperatures, as CH_4 oxidation is a microbial process.

Materials and methods

Site description

Experiments have been carried out at two sites in the centre of the Netherlands in the province Gelderland. They were selected for their low N input, since N input may reduce net atmospheric CH_4 uptake (Hütsch et al., 1994; Mosier et al., 1991; Steudler et al., 1989; Willison et al., 1995).

1. Wolfheze. Heather grassland on dry sandy soil, situated in a nature preserve. The site has not been fertilised for several centuries. In summer, few cattle graze the area. N input via atmospheric deposition is estimated at 40 kg N ha^{-1} yr^{-1} (Erisman & Draaijers, 1995). Dry bulk density was 1.09 g cm^{-3} in the layer 0-5 cm and loss-on-ignition (at 550 °C) was 7.3% in the layer 0-20 cm. Ground water is at least 3 m below the surface. The vegetation is dominated by grasses (*Deschampsia flexuosa* (L.) Trin., *Molinia caerulea* (L.) Moench), heather (*Ericaca tetralix L.*), and mosses.

2. Bovenbuurtse Weilanden. Poorly drained grassland on sandy soil. Until 1972, the site received about 200 kg N ha^{-1} yr^{-1}. Several grazing and mowing treatments were established in 1972. We studied a treatment, which is mown twice a year and received 50 kg N ha^{-1} yr^{-1} in the period 1972-1985 and no N application since 1986. N input via atmospheric deposition is estimated at 45 kg N ha^{-1} yr^{-1} (Erisman & Draaijers, 1995). Dry bulk density was 1.20 g cm^{-3} in the layer 0-5 cm and loss-on-ignition (at 550 °C) was 4.7% in the layer 0-20 cm. The vegetation is dominated by grasses (*Agrostis* spp., *Arrhenatherum elatius* (L.) Beauv. ex J. & C. Presl, *Anthoxanthum odoratum* L., *Festuca rubra* L., *Holcus* spp., *Lolium perenne* L., *Poa trivialis* L.).

Field experiments

Experimental set-up

At Wolfheze, net CH_4 emissions were measured with vented closed flux chambers (Hutchinson & Mosier, 1981; Mosier, 1989) weekly to monthly from March 1996 to March 1997 with 12 to 24 flux chambers per measurement date. The circular, stainless steel flux chambers (I.D. 20 cm, height 16 cm) were located at random in the field and carefully inserted in the soil to a depth of 2 to 4 cm. Thereafter, flux chambers were closed by a stainless steel lid and covered with insulating sheets to prevent temperature changes within the chambers. At regular time intervals (20 to 40 minutes) four gas samples were taken from the headspace of the chambers with glass syringes. They were transported to the laboratory and analysed for CH_4 within 24 h by injecting them manually into a gas chromatograph (PU 4400, Unicam Limited, Porapak R column, nitrogen as carrier gas) using a flame ionisation detector (coefficient of variation: 0.08%). A standard CH_4 concentration of 2.0 $\mu L\ L^{-1}$ (±5%) was used for calibration. Net CH_4 emissions were calculated from linear regression of the time course of CH_4 concentration in the headspace of the chambers. Annual mean net CH_4 emission was estimated by trapezoidal integration of mean net CH_4 emissions over time. At each measurement date, soil temperatures were recorded at 0, 10, 20, 30, 40, and 50 cm depth. Soil moisture content of the upper 10 cm was determined gravimetrically after drying at 105 °C for 24 h. Percentage ground cover by vegetation types (grasses, heather, mosses, bare ground) was recorded for each individual flux chamber.

Two measurement campaigns were carried out to study spatial variability, using the same equipment as described above. At Bovenbuurtse Weilanden, net CH_4 emissions and soil moisture content were measured in 60 flux chambers in a total area of 30 by 45 m in May 1995. Thirty of these flux chambers were placed in small drainage trenches. These trenches were 5 to 10 cm lower than the grassland. They occupy about 5% of the total surface area of Bovenbuurtse Weilanden. The second measurement campaign was carried out at Wolfheze in June 1996. The field was divided into 100 (10x10) grids of 9 by 9 m. Net CH_4 emissions, soil moisture content and dry bulk density were measured in the centre of each grid.

Statistics and geostatistics

Simple and multiple regression analyses were performed with net CH_4 emissions as dependent variable and soil temperature at the various depths, soil moisture content, dry bulk density and percentage ground cover by different vegetation types as independent variables.

Spatial variability of net CH_4 emissions in the measurement campaigns was analysed using geostatistics. Spatial variability contains systematic and random components. In standard statistics, the variability around the mean is assumed to be random and spatially independent. However, variability of soil variables often contains a spatially dependent component, i.e. the variability can be described as a function of spatial separation. Geostatistics, i.e. the theory of regionalised variables, takes into account both the systematic and random characteristics, and models the spatial dependence of spatially distributed variables (e.g. Trangmar et al., 1985). Geostatistical techniques are currently widely applied by soil scientists, e.g. in analysing spatial variability of nutrient contents (Borges & Mallarino, 1997) and hydraulic properties (Mallants et al., 1996). The degree of spatial dependence was determined with variograms. A variogram $\gamma(h)$ describes the variance of the squared difference of a spatial variable between pairs of points at distance h. Variograms were constructed, assuming isotropic (i.e. direction independent) spatial variation, by:

$$\gamma(h) = \frac{1}{2N(h)} \sum_{i=1}^{N(h)} \left\{ Z(x_i) - Z(x_i + h) \right\}^2 \qquad (7.1)$$

in which for each distance approximately equal to h, $N(h)$ is the number of pairs of observations at this distance and $Z(x_i)$ and $Z(x_i+h)$ represent the value of the spatial variable Z at two positions separated by h. Many variograms show a nugget variance, i.e. a non-zero value as the distance tends to zero. It represents variability not accounted for, either because of microscale variability, which can not be detected at the scale of sampling, or because of measurement errors. In the absence of spatial dependence, i.e. in cases of large point-to-point variation at short distances, the variogram will show a pure nugget effect.

Incubation experiments

Experimental set-up

Incubation experiments were carried out to assess the effect of soil moisture content and temperature on CH_4 consumption capacity of soil samples. The effect of soil moisture content on CH_4 consumption capacity was assessed on samples from Bovenbuurtse Weilanden and from Wolfheze. Soil of the top 10 cm of the profile was incubated at 20 °C with an initial CH_4 concentration of about 2 $\mu L\ L^{-1}$. Soil moisture content was adjusted, either by gently air-drying or adding water, to create a range of soil moisture contents from 0 to 70% (g H_2O g^{-1} dry soil). At the end of the experiment the actual soil moisture contents were determined.

The effect of temperature on CH_4 consumption capacity was studied on samples from Bovenbuurtse Weilanden. Soil of the top 10 cm was incubated at 4, 12, and 20 °C with an initial CH_4 concentration of about 2 μL L^{-1}. The moisture content of the collected soil was 23% (w/w); via air-drying or adding water, soil moisture contents of 18% and 47% (w/w) were also created. There were five replicates per treatment.

Analytical procedures and data acquisition

About 20 to 50 g (dry weight) homogenised soil was placed in incubation bottles of 580 ml and incubated aerobically. Preliminary experiments showed that the total weight of incubated soil per bottle did not affect CH_4 consumption capacity per unit of weight (data not shown). Control bottles containing no soil were included. Gas samples from the headspaces of the bottles were taken through rubber septa. Immediately after sampling, gas samples were analysed for CH_4 using a gas chromatograph (described above). Incubation lasted two to four days in which on average five samples were taken. The decrease in CH_4 concentration in the headspaces of the incubation bottles showed typical first-order-kinetics ($r^2 > 0.98$) (Hütsch et al., 1993):

$$\ln Y_t = \ln Y_0 - kt \qquad (7.2)$$

Y = CH_4 concentration in headspace, μL L^{-1};
t = time, d;
k = rate constant, d^{-1}

The rate constant k was normalised to dry weight (d^{-1} g^{-1} dry soil) and used as a characteristic value for CH_4 consumption capacity. Large values of k indicate high CH_4 consumption capacities.

Results

Field experiments

At Wolfheze, temporal variability of net CH_4 emissions was relatively high, with highest CH_4 uptake in spring and summer (Fig. 7.1). Spatial variability was large

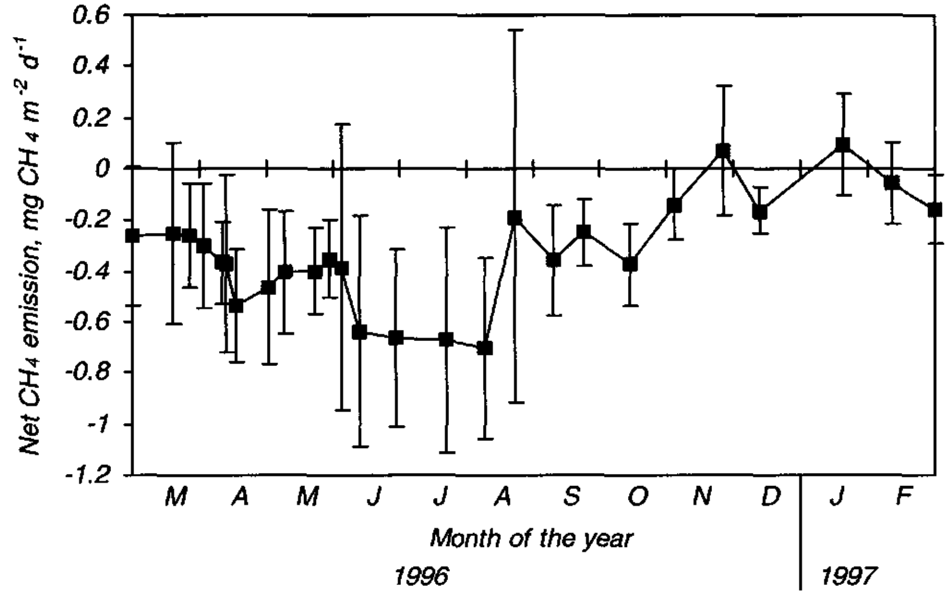

Fig. 7.1. Time course of mean net CH_4 emissions (mg CH_4 m^{-2} d^{-1} ± SD) from heather grassland at Wolfheze. Negative numbers indicate net consumption of atmospheric CH_4, positive numbers indicate net CH_4 production. Each data point is the average of 12 to 24 measurements.

throughout the year; coefficients of variation ranged from 40 to 400%. On an annual basis, grasslands at Wolfheze were a net sink of 1.1 ± 0.1 kg CH_4 ha^{-1} yr^{-1} (±SE). Simple linear regression analyses with net CH_4 emission as the dependent variable and soil temperature as the independent variable showed that net CH_4 emission was best related to soil temperature at the surface ($r^2 = 0.57$) (Fig. 7.2). For soil moisture contents higher than 20% (w/w), the relation between soil moisture content and net CH_4 emission was best fitted with a linear regression model. If all available data points were included, however, the relation was best fitted with a non-linear regression model ($r^2 = 0.73$) (Fig. 7.3).

During the measurement campaign at Bovenbuurtse Weilanden, net CH_4 emissions (±SD) were on average -0.06±0.42 mg CH_4 m^{-2} d^{-1} at a soil moisture content (±SD) of 32±4% (w/w) (trenches excluded). In the trenches, which occupy

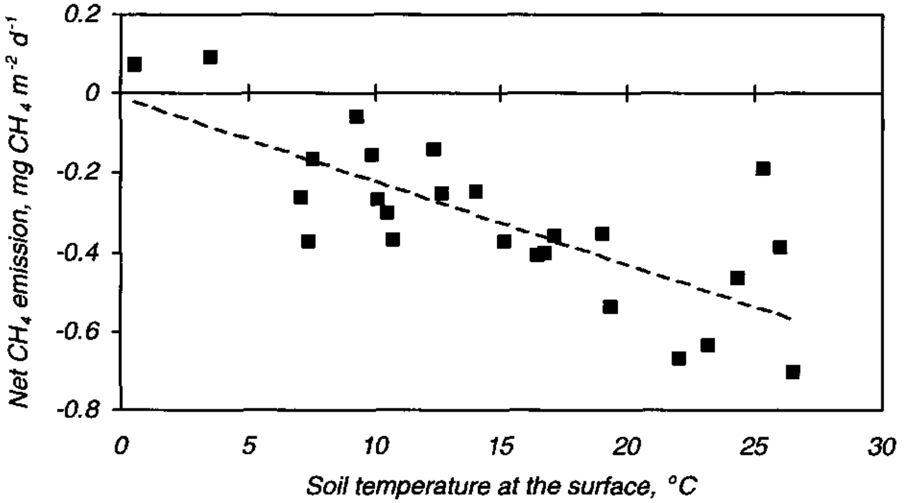

Fig. 7.2. Relationship between soil temperature at the surface (°C) and mean net CH_4 emissions (mg CH_4 m^{-2} d^{-1}) from heather grassland at Wolfheze. Each data point is the average of 12 to 24 measurements. Dashed line is best fit ($r^2 = 0.57$), obtained via linear regression analysis.

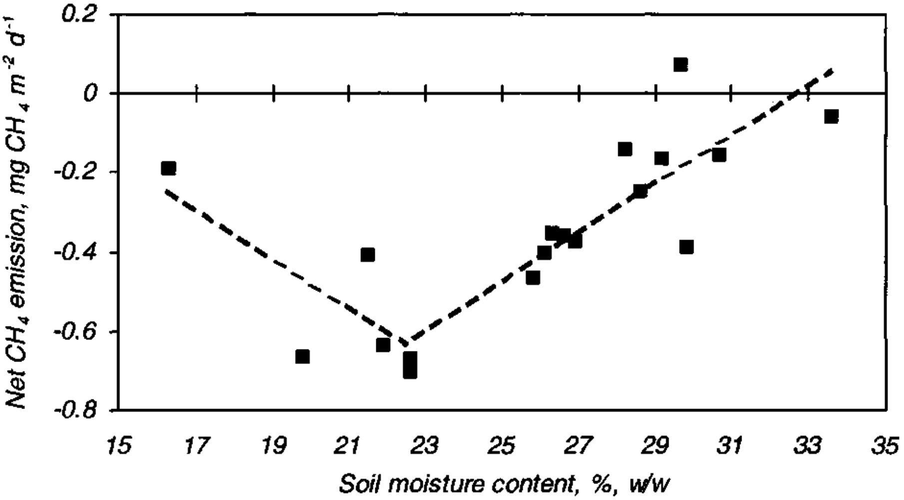

Fig. 7.3. Relationship between soil moisture content (%, w/w) and mean net CH_4 emissions (mg CH_4 m^{-2} d^{-1}) from heather grassland at Wolfheze. Each data point is the average of 12 to 24 measurements. Dashed line is best fit ($r^2 = 0.73$), obtained via non-linear regression analysis.

about 5% of the total surface area, net CH_4 emissions were on average 0.23±0.67 mg CH_4 m^{-2} d^{-1} at a soil moisture content of 39±5% (w/w). Even though this suggests a clear effect of soil moisture content on net CH_4 emission, the relationship between net CH_4 emission and soil moisture content of individual flux chambers was poor (r^2 = 0.03).

During the measurement campaign at Wolfheze, CH_4 uptake (±SD) was on average 0.65±0.44 mg CH_4 m^{-2} d^{-1}. Soil moisture content (±SD) was 22±5% (w/w). Analysis of spatial variability using geostatistics revealed a periodic component in the variogram of net CH_4 emissions (Fig. 7.4a). Soil moisture content showed spatial dependence throughout the whole site (Fig. 7.4b). Dry bulk density did not show clear spatial dependence (Fig. 7.4c). All variograms showed a positive nugget effect. Multiple linear regression analyses with net CH_4 emission as the dependent variable

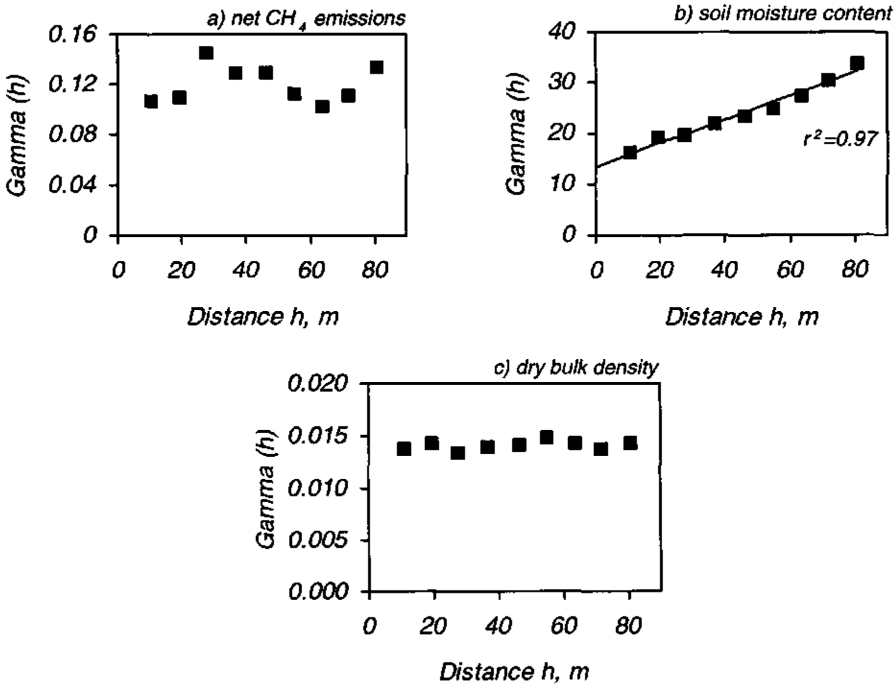

Fig. 7.4. Variogram of **a**) net CH_4 emissions, **b**) soil moisture content, and **c**) dry bulk density of heather grasslands at Wolfheze: variance $\gamma(h)$ versus distance h (m) (see Equation 7.1).

and soil moisture content and dry bulk density of each individual flux chamber as the independent variables, showed that relationships were poor (r^2 = 0.01-0.05). This may have been due to the small range of soil moisture contents and dry bulk densities found in the field.

Vegetation type was neither significantly related to net CH_4 emission at Wolfheze nor to net CH_4 emission at Bovenbuurtse Weilanden, although there was a tendency of atmospheric CH_4 uptake to increase in the order heather, grasses, mosses and no vegetation (not shown). This might be due to facilitation of CH_4 diffusion into the soil.

Incubation experiments

There was a distinct effect of soil moisture content on CH_4 consumption capacity of soil samples from Bovenbuurtse Weilanden and from Wolfheze (Fig. 7.5). At soil moisture contents lower than 5% (w/w), CH_4 consumption capacity was completely inhibited. Optimum soil moisture contents ranged from 22.5 to 37.5% (w/w) for Bovenbuurtse Weilanden and from 17.5 to 27.5% (w/w) for Wolfheze. At soil moisture contents higher than 50% (w/w), CH_4 consumption capacity was greatly reduced. CH_4 consumption capacity was higher in soil samples from Wolfheze than in soil samples from Bovenbuurtse Weilanden.

The effect of soil temperature on CH_4 consumption capacity of soil samples from Bovenbuurtse Weilanden is shown in Fig. 7.6. An increase in temperature from 4 to 12 °C more than doubled CH_4 consumption capacity. The effect of a further increase to 20 °C was smaller, especially at high soil moisture contents.

Discussion

Spatial and temporal variability of CH_4 uptake

Variability of soil variables can often be described as a function of spatial separation (e.g. Trangmar et al., 1985). Little is known about spatial dependence of net CH_4 emissions. Priemé et al. (1996) found spatial dependence of net CH_4 emissions from two forest soils to occur at distances smaller than about 10-12 m. Van den Pol-van Dasselaar et al. (1998a) found that the range of spatial dependence of net CH_4 emissions varied from 50 to more than 100 m on drained grasslands on peat soil,

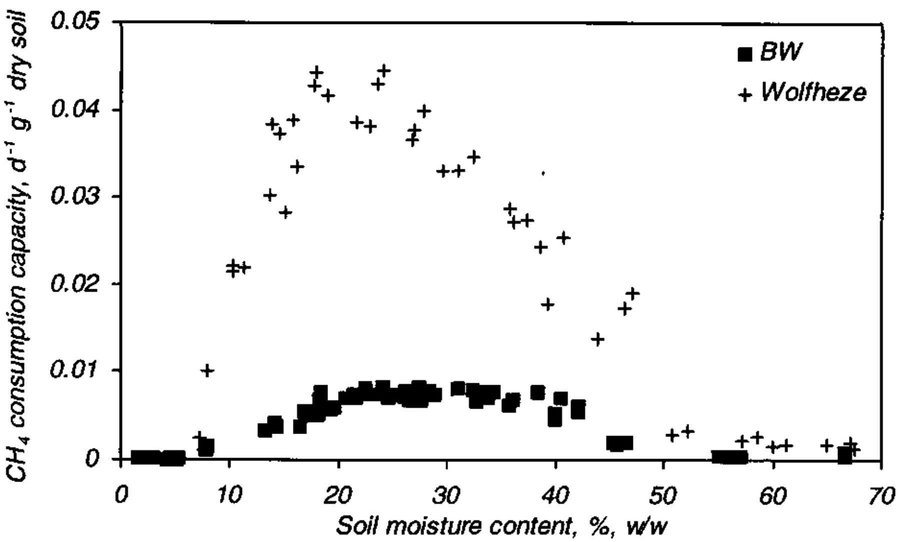

Fig. 7.5. Effect of soil moisture content (%, w/w) on normalised rate constants of CH_4 consumption capacity (d^{-1} g^{-1} dry soil) (see Equation 7.2) of the 0-10 cm layer of Bovenbuurtse Weilanden (BW) and of Wolfheze.

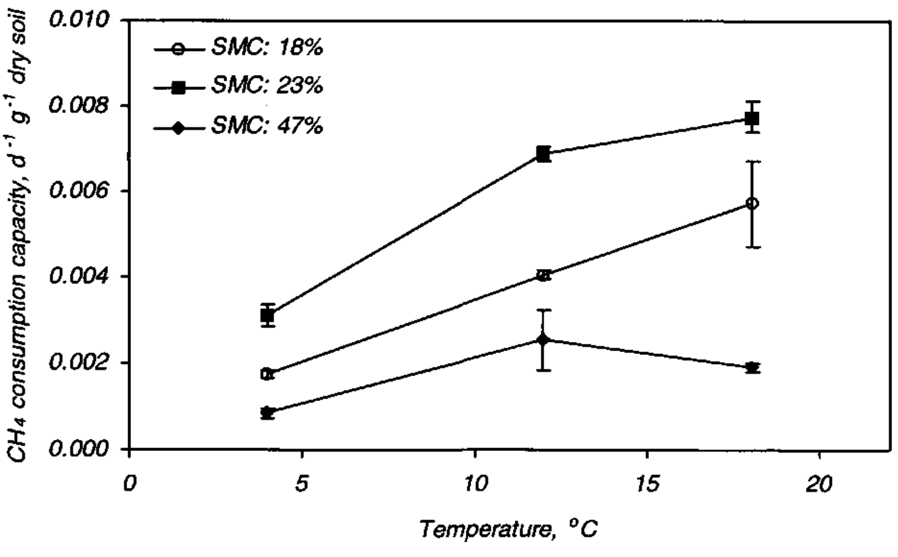

Fig. 7.6. Effect of temperature (°C) on normalised rate constants of CH_4 consumption capacity (d^{-1} g^{-1} dry soil ± SE) (see Equation 7.2) of the 0-10 cm layer of Bovenbuurtse Weilanden for different soil moisture contents (SMC) (%, w/w). Each data point is the average of five measurements.

possibly due to a rather homogeneous distribution of determining soil variables in the intensively managed grassland soil. Van den Pol-van Dasselaar et al. (1998a) showed that spatial dependence of emissions may change with time, such as within a day. Therefore, variograms of emissions should be interpreted with care. The variogram of net CH_4 emission in the measurement campaign at Wolfheze (Fig. 7.4a) reveals a periodic component. However, it is not possible to give a plausible reason for the occurrence of this periodic component. The variograms of net CH_4 emission, soil moisture content, and dry bulk density (Fig. 7.4) exhibited a relatively high nugget variance, indicating that non-spatial variability was high compared with spatial variability. If the variables would have been studied in greater detail, heterogeneity that was defined as non-spatial variability may have been found to contain a spatial component. We expect that in our study the nugget variance represents both variance due to measurement errors and due to microvariability of the variables.

Spatial variability of net CH_4 emissions was not related to differences in soil conditions like soil moisture content or dry bulk density, nor to differences in vegetation type. This may be due to small ranges of these variables. Furthermore, net CH_4 emission integrates the activity of the whole soil profile while soil moisture content and dry bulk density were determined only in the upper 10 cm of the soil.

Temporal variability of net CH_4 emissions at Wolfheze (Fig. 7.1) was related to differences in soil temperature and soil moisture content (Fig. 7.2 and 7.3). It must be noted however that high temperatures often coincided with low soil moisture contents, and vice versa. Hence, it is not clear whether the relationships shown in Fig. 7.2 and 7.3 are a direct effect of temperature or of soil moisture content, or a combined effect of both temperature and soil moisture content.

Effect of soil moisture content

An important factor determining CH_4 uptake is gas transport resistance, which is influenced by soil wetness and structure. The effect of soil moisture content has been studied in many recent incubation experiments (e.g. Adamsen & King, 1993; Dunfield et al., 1995; Koschorreck & Conrad, 1993; Nesbit & Breitenbeck, 1992; Schnell & King, 1996; Whalen & Reeburgh, 1996) and field experiments (e.g. Castro et al., 1994; Castro et al., 1995; Koschorreck & Conrad, 1993; Striegl et al., 1992). Both at low and high soil moisture contents, CH_4 consumption capacity may be suppressed, either by physiological water stress of methanotrophs or by restriction of diffusive CH_4 and O_2 transport. The optimum soil water content for CH_4 uptake reflects the balance between gas transport rates and physiological water stress. An increasing soil moisture content may also decrease net atmospheric CH_4 uptake, due to increased CH_4 production as a result of an increasing proportion of anaerobic sites (Yavitt et al., 1995). Czepiel et al. (1995) suggested that CH_4 oxidation is influenced by both soil moisture content and organic matter content, with differences in organic matter content explaining the spatial variability of CH_4 oxidation at optimum soil moisture contents. We found a clear effect of soil moisture content on CH_4 consumption capacity (Fig. 7.5). Optimum soil moisture contents for CH_4 consumption were in the same range as the prevailing field soil moisture contents. This means that methanotrophs are adapted to the field situation. The lower CH_4 consumption capacity of soil samples from Bovenbuurtse Weilanden compared to soil samples from Wolfheze (Fig. 7.5) may have been due to a higher N turnover rate at Bovenbuurtse Weilanden (Van den Pol-van Dasselaar et al., 1998e), since it has been suggested that a high N turnover rate decreases CH_4 uptake (Mosier et al., 1991).

Effect of temperature

It has been reported that the temperature dependence of CH_4 oxidation is smaller than that of many other biological processes like carbon dioxide production and CH_4 production (Dunfield et al., 1993; King & Adamsen, 1992). This can be explained by the fact that CH_4 oxidation is often limited by CH_4 diffusion, which is a physical process. The optimum temperature for CH_4 oxidation is lower than the optimum temperature for CH_4 production (Nesbit & Breitenbeck, 1992). According to Dunfield et al. (1993), the optimum temperature for CH_4 oxidation is 20-25 °C. Castro et al. (1995) found in their field experiments that soil temperature is an important controller of CH_4 uptake at temperatures between 5 and 10 °C, but not at temperatures between 10 and 20 °C. In agreement with this, we found that the effect of temperature on CH_4 consumption capacity was more pronounced between 4 and 12 °C than between 12 and 18 °C (Fig. 7.6).

Climate change

CH_4 uptake is restricted at low and high soil moisture contents (Fig. 7.3 and 7.5), and at low temperatures (Fig. 7.2 and 7.6). If, as a consequence of increased greenhouse gas emissions, temperatures on earth will increase, then CH_4 uptake may also increase, unless, as a consequence of a warmer climate, soil moisture content decreases drastically. Further on, the sensitivity of CH_4 consumption to soil moisture content may result in short-term variability of CH_4 uptake in response to precipitation and evapotranspiration, as well as in long-term variability due to changing precipitation patterns as a result of climate change.

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Chapter 8

Effects of nitrogen input and grazing on methane fluxes of extensively and intensively managed grasslands in the Netherlands

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Effects of nitrogen input and grazing on methane fluxes of extensively and intensively managed grasslands in the Netherlands

Abstract

Generally, grasslands are considered as a sink of atmospheric methane (CH_4), and nitrogen (N) input as a reducing factor for CH_4 uptake. We aimed to assess the short- and long-term effects of a wide range of N inputs, and of grazing versus mowing, on CH_4 uptake by grasslands in the Netherlands. These grasslands are mostly intensively managed with a total N input via fertilisation and atmospheric deposition in the range of 300-500 kg N ha^{-1} yr^{-1}. Net CH_4 emissions were measured with vented closed flux chambers at four contrasting sites, which were chosen to represent a range of N inputs. There were no significant effects of grazing versus mowing, stocking density, and withholding N fertilisation for three to nine years, on net CH_4 emissions. When ground water level was close to the soil surface, injection of cattle slurry resulted in significant net CH_4 production. Highest atmospheric CH_4 uptake was found at the site with the lowest N input and the lowest ground water level, with an annual CH_4 uptake of 1.1 kg CH_4 ha^{-1} yr^{-1}. This is assumed to be the upper limit of CH_4 uptake by grasslands in the Netherlands. We conclude that grasslands in the Netherlands are a net sink of CH_4 with an estimated CH_4 uptake of 0.5 Gg CH_4 yr^{-1}. At the current rates of total N input, the overall effect of N fertilisation on net CH_4 emissions from grasslands is small or negligible.

Introduction

The atmospheric methane (CH_4) concentration is increasing, which is of concern as it may contribute to global warming. Methane (CH_4) is the most important greenhouse gas next to CO_2. It has been reported to contribute about 20% to the enhanced greenhouse effect (IPCC, 1995a). CH_4 can be formed in soils wherever organic matter is degraded by micro-organisms under anaerobic conditions. Under aerobic conditions, the soil may be a sink where methanotrophic micro-organisms oxidise either atmospheric CH_4, or CH_4 that has been produced in anaerobic parts of the soil. On a global scale, soils contribute 3 to 9% to the total sink strength for atmospheric CH_4 (IPCC, 1995a).

Grasslands are generally considered as sinks of CH_4. The net uptake rate depends on environmental conditions, like ground water level, soil moisture content, temperature, and grassland management (Czepiel et al., 1995; Dunfield et al., 1995; Van den Pol-van Dasselaar et al., 1998d). The carbon (C) and nitrogen (N) dynamics of grazed grasslands differ from that of mown grasslands, but whether this difference causes a difference in CH_4 flux has not yet been demonstrated. Several authors have observed that N fertilisation decreases CH_4 uptake by soil. This is caused either by an immediate inhibition of methanotrophy (short-term effect) or by a change in the composition and size of the microbial community due to repeated fertiliser N application (long-term effect). The decrease of CH_4 uptake by soil may be associated with (i) ammonium (Dunfield & Knowles, 1995; Hütsch et al., 1994; King & Schnell, 1994; Willison et al., 1995), (ii) nitrite (Dunfield & Knowles, 1995; King & Schnell, 1994), and (iii) high N turnover rates (i.e. mineralisation and nitrification) (Hütsch et al., 1993; Mosier et al., 1991). Understanding of the mechanisms of decreasing CH_4 consumption capacity due to N fertilisation is limited, because the knowledge of the underlying microbiology is poor (Roslev et al., 1997).

Most grasslands in the Netherlands are intensively managed with a total N input via fertilisation and atmospheric deposition in the range of 300-500 kg N ha^{-1} yr^{-1}. High N inputs commenced after 1950 following a rapid intensification of livestock farming and industry. Unfertilised and extensively managed grasslands have a total N input of about 30-50 kg N ha^{-1} yr^{-1} via atmospheric deposition of NH_y and No_x (Erisman & Draaijers, 1995).

We aimed to assess the short- and long-term effects of a wide range of N inputs, and of grazing versus mowing, on CH_4 uptake by grasslands in the Netherlands. We investigated the effects of withholding N fertilisation, application of different types of mineral N fertilisers, and application of cattle slurry. Net CH_4 emissions were measured at four contrasting sites, which were chosen to represent a range of N inputs. We hypothesised (i) that N fertilisation reduces net CH_4 uptake in the short-, and (ii) the long-term, causing differences in net CH_4 uptake between sites with different N inputs, (iii) that withholding of N fertilisation increases net CH_4 uptake, and (iv) that grazing decreases net CH_4 uptake compared to mowing, as the annual amount of C and N applied to the soil is larger under grazing than under mowing (Van den Pol-van Dasselaar & Lantinga, 1995).

Materials and methods

Site description

Field experiments have been carried out at four sites in the Netherlands. Characteristics of these sites are shown in Table 8.1.

1. Wolfheze. Heather grassland on dry sandy soil, situated in a nature preserve. The vegetation is dominated by grasses, heather and mosses. Ground water is at least 3 m below the surface. The site has not been fertilised for several centuries. In summer, few cattle graze the area to maintain the grassland. N input via atmospheric deposition is estimated at 40 kg N ha^{-1} yr^{-1} (Erisman & Draaijers, 1995).

2. Bovenbuurtse Weilanden. Poorly drained grassland on sandy soil. The vegetation is dominated by grasses. N input via atmospheric deposition is estimated at 45 kg N ha^{-1} yr^{-1} (Erisman & Draaijers, 1995). Between 1950 and 1972, the site received about 200 kg N ha^{-1} yr^{-1} via fertilisation. Different treatments were established in 1972:

G_{low}: grazing with low stocking density, about 3 animals ha^{-1}, no mineral N application;
G_{high}: grazing with high stocking density, about 4.5 animals ha^{-1}, no mineral N application;
M-: mowing twice a year, N application of 50 kg N ha^{-1} yr^{-1} as calcium ammonium nitrate in the period 1972-1985, no N application since 1986;
M+: mowing twice a year, N application of 50 kg N ha^{-1} yr^{-1} as calcium ammonium nitrate.

3. Wildekamp. Poorly drained grassland on sandy soil with a total N input via fertilisation and atmospheric deposition of about 400-500 kg N ha^{-1} yr^{-1}. The vegetation is dominated by grasses. Three treatments were investigated:

CaNi: fertilisation with calcium nitrate, 80 kg N ha^{-1} cut^{-1};
AmSu: fertilisation with ammonium sulphate, 80 kg N ha^{-1} cut^{-1};
Slur: fertilisation with cattle slurry, injected with a sod-injector to a depth of 5 cm, 15 m^3 ha^{-1} cut^{-1}, which was equal to a mineral N application rate of about 45 kg N ha^{-1} cut^{-1}. In the Netherlands, animal manure must be injected into the soil in order to reduce ammonia volatilisation.

Table 8.1. Characteristics of the sites Wolfheze, Bovenbuurtse Weilanden, Wildekamp and Zegveld.

	Wolfheze	Bovenbuurtse Weilanden	Wildekamp	Zegveld
Soil type	sand	sand	sand	peat
Mean ground water level (m below the surface)	>3	0.5-1	0.5-1	0.2-0.4
Grassland management	extensive grazing	extensive mowing, extensive grazing	intensive mowing	intensive mowing, intensive grazing
N input via fertilisation and atmospheric deposition (kg N ha^{-1} yr^{-1})	40	45-95	400-500	35-460
LOI in the layer 0-20 cm, %	7	5	3	42
pH	3.7 (pH-H_2O)	n.d.*)	4.8 (pH-KCl)	4.9 (pH-KCl)
Major plant species	*Erica tetralix* *Deschampsia flexuosa* *Molinia caerulea*	*Agrostis* spp. *Arrhenatherum elatius* *Festuca rubra* *Holcus* spp.	*Lolium perenne*	*Lolium perenne*

*)n.d. = not determined

4. Zegveld. Grassland on intensively managed and moderately drained peat soil with both high and low N inputs. The vegetation is dominated by grasses. There were two sites:

Z_{low}: relatively low mean ground water level of 42 cm below soil surface in the experimental year 1994;

Z_{high}: relatively high mean ground water level of 22 cm below surface in 1994.

On both sites, there were three treatments:

M- : mowing, withholding N fertilisation since 1992 (before 1992 about 400 kg fertiliser N ha^{-1} yr^{-1} was applied as calcium ammonium nitrate);

M+ : mowing, N application;

G+ : grazing, N application.

In 1994, cumulative N application rates for both M+ and G+ were 378 kg N ha^{-1} yr^{-1} for site Z_{low} and 426 kg N ha^{-1} yr^{-1} for site Z_{high}. Fertiliser N was applied as calcium ammonium nitrate. Input via atmospheric deposition is estimated at 35 kg N ha^{-1} yr^{-1} (Erisman & Draaijers, 1995).

Monitoring net CH_4 emissions in the field

Net CH_4 emissions were measured with vented closed flux chambers (Hutchinson & Mosier, 1981; Mosier, 1989). At Wolfheze, net CH_4 emissions were measured weekly to monthly from March 1996 to March 1997 with 12 to 24 flux chambers. At Bovenbuurtse Weilanden, net CH_4 emissions were measured three to four times in the period 6 May to 5 July 1994 with four to six flux chambers per treatment, and three to five times in the period 20 January to 28 February 1995 with six to 12 flux chambers per treatment. At Wildekamp, net CH_4 emissions were measured during 29 days following fertiliser application on 30 May 1994, with four flux chambers per treatment. At Zegveld, net CH_4 emissions were measured weekly to biweekly from December 1993 to January 1995 with six to 12 flux chambers per site (two to four flux chambers per treatment). At the start of the measurements, circular, stainless steel flux chambers (I.D. 20 cm, height 16 cm) were carefully inserted in the soil to a depth of 2 to 4 cm at all sites, except treatment Slur at Wildekamp. For this treatment rectangular flux chambers were used (length 80 cm, width 20 cm, height 16 cm) to cover a representative fraction of grassland with and without injected slurry per flux chamber. Flux chambers were closed by a stainless steel lid and covered with insulating sheets to prevent temperature changes within the chambers. At regular time intervals (20 to 40 minutes), four gas samples were taken from the headspace of the chambers with glass syringes. At each measurement date, soil temperatures were recorded at 0, 10, 20, 30, 40, and 50 cm depth. Soil moisture content of the upper 10 cm was determined gravimetrically after drying at 105 °C for 24 h.

Incubation experiments

The effect of N fertilisation and grazing versus mowing on CH_4 consumption capacity was studied in an incubation experiment with field moist soil samples from different depths of Bovenbuurtse Weilanden. About 25 g (dry weight) homogenised soil of the layers 0-5, 5-10, 10-20, and 20-30 cm depth was incubated aerobically in bottles of 580 ml at 20 °C with an initial CH_4 concentration of about 80 $\mu L\ L^{-1}$. There were three replicates per soil layer. Control bottles containing no soil were included. Gas samples from the headspace of the bottles were taken through rubber septa. Incubation lasted one week in which four gas samples were taken.

Analytical procedures and data acquisition

Gas samples from the field experiments were analysed for CH_4 within 24 h by gas chromatography (PU 4400, Unicam Limited) using a flame ionisation detector (coefficient of variation: 0.08%). A standard CH_4 concentration of 2.0 $\mu L\ L^{-1}$ (±5%) was used for calibration. Gas samples of the incubation experiments were analysed for CH_4 immediately after sampling.

In the incubation experiments, the decrease in CH_4 concentration in the headspace of the incubation bottles showed typical first-order-kinetics:

$$\ln Y_t = \ln Y_0 - kt \tag{8.1}$$

Y = CH_4 concentration in headspace, $\mu L\ L^{-1}$;
t = time, d;
k = rate constant, d^{-1}

The rate constant k was normalised to dry weight ($d^{-1}\ g^{-1}$ dry soil) and used to characterise CH_4 consumption capacity.

Net CH_4 emissions were calculated for the field experiments from linear regression of the time course of CH_4 concentration in the headspace of the flux chambers. Annual mean net CH_4 emissions were estimated by trapezoidal integration of mean net CH_4 emissions over time. Statistical differences between treatments were tested with an analysis of variance procedure (ANOVA) with factor treatment (P=0.05).

Results

Net CH_4 emissions

Net CH_4 emissions from grasslands at Wolfheze and Zegveld throughout the year are shown in Table 8.2. On average, Wolfheze was a net sink of CH_4 with an annual mean uptake of 1.1 kg CH_4 ha^{-1} yr^{-1}. At Zegveld, differences in CH_4 emissions between sites were significant ($P<0.05$). The site with a relatively low ground water level (Z_{low}) consumed on average 0.3 kg CH_4 ha^{-1} yr^{-1}, while the site with a relatively high ground water level (Z_{high}) consumed on average 0.1 kg CH_4 ha^{-1} yr^{-1}. There were no significant differences in net CH_4 emissions between the treatments. Temporal and spatial variability of net CH_4 emissions at Zegveld and Wolfheze are discussed by Van den Pol-van Dasselaar et al. (1997) and Van den Pol-van Dasselaar et al. (1998d), respectively.

Net CH_4 emissions at Bovenbuurtse Weilanden are shown in Table 8.2. On average, treatment M+ was a small sink, while the other treatments were small sources of CH_4. Soil moisture content was on average 37% (w/w) at the mowing treatments and 43% at the grazing treatments, both in summer and winter. Soil temperature of the upper 5 cm was on average 18 °C in summer and 7.5 °C in winter. Spatial variability of net CH_4 emissions was high, especially at the grazing treatments. Often, one or two high net CH_4 production values and several low net CH_4 consumption values were found. There were no significant differences in net CH_4 emissions between different treatments, or between the two measurement periods.

Table 8.2. Mean net CH_4 emissions from grasslands at Wolfheze, Zegveld and Bovenbuurtse Weilanden throughout the year. Negative numbers indicate net consumption of atmospheric CH_4, positive numbers indicate net CH_4 production. Z_{low} = Zegveld, relatively low ground water level, Z_{high} = Zegveld, relatively high ground water level; B = Bovenbuurtse Weilanden, M = mowing, G = grazing (G_{low} = low stocking density, G_{high} = high stocking density); - = no mineral N application, + = mineral N application.

	March-May	June-Aug	Sept-Nov	Dec-Feb
		mg CH_4 m^{-2} d^{-1} ± SD		
Wolfheze	-0.36±0.09	-0.54±0.19	-0.17±0.16	-0.07±0.10
Z_{low}.M-	-0.04±0.03	-0.17±0.14	-0.12±0.03	-0.05±0.11
Z_{low}.M+	-0.07±0.05	-0.24±0.14	-0.06±0.13	0.02±0.05
Z_{low}.G+	-0.04±0.07	-0.22±0.14	-0.04±0.10	0.00±0.06
Z_{high}.M-	-0.05±0.05	-0.11±0.20	0.02±0.09	0.04±0.06
Z_{high}.M+	-0.00±0.17	-0.19±0.15	-0.03±0.02	0.00±0.05
Z_{high}.G+	0.21±0.26	-0.18±0.10	0.01±0.05	0.04±0.05
		May-July		Jan-Feb
B.M-		0.11±0.62		-0.05±0.14
B.M+		-0.00±0.61		-0.04±0.21
B.G_{low}		-0.22±0.16		0.91±3.65
B.G_{high}		0.23±0.83		0.19±1.03

Effect of type of fertiliser application

Type of fertilisation, i.e. mineral fertiliser or cattle slurry, markedly influenced net CH_4 emissions between soil and atmosphere at Wildekamp (Fig. 8.1). Mean net CH_4 emissions (± SE) were 0.02 ± 0.06 mg CH_4 m^{-2} d^{-1} for CaNi (fertilisation with calcium nitrate), -0.09 ± 0.05 for AmSu (fertilisation with ammonium sulphate) and 7.76 ± 2.29 for Slur (fertilisation with cattle slurry, injected with a sod-injector). There were no significant differences between the two mineral fertiliser treatments. However, net CH_4 emissions were significantly different for the treatment with cattle slurry compared to the mineral fertiliser treatments ($P<0.05$). Treatment Slur showed net CH_4 production during the experimental period. The time course of net CH_4 production followed the time course of ground water level with a delay of three to four days (Fig. 8.1). During the first two weeks of the experimental period, soil temperature at 5 cm depth was about 15 °C. In the last two weeks, it gradually increased to 23 °C. Net uptake of CH_4 for all treatments at the end of the experimental period coincided with a low ground water level and high soil temperatures.

Effect of withholding N fertilisation and grazing versus mowing

Within sites, the effect of withholding N fertilisation for three years (comparison of treatments M- and M+ at Zegveld) or nine years (comparison of treatments M- and M+ at Bovenbuurtse Weilanden) on net CH_4 emissions was not significant ($P<0.05$) (Table 8.2). Furthermore, the incubation experiments with soil samples from Bovenbuurtse Weilanden showed no differences in CH_4 consumption capacities of soil samples from treatments M- and M+ ($P<0.05$). The average profile of CH_4 consumption capacity of all treatments at Bovenbuurtse Weilanden showed a subsurface maximum at 10-20 cm depth (Fig. 8.2).

At Bovenbuurtse Weilanden, the effect of grazing versus mowing (comparison of M and G) and stocking density (comparison of G_{low} and G_{high}) on net CH_4 emissions was not significant ($P<0.05$) (Table 8.2). Furthermore, at Zegveld the effect of grazing versus mowing (comparison of M+ and G+) on net CH_4 emissions was not significant ($P<0.05$) (Table 8.2).

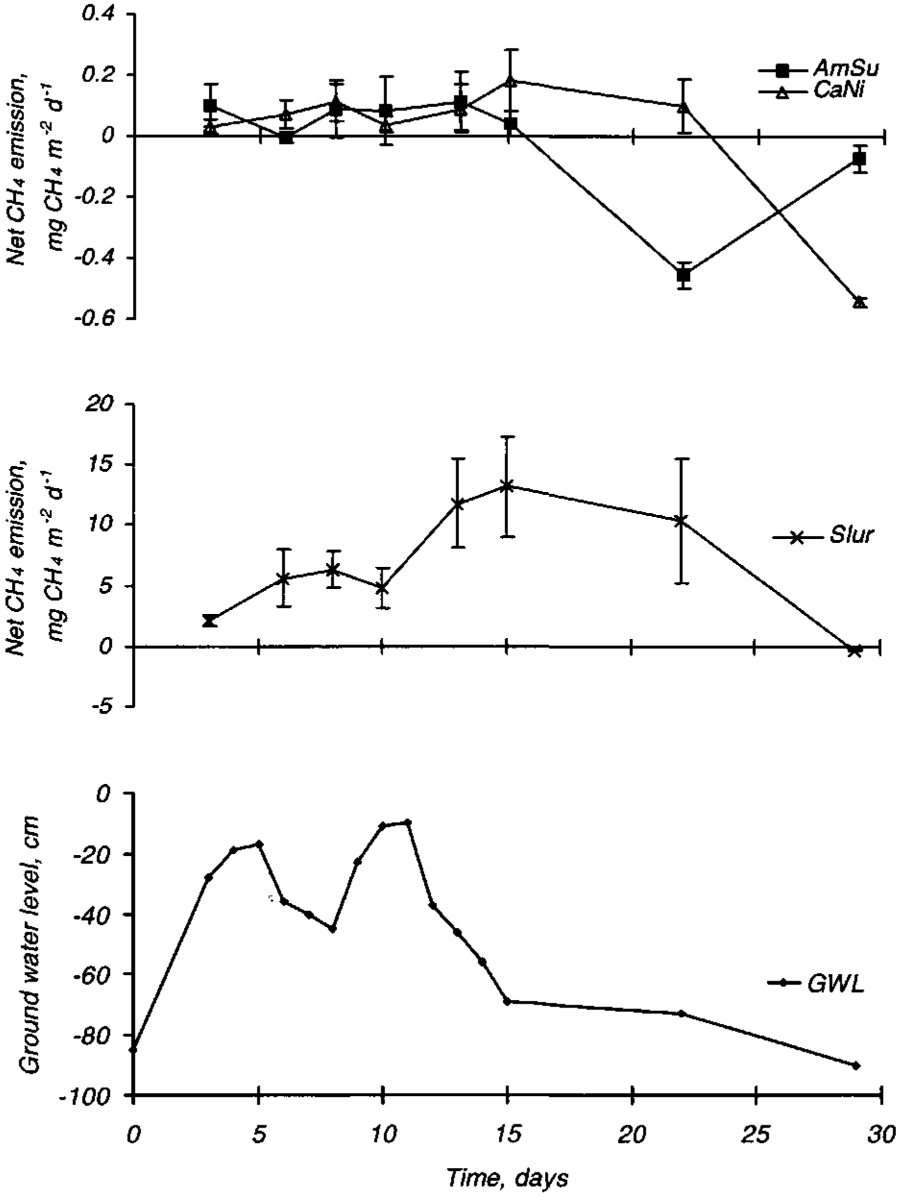

Fig. 8.1. Mean net CH_4 emissions (mg CH_4 m^{-2} d^{-1} ± SE) after N application via ammonium sulphate (AmSu), calcium nitrate (CaNi) and cattle slurry injected into the soil (SLur), and ground water level (GWL) (cm below the surface) at Wildekamp. Each data point is the average of four measurements.

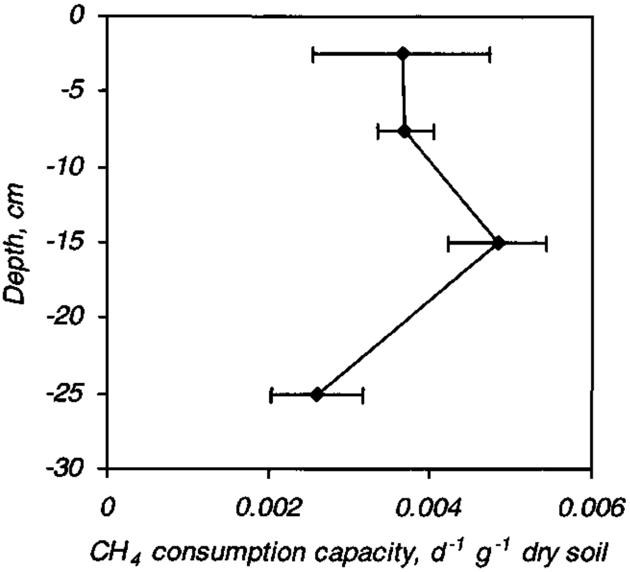

Fig. 8.2. Mean normalised rate constants of CH_4 consumption capacity (d^{-1} g^{-1} dry soil ± SE) of soil samples from different layers (cm below the surface) at Bovenbuurtse Weilanden, determination in aerobic incubations at 20 °C.

Discussion

Net CH_4 uptake

Temperate grassland soils, especially when they are well-drained and thus mainly aerobic, are generally a sink for atmospheric CH_4 with an average uptake of 0-1 mg CH_4 m^{-2} d^{-1} (Dobbie et al., 1996; Jarvis et al., 1994; Kruse & Iversen, 1995; Mosier et al., 1991). Our results fit in this range (Table 8.2). However, several sites sometimes acted as a source instead of a sink. This may have been due to relatively high ground water levels and soil moisture contents, which will have created anaerobic microsites near the soil surface. These anaerobic microsites may have produced CH_4, thus decreasing the sink function of the soil.

The incubation experiments with soil samples from Bovenbuurtse Weilanden showed a subsurface maximum in CH_4 consumption capacity at 10-20 cm depth

(Fig. 8.2). This phenomenon has been found earlier (e.g. Dunfield et al., 1995; Koschorreck & Conrad, 1993; Schnell & King, 1996; Van den Pol-van Dasselaar et al., 1997). It might be caused by outcompetition of methanotrophic bacteria by other, better adapted organisms in the top soil (Koschorreck & Conrad, 1993). Alternatively, methanotrophs could be sensitive to moisture changes, and so may be unable to survive in the top soil (Schnell & King, 1996). The latter hypothesis is supported by results from Nesbit & Breitenbeck (1992), who showed that methanotrophic activity of completely air-dried soils did not recover after rewetting.

Effect of N fertiliser type

At Wildekamp, we tested the effect of N fertiliser type on net CH_4 emissions. Net CH_4 emissions after N application via ammonium sulphate were not significantly different from net CH_4 emissions after N application via calcium nitrate, even though ammonium is often associated with a decrease in CH_4 uptake (Dunfield & Knowles, 1995; Hütsch et al., 1994; King & Schnell, 1994; Willison et al., 1995). This may have been due to conditions being favourable for CH_4 production during the first weeks. Net CH_4 emissions from the injected slurry treatment were significantly higher than from the mineral fertiliser treatments (Fig. 8.1). The net CH_4 production from the slurry treatment is probably caused by a combination of wet soil, application of easily decomposable organic material and anaerobic conditions in the slurry itself. When ground water level dropped at the end of the measurement period, the slurry treatment became a small net sink of CH_4. It has been shown that farm yard manure may stimulate CH_4 consumption capacity by increasing the microbial biomass (Willison et al., 1996). Hütsch et al. (1993) showed that application of farm yard manure did not have a significant long-term effect on CH_4 consumption capacity of soils. Probably, application of organic manure does not affect the long-term CH_4 consumption capacity of the soil.

Effect of N input

N input, especially in form of ammonium, and high N turnover rates may decrease CH_4 uptake (e.g. Hütsch et al., 1994; Mosier et al., 1991; Willison et al., 1995), either by an immediate inhibition of methanotrophy (short-term effect) or by a change in the composition and size of the microbial community due to repeated fertiliser N application (long-term effect).

We did not find any significant short-term effects of N fertilisation on net CH_4 emissions. At Zegveld, there was no significant effect of withholding N fertilisation for three years. At Bovenbuurtse Weilanden, we also found no significant effect of withholding N fertilisation, even though the period without N fertilisation at M- was nine years and before that period only 50 kg N ha^{-1} yr^{-1} was applied. However, combination of data from Wolfheze and Zegveld, the two sites with year-round measurements, suggest that there might be a significant long-term effect of N input causing differences in annual mean net CH_4 emissions (Fig. 8.3). This could however also be partly due to differences in moisture content, since Wolfheze, the site with the highest CH_4 uptake, does not only have the lowest N input, but also the lowest ground water level. Ground water level and soil moisture content were found to be important determining factors for temporal variability of net CH_4 emissions from Zegveld and Wolfheze (Van den Pol-van Dasselaar et al., 1997; Van den Pol-van Dasselaar et al., 1998d). Whatever the precise cause, our data indicate that the overall effect of N fertilisation on net CH_4 emissions from grasslands is small or negligible at the current rates of N input in the Netherlands.

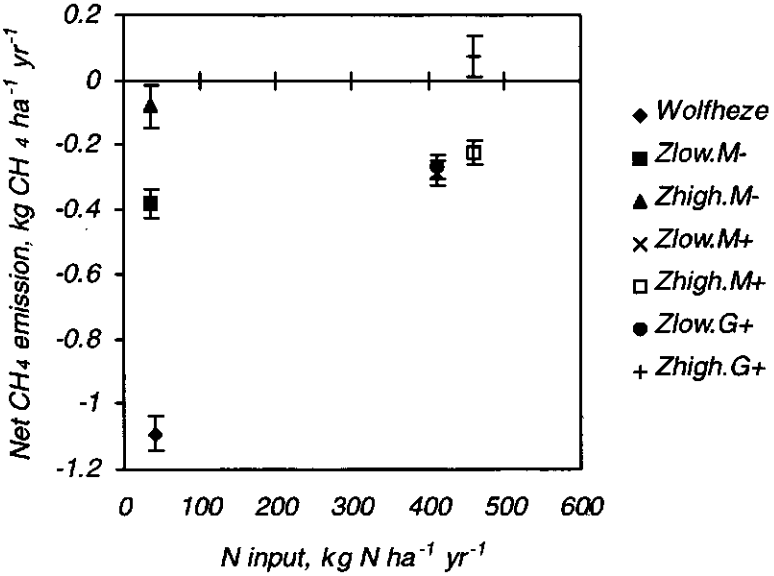

Fig. 8.3. Relationship between estimated mean annual N input (kg N ha^{-1} yr^{-1}) and mean annual net CH_4 emissions (kg CH_4 ha^{-1} yr^{-1} ± SE).

Effect of grazing versus mowing

Grazing versus mowing may affect CH_4 uptake through differences in annual supply of C and N to the soil (Van den Pol-van Dasselaar & Lantinga, 1995). At Zegveld, the effect of grazing versus mowing on net CH_4 emissions was not significant. This may have been due to the relatively short period of three years in which different treatments had been present. However, also at Bovenbuurtse Weilanden, the effect of grazing versus mowing and stocking density on net CH_4 emissions was not significant, despite the fact that treatments were established more than 20 years before. Spatial variability was often higher at the grazing treatments than at the mowing treatments (Table 8.2), probably as a result of cattle excreta creating a heterogeneous pattern of methanogenesis and methanotrophy in the soil. On spots with fresh dung, conditions are favourable for methanogenesis (high organic matter content and low oxygen content). Methanogenesis causes relatively high CH_4 concentrations in the soil, which may stimulate methanotrophs. Increased methanotrophy may still continue when methanogenesis has decreased. Clearly, these factors contribute to a large spatial variability of CH_4 emissions from grazed grasslands. However, our data indicate that the overall effect of grazing on net CH_4 emissions from grasslands is negligible. It has to be emphasised that CH_4 production by cattle themselves is not taken into account in these estimates.

CH_4 uptake by grasslands in the Netherlands

About 30% of the total surface area in the Netherlands is occupied by intensively managed grasslands, amounting to 1,050,000 ha. Recent estimates of national CH_4 emissions (Van Amstel et al., 1993; Van den Born et al., 1991) did not consider grasslands as a sink of CH_4. By contrast, drained organic soils, which are mainly used as grasslands, have been estimated to be a net CH_4 source of 32-89 Gg CH_4 yr^{-1} (35-700 kg CH_4 ha^{-1} yr^{-1}) (Van Amstel et al., 1993). Van Amstel et al. (1993) based their estimates on research at organic soils in other countries (Aselmann & Crutzen, 1989; Moore & Knowles, 1989). However, our year-round measurements at Zegveld show that drained organic soils in the Netherlands are a net sink of CH_4 with an annual mean net CH_4 uptake of 0.1 to 0.3 kg CH_4 ha^{-1} yr^{-1} (see also Van den Pol-van Dasselaar et al., 1997). We consider 0.1 kg CH_4 ha^{-1} yr^{-1}, which is the mean annual CH_4 uptake at site Z_{high}, the site with the relatively high ground water level, as the lower limit of CH_4 uptake by intensively managed grasslands in the Netherlands, because conditions of this site are least favourable for CH_4 uptake, as follows: (i) peat soil, which has the potential of emitting CH_4 as it has a high organic matter content and is largely anaerobic due to the high ground water level, (ii) intensive management with high N fertilisation and a high N turnover rate. The net uptake of 1.1 kg CH_4 ha^{-1} yr^{-1} at Wolfheze may be the upper limit of CH_4 uptake by grasslands in the Netherlands, because conditions at this site are most favourable for CH_4 uptake: (i) prevailing optimal soil moisture contents for CH_4 uptake (Van den Pol-van Dasselaar et al., 1998d), (ii) extensive management with no N fertilisation during past centuries and a low N turnover rate. We conclude that grasslands in the Netherlands (excluding wetlands/undrained peatlands that occupy only 0.5% of the total surface area) are a net sink of CH_4 with an estimated CH_4 uptake of 0.5 Gg CH_4 yr^{-1}. Estimates of CH_4 emissions in the Netherlands need to take account of the role of grasslands in the national CH_4 budget.

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Chapter 9

General discussion

GENERAL DISCUSSION

Introduction

The major aims of this thesis are (i) to provide insight into the major factors that contribute to net CH_4 emissions from grasslands, and (ii) to provide quantitative data on net CH_4 emissions from typical grasslands with a range of soil wetness and N input in the Netherlands. Monitoring of CH_4 emissions and possible determining factors was done at several sites: wet peat soils with low N input (Chapters 2, 3 and 4), drained peat soils with low to high N input (Chapters 5, 6 and 8), and relatively dry sandy soils with low N input (Chapters 7 and 8). In this final chapter, main findings of this study are discussed.

Until this study,

- Determining factors of CH_4 emissions from grasslands in the Netherlands had not been assessed experimentally;
- Measurements of CH_4 emissions from grasslands with a range of soil wetness and N input in the Netherlands were not available;
- No attempts had been made to quantify CH_4 production capacities of individual soil fractions;
- It was unknown whether agriculturally used grasslands on drained peat soil were a source or a sink of CH_4;
- It was not clear whether emissions of greenhouse gases from grasslands were spatially dependent;
- The effect of grazing versus mowing on net CH_4 emissions was not known;
- Short-term and long-term effects of a wide range of N input on CH_4 uptake by grasslands with on average high N input were not known.

Determining factors for CH_4 emissions

Determining factors for CH_4 emissions from grasslands can be split up into environmental factors and management factors. Important environmental factors examined in this thesis are: soil organic matter, ground water level, soil moisture content, temperature, pH, and vegetation. Important management factors examined in this thesis are: drainage, N fertilisation and grazing versus mowing.

Environmental factors

Soil organic matter

CH_4 production is influenced by both quality and quantity of organic matter present (e.g. Crozier et al., 1995; Schimel, 1995; Whiting & Chanton, 1993; Yavitt & Lang, 1990). If metabolisable carbon and micro-organisms are abundant, carbon turnover, methanogenesis and methanotrophy will be stimulated. Methanogenesis may be positively related to C mineralisation. However, C mineralisation is not a good predictor of methanogenesis, since relations between C mineralisation and methanogenesis may be highly variable (Segers, 1998; Chapter 3, Chapter 4).

Some researchers believe that soil organic matter is the main substrate for methanogens (e.g. Schimel et al., 1995), while others believe that fresh plant material is the main substrate (e.g. Chanton et al., 1995; Whiting & Chanton, 1993). In Chapter 4, we showed for wet peat soils that CH_4 production capacity was highest for soil fractions with a large particle size, and that CH_4 production capacity decreased with soil depth. This indicates that the major substrate for methanogens in wet peat soils is recently died plant material. Probably, this also accounts for drained peat soils, since only the top layer of these soils is capable of producing significant amounts of CH_4 (Chapter 5).

Ground water level

Ground water level is one of the main influencing factors of CH_4 emissions (e.g. Bartlett & Harriss, 1993; Moore & Roulet, 1993). Ground water level roughly indicates the transition zone between anaerobic (potential CH_4 producing) and aerobic (potential CH_4 consuming) layers in the soil. Significant CH_4 emissions occur only when mean ground water level is near the surface. Below the ground water level of peat soils, significant CH_4 accumulation may occur (e.g. Dinel et al., 1988; Fig. 2.5; Fig. 5.5). When the ground water level drops, this CH_4 may be released by degassing (e.g. Moore & Dalva, 1993).

At a relatively low mean ground water level, grasslands act as a CH_4 sink. Kengen & Stams (1995) suggested that for drained peat soils at least 40 days complete anaerobiosis at 15 °C are required before significant CH_4 production occurs. Further on, only the top layer appeared to be capable of producing significant amounts of CH_4 (Kengen & Stams, 1995; Chapter 5). In practice this means that temporary waterlogging of drained peat soils during autumn and winter has relatively little effect on CH_4 emissions (Chapter 5).

In the Netherlands, ground water level is managed via the water level in the ditches. Fig. 9.1 suggests that for peat soils in the Netherlands, the mean ground water level where a site turns from a potential source into a potential sink is situated about 20 cm below the surface. Temporal variations in ground water level may cause a large response of CH_4 emissions, as shown by the relationships between ground water level and net CH_4 emissions throughout the year (e.g. Equation 2.1, Fig. 5.4).

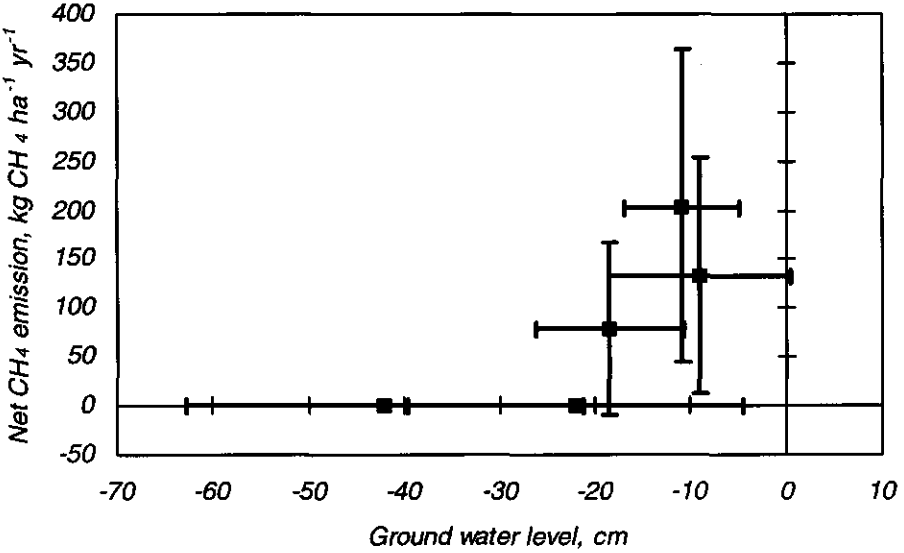

Fig. 9.1. Relationship between mean ground water level (cm below the surface ± SD; $n = 25$ to 65) and mean annual net CH_4 emissions from grasslands on peat soil (kg CH_4 ha^{-1} yr^{-1} ± SD; $n = 1$ to 3) (data from Chapter 2 and 5).

Soil moisture content

Both at low and high soil moisture contents, CH_4 consumption capacity may be suppressed by, respectively, physiological water stress of methanotrophs and restriction of diffusive CH_4 and O_2 gas transport from the atmosphere to the soil (e.g. Castro et al., 1995; Czepiel et al., 1995; Schnell & King, 1996). In incubation experiments with soil samples from two sandy soils (Chapter 7), CH_4 consumption capacity was completely inhibited at soil moisture contents lower than 5% (w/w). Optimum soil moisture contents ranged from 20 to 35%. At soil moisture contents higher than 50%, CH_4 consumption capacity was greatly reduced (Fig. 7.5). The importance of soil moisture content in the field is illustrated in Fig. 9.2 (note that the soils of Fig. 9.2 have different bulk densities). At high soil moisture contents in the field, net atmospheric CH_4 uptake may also be decreased due to CH_4 production as a result of an increased proportion of anaerobic sites in the soil (Yavitt et al., 1995).

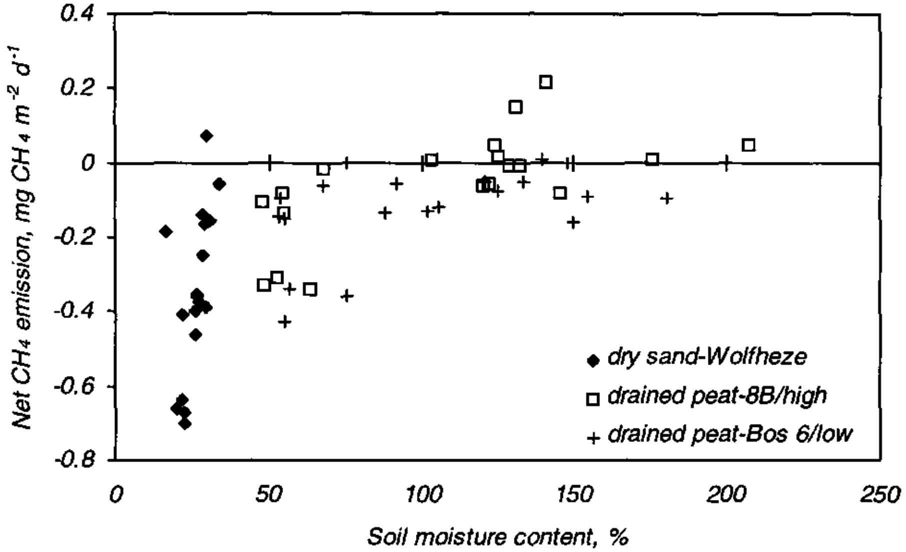

Fig. 9.2. Relationship between soil moisture content (%, w/w) and net CH_4 emissions (mg CH_4 m^{-2} d^{-1}) from grasslands on drained peat soils (data from Chapter 5) and relatively dry sandy soil (data from Chapter 7).

Temperature

Soil temperature is one of the main influencing factors of CH_4 emissions (Bartlett & Harriss, 1993; Castro et al., 1995; Dunfield et al., 1993; Chapter 2; Chapter 5; Chapter 7), since both CH_4 production and CH_4 consumption are microbiological processes. The importance of soil temperature for CH_4 emissions from both wet grasslands and drained or dry grasslands is illustrated in Fig. 9.3.

High temperatures may have a considerable impact on CH_4 emissions, as shown in an experiment under controlled conditions in the greenhouse with soil columns from intensively managed grasslands on peat soil (unpublished data,

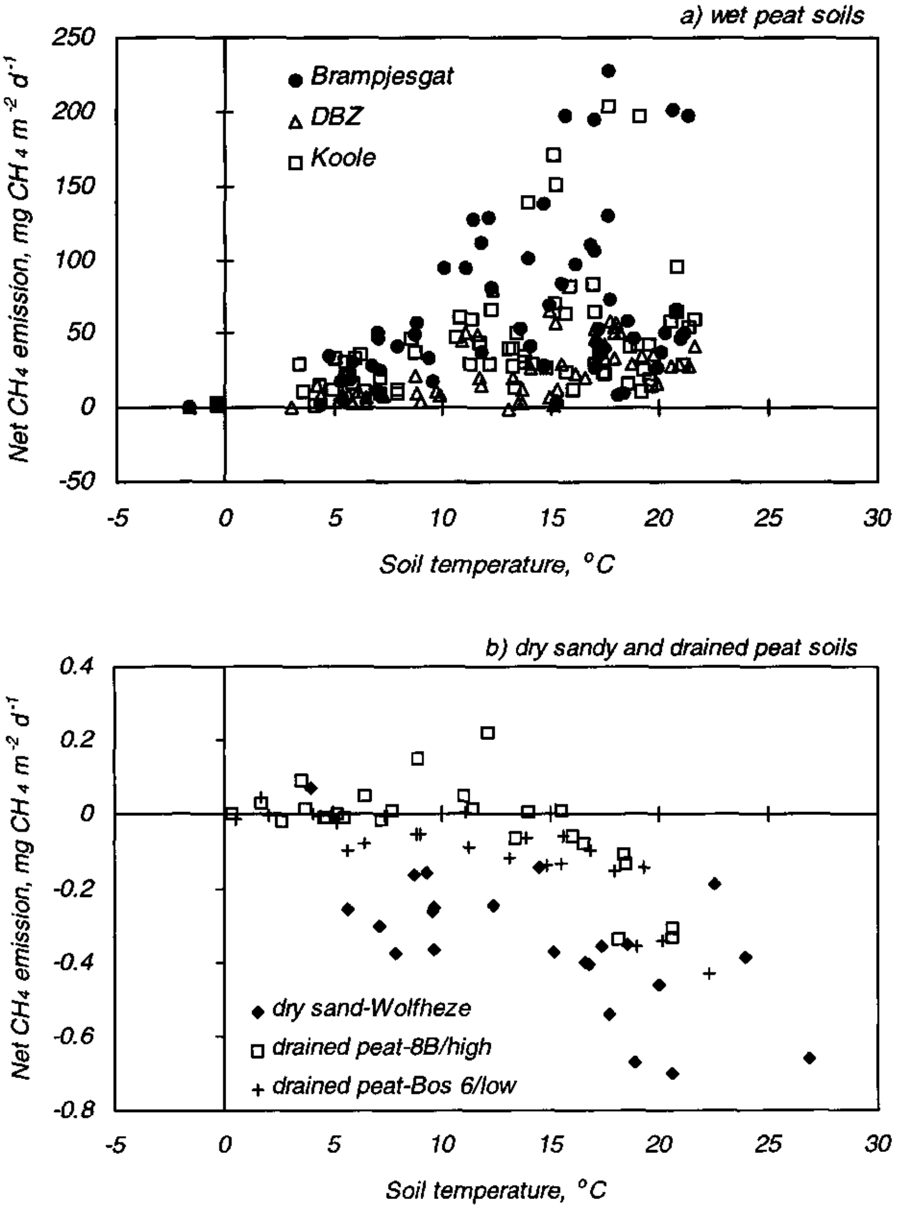

Fig. 9.3. Relationship between soil temperature at 10 cm depth (°C) and net CH_4 emissions (mg CH_4 m^{-2} d^{-1}) from **a)** wet grasslands (data from Chapter 2) and **b)** grasslands on drained peat soils (data from Chapter 5) and relatively dry sandy soil (data from Chapter 7).

Van den Pol-van Dasselaar). Even though in the field these soils were a net sink of CH_4 (Chapter 5), the soil columns were able to produce significant amounts of CH_4 within a few days (1-10 mg CH_4 m^{-2} d^{-1}) when kept at temperatures in the range of 20-30 °C. The rapid production of CH_4 may have been caused by a more pronounced stimulation of methanogenesis compared to methanotrophy, since the temperature dependence of CH_4 production is larger than the temperature dependence of CH_4 consumption (Dunfield et al., 1993; King & Adamsen, 1992).

pH

CH_4 production, CH_4 consumption, and thus CH_4 emissions may be reduced at low pH (Dunfield et al., 1993; Hütsch et al., 1994; Syamsul Arif et al., 1996; Chapter 3). However, the effect of pH can be highly variable. In Chapter 3 for example, wet grasslands with a low pH (3.5 to 5.5) showed significant CH_4 production and CH_4 emissions. The highly variable effect of pH may be due to different strains, for example acid-tolerant methanogenic bacteria (Williams & Crawford, 1985). It remains to be examined whether at the wet grasslands of Chapter 3 different strains were involved in methanogenesis. The results suggest, however, that pH is not a reliable factor for predicting CH emissions.

Vegetation

Vegetation may affect CH_4 emissions from wet grasslands (e.g. Bubier et al., 1995b; Chanton et al., 1993; Whiting & Chanton, 1993; Chapter 3; Fig. 9.4). Firstly, the organic material from plants may serve as substrate for methanogens. Secondly, plant species with aerenchymatous tissues may transport CH_4 and O_2 in the soil. Correlations between sedge biomass and net CH_4 emissions have often been found (e.g. Klinger et al., 1994; Torn & Chapin, 1993; Whiting & Chanton, 1992; Chapter 3).

The effects of vegetation on net CH_4 emissions from relatively dry grasslands are not well documented. In Chapter 7, there were some indications that atmospheric CH_4 uptake increased slightly in the order heather, grasses, mosses and no vegetation as dominant ground cover. This might be due to facilitation of CH_4 transport into the soil. However, the effect of vegetation type was not significant.

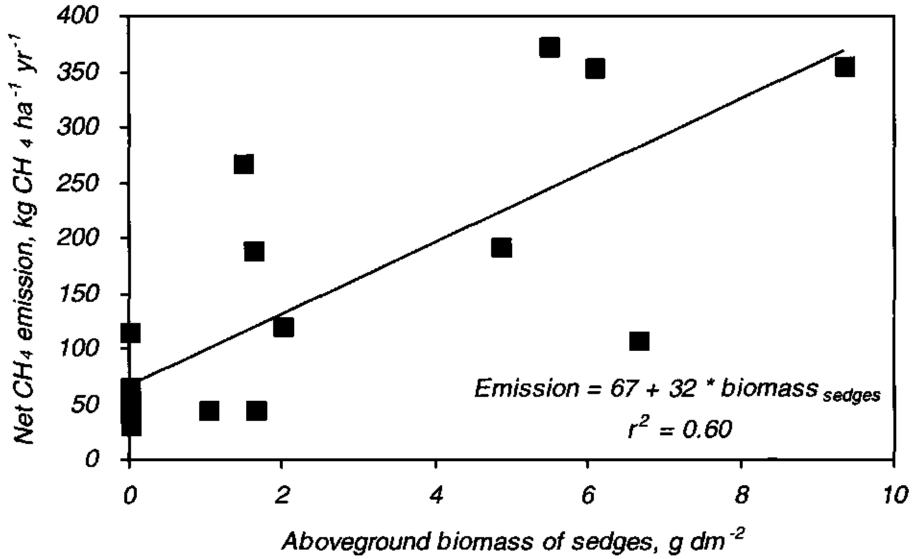

Fig. 9.4. Relationship between sedge biomass (g dm^{-2}) and mean annual net CH_4 emissions (kg CH_4 ha^{-1} yr^{-1}) (data from Chapter 3).

Management factors

Drainage

In the majority of grasslands in the Netherlands, ground water level can be managed via the water level in the surrounding ditches. Drainage decreases CH_4 emissions. As the ground water level drops, the degree of aerobiosis in the soil will increase. Conditions become less favourable for CH_4 production and more favourable for CH_4 consumption. Differences between drained and undrained sites are most distinct if the undrained sites have a ground water level close to the surface (e.g. Martikainen et al., 1992; Roulet et al., 1993; Chapter 2, Chapter 5).

N fertilisation

Most grasslands in the Netherlands are intensively managed. This management is characterised by large N inputs via N fertilisers and animal excreta. High nitrate concentrations in the soil may reduce or completely inhibit CH_4 production, since nitrate may serve as an alternative electron acceptor. Furthermore, N input, especially in the form of ammonium, may reduce CH_4 consumption.

The effect of N input on atmospheric CH_4 uptake has been described in detail in Chapter 8. In short, N input may reduce CH_4 consumption either by an immediate inhibition of methanotrophs or by a change in the microbial community due to repeated fertiliser N application. The decrease of CH_4 consumption may be associated with ammonium, nitrite, and high N turnover rates (e.g. King & Schnell, 1994; Mosier et al., 1991). It has been suggested that organic manure does not significantly affect long-term CH_4 consumption capacity of the soil, even in situations where manure contains much of its N in the ammonium form (Hütsch et al., 1993; Willison et al., 1996). However, the understanding of the mechanisms of a decreasing CH_4 consumption capacity due to N fertilisation is still limited, because knowledge of underlying microbiology is poor (e.g. Roslev et al., 1997).

Fertilisation with cattle slurry, injected with a sod-injector to a depth of 5 cm, resulted in significant CH_4 production for about one month (Chapter 8). This was probably caused by a combination of wet soil, application of easily decomposable organic material and anaerobic conditions in the slurry itself. Short-term effects of mineral N fertilisation were not found (Chapter 8). Furthermore, withholding N fertilisation for three years did not significantly affect CH_4 emissions from intensively managed grasslands on peat soil with a high N turnover rate (Chapter 5). Also, withholding N fertilisation for three to nine years did not significantly affect CH_4 emissions from extensively managed grasslands on sandy soil with a low N turnover rate (Chapter 8). However, combination of data from Chapters 5, 7, and 8 (Fig. 9.5) suggests that there might be a significant long-term effect of N input on CH_4 emissions. This may, however, also have been partly caused by differences in soil moisture content and/or ground water level, since mean ground water level was much higher at the drained peat soils compared to the dry sandy soil. In conclusion, at the current rates of N input in the Netherlands, the overall net effect of N fertilisation on net CH_4 emissions from grasslands is small or negligible.

Grazing versus mowing

Before this study, the effect of grazing versus mowing on CH_4 emissions from grasslands was not well documented. Grazing versus mowing may affect CH_4 emissions as the annual supply of C and N to the soil is higher on grazed grassland than on mown grassland (Van den Pol-van Dasselaar & Lantinga, 1995). However, grazing during three years did not significantly affect net CH_4 emissions compared to mowing at intensively managed grasslands on peat soil with a high N turnover rate (Chapter 5). Also, at extensively managed grasslands on sandy soil with a low N turnover rate, grazing during more than 20 years did not significantly affect net CH_4 emissions compared to mowing (Chapter 8). Furthermore, at these extensively managed grasslands, the effect of stocking density was not significant. It has to be emphasised that CH_4 production by cattle themselves is not included in the estimates. CH_4 production from cattle faeces dropped in the field is, however, included. In conclusion, the overall net effect of grazing on net CH_4 emissions from grasslands is negligible.

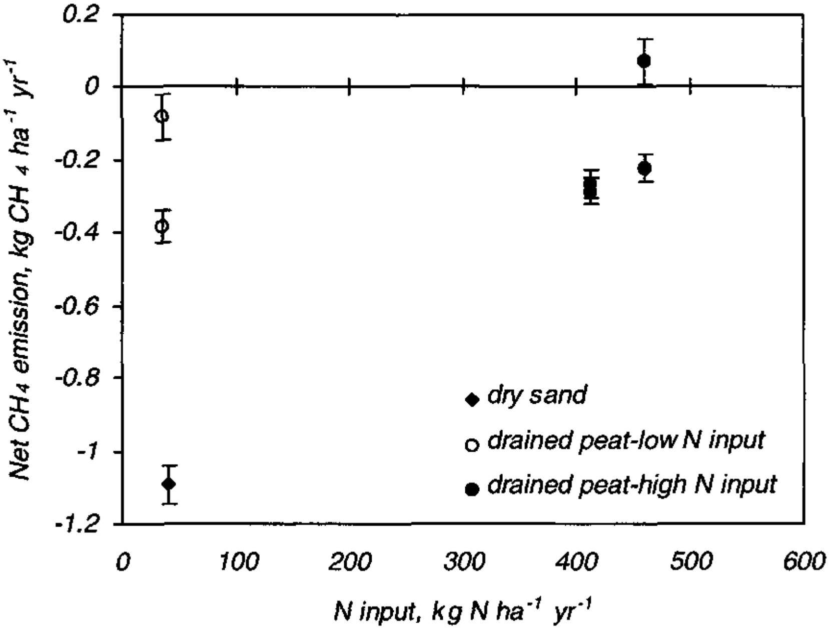

Fig. 9.5. Relationship between estimated mean annual N input (kg N ha^{-1} yr^{-1}) and mean annual net CH_4 emissions from drained peat soils and dry sandy soil (kg CH_4 ha^{-1} yr^{-1} ± SE) (data from Chapter 5 and 7, based on Fig. 8.3).

Options to mitigate CH_4 emissions from grasslands

Grassland management

The major determining factor of CH_4 emissions with respect to grassland management is ground water level. Ground water level can be managed via ditches, which are present at the majority of the grasslands in the Netherlands. To minimise CH_4 emissions, mean ground water level should be kept below about 20 cm (Fig. 9.1). In grasslands with a mean ground water level close to the surface, a considerable decrease of CH_4 emissions may be possible by drainage. However, the ground water level of these wet grasslands is often kept high to maintain a particular natural ecosystem. Also, drainage may affect emissions of other greenhouse gases. In developing mitigation policies, these effects should be taken into account.

The possible impact of grassland management on CH_4 emissions from agriculturally used grasslands is rather limited. The effects of N fertilisation, withholding N fertilisation, grazing versus mowing and stocking density on CH_4 emissions were negligible or small (Chapter 5, Chapter 8). Moreover, net CH_4 emissions from these grasslands were low. In conclusion, grassland management other than drainage is not an option to mitigate net CH_4 emissions from grasslands.

Land use change

A few centuries ago, a large part of the Netherlands consisted of wetlands. Large areas of these wetlands have been drained and are utilised for agricultural purposes. This type of land use change will have decreased CH_4 emissions considerably due to a lowering of the ground water level.

At present, the most important type of land use change with respect to CH_4 emissions from grasslands is the conversion of intensively managed grasslands (which predominantly have a low atmospheric CH_4 uptake of 0.1 to 0.3 kg CH_4 ha^{-1} yr^{-1}, Chapter 5) to nature preserves. The effect of this type of land use change depends on the characteristics of the nature preserve.

If grasslands are converted into 'dry' nature preserves with a low ground water level, the effect on CH_4 emissions will be relatively small. Uptake of atmospheric CH_4 may eventually show a small increase (e.g. to about 1 kg CH_4 ha^{-1} yr^{-1}, Chapter 7). However, it has been reported that diminished capacity for CH_4 consumption is typically persistent with slow recovery times (e.g. Hütsch et al., 1994; King, 1997;

Nesbit & Breitenbeck, 1992; Mosier et al., 1996).

The effect of conversion of intensively managed grasslands into 'wet' nature preserves with a high ground water level may be considerable. The government of the Netherlands intends to convert part of the intensively managed grasslands from agriculture into ecosystems with a high ground water level. Eventually, these grasslands will be comparable to the wet grasslands of Chapter 2 with an estimated mean annual net CH_4 emission of 80-200 kg CH_4 ha^{-1} yr^{-1}. A small CH_4 sink will thus be turned into a large CH_4 source.

Variability of CH_4 emissions

Temporal and spatial variability

Large variability, both temporal and spatial, of net CH_4 emissions from soils is a common phenomenon (e.g. Ambus & Christensen, 1995; Bartlett & Harriss, 1993; Bianchi et al., 1996; Waddington & Roulet, 1996; this thesis). Determinants of variability are related to the temporal and spatial scale at which these determinants dominate (Klinger et al., 1994) (Fig. 9.6).

In situations of net CH_4 production, most important determining factors of net CH_4 emissions will be those that create conditions favourable for methanogenesis. Once these conditions have been established, factors like soil temperature and CH_4 transport will become important (e.g. Morrissey & Livingston, 1992). In situations of net CH_4 consumption, an important determining factor will be soil moisture content (King, 1997).

Due to the complexity of CH_4 dynamics in soils, simulation models can be a useful tool for estimating CH_4 emissions and studying the effect of environmental and management factors on these emissions. In coherence with this study, Segers (Wageningen Agricultural University, former Department of Theoretical Production Ecology) has developed process based models of CH_4 production, consumption and transport in wetlands. In these models, CH_4 emissions are explained from weather, soil and vegetation data via the underlying processes, thus improving insight in the temporal and spatial variability of CH_4 emissions. However, predictive power of these models may be restricted, since quantitative knowledge about the gas transport system and the carbon supply by plants is scarce (Segers, 1998; Segers & Kengen, 1998; Segers & Leffelaar, 1995; Segers & Leffelaar, 1996).

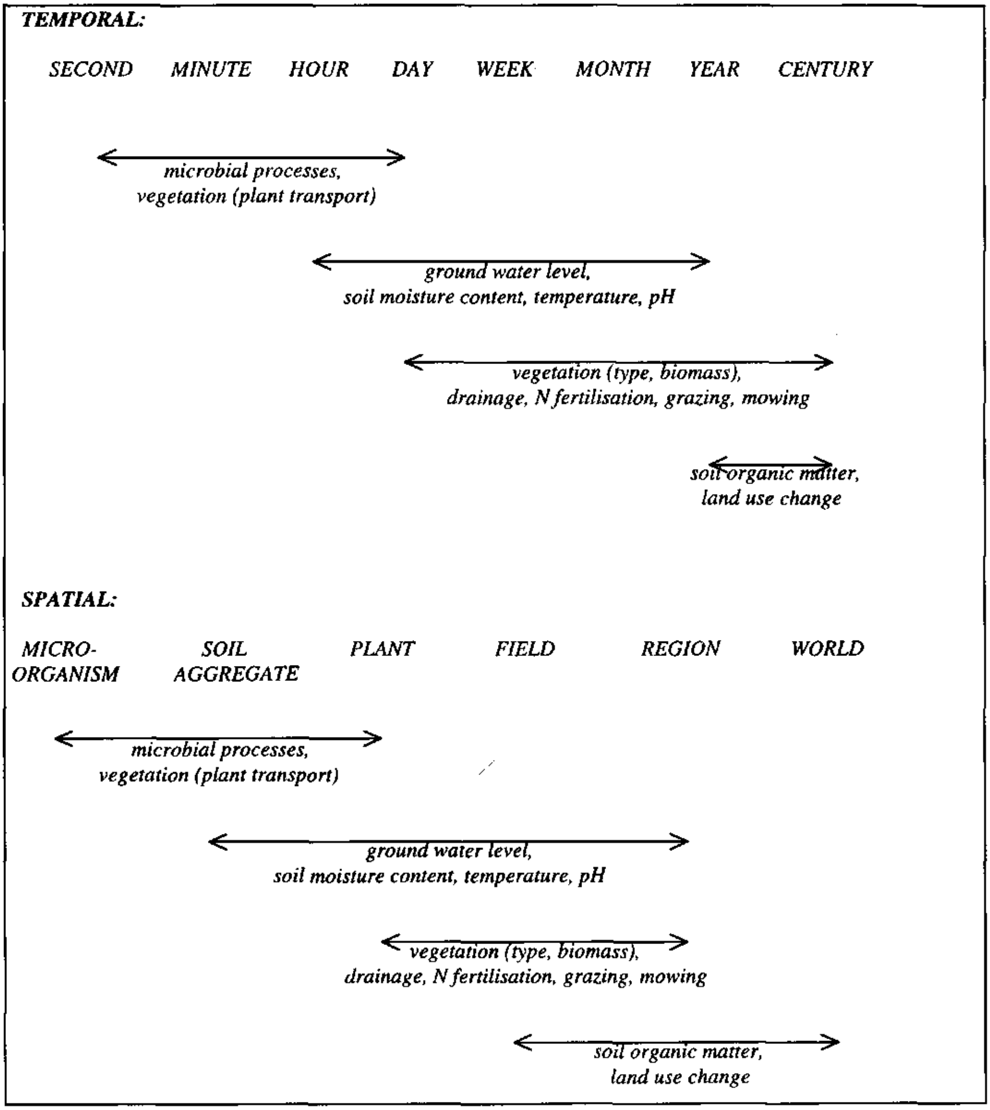

Fig. 9.6. Determinants of CH_4 emissions in relation to their temporal and spatial scales.

Spatial dependence

Spatial variability of soil factors often contains a spatial dependent component., i.e. the variability can be described as a function of spatial separation. If a factor is spatially dependent, then its value at unsampled locations may be estimated by the use of a technique called kriging (see Trangmar et al., 1985; Chapter 6).

Before this study, little was known about spatial dependence of CH_4 emissions. Priemé et al. (1996) found spatial dependence of net CH_4 emissions from two forest soils to occur at distances smaller than about 10-12 metres. In this thesis, the range of spatial dependence of net CH_4 emissions varied from 50 to more than 100 metres at intensively managed grasslands on peat soils (Chapter 6). This relatively large range of spatial dependence was possibly due to a relatively homogeneous distribution of determining soil variables. In contrast, at extensively managed grasslands on sandy soil spatial dependence of net CH_4 emissions was not clearly present (Chapter 7). This indicates that net CH_4 emissions from these grasslands exhibited a relatively heterogeneous pattern.

Spatial dependence of emissions should be interpreted with care, since it was found to differ between sites, and also between succeeding days (Chapter 6). The existence of short-term temporal variability of spatial dependence of emissions implies that emissions can not realistically be estimated by the use of kriging.

Quantification of CH_4 emissions from grasslands in the Netherlands

Thus far, CH_4 emissions from grasslands in the Netherlands were not well-documented (Van Amstel et al., 1993). To obtain realistic estimates of CH_4 emissions, measurements should cover at least one and preferably more years, since there may be large interannual variability (e.g. Mosier et al., 1996; Chapter 2). Monitoring of emissions over a long period of time is, however, not common practice, since it is very time consuming.

CH_4 emissions have been measured in a period of more than three years at grasslands in the Netherlands with a range of soil wetness and N input. Grasslands on wet peat soils with a mean ground water level ranging from 0.1 to 0.2 m below the surface and an annual N input of 30-50 kg N ha^{-1} yr^{-1} emitted 80-200 kg CH_4 ha^{-1} yr^{-1}. Ditches near these grasslands emitted 40-225 kg CH_4 ha^{-1} yr^{-1} (Chapter 2). Intensively managed grasslands on drained peat soils with a mean ground water level ranging from 0.2 to 0.4 m below the surface and an annual N input ranging from 35 to 460 kg N ha^{-1} yr^{-1} consumed 0.1-0.3 kg atmospheric CH_4 ha^{-1} yr^{-1} (Chapter 5). An extensively managed grassland on relatively dry sandy soil with a mean ground water level more than 3 m below the surface and an annual N input of 40 kg N ha^{-1} yr^{-1} consumed on average 1.1 kg atmospheric CH_4 ha^{-1} yr^{-1} (Chapter 7). Incidental measurements showed that intensively managed grasslands on drained peat soil in Sweden were net sinks of CH_4 (average net CH_4 emission: -0.04 mg CH_4 m^{-2} d^{-1}; range: -0.03 to -0.05) (Chapter 6). Extensively managed grasslands on sandy soils in the Netherlands were small net sources of CH_4 (average net CH_4 emission: 0.14 mg CH_4 m^{-2} d^{-1}; range -0.22 to 0.91) (Chapter 8). However, this was probably related to wet conditions during the measurement period.

The fact that even intensively managed grasslands on peat soil with a relatively high ground water level and high N input (Chapter 5, Chapter 6) are on average net sinks of CH_4 may suggest that all intensively managed grasslands are net sinks of CH_4. Only grasslands with an annual mean ground water level close to the surface (<20 cm; Fig. 9.1) may be a source of CH_4.

The role of soils in the national CH_4 budget

Thus far, grasslands on mineral soils were not considered as a CH_4 sink in national CH_4 budgets (Van Amstel et al., 1993). By contrast, drained peat soils, which are mainly used as grasslands, were considered a CH_4 source. Results presented in this thesis show that estimates of CH_4 emissions in the Netherlands should be adjusted to put straight the role of grasslands in the national CH_4 budget.

Net CH_4 uptake by grasslands in the Netherlands (excluding wetlands / undrained peatlands that occupy only 0.5% of the total surface area) is estimated at 0.5 Gg CH_4 yr^{-1} (Chapter 8). Wetlands/undrained peatlands emit 80-200 kg CH_4 ha^{-1} yr^{-1} (Chapter 2). Clayey soils have not been studied. CH_4 uptake by clayey soils might be smaller than CH_4 uptake by sandy soils (Boeckx et al., 1997; Born et al., 1990), but this was not taken into account in our estimates. Net CH_4 uptake by arable land is usually much smaller than net CH_4 uptake by grasslands (e.g. Ambus & Christensen, 1995; Glenn et al., 1993; Hütsch et al., 1994); therefore CH_4 uptake by arable land in the Netherlands is considered negligible. Net CH_4 uptake by forests is estimated at 2.0 Gg CH_4 yr^{-1} (Van Amstel et al., 1993).

Van Amstel et al. (1993) estimated total national CH_4 source strength at 1227 Gg CH_4 yr^{-1}. The contribution of soils was on average 70 Gg CH_4 yr^{-1}. Based on the updated CH_4 emission estimates from soils (Table 9.1), total national CH_4 source strength is estimated at 1165 Gg CH_4 yr^{-1}.

Table 9.1. Update of CH_4 emission estimates from soils in the Netherlands.

	Van Amstel et al. (1993)		This thesis	
	kg CH_4 ha^{-1} yr^{-1}	Gg CH_4 yr^{-1}	kg CH_4 ha^{-1} yr^{-1}	Gg CH_4 yr^{-1}
Wet soils	300 to 750	10 to 27	80 to 200	5 to 10
Drained grasslands	35 to 750	32 to 89	-0.1 to -1.1	-0.5
Forest soils		-2.0		-2.0

Main findings of this study

Results presented in this thesis contribute to understanding of national and global CH_4 emissions. CH_4 emissions from grasslands in the Netherlands with a range of soil wetness and N input have been quantified. Ground water level management of grasslands on wet peat soils with a ground water level ranging from 0 to 20 cm below the surface is a critical factor, since the impact of ground water level on CH_4 emissions from these grasslands may be considerable (Chapter 2). Variability of CH_4 emissions from grasslands on wet peat soils is mainly due to differences in ground water level, soil temperature, CH_4 production capacity, vegetation type and plant biomass (Chapters 2 and 3). CH_4 production capacities of individual soil fractions showed that recently died plant material is a major substrate for methanogens in wet peat soils (Chapter 4).

Intensively managed grasslands on drained peat soils are small net sinks of CH_4 (Chapter 5). Spatial dependence of greenhouse gas emissions may differ between sites, and also between succeeding days (Chapter 6). Variability of net CH_4 uptake by grasslands is mainly due to differences in ground water level, soil moisture content and soil temperature (Chapters 5 and 7).

The overall net effect of grazing versus mowing on net CH_4 emissions from grasslands is negligible (Chapter 8). At the current rates of N input in the

Netherlands, the overall net effect of N fertilisation on net CH_4 emissions from grasslands is small or negligible (Chapter 8). Grassland management other than drainage is not a suitable tool for reducing net CH_4 emissions (Chapter 9).

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Summary

Introduction

Methane (CH_4) is an important greenhouse gas. The concentration of greenhouse gases in the atmosphere has been increasing since pre-industrial times, mainly due to human activities. This increase gives concern, because it may cause global warming due to an enhanced greenhouse effect.

In the soil, CH_4 may be produced under anaerobic conditions, and consumed under aerobic conditions. Net CH_4 emissions, i.e. the resultant of CH_4 exchanges between soil and atmosphere, encompasses the processes CH_4 production, CH_4 consumption, and CH_4 transport. If CH_4 production exceeds CH_4 consumption, the soil is a source of CH_4; if CH_4 consumption exceeds CH_4 production, the soil is a sink of CH_4. The contribution of soils to the global CH_4 balance is significant: 14-47% of the total source and 3-9% of the total sink (IPCC, 1995a).

The major aims of this study were to provide insight into the major factors that contribute to net CH_4 emissions from grasslands, and to provide quantitative data on net CH_4 emissions from typical grasslands with a range of soil wetness and N input in the Netherlands. CH_4 emissions from grasslands were measured with flux chambers at a number of sites in the period 1994-1997. Furthermore, several incubation experiments were carried out. Since large variability of net CH_4 emissions is a common phenomenon, special attention was paid to temporal and spatial variability.

Wet grasslands with low N input on peat soils

The area of wet grasslands on peat soil in the Netherlands is small, but slowly increasing at the expense of drained, agriculturally used grasslands. Net CH_4 emissions were measured at wet grasslands on peat soil in the nature preserve "Nieuwkoopse Plassen", which is a former peat mining and agricultural area with narrow grassland and reed fields surrounded by ditches. Ground water level is kept near the surface via the water level of the ditches. Mean ground water level is 10-20 cm below the surface. N input is 30-50 kg N ha^{-1} yr^{-1} via atmospheric deposition. Measurements were carried out at three sites, Drie Berken Zudde, Koole and Brampjesgat, during three years. The sites were considerable sources of CH_4 with average CH_4 emissions of 79, 133 and 204 kg CH_4 ha^{-1} yr^{-1}, respectively. Ditches near the sites emitted 42-225 kg CH_4 ha^{-1} yr^{-1}. The time course of CH_4 emissions for all experimental sites and years was fit with a multiple linear regression model with ground water level and soil temperature as independent variables. Lowering or raising the ground water level by 5 cm could decrease or increase CH_4 emissions by 30-50%. Therefore, ground water level management of these grasslands should be done with care (Chapter 2). Spatial variability of CH_4 emissions was high. Most important determinants of spatial variability were CH_4 production capacity and aboveground biomass of sedges (*Carex* spp.). Sedges and other plants may affect CH_4 emissions by stimulating CH_4 transport from anaerobic layers in the soil to the surface and by serving as substrate for methanogens (Chapter 3).

In order to improve our understanding of CH_4 emissions, wet peat soils were fractionated into different size and density fractions. Incubation experiments showed that the individual fractions were rather similar with respect to C mineralisation capacity and C/N ratio, but not with respect to CH_4 production capacity. Significant CH_4 production only occurred for fractions with a large particle size. Furthermore, CH_4 production capacity strongly decreased with depth. This indicates that in these wet peat soils recently died plant material is a major substrate for methanogens (Chapter 4).

Intensively managed grasslands with a range of N input on drained peat soils

Peat soils are often considered to have a high CH_4 emitting potential, because they are anoxic at shallow depth and have high organic matter contents. In the Netherlands, the majority of the peat soils is drained. Before this study, it was not known whether drained peat soils would be a source or a sink of CH_4. Net CH_4 emissions from drained peat soils were measured at grasslands at the experimental farm Zegveld with mean ground water levels of 20-40 cm below the surface and an annual N input via fertilisation and atmospheric deposition ranging from 35-460 kg N ha^{-1} yr^{-1}. Net CH_4 emissions from these grasslands were low; they consumed 0.1 to 0.3 kg CH_4 ha^{-1} yr^{-1}. Effect of mean ground water level, in the range of 20-40 cm below the surface, was significant, but small. There were no significant effects of grazing versus mowing and withholding N fertilisation for three years on net CH_4 emissions (Chapter 5).

In order to assess the spatial variability and spatial dependence of greenhouse gas emissions (CH_4, N_2O, and CO_2), and their underlying soil processes and properties, a field campaign was carried out at grasslands on drained peat soil in Fallköping, Sweden. Emissions and potential factors controlling CH_4 emissions were measured on two adjacent sites on two successive days for each site. Spatial variability was analysed using geostatistics. Both sites were small sinks of CH_4. Spatial variability of emissions was high with coefficients of variation of 50 to 1400%. Emissions either showed a spatial trend or were spatially dependent. However, spatial dependence of emissions showed differences between sites and also between succeeding days. This implies that emissions can not realistically be estimated by the use of geostatistics (Chapter 6).

Extensively managed grasslands with low N input on relatively dry soils

To explore the maximum uptake of atmospheric CH_4 by grasslands in the Netherlands, net CH_4 emissions were measured at extensively managed heather grasslands on sandy soil in Wolfheze with a mean ground water level more than 3 m below the surface and an annual N input via atmospheric deposition of 40 kg N ha^{-1} yr^{-1}. These grasslands consumed 1.1 kg CH_4 ha^{-1} yr^{-1}. Temporal variability of net CH_4 emissions at Wolfheze was related to differences in soil temperature and soil moisture content. CH_4 uptake was highest at high soil temperatures and intermediate soil moisture contents. Incubation experiments showed that at low soil moisture contents, CH_4 consumption was completely inhibited, probably due to physiological water stress of methanotrophs. At high soil moisture contents, CH_4 consumption was greatly reduced, probably due to the slow down of diffusive CH_4 and O_2 transport in the soil. Optimum soil moisture contents were in the same range as prevailing in the field (Chapter 7).

Determining factors for CH_4 emissions

Important environmental factors determining CH_4 emissions are soil organic matter, ground water level, soil moisture content, temperature, and vegetation characteristics. Ground water level exerts a primary control over CH_4 emissions, since ground water level forms the transition zone between anaerobic (potential CH_4 producing) and aerobic (potential CH_4 consuming) layers in the soil. Therefore, drainage of wet grasslands is an important management factor determining CH_4 emissions. At the current rates of N input via fertilisation and atmospheric deposition in the Netherlands, the overall net effect of N fertilisation on net CH_4 emissions from grasslands is small or negligible. Furthermore, the effects of grazing versus mowing and stocking density on net CH_4 emissions are negligible (Chapter 8). In conclusion, grassland management, other than drainage, is not an option to mitigate net CH_4 emissions from grasslands in the Netherlands.

Quantification of net CH_4 emissions from grasslands in the Netherlands

Most grasslands in the Netherlands are intensively managed with a total N input via fertilisation and atmospheric deposition in the range of 300-500 kg N ha^{-1} yr^{-1}. Thus far, CH_4 emissions from grasslands in the Netherlands were not well-documented. However, in this study, CH_4 emissions from a number of grasslands with a range of soil wetness and N input have been quantified. Total net CH_4 uptake by grasslands in the Netherlands (excluding wet grasslands) is estimated at 0.5 Gg CH_4 yr^{-1}. Wet soils, which occupy only 0.5% of the total surface area, emit 5-10 Gg CH_4 yr^{-1}. Estimates of CH_4 emissions in the Netherlands should be adjusted to put straight the role of grasslands in the national CH_4 budget.

Samenvatting

Inleiding

Methaan (CH_4) is een belangrijk broeikasgas. De concentratie aan broeikasgassen in de atmosfeer is sinds de industrialisatie toegenomen. Deze toename geeft reden tot bezorgdheid, omdat ze kan leiden tot een stijging van de temperatuur op aarde als gevolg van het zogenaamde 'broeikaseffect'.

Methaan kan in de bodem door micro-organismen onder anaërobe omstandigheden geproduceerd en onder aërobe omstandigheden geconsumeerd worden. De uitwisseling van methaan tussen bodem en atmosfeer wordt aangeduid met netto methaanemissie. Netto methaanemissie is het resultaat van methaanproduktie, methaanconsumptie en methaantransport in de bodem. Als de methaanproduktie groter is dan de methaanconsumptie, is de bodem een bron van methaan; als de produktie kleiner is dan de consumptie, is de bodem een sink van methaan. De bodem speelt een belangrijke rol in de methaanbalans: 14-47% van de totale bronnen en 3-9% van de totale sinks wordt ingenomen door bodems (IPCC, 1995a).

De hoofddoelen van deze studie waren het vergroten van de kennis over de factoren die netto methaanemissie uit grasland bepalen en het kwantificeren van de netto methaanemissie uit graslanden met verschillende vochttoestanden en stikstofaanvoer in Nederland. In de periode 1994-1997 zijn op diverse plaatsen methaanemissies gemeten met behulp van fluxkamers. Ook zijn verschillende incubatie-experimenten uitgevoerd. Omdat de variabiliteit van de netto methaanemissie in het algemeen erg groot is, is ruime aandacht besteed aan zowel temporele als ruimtelijke variabiliteit.

Natte graslanden met een lage stikstofaanvoer op veengronden

Het oppervlak aan natte graslanden op veengrond in Nederland is klein, maar neemt langzaam toe ten koste van ontwaterde, landbouwkundig gebruikte graslanden. In het natuurgebied "Nieuwkoopse Plassen" is de netto methaanemissie uit natte graslanden op veengrond bepaald. Dit natuurgebied was vroeger een veenontginnings- en landbouwgebied. In het gebied worden smalle stroken grasland en rietveld afgewisseld met sloten. Via het slootpeil wordt de grondwaterstand hoog gehouden. De grondwaterstand ligt gemiddeld op 10-20 cm onder het maaioppervlak. De stikstofaanvoer bedraagt 30-50 kg N ha^{-1} jr^{-1} via atmosferische stikstofdepositie. Er is gedurende drie jaar op drie locaties gemeten: Drie Berken Zudde, Koole en Brampjesgat. Deze graslanden waren aanzienlijke bronnen van methaan en stootten gemiddeld 79, 133 en 204 kg CH_4 ha^{-1} jr^{-1} uit. Sloten rondom de graslanden stootten 42-225 kg CH_4 ha^{-1} jr^{-1} uit. Het verloop van de methaanemissie in de tijd kon voor alle velden en jaren beschreven worden met behulp van een meervoudig lineair regressiemodel met als verklarende variabelen grondwaterstand en bodemtemperatuur. Volgens het model kan een daling of stijging van de grondwaterstand met 5 cm een afname of toename van de netto methaanemissie met 30-50% tot gevolg hebben. Dit betekent dat bij het beheer van de grondwaterstand van deze graslanden voorzichtigheid geboden is (hoofdstuk 2). De ruimtelijke variabiliteit van de netto methaanemissie was groot en kon het best verklaard worden met behulp van verschillen in methaanproduktiecapaciteit en bovengrondse biomassa van zeggen (*Carex* spp.). Zeggen en andere planten kunnen de methaanemissie op tweeërlei wijze beïnvloeden, namelijk door methaan van de anaërobe lagen in de bodem naar het oppervlak te transporteren en door te fungeren als substraat voor methaanproducerende micro-organismen (hoofdstuk 3).

Om het inzicht in methaanemissie te vergroten, is de bodem van natte veengronden in verschillende fracties gescheiden op basis van verschillen in deeltjesgrootte en -dichtheid. Incubatie-experimenten toonden aan dat de afzonderlijke fracties zowel een vergelijkbare koolstofmineralisatiecapaciteit als C/N verhouding hadden. Er traden echter wel verschillen in methaanproduktiecapaciteit tussen de fracties op. Aanzienlijke methaanproduktie trad alleen op bij fracties met grote bodemdeeltjes. Verder nam de methaanproduktiecapaciteit sterk af met de diepte. Dit wijst erop dat in deze natte veengronden recent gestorven plantenmateriaal een belangrijk substraat voor methaanproducerende micro-organismen is (hoofdstuk 4).

Intensief beheerde graslanden met een lage en een hoge stikstofaanvoer op ontwaterde veengronden

Veengronden zijn in staat om veel methaan te produceren, omdat zij in het algemeen op geringe diepte anaëroob zijn en een hoog organische stofgehalte bezitten. In Nederland is het merendeel van de veengronden echter ontwaterd. Aan het begin van deze studie was het niet duidelijk of ontwaterde veengronden een bron of een sink van methaan zouden zijn. Op ontwaterde veengraslanden van de proefboerderij Zegveld met een gemiddelde grondwaterstand van 20-40 cm onder het maaioppervlak en een jaarlijkse stikstofaanvoer via bemesting en atmosferische stikstofdepositie van 35-460 kg N ha^{-1} jr^{-1} is de netto methaanemissie gemeten. Deze graslanden waren een kleine sink van methaan; ze consumeerden 0,1 tot 0,3 kg CH_4 ha^{-1} jr^{-1}. De invloed van de grondwaterstand (bij gemiddelde grondwaterstanden van 20 tot 40 cm onder het maaioppervlak) was significant, maar gering. Beweiding versus maaien en het achterwege laten van stikstofbemesting gedurende drie jaar had geen significant effect op de netto methaanemissie (hoofdstuk 5).

Op ontwaterde veengronden in Fallköping, Zweden, is een veldcampagne uitgevoerd om de ruimtelijke variabiliteit en de ruimtelijke afhankelijkheid van zowel broeikasgasemissies (methaan, lachgas en kooldioxide), als hun onderliggende bodemprocessen te bepalen. Metingen zijn verricht op twee graslanden op twee aaneengesloten dagen per grasland. De ruimtelijke variabiliteit is geanalyseerd met behulp van geostatistiek. Beide graslanden bleken kleine sinks van methaan te zijn. De ruimtelijke variabiliteit van de emissies was groot met variatiecoëfficiënten van 50 tot 1400%. De emissies vertoonden ofwel een ruimtelijke trend of ze waren ruimtelijk afhankelijk. Deze ruimtelijke afhankelijkheid was echter verschillend voor de verschillende graslanden en op de verschillende dagen. Dit betekent dat het niet mogelijk is om met behulp van geostatistiek emissies realistisch te schatten (hoofdstuk 6).

Extensief beheerde graslanden met een lage stikstofaanvoer op relatief droge zandgrond

Om de maximale methaanopname door bodems in Nederland te bepalen, zijn metingen verricht op extensief beheerde heidegraslanden op zandgrond in Wolfheze met een grondwaterstand dieper dan drie meter onder het maaioppervlak en een jaarlijkse stikstofaanvoer via depositie van 40 kg N ha^{-1} jr^{-1}. Deze graslanden consumeerden 1,1 kg CH_4 ha^{-1} jr^{-1}. De temporele variabiliteit van methaanopname in Wolfheze was gerelateerd aan verschillen in temperatuur en vochtgehalte van de bodem. De methaanopname was het hoogst bij hoge bodemtemperaturen en gemiddelde vochtgehaltes. Incubatie-experimenten lieten zien dat bij lage vochtgehaltes geen methaanconsumptie optrad, waarschijnlijk door fysiologische waterstress van de methaanconsumerende micro-organismen. Bij hoge vochtgehaltes was de methaanconsumptie aanzienlijk verminderd, waarschijnlijk doordat diffusie van methaan en zuurstof in de bodem werd bemoeilijkt. De voor consumptie optimale vochtgehaltes waren gelijk aan de meest voorkomende vochtgehaltes in het veld (hoofdstuk 7).

Factoren die de netto methaanemissie uit grasland bepalen

Belangrijke bodem- en klimaatfactoren, van invloed op de netto methaanemissie, zijn het organische stofgehalte van de bodem, de grondwaterstand, het vochtgehalte van de bodem, de temperatuur en de vegetatie. De grondwaterstand oefent een primaire invloed uit op methaanemissie, omdat de grondwaterstand de grens aangeeft tussen anaërobe (potentieel methaanproducerende) en aërobe (potentieel methaanconsumerende) lagen in de bodem. Dit betekent dat drainage van natte graslanden grote invloed heeft op methaanemissies. Het netto effect van stikstofbemesting is klein of verwaarloosbaar bij de huidige stikstofaanvoer via bemesting en atmosferische stikstofdepositie in Nederland. Verder zijn de effecten van beweiding versus maaien en veedichtheid verwaarloosbaar (hoofdstuk 8). Dit betekent dat graslandbeheer, met uitzondering van drainage, geen mogelijkheid biedt om de netto methaanemissie uit grasland in Nederland te verminderen.

Kwantificering van de netto methaanemissie uit grasland in Nederland

De meeste graslanden in Nederland worden intensief beheerd en hebben een totale stikstofaanvoer van 300-500 kg N ha^{-1} jr^{-1} via bemesting en atmosferische stikstofdepositie. Voordat deze studie werd uitgevoerd, was er niet veel bekend over de netto methaanemissie uit graslanden in Nederland. Het resultaat van deze studie is een kwantificering van de methaanemissies uit verschillende graslanden met verschillende vochttoestanden en stikstofaanvoer. De totale netto methaanopname door graslanden in Nederland (met uitzondering van natte graslanden) is geschat op 0,5 kton CH_4 jr^{-1}. Natte bodems, die slechts 0,5% van het totale oppervlak innemen, stoten naar schatting 5-10 kton CH_4 jr^{-1} uit. De huidige schattingen van methaanemissie in Nederland zullen aangepast moeten worden om de rol van graslanden in de nationale methaanbalans recht te doen.

CURRICULUM VITAE

Agnes van den Pol-van Dasselaar werd op 25 november 1968 geboren te Nijkerk. In 1987 behaalde zij op het Farel College te Amersfoort haar VWO-diploma. Aansluitend ging zij studeren aan de Landbouwuniversiteit Wageningen in de richting Landbouwplantenteelt met als specialisatie graslandkunde. Haar afstudeervakken waren graslandkunde en theoretische productie-ecologie. Zij liep stage bij de Grasslands Division of the Department of Scientific and Industrial Research in Palmerston North, Nieuw Zeeland. In januari 1993 studeerde zij af met lof. Van december 1992 tot juli 1993 werkte zij bij het toenmalige Nederlands Meststoffen Instituut (NMI) aan een studie naar het energieverbruik in de Nederlandse landbouw. Vervolgens werkte zij vier jaar (1993-1997) als toegevoegd onderzoeker bij de vakgroep Bodemkunde en plantenvoeding aan de Landbouwuniversiteit Wageningen. Hier deed zij onderzoek naar methaanemissies uit graslanden, hetgeen resulteerde in dit proefschrift. Momenteel werkt zij bij het Praktijkonderzoek Rundvee, Schapen en Paarden (PR) te Lelystad op de afdeling Weidebouw aan onderzoek op het gebied van gewasbescherming en bemesting van grasland en voedergewassen. Sinds juli 1998 is zij aangesteld als projectleider/onderzoeker op het gebied van maïs.