-Flushing meadows-

The influence of management alternatives on the greenhouse gas balance of fen meadow areas

A.P. Schrier-Uijl

Thesis Committee

Thesis supervisor:
Prof. dr. F. Berendse
Professor of Nature Conservation and Plant Ecology
Wageningen University

Thesis co-supervisors:
Dr. E.M. Veenendaal
Associate professor at the Nature Conservation and Plant Ecology Group
Wageningen University

Dr. ir. P.A. Leffelaar
Associate professor at the Plant Production Systems Group
Wageningen University

Other members:
Prof. dr. B. Holtslag, Wageningen University
Prof. dr. J.T.A. Verhoeven, Utrecht University
Prof. dr. J. Roelofs, Radboud University, Nijmegen
Prof. dr. ir. O. Oenema, Wageningen University

This research was funded by the Dutch research program 'Climate changes Spatial Planning', the province of North Holland and the European research program CarboEurope-IP. This research was conducted under the auspices of the C.T. de Wit Graduate School for Production Ecology and Resource Conservation (PE&RC).
ISBN 978-90-8585-718-1

-Flushing meadows-

The influence of management alternatives on the greenhouse gas balance of fen meadow areas

By

Adriana Pia Uijl
Born in Tholen
Netherlands

ACADEMIC THESIS
Submitted in partial fulfillment of the requirements for the degree of doctor
At Wageningen University
By the authority of the Rector Magnificus
Prof. dr. M.J.Kropff
In the presence of the
Thesis Committee appointed by the Doctorate Board
To be defended in public
On Friday 10 September 2010
at 4 p.m. in the Aula

A.P. Schrier-Uijl
Flushing meadows - The influence of management alternatives on the greenhouse gas balance of fen meadow areas, 198 pages.

Thesis Wageningen University, the Netherlands (2010)
With references, with summaries in Dutch and English

ISBN 978-90-8585-718-1

Acknowledgements

One of the commonest misconceptions about research is that it is an ‘ivory tower’ activity, far removed from reality and from social contact with others. But I have to say that my PhD work was FAR from working in an ‘ivory tower’. Fieldwork, laboratory work, work in the greenhouse, teaching, supervising, writing, publishing, taking postdoctoral courses, going to international conferences and workshops were all part of the game in the past five years. Although…I must say that especially in my writing/publishing period (September 2009 – May 2010) I was mainly working in the office on my own. Looking back on the whole journey, this last period was the toughest (I missed the field work and the contacts with other people!).

When I started as a PhD student I had no idea what to expect for the road ahead. During the ‘journey’ I realised that I would finally complete the project through God Who gave me the strength to complete this thesis and through all those who helped me along the way…

First, I wish to thank my colleagues, who were always available for chats, serious talks, discussion and advice: Elmar Veenendaal, Petra Kroon, Peter Leffelaar, Linda Nol, Annelies Veraart, Dimmie Hendriks, Petra Stolk, Ko van Huissteden and Arjan Hensen working at various groups (KNMI, ,ECN, VU, WUR and Alterra). I would especially like to thank Petra, Dimmie and Linda for the fun we had, the dinners we had and for being fantastic company on many of the trips to international conferences.

Someone who brought me down to earth when I was ‘flying’ (by singing: ‘Hi ho, hi ho, je krijgt het niet kado’) and who supported me in periods of doubt (‘we are laughing all the way to the finish!’), was Elmar my daily supervisor. As my co-supervisor, Elmar had the qualities to let me make mistakes and discover new possibilities, and to support me in new ideas. I realise that for him and also for my second co-supervisor Peter Leffelaar and thesis supervisor Frank Berendse, it was probably a hard job guiding me as I was sometimes an unguided, headstrong missile (as they had already been warned even before I had my intake interview ☺). I wish to thank you all very much for being my (co)-promotors. Although we had sometimes different opinions, Frank, I would like to thank you for your critical notes, your ideas and your ability to slow me down sometimes if needed.

Next, I would like to thank the following indispensable people that helped me out in the field under sometimes harsh conditions: the technicians Jan and Frans. You remember the

sampling on the lakes and still feeling the waves at night? Or being in the field for the entire day in freezing weather? Thanks for all the pea soup when it was cold and for saving me when I broke through the ice of the ditch with instruments worth 40,000 euros. And thanks for knocking on my window every day until I came for coffee! In the field a received also support from the van Eijk en van Eeuwijk families in Reeuwijk and from Nico de Bruin at Staatsbosbeheer, who kindly let me have access to sites en provided important information. Petra and Gerda cared for important aspects of my office life.

I would also like to thank the PhD students in our department: Veronique, Daan, William, Marije, Marjolein, Ake, Flavia, Hanneke and Marleen. Thank you so much for the nice movie-watch moments, dinners, discussions, coffee breaks, and some of you for being my models and later my clients for hair cutting. I am not yet sure, but if I do make the career switch from scientist to hairdresser, the pimped-up Volkswagen old-timer bus will show up next to the Chinese food-bus at the WUR campus ☺.

I realize that, as a PhD, I had a 'double-life'. Being in Wageningen I was part of the scientific network, project member, teacher, supervisor. Being in Ede I was a wife, mum, part of social life, church member, sport club member, housekeeper (although sharing this last mentioned job with Helma). The switch from one life to the other happened in the 15-minute journey from home to the University or back. I experienced this 'double-life' as a great advantage and I wouldn't have done so well without it!

To start, I would like to thank my former University study mates, friends and partly also photographers club: Eke, Nynke, Evert (and of course Katrijn), Fenny and Marieke. I will never forget our soil science excursions (Spain, France, Germany, Belgium), our camps, dinners, photography trips and chats.

I would also like to thank my church friends: the van Hijum, Toonstra, Meijer, Vervoorn, Boer, Pennings, Groenleer, Scharrenburg and Negeri families for all the get-out-of-the-house events we had together: the camping, the Sunday afternoon trips, the ice-skating, the committee work and the numerous parties we had. I would like to thank my friends the Lammers and Roeleveld families for the exclusive self-made dinners and for the trips we made together.

I would like to thank my entire family right through from my mum, my brother, my sisters to my family in law. I wish dad would have been here, he'd been so proud. I'd like to thank the dearest people in my life: Elbert, Sharon and Quinn. Elbert, thank you so much for being patient with me and for your support by being so flexible in your working hours! Although Quinn and Sharon sometimes felt very disappointed in times I had to work when instead they wanted me to go with them to buy ice cream at Bernardo's (which they actually want every day) or go with them to the swimming pool, they always truly believed

in me. All three supported me by taking me out of my work-life right into the wife-and-mum-life, thank you for doing so!!. I remember that when the teacher at school asked Quinn what kind of work I did, he saying that his mum worked in the field in between the cows, and the rest of the time played computer games at the University…and he was partly right?

Concluding my acknowledgements: thank you all so much for your support and let's keep on meeting in the future!

Table of Contents

Chapter 1

General introduction

1.1. Background

It has been recognised for at least a decade that humans have become a global geophysical force in nature. For example, human activities such as agriculture, industry and biomass burning have raised the concentrations of the greenhouse gases carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) by 70% between 1970 and 2004 (IPCC 2007) and therefore have influenced global climate change. Under the Kyoto Protocol, which was an important first step in combating human affected climate change, most developed countries have to reduce their greenhouse gas emissions; the Netherlands is committed to reduce its annual greenhouse gas (GHG) emissions by an average of 6% below the level of 1990 between 2008 and 2012. The Kyoto commitments have encouraged countries to investigate the possibilities for reducing emissions in various sectors.

In the Netherlands, by far the largest contributors to the total CO_2 emissions are the industry and transport sectors, while the highest CH_4 and N_2O emissions are due to agriculture (Maas et al., 2008). Emissions originating from agriculture account for a considerable part of the national GHG emission, which is why it is important to quantify these fluxes and to understand the mechanisms that control them in order to find ways of reducing these emissions. Contrary to other emissions, such as those from industry, the terrestrial emissions depend on many uncertain variables, such as climate, management and water table. Because the influence of these variables on emissions and underlying processes is largely unknown, estimates of GHG emission are currently mostly based on global trends which is uncertain and often not applicable to specific areas.

Among the specific areas which play an important role in the terrestrial carbon cycle, but which are not well understood in terms of emissions, are temperate peat areas. These peatland systems are vulnerable to changes and over the last centuries, anthropogenic pressure for land in Europe has resulted in the reclamation of large areas of peat swamps to make them suitable for agriculture or other land uses. The required drainage to reclaim peatlands results in subsidence, degradation of peat soils and release of carbon to the atmosphere. In the Netherlands peat areas make up almost 7% of the land, but this is decreasing as a result of the transformation of organic soils into mineral soils due to peat oxidation. Typical peat subsidence rates in Europe range from a few millimetres to as much as 3 cm/year depending on drainage and climatic (e.g. temperature) conditions (Kasimir-Klemedtsson et al., 1997). Estimates of emissions from peat areas including CO_2, CH_4 and N_2O are scarce and emissions of water bodies have never been included in regional greenhouse gas budgets. At present, approximately 294 000 ha of the Netherlands consists of peat land, defined as soils with an organic matter content of more than 50 % in the upper 80 cm (Hendriks, 1991). The peat areas in the Netherlands are situated in the west (Groene Hart) and east (provinces of Friesland and Overijssel) of the country. These landscapes typically consist of land, drainage ditches and large shallow fresh water bodies. Most of the

peat soils have been drained and are used as fen meadow (grassland). As a result of the lowering of the water table in the past, Dutch fen meadow ecosystems have for long been a strong net source of CO_2 because of the increased peat oxidation (Langeveld et al., 1997). High oxidation rates cause subsidence of soils and it has been estimated that in the past 30-40 years 20% of our peat soils have transformed into a mineral soil (Kempen et al., 2009). Yet reliable estimates of emissions of CO_2 and especially of CH_4 and N_2O are still lacking. Burgerhart (2001) suggests that peat areas could turn into sinks of CO_2 if water levels were raised.

The research described in this thesis was part of the larger BSIK-KvR ME-1 project and of the CarboEurope-IP programme (www.klimaatvoorruimte.nl and www.carboeurope.org). The BSIK-KvR ME-1 project focussed on integrated observations and modelling of greenhouse gas budgets at the ecosystem level in the Netherlands, whereas the CarboEurope programme focussed on understanding and quantifying the present terrestrial carbon balances of Europe and the associated uncertainties at local, regional and continental scales. All the research was funded by the Province of North Holland, BSIK and CarboEurope and was conducted in collaboration with research groups at the Vrije Universiteit, Energy research Centre of the Netherlands and Wageningen University.

1.2. Research objectives

Regional greenhouse gas estimates are uncertain and the processes that control the greenhouse gas budgets are poorly understood. A major question is: does the restoration of peat areas convert them from greenhouse gas sources into greenhouse gas sinks? In addition, the spatial and temporal variability of emissions is high, and therefore there is a need for suitable methods to measure fluxes and to up-scale them to the landscape scale.

My objectives were:

- to quantify the emission of greenhouse gases (CO_2, CH_4, N_2O) that are produced at small-scale (1mx1m), plot scale (ha scale) and landscape scale (whole polder scale) in Dutch fen meadows;
- to determine the spatial and temporal variability of fluxes at different scales;
- to develop a system for upscaling GHG emission from a small-scale to landscape scale and to provide predictions based on regression models relating emissions with measured explanatory variables;
- to answer the question: can restoration of fen meadows turn these areas into sinks when the greenhouse gases CO_2, CH_4 and N_2O are included in the emission balance?

1.3. Contents of thesis

This thesis comprises 8 chapters including the introduction (Chapter 1). In Chapter 2 an overview is given of the scientific background of this research and of past research. Chapter 3 discusses the methane budgets of intensively managed and extensively managed peatlands on the basis of chamber measurements, and emissions are compared with the emissions of an unmanaged peatland. Chapter 4 deals with the reliability of different measurement methods and the comparison of small-scale flux measurements and landscape-scale flux measurements. Methods are cross-validated by comparing spatially up-scaled small-scale chamber measurements with integrated landscape-scale eddy covariance measurements. Chapter 5 focuses on methane and nitrous oxide emissions from an intensively managed area, estimated from continuous eddy covariance measurements. In Chapter 6, greenhouse gas emissions from lakes and drainage ditches in peat areas are discussed in detail. Chapter 7 comprises an integrated general synthesis of the research performed within the ME 1 project. It gives the full greenhouse gas balances of three peat areas and includes a discussion of the influence of restoration on the greenhouse gas balance. Additionally, future land use change scenarios are discussed and management options to reduce emissions are suggested. Chapter 8 summarizes all the research presented in this thesis and based on the insights gained in this research conclusions per chapter are given and mitigation options are discussed.

1.4. References

Burgerhart, N. (2001) in Dutch: Mogelijkheden voor koolstofopslag in Nederlandse ecosystemen. Dutch National Inventory Report for LULUCF sector, category 5D1.

Hendriks, R.F.A. (1991) in Dutch: afbraak en mineralisatie van veen. Sc-DLO, Wageningen, Rapport 199.

IPCC 2007. Climate change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Kasimir-Klemedtsson, A., Klemedtsson, L., Berglund, K., Martikainen, P., Silvola, J., Oenema, O. (1997). Greenhouse gas emissions from farmed organic soils: a review. Soil Use and Management, 13 pp. 245-250.

Kempen, B., Brus, D.J., Heuvelink, G.B.M., Stoorvogel, J.J. (2009): Updating the 1:50,000 Dutch soil map using legacy soil data: A multinomial logistic regression approach. Geoderma 151, pp. 311-326.

Langeveld, C.A., Segers, R., Dirks, B.O.M., vandenPolvanDasselaar, A., Velthof, G.L., Hensen, A (1997): Emissions of CO_2, CH_4 and N_2O from pasture on drained peat soils in the Netherlands, European journal of agronomy, Vol 7, Issue 1-3, 35-42.

Maas, C.W.M., Coenen, P.W.H.G., Ruyssenaars, P.G., Vreuls, H.H.J., Brandes, L.J., Baas, K., Van den Berghe, G., Van den Born, G. J., et al., 2008. Greenhouse gas emissions in the Netherlands 1990-2006, National inventory report 2008. MNP report 500080009, Netherlands Environmental Assessment Agency (MNP).

Chapter 2

Background

2.1. Humans, climate change and peatlands

The human imprint on the Earth's ecosystems has important effects on the Earth's climate, and anthropogenic changes to ecosystems are expected to continue at a fast pace in the near future. The enormous growth of the world's population and industrialisation have led to rapid increases in the use of fossil fuel, biomass burning, agricultural activities and land use changes, resulting in enhanced emissions of aerosols and greenhouse gases into the atmosphere. Changes in the biogeochemical cycles of terrestrial ecosystems, such as the carbon and nitrogen cycles and their influence on the dynamics of the atmosphere, influence the climate in terms of temperature and precipitation. In Melbourne, Australia, for example, people had few concerns about water supply until a decade ago. But after ten years of drought, probably an impact of climate change, Melbourne is now facing a severe water shortage. And in arid and semi-arid African countries, the changing climate and increased human activities have led to land degradation and desertification (IPCC 2007). Another example is that of the Amazon region, where observations suggest that the soil surface temperature has increased, the wet season is beginning at a later date and droughts have become more severe over the last few decades. Global climate models suggest that temperature increase in the Arctic will be considerably stronger than in the rest of the world; this warming can result in permafrost thawing and ice melting. The main projected biophysical effects in polar regions are reductions in thickness and extent of glaciers, ice sheets and sea ice, and changes in natural ecosystems, with detrimental effects on many organisms – including migratory birds, mammals and higher predators. Besides the anthropogenic influences on the Earth's climate, other major factors that also probably influence the climate on larger temporal scales are varying solar activity, volcanic activity, and changes in the ocean currents and atmospheric circulation.

Greenhouse gases play a role in preventing heat from escaping from the Earth's surface and thus changes in greenhouse gas concentrations in the atmosphere will have a strong impact on climate; without greenhouse gases, scientists estimate that the average temperature on Earth would be approximately 30 degrees Celsius cooler. But if the concentrations increase, so does absorption of thermal radiation, followed by rising tropospheric and soil surface temperatures. The key greenhouse gases of concern are carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) (IPCC 2007). The global warming potentials (GWP) of these gases over a 100-year time horizon are 1 for CO_2, 25 for CH_4 and 298 for N_2O (Forster et al., 2007). The GWP takes account of the atmospheric life-times of the different gases. Anthropogenic sources of CO_2 include: the combustion of fossil fuels to generate energy; industry; and land use changes that accelerate organic matter oxidation (IPCC 2007). The main sources of CH_4 include the burning of fossil fuels and agricultural practices, whereas the main source of N_2O is agriculture (IPCC 2007) (see *Fig 2.1* for an overview of greenhouse gas sources).

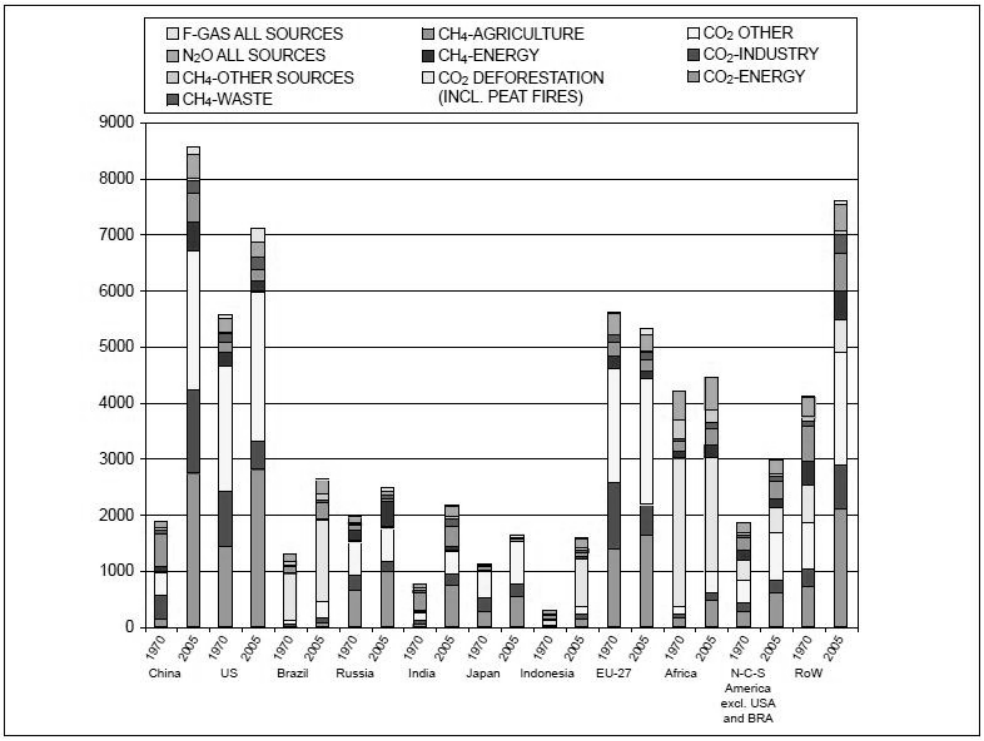


Fig 2.1 *Greenhouse gas emissions (Mton CO_2-eq) in 1970 and 2005 for eleven regions in the world (http://edgar.jrc.ec.europe.eu/, 2009).*

Natural sources also add to the total emission. A comparison of observed global-scale changes in surface temperature with results simulated by climate models using natural or natural plus anthropogenic forcing is shown in *Fig 2.2*.

Peatlands contribute considerably to the global greenhouse gas budget (Blodau and Moore, 2003). These ecosystems occupy 6% of the world's surface and function as CO_2 sinks and CH_4 sources if not drained. Human activity such as drainage has turned these areas into CO_2 sources. However, quantitative estimates of CO_2 emissions and of total greenhouse gas balances (including CH_4 and N_2O) are uncertain, while aquatic ecosystems such as freshwater ditches and lakes in peat landscapes have generally been neglected. But given the wide interest in restoring peatlands, it is essential that management prescriptions are based on proper quantification of fluxes and understanding of processes.

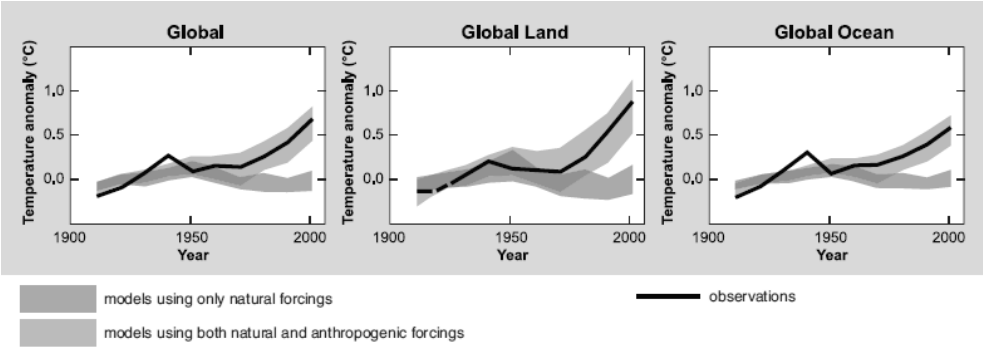


Fig 2.2. *Global changes in surface temperature derived from climate models and observations. Dark shaded bands show the 5 to 95% range for 19 simulations from five climate models using only the natural forcings due to solar activity and volcanoes. Light shaded bands show the 5 to 95% range for 58 simulations from 14 climate models using both natural and anthropogenic forcing (IPCC 2007).*

2.2. History of Dutch peatlands

The peat areas in the Netherlands have developed from large peat bogs created from peat accumulating after the last ice age when the sea level in the North Sea rose rapidly (Zagwijn, 1991). When the rate of sea level rise declined, a system of barrier beaches and islands developed, situated approximately along the present western and northern coastlines. With a sufficient supply of sediment the tidal basins behind the barrier coasts filled in until the tidal inlets could be closed by wave action and deposition of sediments. In the former tidal basins, marsh vegetation developed which gradually changed into ombrotrophic raised peat bogs, which receive all of their water and nutrients from precipitation, rather than from streams or springs. At larger tidal amplitudes and/or in areas with a smaller supply of sediment, the coastline did not close and estuarine conditions persisted. Here, extensive areas of saline and brackish marshes developed, merging into a belt of freshwater marshes and peat bogs at the landward side in the west and north-east of the country. In the same period, ombrothophic bogs also developed in depressions inland.

Later, the North Sea started to erode the coastal area again, which caused increased drainage of the coastal moors. Over large areas, peat stopped accumulating. During the Roman occupation, the human settlements were established on dune ridges and other sandy outcrops in these areas. The new colonists cultivated the peat land and other wetland soils. In the 10th century reclamation and cultivation increased and to create conditions suitable for crop growth, drainage ditches were dug and the existing drainage canals were enlarged to dewater the large peat bogs (Edelman, 1974). The drainage ditches were usually dug at right-angles to the natural watercourses, thereby producing the typical rectangular patterns of fields and ditches (*Fig 2.3.*).

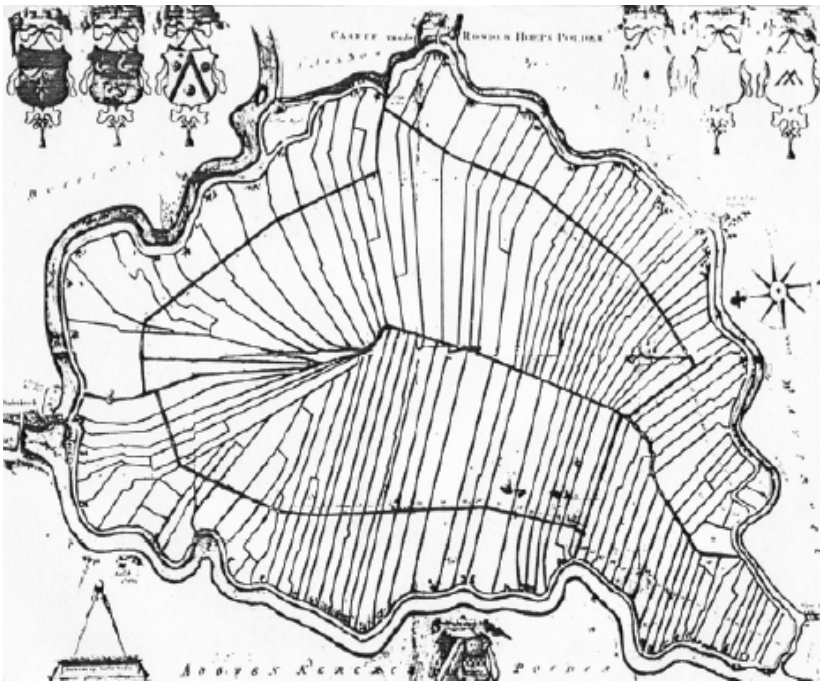


Fig 2.3. *Typical rectangular patterns of fields and ditches, polder Ronde Hoep.*

The consequence of this dewatering was that due to falling water tables the peat dried out and shrank; this, combined with peat removal, led to the soil subsidising (de Bont, 2009). The process of drainage continued and in addition the top layer of the peat was burned to provide extra nutrients for agriculture. Also, the extraction of peat for fuel started. This combination of drainage, peat cutting and burning led to the disappearance of the coastal ombrothophic peat ecosystems. In the late Middle Ages the land subsided rapidly, to below sea level, and during periods of high water, dikes broke, large areas were flooded and a large part of the peat was removed by the sea. Some areas became permanently inundated.

In the 16th century, the newly gained independence of the Netherlands stimulated the economic and technological development of the country. New inventions such as windmills were used to drain and reclaim many shallow freshwater and brackish lakes (van Veen, 1962), to make dairy farming possible, causing a further acceleration of the processes that induce land subsidence. However, not all the freshwater lakes are reclaimed and today a large part of the peat area still consists of marshland and water bodies.

Peat extraction for fuel became more and more important and peat was cut on a large scale, sometimes until the mineral subsoil was exposed. The peat cutting was carried out systematically, and the size and form of the drainage ditches and fields were agreed on

beforehand. The process peaked in the 19th century, but in some places continued until well into the 20th century (Stol et al., 1986). Currently, a large part of the peat area is mainly used for dairy farming. After World War II, dairy farming was modernised and the drainage of fields and application of fertiliser and manure were intensified, to boost yields. It has now been realised that such intensive management has consequences for the vulnerable peat marshes and that if deep drainage is continued in the future, the peat will oxidise and disappear as CO_2 into the atmosphere until no peat is left. There are therefore plans to restore peat areas in the Netherlands and to convert agricultural peatland back into less intensively managed peatland or wetland-nature for the purpose of ecological improvement, peat conservation and water storage (Van den Bos, 2003). This being the case, it is important to know whether the implementation of these plans will have consequences for carbon storage in the peatlands and for the total greenhouse gas balances of these areas.

2.3. Greenhouse gases and peatlands

CO_2 fluxes between the atmosphere and ecosystems are primarily controlled by the **photosynthesis**[*] and **respiration** of vegetation, animals and soil and by decomposition processes. The balance between the production and decomposition of organic compounds determines whether a system is a sink (uptake) or a source (release) of CO_2 (Valentini et al., 2000). Peatlands generally act as sources of CO_2 during the night (respiration only) and act as sinks for CO_2 during the day (when photosynthesis exceeds respiration). In wetland soils, the decomposition of organic material is slow because shallow water tables prevent O_2 from penetrating deeply into the soil. Consequently, the degradation of the peat is slow and net peat formation can take place (Alm, 1997). Areas of peat are commonly seen as overall sinks for CO_2. When the peat areas are drained, however, the situation is different. The availability of O_2 results in much faster degradation rates, and also in addition the application of fertilizer and manure, liming and repeated ploughing influence carbon and nitrogen cycling (e.g. Pessi, 1996).

CH_4 (methane) is emitted to the atmosphere as a result of production, consumption and transport through the soil or through water. It is produced through **methanogenesis** under anaerobic conditions in soil, water bodies and stored manure, and by enteric fermentation. Methane production is a microbiological process, which can occur when organic matter is degraded anaerobically and when most alternative terminal electron acceptors (O_2, NO_3^-, Fe^{+3},and SO_4^{2-}) are depleted by the microbial community (Zehnder & Stumm, 1988). The factors that affect how much of the CH_4 produced is consumed by oxidation on its way through the oxygen-rich soil before it reaches the atmosphere include the residence time of CH_4 within a biologically active environment, the oxygen status of the transport route and

[*] processes in bold are explained *Table 2.1.*

the biological activity of that environment. Wetlands, including peatlands, are considered the largest single source of atmospheric CH_4 (Denman et al., 2007). Methane emission from wetlands shows large spatial and temporal variability. The main factors determining this variability are management, land use history, moisture conditions and environmental conditions such as temperature (Moore & Knowles, 1989; Bridgeham & Richardson, 1992; Roulet, 1993; Dise, 1993; Segers, 1998; Van den Pol – van Dasselaar et al., 1998ab; Arah & Stephen, 1998; Van de Pol – van Dasselaar et al., 1999; Bazhin, 2003).

N_2O is primarily emitted from agricultural and natural ecosystems as a by-product of **nitrification** and **denitrification** (e.g. Rogers and Whiteman, 1991). Natural wetlands with high water tables do not necessarily produce N_2O (Nykänen et al., 2002) but may consume small amounts of N_2O in denitrification, when atmospheric N_2O is reduced to N_2 (Schiller and Hastie, 1994; Regina et al., 1996). However, agricultural soils are significant sources of N_2O (Mosier, 1991; Kroeze et al., 1999), and direct N_2O emissions from agricultural soils are believed to contribute about one-third of total global emissions (Mosier et al., 1991). N_2O fluxes also have a high spatial and temporal variability, and are therefore difficult to predict (Denmead, 1979; Groffmann et al., 2000; Velthof et al., 1996).

Table 2.1. *Explanation of the processes given in bold in the text above.*

Process	Process description	Pathway
Photosynthesis	Conversion of CO_2 into sugars, using the energy from sunlight	$6CO_2 + 6H_2O \rightarrow C_6H_{12}O_6 + 6O_2$
Respiration	Breakdown of organic material into CO_2, using O_2	$6CO_2 + 6H_2O \leftarrow C_6H_{12}O_6 + 6O_2$
Methanogenesis	Form of anaerobic respiration: on the right, the best described pathway is shown	$CO_2 + 4H_2 \rightarrow CH_4 + 2H_2O$
Nitrification	Biological oxidation of ammonia (NH_4^+) into ammonium (NH_3) into nitrate (NO_3^-) by bacteria. This process has N_2O as a by-product.	$NH_3 + 2O_2 \rightarrow NO_3^- + H^+ + H_2O$
Denitrification	Conversion of NO_3^- under anaerobic conditions with N_2 and N_2O as by-products	$NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2$

Integrated greenhouse gas balances in wetlands are mainly dependent on land use (and landuse change) and management practices. Half of the peatland in Europe is used in some way or other, and often this use involves drainage (Joosten and Clarke, 2002). Drainage allows O_2 to penetrate deeper into the soils, causing respiration rates to increase, followed by release of CO_2 and thus subsidence of soils. Because methanogenesis is a form of anaerobic respiration, the emission of CH_4 will decrease when water tables fall. Nitrous oxide emissions, which are mainly dependent on the application of fertiliser and manure, will decrease if management becomes less intensive. Restoration of drained peatlands will cause a change in the total carbon balance and total greenhouse gas balance, but the estimates are uncertain and the underlying processes are largely unknown.

2.4. Previous research in peatlands

Peat areas make up almost 7% of the land area of the Netherlands and occur mainly in the west of the country. These peat areas have a typical landscape of narrow fields, surrounded by large ditches and lakes. *Fig 2.4.* shows some of the areas in which research has been done. Most of the fields in the peat areas are used for grass production and grazing and therefore in about 85% of the Dutch peatland the water table has been lowered.

Most of the previous research on greenhouse gas emission from soils in the Netherlands has been done since the early 1990s as part of the Dutch National Research Programme on Global Air Pollution and Climate Change (NOP). Within this programme, Hensen et al. (1999) measured CO_2 fluxes in grassland on clay-on-peat soils (Cabauw) and found a net emission of 0.005 – 1.2 kg CO_2 m^{-2} yr^{-1} for the period 1993 – 1995.

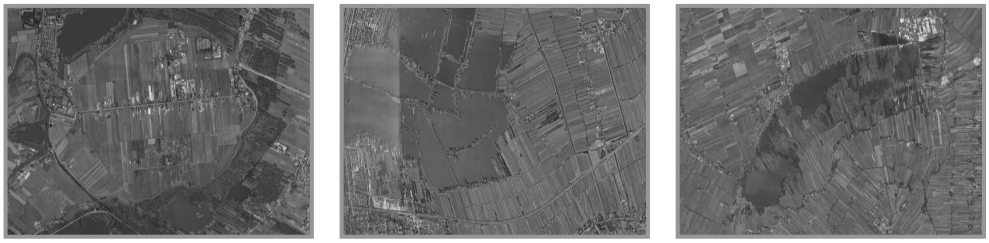


Fig 2.4. *Typical peat landscapes. The research areas shown are: left) Horstermeer polder (52.14°N,5.04°E), middle) Oukoop (52.03°N,4.78°E) and Stein (52.02°N,4.77°E) polders and lake Reeuwijk (52.03°N,4.75°E) and right) Lake Nieuwkoop (52.15°N,4.79°E).*

It was also found that the CO_2 fluxes from a site with a shallow water table (30 cm below field level) and from a site with a deep water table (60 cm below field level), both on peat soils, were not significantly different.

Night-time CO_2 emission (plant plus soil respiration) correlated positively with seasonal temperature and ranged from 0 to about 800 mg CO_2 m^{-2} hr^{-1} at Zegveld, and up to about 1500 mg CO_2 m^{-2} hr^{-1} at Cabauw. Veenendaal et al. (2007) studied CO_2 fluxes in two agricultural peat ecosystems, one intensively managed (Oukoop) and one extensively managed (Stein), and Hendriks et al. (2007) studied a nature reserve in a peat area. Over the year 2005 the extensively managed site was estimated to be a sink of -62.4 g CO_2-C m^{-2}, the intensively managed site was estimated to be a source of 157 g CO_2-C m^{-2} yr^{-1} and the natural site was estimated to be a sink of -311 g C-CO_2 m^{-2}. The researchers concluded that the differences in net ecosystem exchange (NEE) were the result of differences in management. Methane fluxes and N_2O fluxes were not studied at that time and therefore not included in the total greenhouse gas balance.

In 1992, the integrated CH_4 grassland project (described by Segers and Van Dasselaar, 1995) was set up. At the start of the project, peat soils were considered as a substantial source of CH_4, with an average emission of about 2.5 mg m^{-2} hr^{-1} contributing about 5 % to the Dutch CH_4 emissions (Van Amstel et al., 1993). However, it was soon discovered that CH_4 emissions from drained peat soils in Europe (mean water table 30 cm below field level) are low (<0.01 mg m^{-2} hr^{-1}) or even negative (Mosier et al., 1991; Van den Pol–van Dasselaar et al., 1998ab; Martikainen et al., 1992; 1993; Roulet et al., 1993; Glenn et al., 1993). In more natural fens, however, CH_4 emissions may be in the order of 1.7 mg CH_4 m^{-2} hr^{-1} yr^{-1} or 3 t CO_2 eq ha^{-1} yr^{-1}; these rates are similar to wet mire emissions (Van den Pol–Van Dasselaar et al. 1999). Other researchers have also found that in Indonesia raising the groundwater level in peat soils caused a significant increase of CH_4 emission: from 0.004 to 0.175 mg m^{-2} hr^{-1}. Lowering the water table from a depth of 20 cm to a depth of 30 cm in some areas led to the CH_4 emission decreasing by 25 % (Furukawa, 2005).

The effects of manure and chemical fertiliser on CH_4 from fen meadows are not yet well known. Van de Pol–van Dasselaar found that CH_4 emission from manure treatments were significantly higher than those from the mineral fertiliser treatments. She suggested that the higher emission from the manured sites were probably caused by the combination of wet soil, the application of easily decomposable organic material and anaerobic conditions in the manure itself (Van de Pol–van Dasselaar et al., 1999). In situ studies have shown that the incorporation of organic matter markedly increases CH_4 emission (Kanno et al., 1997 in Le Mer, 2001). Grazing and mowing may affect CH_4 uptake through differences in the annual supply of C and N to the soil; however, no clear effects of grazing or mowing have yet been found, except for greater spatial variability of emissions in grazed sites compared with sites that are only mown (Van Dasselaar and Lantinga, 1995). Few studies have been done on the CH_4 emissions from fen meadow systems, and the uncertainties in the spatial and temporal variability of the data presented in these systems are large (Smemo and Yavitt, 2006).

Previous studies on N_2O emissions suggest that 90% of all N_2O emission originates from biological processes in soils (e.g. Bouwman, 1990). The factors controlling N_2O emission are soil conditions such as soil moisture, soil temperature and the availability of ammonium and nitrate. Natural wetlands with high water tables do not necessarily produce N_2O (Nykänen et al., 2002) but may consume small amounts of N_2O in denitrification, when atmospheric N_2O is reduced to N_2 (Schiller and Hastie, 1994; Regina et al., 1996). However, drained agricultural wetlands are known to be N_2O emitters when grasslands are fertilised (Schothorst, 1977; Langeveld et al., 1997; Oenema et al., 1997). Velthof et al. (1996) reported a large difference between unfertilised and fertilised soils, with emissions of 0.05-1.29 g N m^{-2} yr^{-1} and 0.73-4.2 g N m^{-2} yr^{-1}, respectively. Flechard et al. (2005) measured N_2O fluxes from soil over three growing seasons in intensively and extensively

managed systems in Central Switzerland. Emissions appeared to be largest following the application of mineral (NH_4NO_3) fertiliser, but there were also substantial emissions following cattle slurry application, after grass cuts and during the thawing of frozen soil. Pihlatie et al. (2004) did an experiment at four different soil moisture conditions (40, 60, 80 and 100% WFPS) but similar N contents in the soil. The highest N_2O emission was measured at 100% WFPS. Others (e.g. Dobbie and Smith, 2001) also found an increase of N_2O emission with increasing WFPS. In addition, Berendse et al. (1994) found that denitrification rates did not differ between dry and wet peat soils from year to year, but did differ from season to season.

The three greenhouse gases of concern are CH_4, N_2O and CO_2. Because of the large spatial and temporal variability of the emission of these three gases it is difficult to predict larger-scale (e.g. site scale, landscape scale or regional scale) emission from up-scaled point measurements. The driving processes are not fully known and therefore model-based predictions are still uncertain. In order to obtain full greenhouse gas balances over larger areas, multiple year measurements of all three gases are needed, to cover the variability. There are three major challenges when upscaling fluxes from small-scale measurements to the landscape scale: 1) selecting the correct ecosystem variables, 2) developing robust predictive relationships (Groffman et al., 2000), and 3) using long-term datasets. Various measurement techniques have been developed in recent decades to provide accurate long-term datasets.

2.5. Methodology

The two main methods of measuring greenhouse gas emissions are (1) the static chamber method (e.g. van Huissteden et al., 2005) (*Fig 2.5.*) and (2) the eddy covariance method (e.g. Veenendaal et al., 2007) (*Fig 2.6.*). Combining these two ”field” methods and also coupling the measured emissions with driving variables is useful for upscaling emissions to landscape scale and for gaining insights into the details of the processes. In the last ten years, numerous synthesis papers have been published on greenhouse gas emissions measured by different methods: the emission estimates cover a broad range of ecosystems. In addition, detailed studies have been done on methodological aspects related to missing data or gap-filling and energy balance closure. However, there are large uncertainties in emission estimates, CH_4 and N_2O eddy covariance data are sparse because this method is still being developed, and there is a lack of comparisons of different measurement techniques.

To achieve the aim of the research described in this thesis, long-term combined data sets were needed and therefore we used both static chamber and eddy covariance methods to measure emissions from fields and water surfaces. For the chamber measurements on water

we used floating chambers, and on land we used rings inserted in the soil at fixed places, on which the chamber was placed. The two methods are described below.

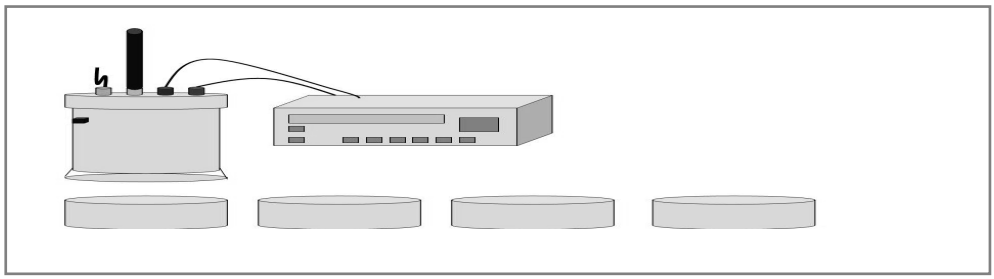


Fig 2.5. *The chamber measurement technique as used in the field experiments. Example of an enclosure for flux chamber measurements, which is connected to a gas analyser.*

Chamber-based methods are often used to determine source and sink distributions in non-uniform landscapes and are used to quantify small-scale spatial differences in CO_2, N_2O and CH_4 fluxes (e.g. Christensen et al., 1995; Hutchinson and Livingston, 2002; Hendriks et al., 2007). The enclosures function by restricting the volume of available air for exchange across the covered surface, so that any net emissions or immission of the enclosed gases can be measured as a concentration change (Livingston and Hutchinson, 1995). Usually, multiple gas concentration measurements are performed at fixed time intervals of e.g. 1 or 2 minutes, to calculate the flux. Chamber-based methods are highly accurate when used properly (Denmead, 2008) and are the most widely used approach for measuring fluxes of CO_2, CH_4 and N_2O from bare soil surfaces and surfaces with short vegetation. Because the spatial integration of measurements is complicated, the chamber-based method is not often used for large-scale estimates of greenhouse gas emission (Flechard et al., 2007). The method is often criticised because of the uncertainties caused by pressure artefacts, temperature effects (e.g. Hutchinson and Livingston, 2002; Rochette and Eriksen-Hamel, 2008), and temporal discontinuity of measurements. Measurements have to be done manually and therefore day-time measurements have usually been performed, and the possibility of diurnal variability has been neglected. Although most of the chamber effects have been eliminated from recent set-ups, the problem of neglecting the influence of wind remains (Denmead, 2008). The drawback of taking only day-time measurements also remains, unless automatic logging instruments are used during the night. In our research we used a Photo Acoustic Field Gas Monitor (INNOVA 1412 sn, LumaSenseTM Technologies, Ballerup, Denmark). The chamber system which was used in this research is compared to other chamber-based-systems and the reliability of the measurement systems was evaluated. For the comparability-test of the different chamber set-ups 12 simultaneous measurements were performed by the three groups within the project (ECN, VU and WUR) during a two day-measurement campaign with three different techniques: 1) a 'leaking' chamber set-up with a tunable diode gas analyzer 2) a closed chamber set-up with a tunable diode gas analyzer and 3) two closed chamber set-ups with an Innova Photo Acoustic Field

Gas Monitor (one of the set-ups is used in this research). The results showed that the observed CH_4 and Ecosystem respiration ($R_{eco)}$ were reliable and did not show significant differences between the closed-chamber-systems. The 'leaking' chamber-system underestimated the emissions (results not published).

Fig 2.6. *Eddy covariance measurements in the field. (licor 7500 Irga with a Campbell Scientific Csat Sonic anemometer and standard meteorological equipment; according To the CarboEurope protocol)*

Eddy covariance (EC) techniques have been used to continuously quantify landscape-scale temporal variability of CO_2 and, to a lesser extent, of CH_4 (e.g. Baldocchi et al., 2001; Aubinet et al., 2000; Veenendaal et al., 2007; Hendriks et al., 2007; Kroon et al., 2007). The EC method is based on measuring turbulent ascending and descending wind fields, temperature, and gas concentrations at high frequency at a certain measurement point (e.g. Baldocchi, 2003). The advantage of this method is that it does not disturb the soil/air environment, integrates over larger areas and has a continuous time coverage. An eddy covariance output flux represents the integrated net flux from the landscape upwind from the measurement point. The extent of the upwind area from which the flux originates, the so-called footprint area (in this research 10^2 to 10^4 m^2), depends on atmospheric stability and surface roughness (e.g. Grelle and Lindroth, 1996; Kormann and Meixner, 2001; Neftel et al., 2007). EC measurements are based on assumptions such as horizontal homogeneity, flat terrain and negligible mean vertical wind velocities over the averaging period. Uncertainties arise for reasons that include one-point sampling and the lack of low and high frequency responses (e.g. Moore, 1986; Aubinet et al., 2000; Kroon et al., 2007). In this research, data were logged with an open-path eddy covariance system for CO_2 (CR5000, Campbell Scientific, USA). The system for CH_4 consisted of a three-dimensional sonic anemometer (model R3, Gill Instruments, Lymington, UK) and a quantum cascade laser (QCL) spectrometer (model QCL-TILDAS-76, Aerodyne Research Inc., Billerica MA, USA).

2.6. Contents of the thesis

Peatlands cover 7% of the Netherlands and are considered as important greenhouse gas emitters. Currently, peat degradation in the Netherlands is still going on, mainly as a result of drainage for agriculture (*Fig 2.7.*). Continued drainage in the future will result in release of the peat as CO_2 in the atmosphere until no peat is left. There are therefore plans to restore peat areas in the Netherlands and to convert agricultural peatland back into less intensively managed peatland or wetland-nature for the purpose of ecological improvement, peat conservation and water storage (van den Bos, 2003). Burgerhart (2001) suggests that possibly intensively managed peat areas can be turned into CO_2 sinks if water levels are increased. They based their suggestions on a comparison between managed sites in the Netherlands versus natural ecosystems in more northern regions. Hence the need was felt for a large landscape scale experiment that involves alternative land-use scenarios and a comprehensive analysis of the major greenhouse gasses. This experiment was funded by Wageningen University, The Province of North Holland, CarboEurope IP and the Dutch National Research Programme Climate Changes (BSIK) and forms the basis of this thesis.

Fig 2.7*. An example of the subsidence of a peat soil in the western peat area of the Netherlands. The building is a remaining from the second world war.*

To determine the effect of different land-use options (management reduction and ground water level increase) on the greenhouse gas balance and the carbon balance, a landscape-scale experiment was started at the end of 2004. Three peat sites were included, each with a different management intensity and ground water level (*Fig 2.8.*). Two of the sites, Oukoop and Stein, were under the aegis of Wageningen University; the third, Horstermeer, was under the aegis of Amsterdam Vrije Universiteit (Hendriks et al., 2007).

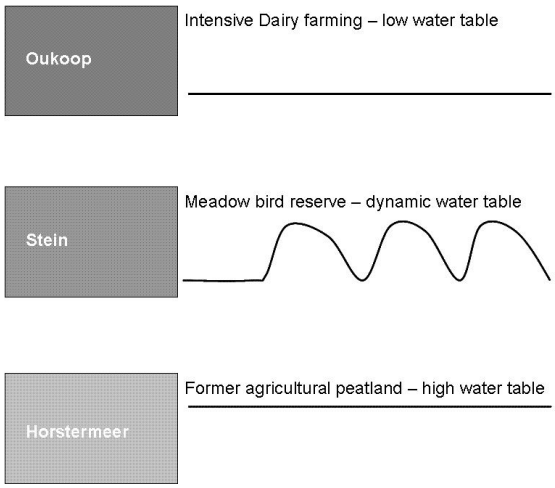


Fig 2.8. *Three experimental peat sites each with a different management intensity and ground water level*

Due to the high spatial and temporal variability of greenhouse gas emissions, reliable estimates of CO_2, CH_4 and N_2O are lacking and most large-scale estimates do not include emissions from aquatic environments (e.g. saturated land, ditches and lakes). However, it is relevant to include all possible emission sources and sinks and to reduce uncertainties in measurements. The reduction of uncertainty is of great importance for generating more reliable model predictions of the emission of greenhouse gases from fen meadows. We therefore set out to quantify greenhouse gases emissions (CO_2, CH_4, N_2O) at different scales in Dutch peatlands and to determine the spatial and temporal variability of fluxes. We aimed to develop a technique for upscaling greenhouse gas emission from small-scale to landscape scale. Therefore, we aimed to quantify robust predictive relationships on the basis of long-term (2005-2008) sets of measurements. . Emissions from water bodies were also included in this study. We ascertained CO_2, CH_4 and N_2O uptake and release from soils and water from direct flux measurements. Closed chambers (Alm et al., 1997) and micrometeorological systems (Moncrieff et al., 1998) were used at different temporal scales in heterogeneous landscapes of fields and water bodies.

The static chamber method was used to study small-scale spatial variability within landscapes and to up-scale fluxes from square metre scale to landscape scale. Fluxes from water bodies were measured by using a floating chamber connected to a gas analyser. Simultaneously with the flux measurements we also measured variables such as soil temperature, water temperature, soil moisture, water table, Electrical Conductivity and pH, to study the dependency of emissions on these variables. Each flux measurement consisted of five concentration measurements taken at one-minute intervals from which dC/dt was calculated where C is the concentration of CO_2, CH_4 or N_2O in the air of the measuring

chamber and *t* is the time. Regression models were developed to describe fluxes and to fill gaps in the dataset. Final emissions were calculated by surface area weighting factors for each contributing source or sink; uncertainties were based on temperature-dependent uncertainties of the parameters in the regression analyses.

The eddy covariance method was used for integrated, continuous measurements over larger areas. Temporal variability could be studied and the appropriateness for measuring CH_4 and CO_2 could be tested by comparing up-scaled chamber measurements with the EC measurements. Net ecosystem exchange of CO_2 was determined directly from the EC flux measurements; respiration was determined using night-time NEE values. Soil respiration is described as a function of soil variables and these functions were used for gap fill procedures. Methane and N_2O fluxes were determined directly from the EC flux measurements, and different gap-filling methods were compared.

The results of this study are published in several Scientific Journals. Chapter 3 is published in Plant & Soil (2010, doi 10.1007/s11104-009-0180-1), chapter 4 is published in Agricultural and Forest Meteorology (2010, doi 10.1016/j.agrformet.2009.11.007), chapter 5 is published in the European Journal of Soil Science (2010, vol 61), chapter 6 is published in Biogeochemistry (2010, doi 10.1007/s10533-010-9440-7) and chapter 7 will be submitted to Global Change Biology. Chapter 8 summarizes and discusses the results of the preceding chapters and presents conclusions about the implications of the results for peat area management.

2.7. References

Alm, J., Talanov A., Saarnio, S., Silvola, J., Ikkonen, E., Aaltonen, H., Nykanen, H. & Martikainen, P. J. (1997): Reconstruction of the carbon balance for microsites in a boreal oligotrophic pine fen, Finland. Oecologia 110: 423-431.

Aubinet, M., Grelle, A., Ibrom, A., Rannik, U., Moncrieff, J., Foken, T., Kowalski, A.S., Martin, P.H., Berbigier, P., Bernhofer, C., Clement, R., Elbers, J., Granier, A., Grunwald, T., Morgenstern, K., Pilegaard, K., Rebmann, C., Snijders, W., Valentini, R. and Vesala T., 2000. Estimates for the annual net carbon and water exchange of forests. The EUROFLUX methodology. Adv. in Ecol. Res.: 30, 113-175.

Arah, J. R. M. and Stephen, K. D. (1998): A model of the processes leading to methane emission from peatland. Atmospheric Environment 32: 3257-3264.

Baldocchi, D., Falge, E., Gu, L.H., et al., 2001. FLUXNET: A new tool to study the temporal and spatial variability of ecosystem-scale carbon dioxide, water vapor, and energy flux densities. Bulletin of the American Meteorological Society 82, 2415–2434.

Baldocchi, D., 2003. Assessing the eddy covariance technique for evaluating carbon dioxide exchange rates of ecosystems: past, present and future. Glob. change Biol.: 9, 479-492.

Bazhin, N. M. (2003): Theoretical consideration of methane emission from sediments. Chemosphere 50: 191-200.

Berendse, F., Oomes, M.J.M., Altena, H.J., Visser, W. (1994). A Comparative Study of Nitrogen Flows in Two Similar Meadows Affected by Different Groundwater Levels. Journ. Of Applied Ecol. 31, pp. 40-48.

Blodau, C. and Moore, T. R. (2003): Experimental response of peatland carbon dynamics to a water table fluctuation. Aquatic Sciences 65: 47-62.

Bont, C. de (2009): in Dutch: Vergeten land; ontginning, bewoning en waterbeheer in de westnederlandse veengebeiden (800-1350), Alterra Scientific Contributions, Wageningen UR.

Bouwman, A.F., (1990). In: Bouwman AF (Ed.): Soils and the Greenhouse effect (pp. 61-127) Exchange of greenhouse gases between terrestrial ecosystems and the atmosphere. John Wiley and Sons. Chichester, UK.

Bridgeham, S. D. and Richardson, C. J. (1992): Mechanisms Controlling Soil Respiration (CO_2 and CH_4) in Southern Peatlands. Soil Biology & Biochemistry 24: 1089-1099.

Burgerhart, N. (2001) in Dutch: Mogelijkheden voor koolstofopslag in Nederlandse ecosystemen. Dutch National Inventory Report for LULUCF sector, category 5D1.

Christensen, T.R., Jonasson, S., Callaghan, T.V., Havström, M., (1995). Spatial variation in high-latitude CH4 flux along a transect across Siberian and Eurasian tundra environments. J. of Geoph. Res.: 100, 21035-21045.

Denman, K. L., Brasseur, G., Chidthaisong, A., Ciais, P., Cox, P. M., Dickinson, R. E., Hauglustaine, D., Heinze, C., Holland, E., Jacob, D., Lohmann, U., Ramachandran, S., Da Silva Dias, P. L., Wofsy S. C., and Zhang, X.: Couplings between changes in the climate system and biogeochemistry in: Climate Change (2007): The Physical Science Basis. Contribution of working group I to the fourth assessment report of the Intergovernmental Panel on Climate Change, edited by: Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K. B., Tignor M., and Miller, H. L., Cambridge University Press, Cambridge, UK and New York, NY, USA, 2007.

Denmead, O. T. (1979): Chamber Systems for Measuring Nitrous-Oxide Emission from Soils in the Field. Soil Science Society of America Journal 43: 89-95.

Denmead, O. T., Freney J. R. & Simpson J. R. (1979): Nitrous-Oxide Emission During Denitrification in a Flooded Field. Soil Science Society of America Journal 43: 716-718.

Denmead, O.T. (2008): Approaches to measuring fluxes of methane and nitrous oxide between landscapes and the atmosphere, Plant and Soil, 309, pp 5-24.

Dise, N. B. (1993): Methane Emission from Minnesota Peatlands - Spatial and Seasonal Variability. Global Biogeochemical Cycles 7: 123-142.

Dobbie, K. E. and Smith, K. A. (1996): Comparison of CH4 oxidation rates in woodland, arable and set aside soils. Soil Biology & Biochemistry 28: 1357-1365.

Dobbie, K. E., Smith, K. A., Prieme, A., Christensen, S., Degorska, A. & Orlanski, P. (1996): Effect of land use on the rate of methane uptake by surface soils in northern Europe. Atmospheric Environment 30: 1005-1011.

Dobbie, K. E. and Smith K. A. (2001): The effects of temperature, water-filled pore space and land use on N_2O emissions from an imperfectly drained gleysol. European Journal of Soil Science 52: 667-673.

Edelman, T. (1974): Bijdrage tot de historische geografie van de Nederlandse kuststreek. Publicatie van de directie waterhuishouding en waterbeweging, nr. 14.

Flechard, C. R., Neftel, A., Jocher, M., Ammann, C. and Fuhrer, J. (2005): Bi-directional soil/atmosphere N_2O exchange over two mown grassland systems with contrasting management practices. Global Change Biology 11: 2114-2127.

Flechard, C.R., Ambus, P., Skiba, U., Rees, R.M., Hensen, A., van Amstel, A., van den Pol-van Dasselaar, A., et al., (2007). Effects of climate and management intensity on nitrous oxide enissions in grassland systems across Europe, Agric. Ecosyst. And Environm.: 121, 135-152.

Forster, P., Ramaswamy, V., Artaxo, P., Berntsen, T., Betts, R., Fahey, D.W., Haywood, J., Lean, J., Lowe, D.C., Myhre, G., Nganga, J., Prinn, R., Raga, G., M., S., Van Dorland, R., (2007): Changes in Atmospheric Constituents and in Radiative Forcing. In: S. Solomon et al. (Editors), Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, U.K.

Furukawa, Y., Inubushi, K., Ali, M., Itang, A.M., Tsuruta, H. (2005): Effect of changing groundwater levels caused by land-use changes on greenhouse gas fluxes from tropical peat lands, Nutrient Cycling in Agro ecosystems 71, 81-91.

Glenn S., Heyes A. & Moore T. (1993): Carbon-Dioxide and Methane Fluxes from Drained Peat Soils, Southern Quebec. Global Biogeochemical Cycles 7: 247-257.

Grelle A., Lindroth, A., 1996. Eddy-correlation systems for long-term monitoring of fluxes of heat, water vapour and CO2, Glob. Change Biol.: 2, 297-307.

Dise, N. B. (1993): Methane Emission from Minnesota Peatlands - Spatial and Seasonal Variability. Global Biogeochemical Cycles 7: 123-142.

Groffman, P.M., Brumme, R., Butterbach-Bahl, K., Dobbie, K.E., Mosier, A.K., Ojima, D., Papen, H., Parton, W.J., Smith, K.A. and Wagner-Riddle, C. (2000): Evaluating annual nitrous oxide emissions at the ecosystem scale, Global Biogeochemical Cycles, 14, 1061-1070.

Hendriks, D.M.D., van Huissteden, J., Dolman, A.J. and van der Molen, M.K. (2007): The full greenhouse gas balance of an abandoned peat meadow, Biogeosciences, 4, 411-424.

Hensen, A., Vermeulen, A.T., Wyers, G. P., and Zhang, Y. (1999): Eddy correlation and relaxed eddy accumulation measurements of CO_2 fluxes over grassland, Physics and Chemistry of The Earth, 21, pp 383-388.

Hensen, A., Groot, T.T., Van den Bulk, W.C.M., Vermeulen, A.T., Olesen, J.E., Schelde, K. (2006): Dairy farm CH_4 and N_2O emissions, from one square meter scale to the full farm scale, Agriculture Ecosystems and Environment, Vol. 112, Issues 2-3, 146-152.

Hutchinson, G.L., Livingston, G.P., 2002. Soil-atmosphere gas exchange. In : J.H. Dane and G.C. Topp, Editors, Methods of soil analysis. Part 4. Physical methods, SSSA Book Series, no. 5, Soil science Society of America, Madison, WI, 1159-1182.

IPCC 2007. Climate change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Joosten, H. and Clarke, D. (2002): Wise use of mires and peatlands – Background and principles including a framework for decision-making. International Mire Conservation Group / International Peat Society, pp 304.

Kanno, T., Miura, Y., Tsuruta, H. and Minami K. (1997): Methane emission from rice paddy fields in all of Japanese prefecture - Relationship between emission rates and soil characteristics, water treatment and organic matter application. Nutrient Cycling in Agroecosystems 49: 147-151.

Kempen, B., Brus, D.J., Heuvelink, G.B.M., Stoorvogel, J.J. (2009): Updating the 1:50,000 Dutch soil map using legacy soil data: A multinomial logistic regression approach. Geoderma 151, pp. 311-326.

Kroeze, C., Mosier, A. and Bouwman L. (1999): Closing the global N_2O budget: A retrospective analysis 1500-1994. Global Biogeochemical Cycles 13: 1-8.

Kroon, P.S., Hensen, A., Jonker, H.J.J., Zahniser, M.S., van 't Veen, W.H.,Vermeulen, A.T. (2007): Suitability of quantum cascade laser spectroscopy for CH_4 and N_2O eddy covariance flux measurements, BG, 4, 715-728.

Langeveld, C.A., Segers, R., Dirks, B.O.M., vandenPolvanDasselaar, A., Velthof, G.L., Hensen, A (1997): Emissions of CO_2, CH_4 and N_2O from pasture on drained peat soils in the Netherlands, European journal of agronomy, Vol 7, Issue 1-3, 35-42.

Le Mer, J. and Roger P. (2001): Production, oxidation, emission and consumption of methane by soils: A review. European Journal of Soil Biology 37: 25-50.

Livingston, G.P., Hutchinson, G.L. (1995): Enclosure-based measurement of trace gas exchange: applications and sources of error. In: Matson PA, Harriss RC, eds. Biogenic trace gases: measuring emissions from soil and water. Oxford: Blackwell Scientific Publications, 14–15.

Kormann, R., Meixner, F.X., 2001. An analytical footprint model for neutral stratification. Boudary-Layer Meteorology: 99, 207-224.

Maas, C.W.M., Coenen, P.W.H.G., Ruyssenaars, P.G., Vreuls, H.H.J., Brandes, L.J., Baas, K., Van den Berghe, G., Van den Born, G. J., et al., (2008): Greenhouse gas emissions in the Netherlands 1990-2006, National inventory report 2008. MNP report 500080009, Netherlands Environmental Assessment Agency (MNP).

Martikainen, P.J., H. Nykänen., P. Crill., J. Silvola,1992. The effect of changing water table on methane fluxes at two Finnish mire sites. Suo 43: 237-240.

Martikainen, P. J., Nykanen, H., Crill, P. and Silvola, J. (1993): Effect of a Lowered Water-Table on Nitrous-Oxide Fluxes from Northern Peatlands. Nature 366: 51-53.

Moncrieff, J. B., Beverland, I. J., O'Neill, D. H. & Cropley, F. D. (1998): Controls on trace gas exchange observed by a conditional sampling method. Atmospheric Environment 32: 3265-3274.

Mosier, A., Schimel, D., Valentine, D., Bronson, K. and Parton, W. (1991): Methane and Nitrous-Oxide Fluxes in Native, Fertilized and Cultivated Grasslands. Nature 350: 330-332.

Moore, T.R and R. Knowles. (1989): The influence of water table levels on methane and carbon dioxide emission from peatland soils. Can J. Soil Sci. 69:33-38.

Neftel, A., Flechard, C., Ammann, C., Conen, F, Emmenegger, L., Zeyer, K., 2007. Experimental assessment of N2O background fluxes in grassland systems. Tellus B: 59, 470-482.

Nykanen, H., Vasander, H., Huttunen, J. T. and Martikainen, P. J. (2002): Effect of experimental nitrogen load on methane and nitrous oxide fluxes on ombrotrophic boreal peatland. Plant and Soil 242: 147-155.

Oenema, O., Velthof, G. L., Yamulki, S. and Jarvis, S. C. (1997): Nitrous oxide emissions from grazed grassland. Soil Use and Management 13: 288-295.

Pihlatie, M., Syvasalo, E., Simojoki, A., Esala, M. and Regina K. (2004): Contribution of nitrification and denitrification to N2O production in peat, clay and loamy sand soils under different soil moisture conditions. Nutrient Cycling in Agroecosystems 70: 135-141.

Regina, K., Nykanen, H., Silvola, J. and Martikainen, P. J. (1996): Fluxes of nitrous oxide from boreal peatlands as affected by peatland type, water table level and nitrification capacity. Biogeochemistry 35: 401-418.

Rochette, P., Eriksen-Hamel, N.S., (2008): Chamber Measurements of Soil Nitrous Oxide Flux: Are Absolute Values Reliable? Soil Sci. Soc. Am. J.: 72, 331-342.

Roulet, N. T., Ash, R., Quinton, W. and Moore, T. (1993): Methane Flux from Drained Northern Peatlands - Effect of a Persistent Water-Table Lowering on Flux. Global Biogeochemical Cycles 7: 749-769.

Schiller, C. L. and Hastie, D. R. (1994): Exchange of Nitrous-Oxide within the Hudson-Bay Lowland. Journal of Geophysical Research-Atmospheres 99: 1573-1588.

Schothorst, C.J., (1977): Subsidence of low moor peat in the Western Netherlands, Geoderma, 17, 265-271.

Segers, R., (1998): Methane production and methane consumption; a review of processes underlying wetland methane fluxes, Biochemistry, 41 23-51.

Segers, R., Dasselaar, A. van (1995): The integrated CH_4 grassland project: aims, coherence and site description, In: Climate change research: evaluation and policy

implications, S. Zwerver et al. (eds.). Elsevier, Amsterdam, Studies in Environmental Science 64A (1995) 573-576.

Smemo, K.A., Yavitt, J.B. (2006): A multi-year perspective on methane cycling in a shallow peat fen in central New York State, USA, Wetlands 26(1) 20-29.

Stol, T., (1986): De turfwinningslandschappen. In: S. Barends, J. Renes, T. Stol, J.C. van Triest, R.J. de Vries & F.J. van Woudenberg, 1986. Het Nederlandse landschap: een historisch-geografische benadering. Stichting Matrijs, Utrecht: 47-56.

Valentini, R., Dore S., Marchi, G., Mollicone, D., Panfyorov, M., Rebmann, C., Kolle, O. and Schulze, E. D. (2000) Carbon and water exchanges of two contrasting central Siberia landscape types: regenerating forest and bog. Functional Ecology 14: 87-96.

Valentini, R., Matteucci, G., Dolman, A. J., Schulze, E. D., Rebmann, C., Moors, E. J., Granier, A., Gross, P., Jensen, N. O., Pilegaard, K., Lindroth, A., Grelle, A., Bernhofer, C., Grunwald, T., Aubinet, M., Ceulemans, R., Kowalski, A. S., Vesala, T., Rannik, U., Berbigier, P., Loustau, D., Guomundsson, J., Thorgeirsson, H., Ibrom, A., Morgenstern, K., Clement, R., Moncrieff, J., Montagnani, L., Minerbi, S. and Jarvis, P. G. (2000): Respiration as the main determinant of carbon balance in European forests. Nature 404: 861-865.

Van Amstel, A.R., Swart R.J., Krol M.S., Beck J.P., Bouwman A.F., and van der Hoek K.W., (1993): Methane, the other greenhouse gas, research and policy in the Netherlands, National Institute of Public Health and Environmental Protection (RIVM), the Netherlands.

Van den Bos, R. (2003): Restoration of former wetlands in the Netherlands; effect on the balance between CO_2 sink and CH_4 source, Netherlands Journal of Geosciences-Geologie En Mijnbouw, 82, 325-331.

Van Dasselaar, A. V. and Lantinga, E. A. (1995): Modeling the Carbon-Cycle of Grassland in the Netherlands under Various Management Strategies and Environmental-Conditions. Netherlands Journal of Agricultural Science 43: 183-194.

Van den Pol-van Dasselaar, A., Corre, W.J., Prieme, A., Klemedtsson, A.K., Weslien, P., Stein, A., Klemedtsson, L. and Oenema, O. (1998a): Spatial variability of methane, nitrous oxide, and carbon dioxide emissions from drained grasslands, Soil Science Society of America Journal, 62, 810-817.

Van den Pol-van Dasselaar, A., van Beusichem, M.L., and Oenema, O. (1998b): Effects of soil moisture content and temperature on methane uptake by grasslands on sandy soils, Plant and Soil, 204, 213-222.

Van den Pol-van Dasselaar A., Van Beusichem, M.L. and Oenema, O. (1999): Methane emissions from wet grasslands on peat soil in a nature preserve, Biogeochemistry, 44, 205-220.

Van Huissteden, J., T. C. Maximov, and A. J. Dolman, (2005): High methane flux from an artic floodplain (Indigirka lowlands, eastern Siberia). J. Geophys. Res.-Biogeosci., 110(G2), doi:10.1029/2005JG000010.

Van Veen, J., (1962): Dredge, drain, reclaim! The art of a nation. 5th edition. Martinus Nijhoff, The Hague, 200 pp.

Veenendaal, E.M., Kolle, O., Leffelaar, P.S. Schrier-Uijl, A.P., Van Huissteden, J., Van Walsem, J., Möller, F. & Berends, F., (2007): CO_2 exchange and the carbon balance in two grassland sites on eutrophic drained peat soils. Biogeosciences, 4, pp. 1027-1040.

Velthof, G. L., Oenema, O., Postma, R. and VanBeusichem, M. L. (1996): Effects of type and amount of applied nitrogen fertilizer on nitrous oxide fluxes from intensively managed grassland. Nutrient Cycling in Agroecosystems 46: 257-267.

Velthof, G.L. and Oenema, O., (1997): Nitrous oxide emission from dairy farming systems in the Netherlands. Netherlands Journal of Agricultural Science, 45, pp. 347-360.

Zagwijn, W.H. (1991): Nederland in het Holoceen. Geologie van Nederland, deel 1. Haarlem - 's Gravenhage.

Zehnder, A.J.B., and W. Stumm (1988), Geochemistry and biogeochemistry of anaerobic habitats, in Anaerobic Microbiology, edited by A.J.B. Zehnder, pp 1-38, Wiley, New York,.

Chapter 3

Methane emissions in two drained peat agro-ecosystems with high and low agricultural intensity

Methane (CH_4) emissions were compared for an intensively and extensively managed agricultural area on peat soils in the Netherlands to evaluate the effect of reduced management on the CH_4 balance. Chamber measurements (photoacoustic methods) for CH_4 were performed for a period of three years in the contributing landscape elements in the research sites. Various factors influencing CH_4 emissions were evaluated and temperature of water and soil was found to be the main driver in both sites. For upscaling of CH_4 fluxes to landscape scale, regression models were used which were specific for each of the contributing landforms. Ditches and bordering edges were emission hotspots and emitted together between 60% and 70% of the total terrestrial CH_4 emissions. Annual terrestrial CH_4 fluxes were estimated to be 203 (±48%), 162 (±60%) and 146 (±60%) kg CH_4 ha^{-1} and 157 (±63%), 180 (±54%) and 163 (±59%) kg CH_4 ha^{-1} in the intensively managed site and extensively managed site, for 2006, 2007 and 2008 respectively. About 70% of the CH_4 was emitted in the summer period. Farm based emissions caused per year an additional 257 kg CH_4 ha^{-1} and 172 kg CH_4 ha^{-1} for the intensively managed site and extensively managed site, respectively. To further evaluate the effect of agricultural activity on the CH_4 balance, the annual CH_4 fluxes of the two managed sites were also compared to the emissions of a natural peat site with no management and high ground water levels. By comparing the terrestrial and additional farm based emissions of the three sites, we finally concluded that transformation of intensively managed agricultural land to nature development will lead to an increase in terrestrial CH_4 emission, but will not by definition lead to a significant increase in CH_4 emission when farm based emissions are included.

Based on: A.P. Schrier-Uijl, P.S. Kroon, P.A. Leffelaar, J.C. van Huissteden, F. Berendse and E.M. Veenendaal
Plant and Soil, Volume 329, Numbers 1-2 (2010): 509-520

3.1. Introduction

The assessment of the contribution of the trace gas methane (CH_4) to the greenhouse gas effect and the related global warming is of great importance. Northern peatlands are significant sources of CH_4, and are estimated to emit between 20 and 50 Tg yr^{-1} (Mikaloff Fletcher et al., 2004a, b). In northern oligotrophic and eutrophic managed peatland systems, net uptake and emission rates of CH_4 have been found to depend on groundwater level, soil moisture content, temperature, and grassland management (Van den Pol-van Dasselaar et al., 1998a; Hargreaves and Fowler, 1998; Christensen et al., 2003; Blodau and Moore, 2003; Hendriks et al., 2007; Pelletier et al., 2007). CH_4 emissions are difficult to estimate because of their large spatial and temporal variability (e.g. Smemo and Yavitt, 2006). Therefore there are two major challenges when upscaling of CH_4 fluxes from small-scale to landscape scale: 1) selecting the correct ecosystem variables for stratification and 2) developing robust predictive relationships (Groffman et al., 2000).

The majority of fen meadow areas in Atlantic Europe are intensively managed. In the Netherlands, eutrophic peatlands have been drained for centuries and in the last 50 years peatlands have been drained even more deeply to make modern agriculture possible. These peatlands are therefore major carbon sources of CO_2 as a result of peat oxidation (Langeveld et al., 1997; Veenendaal et al., 2007). Burgerhart (2001) has suggested that peat oxidation can be reduced if agricultural peatlands are transformed into wetland nature by raising the water table and by reducing agricultural intensity. These measures alter the carbon cycle and probably turn carbon sources into carbon sinks. Large uncertainties exist, however, of such measures on the CH_4 balance. Hendriks et al., 2007 studied a nature restoration site with no agricultural practices since 1997. They found that a sink of 71 g CO_2-equiv m^{-2} yr^{-1} had developed and they attributed this to a decrease in CO_2 emissions from fields and an increase in CH_4 emissions from ditches and waterlogged soil after stopping agricultural activity. However, in this case the previous situation for CH_4 emission in the time that this area was used for intensive farming is unknown.

The objective of this study is 1) to find empirical relationships that describe the CH_4 emissions from the different landscape elements and to provide a spatially integrated flux for the intensively managed and extensively managed area and 2) to investigate the influence of agricultural activity on the CH_4 balance. Firstly, terrestrial methane fluxes were studied in an intensively managed area, where manure and fertiliser are applied and frequent mowing is practiced. These fluxes are compared to terrestrial CH_4 fluxes measured in an extensively managed area with reduced agricultural activity. Secondly, farm based methane fluxes were estimated using an emissions factor approach. Methane fluxes of the two managed sites were compared to methane fluxes in a former agricultural site which was converted to a nature reserve 15 years ago by stopping intensive dairy farming, raising the water table (> 20 cm below field level for most of the year) (Hendriks et al., 2007).

3.2. Experimental sites and methodology

3.2.1. Experimental sites

The experimental areas Oukoop, an intensively managed dairy farm and Stein, an extensively managed area are located in a polder in the western part of the Netherlands 52°02'01"N" 04°46"E" and 52°01'07"N" 04°46"E", respectively (*Fig 3.1.*). The climate is temperate and humid, with mean annual precipitation of about 800 mm and an annual long-term mean temperature of 9.8 °C. Nol et al. (2008) estimated that 21% of the polder is open water (ditches and small permanent pools), 6% is ditch edges (waterlogged land bordering the ditches) and 73% is drier land with a fluctuating water table. The soils consist of a clayey peat or peaty clay top layer of about 25 cm overlying 12 m eutrophic peat deposits. The polder is between 1.6 and 1.8 m below sea level. Perched water tables occur after heavy rains, when the soil impedes water infiltration. Both sites have been described in more detail by Veenendaal et al. (2007) and Schrier-Uijl et al. (2010b).

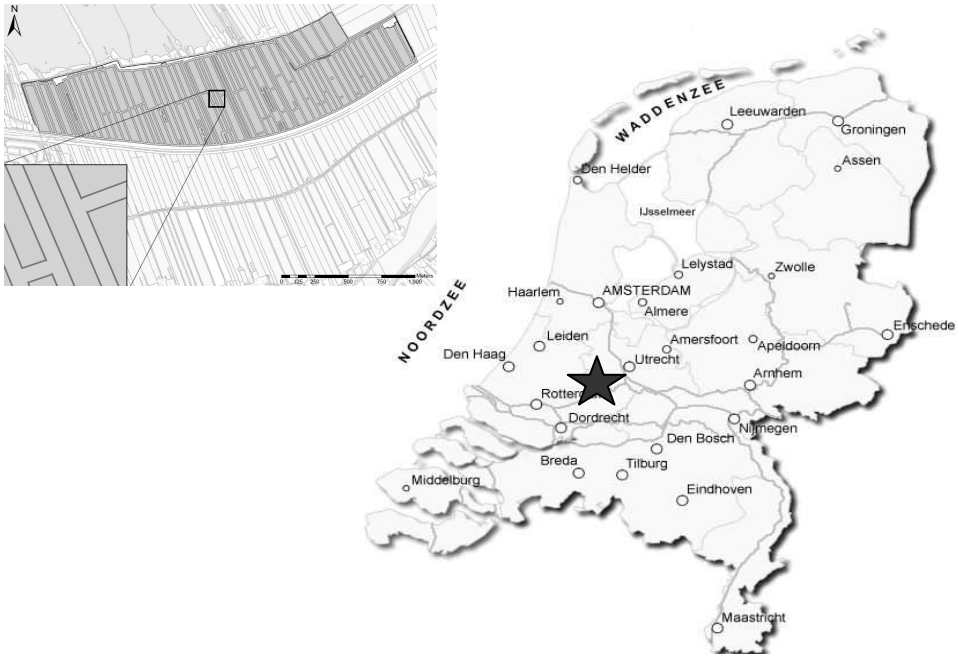


Fig 3.1. *The location of the managed experimental areas in the Netherlands (right) and a close up (left) of the intensively managed area (Oukoop) which shows the characteristic parcellation of the polder landscape.*

The Oukoop experimental area is situated on an intensively managed dairy farm. The management regime varies, but on average consisted of mowing three to four times a year, whereas applications of manure and/or fertiliser took place 5 to 6 times a year. *Lolium perenne* is the most dominant species and *Poa trivialis* is co-dominant with no plants in the drainage ditches. On average the water table is being kept at 50 cm below the surface with perching water tables after heavy rain. Main characteristics of the Oukoop site are given in *Table 3.1*.

The management in Stein has been changed from intensive to extensive some 20 years ago and the area has been turned into a bird reserve. The research area was used as hayfield without application of manure and fertiliser during the measurement period and was mown three times each year after June 15 (2006, 2007 and 2008). The water table was adjusted since 2006, with a high water table (15-20 cm below the surface) in winter and a low water table in summer (50 cm below the surface). In Stein, *Lolium perenne* is also dominant, often with *Poa trivialis* co-dominant and no plants are present in the drainage ditches. Over time, *Holcus lanatus*, *Anthoxanthum odoratum* and *Rumex acetosa* have become more abundant. Characteristics of the Stein site are given in *Table 3.1*.

Table 3.1. *Main characteristics of the Oukoop and Stein site in the Netherlands*

	Soil class, topography and landscape element	Human influence	Parent material	Drainage
Oukoop	Fibric rheic eutric Histosol	0-23 cm: anthropogenic topsoil	0-23 cm: anthropogenic	Poorly drained
	Flat, (alluvial) plain	Peat from ditches on edges	23-50 cm: clayey peat	Saturated for long periods during winter as a result of compaction
		Application of fertiliser: 370, 350 and 260 kg ha-1 in 2006, 2007 and 2008, resp. Application of cow manure: 55, 57 and 90 m3 ha-1 in 2006, 2007 and 2008, resp	> 50 cm: peat, 70% discernible remnants of wood and reed	Mean highest WT: ca. 35 cm Mean lowest WT: ca. 60 cm
Stein	Fibric rheic eutric Histosol	0-23 cm: anthropogenic topsoil	0-29 cm: anthropogenic	Poorly drained since 2005
	Flat, (alluvial) plain	Peat from ditches on edges	29-50 cm: clayey peat	Saturated for long periods every year
		No application of fertiliser or cow manure	> 50 cm: peat, few recognizable plant remnants	Mean highest WT: ca. 25 cm Mean lowest WT: ca. 50 cm

3.2.2. Methodology

3.2.2.1. Measurements

Terrestrial emissions of CH_4 were determined using a closed chamber method (Hutchinson and Mosier, 1981). Methane concentrations were measured using a Photo Acoustic Field Gas Monitor (INNOVA 1412 sn, 710-113, ENMO services, Belgium) connected by Teflon tubes to a PVC chamber (e.g. van Huissteden et al., 2005). Samples were taken from the headspace of a closed cylindrical, dark chamber (30 cm diameter, 25 cm height) that was placed on a collar. A small fan was installed in the chamber to homogenize the inside air and a water lock was placed to control inside pressure. On land we used water between the chamber and the collar to seal the chamber from the ambient air during the measurement. For the ditches we used floaters and a lever system to gently lower the chamber onto the water surface, carefully avoiding the effect of pressure differences and the disturbance of the water surface. We used external silica gel and soda lime filters to minimize cross-interference of CO_2 and water vapour when CH_4 was measured at high CO_2 concentrations. The gas analyser was annually calibrated and tested for drift at the NMI (The Netherlands Institution of Standards, Delft, the Netherlands). In addition, the equipment was occasionally cross-checked for drift with a standard calibration gas. During the measurement period however, the equipment was not found to drift. All measurements were taken during the day, between 9 am and 4 pm. Each flux measurement consisted of five point-measurements taken at one-minute intervals from which dC/dt was calculated.
At the beginning of January 2005, 6 PVC collars (diameter 30 cm) were installed randomly in the footprint area of the eddy covariance systems at both experimental sites, so that gas emissions could be sampled. The measurements were carried out from January 2005 to November 2008 once a month up to twice a month or more during intensive field campaigns.
From February 2006 until November 2008, 19 additional sampling points, distributed over the experimental sites, were sampled every month at both research locations to study spatial variability. At both locations three landscape elements were distinguished with different soil/water temperatures and soil moisture conditions. They were: permanently water-filled ditches, ditch edges, and the field area with a fluctuating water table. Acreage of the landforms was determined by measurements in the field (Nol et al., 2008), the use of GIS maps and use of aerial photographs. In each of the two fields, four sample points were located on the water surface of the ditches, four points on the ditch edges and 11 to 14 sample points in the fields. In total over 1200 measurements were taken.

In addition to each flux measurement, soil or water temperature was measured at 5 cm depth and soil moisture content was determined in the top 5 cm of soil at the sample points, using a HH2 Delta-T device (Delta T Devices, Llandindrod Wells, USA) calibrated for the soil type. Soil and water temperatures were measured every half hour by sensors installed at a depth of 4 cm (Campbell scientific, USA and e+ sensor L-50, Eijkelkamp, the Netherlands, respectively). Water table depth was recorded hourly with pressure sensors

installed in a steel frame to a depth of 70 cm into the soil at one or two places in the field (e+ sensor L-50, Eijkelkamp Agrisearch Equipment BV, Giesbeek, the Netherlands). In the ditch water of both experimental sites, electrical conductivity (*EC*) (Multi 350i, Fisher Scientific, the Netherlands) and *pH* (Cond340i-SET, Fisher Scientific, the Netherlands) was measured in 2008. In Stein, water tables were determined during each field CH_4 measurement in 2008 at the location of the measurement.

Wind speed, air temperature and water vapour pressure were measured with an eddy covariance system consisting of a Campbell Csat C3 Sonic anemometer (Campbell Scientific, Logan, Utah, USA) directed into the main wind direction and a Licor 7500 open path Infrared gas analyser (LI-COR Lincoln, NE, USA). The height of the mast was 305 cm and was located in the middle of the field sample points in both areas (Veenendaal et al., 2007; Schrier-Uijl et al., accepted AFM). For continuous measurements of CH_4 in the intensively managed site, we used the eddy covariance dataset of Kroon et al. (2007) where CH_4 concentrations were measured with a system consisting of a three-dimensional sonic anemometer and a QCL spectrometer (model QCL-TILDAS-76, Aerodyne Research Inc., Billerica MA, USA).

3.2.2.2. Calculations and statistical analyses

Each flux measurement consisted of five points taken at one-minute intervals. The slope d*C*/d*t* of the gas concentration curve at time t=0 s was estimated using linear regression and the slope-intercept method as described by Kroon et al. (2007). The average flux values for CH_4 estimated by the slope intercept method were not significantly different from those estimated by the linear method and therefore, linear regression was used to calculate CH_4 fluxes. The short measurement period of 240 seconds, the rejection of the last point in the case of levelling, and additional measures taken to prevent leakage, mixing and temperature artefacts made it possible to use linear regression (Schrier-Uijl et al., accepted AFM). First, the data quality was assessed: outliers resulting from disturbances, chamber leakage or instrument failures were removed from the data set. Annual mean net CH_4 emissions were estimated by linear regression models of natural-logarithm transformed CH_4 data using T_{soil}/T_{water} as explanatory variables (model-based approach, e.g. Hargreaves and Fowler, 1998). The reliability of this model based approach has been tested previously for the Oukoop experimental site by performing a comparison of the chamber method measurements with eddy covariance measurements of CH_4. The methods agreed well and the difference in cumulative emissions was 17% over a three months period (Schrier-Uijl et al. 2010b).

The statistical significance of differences between landscape elements within sites was calculated with one-way ANOVA. Analysis of covariance, with temperature as covariate,

was used to ascertain the statistical significance of differences in the emissions from the landscape elements of the two sites. Correlations between natural-logarithm transformed CH_4 emissions and independent variables were calculated using step-wise multiple linear regression analysis (case-wise elimination of variables). Paired T-tests were used to calculate the statistical difference in emission between sites at the same day. Uncertainties per landscape element were estimated with a temperature dependent approach and were weighted for the proportion of each landscape element in the total landscape. Statistical analyses were carried out with SPSS. For the calculations of the contribution of animals and manure in the total CH_4 balance, we used the method as described by Hensen et al. (2006) which uses simple emission factors for dairy cows, heifers, calves, manure and farmyard manure.

3.3. Results

3.3.1. Landscape elements

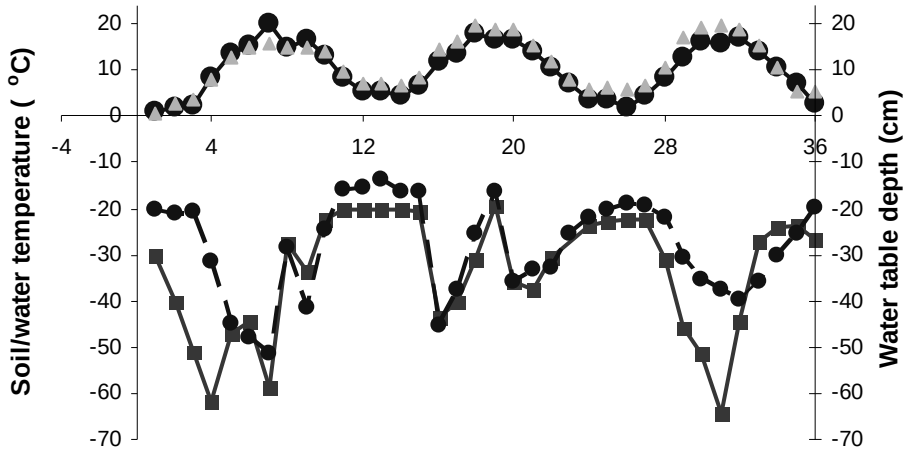


Fig 3.2. *Monthly averaged water temperatures (triangles) and soil temperatures (dots) at 4 cm depth in Stein (upper graph) and water table depths (lower graph) for the intensively managed (squares) and extensively managed (dots) areas from month 1 (January 2006) to month 36 (December 2008).*

We compared the landscape elements for the period that measurements of emissions from all landscape elements ran parallel (January 2006 to December 2008). The average water table depth, water temperature and soil temperature during this period are shown in *Fig 3.2.*

Because temperatures of the two experimental areas are similar (within 0.5 ^{o}C) for most of the time, only Stein is shown.

At both sites, the CH_4 emissions from the ditches (21% of the area) and ditch edges (6% of the area) were significantly higher than those from the fields (P<0.01) and CH_4 emissions from ditches were significantly higher that those from edges (P<0.01). Ditches showed sometimes episodic, exceptionally high emission values. For example in 2006 on September 27 and 28 emissions were 366 (n=6) and 123 (n=4) mg m^{-2} hr^{-1} for the intensively managed and extensively managed areas, respectively versus 5 mg m^{-2} hr^{-1} averaged for the whole summer. During these extreme measurements wind velocity was high and turbulent water surface conditions were observed visually. Field fluxes in Stein were slightly higher compared to field fluxes in Oukoop, with maximum values of 22 and 19 mg m^{-2} hr^{-1}, respectively. Field emission rates were highest in March, April, May and June in Oukoop and in May, June and July in Stein. Field fluxes were found to be increased by about 4% after manure application, which may cause the slightly higher emissions in March and April in Oukoop compared to Stein. Ditches and ditch edges together emit 60% and 68% of the total terrestrial emission in Stein and Oukoop, respectively (*Fig 3.3.*).

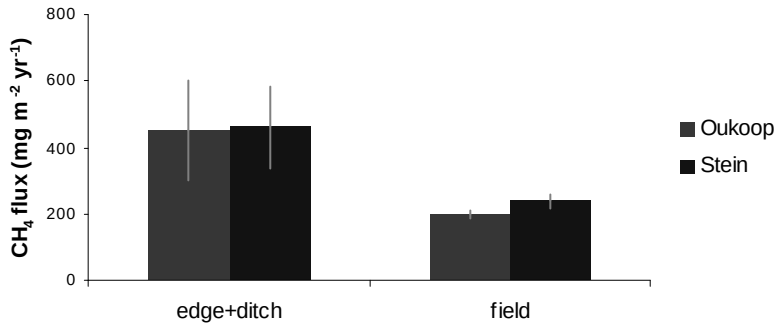


Fig 3.3*. CH_4 emissions from the different landforms for Oukoop grey and Stein black averaged over 2006, 2007 and 2008.*

3.3.2. Driving variables

Methane emission was found to be correlated with several parameters (*Table 3.2.*), with temperature being to be the most important driver for CH_4 emission in all three landscape elements for both Oukoop and Stein. When pooling water temperatures into classes, ln-transformed CH_4 emissions ($\ln(CH_4)$) and water temperature in ditches showed a strong positive correlation in both sites (r^2 = 0.996; P < 0.01; and r^2 = 0.947; P < 0.01 for Oukoop

***Table 3.2**. Matrix of Pearson correlations (r) of ln transformed methane emissions and related parameters for the different landscape elements in Oukoop (intensively managed) and Stein (extensively managed) measured by chambers (left) and eddy covariance (right).*

	*T*soil	*T*air	Moist.	WT	*T*water	*pH*	*EC*	*T*soil	WT	*U*
Oukoop chamber measurements								Oukoop EC measurements		
Field	0.265** (n=195)	0.545** (n=134)	-0.235* (n=119)	0.266[5] (n=40)				0.481** (n=2540)	-0.081[5] (n=77)	0.063**[5] (n=2545)
Edge	0.397** (n=80)	0.350** (n=74)	-0.237* (n=73)							
Ditch		0.422** (n=90)			0.478** (n=97)	0.173[1] (n=30)	0.503**[3] (n=26)			
Stein chamber measurements								Stein EC measurements		
Field	0.474** (n=186)	0.279* (n=190)	-0.166* (n=146)	0.177[5] (n=36)				-	-	-
Edge	0.500** (n=64)	0.473** (n=60)	0.05 (n=59)							
Ditch		0.424** (n=59)			0.496** (n=76)	0.035[2] (n=30)	-0.072[4] (n=30)			

Tsoil: soil temperature; Tair: air temperature; Moist: soil moisture; WT: water table; U: wind velocity.
** or ** statistically significant at P<0.05 and P<0.01, respectively.*
1. range 6.6-7.5; 2. range 6.6-7.4; 3. range 430-650; 4. range 140-346.
5. Corrected for temperature

and Stein, respectively; *Fig 3.4.*). The ln(CH_4) emission in fields was strongly correlated with soil temperature (r^2 = 0.939; P<0.01; and r^2 = 0.865; P<0.01, for Oukoop and Stein respectively; *Fig 3.4.*). For ditch edges the r^2 was 0.941 (P<0.01) and 0.861 (P<0.01) for Oukoop and Stein, respectively.

Average flux values were not significantly different between Oukoop and Stein. In the ditch edges highest maximum values were observed in the spring period in Oukoop and in the summer period in Stein (365 and 118 mg m^{-2} hr^{-1}, respectively), the difference between the average edge fluxes of the sites was not significant. At both sites in 2006, 2007 and 2008 ditch emission rates were highest in June, July, August and September with average summer fluxes of 25 and 31 mg m^{-2} hr^{-1} for Oukoop and Stein, respectively. Emission rates varied greatly depending on the time of the year and about 70% of the total annual CH_4 emissions were observed in summer. Plant cover, composition and biomass in both areas were mostly similar, and field fluxes appeared not influenced by differences in vegetation.
Correlation of field CH_4 emissions with soil moisture were significant (*P*<0.05) in both the intensively and the extensively managed sites (*Table 3.2.*), but when using non-linear multiple regression with temperature as explanatory variable, adding moisture did not significantly improve the predictive power of the regression equation. However, comparing monthly datasets revealed an exception for April in 2006 and 2007, when soil moisture was a stronger predictor than soil temperature for ln(CH_4) at both sites. The water table did not show a correlation with ln(CH_4) neither measured by eddy covariance in Oukoop nor

measured by chambers in Stein (*Table 3.2.*). Eddy covariance measurements show that after temperature correction variability in fluxes over the total area seem to be significantly correlated to wind velocity (*Table 3.2.*). However, because fluxes measured by eddy covariance are integrated over a large area it is not clear if this dependence on turbulence appears in fields, edges, ditches or in a combination of these landscape elements. Ditch water temperature and electrical conductivity (*EC*) in Oukoop were positively correlated with ln(CH_4) fluxes and *pH* was not (*Table 3.2.*). Correlation statistics of chamber measurements (over three years) and eddy covariance measurements (over three months) are given in *Table 3.2.*

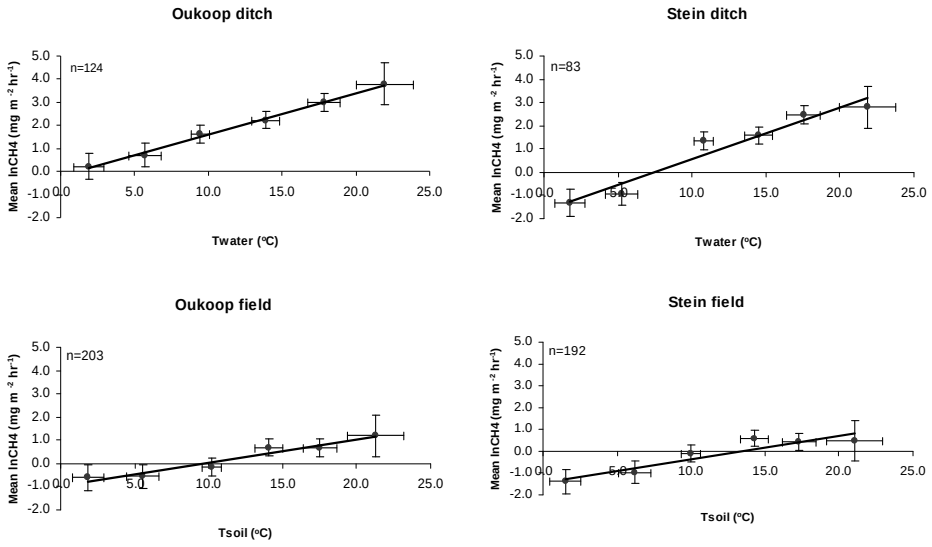


Fig 3.4. *The dependency of ln transformed CH_4 fluxes (mg m^{-2} hr^{-1}) on temperature (^oC) in ditches and fields in Oukoop (left) and Stein (right). Temperatures are pooled. The x-error bars represent the standard deviation of the temperature per class and the y-error bars represent the standard deviation of the ln(CH_4) flux.*

Water temperature was the main driving variable for CH_4 emission in ditches and soil temperature in soils and therefore, water and soil temperature based regressions were used to estimate annual CH_4 balances.

3.3.3. Annual terrestrial methane balances

To estimate annual terrestrial CH_4 balances over 2006 – 2008, we used non-linear regression with temperature as explanatory variable

$$F_{CH_4} = e^{a+bT} \qquad \textbf{3.1.}$$

as described by Schrier et al. (2010b) where F_{CH4} is the CH_4 flux (mg m^{-2} hr^{-1}) measured by the chamber, T is the temperature of soil or water (^oC) and a and b are the coefficients which were derived for each landscape element using the measured temperatures and CH_4 fluxes. Uncertainties in the chamber based regressions were temperature dependent and were different for Oukoop and Stein and for each landform. Average daily uncertainty was about ±60% of the average flux. To further evaluate the reliability of the use of day-time measurements for upscaling to 24-hour fluxes, diurnal variation of CH_4 fluxes was tested using an independent continuous eddy covariance dataset of Kroon et al. (2007). Residual fluxes $\ln(F_{Res})$ (in mg m^{-2} hr^{-1}) were determined after CH_4 fluxes as measured by eddy covariance $\ln(F_{EC})$ (in mg m^{-2} hr^{-1}) were corrected for temperature dependency using

$$\ln(F_{\mathrm{Re}\,s}) = \ln(F_{EC}) - (a + bT_{soil}) \qquad \textbf{3.2.}$$

with a = -1.8, b = 0.17 and T_{soil} = soil temperature (^oC). The residual ln-fluxes were normally distributed (data not shown) and did not differ significantly between day and night (*Fig 3.5.*).

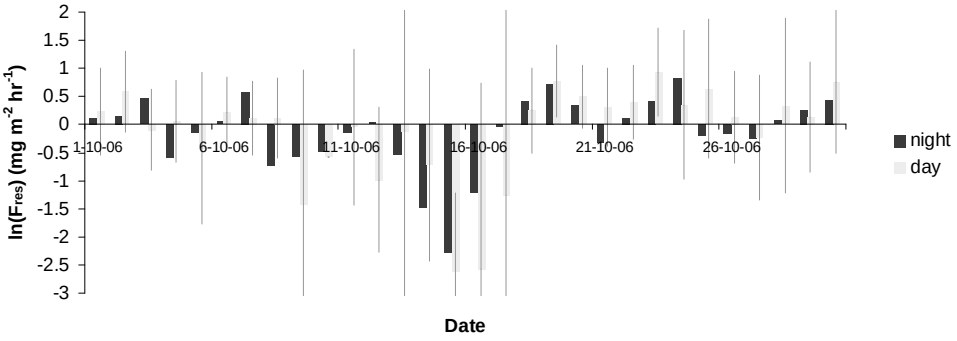


Fig 3.5. *Day/night dynamics of CH_4 fluxes represented by the residual fluxes $\ln(F_{Res})$ after correction for temperature by $\ln(F_{Res}) = \ln(F_{EC}) - (a + bT_{soil})$ for October 2006 based on eddy covariance flux measurements given in Kroon et al. (2007). Error bars show the uncertainty in the mean residual CH_4 flux given by $2\sigma_{\ln(F_{res})} / \sqrt{n}$ with σ the standard deviation.*

Half hourly soil temperature and water temperature data and equation 3.1 were used to estimate half hourly CH_4 fluxes for Oukoop and Stein. Because fluxes were significantly different between the landforms, daily emissions were estimated per landscape element and

multiplied by the area occupied by the landscape element, thus providing a spatially integrated daily flux (*Fig 3.6.*). Total terrestrial modelled fluxes ranged from 203 (±48%) to 146 (±60%) kg ha^{-1} yr^{-1} for Oukoop and from 180 (±54%) to 157 (±63%) kg CH_4 ha^{-1} yr^{-1} for Stein. Methane fluxes did not differ significantly between Oukoop and Stein and did not differ significantly between years. Estimates of cumulative fluxes per site and per year are given in *Table 3.3*.

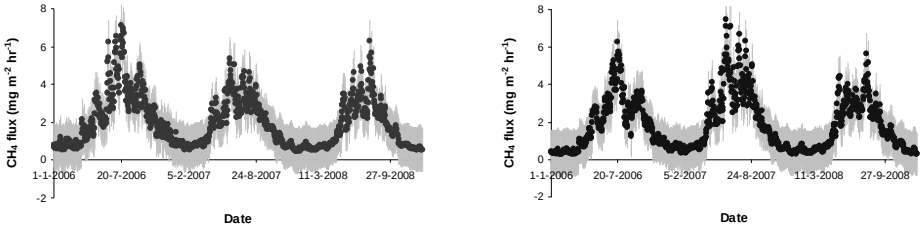


Fig 3.6. *Time series of up-scaled daily spatially integrated CH_4 fluxes in Oukoop (left) and Stein (right) during the period 1 January 2006 to 31 December 2008. The temperature dependent uncertainty band around the dotted line represents plus and minus one standard error for mean prediction, based on the regression analyses, and calculated for each day.*

3.4. Discussion and Conclusions

The objective of this research was to investigate the influence of reduced agricultural activity on the CH_4 balance in two distinct areas. Therefore, the CH_4 emission from a drained, intensively managed peat area was compared to the CH_4 emission from a drained, extensively managed peat area.

First, the spatial and temporal variation of fluxes was investigated and a regression based approach was used to get spatially integrated annual CH_4 emissions. Both areas were stratified into fields, ditches and ditch edges and in both sites the permanently water-filled, 30-60 cm deep ditches and ditch edges turned out to be CH_4 hotspots that together emit 60% and 68% of the total terrestrial emission in Stein and Oukoop, respectively. Apparently, in the procedure of upscaling of the chamber based fluxes to landscape scale it is of great importance to determine the flux associated with each landscape element because they contribute significantly different to the CH_4 balance.

Table 3.3. *Mean soil temperatures, water temperatures (^oC) and temperature based estimates of annual terrestrial methane emissions (kg CH_4 ha^{-1}) for the intensively (Oukoop) and extensively (Stein) managed areas.*

	2006	2007	2008
*T*soil	10.5	10.5	10.4
*T*water	10.9	12.6	12.6
Terrestrial CH_4 emission Oukoop	203 (±48%)	162 (±60%)	146 (±60%)
Terrestrial CH_4 emission Stein	157 (±63%)	180 (±54%)	163 (±59%)

The edges of the ditches were saturated for most of the year, with soil moisture contents >60%. These conditions can give rise to a differently vegetation than in the drier fields and in some places *Iris pseudacorus* and *Typha angustifolia* are present. These aerenchymatic plants may cause additional emissions in the edges because CH_4 diffuses rapidly through their stems (e.g. Chanton et al., 1995; Hendriks et al., 2007).

The CH_4 emission rates from all three landscape elements vary seasonally. In the extensively managed site, high CH_4 emissions were concentrated in the summer period, while in the intensively managed site they were concentrated in early spring and summer, partly associated with field applications of slurry. Van den Pol-van Dasselaar et al. (1999) also reported higher CH_4 emissions after manure application suggesting the combination of higher temperatures in summer, wet soil, the application of easily decomposable organic material and the anaerobic conditions in the slurry as the reason for enhanced CH_4 production. In both sites, over 70% of the total terrestrial CH_4 flux is emitted in summer. The exceptionally high emission peaks from ditches (up to 800 mg m^{-2} hr^{-1} in the summer) may have resulted from ebullition events, during which CH_4 quickly passes through the top layer in the water column. These events mainly occurred when both the water temperature and the wind velocity were high. Ebullition can play an important role during turbulent conditions when bubbles are triggered to escape from the ditch bottom and also low air pressure can cause an increase in ebullition (Hendriks et al., 2008). Besides, during turbulent conditions the aquatic boundary layer thickness is decreased and the potential diffusion rate increases (Kremer et al., 2003a; Minkkinen and Laine, 2006).

Because an integration-based approach may lead to overestimation due to exceptionally high emissions eg. in the summer months, a regression based approach was used to calculate annual CH_4 fluxes (eg. Mikkelä et al., 1995; Chanton et al., 1995). Therefore, we aimed to find empirical relationships that describe the CH_4 emissions from the landscape elements of the two drained peat areas and we found temperature to be the main driver. Except for temperature, no other measured variables did contribute significantly to the predictive power of the chamber based regression. Electrical conductivity was significantly correlated to CH_4 emission from ditches, but to less data was available to include this in the regression. The only month for which we found a strong significant correlation between volume fraction of water in the soil and CH_4 emission rates was April – the period when the field begins to dry out after being waterlogged in winter and when air temperature may rise

rapidly from 10 to 25 ^{o}C. Water table did not influence CH_4 emissions significantly, but the highest emissions occurred at intermediate and sometimes high water tables. In the study areas the water table fluctuated seasonally and within seasons also due to rainfall and polder level regulation by the Dutch water board with high (-15 cm) average water table depths in winter and low (-50 cm) in summer in Stein. In Oukoop water tables depths were low (-65 cm) in summer and high (-20 cm) in winter, and in the winter period perched water tables occurred after heavy rains. Most strong correlations between water table and CH_4 emission in other studies have been found at water tables between 0 and 20 cm below field level (eg. Furukawa et al., 2005; Hargreaves and Fowler, 1998; Strack et al., 2004). The mostly lower water table (>20 cm below field level) in our site could be the cause for the lack of correlation between water table and CH_4 emission.

Eddy covariance measurements showed that no significant differences in CH_4 flux occurred between day and night after correction for temperature. Thus, empirical relations based on temperature can be used to predict 24-hour fluxes of CH_4 per landscape element. The predictions of monthly to annual CH_4 emission have found to be in good agreement with values as measured by eddy covariance (Schrier et al, 2010b). However, considering the daily variability in CH_4 fluxes a better understanding of the influence of variables such as wind, precipitation, air pressure and application of manure and fertiliser on CH_4 emission is needed.

The measurements showed that both areas in this study are a net terrestrial source of CH_4 with annual regression-based means ranging from 146 to 203 kg ha^{-1} for Oukoop and from 156 to 180 kg ha^{-1} for Stein. These values are compared to emissions found in other managed and unmanaged peatland ecosystems (*Table 3.4.*). Van den Pol-van Dasselaar et al. (1998a) studied CH_4 emissions in peat soils in a nature reserve elsewhere in the Netherlands and reported also a large spatial variability. Though the emission values in fields were quite similar to ours, the emission rates they found in saturated land were higher.

The field fluxes found in this study (water tables of 20 to 60 cm below field level) are lower compared to fluxes found in some other eutrophic systems with water tables between 0 and 20 cm below field level for most of the year (eg. Bellisario et al., 1999; Hendriks et al., 2007; Furukawa et al., 2005; Hargreaves and Fowler, 1998; Strack et al., 2004). The extremely high emission rates from ditches found by Minkkinen and Laine (2006) and by Bubier et al. (1993) were similar to the extreme values we found at turbulent water conditions in summer, but average summer fluxes of 18.72 mg m^2 hr^{-1} were higher.

The total ecosystem CH_4 emissions for both areas includes besides the terrestrial emissions also emissions from the farm itself. We estimated the total farm emissions (Q) for the farm

located in the Oukoop area using a general accepted emission factor approach as described by Hensen et al. (2006) by

$$Q = \mathrm{N}_{\mathrm{Dairy}} E_d + N_{Heifer} E_y + N_{Calve} E_c + A_{\mathrm{manure}} E_m + A_{farmyardmanure} E_f \qquad \textbf{3.3}$$

with N the number of animals and with emission factors for dairy cows (E_d), heifer (E_y), calves (E_c), manure (E_m) and farmyard manure (E_f): 274, 170, 48, 53 and 40 g CH_4 day^{-1} $animal^{-1}$ or m^{-3} for farm yard manure, respectively (Sneath et al., 2006; van Amstel et al., 2003).

Table 3.4*. Comparison between the CH_4 emission rates in this study and the CH_4 emission rates reported in other studies on peatland ecosystems. Mean CH_4 emission rates are in mg CH_4 m^{-2} hr^{-1} and the three last columns represent the landscape elements.*

References	ecosystem	field	edge/saturated land	ditch/pond
Minkkinen and Laine (2006)	Boreal fen	-0.04-0.04	-	15.72 up to 25 in summer
Hendriks et al. (2007)	Eutrophic fen abandoned agriculture	1.6	15.3	5.6
Bubier et al. (1993)	Boreal fen	0.0-1.0	-	5.8 up to 38.2 in summer
Bellisario et al. (1999)	Less eutrophic fen	1.0-10.0	-	-
Pelletier et al. (2007)	Boreal fen	0.1-0.9	1.2-8.2	-
Liblik et al. (1997)	Boreal fen	-	2.0-9.2	-
Van den Pol-van Dasselaar et al. (1998b)	Less eutrophic fen	0.9-2.3	11.8	-
Waddington and Day (2007)	Less eutrophic fen	-	-	2.9
Adrian et al. (1994)	Eutrophic aquifer	0.0-8.0	-	
Huttunen et al., (2003)	Boreal fen	-	-	up to 8.0
Hamilton et al. (1994)	Less eutrophic fen	-	-	4.6 - 7.5
Chanton et al. (1993)	Less eutrophic fen	-	5.3-12.4 (aerenchym plants)	-
Schrier et al. (this study)	Eutrophic fen (intensively managed)	0.7-0.8	4.8-6.0	4.5-7.0
Schrier et al. (this study)	Eutrophic fen (extensively managed)	0.8-0.9	2.7-4.4	4.5-5.3

The Oukoop farm owned during the measurement period 50 ha land with 65 adult cows, 20 heifer, 10 calves and had a storage of 700 m^3 slurry and no other farm yard manure. The total farm emissions were estimated at 429 kg CH_4 ha^{-1} yr^{-1}. In addition, this farm used about 20 ha extra land in the Stein area for grass production. The grass is being removed to feed cows in the Oukoop area, and thus the estimated total farm emissions have to be distributed by approximately 0.4 and 0.6 over the extensively managed area and the intensively managed area, respectively. The mean total CH_4 emissions (terrestrial + farm based) for Oukoop and Stein are given in *Table 3.5*.

Table 3.5. *Estimates of annual terrestrial and annual total (terrestrial + farm based) methane emissions (kg ha^{-1}) for the intensively (Oukoop) and extensively (Stein) managed areas.*

	2006	2007	2008
Terrestrial CH_4 emission Oukoop	203 (±48%)	162 (±60%)	146 (±60%)
Total CH_4 emission Oukoop	460 (±48%)	419 (±60%)	403 (±60%)
Terrestrial CH_4 emission Stein	157 (±63%)	180 (±54%)	163 (±59%)
Total CH_4 emission Stein	329 (±63%)	352 (±54%)	335 (±59%)

To investigate the influence of agricultural activity on the CH_4 balance, CH_4 fluxes of our two managed sites were also compared to a former agricultural peat site which was converted to a nature reserve 11 years ago by raising the water table (> 20 cm below field level for most of the year) and cessation of any agricultural management (Hendriks et al., 2007). Without farm based emissions, the intensively managed area and extensively managed area are a net source of CH_4 ranging from 146 to 203 kg ha^{-1} and from 156 to 180 kg ha^{-1}, respectively and the natural site is a larger source of 417 kg ha^{-1}. When we took account of the farm-based emissions, the estimated total annual CH_4 flux was 427 kg ha^{-1} in the intensively managed site and 339 kg ha^{-1} in the extensively managed site averaged over the three years compared to 417 kg ha^{-1} in the natural site of Hendriks et al (2007). This suggests that transformation of intensively managed agricultural land to nature development will lead to an increase in terrestrial CH_4 emission, but not by definition lead to a significant increase in CH_4 emission when farm based emissions are included.

3.5. Acknowledgements

This research project was funded by Wageningen University, The Province of North Holland, CarboEurope IP (contract number: GOCE-CT-2003-505572) and the Dutch National Research Programme Climate Changes Spatial Planning (www.klimaatvoorruimte.nl). Many thanks are due to Jan van Walsem and Frans Möller for their help in the field and in the lab and to the van Eijk and van Eeuwijk families for their logistical support. We are grateful to the State Forestry Service for access to the site and to Nico de Bruin for his collaboration. J. Burrough advised on the English and E.J. Bakker advised on the statistical analyses.

3.6. References

Adrian, N.R., Robinson, J.A., and Suflita, J.M. (1994): Spatial variability in biodegradation rates as evidenced by methane production from an aquifer, Applied and Environmental Microbiology, 60, 3632-3639.

Bellisario, L.M., Bubier, J.L., Moore, T.R., and Chanton, J.P. (1999): Controls on CH_4 emissions from a northern peatland, Global Biogeochemical Cycles, 13, 81-91.

Blodau, C. and Moore, T.R. (2003): Experimental response of peatland carbon dynamics to a water table fluctuation, Aquatic Sciences, 65, 47-62.

Bubier, J., Costello, A., Moore T.R., Roulet, N.T. and Savage, K. (1993): Microtopography and methane flux in boreal peatlands, northern Ontario, Canada, Canadian Journal of Botany-Revue Canadienne De Botanique, 71, 1056-1063.

Burgerhart, N (2001): Mogelijkheden voor koolstofopslag in Nederlandse ecosystemen, Dutch National Inventory Report for LULUCF sector, category 5D1.

Chanton, J.P., Whiting, G.J., Happell, J.D. and Gerard, G. (1993): Contrasting rates and diurnal patterns of methane emission from emergent aquatic macrophytes, Aquat. Bot., 46, 111-128.

Christensen, T.R., Ekberg, A., Strom, L., Mastepanov, M., Panikov, N., Mats, O., Svensson, B.H., Nykanen, H., Martikainen, P.J. and Oskarsson, H. (2003): Factors controlling large scale variations in methane emissions from wetlands, Geophysical Research Letters, 30(7), 1414.

Furukawa, Y., Inubushi, K., Ali, M., Itang, A.M., Tsuruta, H. (2005): Effect of changing groundwater levels caused by land-use changes on greenhouse gas fluxes from tropical peat lands, Nutrient Cycling in Agro ecosystems 71, 81-91.

Groffman, P.M., Brumme, R., Butterbach-Bahl, K., Dobbie, K.E., Mosier, A.K., Ojima, D., Papen, H., Parton, W.J., Smith, K.A. and Wagner-Riddle, C. (2000): Evaluating annual nitrous oxide emissions at the ecosystem scale, Global Biogeochemical Cycles, 14, 1061-1070.

Hamilton, J.D., Kelly, C.A., Rudd, J.W.M., Hesslein, R.H. and Roulet, N.T. (1994): Flux to the atmosphere of CH4 and CO2 from wetland ponds on the hudson-bay lowlands (hbls), Journal of Geophysical Research-Atmospheres, 99, 1495-1510.

Hargreaves, K.J. and Fowler, D. (1998): Quantifying the effects of water table and soil temperature on the emission of methane from peat wetland at the field scale, Atmospheric Environment, 32, 3275-3282.

Hendriks, D.M.D., van Huissteden, J., Dolman, A.J. and van der Molen, M.K. (2007): The full greenhouse gas balance of an abandoned peat meadow, Biogeosciences, 4, 411-424.

Hendriks, D.M.D., Dolman, A.J., van der Molen, M.K., van Huissteden, J. (2008): A compact and stable eddy covariance set-up for methane measurements using off-axis integrated cavity output spectroscopy, Atmospheric Chemistry and Physics, 8(2), 431-443.

Hensen, A., Groot, T.T., Van den Bulk, W.C.M., Vermeulen, A.T., Olesen, J.E., Schelde, K. (2006): Dairy farm CH_4 and N_2O emissions, from one square meter scale to the full farm scale, Agriculture Ecosystems and Environment, Vol. 112, Issues 2-3, 146-152.

Hutchinson, G.L., Mosier, A.R. (1981): Improved soil cover method for field measurement of nitrous-oxide fluxes, Soil Science Society of America Journal, 45(2), 311-316.

Huttunen, J.T., Nykanen, H., Turunen, J. and Martikainen, P.J. (2003): Methane emissions from natural peatlands in the northern boreal zone in finland, fennoscandia, Atmospheric Environment, 37, 147-151.

Kroon, P.S., Hensen, A., Jonker, H.J.J., Zahniser, M.S., van 't Veen, W.H.,Vermeulen, A.T. (2007): Suitability of quantum cascade laser spectroscopy for CH4 and N2O eddy covariance flux measurements, BG, 4, 715-728.

Kroon, P.S., Hensen, A., van den Bulk, W.C.M., Jongejan, P.A.C., Vermeulen, A.T. (2008): The importance of reducing the systematic error due to non-linearity in N_2O flux measurements by static chambers, Nutrient Cycling in Agro Ecosystems, 82, 2, 175-186.

Kremer, J.N., Reischauer, A., D'Avanzo, C. (2003): Estuary-specific variation in the air-water gas exchange coefficient for oxygen, ESTUARIES , 26(4a), 829-836.

Langeveld, C.A., Segers, R., Dirks, B.O.M., vandenPolvanDasselaar, A., Velthof, G.L., Hensen, A. (1997): Emissions of CO_2, CH_4 and N_2O from pasture on drained peat soils in the Netherlands, European journal of agronomy, Vol 7, Issue 1-3, 35-42.

Liblik, L.K., Moore, T.R., Bubier, J.L. and Robinson, S.D. (1997): Methane emissions from wetlands in the zone of discontinuous permafrost: Fort simpson, northwest territories, Canada, Global Biogeochemical Cycles, 11, 485-494.

Mikaloff, Fletcher S.E., Tans, P.P., Bruhwiler, L.M., Miller, J.B. and Heimann, M. (2004a): CH_4 sources estimated from atmospheric observations of CH_4 and 13C/12C isotopic ratios: 1. Inverse modeling of source processes, Global Biogeochemical Cycles, 18(4), doi:10.1029/2004GB002223.

Mikaloff, Fletcher S.E., Tans, P.P., Bruhwiler, L.M., Miller, J.B. and Heimann, M. (2004b): CH_4 sources estimated from atmospheric observations of ch_4 and 13C/ 12C isotopic ratios: 2. Inverse modeling of CH_4 emissions from geographical regions, Global Biogeochemical Cycles, 18.

Mikkelä, C., Sundh, I., Svensson, B.H., Nilsson, M. (1995): Diurnal variation in methane emission in relation to the water table, soil temperature, climate and vegetation cover in a Swedish acid mire, biogeochemistry, Vol 28, Issue 2, 93-114.

Minkkinen, K. and Laine, J. (2006): Vegetation heterogeneity and ditches create spatial variability in methane emissions from peatlands drained for forestry, Plant and Soil, 285, 289-304.

Nol, L, Verburg, P.H., Heuvelink, G.B.M., Molenaar, K. (2008): Effect of land cover data on nitrous oxide inventory in fen meadows. J. of Envir. Qual. 37, 1209-1219.

Pelletier, L., Moore, T.R., Roulet, N.T., Garneau, M. and Beaulieu-Audy, V. (2007): Methane emissions from three peatlands in the La Grande Riviera watershed, James Bay lowland, Canada, J. Geophys. Res.-Biogeosci., 112, noG1, Note(s): G01018.1-G01018.12] (3/4 p.).

Smemo, K.A., Yavitt, J.B. (2006): A multi-year perspective on methane cycling in a shallow peat fen in central New York State, USA, Wetlands 26(1) 20-29.

Sneath, W., Beline, F., Hilhorst, A., Peu, P. (2006): Monitoring GHG from manure stores on organic and conventional dairy farms, Agric. Ecosyst. Environ. 112, pp. 122–128.

Strack, M., Waddington, J.M., Tuittila, E.S. (2004): Effect of water table drawdown on northern peatland methane dynamics: Implications for climate change, Global Biochemical Cycles, DOI: 10.1029/2003GB002209.

Van Amstel, A., Kroeze, C., van Eerdt, M., Dumont, M., Both, D. (2003): National emissions of methane and nitrous oxide from agriculture in The Netherlands: current estimates and future trends, Workshop on Inventories and projections of greenhouse gas and ammonia emissions from agriculture, 27–28 February 2003, European Environmental Agency, Copenhagen.

van den Bos, R. (2003b): Restoration of former wetlands in the Netherlands; effect on the balance between CO_2 sink and CH_4 source, Netherlands Journal of Geosciences-Geologie En Mijnbouw, 82, 325-331.

van den Pol-van Dasselaar, A., Corre, W.J., Prieme, A., Klemedtsson, A.K., Weslien, P., Stein. A,, Klemedtsson. L, and Oenema. O, (1998a): Spatial variability of methane, nitrous oxide, and carbon dioxide emissions from drained grasslands, Soil Science Society of America Journal, 62, 810-817.

van den Pol-van Dasselaar, A., van Beusichem, M.L., and Oenema, O. (1998b): Effects of soil moisture content and temperature on methane uptake by grasslands on sandy soils, Plant and Soil, 204, 213-222.

van den Pol-van Dasselaar, A, Van Beusichem, M.L. and Oenema, O. (1999): Methane emissions from wet grasslands on peat soil in a nature preserve, Biogeochemistry, 44, 205-220.

van Huissteden, J., Maximov, T.C. and Dolman, A.J. (2005): High methane flux from an arctic floodplain (Indigirka lowlands, eastern Siberia), J. Geophys. Res.-Biogeosci., 110(G2), G02002.

Veenendaal, E.M., Kolle, O., Leffelaar, P.A., Schrier-Uijl, A.P., Van Huissteden, J., van Walsem, J, Möller, F, and Berendse, F. (2007): CO_2 exchange and carbon balance in two grassland sites on eutrophic drained peat soils, Biogeosciences: 4, 1027-1040Waddington JM and Day SM (2007) Methane emissions from a peatland following restoration, J. Geophys. Res.-Biogeosci., doi 10.1029/2007JG000400.

Chapter 4

Comparison of chamber and eddy covariance based CO_2 and CH_4 emission estimates in a heterogeneous grass ecosystem on peat

Fluxes of methane (CH_4) and carbon dioxide (CO_2) estimated by empirical models based on small-scale chamber measurements were compared to large scale eddy covariance (EC) measurements. For CH_4 direct EC measurements were used and for CO_2 a combination of EC measurements and EC based models were used. The experimental area was a flat peat meadow in the Netherlands with heterogeneous source strengths for both greenhouse gases. Two scenarios were used to assess the importance of stratifying the landscape into landscape elements before up-scaling the fluxes measured by chambers to landscape scale: one took the main landscape elements into account (field, ditch edge ditch), the other took only the field into account. Non-linear regression models were used to up-scale the chamber measurements to field emission estimates. EC CO_2 respiration consisted of measured nighttime EC fluxes and modeled daytime fluxes using the Arrhenius model. EC CH_4 flux estimate was based on daily averages and the remaining data gaps were filled by linear interpolation. The EC and chamber based estimates agreed well when the three landscape elements were taken into account with 16.5% and 13.0% difference for CO_2 respiration and CH_4, respectively. However, both methods differed 31.0% and 55.1% for CO_2 respiration and CH_4 when only field emissions were taken into account when up-scaling chamber measurements to landscape scale. This emphasizes the importance of stratifying the landscape into landscape elements. The conclusion is that small-scale chamber measurements can be used to estimate fluxes of CO_2 and CH_4 at landscape scale if fluxes are scaled by different landscape elements.

Based on: A.P. Schrier-Uijl, P.S. Kroon, A. Hensen,
P. A. Leffelaar, F. Berendse and E.M. Veenendaal
Agricultural and Forest Meteorology (2010) (doi 10.1016/j.agrformet.2009.11.007)

4.1. Introduction

Net emission of greenhouse gases in an ecosystem is a result of uptake and emission and will change depending on variables such as temperature, moisture conditions, soil physics, topography, management practices and vegetation cover (e.g. Riutta et al., 2007). The greenhouse gases carbon dioxide (CO_2) and methane (CH_4) have a significant impact on the greenhouse gas balance and account for over 60% and 20% of global warming, respectively (IPCC, 2007a). Variables that define greenhouse gas production and consumption in ecosystems vary spatially and temporally. In order to determine the origin of fluxes, to properly extrapolate fluxes to the ecosystem scale and to ascertain possible management interventions in heterogeneous landscapes, it is necessary to combine small-scale gas measurement techniques and large-scale measurement techniques. Comparison of the different scale techniques can also give independent information about the reliability of the methods. Three widely used and totally independent methods for determining fluxes at different spatial scales are chamber-based methods, micrometeorological towers (e.g. eddy covariance systems), and calculations based on equations for diffusion at the soil/air and water/air interfaces (Denmead et al., 2008).

In this study, the first two methods are compared for CO_2 respiration and CH_4 emissions in a peat meadow and estimates of cumulative emissions are given over one year and three months, respectively. With the chamber-based method, the emissions are up-scaled to field scale using a temperature regression model which takes into account the heterogeneity of the ecosystem. Three regression models are determined for the three main landscape elements (field, ditch edges and ditches) based on chamber flux measurements. The up-scaled chamber fluxes are compared to EC CO_2 respiration and CH_4 fluxes which were measured at the same site. CO_2 respiration fluxes are based on measured night time fluxes and modelled daytime fluxes using the Arrhenius model. EC CH_4 fluxes are based on daily averages and the remaining data gaps are filled by linear interpolation.

Chamber-based methods are often used to determine source and sink distributions in non-uniform landscapes and are used to quantify small-scale spatial differences in CO_2 and CH_4 fluxes (e.g. Christensen et al., 1995, 2000; Hutchinson and Livingston, 2002; Kutzbach et al., 2004; Hendriks et al., 2007; Schrier-Uijl et al., 2009). Because of the large increase in gas concentration in the headspace, chamber-based methods are highly sensitive (Denmead, 2008). However, this methodology is often criticised because of uncertainties due to pressure artefacts, temperature effects (e.g. Hutchinson and Livingston, 2002; Rochette and Eriksen-Hamel, 2008), discontinuity of measurements and lack of spatial integration (Flechard et al., 2007a). Although in recent set-ups most of the direct chamber effects have been eliminated, the problem of neglecting the influence of wind remains (Denmead, 2008). In this study closed static chambers were used, in which the air is circulated between the

headspace of a dark chamber and a gas analyser. Measures were taken to minimise pressure artefacts, cross interference and mixing artefacts.

Eddy covariance (EC) techniques have been used for continuous quantification of landscape-scale temporal variability of CO_2 and CH_4 (e.g. Baldocchi et al., 2001; Aubinet et al., 2000; Veenendaal et al., 2007; Hendriks et al., 2007; Kroon et al., 2007). This technique has been used to measure CO_2 fluxes, and many studies have been published in which CO_2 EC fluxes are discussed; However, only a few instruments are available for EC measurements of CH_4 and until recently only a few studies have tested the appropriateness of EC measurements of CH_4 (e.g. Kroon et al., 2007; Hendriks, 2008; Kroon et al.,2009a). The EC method is based on sensing turbulent wind fields, temperature, and gas concentrations at high frequency at a certain measurement point (e.g. Baldocchi, 2003). The advantage of this method is that it does not disturb the soil/air environment, and integrates over larger areas and has continuous time coverage. EC fluxes represent the integrated net fluxes from the landscape upwind from the measurement point. The extent of the upwind area from which the flux originates, the footprint area, depends on atmospheric stability and surface roughness (e.g. Grelle and Lindroth, 1996; Kormann and Meixner, 2001; Neftel et al., 2007). However, EC measurements are based on assumptions, such as horizontal homogeneity, flat terrain and negligible mean vertical wind velocities over the averaging period. Furthermore, they are beset by uncertainties, among others due to one-point sampling and the lack of low and high frequency responses (e.g. Moore, 1986; Aubinet et al., 2000; Kroon et al., 2009a; Kroon et al., 2009b).

In this study, the cumulative CO_2 respiration and CH_4 emissions are estimated over one year (2006) and three months (August - November 2006), respectively. The experimental area is a flat peat meadow in the Netherlands with heterogeneous CO_2 and CH_4 source strengths. The objective is to compare the cumulative field emissions derived based on chamber measurements with the cumulative emissions based on EC flux measurements. Two emission scenarios are compared: the first scenario took only fields into account, the second scenario took the three main landscape elements (fields, ditches, ditch edges) into account when up-scaling the chamber measurements to landscape scale.

4.2. Experimental site and climatic conditions

The experimental site Oukoop, is an intensively managed dairy farm area in a polder in the west of the Netherlands (52°02’11.22”N” 04°46 ‘49.53”E”). The site is divided into three landscape elements according to micro topography and soil moisture condition: permanently water-filled ditches, almost saturated ditch edges and the relatively dry field area with fluctuating water table (*Fig 4.1.*). Ditches, ditch edges and fields account for 16%,

5% and 79% of the average footprint area of the EC system, as estimated from areal photographs.

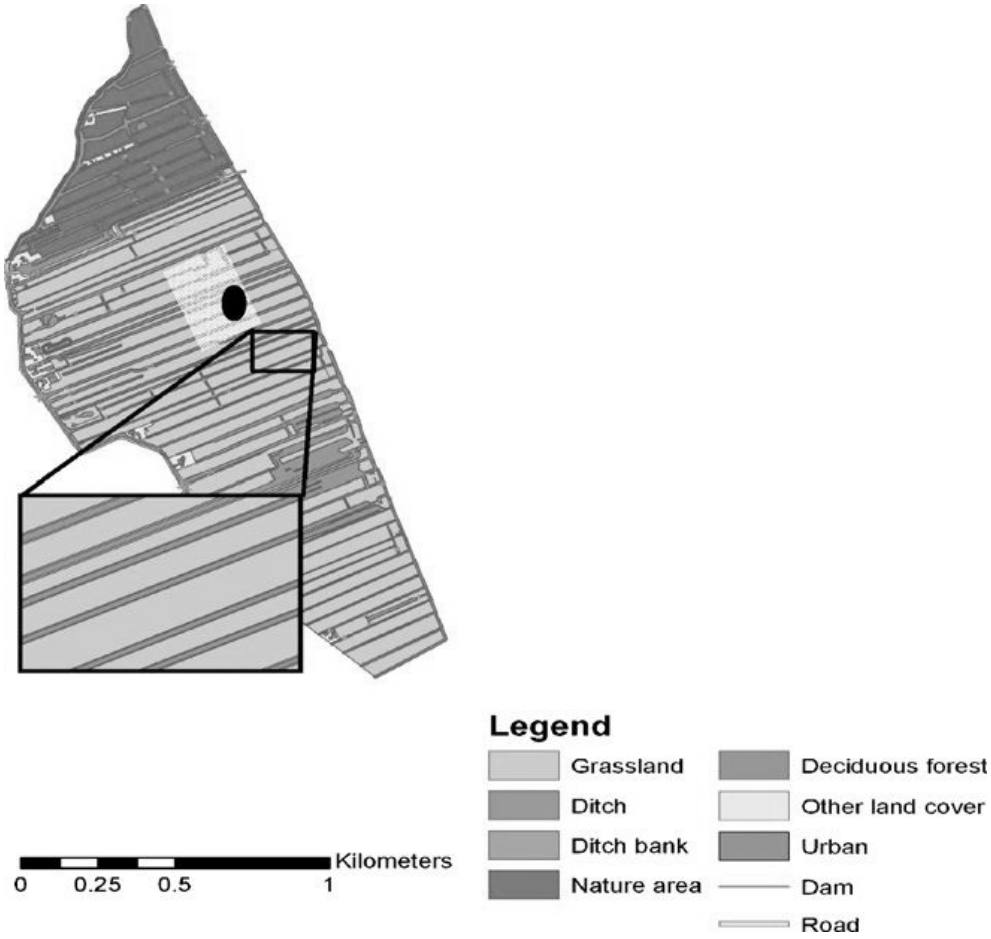


Fig 4.1. *The Oukoop experimental site. The inset is a close-up of the managed peat area around the measurement set-ups, showing the characteristic field pattern of the polder landscape (Nol et al., 2008). The black dot is the location of the EC systems and the white area is the area where chamber measurements were performed.*

The soil consists of a clayey peat layer of 0.5 m overlying 12 m eutrophic peat deposits. *Table 4.1*. gives more details on the site properties.
The climate is temperate and humid, with an annual mean precipitation of 807 mm and an annual mean temperature of 11.2 °C in 2006. The elevation of the polder is between 1.6 and 1.8 m below sea level. The depth of the groundwater varies from 0.65 to 0.20 m below field level and perched water tables occur after heavy rain, when the soil impedes water

infiltration. Manure and artificial fertiliser are applied about five times a year from February to September. The application rates in 2006 were 55 m^3 ha^{-1} (253 kg N ha^{-1} yr^{-1}) for cow manure and 320 kg ha^{-1} (84 kg N ha^{-1} yr^{-1}) for artificial fertiliser. The ecosystem is eutrophic and in terms of vegetation the area is homogeneous, with Rye grass (*Lolium perenne*) the dominant grass species, and Blue grass (*Poa trivialis*) the co-dominant species (Veenendaal et al., 2007). The grass height ranged from 0.05 – 0.35 m and is harvested about four times a year. Based on farmers data and field measurements, the yield is estimated at $6.5x10^3$ kg dry matter ha^{-1} y^{-1}. Grazing removed an additional 2- $3x10^3$ kg dry matter ha^{-1} y^{-1} giving a total dry matter offtake in the order of $9x10^3$ kg dry matter ha^{-1} y^{-1} (Veenendaal et al., 2007). The area is a source of CO_2 and CH_4 (Veenendaal et al., 2007; Kroon et al., 2007; Schrier-Uijl et al., 2010a). Spatial variability of CO_2 and CH_4 is known to be high (Schrier-Uijl et al., 2010a).

Table 4.1. *Main characteristics of the Oukoop site in the Netherlands*

Soil class, topography and landscape element	Human influence	Parent material	Drainage
Fibric rheic eutric Histosol	0-23 cm anthropogenic topsoil Peat from ditches on edges Application of fertiliser	0-23 cm: anthropogenic 23-50 cm: clayey peat >50 com: peat, 70% discernible remnants of wood and reed	Poorly drained Saturated for long periods during winter as a result of compaction Mean highest WT: ca. 35 cm Mean lowest WT: ca. 50 cm
Flat (alluvial) plain			

4.3. Instrumentation and methodology

4.3.1. Chamber based emission estimates methodology

4.3.1.1. Chamber measurements

CO_2 and CH_4 fluxes from the field, ditches and ditch edges were measured using a static chamber method (Hutchinson and Mosier, 1981). A Photo Acoustic Field Gas Monitor (INNOVA 1412 sn, LumaSenseTM Technologies, Ballerup, Denmark) was connected by Teflon tubes to a dark PVC chamber with a surface area of 0.072 m^2 and a height of 0.20 m. This methodology is often criticised because of uncertainties due to pressure and temperature effects (e.g. Hutchinson and Livingston, 2002; Rochette and Eriksen-Hamel, 2008), discontinuity of measurements and lack of spatial integration (Flechard et al., 2007a) and therefore measures are taken to avoid these artefacts. A small fan was installed in the chamber to homogenize the inside air; a small water lock on top of the chamber adapted inside pressure to air pressure. In the field and at the ditch edges the chamber was put into a

water-filled groove on a 0.15 m high collar inserted into the soil to prevent leakage. At the ditches, floaters and a lever system were used to lower the chamber onto the water surface gently, to avoid the effect of pressure differences. Since the gas monitor software does not compensate fully for cross-interference of CO_2 and water vapour at high concentrations, air was lead through silica gel and soda lime filled glass tubes before it entered the gas analyzer. The soda lime filter was not applied in months with low CO_2 concentrations (fluxes ranging from 0-600 mg m^2 hr^{-1}). The gas analyzer was annually calibrated and tested for drift at the NMI (Nederlands Meet Instituut: Delft, The Institution of Standards, the Netherlands). Detection limits of the gas analyzer were 1.5 ppm for CO_2 and 0.1 ppm for CH_4.

To estimate the quality of the measurement set-up we used the Rochette and Eriksen-Hamel (2008) method, which defines the quality of different chamber systems on a scale of 1 (very poor) to 4 (high), basing the overall confidence in the flux measurement on the weakest factors in each study. The quality of the measurement set-up used in this study was estimated to be 4.

To study the spatial variability of CO_2 and CH_4 emissions we measured the fluxes at 21 locations: 4 on the ditches, 4 on the ditch edges and 13 on the field. The measurements were performed between 9 am and 3 pm at bi-weekly intervals in spring and summer and four-weekly intervals in autumn and winter. For the regression analyses we used 300 measurements of CO_2 obtained in 2006 and 600 measurements of CH_4 obtained over the period 2005-2008.

Each flux measurement consisted of five points taken at one-minute intervals to estimate the slope dC/dt at time t=0. Concentration profiles were visually analyzed on linearity; the first or last measurement point was rejected if pressure artifacts were visible after the chamber had been positioned or when leveling occurred due to leakage. This left 85% of the CO_2 data and 65% of CH_4 data for analysis. The slope dC/dt of the gas concentration curve at time t=0 s was estimated using linear regression and the slope-intercept method (Kroon et al., 2008). The average flux values for CO_2 and CH_4 estimated by the slope intercept method were about 3.5% and 4.0% higher than those estimated by the linear method. This small difference between the two methods is a result of the short measurement period of 240 seconds, the rejection of the last point in the case of leveling, and by additional measures taken to prevent leakage, mixing and temperature artifacts (e.g. water seal around the chambers, small fan and water lock). Because the difference was not significant, the linear method was used to calculate the flux

$$F_{ch} = \frac{V}{A}\left[\frac{dC}{dT}\right]_{t=0} \qquad \textbf{4.1}$$

where F_{ch} is the flux measured by the chamber, V is the volume of the chamber (m^3), A is the surface of the chamber (m^2), C is the gas concentration in the chamber at ambient temperature and pressure (mg m^{-3}) and t is the length of the measurement period (s). Since the output of the gas analyser is in ppm, dC/dt is calculated by

$$\frac{dC}{dt} = 0.036\frac{PM}{RT}\frac{d\hat{C}}{dt} \qquad \textbf{4.2}$$

where P is the air pressure (Pa) , $\hat{C}$ is the gas concentration (ppm), R the universal gas constant (8.314 J.mol^{-1}K^{-1}), T the absolute temperature during the measurement (K), M the molecule weight and 0.036 is a conversion factor for time.

4.3.2. Empirical models based on chamber data

The empirical model for CO_2 respiration was based on a modified Arrhenius equation which is given by (Lloyd and Taylor, 1994)

$$R_{\text{Res}} = R_{10}\, e^{E_0\left(\frac{1}{(283.15-To)}-\frac{1}{(T-To)}\right)} \qquad \textbf{4.3}$$

in which R_{Res} is the respiration measured by the chambers (μmol m^{-2} s^{-1}), E_0 is the activation energy (K), T_0= 227.13 K, T is the soil or water temperature and R_{10} is the ecosystem respiration at 10 ^{o}C. The R_{10} and E_0 were determined for each landscape element using the measured CO_2 respiration and the measured soil and water temperatures during the respiration measurement (e.g. Reichstein et al., 2005).

Methane emission was found to be correlated with several parameters, with temperature being the most important driver (Schrier-Uijl et al., 2010a). The dependence on temperature followed an exponential function which differed per landscape element; this finding was in line with earlier studies (e.g. Hargreaves et al., 2001; Hendriks et al., 2007). The effect of soil moisture or water table did not enhance the predictive power of the regression, probably because the water levels in the field (a polder) are controlled by the water board. We used an empirical model for CH_4, which was based on a non-linear regression with temperature as only explanatory variable

$$F_{CH_4} = e^{a+bT} \qquad \textbf{4.4}$$

where F_{CH4} is the CH_4 flux measured by the chamber, T is the temperature of soil or water and a and b are the coefficients which were derived for each landscape element using the

measured temperatures and CH_4 fluxes. This resulted in three landform-dependent empirical relationships.

Half-hourly soil and water temperatures were used to calculate half-hourly emissions for CO_2 and CH_4 for each landscape element using equations 4.3 and 4.4, respectively. Because soil and water temperatures were measured by different instruments, the temperatures were corrected for the offset between the average temperature derived from the manual sensor used at each chamber measurement and the sensors installed in the soil at 4 cm depth and in the water. The final emission was calculated by the weight-factors for each landscape element within the average footprint of the EC systems. The uncertainties around the CO_2 and CH_4 lines were based on temperature-dependent uncertainties of the parameters in the regression analyses and were calculated for each day.

4.3.3. Eddy covariance emission estimates methodology

4.3.3.1. Eddy covariance measurements

EC fluxes of CO_2 and CH_4 were both measured at a height of 3.05 m in the middle of the field, 5 m apart from each other. The footprint is estimated each half hour by the model of Kormann and Meixner (Kormann and Meixner, 2001; Neftel et al., 2007). Monthly average length of the 1%-ellips varied between 300 and 500 meters and monthly average half width of the 1%-ellips varied from 20 to 90 meters in the period August – December 2006. The 1%-ellips is the area where the footprint function reaches 1% of its maximum value. Under the predominant wind direction (W/SW) the proportions of field, ditch and ditch edge within the footprint remained approximately 79%, 16% and 5%, respectively. When the wind shifted to N/NW, which occurred in 8.5% of the measurement moments in 2006, then the proportion of ditch in the footprint was 16–19%.

The terrain around both towers was flat and free of obstruction for at least 600 m in all directions, except for the container in which instruments were placed. The CO_2 mast consisted of a Campbell Csat C3 Sonic anemometer (Campbell Scientific, Logan, Utah, USA) oriented towards the predominant wind direction and a Licor 7500 open path Infrared gas analyzer (LI-COR Lincoln, NE, USA). The CO_2 gas analyzer was calibrated annually using CO_2 concentrations of 370 and 400 ppm. Data were logged with a data logger (CR5000, Campbell Scientific, USA).

Night time EC fluxes of CO_2 were integrated as half-hourly means with the EDDYFLUX software (O. Kolle MPI-BGC Jena following Carbo-Europe protocols: Aubinet et al., 2000). Data were filtered for spikes and linear detrending was used. A Webb correction for density fluctuations was applied (Webb et al., 1980). Quality control criteria according to

Foken and Wichura (1996) were used to reject bad data. In addition, we also removed bad quality data due to temporary frost and dew, or moisture formation on the open path gas analyzer sensor head and we removed the data for which respiration values were reduced below a certain friction velocity (u*). From the remaining data set, which varied from 42% data coverage in July to 55% data coverage in December, storage fluxes were calculated from the CO_2 measurements and added to the EC flux for each 30 min period according to Hollinger et al. (1994). For a detailed description of the CO_2 meteorological system, see Veenendaal et al. (2007).

Up to recently, it was not possible to obtain EC flux measurements of CH_4. However, instrumentation that meets the requirements for continuous measurements of CH_4 is now becoming available (e.g. Laurila et al., 2005; Rinne et al., 2007). One of these systems is that of Kroon et al (2007) which measured EC fluxes of CH_4 in the same field where chamber measurements of CH_4 were performed from August – September 2006. The system consisted of a three-dimensional sonic anemometer (model R3, Gill Instruments, Lymington, UK) and a quantum cascade laser (QCL) spectrometer (model QCL-TILDAS-76, Aerodyne Research Inc., Billerica MA, USA). The QCL spectrometer was calibrated at least once a week using mixtures in N_2/O_2 of CH_4 concentrations of 1700 and 5100 ppb (Scott specialty gases, the Netherlands). The sonic anemometer data and the QCL spectrometer output were logged and processed using a data acquisition program developed at ECN, following the procedures of McMillen (1988).

The CH_4 fluxes were measured according to Kroon et al. (2007) and were calculated according to Kroon et al. (2009a,b). The EC fluxes were corrected for changing calibration, frequency response losses and density fluctuations. The net CH_4 exchange was calculated by adding the storage change term to the EC flux term. The data was flagged using the instationarity tests of Foken and Wichura (1996) and was rejected when the flag was larger than 2. In addition, the fetch was checked by Kormann and Meixner footprint model (Kormann and Meixner, 2001) and the flux value was removed when less than 70% of the flux came from the dairy farm site. No u*-filtering was applied on this data set since the CH_4 was probably partly stored in the soil during periods of low u* and escaped to the atmosphere during periods of high u*-values (i.e. pumping effects) (e.g. Gu et al., 2005; Flechard et al., 2007b; Kroon et al., submitted). We used daily CH_4 fluxes to minimize the uncertainty in the used estimates. Daily values were derived when 12 or more 30 min fluxes were available during a day (Kroon et al., submitted) which occurred in 84% of the days and the remaining gaps were filled by an linear interpolation procedure.

The uncertainty in a single 30 min EC flux measurement consists of several uncertainties either linked to the correction algorithm of the systematic errors or linked to processes for which no corrections could be made. All uncertainties are random and decrease with increasing independent realizations (Kroon et al., 2009a). The uncertainty in a 30 min EC

flux can be even larger than 100% and is mainly caused by the one-point uncertainty (e.g., Businger et al., 1986; Kroon et al., 2009a). However, the uncertainty in emission estimates over longer time spans, like a day, month or year, will be much smaller. Annual uncertainty in the CO_2 and CH_4 measurements are in the order of 15% for both instrumental set-ups (Veenendaal et al., 2007; Hendriks et al., 2007; Baldocchi et al., 2001).

4.3.3.2. Model based on eddy covariance data

Net eco-system exchange of CO_2 (NEE) was determined directly from the eddy covariance flux measurements and is considered to be the sum of the gross eco-system production (GEP) and ecosystem respiration (R_{eco}). The respiration was determined using nightly NEE values, when photosynthetic active radiation (PAR) = 0, assuming that photosynthesis is absent and the NEE consists only of R_{eco}. Next, the soil respiration is described as a function of the half hourly soil temperatures (N= 2710) by using the Arrhenius relation (equation 4.3) (eg. Béziat et al., 2009) and parameters were estimated from the one year dataset (2006). In some months there was too less data available for calculating monthly R_{10} and E_0 values and therefore yearly parameters were estimated. The missing night- and day time CO_2 respiration data (PAR>0) were estimated with this model (e.g. Béziat et al., 2009; Veenendaal et al., 2007; Hendriks et al., 2007; Reichstein et al., 2005). The data gaps in CH_4 flux measurements were filled using a linear interpolation.

4.3.4. Additional measurements

Soil temperature, water temperature, air temperature and soil moisture content were recorded during each chamber measurement and grass height was determined every month. The mast was equipped with micrometeorological sensors to measure short and long wave radiation (CRN1 Kipp & zonen, Delft, the Netherlands), photosynthetic photon flux density (Parlite, Kipp & zonen, the Netherlands), air temperature and humidity (HMP 45a, Vaisala, Finland) and air pressure (Druck CS115, Campbell Scientific, USA). Soil heat flux plates (HPF01, Campbell Scientific, USA) were installed in the field to estimate the soil heat flux at depths of 0.02, 0.04, 0.08, 0.16, and 0.32 m. Soil temperature sensors were installed at depths of 0.02, 0.04, 0.08, 0.16, and 0.32 m (Campbell Scientific, USA). Precipitation rates were measured with a tipping bucket rain gauge (Young, Traverse City Michigan, USA). Water level was measured with pressure transducers (Eijkelkamp, Giesbeek, the Netherlands).

4.4. Results and discussion

4.4.1. Parameter estimation

Night time CO_2 respiration measurements by EC were performed over 2006. Fluxes were found to be reduced below u^* of 0.16 m s^{-1} and therefore those fluxes were eliminated from the dataset. To accurately calculate ecosystem respiration at 10 ^oC and the activation energy in equation 4.3 (R_{10} and E_0), night time fluxes over 2006 were analyzed and non-linear regression was applied. Yearly parameters were used, which were estimated at 4.1 µmol CO_2 m^{-2} s^{-1} (95% confidence interval 3.9–4.3 µmol CO_2 m^{-2} s^{-1}) and 306 K (95% confidence interval 277–335 K), respectively. Monthly estimated parameters would improve the temporal variation component of the models, but there was too less data available for some months.

Empirical models derived from the chamber data were based on measured respiration and the corresponding soil temperature and water temperature. They were different for the three landscape elements within the average footprint area of the masts: CO_2 respiration rates were highest in the field and lowest in the ditches.
The parameters R_{10} and E_0 for each landscape element and their uncertainties were determined by non-linear regression using equation 4.3, see *Table 4.2.*

Table 4.2. *The three landscape elements with their proportional coverage within the footprint, average R10, a and b values and uncertainties (u) or 95% confidence intervals.*

		CO_2 respiration				CH_4 emission			
	Coverage	R_{10}	95% CI (R_{10})	E_0	95% CI (E_0)	a	u (a)	b	u (b)
Ditch	0.16	0.56	0.4-0.7	269	142-356	-0.75	0.48	0.19	0.03
Edge	0.05	4.66	3.9-6.1	306	208-421	0.37	0.38	0.12	0.03
Field	0.79	6.26	6.0-7.1	335	300-375	-1.03	0.19	0.07	0.02

Large-scale CH_4 flux measurements by EC were performed over a three-month period in 2006; they are described in Kroon et al. (2007). We used this data set to compare large-scale measurements with up-scaled small-scale measurements.
At the small-scale, CH_4 fluxes were based on the chamber measurements of 2005–2008. Temperature was found to be significantly related to emission of CH_4 and was used to fit an exponential function using equation 4.4. The parameters *a* and *b* and their uncertainties are given in *Table 4.2.* The magnitude of the fluxes depended on the landscape element: CH_4 emissions were highest from the ditches and lowest from the fields.

4.4.2. Comparison of EC and empirical models over larger temporal scales

The CO_2 respiration model and the CH_4 model were compared using two different scenarios. The first scenario is based on the model that considers the field only and in the second scenario fluxes are weighted with fixed coverage fractions for the three main landscape elements within the average footprint area of the masts (*Table 4.2.*). The CO_2 respiration rates and estimates of CH_4 fluxes from the peat meadow obtained using the weighted non-linear models based on chamber data (second scenario) agreed well with the EC fluxes (*Fig 4.2.* and *Fig 4.3.*, respectively) over one year and three months, respectively, while respiration rate estimates based on the model that considers the field (first scenario) did not. This emphasizes the importance of stratifying the landscape into landscape elements that contribute differently to the greenhouse gas emission.

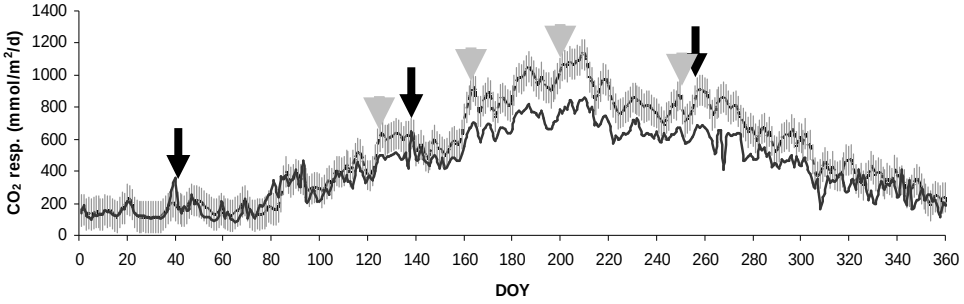


Fig 4.2. *Comparison of a model based on chamber measurements (dashed line) and respiration rates derived by EC (solid line) for 2006. The uncertainty band around the dotted line represents plus and minus one standard error calculated for for mean prediction, based on the regression analysis, and each day. Arrows indicate manure events (large) and mowing events (small).*

In the first scenario the difference between cumulative values for CO_2 respiration measured by EC and chamber based values was 31.0%, and in the second scenario the model for CO_2 respiration estimated 16.5% higher cumulative emissions compared to EC measurements: $188x10^{-3}$ mmol m^{-2} CO_2 versus $157x10^{-3}$ mmol m^{-2} CO_2, respectively (*Fig 4.4.*). In the second scenario, the largest differences occurred in August, September and October 2006: 20.8%, 24.4%, 21.9%, respectively, with lower fluxes measured by EC (*Fig 4.2.*), while the respiration rates in spring, early summer and winter months agreed well (within 10%).

In the first scenario, cumulative CH_4 fluxes over the three-month period estimated by the model were 55.1% lower compared to the EC fluxes and in the second scenario fluxes were 13.0% lower measured by EC: the estimates were 247.6 and 205.2 mg m^2 for chambers and EC, respectively (*Fig 4.4.*).

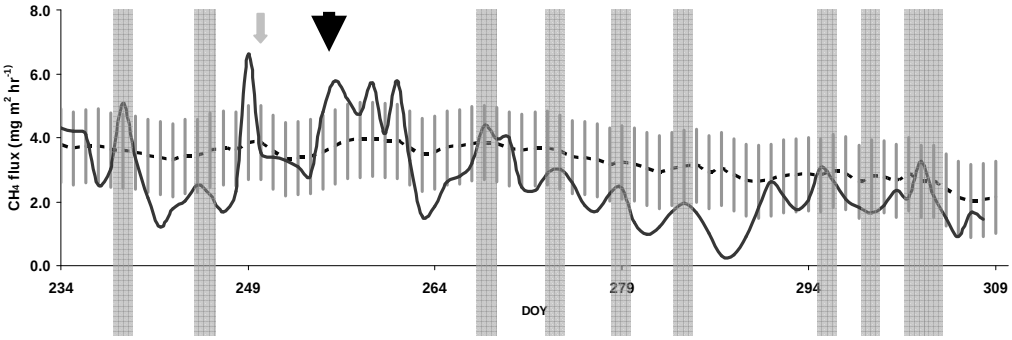


Fig 4.3. *Time series of measured solid line) and modeled (scenario 2, dashed line) daily averaged CH_4 fluxes during the period 21 August to 5 November, 2006. The uncertainty band around the dotted line represents plus and minus one standard error for mean prediction, based on the regression analyses, and calculated for each day. The bars represent days on which wind velocity exceeded 5 m s^{-1}; the large arrow represents a manure application event and the small arrow represents a mowing event.*

Higher emission rates for CO_2 based on chamber data compared with emissions measured by EC have been reported earlier in the literature (e.g. Norman et al., 1997; Jansen et al., 2001; Davidson et al., 2002; Reth et al., 2005). For example: Kabwe et al. (2005) found 12% higher fluxes measured by chambers in the summer. Twine et al. (2000) tested 9 EC

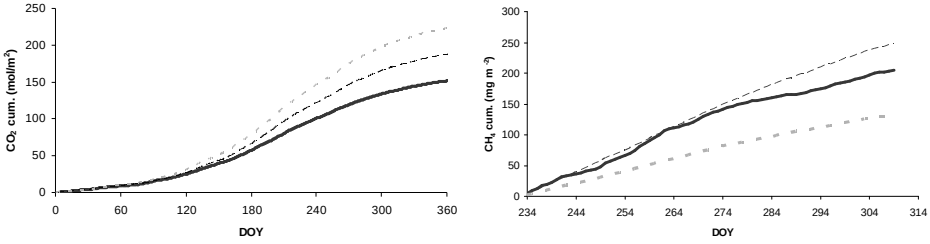


Fig 4.4. *Cumulative CO_2 respiration (left) and CH_4 emission (right) over one year and three months, respectively. The blue dashed line is the model based on chamber measurements and weighted for the landscape elements (scenario 2), the red solid line shows the EC results and the green dotted lines show the cumulative values based on upscaling from field measurements (scenario 1).*

systems for CO_2 and showed that surface fluxes measured by EC tend to be underestimated for a number of reasons, including mismatched sources of latent heat fluxes (*LE*) and sensible heat fluxes (*H*), inhomogeneous surface cover and soil characteristics, flux divergence or dispersion, non-stationarity of the flow, lack of a fully developed turbulent surface layer, flow distortion, sensor separation, topography and instrument error. In our

case, the energy balance was not fully closed (Veenendaal et al., 2007). This discrepancy is not unusual for EC measurements (e.g. Corradi et al., 2005; Nieveen et al., 2005) and might lead to underestimation of the actual fluxes (Twine et al., 2000). Further inter-comparison studies at different sites are needed.

4.4.3. Comparison of EC and empirical models on a daily basis

The daily averaged EC fluxes were compared with the daily averaged fluxes estimated by the empirical model, to validate the response to meteorological conditions and management. The temporal variation of the EC averaged daily fluxes was large, whereas the emission rates as estimated by the empirical model were smoothed. The difference in fluxes between the two methods was especially marked in periods of manure application: the chamber method did not follow the variation in CO_2 and CH_4 as measured by the EC systems (see also e.g. Reth et al., 2005). Neither did the chamber method capture the change in CO_2 respiration rates after mowing, whereas the EC system did.

Furthermore, the EC system sometimes measured increased CH_4 (*Fig 4.3.*) and CO_2 emissions at high wind velocity followed by a short period of reduced emissions. We corrected the EC fluxes for temperature and we found a significant positive correlation between these EC emission values of both gases and wind velocity at three meters (P<0.01). Other researchers (Sachs et al., 2008 and Wille et al., 2008) have recently also reported a correlation between horizontal wind speed and CH_4 fluxes and they suggested a possible underestimation by chambers. They stated that higher CH_4 fluxes from water bodies at high wind velocity may be due to turbulence induced ebullition and indicate a possible threshold of wind speed for the triggering of this process. Ebullition in water could also be triggered by changes of air pressure (Frokling and Crill, 1994). Besides, increased turbulence could flush out the CH_4 stored in the surface layer at non-turbulent periods at night (Hargreaves et al., 2001), but the total effect could perhaps be neglected over longer time spans because flushing and storage of gases in the vegetation layer or in the upper water layer could be in balance. More research is needed, especially in water bodies, to capture this flux variability and to also determine the influence of u^* on CH_4 fluxes from water bodies and soil.

4.4.4. Conclusion

Two independent methods for measuring CH_4 and CO_2 at large scale (EC) and small-scale (chambers) have been tested at different temporal scales in a heterogeneous landscape of fields and ditches. When only the field emissions are included, the difference between EC and up-scaled chamber-based cumulative emissions is 31.0% for CO_2 and 55.1% for CH_4.

However, when the representative landscape elements within the ecosystem are taken into account and a regression model is created for the different components, the EC and up-scaled chamber- based cumulative emissions agree well with 16.5% difference for CO_2 and 13.0% difference for CH_4. This difference can become even smaller if the regression models will be refined by e.g. management influences. To conclude, small-scale chamber measurements can thus certainly be used to estimate fluxes of CO_2 and CH_4 at landscape scale when all different landscape elements are taken into account.

4.5. Acknowledgements

This study was funded by Wageningen University, The Province of North Holland, CarboEurope IP (contract number: GOCE-CT-2003-505572) and the Klimaat voor Ruimte Project (BSIK). Many thanks to Jan van Walsem and Frans Möller for their help in the field and the lab and to Theo van Eijk and the van Eeuwijk family for their logistical support. We are grateful to the State Forestry Service for access to the site and to Nico de Bruin for his collaboration. Joy Burrough advised on the English.

4.6. References

Aubinet, M., Grelle, A., Ibrom, A., Rannik, U., Moncrieff, J., Foken, T., Kowalski, A.S., Martin, P.H., Berbigier, P., Bernhofer, C., Clement, R., Elbers, J., Granier, A., Grunwald, T., Morgenstern, K., Pilegaard, K., Rebmann, C., Snijders, W., Valentini, R. and Vesala T., 2000. Estimates for the annual net carbon and water exchange of forests. The EUROFLUX methodology. Adv. in Ecol. Res.: 30, 113-175.

Baldocchi, D., Falge, E., Gu, L.H., et al., 2001. FLUXNET: A new tool to study the temporal and spatial variability of ecosystem-scale carbon dioxide, water vapor, and energy flux densities. Bulletin of the American Meteorological Society 82, 2415–2434.

Baldocchi, D., 2003. Assessing the eddy covariance technique for evaluating carbon dioxide exchange rates of ecosystems: past, present and future. Glob. change Biol.: 9, 479-492.

Béziat, P., Ceschia, E., Dedieu, G., 2009. Carbon balance of a three crop succession over two cropland sites in South West France., Agric. Forest. Meteorol., 149, 1628–1645.

Businger, J. A., 1986. Evaluation of the accuracy with whch dry deposition can be measured with current meteorological techniques. Journal of Climate and Applied Meteorology 25, 100-1124.

Christensen, T.R., Jonasson, S., Callaghan, T.V., Havström, M., 1995. Spatial variation in high-latitude CH4 flux along a transect across Siberian and Eurasian tundra environments. J. of Geoph. Res.: 100, 21035-21045.

Conrad, R., 1989. Control of methane production in terrestrial ecosystems. In: exchange of trace gases between terrestrial ecosystems and the atmosphere: 39-58.

Corradi, C., Kolle, O., Walter. L., Zimov, S.A., Schulze, E.D, 2005. Carbon dioxide and methane exchange of a north-east Siberian tussock tundra. Glob. change Biol.: 11, 1-16.

Davidson, E.A., Savage, K., Bolstad, P., Clark, D.A., Curtis, P.S., Ellsworth, D.S., Hanson, P.J., Law, B.E., Luo, Y., Pregitzer, K.S., Randolph, J.C., Zak, D., 2002. Belowground carbon allocation in forests estimated from litter fall and IRGA-based soil respiration measurements, Agric. Forest Meteorol.: 113, 39-51.

Denmead O.T., 2008. Approaches to measuring fluxes of methane and nitrous oxide between landscapes and the atmosphere, Plant Soil: 309, 5-24.

Flechard, C.R., Ambus, P., Skiba, U., Rees, R.M., Hensen, A., van Amstel, A., van den Pol-van Dasselaar, A., et al., 2007. Effects of climate and management intensity on nitrous oxide enissions in grassland systems across Europe, Agric. Ecosyst. And Environm.: 121, 135-152.

Flechard, C.R., Neftel, A., Jochner, M., Ammann, C., Leifeld, J., Fuhgrer, J., 2007b. Temporal changes in soil pore space CO_2 concentration and storage under permanent grassland. Agric. Forest. Meteorol.: 142, 66-84.

Foken, Th., and Wichura, B., 1996. Tools for quality assessment of surface-based flux measurements. Agric. Forest Meteorol.: 78, 83-105.

Frolking, S., Crill, P, 1994. Climate controls on temporal variability of methane flux from a poor fen in southeastern New-Hampshire – measurement and modeling. Global biogeochem. Cycl. Vol. 8 issue 4. 385-397.

Grelle A., Lindroth, A., 1996. Eddy-correlation systems for long-term monitoring of fluxes of heat, water vapour and CO2, Glob. Change Biol.: 2, 297-307.

Hargreaves, K.J., Fowler, D., Pitcairn, C.E.R., et al., 2001. Annual methane emission from Finnish mires estimated from eddy covariance campaign measurements, Theor. Appl. Climatol., 70(1- 4), 203-213.

Hendriks, D. M. D., van Huissteden, J., Dolman, A.J., van der Molen, M.K., 2007. The full greenhouse gas balance of an abandoned peat meadow. Biogeosciences Discussions: 4, 277-316.

Hendriks, D.M.D., Dolman, A.J., van der Molen, M.K., van Huissteden, J., 2008. A compact and stable eddy covariance set-up for methane measurements using off-axis integrated cavity output spectroscopy. Atmos. Chem. Phys., 8, 1-13.

Hollinger, D.Y., Kelliher, F.M., Byers,J.N., Hunt J.E., Mcseveny, T.M. and Weir P.L.., 1994. Carbon dioxide exchange between an undisturbed old-growth temperate forest and the atmosphere. Ecology: 75, 134-150.

Hutchinson, G.L., Livingston, G.P., 2002. Soil-atmosphere gas exchange. In : J.H. Dane and G.C. Topp, Editors, Methods of soil analysis. Part 4. Physical methods, SSSA Book Series, no. 5, Soil science Society of America, Madison, WI, 1159-1182.

IPCC 2007a: The Physical Science Basis. Contributions of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change: 996, Cambridge University Press, Cambridge, UK.

Jansen, I.A., Kowalski, A.S., Ceulemans, R.,2001. Forest Floor CO2 fluxes estimated by eddy covariance and chamber based model. Agric. Forest Meteorol.: 106(1), 61-69.

Kabwe, L.K., Hendry, M.J., Wilson, G.W., Lawrence, J.R., 2005. Characterizing spatial and temporal variations in CO2 fluxes from ground surface using three complimentary measurement techniques. J. of Hydrology.: 311: 80-90.

Kormann, R., Meixner, F.X., 2001. An analytical footprint model for neutral stratification. Boudary-Layer Meteorology: 99, 207-224.

Kroon, P.S., Hensen, A., Zahniser, M.S., van t' Veen, W.H., Vermeulen, A.T., 2007. Suitability of quantum cascade laser spectrometry for CH4 and N2O eddy covariance measurements. Biogeosciences: 4, 715-728.

Kroon, P.S., Hensen, A., Jonker, H.J.J., Ouwersloot, H.G., Vermeulen, A.T., Bosveld, F.C., 2009 this issue. Uncertainties in eddy covariance flux measurements assessed form CH4 and N2O observations, Agric. Forest. Meteorol. doi:10.1016/j.agrformet.2009.08.008.

Kroon, P.S., Schuitmaker, A., Jonker, H.J.J.,, Tummers, M.J., Hensen, A., Bosveld, F.C., 2009 this issue. An evaluation by laser Doppler anemometry of the correction algorithm base don Kaimal co-spectra for high frequency losses of EC flux measurements of CH_4 and N_2O, Agric. Forest. Meteorol., doi:10.1016/j.agrformet.2009.09.009.

Kroon, P.S., Schrier-Uijl, A.P., Hensen, A., Veenendaal., E.M., Jonker, H.J.J., submitted, Annual balances of CH_4 and N_2O from a managed fen meadow using eddy covariance flux measurements. Eur. J. Soil Sci.

Kroon, P.S., Hensen, A., van den Bulk, W.C.M., Jongejan, P.A.C., Vermeulen, A.T., 2008. The importance of reducing the systematic error due to non-linearity in N2O flux measurements by static chambers. Nutr. Cycl. Agroecosyst.: 82, 175-186, DOI 10.1007/s10705-008-9179-x

Kutzbach, L., Wagner, D., Pfeiffer, E.M., 2004. Effect of micro relief and vegetation on methane emission from wet polygonal tundra, Lena Delta, Northern Siberia. Biogeochemistry: 69, 341-362.

Laurila, T., Tuovinen, J. P., Lohila, A., Hatakka, J., Aurela, M., Thum, T., Pihlatie, M., Rinne, J. and Vesala, T., 2005. Measuring methane emission rates from a landfill using a cost-effective micrometeorological method. Geophysical Research Letters, 32, L19808, doi:10.1029/2005GL023462, 2005

Lloyd, J., Taylor, J.A., 1994. On the temperature dependency of soils. Functional Ecology: 8, 315-323.

McMillen, R.T., 1988. An eddy correlation technique with extended applicability to non-simple terrain. Boundary-Layer Meteorology 43, 231–245.

Moore, C.J., 1986. Frequency response corrections for eddy correlation systems. Boundary-Layer Meteorology: 37, 17-35.

Neftel, A., Flechard, C., Ammann, C., Conen, F, Emmenegger, L., Zeyer, K., 2007. Experimental assessment of N2O background fluxes in grassland systems. Tellus B: 59, 470-482.

Nieveen, J.P., Campbell, D.I., Schipper, L.A., Blair, I.J, 2005. Carbon exchange of grazed pasture on a drained peat soil. Glob. change Biol.: vol 11, 4, 607-618.

Nol, L., Verburg, P.H., Heuvelink, G.B.M., Molenaar, K., 2008. Effect of land cover data on nitrous oxide inventory in fen meadows. J. of Envir. Qual.: 37, 1209-1219

Norman, J.M., Kucharik, C.J., Gower, S.T., Baldocchi, D.D., Crill, P.M., Rayment, M., Savage, K., Striegel, R.G., 1997. A comparison of six methods for measuring soil-surface carbon dioxide fluxes. J. Geophys. Res.: 102 28, 771-777.

Reichstein, M., Falge, E., Baldocchi, D., Papale, D., Aubinet, M., Berbigier, P., Bernhofer, C., Buchmann, N., Gilmanov,T., Granier, A., Grunwald, T., Havrankova, K., Ilvesniemi, H.,Janous, D., Knohl, A., Laurila, T., Lohila, A., Loustau, D., Matteucci,G., Meyers, T., Miglietta, F., Ourcival, J. M., Pumpanen, J., Rambal, S., Rotenberg, E., Sanz, M., Tenhunen, J., Seufert,G., Vaccari, F., Vesala, T., Yakir, D., Valentini, R., 2005. On the separation of net ecosystem exchange into assimilation and ecosystem respiration: review and improved algorithm, Glob. Change Biol.,11, 1424–1439.

Reth, S., Göckede, M., Falge, E., 2005. CO2 efflux from Agricultural soils in Eastern Germany – comparison of a closed chamber system with eddy covariance measurements, Theoretical and Applied Climatol.: 80, 105-120.

Rinne, J., Riutta, T., Pihlatie, M., Aurela, M., Haapanala, S., Tuovinen, J.-P., Tuittila, E.-S., and Vesala, T., 2007. Annual cycle of methane emission from a boreal fen measured by the eddy covariance technique. Tellus 59B, 449-457

Riutta, T., Laine, J., Aurela, M., Rinne, J., Vesala, T., Laurila, T., Haapanala, S., Pihlatie, M., Tuittila, E. S., 2007. Spatial variation in plant community functions regulates carbon gas dynamics in a boreal fen ecosystem. Tellus 59B, 838-852. doi: 10.1111/j.1600-0889.2007.00302.x

Rochette, P., Eriksen-Hamel, N.S., 2008. Chamber Measurements of Soil Nitrous Oxide Flux: Are Absolute Values Reliable? Soil Sci. Soc. Am. J.: 72, 331-342.

Sachs, T., Wille, C., Boike, J., Kutzbach, L., 2008. Environmental controls on ecosystem-scale CH4 emission from polygonal tundra in the Lena River Delta, Siberia, J. Geophys. Res.: 113, G00A03, doi:10.1029/2007JG000505.

Schrier-Uijl, A.P., Veenendaal, E.M., Leffelaar, P.A., van Huissteden, J.C., Berendse, F., 2009. Methane emissions in two drained peat agro-ecosystems with high and low agricultural intensity. Plant Soil, doi:10.1007/s11104-009-0180-1.

Twine T.E., Kustas, W.P., Norman, J.M., Cook, D.R., Houser, P.R., Meyers, T.P., Prueger, J.H., Starks, P.J., Wesely, M.L., 2000. correcting eddy-covariance flux underestimates over a grassland, Agric. Forest Meteorol.: 103, 279-300.

Veenendaal, E. M., Kolle, O., Leffelaar, P. A., Schrier-Uijl, A. P., Van Huissteden, J., Van Walsem, J., Möller, F., and Berendse, F., 2007. CO2 exchange and carbon balance in two grassland sites on eutrophic drained peat soils, Biogeosciences: 4, 1027-1040.

Webb, E.K., Pearman, G.I., and Leuning, R., 1980. Correction of flux measurements for density effects due to heat and water vapour transfer. Quarterly Journal of the Royal Meteorological Society: 106, 85-100.

Wille, C., Kutzbach, L., Sachs, T., Wagner, D., Pfeiffer, E.M., 2008. Methane emission from Siberian arctic polygonal tundra: eddy covariance measurements and modeling, Glob. Change Biol.: 14, 1395-1408.

Chapter 5

Annual balances of CH_4 and N_2O from a managed fen meadow using eddy covariance flux measurements

Annual terrestrial balances of methane (CH_4) and nitrous oxide (N_2O) are presented for a managed fen meadow in the Netherlands for 2006, 2007 and 2008 using eddy covariance (EC) flux measurements. Annual emissions derived from different methods are compared. The most accurate annual CH_4 flux is achieved by gap filling EC fluxes with an empirical multivariate regression model with soil temperature and mean wind velocity as driving variables. This model explains about 60% of the variability in observed daily CH_4 fluxes. Annual N_2O emissions can be separated into background emissions and event emissions due to fertilization. The background emission is estimated using a multivariate regression model also based on EC flux data with soil temperature and mean wind velocity as driving variables. The event emissions are estimated using emission factors. The minimum direct emission factor is derived for six fertilization events with subtracting the background emission and the IPCC default emission factor of 1% is used for the other events. In addition, the maximum direct emission factors are determined for the six events without subtracting the background emission. The average direct emission factor ranges from 1.2 to 2.8% which is larger than the IPCC default value. Finally, the total terrestrial greenhouse gas balance is estimated at 16 Mg ha^{-1} $year^{-1}$ in CO_2-equivalents with contributions of 30%, 25% and 45% by CO_2, CH_4 and N_2O, respectively.

Based on P. S. Kroon, A. P. Schrier-Uijl, A.Hensen, E. M. Veenendaal en H. J. J. Jonker
European Journal of Soil Science (2010), Vol 61.

5.1. Introduction

Peatland ecosystems cover approximately 3% of the global land surface and have evolved as globally important sinks of atmospheric carbon dioxide (CO_2) since the last ice age. This is the result of their ability to accumulate more organic matter through photosynthesis than is released through respiration, mainly as a result of high water tables and therefore anaerobic conditions. The carbon (C) accumulated in peatlands is equivalent to almost half the total atmospheric content (Drösler *et al.*, 2008). Peatlands have helped to remove significant amounts of CO_2 from the atmosphere over the last 10,000 years. However, peatlands are not just acting as a sink for CO_2. The wet and anaerobic conditions can lead to formation of the important greenhouse gas methane (CH_4).

Over the last few centuries, many peatland areas have been converted into agricultural land by artificially lowering the water table. As a result peatlands have become a strong source of CO_2 and nitrous oxide (N_2O) due to an increase in peat oxidation and fertilizer application (Schothorst, 1977; Langeveld *et al.*, 1997). Peatland restoration by elevating the water level and decreasing the agricultural intensity may return peatland areas to a sink of CO_2 and also modify the emissions of CH_4 and N_2O (e.g., Van den Bos *et al.*, 2003). All three greenhouse gases should thus be taken into account to assess the total effect of restoration on the greenhouse gas (GHG) balance.

In order to compile the full GHG balance and to understand the processes that affect this balance long term (multiple year) measurements are needed covering all three species. There are only a few studies published with annual records of CO_2, CH_4 and N_2O in peatlands drained for agriculture (e.g., Maljanen *et al.*, 2003; Regina *et al.*, 2004; Regina *et al.*, 2007; Veenendaal *et al.*, 2007). Most studies only assess annual emissions of CO_2 (e.g., Veenendaal *et al.*, 2007), and the published annual estimates of CH_4 and N_2O have uncertainties that can exceed 50% (e.g., Flechard *et al.*, 2007a). The relatively large uncertainties in CH_4 and N_2O are due to a combination of complexity of the source (i.e. spatial and temporal variation), limitations in the measurement equipment and the methodology used to quantify the emissions. High-frequency micrometeorological methods are a good candidate to determine integrated emission estimates on a hectare scale that also have continuous time coverage. Instrumentation has recently been proven to be suitable for CH_4 and N_2O eddy covariance (EC) flux measurements (e.g., Eugster *et al.*, 2007; Kroon *et al.*, 2007; Neftel *et al.*, 2007; Hendriks *et al.*, 2008).

The main objective of this study is to obtain annual terrestrial CH_4 and N_2O balances from an intensively managed site on peat grassland in the Netherlands over 2006, 2007 and 2008 using EC flux measurements. The terrestrial emission estimates include emissions from field, ditch and ditch edges. Different methods are compared to estimate the annual emissions. The magnitude and the uncertainty in the annual terrestrial CH_4 emission

measured by EC are compared to the annual emission based on static chamber measurements (Schrier-Uijl *et al.*, 2009a). In addition, the direct emission factor EF_1 (ratio of emitted N_2O-N to applied nitrogen (N)) is compared with the default IPCC value of 1% (IPCC, 2006). Finally, a first estimate of the total terrestrial GHG balance is made using the CH_4 and N_2O balances derived in this study and the CO_2 annual balance given in Veenendaal *et al.* (2007). This GHG balance is compared to the terrestrial GHG emissions measured at other peatlands.

5.2. Experimental site and climatic conditions

The measurements were performed at an intensively managed dairy farm area. This farm is located at Oukoop near the town of Reeuwijk in the Netherlands (52º 02' 11" N, 4º 46' 49" E). The site has peat soil with a clayey peat or peaty clay surface layer of about 0.25 m on 12 m eutrophic peat deposits. Average C and N content in the top 0.20 m of the soils are 24% and 2.4%, respectively (Veenendaal *et al.*, 2007). Rye grass (*Lolium perenne*) is the most dominant grass species with often co-dominant rough bluegrass (*Poa trivialis*) (Veenendaal *et al.*, 2007). The site consists of three main landscape elements according to micro topography and soil moisture conditions: permanently water-filled ditches, almost saturated ditch edges and the relatively dry field area. Ditches, ditch edges and fields account for 16%, 5% and 79% to the average footprint area of the EC flux system (Schrier-Uijl, *et al.*, 2009b). The contribution over the three different landscape elements is approximately equal for all wind directions. The mean elevation of the polder is between 1.6 and 1.8 m below sea level (ASL). The ditch water level in the polder is being kept at -2.39 in winter and -2.31 m ASL in summer (Veenendaal *et al.*, 2007). The groundwater depth varies from 0.70 to 0.15 m below field level and perched water tables occur after heavy rain.

Table 5.1. *Climatic and management characteristics of Oukoop site in the Netherlands*

	2006	2007	2008
Mean annual temperature at 10 m /ºC[b]	11.1	11.1	10.6
Mean WFPS[c] at 10 cm depth /%	83	91	88
Mean WFPS[c] at 30 cm depth /%	97	98	99
Annual precipitation /mm[b]	767	1087	736
Cow manure application /kg N ha^{-1}	253	262	414[a]
Artificial fertilizer application /kg N ha^{-1}	100	95	70

[a]Farmer stopped the agricultural activities and emptied slurry storage. [b]Temperature and precipitation data were made available by KNMI. [c] WFPS means water-filled pore space.

Cow manure and artificial fertilizer were applied about two to six times a year in the period from February to October. There were four harvest events each year. In 2008 the fertilization protocol was adapted from October since the farmer stopped the agricultural

activities at this peatland. A summary of the main climatic and management characteristics of the site is given in *Table 5.1*.

5.3. Instrumentation and methodology

5.3.1. Instrumentation

Two separate EC systems were used, one was used for CO_2 and latent heat fluxes (Veenendaal *et al.*, 2007), and the other one with a quantum cascade laser (QCL) for CH_4 and N_2O fluxes (Kroon *et al.*, 2007). Both masts of 3 m height were positioned about 5 m apart in the middle of the field. Terrain around the towers was flat and free of obstruction for at least 600 m in all directions, except for the 2x2x2 m container positioned downwind in which the QCL spectrometer was placed. CO_2 and latent heat fluxes were derived from an EC system consisting of a Campbell SCAT C3 sonic anemometer (Campbell Scientific. Logan. Utah., USA) and a Licor 7500 open path Infrared gas analyzer (LICOR Lincoln, NE, USA). Wind speed, air temperature, CH_4 and N_2O concentrations were measured with a system consisting of a three-dimensional sonic anemometer and a QCL spectrometer (model QCL-TILDAS-76, Aerodyne Research Inc., Billerica MA, USA).
Two sonic anemometer types were used, a WMPRO (Gill Instruments, Lymington, UK) from 26 October 2007 to 9 July 2008 and during the rest of the measurement period an R3 (Gill Instruments, Lymington, UK). The sonic anemometer data and the QCL spectrometer output were logged using the RS232 output and processed using a data acquisition program developed at ECN, following the procedures of McMillen (1988). A more detailed explanation of this measurement set-up can be found in Kroon *et al.* (2007).
Soil measurement sensors included soil heat flux plates (HPF01, Campbell Scientific, USA), soil temperature sensors at depths of 0.02, 0.04, 0.08, 0.16 and 0.32 m (Campbell Scientific, USA) and soil moisture probes by volume at depths of 0.10, 0.20 and 0.30 m (Theta probes ML 2x; Delta T devices Burwell, UK). Additional micrometeorological observations were used from the Cabauw site which is located about 14 km from Oukoop. These data were made available by the Royal Netherlands Micrometeorological Institute (KNMI).

5.3.2. Methodology

5.3.2.1. Emission measurements

The expression for the net ecosystem exchange of CH_4 (F_{CH4}) and N_2O (F_{N2O}) is derived from the tracer conservation equation (e.g., Aubinet *et al.*, 2000). Assuming horizontal homogeneity and a flat terrain within the averaging time of 30 minutes, the net ecosystem exchange F_{wc} consists of two contributions, the storage term St_c and the eddy covariance flux term EC_{wc}, and is given by

$$\underbrace{\int_0^h \overline{S}_c dz}_{F_{wc}} = \underbrace{\frac{\overline{c}_i - \overline{c}_{i-1}}{T_{\mathrm{av}}} h}_{\mathrm{St}_c} + \underbrace{\overline{w'c'}|_{z=h}}_{\mathrm{EC}_{wc}} \qquad \textbf{5.1}$$

where h is the measurement height in m, S_c the source/sink term in ppb s^{-1}, c the gas concentration in ppb, i the flux number, T_{av} the averaging time in s and w the vertical wind velocity in m s^{-1}. We calculate the storage term St_c using the average values of CH_4 and N_2O at 3 m height over each 30 minutes period.
We determine the EC flux EC_{wc} by

$$\mathrm{EC}_{wc} = \chi_{\mathrm{cal}} \chi_{\mathrm{res}} \mathrm{EC}_{wc}^{\mathrm{meas}} + \chi_{\mathrm{cal}} \chi_{\mathrm{Webb}} \qquad \textbf{5.2}$$

With $\mathrm{EC}_{wc}^{\mathrm{meas}}$ the measured EC flux in ppb m s^{-1}, χ_{cal} the calibration correction, χ_{res} the frequency response correction and χ_{web} the Webb-correction. More information about the correction process is given in Kroon *et al.* (2009a, b).

5.3.2.2. Annual terrestrial balances

The annual terrestrial emission of CH_4 E_{CH4} in kg CH_4 ha^{-1} $year^{-1}$ is given by

$$E_{\mathrm{CH4}} = \left(\frac{16}{12}\right)\left(\frac{1x10^4}{1x10^{12}}\right)\sum_{i=1}^{N} F_{\mathrm{CH4}_i} T_{\mathrm{av}} \qquad \textbf{5.3}$$

where F_{CH4} is the emission in ng C m^{-2} s^{-1}, the factor (16/12) is the conversion from kg C to kg CH_4, the factor $1x10^4$ is used for the conversion from m^{-2} to ha^{-1}, the factor $1x10^{12}$ for the conversion from ng to kg and N represents the amount of time slots within a year which equals to $T_{\mathrm{year}}/T_{\mathrm{av}}$.
Below we distinguish three methods to deal with data gaps in CH_4 flux data. Method 1 is based on the average CH_4 EC flux where the average flux is extrapolated to an annual

value, in other words the data gaps are filled with the average emission. Method 2 uses all available EC flux measurements and the remaining data gaps are filled by an empirical multivariate regression. Method 3 is totally based on the empirical regression. We derive this regression using the statistical method ANOVA.

The annual terrestrial emission of N_2O E_{N2O} in kg N_2O ha^{-1} $year^{-1}$ consists of a background emission level on which emission peaks are superimposed. These peaks are related to manure and fertilizer application (direct agricultural soil emission). The background emission level derived from the EC data includes a component that is caused by atmospheric N deposition. Outside the measurement fetch indirect N_2O emission is caused by nitrogen emitted from the field as NH_3 or N leached to the ditches-lakes-rivers and finally to the sea. These two processes should be taken into account in the full terrestrial N_2O balance.

Consequently, the annual emission of N_2O E_{N2O} in kg N_2O ha^{-1} $year^{-1}$ is given by

$$E_{N2O} = E_{EC} + E_l + E_d \qquad \textbf{5.4.}$$

with E_{EC} the emissions measured by the EC flux technique, E_l the indirect emissions due to leaching and runoff and E_d the indirect emissions due to deposition. The contribution E_l is estimated using the IPCC methodology (IPCC, 2006)

$$E_l = \left(\frac{44}{28}\right)\left(N_{synth} + N_{cow}\right)\mathrm{Frac}_{leach}\,\mathrm{EF}_5 \qquad \textbf{5.5}$$

where the factor (44/28) is used for the conversion from kg N to kg N_2O and N_{synth} the annual amount of synthetic fertilizer in kg N ha^{-1} $year^{-1}$, N_{cow} the annual amount of applied cow manure in kg N ha^{-1} $year^{-1}$, Frac_{leach}=0.3 and EF_5=0.75%. The contribution E_d is partly based on the IPCC methodology (IPCC, 2006)

$$E_d = \left(\frac{44}{28}\right)\left(\left(\sum_{m=1}^{N_{event}} N_{fert_m}\left(1-k_m\right)\right) - \overline{N}_d\right)\mathrm{EF}_4 \qquad \textbf{5.6}$$

with N_{event} the number of fertilization events, N_{fert} the amount of applied fertilizer in kg N ha^{-1}, k is equal to 0.9 for synthetic fertilizer or 0.8 for organic fertilizer (IPCC, 2006), $\overline{N}_d$ is the average annual N deposition in the Netherlands which equals to 30.8 kg N ha^{-1} $year^{-1}$ (Ruiter *et al.*, 2006) and EF_4=1% (IPCC, 2006).

The annual emission of N_2O E_{EC} can be derived in a similar way as the annual emission of CH_4 when no data gaps are available. The annual emission is then given by

$$E_{EC} = \left(\frac{44}{28}\right)\left(\frac{1x10^4}{1x10^{12}}\right)\sum_{i=1}^{N} F_{N2O_i} T_{av} \qquad \textbf{5.7}$$

where F_{N2O} is the N_2O emission in ng N m^{-2} s^{-1}.

However, the data coverage is hardly ever 100%. In that case, we distinguish two contributions in the annual N_2O emission E_{EC}, the background emissions E_{bgnd} and the emissions due to fertilizer application E_{fert}. The background emission is defined as the N_2O emission which occurs when no fertilizer is applied. We estimate the background fluxes F_{bgnd} using a multivariate regression model based on all EC fluxes excluding EC fluxes measured around a management event. This background emission is probably larger than the true background emission (unmanaged situation) since it will include emissions from the accumulated N in the soil due to previous fertilization events. The background emission level also includes the emission due to atmospheric N deposition on the field. Therefore, we use an adapted direct emission factor EF_1^{min} for estimating the direct emissions due to fertilizing application. This factor is calculated by

$$EF_1^{min} = \left(\frac{1x10^4}{1x10^{12}}\right)\frac{\sum_{i=1}^{N}\left(F_{N20_i} - F_{bgnd_i}\right)T_{av}}{kN_{fert}} x100\% \qquad \textbf{5.8}$$

with N the amount of time slots within a fertilization event which equals to T_{event}/T_{av}, N_{fert} the amount of applied fertilizer in kg N ha^{-1} and k is equal to 0.9 for synthetic fertilizer or 0.8 for organic fertilizer (IPCC, 2006). The factor k is used for adjusting to account for the amount of N that volatilizes as NH_3 and NO_x (IPCC, 2001a; IPCC, 2006). The total annual N_2O emission E_{EC} is then derived from

$$E_{EC} = \left(\frac{44}{28}\right)\left(E_{bgnd} + E_{fert}\right) \qquad \textbf{5.9}$$

where the factor (44/28) is used for the conversion from kg N to kg N_2O.The background emission E_{bgnd} in kg N ha^{-1} $year^{-1}$ is

$$E_{bgnd} = \left(\frac{1x10^4}{1x10^{12}}\right)\sum_{i=1}^{N} F_{bgnd_i} T_{av} \qquad \textbf{5.10}$$

where N represents the amount of time slots within a year which equals to T_{year}/T_{av} and the fertilizer emission E_{fert} in kg N ha^{-1} $year^{-1}$ is

$$E_{\text{fert}} = E_{\text{direct}} = \sum_{m=1}^{N_{\text{event}}} \text{EF}_{1_m}^{\text{min}} k_m N_{\text{fert}_m} \qquad \textbf{5.11}$$

where E_{direct} indicates the direct N_2O emissions due to fertilizing and N_{event} is the number of fertilization events.
However, we realize that the emission factor EF_1 used in the inventory reports is higher than EF_1^{min} because the real background emission is between 0 ng N m^{-2} s^{-1} and the measured background emission. To check the reliability of the IPCC default value of 1%, we compare this default value with an emission factor range which is set from EF_1^{min} to EF_1^{max}, where EF_1^{max} is determined by

$$\text{EF}_1^{\text{max}} = \left(\frac{1x10^4}{1x10^{12}} \right) \frac{\sum_{i=1}^{N} F_{\text{N20}_i} T_{\text{av}}}{kN_{\text{fert}}} x100\% \qquad \textbf{5.12}$$

with N the amount of time slots within a fertilization event which equals to $T_{\text{event}}/T_{\text{av}}$
In this study, we use three methods to derive the annual terrestrial N_2O emission E_{N2O}. Method 1 uses the average measured N_2O flux over the available EC fluxes within a year and this value is extrapolated to an annual E_{EC} value and the indirect annual emissions E_{l} and E_{d} are added. In method 2, we first determine for each day number, i.e. from number 1 to 365, the average N_2O emission $\overline{F}_{\text{N2O}}$ over the three years, while the data gaps are filled by the last available daily average. The annual emission E_{EC} is then calculated using equation 5.7, and the contributions E_{l} and E_{d} are again added. Method 3 uses equation 5.9 for E_{EC} where we determine the emission factor EF_1^{min} for each available measured fertilization event with equation 5.8 and we use an emission factor of 1% for the remaining fertilization events. The total terrestrial N_2O emission is again obtained by adding E_{EC} to E_{l} and E_{d}. In addition, we compare the direct emission factors to the 1% factor to check whether it is reasonable to use this value for the events which are not available.

We finally derive the total terrestrial GHG emission E_{GHG} in kg CO_2-equivalents ha^{-1} $year^{-1}$ by

$$E_{\text{GHG}} = E_{\text{CO2}} + 25E_{\text{CH4}} + 298E_{\text{N2O}} \qquad \textbf{5.13}$$

with E_{CO2} the annual terrestrial CO_2 emission in kg CO_2 ha^{-1} $year^{-1}$. The factor 25 and 298 are the global warming potential of CH_4 and N_2O over a 100 year time horizon, respectively, in relation to CO_2 (IPCC, 2007).

5.4. Results and discussion

5.4.1. Data coverage

The EC flux measurement set-up was installed for one week in February 2006, and from April 2006 to October 2008. The data coverage is 48% over the measurement periods (N=20,957 30 minutes EC fluxes); data losses occurred among others due to problems with the QCL (28%), computer (9%), automatic liquid nitrogen filling system (6%) and sonic anemometer (2%). The relatively high data losses of the QCL were due to the development stage of the QCL at the beginning of the measurement period. The footprint check and non steady state tests result in data rejection of 19% and 23% for CH_4 and N_2O, respectively. We correct the EC fluxes for systematic errors using equation 5.2. The total CH_4 and N_2O flux F_{wc} are determined by adding the storage term to the EC flux for each 30 minutes period (Eq. 5.1).

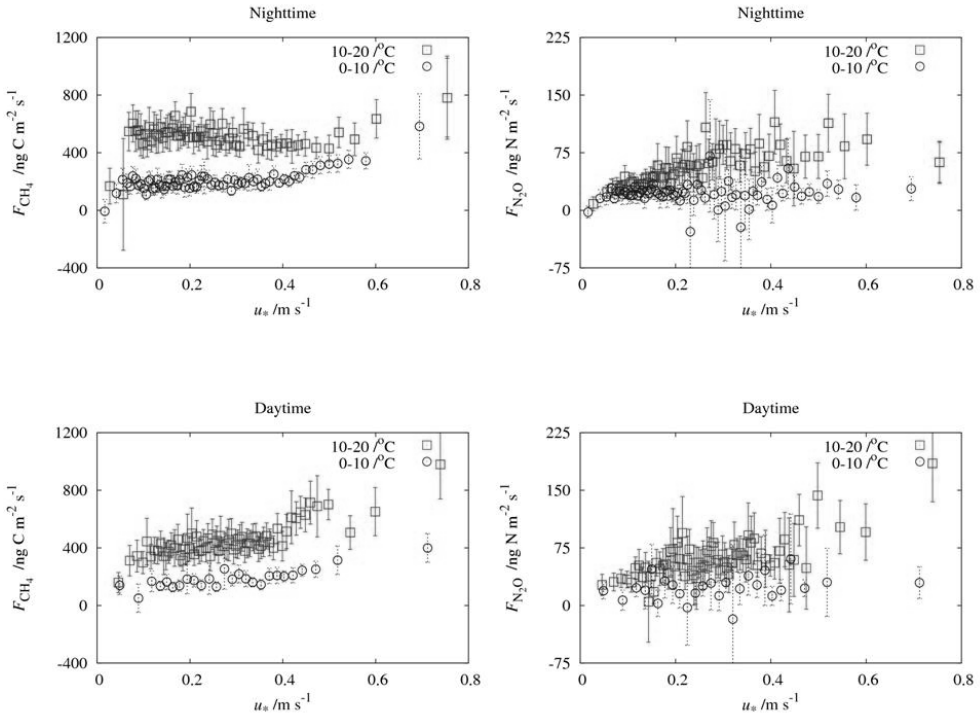


Fig 5.1. *Average CH_4 fluxes (left) and N_2O fluxes (right) as function of friction velocity u_*. Data is split into night- and daytime, and in two temperature classes. The data is pooled into groups of 50 fluxes each. Error bars show the uncertainty in the average CH_4 and N_2O flux given by the standard error* $\mathrm{SE} = 2\sigma / \sqrt{N}$.

It is often stated that the measured total flux can be underestimated under conditions of low turbulence. A common procedure for CO_2 fluxes is to investigate the effect of the friction velocity u^* on the measured nighttime CO_2 fluxes. CO_2 fluxes are then rejected below a threshold u^* for which the average fluxes decrease significantly (e.g., Nieveen *et al.*, 2005; Wohlfahrt *et al.*, 2005; Veenendaal *et al.*, 2007). All these studies were focused on nighttime fluxes.

In order to compare the night- and daytime u^*-dependence of CH_4 and N_2O fluxes, we select the data with shortwave radiation of 0 W m^{-2} for nighttime and shortwave radiation larger than 50 W m^{-2} for daytime. The fluxes are additionally divided into groups, with air temperature between 0 and 10 ºC, and 10 and 20 ºC. The fluxes in each group are sorted by u^* and pooled into groups of 50 fluxes each. This leads to a similar behavior for night- and daytime u^*-dependence of CH_4 and N_2O (*Fig 5.1.*). The fluxes show a marked decrease for $u^* < 0.07$ m s^{-1} which is comparable with the threshold $u^* = 0.1$ m s^{-1} of CO_2 fluxes measured at the same site (Veenendaal *et al.*, 2007). For $u^* > 0.45$ m s^{-1}, the average CH_4 and N_2O EC fluxes show a small increase which is not caused by higher temperatures.

When dividing the fluxes into air temperature classes of 2 ºC and three soil moisture classes, always a similar pattern is found for which the fluxes decrease for $u^* < 0.07$ m s^{-1} and increase for $u^* > 0.45$ m s^{-1}. Gu et al. (2005) suggested that there are two thresholds for CO2 nighttime fluxes, u^*_L and u^*_H delimiting three zones in the u^*-range, with a decrease of flux below u^*_L and an increase above u^*_H. Fluxes below u^*_L would be lower due to storage increase and fluxes above u^*_H are higher due to pressure pumping effects (Gu et al., 2005; Flechard et al., 2007b). Flechard et al. (2007b) also showed that the addition of soil storage change to the flux removed a large fraction of the u^*-dependence which suggests that no u^*-filtering should be applied to avoid possible double-counting. Therefore no u^*-filtering is applied on the EC flux data set in this study.

Another point of concern is related to the uncertainty in a 30 minutes CH_4 and N_2O EC flux which can be larger than the flux itself (Kroon *et al.*, 2009a). This uncertainty is mainly caused by the random one-point uncertainty (e.g., Businger, 1986). To reduce the uncertainty, averages over longer time span than 30 minutes can be used since the random uncertainty decreases with the number of 30 minutes EC fluxes. Therefore, we assess the suitability of using daily average emissions instead of 30 minutes F_{wc} values. It is justified to use daily average emissions when the data coverage is 100%, when there is no diurnal variation or when the number of night and day fluxes balances.

Several studies investigated the diurnal cycle of CH_4 and N_2O emissions e.g., Mikkelä *et al.*, 1995; Duan *et al.*, 2005; Hendriks *et al.*, 2008 for CH_4 and e.g., Skiba *et al.*, 1996; Crill *et al.*, 2000; Maljanen *et al.*, 2002 for N_2O. For both gases some studies have observed a diurnal cycle and some studies have not. For CH_4, the diurnal pattern is maybe dependent on the vegetation type (e.g., Mikkelä *et al.*, 1995; Duan *et al.*, 2005). In this study, we

check the diurnal cycles using the whole data set *(Fig 5.2.*). A diurnal pattern is observed for CH_4, with larger CH_4 fluxes in the late afternoon, while no clear pattern is found for N_2O. Consequently, we evaluate further the diurnal variation of N_2O for days with high emissions or high temperatures; however, we still did not observe a clear diurnal pattern. The average soil temperature, in *Fig 5.2.*, shows the same diurnal pattern as CH_4 supporting that soil temperature is an important driver for CH_4 fluxes which is also found in other studies (e.g., Hendriks *et al.*, 2007). The absence of a clear diurnal pattern for N_2O allows for the use of daily average values for N_2O for further analyses. For CH_4, however, daily averages can only be used when balanced numbers of day and night observations are available, which is valid in this study.

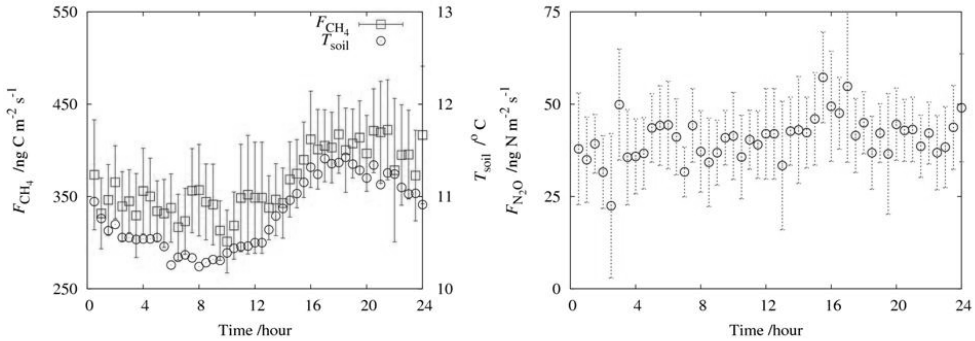


Fig 5.2. *Diurnal pattern of CH_4 fluxes and soil temperature (left) and N_2O fluxes (right) at a peat area in the Netherlands. Each point represents the average of about N=320 30 minutes EC fluxes for CH_4 and N_2O, and the average of about N=1000 30 minutes T_{soil} values. The error bars indicate the uncertainty in the average given by the standard error* $\mathrm{SE} = 2\sigma/\sqrt{N}$.

Consequently, we calculate daily averages for days with more than 12 30 minutes EC fluxes balanced over day and night which lead to 119, 106 and 188 daily CH_4 fluxes and 117, 102 and 186 for N_2O in 2006, 2007 and 2008, respectively. The average daily CH_4 and N_2O flux and standard deviation are 368 and 251 ng C m^{-2} s^{-1}, and 42 and 52 ng N m^{-2} s^{-1}, respectively. The fluxes of both gases are approximately lognormally distributed (*Fig 5.3.*).

The geometric average and geometric standard deviation are defined by $\mu_{Geo} = e^{\mu}$ and $\sigma_{Geo} = e^{\sigma}$ with μ and σ the mean and standard deviation of the logarithm of the observed flux values which resembles a normal distribution. The geometric average indicates the emission that occurred most often. When omitting a small number of negative values (<1%) from the data set, the geometric average fluxes in the lognormal distribution are 291 ng C m^{-2} s^{-1} and 26 ng N m^{-2} s^{-1} for CH_4 and N_2O, respectively. In all further analyses, we do not omit the negative numbers since uptake fluxes of CH_4 and N_2O could really occur.

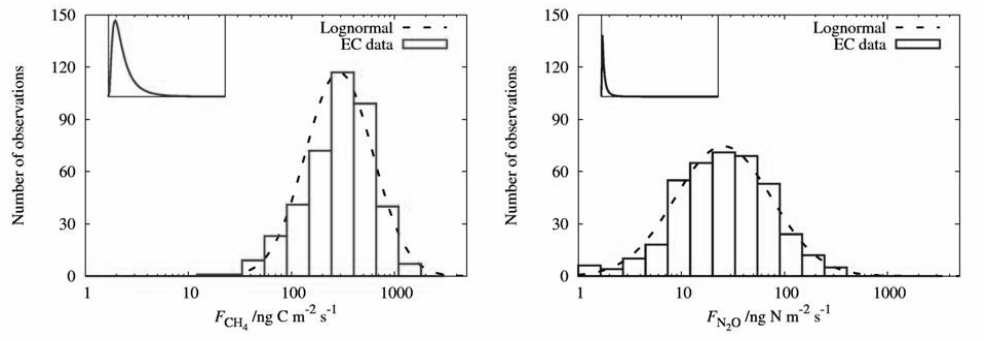


Fig 5.3. *Distribution of daily average CH_4 (left) and N_2O fluxes (right) with the lognormal distribution. The insert shows the shape of the distribution on a linear scale.*

5.4.2. Annual terrestrial CH_4 balance

A regression model is often used to fill the data gaps and to derive annual balances. Several studies have used a regression with temperature as only driving variable explaining about 90% of the variation (Hargreaves *et al.* 2001) or more commonly lower values of about 30% (Hendriks, *et al.*, 2007). The uncertainty in annual emissions based on these models is sometimes even larger than 50%.

In this study, we perform a step-wise multivariate regression analysis using the daily CH_4 values for which data is available for all possible regression parameters (N=265). The correlation of F_{CH4} and ln(F_{CH4}) is tested for air temperature (T_{air}), soil temperature (T_{soil}), rain intensity (R), wind velocity (U), air pressure (P_{air}) and soil moisture (θ). We do not include water table information since Schrier-Uijl *et al.* (2010a) showed that the CH_4 emissions did not correlate with the (actively managed) water table at our study site. The Pearson correlations (r) appear to be somewhat more significant for ln(F_{CH4}) than F_{CH4} with the different variables (*Table 5.2.*). Significant correlations (P<0.05) are found for ln(F_{CH4}) with T_{air}, T_{soil}, P_{air} , θ and R. In these cases r values are larger than those derived from static chamber measurements at the same site (Schrier-Uijl *et al.*, 2010a). The best regression is found by including T_{soil} and U; while adding other variables does not improve the regression. The regression of ln(F_{CH4}) versus T_{soil} and U explains about 60% of the variability in the observed daily CH_4 values (R^2=0.63; P<0.001). Adding soil moisture does not improve the regression which is a remarkable result. However, the reason can be explained by the fact that the effect of temperature on CH_4 fluxes is larger than the effect of soil moisture. In addition, the parameters soil moisture and temperature are significantly anti-correlated (r for soil moisture at 30 cm depth and T_{soil} is -0.82 and P<0.001). In other words, the CH_4 fluxes are higher for higher soil temperatures, but at the same time higher soil temperatures lead to lower soil moisture contents.

Table 5.2. *Pearson correlations (r) of CH_4 fluxes and N_2O background fluxes in Oukoop in the Netherlands. Fluxes have been measured by EC flux technique and are correlated with soil temperature (T_{soil} at 4 cm depth), air temperature (T_{air}), atmospheric pressure (P_{air}), windspeed (U), volumetric soil moisture content (θ at 10 cm depth) and daily rainfall (R).*

	N /-	T_{soil} /ºC	T_{air} /ºC	P_{air} /hPa	U /m s^{-1}	$\theta_{10\ cm}$ /m^3 m^{-3}	R /mm day^{-1}
Ln(CH_4)	265	0.749[a]	0.688[a]	-0.280[a]	0.050	-0.510[a]	0.193[b]
CH_4	265	0.714[a]	0.648[a]	-0.189[a]	-0.065	-0.553[a]	0.142[b]
Ln(N_2O)	252	0.526[a]	0.456[a]	-0.141[b]	-0.031	-0.401[a]	0.112
N_2O	252	0.394[a]	0.341[a]	-0.165[a]	-0.057	-0.367[a]	0.130[b]

[a]Correlation is significant at the 0.01 level. [b]Correlation is significant at the 0.05 level.

A correlation between CH_4 fluxes and U has also recently reported by e.g., Wille *et al.*, 2008 and Schrier-Uijl *et al.*, 2010b. They stated that higher CH_4 fluxes occur from water bodies at high wind velocities. These emissions are maybe due to the turbulence induced ebullition. In addition, Frolking and Crill (1994) indicated that ebullition could be triggered by changes of air pressure. This could explain the correlation between air pressure and CH_4 fluxes. Next to the enhanced emissions from water, enhanced fluxes from field could occur at higher mean wind velocities due to the soil storage effect discussed in this study.

The empirical regression of F_{CH4} against T_{soil} and U is given by

$$F_{CH4} = \exp\left(3.796(\pm 0.105) + 0.136(\pm 0.006)T_{soil} + 0.113(\pm 0.016)U\right) \qquad \textbf{5.14}$$

with F_{CH4} in ng C m^{-2} s^{-1}, T_{soil} at 4 cm depth in ºC and U at 3 m in m s^{-1}. In further research, the regression could possibly be improved by adding management information. For example Kroon *et al.* (2007) and Schrier-Uijl *et al.* (2010a) showed that CH_4 emissions are larger after cow manure application. The regression based emissions F_{CH4} and their uncertainties are represented together with the daily measured emissions and their uncertainties in *Fig 5.4.*

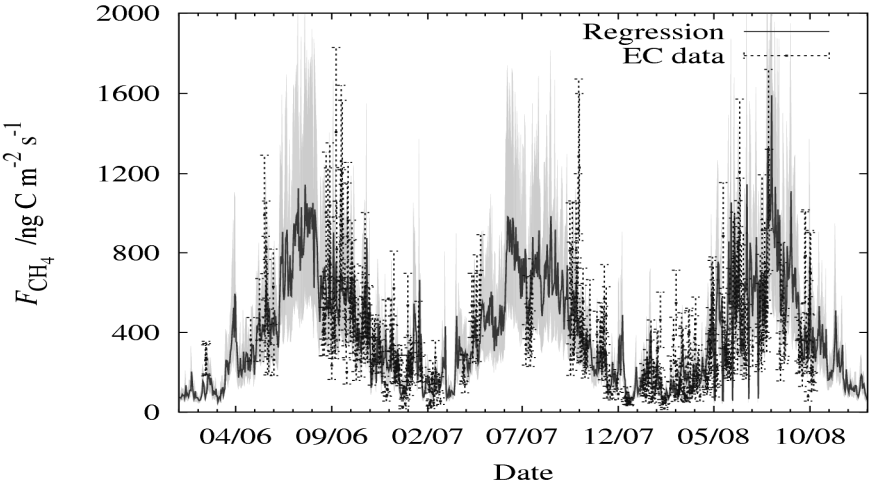


***Fig 5.4**. Measured and modeled CH4 emissions in Oukoop in 2006, 2007 and 2008. The model is a multivariate regression based on EC flux measurements. Emissions are given together with their uncertainty where the gray surface represents the uncertainty range of the empirical regression.*

The three methods to estimate the annual CH_4 balances (see methodology section) result in average annual emissions over the period 2006 to 2008 of 157 (±33%), 165 (±17%) and 164 (±21%) kg CH_4 ha^{-1} year^{-1} for method 1, 2 and 3, respectively (*Table 5.3.*). The terms within brackets denote the relative uncertainties which are calculated by the methods described in the appendix. The three estimates compare well, and these annual estimates are in good agreement with the annual estimate of 170 (±32%) kg CH_4 ha^{-1} year^{-1} determined by Schrier-Uijl *et al.* (2009a) using static chamber measurements at the same site. The estimate by method 2 has the smallest uncertainty and has therefore been used in the rest of this study. The uncertainty in the estimate by method 2 is much smaller than the uncertainty in the estimate by static chamber measurements. In addition, much more accurate estimates can be derived from EC flux measurements for measurement sets with larger data coverage. If a data coverage above 80% is attained, the uncertainty in the annual balances could be even smaller than 10% (based on Kroon *et al.*, 2009a).

This agricultural peatland is thus a serious source for CH_4 emissions in contrast to other published studies which indicate a small source (Regina *et al.*, 2007) or even a sink (Van den Pol-van Dasselaar *et al.,* 1999). The difference can be explained mainly by the ability of the EC flux method to include the CH_4 emission from the ditches and ditch edges at our site. Schrier-Uijl *et al.*, (2010a) indicated that about 70% of the emissions are caused by the ditches and ditch edges at our site. This means that only about 50 kg CH_4 ha^{-1} year^{-1} is emitted by the field. The remaining difference between the northern peat grassland emissions given in Regina *et al.,* (2007) can be explained by the temperature difference. Van den Pol-van Dasselaar *et al.*, (1999) indicated an uptake, however, their emissions

values are only based on field emissions. Consequently, it is important that emissions of ditch and ditch edges are taken into account when they occur within the investigated area.

Table 5.3. *Annual terrestrial CH_4 emission in kg CH_4 ha^{-1} $year^{-1}$ and its relative uncertainty in Oukoop in the Netherlands.*

	2006	2007	2008	Average
Method 1[a]	194(±54%)	140(±63%)	138(±53%)	157(±33%)
Method 2[b]	176 (±30%)	169(±31%)	149(±26%)	165(±17%)
Method 3[c]	172(±37%)	166(±37%)	155(±37%)	164(±21%)
Method 4[d]	203 (±48%)	162(±60%)	146(±60%)	170(±32%)

[a]Average EC flux is extrapolated. [b]Based on EC flux measurements where the remaining data gaps are filled by a multivariate regression model. [c]Based on multivariate regression model derived from EC flux measurements only. [d] Based on multivariate regression model derived from static chamber measurements only (Schrier-Uijl et al., 2009a).

5.4.3. Annual terrestrial N_2O balance

The annual terrestrial N_2O emission E_{N2O} and its relative uncertainty is derived from three methods (see methodology section and appendix). We estimate an average annual N_2O emission E_{EC} of 21 (±36%) kg N_2O ha^{-1} $year^{-1}$ with method 1 and of 18 (±21%) kg N_2O ha^{-1} $year^{-1}$ with method 2. The average annual N_2O emission E_l and E_d are both estimated at 1 (±50%) kg N_2O ha^{-1} $year^{-1}$. This leads to an average annual terrestrial N_2O emission E_{N2O} of 23 (±31%) kg N_2O ha^{-1} $year^{-1}$ and 20 (±19%) kg N_2O ha^{-1} $year^{-1}$ with method 1 and 2, respectively. For method 3, we first determine the background fluxes F_{bgnd} and the emission factors EF_l^{min} (see Eq. 5.9). The background fluxes are determined using a multivariate regression model based on EC flux data excluding the EC fluxes around fertilization events. We select the N_2O EC fluxes for which all possible driving variables are available (N=252). The correlations of F_{N2O} and $\ln(F_{N2O})$ are tested for air temperature (T_{air}), soil temperature (T_{soil}), rain intensity (R), wind velocity (U), air pressure (P_{air}) and soil moisture (θ) (*Table 5.2.*). The best linear regression is found by including T_{soil} and U (R^2=0.29 and P<0.001) and is given by

$$F_{bgnd} = \exp\left(1.777(\pm 0.200) + 0.122(\pm 0.012)T_{soil} + 0.063(\pm 0.030)U\right) \qquad \textbf{5.15}$$

with F_{bgnd} in ng N m^{-2} s^{-1}, T_{soil} the soil temperature at 4 cm depth in ºC and U the mean wind velocity at 3 m in m s^{-1}. The annual background emissions E_{bgnd} and their relative uncertainties are then estimated at 18 (±71%), 17 (±71%) and 17 (±71%) kg N_2O ha^{-1} $year^{-1}$ for 2006, 2007 and 2008, respectively, corresponding to an average background N_2O flux

F_{bgnd} of about 35 ng N m^{-2} s^{-1}. These values are larger than the background emission previously reported in Velthof *et al.* (1997) for peat soils in the Netherlands (5.3 ± 5.2 kg N_2O ha^{-1} $year^{-1}$). This is explainable since they derived the estimates from unfertilized and non-grazed cut grassland sites.
We determine the emission factors EF_l^{min} for six fertilization events using equation 5.8. The factor range is from 0.2 to 2.2% with an average and standard deviation of 1.2 and 0.8%, respectively, which is close to the IPCC default value of 1%. The IPCC default EF_1 value is therefore used for fertilization events for which no data is available. This leads to 5.2 (±50%), 5.3 (±50%) and 4.8 (±50%) kg N_2O ha^{-1} $year^{-1}$ for 2006, 2007 and 2008, respectively. The total annual terrestrial N_2O emission estimated by method 3 is then derived from adding the fertilization event emission E_{fert} to the background emission E_{bgnd}, the indirect emissions due to leaching E_l and deposition E_d (Eq. 5.4, 5.5, 5.6 and 5.9).

Table 5.4. *Annual terrestrial N_2O emission in kg N_2O ha^{-1} $year^{-1}$ and its relative uncertainty in Oukoop in the Netherlands.*

	2006	2007	2008	Average
Method 1[a]	30(±58%)	20(±57%)	19(±47%)	23(±31%)
Method 2[b]				20(±19%)
Method 3[c]	25(±52%)	24(±51%)	24(±50%)	24(±28%)

[a]*Average EC flux is extrapolated and the indirect emissions are added.* [b]*Integrating over average EC flux over the three years of each day number, i.e. from number 1 to 365, the data gaps are filled by the last available daily average. The indirect emissions are added.* [c]*Background emission is determined by a multivariate regression model based on EC flux data and the emissions due to fertilizing, leaching and deposition are added.*

This results in annual terrestrial N_2O emissions E_{N2O} of 25 (±52%), 24 (±51%) and 24 (±50%) kg N_2O ha^{-1} $year^{-1}$ for 2006, 2007 and 2008, respectively. The terms within brackets denote the relative uncertainty in the annual N_2O emission (Kroon et al., 2010, online version). The average emission of method 3 is close to the average annual terrestrial N_2O emission of method 1 and 2 (*Table 5.4.*) and the average terrestrial N_2O emission is comparable to reported annual terrestrial emissions of other peat soils in the Netherlands (Langeveld *et al.*, 1997). A representation of the background emission, the annual average emission E_{EC} and the EC flux data is shown in *Fig 5.5.*

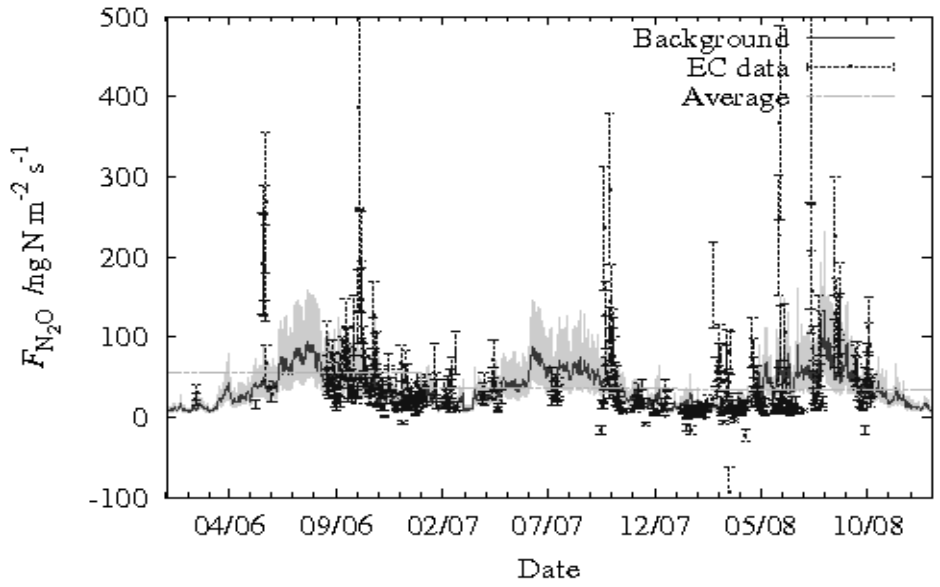


Fig 5.5. *Measured N_2O emissions and modeled background N_2O emissions in Oukoop in 2006, 2007 and 2008. The model is a multivariate regression based on EC flux measurements. They are given together with their uncertainty where the gray surface represents the uncertainty range of the empirical regression. In addition, the average measured EC flux over each year is indicated.*

We can not compare the annual terrestrial N_2O emission estimates derived from EC flux measurement to static chamber measurements since N_2O emissions were mostly too small to detect with the static chamber method used by Schrier-Uijl *et al.* (2010a, b). They used a closed dark chamber of 25 cm height from which air samples were taken at one-minute intervals. Each flux measurement consisted of five point-measurements and the concentrations were analyzed by a Photo Acoustic Field Gas Monitor (INNOVA 1412 sn, 71-113, ENMO services, Belgium). The minimum detection limit of this static chamber system was 39 ng N $m^{-2}s^{-1}$ and for the EC flux system 6 ng N $m^{-2}s^{-1}$ (Kroon *et al.*, 2007). Thus, the use of this EC flux set-up was a big step forward in obtaining more accurate emission estimates. In addition, more accurate estimates can be obtained when the data coverage increases (Kroon *et al.*, 2009a).

After estimating the annual balances, we investigate in more detail the emission factor EF_1 which is used in the IPCC guidelines for estimating the direct N_2O emissions from agricultural soils. In this study, we calculate the possible EF_1 ranges for six fertilization events ranging from EF_1^{min} to EF_1^{max}. The minimum EF_1 has already been determined and ranges from 0.2 tot 2.2% with an average and standard deviation of 1.2 and 0.8%, respectively. The maximum EF_1 is estimated using equation 5.12 and ranges from 0.9 to 4.6% with an average and standard deviation of 2.8 and 1.9%. Thus, the average EF_1 range

is from 1.2 to 2.8% which is larger than the IPCC default EF_1 value of 1% (IPCC, 2006). Kuikman *et al.* (2006) has also reported a larger EF_1 than 1% for peat soils in the Netherlands, respectively, an EF_1 of 0.55%, 1.24% and 3% for sand, clay and peat soils. This emphasizes the fact that separate parameterizations should be derived for each soil type.

In addition, the emission factors EF_1 should be determined more accurately since the uncertainty in the direct N_2O emission from agricultural soils is a main contributor to the uncertainty in the total national GHG emission estimate (Maas *et al.*, 2008). The uncertainty is large due to large uncertainties in the measured N_2O emissions F_{N2O}, the background emissions F_{bgnd} and the *k* factor. We recommend performing simultaneously continuous terrestrial measurements of N_2O and NH_3 at representative non agricultural and a managed sites at national and international scale. These measurements could be done using micrometeorological techniques, like the EC flux technique. An overview is given of EF_1^{min}, EF_1^{max} and the micrometeorological properties of the six fertilization events in *Table 5.5*.

Table 5.5. *Emission factor range of six fertilization events of one month in Oukoop in the Netherlands.*

Date Fertilizing	Cow manure /kg N ha^{-1}	Artificial fertilizer /kg N ha^{-1}	Tsoil $_{4\ cm}$ /ºC	WFPS $_{10\ cm}$ [a] /%	R /mm $month^{-1}$	EF_1^{min} [b] - EF_1^{max} [c] /%
14-09-06	55	-	16	85	47	1.9 – 4.6
15-09-07	46	-	14	91	68	2.2 – 4.6
05-02-08	138	-	3	96	32	0.6 – 1.0
03-04-08	46	43	8	95	29	0.2 – 0.9
19-05-08	83	27	15	77	48	0.5 – 1.2
30-06-08	46	-	16	60	104	1.7 – 4.2

[a] *WFPS means water-filled pore space.* [b] EF_1^{min} *is the factor for direct N_2O emissions where the background emission derived from EC flux measurements is subtracted.* [c] EF_1^{max} *is the factor for direct N_2O emission where the background emission is set to 0 ng N m^{-2} s^{-1}.*

5.4.4. Annual terrestrial GHG balance

The total terrestrial GHG balance is estimated by equation 5.13. We use the annual CH_4 emission derived from method 2 and the annual N_2O emissions derived from method 3. The average annual CH_4 emission is 165 kg CH_4 ha^{-1} $year^{-1}$ and the average annual N_2O is 24 kg N_2O ha^{-1} $year^{-1}$. The annual terrestrial CO_2 emission, net ecosystem exchange (NEE), is taken from Veenendaal *et al.* (2007), who calculated a value of 134 g C m^{-2} $year^{-1}$ for October 2004 to October 2005 which equals to 4910 kg CO_2 ha^{-1} $year^{-1}$. This leads to an average total terrestrial GHG emission of 16 (±15%) Mg ha^{-1} $year^{-1}$in CO_2-equivalents with 30%, 25% and 45% from CO_2, CH_4 and N_2O, respectively (*Fig 5.6.*). This means that the agricultural peatland is an important source for all three greenhouse gases.

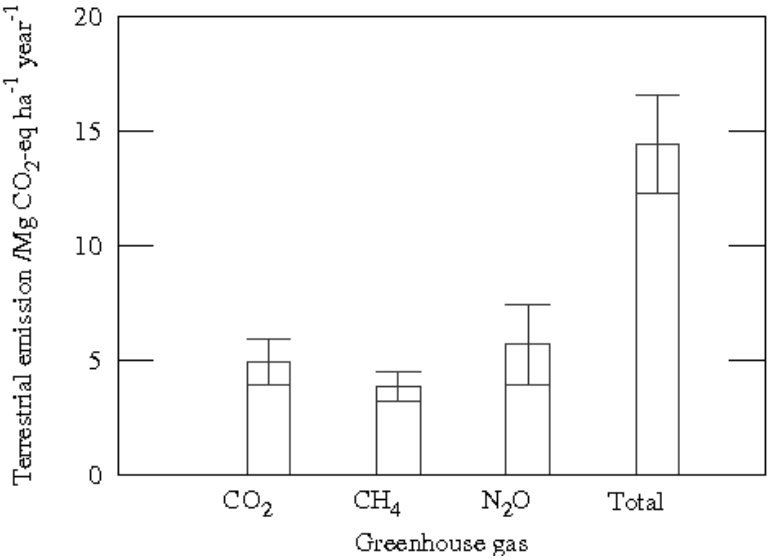


Fig 5.6. *Annual terrestrial emissions of CO_2, CH_4 and N_2O and the sum of the emissions of these three gases in CO_2-equivalents ha^{-1} $year^{-1}$ with their uncertainty measured at a managed peat area in the Netherlands.*

To put this into the right perspective, it is important to realize that there are substantial additional sources due to the full range of agricultural activities on this peatland. If CO_2 emissions due to biomass removal (i.e. grass harvest) and CH_4 farm emissions (i.e. enteric fermentation by dairy cows) are included, the total emission will increase by $10.5x10^3$ kg CO_2 ha^{-1} $year^{-1}$ (Veenendaal *et al.*, 2007) and 5911 kg CO_2 $ha^{-1}$$year^{-1}$ (257 kg CH_4 ha^{-1} $year^{-1}$) (Schrier-Uijl *et al.*, 2010a), respectively. The total balance will thus increase by even more than 250%, which emphasizes the importance of further research in which all GHG components will be taken into account.

We also compared the total field GHG emissions at this site (Oukoop) to the GHG emissions in a former agricultural site (Horstermeer) which was converted to a nature reserve 15 years ago by stopping intensive farming and raising the water table (Hendriks *et al.*, 2007). The terrestrial CO_2, CH_4 and N_2O emissions at the Horstermeer site are -11403 kg CO_2 ha^{-1} $year^{-1}$, 417 kg CH_4 ha^{-1} $year^{-1}$ and negligible (Hendriks *et al.*, 2007). This leads to a total GHG uptake of -2 Mg ha^{-1} $year^{-1}$ in CO_2-equivalents. Consequently, a transformation of an intensively agricultural site to a nature reserve will probably lead to a decrease in total GHG emission since the Oukoop site is a strong source of 16 Mg ha^{-1} $year^{-1}$ in CO_2-equivalents. The CO_2 and N_2O emissions are thus much larger at the intensively managed Oukoop site than at the restored Horstermeer site. In addition, if we also include the farm based emissions to the CH_4 emissions, the total CH_4 emissions at Oukoop will be also larger than the CH_4 emissions at the Horstermeer site (e.g., Schier-Uijl *et al.*, 2010a).

5.5. Conclusions

The annual CH_4 and N_2O terrestrial balances were derived from a dairy farm on peat grassland in the Netherlands over 2006, 2007 and 2008 using EC flux measurements. The average terrestrial CH_4 balance was 165 kg CH_4 ha^{-1} $year^{-1}$ and the average terrestrial N_2O balance 24 kg N_2O ha^{-1} $year^{-1}$. The CH_4 estimate was more accurate than the estimate by static chambers and the N_2O estimate was a step forward since the background fluxes could not be detected by static chamber measurements using a photo acoustic instrument at the same site.

Furthermore, we investigated the direct emission factor EF_1 from agricultural soils which is used in the IPCC guidelines. We calculated the possible EF_1 range for six fertilization events ranging from EF_1^{min} to EF_1^{max} since the real background emission was not known. The minimum EF_1 was determined by subtracting the background emission and the maximum EF_1 without subtracting the background emission. The average EF_1^{min} was 1.2% and the average EF_1^{max} 2.8%. Both values are larger than the IPCC default EF_1 value of 1%.

Finally, the total terrestrial GHG balance was estimated at 16 Mg ha^{-1} $year^{-1}$ in CO_2-equivalents averaged over 2006, 2007 and 2008 with contributions of 30%, 25% and 45% by CO_2, CH_4 and N_2O. This agricultural peatland was thus a serious source for all three greenhouse gases. The total emission would be even larger since the farm emissions and the biomass removal were not taken into account.

5.6. Acknowledgements

This research was part of the Dutch National Research Program BSIK ME1. We are grateful to two anonymous referees for their remarks and suggestions. Thanks are due to our colleagues P. van den Bulk, P. Fonteijn and H. van ‘t Veen for their assistance during

these measurements. We are also grateful to A. Vermeulen and E. de Beus for their assistance in data analyses. In addition, we would like to take the opportunity of thanking C. Flechard from Inra in France for the productive discussion and advices. Finally, we owe a special debt of gratitude to the farmer T. Van Eyk for using his farm site.

5.7. References

Aubinet, M., Grelle, A., Ibrom, A., Rannik, Ü, Moncrieff, J., *et al.*, 2000. Estimates of the annual net carbon and water exchange of forests: The EUROFLUX methodology. *Advances in ecological research*, **30**, 113-175.

Businger, J. A., 1986. Evaluation of the accuracy with which dry deposition can be measured with current micrometeorological techniques. *Journal of Climate and applied meteorology*, **25**, 1100-1124.

Crill, P.M., Keller, M. & Weitz, A., 2000. Intensive field measurements of nitrous oxide emissions from a tropical agricultural soil. *Global Biogeochemical Cycles*, **14**, 85-95.

Drösler, M., Freibauer, A., Christensen, T.R. & Friborg, T., 2008. Observations and status of peatland greenhouse gas emissions in Europe. In: *Observing the continental scale greenhouse gas balance.* (ed. Dolman A. J, Valentini R, & Freibauer A.), pp. 243-262. Springer Ecological series 203.

Duan, X., Wang, X., Mu, Y. & Ouyang, Z., 2005. Seasonal and diurnal variations in methane emissions from Wuliangsu Lake in arid regions of China. *Atmospheric Environment*, **39**, 4479-4487.

Eugster, W., Zeyer, K., Zeeman, M., Michna, P., Zingg, A., Buchmann, N. & Emmenegger, L., 2007. Methodical study of nitrous oxide eddy covariance measurements using quantum cascade laser spectrometry over a Swiss forest. *Biogeosciences*, **4**, 927-939.

Flechard, C.R., Ambus, P., Skiba, U., Rees, R. M. *et al.*, 2007a. Effects of climate and management intensity on nitrous oxide emissions in grassland systems across Europe. *Agriculture, Ecosystems and Environment*, **121**, 135-152.

Flechard, C.R., Neftel, A. Jochner, M., Ammann, C., Leifeld, J. & Fuhgrer, J., 2007b. Temporal changes in soil pore space CO_2 concentration and storage under permanent grassland. *Agric. For. Meteorol.*, **142**, 66-84.

Foken, T. & Wichura, B., 1996. Tools for quality assessment of surface-based flux measurements. *Agric. For. Meteorol.*,**78**, 83-105.

Frolking, S., & Crill, P, 1994. Climate controls on temporal variability of methane flux from a poor fen in southeastern New-Hampshire – measurement and modeling. Global biogeochem. Cycl., 8, 385-397.

Gu, L., Falge, E.M., Boden, T., Baldocchi, D.D., Black, T.A., *et al.*, 2005. Objective threshold determination for nighttime eddy flux filtering. *Agric. For. Meteorol.*, **128**, 179-197.

Hargreaves, K.J., Fowler, D., Pitcairn, C.E.R. & Aurela, M., 2001. Annual methane emission from Finnish mires estimated from eddy covariance campaign measurements. *Theor. Appl. Climatol.*,**70**, 203-213.

Hendriks, D.M.D., Van Huissteden, J., Dolman, A.J. & Van der Molen, M.K., 2007. The full greenhouse gas balance of an abandoned peat meadow. *Biogeosciences*, **4**, 411-424.

Hendriks, D.M.D., Dolman, A.J. Van der Molen, J.K. & Van Huissteden, J., 2008. A compact and stable eddy covariance set-up for methane measurements using off-axis integrated cavity output spectroscopy. *Atmos. Chem. Phys*., **8**, 431-443.

IPCC, 2001a. Good Practice Guidance and Uncertainty Management in National Greenhouse gas Inventories. Intergovernmental Panel on Climate Change (IPCC) Institute for Global Environmental Strategies, Tokyo, Japan.

IPCC, 2006. 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Prepared by the National Greenhouse Gas Inventories Programme, Eggleston, H.S., Buendia, L., Miwa, K., Ngara T. & Tanabe, K., (eds), IGES, Japan.

IPCC 2007. Climate change 2007.Tthe Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Kormann, R. & Meixner, F.X., 2001. An analytical footprint model for non-neutral stratisfication. *Bound.-Lay. Meteorol*., **99**, 207-224.

Kroon, P.S., Hensen, A., Jonker, H.J.J., Zahniser, M.S., Van het Veen, W.H. & Vermeulen, A.T., 2007. Suitability of quantum cascade laser spectroscopy for CH_4 and N_2O eddy covariance flux measurements. *Biogeosciences*, **4**, 715-728.

Kroon, P.S., Hensen, A., Jonker, H.J.J., Ouwersloot, H.G. Vermeulen, A.T. & Bosveld, F.C., 2009a. Uncertainties in eddy covariance flux measurements assessed from CH_4 and N_2O observations. *Agric. For. Meteorol.*, doi:10.1016/j.agrformet.2009.08.008.

Kroon, P.S., Schuitmaker, A., Jonker, H.J.J., Tummers, M.J., Hensen, A., & Bosveld, F.C., 2009b. An evaluation by laser Doppler anemometry of the correction algorithm based on Kaimal co-spectra for high frequency losses of EC flux measurements of CH_4 and N_2O. Agric. For. Meteorol., doi:10.1016/j.agrformet.2009.08.009.

Kuikman, P.J., Van der Hoek, K.W., Smit, A. & Zwart, K., 2006. Update of emission factors for nitrous oxide from Agricultural soils on the basis of measurements in the Netherlands, *Alterra-report 112*, Wageningen, The Netherlands.

Langeveld, C.A., Segers, R., Dirks, B.O.M., Van den Pol-van Dasselaar, A., Velthof, G. C. & Hensen, A., 1997. Emissions of CO_2, CH_4 and N_2O from pasture on drained peat soils in the Netherlands. *Eur. J. Agron*., **7(1-3)**, 25-42.

Lee, X.L., Massman, W. & Law, B., 2004. Handbook of micrometeorology, Kluwer Academic Publishers, Dordrecht, The Netherlands.

Maas, C.W.M., Coenen, P.W.H.G., Ruyssenaars, P.G., Vreuls, H.H.J., Brandes, L.J., Baas, K., Van den Berghe, G., Van den Born, G. J., *et al.*, 2008. Greenhouse gas emissions in the Netherlands 1990-2006, National inventory report 2008. MNP report 500080009, Netherlands Environmental Assessment Agency (MNP).

Maljanen, M., Martikainen, P.J., Aaltonen, H. & Silvola, J., 2002. Short-term variation in fluxes of carbon dioxide, nitrous oxide and methane in cultivated and forested organic boreal soils. *Soil Biology & Biochemistry*, **34**, 577-584.

Maljanen, M., Liikanen, A., Silvola, J., & Martikainen, P.J., 2003. Nitrous oxide emissions from boreal organic soil under different land-use. *Soil Biology & Biochemistry*, **35**, 1-12.

McMillen, R.T., 1988. An eddy correlation technique with extended applicability to non-simple terrain. *Bound.-Lay. Meteorol.*, **43**, 231-245.

Mikkelä, C., Sundh, I., Svensson, B.H. & Nilsson, M., 1995: Diurnal variation in methane emission in relation to the water table, soil temperature, climate and vegetation cover in a Swedish acid mire. *Biogeochemistry*, **28**, 93-114.

Neftel, A., Flechard, C., Ammann, C., Conen, F., Emmenegger, L. & Zeyer, K., 2007. Experimental assessment of N_2O background fluxes in grassland systems. *Tellus B*, **59**, 470-482.

Nieveen, J.P., Campbell, D.I., & Schipper, L.A., 2005. Carbon exchange of grazed pasture on a drained peat soil. *Glob. Change Biol.*, **11**, 607-618.

Regina, K., Syväsalo, E., Hannukkala, A. & Esala, M., 2004. Fluxes of N_2O from farmed peat soils in Finland. *Eur. J. Soil Sci.*, **55(3)**, 591-599.

Regina, K., Pihlatie, M., Esal, M., & Alakukku, L., 2007. Methane fluxes on boreal arable soils. *Agriculture, Ecosystems and Environment*, **119**, 346-352.

Ruiter, J.F., Van Pul, W.A.J., Van Jaarsveld, J.A., & Buijsman, E., 2006. Zuur- en stikstofdepositie in Nederland in de periode 1981-2002. RIVM rapport 500037005/2006 (In Dutch).

Schothorst, C.J., 1977. Subsidence of low moor peat in the Western Netherlands, *Geoderma*, **17**, 265-271.

Schrier-Uijl, A.P., Kroon, P.S., Leffelaar, P.A., Van Huissteden, J.C., Berendse, F. & Veenendaal, E.M., 2009a. Methane emissions in two drained peat ecosystems with high and low agricultural intensity, *Plant Soil*, doi :10.1007/s11104-009-0180-1.

Schrier-Uijl, A.P., Kroon, P.S., Hensen, A., Leffelaar, P.A., Berendse, F. & Veenendaal, E.M., 2009b. Comparison of chamber and eddy covariance based CO_2 and CH_4 emission estimates in a heterogeneous grass ecosystem on peat, *Agric. For. Meteorol.*, doi:10.1016/j.agrformet.2009.11.007.

Skiba, U., Hargreaves, K.J., Beverland, I.J., ONeill, D.H., Fowler, D. & Moncrieff, J.B., 1996. Measurement of field scale N_2O emission fluxes from a wheat crop using micrometeorological techniques. *Plant Soil*, **181**, 139-144.

Van den Bos, R. Van Huissteden, J. & Van de Plassche, O.: A model-based assessment of CO_2 and CH_4 fluxes in coastal peatlands (Western Netherlands) for different climate

and management scenarios, In: *Human influences on carbon fluxes in coastal peatlands; processs analysis, quantification and prediction* (ed. Van den Bos, R. M.), Thesis, Vrije Universiteit, pp.91-101, 2003.

Van den Pol-van Dasselaar, A., Van Beusichem, M.L., & Oenema, O., 1999. Effects of nitrogen input and grazing on methane fluxes of extensively and intensively managed grasslands in the Netherlands. *Biol. Fertil. Soils*, **29**, 24-30.

Veenendaal, E.M., Kolle, O., Leffelaar, P.S. Schrier-Uijl, A.P., Van Huissteden, J., Van Walsem, J., Möller, F. & Berends, F., 2007. CO_2 exchange and carbon balance in two grassland sites on eutrophic drained peat soils. *Biogeosciences*, **4**, 1027-1040.

Velthof, G.L & Oenema, O., 1997. Nitrous oxide emission from dairy farming systems in the Netherlands. *Netherlands Journal of Agricultural Science*, **45**, 347-360.

Wille, C., Kutzbach, L., Sachs, T., Wagner, D., & Pfeiffer, E.M., 2008. Methane emission from Siberian arctic polygonal tundra: eddy covariance measurements and modelling. Glob. Change Biol., 14, 1395-1408.

Wohlfahrt, G., Anfang, C., Bahn, M., Haslwanter, A., Newesely, C., *et al.*, 2005. Quantifying nighttime ecosystem respiration of a meadow using eddy covariance, chambers and modelling. *Agric. For. Meteorol.*, **128(3-4)**, 141-162.

Chapter 6

Release of CO_2 and CH_4 from lakes and drainage ditches in temperate wetlands

Shallow fresh water bodies in peat areas are important contributors to greenhouse gas fluxes to the atmosphere. In this study we determined the magnitude of CH_4 and CO_2 fluxes from 12 water bodies in Dutch wetlands during the summer season and studied the factors that might regulate emissions of CH_4 and CO_2 from these lakes and ditches. The lakes and ditches acted as CO_2 and CH_4 sources of emissions to the atmosphere; the fluxes from the ditches were significantly larger than the fluxes from the lakes. The mean greenhouse gas flux from ditches and lakes amounted to 129.1 ± 8.2 (mean ± SE) and 61.5 ± 7.1 mg m^{-2} hr^{-1} for CO_2 and 33.7 ± 9.3 and 3.9 ± 1.6 mg m^{-2} hr^{-1} for CH_4, respectively. In most water bodies CH_4 was the dominant greenhouse gas in terms of warming potential. Trophic status of the water and the sediment was an important factor regulating emissions. By using multiple linear regression 87% of the variation in CH_4 could be explained by PO_4^{3-} concentration in the sediment and Fe^{2+} concentration in the water, and 89% of the CO_2 flux could be explained by depth, EC and pH of the water. Decreasing the nutrient loads and input of organic substrates to ditches and lakes by for example reducing application of fertilizers and manure within the catchments and decreasing upward seepage of nutrient rich water from the surrounding area will likely reduce summer emissions of CO_2 and CH_4 from these water bodies.

Based on A.P. Schrier-Uijl, A.J. Veraart, P. A. Leffelaar, F. Berendse and E.M. Veenendaal
Biogeochemistry (2010), (doi 10.1007/s10533-010-9440-7)

6.1. Introduction

Freshwater bodies such as ditches, streams, wetlands and lakes contribute appreciably to the processing of carbon and its transport to the atmosphere (e.g Bastviken et al. 2004; Walter et al. 2006; Wang et al. 2006). It has been estimated that lakes annually emit 8-48 Tg methane (CH_4), which is 6-16% of the global natural CH_4 emissions (Bastviken et al. 2004; St. Louis et al. 2000), and 513 Tg C carbon dioxide (CO_2) (Cole et al. 1994). Saarnio et al. (2009) have estimated that large lakes alone account for 24% of all wetland CH_4 emissions in Europe. It has been shown that small water bodies also significantly contribute to the landscape-scale CH_4 budgets in wetland regions (eg. Schrier-Uijl et al. 2009; Juutinen et al. 2009; Repo et al. 2007; Walter et al. 2007; Roulet et Moore, 1995). Yet though it is likely that both lakes with organic-rich sediment and also eutrophic ditches contribute especially significantly to regional greenhouse gas balances, they are poorly studied and little is known about their underlying biogeochemical processes (Saarnio et al. 2009).

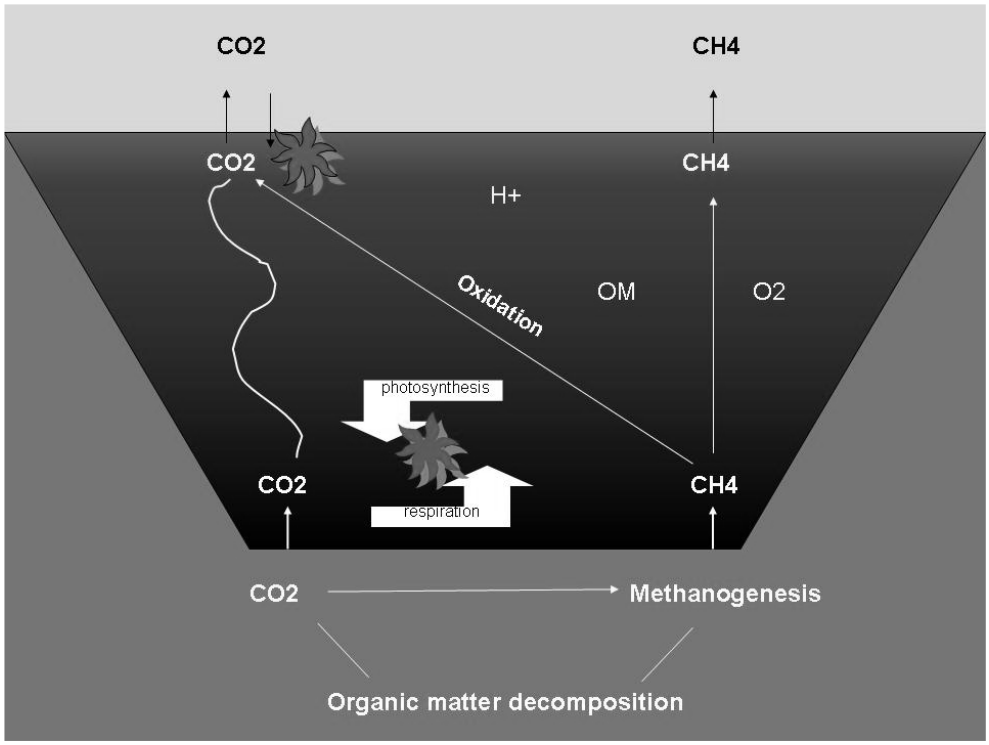


***Fig 6.1**. Simplified illustration of CO_2 and CH_4 dynamics in water bodies, with OM= organic matter.*

CO_2 is produced by respiration in sediments and throughout the water column and can also be a product of biological processes in the sediment (*Fig 6.1.*). As CO_2 is highly soluble, high concentrations can accumulate near the sediment/water interface, which results in oversaturation and release to the atmosphere. It has been suggested that the transport of dissolved organic carbon (DOC) from terrestrial environments is an important source of carbon in aquatic environments. If this is the case, lakes in organic-rich peatlands have larger CO_2 fluxes than lakes in mineral catchments (Rantakari and Krotelaines, 2005; Huttunen et al. 2002).

CH_4 emission is the balance of two counteracting processes: methanogenesis in anoxic conditions and the oxidation of the generated CH_4 (Minkkinen et al. 2006; Bastviken et al. 2002), (*Fig 6.1.*). CH_4 is a major product of carbon metabolism in lakes; its production depends on the availability of alternative electron acceptors such as O_2, NO_3^-, Fe^{3+} and $SO4^{2-}$ (van Bodegom and Scholten, 2001). After these electron acceptors have been used up, CH_4 production becomes the dominating degradation process of organic matter and is the terminal microbial process in the anaerobic degradation of organic matter. The CH_4 travels from the sediment through the water column to the atmosphere and on the way it can be oxidised into CO_2 (Whiting and Chanton, 1993). Most of the CH_4 that remains unoxidised will be emitted by diffusive flux to the atmosphere.

The underlying microbial processes affecting CO_2 and CH_4 production and emission are regulated by variables such as sediment and water temperature, oxygen availability, organic matter availability and composition, sediment and water chemistry, the presence of electron acceptors (redox conditions), pH, electrical conductivity (EC) and factors such as water depth and lake size (e.g. Stadmark and Leonardson, 2005; Juutinen et al. 2009; Repo et al. 2007; Frei et al. 2006; Loeb et al. 2007; Casper et al. 2003).

Most of the freshwater lakes in the Netherlands are in peat areas, are very shallow (<2 m), and were created by large-scale dredging and removal of peat during the early 17th century (Gulati and van Donk, 2002). They vary in area, depth, hydrology and physico-chemical characteristics, but most of them are eutrophic, due to the application of fertilisers and manure within their catchments, the oxidation of peat and the upward seepage of nutrient-rich water from the surrounding area. Drainage since the Middle Ages has resulted in the typical landscape of narrow fields separated by drainage ditches (*Fig 6.2.*).

Shallow fresh water bodies are not well understood in terms of their greenhouse gas emissions and have not been incorporated in previous regional or global greenhouse gas budgets. The emission from these water bodies is probably high and poses an international rather than a domestic problem because in many lowland regions of Europe agriculture continues to contribute appreciably to the nutrient loading of lakes and ditches (Gulati and van Donk, 2002; Lamers et al.1998).

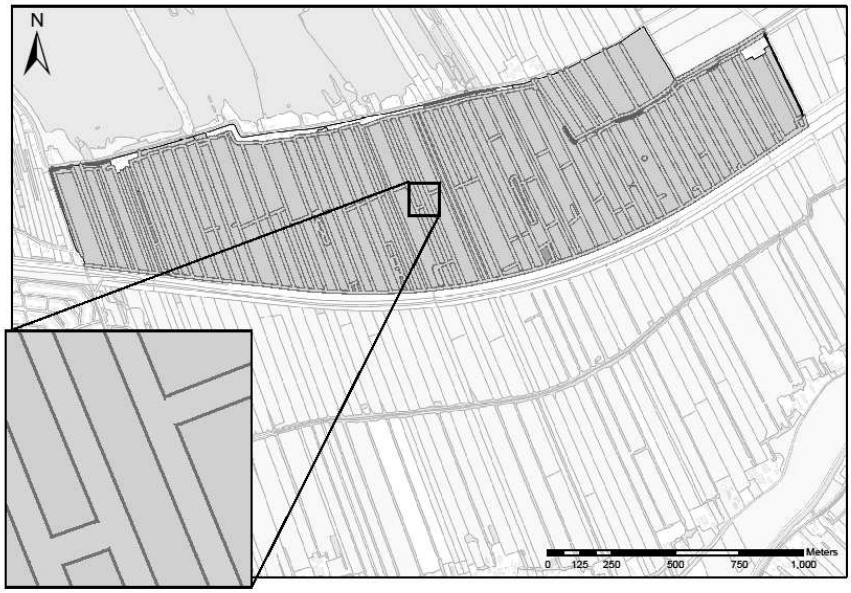


Fig 6.2. *The typical Dutch peat area landscape of lakes and narrow fields separated by drainage ditches (Nol et al. personal communication).*

About 16% of the total area (41 864 km^2) of the Netherlands is covered by water, mostly classified as wetland (Gulati and van Donk, 2002) and with over 300,000 km of drainage ditches. Much of this wetland is in peat areas. It is important to quantify how these wetlands contribute to the greenhouse gas balance and which factors regulate the emission. In this study we focus on the high-emitting temperate lakes and drainage ditches in peat areas in the Netherlands and on many variables that can alter the emission of CH_4 and CO_2. The two aims of this study were (1) to quantify CH_4 and CO_2 fluxes from shallow lakes and drainage ditches in the Netherlands during a three-week period in the summer season and (2) to identify the factors that regulate the emissions of CH_4 and CO_2 from lakes and ditches.

6.2. Materials and methods

6.2.1. Study sites

Measurements were performed in a three-week period between June 16th and July 6th in the summer of 2009 in 5 shallow fresh water lakes and 14 drainage ditches at 7 locations in peat areas in the Netherlands (*Fig 6.3.*).

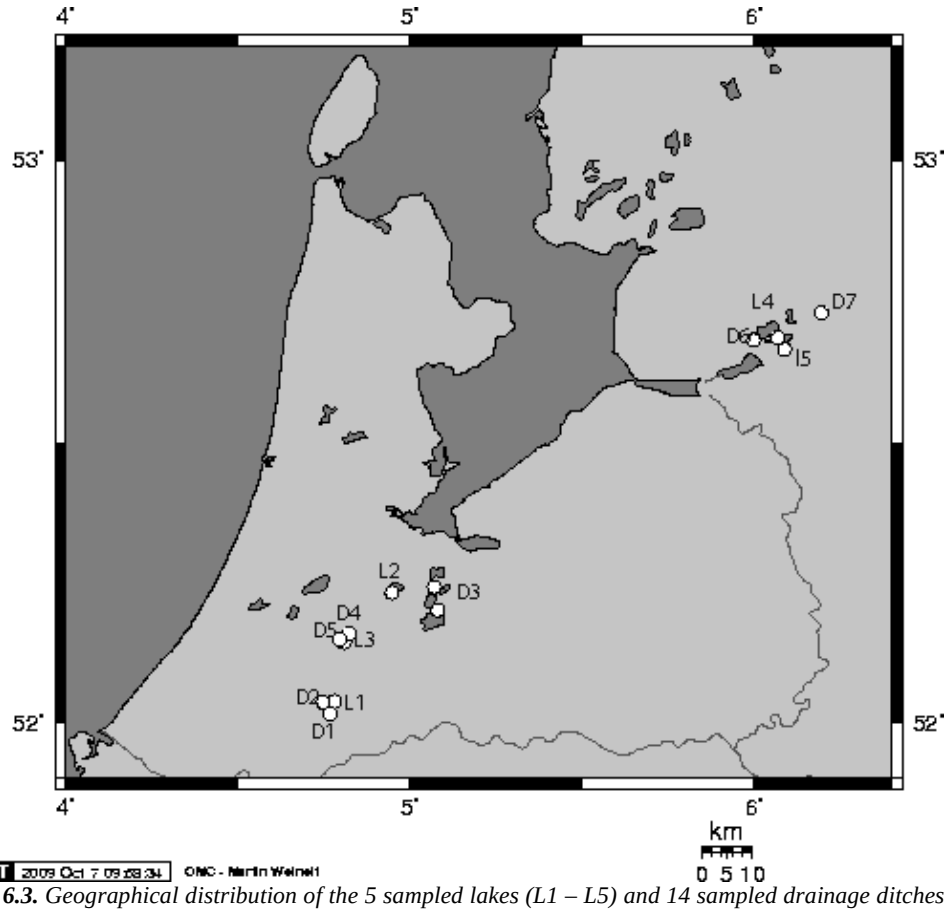


***Fig 6.3.** Geographical distribution of the 5 sampled lakes (L1 – L5) and 14 sampled drainage ditches (at 7 locations) (D1 – D7) in peat areas in the Netherlands.*

The 5 lakes are located in peat areas in the Netherlands and differ in trophic status (de Haan et al. 1993) and depth (*Table 6.1.*). L4 and L5 are located in the east of the Netherlands where the subsoil consists of mesotrophic to oligotrophic sedge-peat overlying sand. The other lakes are located in the southwest of the Netherlands where the subsoil consists of eutrophic to mesotrophic reed–sedge peat and alder carr peat (Rienks and Gerritsen, 2005).

Table 6.1. *Characteristics of the lakes sampled.*

Lake	Abbreviation	Trophic status	Lake size (ha)	Average depth (m)
Reeuwijkse plas	L1	eutrophic	927	2.1
Vinkeveense plas	L2	eutrophic	1079	2.4
Nieuwkoopse plas	L3	eutrohpic	676	2.5
Belterwiede	L4	mesotrophic to eutrophic	613	1.8
Schutsloterwiede	L5	mesotrophic to eutrophic	141	1.2

Drainage ditches at 7 locations in different peat areas in the Netherlands were sampled. They differed in trophic status and water depth (*Table 6.2.*). At each location 2 connected ditches were sampled and because there were no significant differences between them related to water quality they were treated as 1 location in the analyses. All the drainage ditches sampled contained some aquatic vegetation.

Table 6.2. *Characteristics of the drainage ditches.*

Drainage ditch	Abbreviation	Trophic status	Ditch width (m)	Mean water depth (m)
Oukoop	D1	eutrophic	8	0.25
Stein	D2	mesotrophic	6	0.28
Horstermeer	D3	eutrophic	3	0.90
Drie berken	D4	mesotrophic	3	0.87
Koole	D5	mesotrophic	3	0.80
Sint Jans	D6	eutrophic	2	0.28
Doosje	D7	mesotrohpic	6	0.43

6.2.2. Measurements

6.2.2.1. Flux measurements and calculation of fluxes

Detailed measurements of CH_4 emission and CO_2 emission were performed with floating chambers from a dinghy at different locations in the lakes and drainage ditches. We measured the emissions from each lake on two different days. On each of these days we measured at three different locations per lake, and repeated the measurements 5 times at each location. This yielded 30 measurements per lake. Each ditch was sampled on one day in the three-week period, with 8 replicates per ditch. This yielded 16 measurements per location in the two connected ditches. All measurements were performed between 10.30 h and 14.30 h. Data quality was assessed and outliers resulting from disturbances were removed from the dataset. Emissions of CH_4, CO_2 and N_2O were determined using a closed dark chamber method and a Photo Acoustic Field Gas Monitor (INNOVA 1412 sn, 710-

113, ENMO services, Belgium) connected to a PVC chamber by Teflon tubing (e.g. van Huissteden et al. 2005; Hendriks et al. 2007). Fluxes of N_2O appeared to be too low to detect with the gas analyzer, therefore the N_2O flux measurements were not included in the analyses. Samples were taken from the headspace of this closed cylindrical dark chamber (30 cm diameter, 25 cm height). Gas samples were taken every minute during a 5-minute period and every single measurements was checked on linearity of the build up of the gas concentration in the chamber. This check eliminated about 30% of the measurements. The slope dC/dt of the gas concentration curve at time t=0 was estimated using linear regression (e.g. van Huissteden et al. 2005; Schrier-Uijl et al. 2010b). A small fan was installed in the chamber to homogenise the inside air and a water lock was used to control pressure in the chamber. We used a floater to place the chamber onto the water surface, carefully avoiding the effect of pressure differences and the disturbance of the water surface (for details, see Schrier-Uijl et al. 2010a). Since the gas monitor software does not compensate fully for cross-interference of CO_2 and water vapour at high concentrations, air was led through glass tubes filled with silica gel and soda lime before it entered the gas analyser, to remove water vapour. To cross-validate the chamber-based measurements, we also performed eddy covariance measurements on L1 at the same time and location and compared these with the chamber measurements within the footprint of the system. The eddy covariance system was located along a boardwalk in L1 and the footprint of the mast was on the lake. Within this footprint chamber measurements were performed on the lake during a period of 4 hours. The two independent methods had previously been compared at different temporal scales in a heterogeneous landscape of fields and ditches (Schrier-Uijl et al 2010b; Kroon et al. 2007).

6.2.2.2. Variables measured

At each lake and drainage ditch we measured water temperature and pH at two depths (10 and 30 cm and at 25 cm depth in D1 and D2), dissolved oxygen at 10 cm intervals from the water surface to the sediment surface, and the EC at 10 cm depth. Oxygen, pH, temperature and EC were measured with an HQ multiprobe with a luminescent dissolved oxygen sensor (Hach Company, Loveland, Colorado, USA). The variables investigated in the ditches were the dissolved CH_4 concentrations at the water surface, the middle of the water column and in the water immediately above the sediment. Samples for dissolved methane analysis were taken using an airtight 20ml glass syringe at three depths in the water column: at the sediment surface, at the water surface, and at a depth half-way in between. The water samples were transferred into airtight glass Exetainers® (Labco, high Wycombe, UK) containing 120µl $ZnCl_2$ to halt biological processes; to prevent air bubbles being trapped in these vials they were filled to overflowing before being capped. The samples were stored in water at 20°C until analysis. Dissolved methane was measured by membrane inlet mass spectrometry (MIMS) (Lloyd & Scott 1983) using an OmniStarTM Gas Analysis System

(Pfeiffer Vacuum, Asslar, Germany), equipped with a quadrupole QMS 200 mass spectrometer with a Channeltron detector (Burle Industries). The MS was operated by Quadstar 32-bit software for data acquisition. The sample was pumped through a water bath at 20°C before passing through silicon membrane tubing in which gases were released to the MS. An inlet as described by Kana (1994) was used for the analysis, but without using a cryotrap, as this would have frozen out the methane. Instead, to prevent confounding effects of water vapour, the inlet at the MS side was heated to 180°C. Methane was measured at mass to charge ration (m/z) of 15, as a pre-calibration experiment had shown that this gave the most reliable results. Concentrations of methane were calculated by comparing the ion current at m/z 15 of the sample to the ion current at m/z 15 of air-saturated water at 20°C.

The water in each lake was sampled at three locations with 3 replicates (mixed sample). The water in each ditch was sampled at two locations with 3 replicates (mixed sample). Undisturbed sediment samples were taken from the sediment top layer (upper 10 cm) by means of a plastic cup perforated with holes 2 cm apart at the end of a length-adjustable pipe.

Two of the three water samples were filtered immediately with a Whatman 0.45µm cellulose membrane filter (Whatman International Ltd, Maidstone, England); the third sample was not filtered. All samples were transported in coolers and stored frozen (-20^{o}C) until analyses. The unfiltered water samples were analysed for organic matter (OM) content, %C, %N, Chlorophyll-a content, total N and total P; the filtered samples were analysed for NO_3^-+NO_2^-, NH_4^+, SO_4^{2-}, Fe_2^+ and PO_4^{3-} using a SANplus autoanalyzer (Skalar Analytical, Breda, the Netherlands). Dissolved organic carbon (DOC) and dissolved inorganic carbon (DIC) were measured in filtered samples using a carbon analyser. Total N and total P were measured using a SANplus auto analyser with laser destructor. All these samples were measured in duplicate. Chlorophyll-a content in unfiltered samples from the microcosms was measured using a phytoPAM fluorometer (Heinz Waltz GmbH, Effeltrich, Germany). For the sediment samples a $CaCl_2$ extraction was used to obtain the available PO_4-P, NH_4-N and NO_3-N, and an ammonium oxalate extraction was used to obtain the active form of Fe.

6.2.3. Data analysis

Correlations between the measured variables and fluxes of CO_2 and CH_4 were first tested by using Pearson correlation analysis. Data were tested for normality. We used stepwise, multiple linear regression analyses to quantify the relationships between environmental variables and fluxes of CO_2 and CH_4 (SPSS 15.0). The variables that significantly enhanced the emissions of CO_2 and CH_4 were selected and were used to build regression models. Differences in the fluxes and variables between and within lakes and ditches were tested using one-way ANOVA (SPSS 15.0).

6.3. Results

6.3.1. Climatic variables

During the sampling period the mean day air temperatures ranged from 15 to 25 ^{o}C, the average temperature at the surface of the water bodies studied ranged from 19.2 ^{o}C to 25.4 ^{o}C and the wind speed at 3 m above water level ranged from 2.1 to 4.5 m s^{-1} (*Table 6.3.*).

Table 6.3*. Average (mean ± SD) water temperatures at 10 cm depth (T_{10} in ^{o}C), at 30 cm depth (T_{30} in ^{o}C) and average wind speed (U in m s^{-1}) during the measurement. For the location codes see Table 6.1. and Table 6.2.*

	Mean T_{10}	Mean T_{30}	Mean U
D1	20.1 (± 0.1)	19.5 (± 0.9)	4.1
D2	25.4 (± 2.5)	25.2 (± 2.4)	3.0
D3	19.2 (± 0.6)	18.4 (± 0.4)	2.7
D4	25 (± 0.5)	25.8 (± 0.3)	3.0
D5	22.3 (± 0.3)	22 (± 0.1)	4.6
D6	20.2 (±0.7)	19.1 (± 1.4)	2.1
D7	22.8 (± 0.2)	21.8 (± 1.2)	3.9
L1	23.6 (± 3.0)	23.7 (± 0.5)	3.0
L2	21.4 (± 1.1)	20.3 (± 0.5)	3.6
L3	23.3 (± 1.1)	23.2 (± 1.1)	3.0
L4	23.2 (±1.4)	22.7 (± 1.2	3.9
L5	25 (± 1.2)	24.9 (± 0.6)	3.9

6.3.2. Characteristics of lakes and drainage ditches

The lakes and drainage ditches were humic, shallow and nutrient-rich. The sediment in D6 and D7 had the lowest organic matter content because these two ditches are located in an area with shallow peat on sand. The EC in all the lakes and ditches sampled ranged from 269 – 866 μS cm^{-1}; the pH ranged from 6.8 – 9.0, with the highest values in the lakes (*Table 6.4.*).

Table 6.4. *General characteristics of the lakes and ditches sampled, with standard deviations for lake/ditch depth, EC, pH and percentage organic matter in the sediment. Lake area was determined from Top10vector maps: TDN (2006).*

Site	Lake size (ha) or Ditch width (m)	Sediment type	Mean depth (cm)	Mean EC (µS cm^{-1})	Mean pH	Organic matter sediment (% dry weight)
L1	927	Peat	209.2 (±42)	508.7 (±53)	9.0 (±0.3)	46.63 (±9. 5)
L2	1079	Peat	240.0 (±42)	866.0 (±13)	8.4 (±0.1)	36. 89 (±12.9)
L3	676	Peat	253.3 (±71)	392.5 (±2)	8.2 (±0.1)	63.08 (±14.1)
L4	613	Peat on sand	178.0 (± 27)	398.0 (±9)	8.4 (±0.1)	58.72 (±3.4)
L5	141	Peat on sand	120.0 (±21)	393.0 (±15)	8.4 (±0.2)	59.81 (±5.8)
D1	8	Peat	25.0 (±3)	523.0 (±38)	9.0 (±0.0)	55.94 (±3.0)
D2	6	Peat	27.5 (±9)	269.3 (±46)	6.8 (±0.3)	57.37 (±7.9)
D3	3	Peat	90.0 (±14)	662.0 (±48)	7.0 (±0.4)	37.77 (±3.7)
D4	3	Peat	86.7 (±8)	386.7 (±2)	7.2 (±0.3)	74.22 (±0.8)
D5	3	Peat	80.0 (±17)	387.0 (±2)	7.7 (±0.1)	73.89 (±0.8)
D6	2	Peat on sand	27.5 (±3)	350.3 (±26)	7.2 (±0.3)	9.53 (±3.1)
D7	6	Peat on sand	42.5 (±3)	395.5 (±6)	8.1 (±0.3)	2.65 (±2.4)

6.3.3. Emissions to the atmosphere

Lakes and drainage ditches studied acted as sources of CO_2 and CH_4 emissions to the atmosphere (*Fig 6.4.* and *Fig 6.5.*), except for L1 where a small uptake of CO_2 was measured. The mean release of both gases to the atmosphere was significantly higher from the ditches than from the lakes ($P<0.001$).

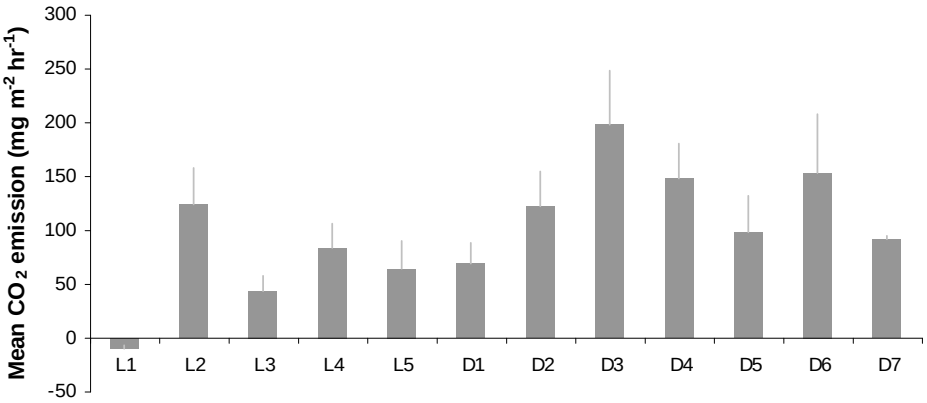


Fig 6.4. *Mean CO_2 fluxes (mg m^{-2} hr^{-1}) and their standard deviations in the sampling period for the ditches (D1 – D7) and lakes (L1 – L5) at different locations in peat areas in the Netherlands. Positive flux values indicate release from the water to the atmosphere.*

The contribution of CO_2 emission compared to CH_4 emission in terms of warming potential is given in *Fig 6.6.*, where CH4 fluxes have been transformed to CO_2 equivalents (CH_4 is 23 times as potent as CO_2).

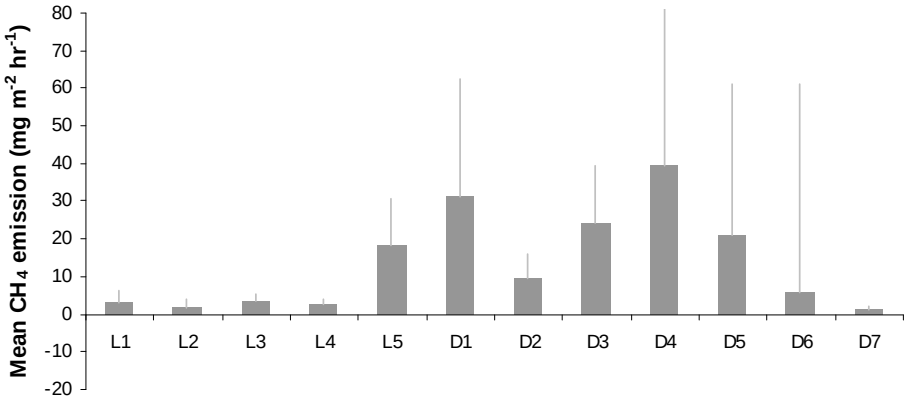


Fig 6.5. *Mean CH4 fluxes (mg m^{-2} hr^{-1}) with their standard deviations for the ditches (D1 – D7) and lakes (L1 – L5) at different locations in peat areas in the Netherlands. Ditches were sampled on one day (n=16 per location) in the period 16 June – 6 July 2009; Lakes were sampled during 2 days in the same period (n=24 per location). Positive flux values indicate release from the water to the atmosphere.*

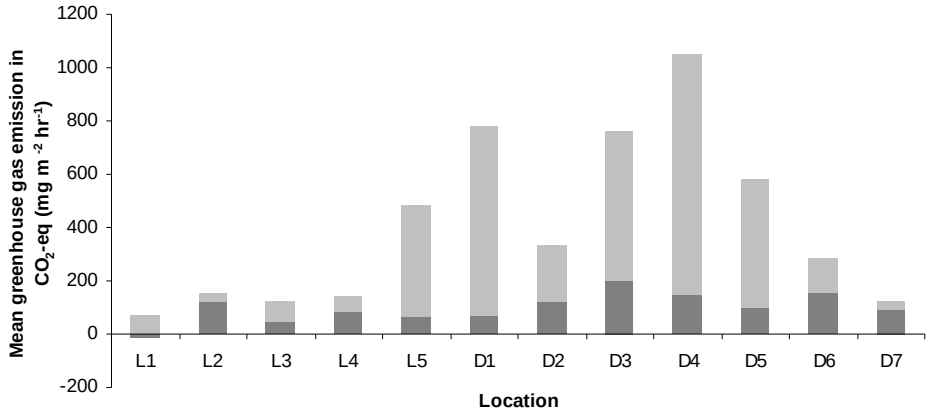


Fig 6.6. *Contribution to greenhouse warming of CO_2 emission (dark grey) compared to CH_4 emission (light grey) in lakes and ditches, given in CO_2 equivalents.*

Lakes

The emission of CO_2 from the lakes (n=93) ranged from -6.0 to 123.9 mg m^2 hr^{-1} and CH_4 emission (n=96) ranged from 1.4 – 18.1 mg m^2 hr^{-1}. The CO_2 fluxes from L1 were significantly lower than those from the other lakes ($P<0.01$); the CO_2 fluxes from L2 were significantly higher than those from L3, L1 and L5 ($P<0.05$). The highest CH_4 emission was measured from L5 and the lowest from L2, but the differences were not significant. The lakes acted as sources of emissions of both gases, except for L1 that acted as a very small sink for CO_2. In terms of warming potential, in 3 lakes the dominant emitted greenhouse gas was CH_4 and in 2 lakes it was CO_2 (*Fig 6.6.*).

Ditches

The emission of CO_2 from the drainage ditches (n=80) ranged from 69.6 mg m^2 hr^{-1} to 199.0 mg m^2 hr^{-1} and CH_4 emission (n=79) ranged from 1.2 to 39.3 mg m^2 hr^{-1}. The CO_2 emission from D3 was significantly higher than the fluxes from D7, D1 and D5. The highest CH_4 emission was measured from D4 and the lowest from D7, but the CH_4 fluxes did not differ significantly because there was great variability among the ditches. In all ditches except D6 and D7, the dominant greenhouse gas in terms of warming potential was CH_4. All ditches acted as sources of emissions of both gases.

Cross-validation

Large-scale CH_4 flux measurements by eddy covariance were performed on one of the lakes (L1) to cross-validate flux values from this homogeneous landscape on a diurnal base. A cross-validation of chamber based CH_4 and CO_2 values and eddy covariance based values is also performed earlier for a more heterogeneous peat area (Schrier-Uijl et al, 2010b). Details for the used eddy covariance instruments have been reported in Veenendaal

et al., 2007 and Kroon et al., 2007. In this study, CH_4 fluxes within the footprint of the eddy covariance system were 5.8 ± 3.26 (mean ± SD, n=24) measured by chambers compared to 4.6 ± 1.3 measured by eddy covariance over a four hour period. It would be of great interest in the future to also use eddy covariance to capture temporal variability of greenhouse gas fluxes (CH_4 and CO_2) from water bodies and to explain more of the measured variability.

6.3.4. Dissolved oxygen and dissolved CH_4

Typical vertical profiles of oxygen saturation during the measurements are shown in *Fig 6.7.*

On average, the lakes had a higher O_2 saturation than ditches. In both types of waterbody, oxygen saturation decreased only slightly at the top of the water column, which suggests that there was hardly any respiration by aquatic organisms. Deeper in the water column the oxygen saturation fell rapidly to values close to 0% just above the sediment. Of the lakes, L2 had the highest O_2 saturation throughout the profile, and of the ditches D3 and D6 had the lowest O_2 saturation.

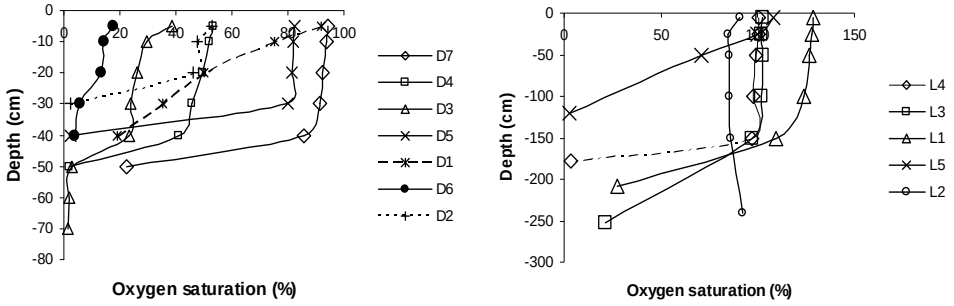


***Fig 6.7**. Oxygen saturation (%) in the water columns of lakes (right) and drainage ditches (left) at different depths in the water column at the time of the measurements.*

Dissolved CH_4 concentrations were measured at three depths: at the top and middle of the water column and just above the ditch sediments. In all the ditches the dissolved CH_4 concentrations increased with depth (*Fig 6.8.*).

Fig 6.8. *Concentrations of dissolved CH_4 ($\mu g\ l^{-1}$) in the water of ditches at the top, middle and bottom of the water column. The y-axis shows the depth (cm).*

D1 and D3 had a high dissolved CH_4 concentration and also a high CH_4 emission (*Fig 6.5.*). None of the following variables correlated significantly with the dissolved CH_4 in the ditch water immediately above the sediment or with the difference between dissolved CH4 concentration at the water surface and at the sediment surface: nutrient content (NO_3^-, NH_4^+, Fe, PO_4^{3-}); sediment oxygen demand (SOD); O_2 saturation of the water, organic matter content (% organic matter, %N, %C); amount of green algae and plants. We did not find any significant correlation between dissolved CH_4 concentration at the water surface and CH_4 release to the atmosphere. The oxygen saturation at the sediment surface correlated negatively with CH_4 emission to the atmosphere (P=0.065).

6.3.5. The variables measured and their correlation with CH_4 and CO_2 emission

Climate, depth, EC and pH

Climatic conditions in the three-week sampling period were stable. No significant correlation was found between the CO_2 and CH_4 fluxes and the temperature or the wind velocity. Neither the depth of water in the ditches (range 0.28 – 0.90 m) or the depth of water in the lakes (range 1.20 m – 2.53 m) correlated significantly with CO_2 or CH_4 release to the atmosphere, although the deepest lakes tended to have the lowest CH_4 and CO_2 emissions. A positive correlation was found between EC and CO_2 flux and a significant negative correlation was found between CO_2 emission and the pH of the water (r=-0.81; p=0.001). Though the correlation between CH_4 emission and pH was also negative, it was not significant (r=-0.23; p=0.41).

Nutrients and organic matter in water and sediment

The percentage of N measured in the lake sediments was significantly positively correlated with the release of CO_2 (P<0.01); in ditches, the %N and the %OM measured in the sediments were significantly positively correlated with the release of CH_4 (p=0.02 and p=0.05, respectively). The lowest organic matter contents of the sediments were found in

D6, D7 and L2 (*Table 6.4.*), which had the lowest CH_4 fluxes (*Fig 6.5.*). In this study neither the DIC nor the DOC correlated significantly with the CO_2 flux or the CH_4 flux.

The ammonium ($NH4^+$) concentration in the water ranged from 0.1 to 478.6 µg NH_4-N l^{-1}: the highest concentrations were found in D3, D6 and L2. Ammonium concentration correlated positively with CO_2 emission (r=0.67; P<0.05). The NO_3^- concentration in the water was around 0 mg N l^{-1} – except for L2 and L4, where the mean concentrations were 0.43 and 0.12 mg N l^{-1}, respectively. In the sediment of the lakes and ditches the NH_4^+ concentrations ranged from 12.3 to 478.1 mg kg^{-1} dry weight with the highest concentrations in D4 (324.2 mg kg^{-1} dry weight) and D5 (478.1 mg kg^{-1} dry weight). The NO_3-N concentration ranged from 0.0 to 3.55 mg kg^{-1} dry weight, with the highest concentration in D4 and the lowest in L4. The only lake with high NO_3 concentrations in the water and sediment was L2: it was also the only lake where measurable N_2O emissions were observed (0.163 mg m^{-2} hr^{-1}, n=23). See *Table 6.5.* for the concentrations of NH_4^+ and *Table 6.6.* for the concentrations of NO_3-N.

Table 6.5. *Chemical composition of the water of lakes and ditches with means for sulphate (SO_4^{2-}), iron (Fe^{2+}), phosphate (PO_4^{3-}), ammonium (NH_4^+), nitrate (NO_3^-) and total P. For location codes see Table 6.1. and Table 6.2.*

Location	SO_4^{2-}	Fe^{2+}	PO_4^{3-}	NH_4^+	NO_3^-	Ptot
	(mg l^{-1})	(µg l^{-1})	(µg P l^{-1})	(µg N l^{-1})	(mg N l^{-1})	(µg P l^{-1})
D1	47.1	324.3	98	13.8	0.00	153.4
D2	16.6	864.5	33	0.1	0.00	66.3
D3	3.4	1032.0	291	478.6	0.00	515.7
D4	21.6	58.5	11	8.8	0.00	18.3
D5	22.1	26.7	24	9.2	0.00	15.0
D6	7.1	470.5	240	166.4	0.00	201.0
D7	19.2	146.7	45	3.8	0.00	15.8
L1	34.8	48	53	12.2	0.00	11.3
L2	53.4	59.7	11	76	0.42	14.0
L3	21.0	31.7	25	11.5	0.00	10.3
L4	18.2	113.5	10	15.8	0.12	12.6
L5	16.9	640.0	18	6.1	0.00	25.4

A weak negative correlation was found between the SO_4^{2-} concentrations in lake water (range 17.8 – 53.4 mg l^{-1}) and ditch water (range 3.4-47.1 mg l^{-1}) and CO_2 and CH_4 fluxes (r=-0.43, p=0.16; r=-0.1, p=0.81). The water of D3 had the lowest SO_4^{2-} concentrations of all the lakes and ditches sampled; this ditch had a high Fe concentration in its water and sediment (*Table 6.6.*), which suggests the binding of SO_4^{2-} to iron. Overall, the SO_4^{2-} concentrations did not significantly differ between the ditches and lakes. The Pearson correlations between CO_2, CH_4 and Fe^{2+} in the water were weakly positive; the Fe^{2+} concentrations we measured ranged from 33.6 µg l^{-1} to 1032 µg l^{-1}, with an average of 301.6 µg l^{-1}. Methane emission correlated significantly positively with the PO_4^{3-} concentration of the sediments of the lakes and ditches (r=0.77, p=0.81). The PO_4^{3-}

concentration of the sediments correlated positively both with the Fe concentration and the SO_4^{2-} concentration of water. The total P concentration in the water correlated positively with CO2 emission, indicating the high availability of organic substrates. The nutrient concentrations are given in *Table 6.5* and *Table 6.6.*

Table 6.6. *Means for Fe, PO_4-P, NH_4-N, NO_3-N in the top10 cm of the bottom sediments of the sampled lakes and drainage ditches.*

Location	Fe	PO_4-P	NH_4-N	NO_3-N
	(mg kg^{-1})	(mg kg^{-1})	(mg kg^{-1})	(mg kg^{-1})
D1	2.7	8.6	154.9	0.4
D2	3.0	0.6	215.4	0.7
D3	7.0	0.3	187.8	0.9
D4	6.0	20.7	478.1	3.6
D5	3.1	8.8	323.6	0.5
D6	2.0	0.2	51.9	0.2
D7	0.8	0.1	12.3	0.1
L1	1.5	2.6	106.0	0.4
L2	1.7	3.3	8.0	1.2
L3	0.5	5.2	123.7	0.3
L4	5.7	0.5	86.7	0
L5	na	na	na	na

6.3.6. Multiple regression analyses

`

Multiple regressions with stepwise elimination of variables showed that for summer CH_4 fluxes the PO_4^{3-} concentrations in the sediment and the Fe^{2+} concentrations in the water explained 87% of the variation when Eq. 6.1 was used:

$$F_{CH4} = -3.482 + 2.183[PO4]_{sediment} + 22.116[FE]_{water} \qquad \textbf{6.1}$$

Where F_{CH4} is the CH_4 flux (mg m^{-2} hr^{-1}), $[PO_4]_{sediment}$ is the PO_4-P concentration in the sediment (mg kg-1) and $[FE]_{water}$ is the concentration of Fe^{2+} in the water (mg l^{-1}). *Table 6.7.* presents statistical details of the model. *Fig 6.9.* shows the measured CH_4 fluxes versus the CH_4 fluxes in the sampled lakes and ditches, modelled by means of Eq. 6.1.

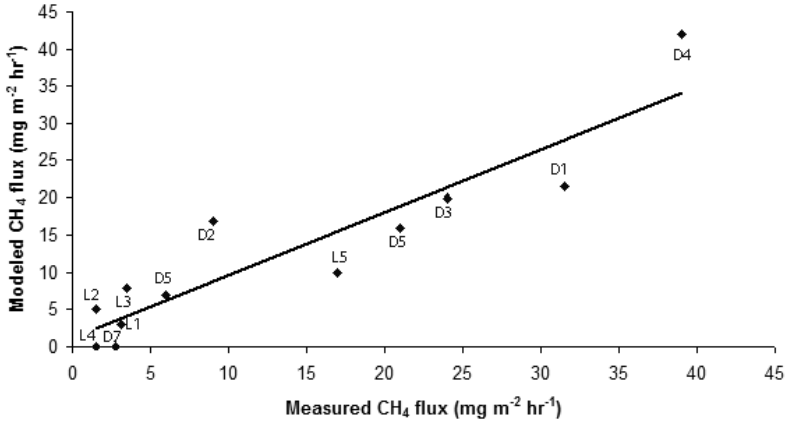


Fig 6.9. *Measured CH_4 fluxes versus modeled CH_4 fluxes (mg m^{-2} hr^{-1}) based on a multiple regression model with PO_4-P concentration in the sediment and Fe^{2+} concentration in the water as explanatory variables.*

By performing regression analyses with ditches only, the fit of the regression improved to $R^2 = 0.94$.

For CO_2, a regression model with mean depth of lake or ditch water and the EC and pH of the water as independent variables explained up to 89% of the variation in summer CO_2 emission at the water–atmosphere interface when Eq. 6.2 was used:

$$F_{CO2} = 477.359 - 0.213depth + 0.171EC - 54.4pH \qquad \textbf{6.2}$$

Where F_{CO2} is the CO_2 flux, depth is the mean depth of the water in the sampled lake or ditch (cm), EC is the mean electrical conductivity and pH is the mean pH in the sampled lakes and ditches. *Fig 6.10.* shows the measured CO_2 fluxes in lakes and ditches versus the modelled CO_2 fluxes by means of Eq. 6.2. *Table 6.7.* presents statistical details of the model.

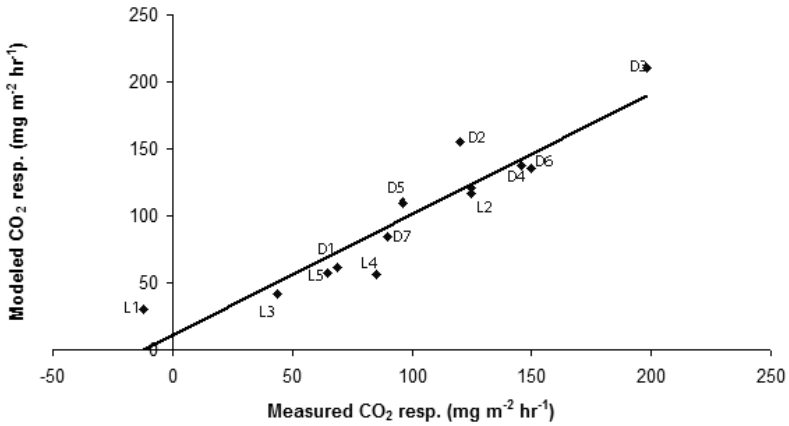


Fig 6.10. *Measured CO_2 flux versus modeled CO_2 flux (mg m^{-2} hr^{-1}) based on a multiple regression model with mean depth of water, EC and pH as explanatory variables.*

Regression analyses on the data from the ditches only improved the predictive power of the regression to $R^2 = 0.91$.

Table 6.7. *Statistical details of the CH_4 and the CO_2 multiple regression models with PO_4 concentration in the sediment, Fe concentration in the water, depth, EC and pH as explanatory variables: R square of the model, significance of the model, Pearson correlations and significance of the separate variables.*

	R^2 model	*P* model	Pearson corr.			*P* parameter		
			$PO_{4sediment}$		Fe_{water}	$PO_{4sediment}$	Fe_{water}	
CH_4 model	0.87	0.000	0.77		0.19	0.000	0.003	
			Depth	EC	*pH*	Depth	EC	*pH*
CO_2 model	0.89	0.000	-0.62	0.17	-0.81	0.06	0.004	0.001

6.4. Discussion

In this study we determined the magnitude of CH_4 and CO_2 fluxes from 12 water bodies in Dutch wetlands during a three weeks period in the summer season and studied the factors that might regulate emissions of CH_4 and CO_2 from these lakes and ditches. During this period the lakes and ditches acted as CO_2 and CH_4 sources of emissions to the atmosphere; the fluxes from the ditches were significantly larger. One lake (L1) was in equilibrium with the atmosphere in terms of CO_2 emission. Kosten et al, 2010 found that <10% of lakes worldwide are in equilibrium with the atmosphere in terms of pCO_2, and they found that most other lakes are CO_2 sources. Compared with other studies, the lake emissions

founding our study were in the intermediate to high range (see *Table 6.8.* for CH_4 fluxes). For example, Rantakari (2008) found CO_2 fluxes in the range 7.48-11.5 mg m^{-2} hr^{-1} in 37 boreal Finnish lakes. The average CH_4 emission from our drainage ditches was higher than the lake fluxes found in other studies (*Table 6.8.*). As the CH_4 emissions measured by the gas analyser within the footprint area of an eddy covariance system at location L2 agreed within the uncertainty limits with the EC system, we are confident that our measurement technique provided reliable flux estimates that are applicable to larger areas (Kroon et al. 2007; Schrier-Uijl et al. 2010b).

Table 6.8. *Comparison between the CH_4 emission rates in this study and the CH_4 emission rates reported in other studies. Mean CH_4 emission rates are in mg CH_4 m^{-2} hr^{-1}. The period of sampling and the location are given.*

Reference	System	Location	Sampling period	Flux CH_4 (mg m^{-2} hr^{-1})
Guerin et al. 2007	Tropical Lake	French Guiana	late spring	4±4.7
Juutinen et al. 2009	30 Eutrophic Boreal Lakes	Finland	all seasons	Median of 0.137
Stadmark and Leonardson., 2005	3 Ponds	South Sweden	summer	10
Huttunen et al. 2002	Boreal lakes	Finland	summer	1.0
Bastviken et al. 2004	11 lakes	North America	summer 2000	Range 0.15 – 3.2
Repo et al. 2007	3 boreal lakes	Siberia	Summer	0.34
Present research	5 temperate lakes	Netherlands	early summer 2009	1.4-18.1, mean 5.0
Present research	7 drainage ditches	Netherlands	early summer 2009	1.2-39.3, mean 18.8

For units: mg CH_4 m^{-2} hr^{-1} refer to mg CH_4 emitted per m^2 of water area per hour.

The temporal variability of emissions of CH_4 and CO_2 from water bodies is normally found to be related to temperature and wind velocity when measuring over longer time spans (e.g. Stadmark et al. 2005; Frei et al. 2006; Hendriks et al. 2007; Repo et al. 2007; Schrier-Uijl et al. 2010a,b; Kroon et al, in press). However, a large part of the variability of fluxes cannot be explained by temperature or wind velocity only. Our results refer to data collected during summer, a period in which around 70% of the annual ditch emissions are generated (Schrier-Uijl et al, 2010). The study did not last long enough to include seasonal patterns of CH_4 and CO_2 production and emission. Diurnal stratification and mixing due to day–night temperature differences may bias flux estimates if the only measurements available are from the daytime (Repo et al. 2007). In Schrier-Uijl et al. (2010b) the diurnal variation of CH_4 fluxes over an area with fields and ditches was tested in October/November 2006. After correction for temperature dependency, the emission of CH_4 did not differ significantly between day and night. Nevertheless, there could be diurnal variation of fluxes from water bodies, because less oxygen will be produced at night, which will result in a lower redox potential and higher CH_4 production. In addition, less CO_2 will be taken up by aquatic plants at night, because then they are not photosynthesising. These effects should be considered when estimating annual fluxes from water bodies by using continuous

measurements such as eddy covariance. In our study, only diffusive fluxes of CO_2 and CH_4 were measured; however, ebullition can also contribute to the emission of CH_4 from water bodies (Walter et al. 2006; Walter et al. 2007). While sampling ditches D5 and D4, we observed ebullition, so it is possible that we underestimated the release of CH_4 fluxes. In a summer study done by Repo et al. (2007) in Siberian water bodies, ebullition was observed in two of the three lakes sampled (depth < 1.5 m) and accounted for 19-37% and 11-40% of the total CH_4 emissions from these two lakes.

The fact that the lakes and ditches acted as sources for CH_4 and CO_2 indicates that CO_2 production exceeded CO_2 uptake during photosynthesis by plants (Riera et al. 1999) and that CH_4 production exceeded CH_4 oxidation. Our observation that the deeper, mostly less eutrophic lakes with low EC had the smallest fluxes agrees with findings reported for lakes in the boreal zone in Finland (depth range 3.8 – 26.5 m) (Juutinen et al. 2009). Deeper water bodies usually have less degradable organic matter and more oxidation of CH_4 than shallow lakes, because the transport pathway is longer (e.g. Borges et al. 2004). The EC, which is an indicator of trophic status in fresh water lakes, and the depth of the water body were two of the three significant predictors in the regression analyses for CO_2 fluxes.

The pH correlated negatively with emissions from both gases, yet at lower pH values (pH < 7) the correlations are usually positive (e.g. Inubushi et al. 2005). CO_2 enters the water as a result of the biological processes of organic carbon degradation and respiration by plants. In our ecosystems it is likely that through uptake of CO_2 by plants during the day in the growing season, HCO_3^- is transformed to CO_2, causing the HCO_3^- concentration to decline and diminishing the buffering effect. This reduced buffering effect can result in pH values above 9.0 and in a negative relation between CO_2 flux and pH. Incorporating pH in the regression equation for CO_2 significantly improved the equation's predictive power. In peat soils in temperate areas the optimum pH for methanogenesis is between 5.5 and 7.0, which explains the slightly negative correlation we found between CH_4 emission and pH (Le Mer and Roger, 2001).

Water turbulence due to wind can increase mixing of oxygen in the water. This is illustrated by the higher oxygen concentration throughout the water column of L2, which had been subjected to high wind speeds on the day before sampling. The high O_2 saturation in L2 corresponded with higher CO_2 fluxes and very low CH_4 fluxes, illustrating the oxidation of CH_4 to CO_2. The opposite can be seen in D3, D6 and L5, where the oxygen concentrations at the water surface were low and the CH_4 fluxes were high, illustrating the low turnover of CH_4 carbon to CO_2. The fast decrease in dissolved CH_4 from the sediments to the water surface in D3 and D1 indicates that most of the dissolved CH_4 is oxidised during transport or passes through the water column and escapes to the atmosphere very quickly. As found in other studies, dissolved CH_4 poorly predicted the diffusive fluxes at the water–air interface (e.g. Huttunen et al. 2006; Juutinen et al. 2009). The factors responsible for this

finding could be variation in duration of storage, release of CH_4 to the atmosphere and complex processes during transport of CH_4 through the water column (e.g. Kankaala et al. 2007).

The input of organic matter as a substrate in the lake or drainage ditch system increases the availability of substrates and this can increase the production of CO_2 and CH_4 (e.g. Casper et al. 1992) and increases the possibility of minimising the competition for electron donors between methanogenesis and other anaerobic processes (Scholten et al. 2002; Scheid et al. 2003). As long as O_2 reaches the sediments, it will act as the primary oxidant of organic matter.

Permanently anaerobic conditions in the sediment may hamper nitrification of NH_4^+ to NO_3^-, but stimulate denitrification of NO_3^- to N_2 by microorganisms, leading to a high NH_4^+ to NO_3^- ratio, as was found in this study. Also, the greater availability of NH_4^+ compared to NO_3^- suggests the occurrence of dissimilatory reduction of NO_3^- to NH_4^+ (DNRA) under anaerobic conditions in these ditches. DNRA is likely to occur in the organic sediments that we sampled, as this process usually occurs at high carbon inputs (Burgin, 2007). As our ditch systems did not contain much aquatic plant biomass, it is unlikely that the NO_3^- uptake by plants and algae was influential in the ditches. Other possible sources of NH_4^+ in the water could be cation exchange of adsorbed NH_4^+ by Fe_2^+ (but this only occurs at very high Fe_2^+ concentrations: Loeb et al. 2007), and leaching through groundwater from surrounding, managed agricultural areas. NH_4^+ inhibits methanotrophy and therefore may reduce CH_4 oxidation and increase its emission (Conrad and Rothfuss, 1991), which may explain the positive correlation between the NH_4^+ and CH_4 fluxes in our study. The positive correlation of NH_4^+ with CO_2 emission is in line with the findings of other studies (e.g. Cicerone and Shetter, 1988).

Our finding is that the two most significant predictors of CH_4 fluxes were the PO_4^{3-} concentration in the sediment of lakes and ditches and the Fe^{2+} concentration in the water of lakes and ditches. In anaerobic sediments, Fe^{3+} will be reduced to Fe^{2+}. At the sediment–water interface some of the Fe^{2+} will be oxidized to Fe^{3+} (how much depends on the oxygen concentration just above the sediment) and some of this will be released into the water. Thus a high concentration of Fe^{2+} in the water is related to anaerobic conditions. Both Fe concentration and SO_4^{2-} concentration correlate with PO_4^{3-} availability at the sediment–water interface. The PO_4^{3-} in sediments is bound to Fe^{3+} and when the Fe^{3+} is reduced to Fe^{2+}, PO_4^{3-} will be released to the water (e.g. Smolders et al. 2006; Smolder and Roelofs, 1993).

Overall, a higher trophic status was positively correlated with summer emissions of CO_2 and CH_4, while the depth of the water and the pH were inversely correlated with CO_2 emission. It is therefore likely that decreasing the inputs of organic matter and nutrients (for

example, by changing the management of the surrounding areas) will reduce emissions and that this effect will be strongest in drainage ditches.

Much of the uncertainty in flux estimates is due to temporal variation. So, also diurnal, seasonal, annual and inter annual variability must be studied in more detail to get insight in climatic responses, extreme drought/rainfall events and the influence of management in the surrounding catchments. In this respect, there is a need for long-term, continuous measurements of emissions (e.g. by eddy covariance).

6.5. Conclusion

The current study focused on emissions from temperate, shallow lakes (n=5) and drainage ditches (n=14) in agricultural peat areas in the Netherlands. It was found that in general, both these types of waterbodies are important sources of CO_2 and CH_4. The ditches had significantly higher CO_2 and CH_4 fluxes than the lakes. Trophic status was an important indicator of the magnitude of fluxes. 87% of the variation in the summer fluxes of CH_4 could be explained by PO_4^{3-} in the sediment and Fe^{2+} concentration in the water, and 89% of the CO_2 flux could be explained by water depth, EC and pH. Our results can be used to refine greenhouse gas emission inventories and to ascertain possible ways of reducing the release of CO_2 and CH_4 from water bodies to the atmosphere. Decreasing the nutrient loads and input of organic substrates to ditches and lakes will likely reduce summer emissions of CO_2 and CH_4 from these water bodies.

6.6. Acknowledgements

Many thanks to Roos Loeb and Joop Harmsen for their critical comments on an earlier version of this manuscript, and to Jan van Walsem, Frits Gilissen and John Beijer for their assistance in the field and in the laboratory. We are also grateful to the State Forestry Service for the use of their boats and to Arjan Hensen and Petra Kroon who carried out the eddy covariance measurements at L1. Joy Burrough advised on the English. This study was funded by Wageningen University, The Province of North Holland, CarboEurope-IP and the Klimaat voor Ruimte Project (BSIK).

6.7. References

Bastviken, D, Ejlertsson, J, Tranvik, L (2002): Measurement of methane oxidation in lakes: A comparison of methods. Env Sci Tech 36 3354–3361.

Bastviken, D, Cole, J, Pace, M et al (2004): Methane emissions from lakes: Dependence of lake characteristics two regional assessments and a global estimate. Glob Biogeochem Cycles 18 GB4009 doi: 10.1029/2004GB002238.

Burgin, AJ and Hamilton, SK (2007): Have we overemphasized the role of denitrification in aquatic ecosystems? A review of nitrate removal pathways. Front Ecol Environ 5 (2) 89-96.

Conrad, R and Rothfuss, F (1991): Methane oxidation in the soil surface-layer of a flooded rice field and the effect of ammonium. Biol And Fert Of soils 12 1: 28-32.

Casper, P (1992): Methane production in lakes of different trophic state. Arch Hydrobiol Beih Ergebn Limnol 37: 149–154.

Casper, P, Chan, OC, Furtrado, ALS et al (2003): Methane in an acidic bog lake: The influence of peat in the catchment on the biogeochemistry of methane. Aquat Sci 65 36-46 doi:1015-1621/03/010036-11.

Cole, JJ, Caraco, NF, Kling, GW et al (1994): Carbon-dioxide supersaturation in the surface waters of lakes. Science 265 5178: 1568-1570.

De Haan, H, van Liere, L, Klapwijk, SP et al (1993): The structure and function of fen lakes in relation to water table management in The Netherlands. Hydrobiologia 265 1-3, doi: 10.1007/BF00007266.

Frei, M, Razzak, MA, Hossain, MM et al (2006): Methane emissions and related physicochemical soil and water parameters in rice-fish systems in Bangladesh. Agr Ecosyst and Env 120:391-398 doi: 10/1016/j.agee.

Gulati, R and van Donk, E (2002): Lakes in the Netherlands their origin eutrophication and restoration: state-of-the-art review. Hydrobiologia 478: 73-106.

Hendriks, DMD, van Huissteden, J, Dolman, AJ et al (2007): The full greenhouse gas balance of an abandoned peat meadow. Biogeosciences Discussions 4 277-316.

Huttunen, JT, Väisänen, T, Heikkinen, M et al (2002): Exchange of CO_2 CH_4 and N_2O between the atmosphere and two northern boreal ponds with catchments dominated by peatlands or forests. Plant Soil 242 137–146.

Huttunen, JT, Väisänen, T, Helllsten, SK et al (2006): Methane fluxes at the sediment-water interface in some boreal lakes and reservoirs. Boreal Env Res 11:27-34 ISSN 1239-6095.

Inubushi, K, Otake, S, Furukawa, Y et al (2005): Factors influencing methane emission from peat soils: Comparison of tropical and temperate wetlands. Nutr Cycl Agroecosyst 71: 93-99.

Juutinen, S, Rantakari, M, Kortelaine, P et al (2009): Methane dynamics in different boreal lake types. Biogeosciences 6 209-223.

Kana, TM (1994): Membrane inlet mass spectrometer for rapid high-precision determination of N_2 O_2 and Air in environmental water samples. Analytical chemistry 66: 4166.

Kankaala, P, Taipale, S, Nykanen, H et al (2003): Oxidation, efflux, and isotopic fractionation of methane during autumnal turnover in a polyhumic, boreal lake. J Geoph Res-Biogeosc. 112 G2 G02003.

Kroon, PS, Hensen, A, Zahniser, MS et al (2007): Suitability of quantum cascade laser spectrometry for CH_4 and N_2O eddy covariance measurements. Biogeosciences 4, 715–728.

Kroon, PS, Schrier-Uijl, AP, Hensen, A et al (accepted): Annual balances of CH_4 and N_2O from a managed fen meadow using eddy covariance flux measurements. Eur J Soil Sci.

Lamers, PM, van Roozendaal, ME, Roelofs GM (1998): Acidification of freshwater wetlands: Combined effects of non-airborne sulfur pollution and desiccation. Water, Air and Soil Poll. 105: 95-106.

Le Mer, J and Roger, P (2001): Production oxidation emission and consumption of methane by soils: A review. Eur J Soil Biol 37 1:25-50.

Lloyd, D and Scott, RI (1983): Direct measurement of dissolved gases in microbiological systems using membrane inlet mass spectrometry. J of Microbiol Meth 1: 313-328.

Loeb, R, van Daalen, E, Lamers LPM et al (2007): How soil characteristics and water quality influence the biogeochemical response to flooding in riverine wetlands. Biogeochemistry 85: 289-302 doi:10.1007/s1-533-007-9135-x.

Minkkinen, K and Laine, J (2006): Vegetation heterogeneity and ditches create spatial variability in methane emissions from peatlands drained for forestry. Plant Soil 285 289-304.

Rantakari, M and Kortelainen, P (2005): Interannual variation and climatic regulation of the CO_2 emission from large boreal lakes. Glob Change Biol 11 1368–1380.

Rantakari, M and Kortelainen, P (2008): Controls of organic and inorganic carbon in randomly selected Boreal lakes in varied catchments. Biogeochem 91 2-3 151-162.

Repo, ME, Huttunen, JT, Naumov, AV et al (2007): Release of CO_2 and CH_4 from small wetland lakes in western Siberia. Tellus 59 788–796 doi:10.1111/j.1600-0889.2007.00301.x.

Roulet, NT and Moore, TR (1995): The effect of forestry drainage practices on the emission of methane from northern peatlands Can Journ Of For Res 25 3: 491-499.

Saarnio, S, Winiwater, W, Leitão J (2009): Methane release from wetlands and watercourses in Europe. Atm Env 43 1421–1429 doi:10.1016/j.atmosenv. 2008.04.007.

Scheid, D, Stubner, S, Conrad, R (2003): Effects of nitrate- and sulfate-amendment on the methanogenic populations in rice root incubations. Fems Microbiol Ecol 43 3: 309-315.

Scholten, JCM, Conrad, R, Stams, AJM (2002): Effect of 2-bromo-ethane sulfonate molybdate and chloroform on acetate consumption by methanogenic and sulfate-reducing populations in freshwater sediment. Fems Microbiol Ecol 32 1: 35-42.

Schrier-Uijl, AP, Kroon, PS, Leffelaar, PA et al (2009a): Methane emissions in two drained peat agro-ecosystems with high and low agricultural intensity. Plant Soil DOI 10.1007/s11104-009-0180-12009.

Schrier-Uijl, AP, Kroon, PS, Hensen, A et al (2009b): Comparison of chamber and eddy covariance based CO_2 and CH_4 emission estimates in a heterogeneous grass ecosystem on peat. Agric. For. Meteorol., doi:10.1016/j.agrformet.2009.11.007.

Smolders, AJP and Roelofs, JGM (1993): Sulphate-mediated iron limitation and eutrophication in aquatic ecosystems. Aquatic Botany 46 247-253.

Smolders AJP, Lamers LPM, Lucassen ECHET et al (2006): Internal eutrophication: How it works and what to do about it - a review. Chem Ecol 22 2 93-111.

Stadmark, J and Leonardson, L (2005): Emissions of greenhouse gases from ponds constructed for nitrogen removal. Ecol Engineering 25:542-551 doi: 10.1016/j.ecoleng.2005.07.004.

St. Louis, VL, Duchemin, CAE, Rudd, JWM et al (2000): Reservoir surfaces as sources of greenhouse gases to the atmosphere: A global estimate. Bioscience 50 766 – 775 doi: 10.1641/0006-3568(2000)050[0766: RSASOG.

TOP10vector Product information (in Dutch). Topografische Dienst Kadaster Emmen the Netherlands. Available at http://www.tdn.nl/ (verified 6 Dec 2007).

Valentine, DW, Holland, EA, Schimel, DS (1994): Ecosystem and physiological controls over methane production in Northern wetlands. J Geophys Res Atmosph 99 D1: 1563-1571.

van Bodegom, PM and Scholten, JCM (2001): Microbial processes of CH_4 production in a rice paddy soil: Model and experimental validation. Geochemica et Cosmochimica Acta 65 13: 2055-2066.

van Huissteden, J, Maximov, TC, Dolman AJ (2005): High methane flux from an arctic floodplain (indigirka lowlands eastern siberia) J Geophys Res-Biogeosci 110.

Veenendaal EM, Kolle O, Leffelaar PA et al (2007) CO_2 exchange and carbon balance in two grassland sites on eutrophic drained peat soils. Biogeosciences: 4 1027-1040.

Walter, KM, Zimov, SA, Chanton, JP (2006): Methane bubbling from Siberian thaw lakes as a positive feedback to climate warming. Nature 443 71– 75 doi:10.1038/nature05040.

Walter, KM, Smith, LC, Chapin, FS (2007): Methane bubbling from northern lakes: present and future contributions to the global methane budget. Phil Trans Royal Soc 365 1657–1676.

Wang, HJ, Lu, JW, Wang, WD (2006): Methane fluxes from the littoral zone of hyper eutrophic Taihu Lake China. J Geophys Res 111 D17109 doi: 10.1029/2005JD006864.

Whiting, GJ and Chanton, JP (2001): Greenhouse carbon balance of wetlands: methane emissions versus carbon sequestration Tellus 53B 521-528 ISSN 0280-6509 2001.

Chapter 7

Agricultural peat lands; towards a greenhouse gas sink – a synthesis

We examined how mitigation through the reduction of management and rewetting may affect the greenhouse gas (GHG) emission and the carbon balance of intensively managed agricultural peatlands. Balances were determined for three peatlands in the Netherlands from 2005 to 2008 by taking into account the landscape elements that contributed differently. One area (Oukoop) is an intensively managed peatland, including a dairy farm, with the average water table at an average depth of 0.55 m. The second area (Stein) is an extensively managed peatland with a dynamic water table at an average depth of 0.45 m, and the third area is an (since 15 years) abandoned former agricultural peatland with the average water table at a depth of 0.2 m. The managed peatlands acted as terrestrial GHG sources: 1.4 kg CO_2-eq m^{-2} yr^{-1} for the intensively managed area and 1.0 kg CO_2-eq m^{-2} yr^{-1} for the extensively managed area; the unmanaged area acted as a GHG sink of 0.7 kg CO_2-eq m^{-2} yr^{-1}. Adding the farm-based CO_2 and CH_4 emissions increased the source strength for the managed sites to 2.7 kg CO_2-eq m^{-2} yr^{-1} for Oukoop and 2.1 CO_2-eq m^{-2} yr^{-1} for Stein. Shifting from intensively managed to extensively managed grass-on-peat reduced GHG emissions mainly because N_2O emission and farm-based CH_4 emissions decreased. Both agriculturally managed sites acted as carbon sources but the former agricultural peatland acted as a carbon sink. An additional study in summer showed that water bodies contribute significantly to the GHG balance and that reduction of anthropogenic disturbances, such as eutrophication, in the catchment areas of these water bodies will probably reduce summer emissions. Overall, this study suggests that intensively managed peatlands are large sources of GHG and carbon, but, if appropriate measures are taken, fen meadow ecosystems can be turned back into GHG and carbon sinks within 15 years of abandonment.

A.P. Schrier-Uijl, D.M.D. Hendriks, P.S. Kroon, A. Hensen, J. van Huissteden, P. A. Leffelaar, L. Nol, E.M. Veenendaal and F. Berendse

7.1. Introduction

As a result of human activities, the global mean surface temperature rose by about 0.6 ± 0.2 ^{o}C (IPCC 2007) during the 20th century, primarily due to increased atmospheric concentrations of the key greenhouse gases (GHGs) carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O). The proportion of greenhouse gas (GHG) emissions caused by agriculture is estimated at 13.5% worldwide (IPCC 2007). Compared to other sectors, such as energy supply (25.9%) industry (19.4%) forestry (17.4%) and transport (13.1%), the agricultural contribution to climate change is significant. Carbon dioxide can be sequestered by storage in stable organic carbon pools in the soil, while commonly used crop production practices generate CO_2 and N_2O. Many studies have investigated the potential of agricultural management to improve the sink function of agricultural land for CO_2 (e.g. Soussana et al., 2004).

Although peatlands cover only 6% of the earth's surface, they play a central role in the global carbon cycle. Natural peatlands capture carbon as CO_2, with a long-term average uptake rate of 25 g C m^{-2} yr^{-1} (Borren et al., 2006). However, the wet soil conditions in peatlands lead to methanogenesis and significant CH_4 emissions. The balance between CO_2 uptake and CH_4 release is dependent on moisture conditions, temperature, vegetation composition, availability of degradable substrates and microbial activity (e.g. Hendriks et al., 2009). Generally, N_2O does not play a significant role in the GHG budgets of natural peatlands.

While natural peatlands act as sinks of carbon, agricultural peatlands act as sources of carbon and GHGs. In agricultural peatlands, N_2O contributes to the GHG balance; generally, these peatlands also release CO_2. For instance in Europe, 50% of all peatlands are subjected to various sorts of agricultural practices (Joosten and Clarke, 2002), often associated with drainage. Anthropogenic alterations to peatlands have led to dramatic changes in the water balance and to the oxidation of peat and release of CO_2 to the atmosphere. The oxidation of peat in combination with soil settling leads to soil subsidence. Besides, high inputs of nutrients from chemical fertiliser and manure alter the GHG balance and have led to increased N_2O emissions. Recently, efforts have been made to return intensively used, drained peatland areas into near-natural areas, in order to reduce emissions and to stop peat subsidence. Burgerhart (2001) stated that drained peatlands can be transformed into sinks of CO_2 when groundwater levels are raised and management is reduced. So far, little research has been carried out on the effects of decreasing the management intensity, or on the source and sink strengths of CO_2, CH_4 and N_2O.

In the Netherlands about 270 000 ha (7 % of the land surface) is peatland, of which around 11,000 ha comprises the water surfaces of drainage ditches (Nol et al., 2008). Large shallow freshwater lakes are also part of this peatland. Since the industrial period, the peat

soils have been intensively drained and fertilised and have become carbon sources (e.g. Langeveld et al., 1997; Veenendaal et al., 2007; Hendriks et al., 2007). Current peat subsidence rates in the Netherlands are up to 10 mm yr^{-1} and in the last 30 years alone, 20% of the peat soils have disappeared, transformed to mineral soil due to intensive drainage (Kempen et al., 2009). Recent research shows that water bodies also contribute very importantly to the GHG balance in agricultural peatlands and therefore have to be considered when calculating regional GHG budgets (Schrier-Uijl et al., 2010abc). Emissions from wetlands like Horstermeer and from water bodies are not yet included in national emission inventories (Maas et al., 2008). The emissions from grassland are only partly included in emission inventories and uncertainties of quantifications are high (Maas et al., 2008).

In order to test the hypothesis of Burgerhart (2001) as mentioned earlier, a landscape-scale experiment was performed in peatlands under different management regimes. To obtain spatially and temporally explicit GHG budgets of sources and sinks, simultaneous measurements were carried out for four years at different spatial scales on the entire suite of key biogenic GHGs (CO_2, CH_4 and N_2O). For full accounting of the carbon and GHG balance, the import and export of carbon resulting from management practices was considered. The sites were an intensively managed agricultural peatland (average water table -0.55 m), an extensively managed agricultural peatland (dynamic water table with high water tables in winter and low water tables in summer, average -0.45 m) and a former agricultural peatland (average water table -0.2 m) where 14 years ago agricultural management ceased and the water table has been raised.

The main goal of this research was to analyse the effect of reducing the management and raising the water table in intensively managed agricultural peatlands on the GHG and carbon balances.

7.2. Materials and methods

At three lowland peat sites in the mid west of the Netherlands, the exchange of CO_2, CH_4 and N_2O between the soil–plant system and the atmosphere was measured in detail at various scales, using different measurement techniques. The three research sites (Oukoop, Stein and Horstermeer) lie below sea level and have a temperate climate. In this article, a general description of materials and methods used during the project is given; for more details on specific sites, specific measurement methods or data analyses the reader is referred to other publications (Schrier-Uijl et al., 2010abc; Kroon et al., 2010; Kroon et al., 2009; Nol et al., 2008; Hendriks et al., 2007; Kroon et al., 2007; Veenendaal et al., 2007). Tables 1a and b give an overview of peat depth, landscape elements, land use, management practices, soil and water characteristics, and micrometeorological conditions per site.

7.2.1. Measurement sites

The Oukoop site (Ou) is an intensively managed grassland polder on peat in the west of the Netherlands (lat. 52^{o} 02'N, long. 4^{o} 47'E, altitude -1.8 m a.s.l.). The grassland site from which biomass is regularly removed for fodder is on an intensive dairy farm. Manure and fertilisers are applied four or five times a year from February to September. The area has a clayey peat or peaty clay top layer of about 0.25 m thickness on a 12 m thick peat layer. Sixteen percent of the total polder is open water (drainage ditches or small ponds), 5% is bordering (waterlogged) edges and the remaining part consists of relatively dry fields with fluctuating water tables at an average depth of 0.55 m. The dominant grass species are *Lolium perenne* and *Poa trivialis*.

The Stein site (St) is an extensively managed grassland polder on peat, 4 km south-west of the Oukoop site. The area has been managed as a meadow bird reserve since 2001: no manure or artificial fertilisers have been applied since then and that the only management is the removal of aboveground biomass three times a year. Before it gradually became a meadow bird reserve, the polder was intensively used for grass production in the same way as the Oukoop polder. The area has the same soil characteristics as the Oukoop site and its landuse history prior to being taken out of production was similar. The water table has been dynamic since 2006: water tables are high in winter and low in summer and the average depth of the water table is 0.45 m. The proportions of open water, waterlogged edges and drier fields are approximately the same as at the Oukoop site. The dominant plant species are *Lolium perenne* and *Poa trivialis*, but over time *Anthoxanthum odoratum* and *Rumex acetosa* have become more abundant.

The Horstermeer site (Ho) is a grassland/wetland polder on peat in a formerly intensively farmed dairying area in the centre of the Netherlands (lat 52^{o} 02'N, 5^{o} 04'E, altitude -2.2 m below sea level). The site was abandoned in 1998 and has not been farmed since. After abandonment, the ditch water table was raised to approximately 0.10 m below the land surface and the vegetation has developed towards semi-natural grassland. Above Pleistocene cover sands is a 2-meter thick peat layer overlain by organic-rich lake sediments. Five percent of the area is open water (ditches), 10% is permanently waterlogged soil (mostly along the ditches), 25% is relatively wet soils and 60% is relatively dry land with a water table fluctuating between 0 – 0.40 m depth) and an aerated top-layer. No management takes place, except for regulation of the ditch water table. Dominant species are *Holcus lanatus, Phalaris arundinacea, Glyceria fluitans*, reeds and high forbs.

Table 7.1. *Site descriptions, land use and management per peat site*

Loc.	Peat thickness	Landscape elements				Land use	Grazing[2]	Biomass Removal[2]	Cow manure Applied[2]	Fertiliser Applied[2]
		Dry land	Wet land	waterlogged land	Water					
	(m)	%	%	%	%			(ton ha^{-1} yr^{-1})	(kg N ha^{-1} yr^{-1})	(kg N ha^{-1} yr^{-1})
Ou[1]	12	79		5	16	intensively managed grassland	2005 and 2006 by some cows	12	300	88
St[1]	12	79		5	16	extensively managed hayfield	young cattle few days per year	10	0	0
Ho[1]	2.1	60	25	10	5	former managed area under restoration	None	0	0	0

[1]*Ou= Oukoop, St= Stein, Ho= Horstermeer.*
[2]*Values related to management are averaged over the years 2006, 2007 and 2008.*

Table 7.2. *Soil and micrometeorological characteristics perpeat site.*

Loc.	Mean annual		Elevation (a.m.s.l.[3])	Soil type	Org. carbon weight (%) (0-0.2 m)	pH-KCL soil (0-0.2 m)	Mean WT[4] (m below surface)	pH KCL ditch water
	Air temp. (^{o}C)[2]	Rainfall (mm)[2]						
Ou[1]	9.8	797	-1.8	Fibric Eutric Histosol	24	6.2	0.55	7.9
St[1]	9.8	797	-1.7	Fibric Eutric Histosol	15	5.7	0.45	7.5
Ho[1]	9.8	797	-2.2	Eutric Histosol	39	5.0	0.20	7.1

[1] Ou= Oukoop, St= Stein, Ho= Horstermeer; [2] Averaged over 10 years (data from KNMI); [3] a.m.s.l. = above mean sea level; [4] WT = water table

7.2.2. Instrumentation and methodology to quantify terrestrial emissions

Measurement techniques of GHG fluxes comprise mainly the chamber method and the micrometeorological eddy covariance (EC) technique (Christensen et al, 2003). The two techniques cover different scales both in time and space: the chamber method covers up to 1 m^2 spatially and measurements are discontinuous, while eddy covariance covers 100 – 1000 m^2 and measurements are continuous over time (Hendriks et al., 2009). The EC technique is best suited for determining average GHG fluxes at the landscape scale and capturing temporal variability. However, EC techniques for CH_4 and N_2O are currently still in development and were not available at all three sites. The chamber technique is best suited for capturing spatial variability, but can also be used to determine average GHG fluxes at the landscape scale (Hendriks et al., 2009). In our study, the landscape elements that contributed differently to the GHG balance were taken into account proportionally (see *Table 7.1*.). The aim was to measure terrestrial emissions (from fields, ditches and ditch edges). Under stable conditions, at low wind velocity sometimes also farm related emissions were captured by the eddy covariance systems. These farm-based emissions were excluded from the datasets by using the footprint analyses of Kormann and Meixner (2001).

To capture temporal variability of fluxes, all three sites were equipped with eddy covariance (EC) systems for CO_2 since the end of 2004 for four years (for details of the systems see Veenendaal et al., 2007 and Hendriks et al., 2007). In addition, Oukoop was equipped with an EC system for CH_4 and N_2O since the beginning of 2006 for three years (details in Kroon et al., 2007, 2009). In April 2006 the Horstermeer site was equipped with an EC system for CH_4 (Hendriks et al., 2008, 2009). Small-scale chamber measurements of CO_2, CH_4 and N_2O were performed at all three sites to capture spatial variability of fluxes within the research sites. At the Oukoop and Stein sites measurements were performed from January 2006 to December 2008 in the Oukoop and Stein polders (Schrier-Uijl et al 2010a,b) and at the Horstermeer polder from January 2005 to December 2008 (Hendriks et al., 2007). In addition to EC devices and chamber measurements, towers with meteorological instruments were installed at each site which provided 30-minute averages of global radiation (R_{in}), net radiation (R_{net}), air temperature (T_{air}), vapour pressure (P_{vap}), wind speed (U), wind direction (D) and precipitation (P). Soil measurement sensors included soil heat flux plates (HPF01, Campbell Scientific, USA), soil temperature sensors at depths of 0.02, 0.04, 0.08, 0.16 and 0.32 m (Campbell Scientific, USA) and soil moisture probes to measure volumetric moisture contents at depths of 0.10, 0.20 and 0.30 m (Theta probes ML 2x; Delta T devices Burwell, UK). These systems provided 30 minute values for soil heat fluxes, soil temperature, soil moisture and water table.
Comparable measurement procedures, flux calculation methods, gap filling techniques and up-scaling methods were used to calculate annual GHG fluxes. An overview of the measurement periods, used instruments and temporal up-scaling methods are given in *Table 7.3*. and the procedures to calculate annual fluxes are given in *Table 7.4*. These methods are

explained in paragraph 7.2.3. For spatial up-scaling of chamber measurements, the landscape elements that contributed differently to the GHG balance were taken into account proportionally.

Table 7.3. *Measurement periods, techniques and temporal up-scaling methods per GHG per site.*

Gas	Measurement methods						Temporal up-scaling methods	
	Static chamber			Eddy covariance			Static chamber	Eddy covariance
	Ou	St	Ho	Ou	St	Ho	All locations	All locations
CO_2	2006 2007 2008	2006 2007 2008	2005 2006 2007 2008	2005 2006 2007 2008	2005 2006 2007 2008	2005 2006 2007 2008	*T* regression*	night time measurements and multiple regression for data gaps, with monthly E_0 and R_{10} values.
CH_4	2006 2007 2008	2006 2007 2008	2005 2006 2007 2008	2006 2007 2008	2006 2007 2008	April 2007	*T* regression*	measured values and multiple regression with *T* (soil temperature and *U* (wind velocity) for data gaps
N_2O	2006 2007 2008	2006 2007 2008	2005 2006 2007 2008	2006 2007 2008	2006 2007 2008	NA**	NA**	The used method separates background emission and event emission due to manure application

** Regression based on temperature.*

*** not available, the detection limit of the gas analyzer was too low for the used chamber design.*

Table 7.4. *Emission calculation methods per site per greenhouse gas.*

Site	GHG	Method	Calculations of emissions	Ref.	Abbreviations
Ou, St, Ho	CO_2	Eddy covariance	Annual NEE_{CO2} =GPP-R_e Annual NEE is calculated from 30 minute night fluxes $R_e = R_{10} \exp^{E_0((1/283.15-T_0)-1/(T-T_0))}$ R_{10} and E_0 are estimated per month and 30 minute day fluxes $F_c = \frac{\alpha.\mathrm{PPFD}.\beta}{\alpha.\mathrm{PPFD}+\beta} + \chi$	1, 2, 3	NEE= net ecosystem exchange GPP= gross primary production R_e = ecosystem respiration R_{10}= respiration at 10 ℃ T_0= fixed T at 227.13 K E_0= activation energy F_c= ecosystem flux PPFD= Photosynthetic photon flux density α, β and χ are parameters
Ou, St, Ho	CO_2	Dark chamber	Annual R_e is calculated from a regression based on three years of chamber measurement given by $R_e = R_{10} \exp^{E_0((1/283.15-T_0)-1/(T-T_0))}$	4	
Ou, Ho	CH_4	Eddy covariance	$\mathrm{NEE}_{CH4} = \sum_{i=1}^{N} F_{CH4_i} T_{av}$ with F_{CH4} 30 minute measured eddy covariance flux or the gap filled flux given by $F_{CH_4} = \exp^{a+bT+cU}$ *a*, *b* and *c* are factors in the regression	5	NEE_{CH4}= annual emissions of CH_4 F_{CH4}= 30 minute flux of CH_4 T_{av} = averaging time T= 30 minute soil temperature U –=30 minute wind velocity
Ou, St, Ho	CH_4	Dark chamber	$\mathrm{NEE}_{CH4} = \sum_{i=1}^{N} F_{CH4_i} T_{av}$ with $F_{CH_4} = \exp^{a+bT}$ *a* and *b* are factors in the regression and are different per site and per landform	6	
Ou	N_2O	Eddy covariance	$\mathrm{NEE}_{N_2O} = E_{EC} + E_l + E_d$ with $E_{EC} = E_{bgnd} + E_{fert}$	5	NEE_{N2O}= annual emissions of N_2O E_{EC} = emission measured by eddy covariance E_l = indirect emission due to leaching and run-off E_d = indirect emission due to deposition E_{bgnd} = background emission E_{fert} = direct emission due to fertilizing events
St	N_2O	Literature		7	
Ho	N_2O	Literature			

1 Lloyd and Taylor (1994), 2 Veenendaal et al. (2007), 3 Falge et al. (2001), 4 Schrier-Uijl et al. (2010b), 5 Schrier-Uijl et al. (2010a), 6 Kroon et al. (2010), 7 Velthof et al. (2007).

7.2.2.1. Small-scale flux measurements

To study the spatial variability of GHG fluxes, small-scale flux measurements were performed on fields and open water for CO_2 and CH_4 using a Photo Acoustic Field Gas Monitor (type 1412, Innova AirTech Instruments, Ballerup, Denmark) connected with Teflon tubes to closed opaque chambers (height 0.2 m, diameter 0.3 m). A fan was installed in the chamber to homogenise the inside air and two external filters were added: a soda lime filter for CO_2 (when measuring CH_4) and a silica gel filter for water vapour. To avoid disturbances, vegetation was not removed prior to the flux measurements (Hendriks et al., 2007; Schrier-Uijl et al., 2010a,b). The rate of increase over time of the gas concentration in the headspace of the chamber provides a direct estimate of the gas exchange flux between the soil and the atmosphere. Concentrations were measured every minute for 5-15 minutes at one minute intervals, and the slope d*C*/d*t* of the gas concentration curve was estimated by linear regression. Because the relations with explanatory variables were non-linear, multiple non-linear regression was used to calculate annual emissions, and the landscape elements (ditches, waterlogged ditch edges and fields) that contributed significantly differently to the GHG balance were taking into account by using a weighting factor. As measurements were available for each separate contributing landscape element, emission scenarios with a changing landscape in terms of decreasing or increasing the amount of open water could be modelled. In the summer of 2009 an additional measurement campaign was performed on 6 large shallow lakes and 14 drainage ditches in peatlands. Measurements of CO_2 and CH_4 were performed using floating chambers. Cross-checks of emission values were performed by comparing eddy covariance measurements with upscaled chamber measurements within the footprint of the eddy covariance systems (Schrier-Uijl 2010bc).

7.2.2.2. Landscape scale flux measurements

Landscape scale flux measurements of CO_2, CH_4 and N_2O were performed using the EC flux technique. Footprints of all EC flux towers were over the entire landscape, including fields, ditches and ditches edges but excluding farms. EC flux systems for CO_2 at the three sites consisted of a sonic anemometer and a fast response CO_2-H_2O analyzer. Open path infrared gas analyzers (LI-COR Lincoln, NE, USA) were used 4.3 meter above the soil surface in Horstermeer and 3.05 meter above the soil surface in Oukoop and Stein. Oukoop was also equipped with an EC flux system for CH_4 and N_2O in the period 2006-2008 consisting of a sonic anemometer and a Quantum cascade laser spectrometer (model QCL-TILDAS-76, Aerodyne Research Inc., Billerica MA, USA). In April 2007 the Horstermeer site was also equipped with an EC flux system for CH_4, consisting of a DLT-100 Fast Methane Analyzer (FMA) from Los Gatos Research Ltd. In all three sites additional micrometeorological measurements were performed. Annual fluxes were calculated by

using measured 30 minute values and data gaps were filled with a multi variate regression (Kroon et al., 2010).

7.2.2.3. Additional measurements

At the beginning and the end of the experiments (2005 and 2008), C and N content, organic matter, NO_3^-, NH_4^+, PO_4^{3-} and pH. Water from drainage ditches was sampled for measurements of pH, dissolved organic C (not for Horstermeer), dissolved organic N content (not for Horstermeer), organic matter, NO_3^-, NH_4^+, PO_4^{3-}, SO_4^{2-}, Fe^{2+}, dissolved CH_4, oxygen saturation and electrical conductivity. Well-stirred samples of slurry manure were sampled just before manure application in the Oukoop site and were analyzed for dry matter and C content. To estimate the aboveground biomass and biomass removal, vegetation height was measured every three to four weeks to estimate the above ground biomass and biomass removal using the relationship between vegetation height and biomass weight discussed in Veenendaal et al. (2007). Biomass samples were taken at 4–-8 week intervals for each of 5 – 8 parcels in the main footprint of the mast. Bulk samples consisted of 20 subsamples of 5–-10 leaves sampled randomly from the grass canopy.

7.2.3. Estimates of GHG balances and carbon balances

The total GHG balance of the managed polders as calculated in this study consists of 1) terrestrial sinks and sources (including fluxes from fields, waterlogged land and drainage ditches, together further refered to as field and 2) sinks and sources related to the way farm animals exploit the net primary production (NPP) when they are not present in the meadow (a typical situation in modern dairy farming). These include emissions from livestock and emissions from manure storage. The latter are refered to as farm emissions.

The terrestrial GHG balance of each experimental site was calculated for three years by summation of the net ecosystem exchange of CO_2, CH_4 and N_2O using the global warming potential (GWP) of each gas at the 100 years time horizon (IPCC, 2007).

$$NEE_{GHG} = NEE_{CO2} + 25NEE_{CH4} + 298NEE_{N2O}$$

where 25 and 298 are the global warming potentials of CH_4 and N_2O for a 100-year time horizon. The GWP is a measure of how much a given mass of GHG is estimated to contribute to global warming as compared to CO_2 (thus for CO_2, the GWP is by definition 1).

The total emissions from the unmanaged polder consisted only of terrestrial emissions. However at the managed sites, besides the terrestrial GHG sources and sinks from fields and ditches, also the emissions due to farm-based activities have to be added to the total GHG balance. For example, CO_2, CH_4 and N_2O are released from a farm through enteric fermentation by cattle and manure deposits (Soussana et al., 2008). Biomass that is produced and removed in Stein is consumed within the Oukoop farm, therefore emission calculations from the farm are attributet in a ratio 3:2 to Oukoop and Stein, respectively (see further Veenendaal et al 2007). An overview of the GHG fluxes considered in the calculation of balances in this study is given in in *Fig 7.1*.

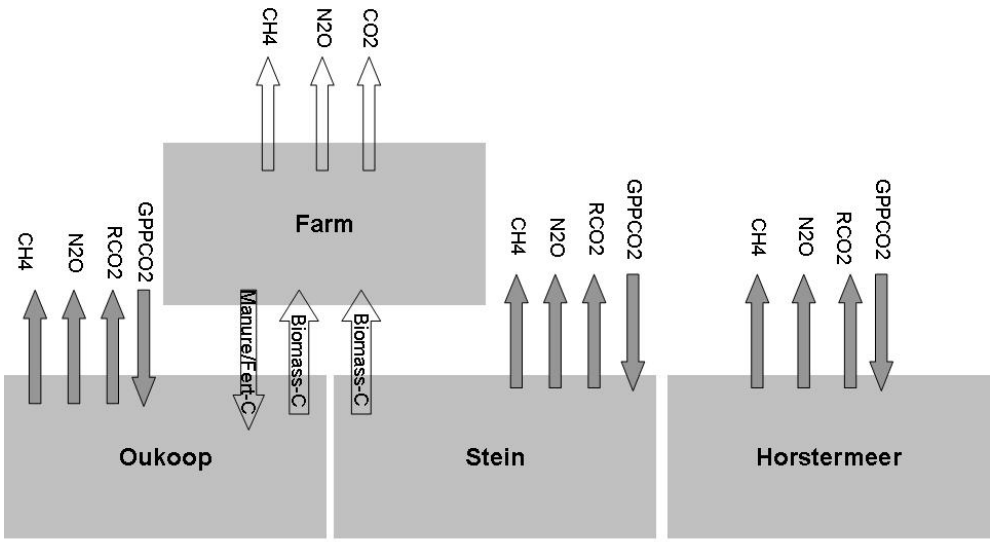


Fig 7.1. *Terrestrial and farm GHG fluxes (CO_2 respiration (RCO_2), CO_2 gross ecosystem production or photosynthesis ($GEPCO_2$), CH_4 and N_2O) and carbon fluxes (CO_2-C, CH_4-C, manure and fertiliser-C, biomass-C) that were considered in the current study for Oukoop, Stein and Horstermeer. White arrows are farm-based fluxes and dark grey arrows are terrestrial fluxes External inputs from imported feeds end outputs to milk and meat are excluded from this balance as are dissolved organic carbon losses (DOC).*

Appendix A gives an overview of all carbon flows in the intensively managed peat area Oukoop. The outside boundaries of the total polder system are given and also the boundaries of the carbon balance considered in this study (from the field point-of-view) are given. For the carbon balance of the three sites, field CO_2-C, field CH_4-C, manure-C input,

fertiliser-C input and export of biomass-C have been included. External inputs from imported feeds and outputs through milk and meat (arrows 12 and 13 in Appendix A) are excluded from this balance and dissolved organic carbon losses are considered to be negligable (DOC; arrow 14 of Appendix A). The external C source from imported animal feeds can vary between 10 to 20% of total feed production (Iepema et al., 2005).

For the total GHG balance terrestrial and farm-based sources and sinks were included. Farm-based CH_4 fluxes consist of emissions due to enteric fermentation by animals and manure management. Farm-based CH_4 emissions from the cattle and the stable were estimated following the simple emission factor approach described by Hensen et al. (2006)

$$F_{CH4farm} = N_{dairy} + N_{heifer}E_y + N_{Calves}E_c + A_{manure}E_m + A_{FYM}E_f$$

with N the number of animals, A the amount of manure or farmyard manure (m^3) and with emission factors for dairy cows (E_d), heifers (E_y), calves (E_c), manure in storages (E_m) and farmyard manure (E_f). The emission parameters were 274 g CH_4 day^{-1} $animal^{-1}$ for cows 170 g CH_4 day^{-1} $animal^{-1}$ for heifers, 48 g CH_4 day^{-1} $animal^{-1}$ for calves, 53 g CH_4 day^{-1} m^{-3} for fertiliser and 40 g CH_4 day^{-1} m^{-3} for farmyard manure – all ±50% (Sneath et al., 2006; van Amstel et al., 2003).

Farm-based CO_2 emissions were estimated from the amount of biomass-C imported into the farm subtracted by the amount of manure-C added on the fields and the amount of C emitted as CH_4. A production efficienty of 7% for large mammals is accounted for (van Raamsdonk et al., 2007; Nieveen et al., 2005; Guinand-Flament et al., 2007).

The farm-based N_2O source strength was estimated by using the farm measurements of Hensen et al. (2005). The terrestrial N_2O fluxes which are shown in *Fig 7.1.* were measured in the Oukoop site, whereas for the Stein and Horstermeer sites these components have been estimated from literature, because at these two sites nitrous oxide emissions were too small to detect with the used gas analyzer. For the considered fluxes in the Horstermeer site, the reader is referred to Hendriks et al., 2007. Indirect emissions due to leaching and run-off were measured in the Horstermeer site, but were not directly measured in the Oukoop and Stein site. In Kroon et al (2010) an estimate of leaching and runoff based on the annual amount of synthetic fertilizer and the annual amount of applied cow manure (IPCC 2006) is given for the Oukoop site.

Emissions from large water bodies were measured in the summer season, and thus no annual values have been presented for these ecosystems in this paper. Summer emissions from water bodies in peatlands and emissions from peatland were therefore compared using June and July data only. These summer fluxes represent around 70% of the annual CH_4 emission from drainage ditches (Schrier-Uijl et al., 2010a).

7.2.4. Up-scaling of fluxes to regional scale

To be able to scale up from the measurement sites to the western peatland area of the Netherlands, a detailed inventory of the distribution of landscape elements is needed. This peatland area covers about 115,000 ha. A detailed database was compiled using the topographic vector-based Top10Vector database (TDN, 2006), a field inventory (Nol et al., 2008), and databases of Dutch natural peatlands (Natuurmonumenten, Staatsbosbeheer, Provinciale landschappen). The resulting database distinguishes between intensively and extensively managed peatland, ditches and ditch edges. Using this database, the areas of these landscape elements were estimated and used together with the emissions from the landscape elements to estimate the total GHG emission from the peatland area under the assumption that fluxes measured in the intensively managed area and the extensively managed area in this study were representative for the western peatland area. *Table 7.5* shows the areas of the landscape elements.

Table 7.5 *Areas (ha) of landscape elements/land use in the Dutch peatlands.*

Landscape element/land use	Area (ha)	Area (%)
Grassland: intensively managed	78,375	68%
Grassland: extensively managed	8,786	8%
Water	6,717	6%
Urban area (incl. greenhouses)	983	1%
Roads	4,490	4%
Forest	2,716	2%
Cropland	1,818	2%
Other land use	11,258	10%
Total	115,142	100%

Kadaster, 2010. Product information TOP10vector (in Dutch). Topografische Dienst Kadaster, Emmen, the Netherlands. Available at http://www.tdn.nl/ (verified 3 May 2010).

7.2.5. Future scenario for an increase in water storage

Simple scenarios for the peatlands were constructed. They were primarily based on the proposed mitigation option of enlarging the area of open water to increase the water storage capacity. Open water emissions were measured separately from the field emissions and waterlogged land emissions in the three sites and therefore by changing the weighting factor for each landscape element in the regressions, emissions could be estimated for different percentages of open water. Because Stein and Oukoop act similarly in relation to emissions from open water and field, only Stein and Horstermeer were considered in the different scenarios. Future climate change in terms of temperature rise and changing rainfall patterns were not accounted for, nor were hydrological changes (e.g. water table changes) emanating from changes in field size in the scenarios

7.3. Results

7.3.1. Carbon dioxide balance

Temporal variability of the annual net ecosystem exchange (NEE) was high (diurnal, seasonal, annual), but NEE of CO_2 in the three areas showed a clear difference between the managed (net release) and the unmanaged (net uptake) peatlands (*Table 7.6.*, *Fig 7.2.*). Over a four-year period the unmanaged peatland (Horstermeer) had an average CO_2 NEE uptake of 1.4 kg CO_2 m^{-2} yr^{-1}, while the two managed peatlands (Stein and Oukoop) had an average release of 0.4 kg CO_2 m^{-2} yr^{-1}. Inter-annual variability was high: for example, the intensively managed area appeared to be a marginal CO_2 sink in 2007, which was a relatively wet and cold year, but a CO_2 source in the other three years. Carbon dioxide estimates for R_{eco} (derived from night-time measurements (Veenendaal et al., 2007)) and gross ecosystem productivity (GEP) (calculated as the difference between R_{eco} and NEE) are also shown. The NEE of shallow lakes was measured in the summer of 2009 (in the months June and July in which high emissions occur) and therefore no annual values are shown in *Table 7.6.*

Table 7.6. *Terrestrial carbon dioxide flux estimates (kg CO_2 m^{-2}) measured by eddy covariance in the period 2005-2008 for the intensively managed production grassland on peat (Ou), the extensively managed hayfield on peat (St) and the unmanaged former agricultural peatland (Ho). Fluxes from shallow lakes were measured in the summer of 2009. Fluxes from removed biomass were not included.*

Site	Carbon dioxide flux per year (kg CO_2 m^{-2} yr^{-1})												Av. flux
	2005			2006			2007			2008			
	R_{eco}[1]	GEP[1]	NEE[1]	R_{eco}[1]	GEP[1]	NEE[1]	R_{eco}[1]	GEP[1]	NEE[1]	R_{eco}[1]	GEP[1]	NEE[1]	NEE[1]
Ou	5.7	-5.3	0.4	5.5	-4.7	0.7	5.7	-5.9	-0.2	4.6	-4.0	0.7	0.4
St	5.6	-5.7	-0.09	5.8	-5.0	0.8	5.4	-5.1	0.3	5.4	-4.6	0.8	0.4
Ho			-1.0			-1.0			-1.9			-1.5	-1.4
Lake													0.5[2]

[1] R_{eco} *= Respiration, GEP= Gross Ecosystem Production, NEE= Net Ecosystem Exchange*

[2] *Fluxes were measured in the summer (June/July) of 2009 in 5 large shallow lakes located in peatlands.*

Fig 7.2. shows the NEE for CO_2 measured daily for all three sites for the period 2005-2008 and the cumulative NEE for each year. The unmanaged site appeared to be a CO_2 sink in all years (range -1034 to -1939 g CO_2 m^{-2} yr^{-1}), with periods of a small net release in late winter/early spring periods and with net uptake in the rest of the year. Daytime variability

showed uptake during daytime and release during night time. The managed sites appeared to be sources of CO_2 in all years, except for Oukoop, which was a marginal sink for CO_2 in 2007 and Stein, which was a marginal sink of CO_2 in 2005 (range -173 to +747 g CO_2 m^{-2} yr^{-1} for Oukoop and -88 to +790 g CO_2 m^{-2} yr^{-1} for Stein).

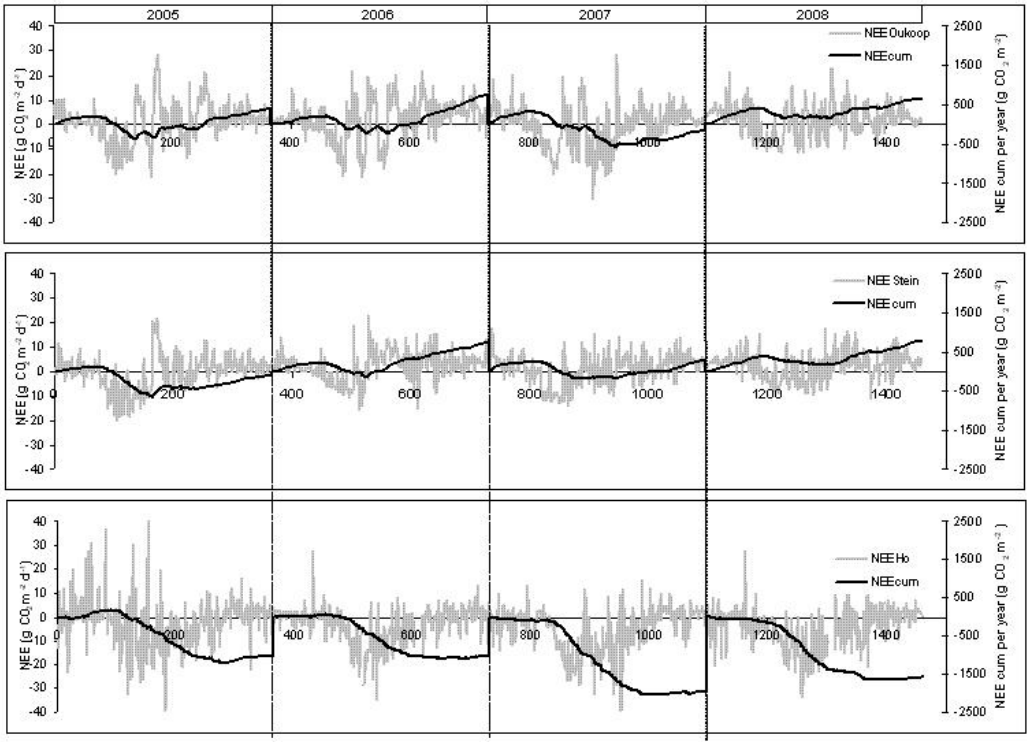


Fig 7.2. *The NEE CO_2 for the Oukoop (intensively managed), Stein (extensively managed) and Horstermeer (unmanaged) peatlands measured by the eddy covariance flux technique. The black line represents the cumulative NEE for each year separately (y-axis on the right) and the grey line represents the temporal variability of NEE on the time scale of a day (y-axis on the left). The x-axis represents the day number since 1 January 2005.*

Monthly CO_2 NEE values show (*Fig 7.3.*) that the difference between the managed areas with low water tables and the unmanaged area with high water table is largest in the growing season. In this season, the Horstermeer, the former agricultural peatland with unmanaged vegetation is a large sink, while Oukoop and Stein are only minor sinks.

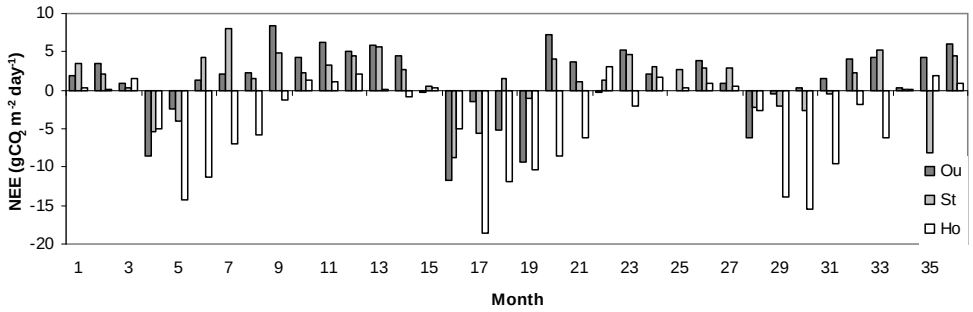


Fig 7.3. *Monthly NEE CO_2 values for the three experimental sites Oukoop (Ou), Stein (St) and Horstermeer (Ho). The CO_2 NEE is given on the y-axes in g CO_2 m^{-2} d^{-1} and the month numbers are given from 2006-2008 on the x-axes.*

Total farm-based CO_2 emissions (arrows 1 + 3 in Appendix A) were calculated from the biomass-C produced on the farm and fed to the cattle in Oukoop (value 400 g C m^{-2} and 420 g C m^{-2} for Oukoop and Stein, respectively; arrow 10 in Appendix A) a production efficiency of 5% for large mammals (production of milk and meat from assimilated feed) was assumed (personal comment Veenendaal), 142 g C m^{-2} yr^{-1} exported as manure to the fields in Oukoop (measured; arrow 15 in Appendix A) and a farm based CH_4 emission of 57 g C m^{-2} yr^{-1} (arrow 2 in Appendix A). The farm based CO_2 release of NPP is estimated at 0.7 kg CO_2 m^{-2} yr^{-1} (arrow 1 Appendix A). This brings the total (terrestrial + farm; arrows 1+3+5-6+8 in Appendix A) CO_2 emission to 1.1, 1.1 and -1.4 kg CO_2 m^{-2} yr^{-1} for Oukoop, Stein and Horstermeer, respectively.

7.3.2. Methane balance

In all three sites, temperatures of soil and water were the most significant predictors of CH_4 emissions and they were therefore used in the non-linear regression. At the Horstermeer site clear diurnal cycles were found in CH_4 fluxes during all seasons synchronous to incoming radiation. This diurnal variation is in agreement with other wetland sites with similar vegetation and is probably caused by stomatal opening and/or pressurized convective gas transport through the vascular plants with highest emissions in the late afternoon and lowest emissions during the night (Hendriks et al. 2009). Based on Hendriks et al. (2009) annual emissions were calculated by using a correction for diurnal variation and by upscaling of emissions from four instead of three contributing landscape elements: relatively dry land (60%), relatively wet land (25%), water saturated land for most of the year (10%) and ditches (5%) (*Table 7.1.*). For the Oukoop and Stein site, after correction for temperature no difference between average day and night flux was found for CO_2, CH_4 and N_2O (Schrier-Uijl et al., 2010b; Kroon et al., 2010); no correction was applied (Schrier-Uijl et al., 2010b).

Annual emissions of the three sites for 2006, 2007 and 2008 and for 2005 for the Horstermeer site are given in *Table 7.7.*

Table 7.7. *Yearly and average methane fluxes (g CH_4 m^{-2} yr^{-1}) measured by the chamber method and for Oukoop also measured by the eddy covariance method. For spatial upscaling of chamber measurements landscape element weighted predictive relationships are used. Farm-based emissions are not included in this table.*

Ecosystem	Annual methane fluxes (g CH_4 m^{-2} yr^{-1})				Average emission (g CH_4 m^{-2} yr^{-1})
Spatially weighted for contribution of fields, ditches and edges	2005	2006	2007	2008	Average
Ou (chamber method)	NA[1]	20.3	16.2	14.6	17.0
Ou (eddy covariance method)	NA[1]	17.2	16.6	15.5	16.4
St (chamber method)	NA[1]	15.7	18.0	16.3	16.7
Ho (chamber method)	19.1	20.5	19.8	17.6	19.2

[1]*NA means not available.*

Fig 7.4. shows the temporal variability (daily values) and the cumulative NEE for terrestrial CH_4 (including fields, waterlogged land and open water) over three years for all three sites. For the Oukoop site three years of eddy covariance measurements are also shown. The modelled emissions based on chamber measurements are smoother than the eddy covariance measurements because only temperature was used as predictive variable. Cumulative CH_4 values are on average over three years similar for Oukoop and Stein but are higher for the Horstermeer site. The CH_4 emissions varied widely with the season, peaking during summer.

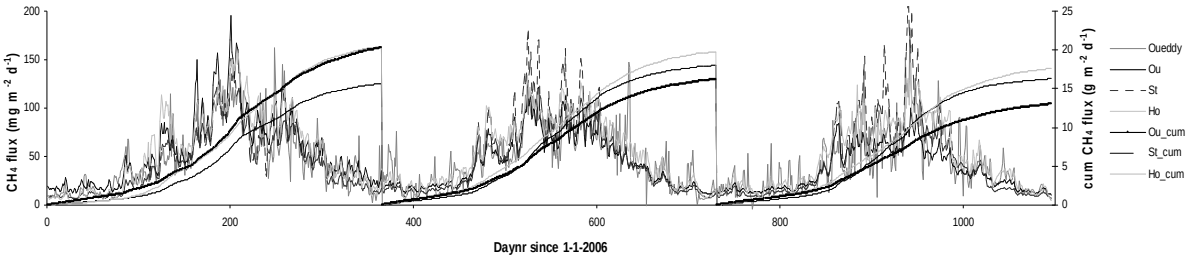


Fig 7.4. *Temporal variability of terrestrial methane fluxes for the three experimental sites(Ou, St, Ho) modelled by predictive relationships based on chamber measurements and additionally for Oukoop measured by continuous eddy covariance (oueddy). The right-hand y-axis represents the cumulative CH_4 flux over the three years and the left-hand y-axis represents the CH4 emissions in mg m^{-2} hr^{-1}. Fluxes are weighted by the contribution of each landscape element.*

Spatial variability had been found to be high between landscape elements within an ecosystem (Schrier-Uijl et al., 2010; Hendriks et al., 2007, 2009) and between ecosystems (this study).

Summer CH_4 emissions from the three peatlands were compared to emissions from 5 large shallow freshwater lakes located in peatlands (*Fig 7.5.*).

Summer emissions from the lakes were significantly higher than the emissions from the managed ecosystems (fields, ditches and ditch edges). The unmanaged peatland had significantly lower summer emissions than the extensively managed systems and the large shallow lakes. In an earlier study (Schrier-Uijl et al. 2010c) the emission from lakes appeared to be smaller than the emission from drainage ditches within the managed and unmanaged ecosystems.

Ecosystem CH_4 emission includes both terrestrial emissions and emissions from the farm itself (arrows 2, 4 and 7 in Appendix A). Schrier-Uijl et al. (2010a) reported additional farm-based emissions of 17 and 26 g CH_4 m^{-2} yr^{-1} for Stein and Oukoop, respectively. The estimates were based on the size of the farm, manure storage and the number of livestock (Hensen et al., 2006; van Amstel et al., 2003; Sneath et al., 2006). The grass in Stein is removed to feed cows in the Oukoop area, and thus the estimated total farm emissions had to be distributed by approximately 0.4 over the extensively managed area and 0.6 over the intensively managed area. The sum of terrestrial CH_4 emissions and farm-based CH_4 emissions was 43.0 g CH_4 m^{-2} yr^{-1} for Oukoop, 33.7 g CH_4 m^{-2} yr^{-1} for Stein and 19.2 g CH_4 m^{-2} yr^{-1} for Horstermeer.

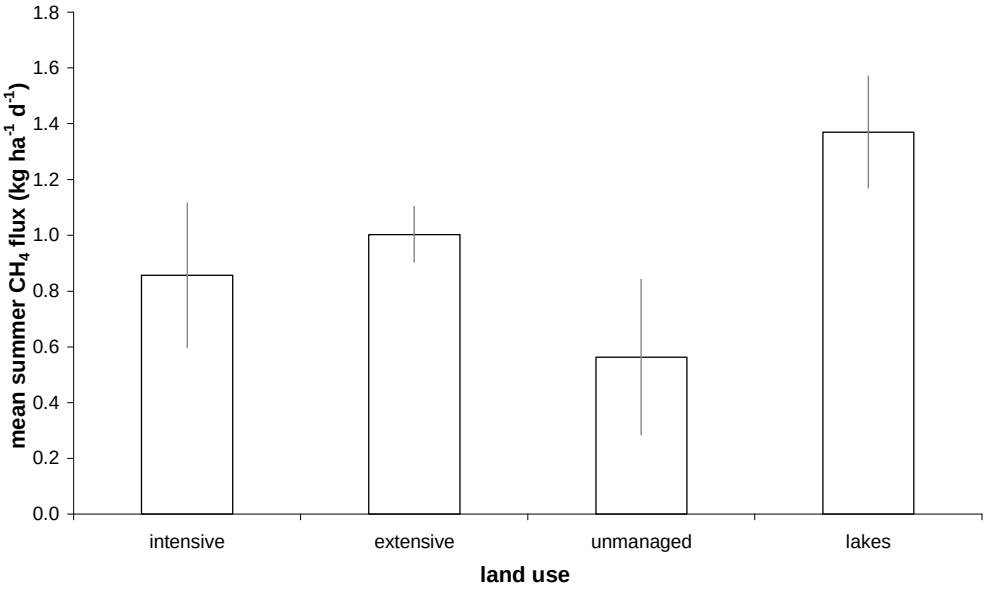


Fig 7.5. *Comparison of summer (June/July) CH_4 fluxes between different ecosystems in peatlands. Fluxes are averaged over three years for the Oukoop (intensive), the Stein (extensive) and the Horstermeer (unmanaged) polders. The lake fluxes are measured in one summer (June/July, n=97) season and are averaged over 5 large shallow lakes; error bars represent the standard deviation of the mean.*

7.3.3. Carbon balances

To obtain a more comprehensive overview of the pathways of carbon at the different sites, the carbon pools of the ecosystems were assessed as shown in *Fig 7.1* and Appendix A. The carbon dioxide source estimates of 0.4 kg CO_2 m^{-2} averaged over four years for Oukoop and Stein result in an average carbon source strength of 1091 kg C ha^{-1} yr^{-1}. For the Horstermeer site the sink strength of 1.4 kg CO_2 m^{-2} yr^{-1} results in an average carbon sink strength of 3818 kg C ha^{-1} yr^{-1}. The earlier reported sink strength of -57 kg C ha^{-1} yr^{-1} for Stein in Veenendaal et al. 2007 was found in 2004-2005. This was the only year that Stein appeared to be a sink (*Fig 7.6.*). The average CH_4 estimates of 17.0 for Oukoop, 16.7 g C m^{-2} for Stein and 19.2 g C m^{-2} for Horstermeer result in an corresponding average carbon source strength of 127.5, 125.3 and 144.0 kg C ha^{-1} yr^{-1}, respectively. At the Horstermeer site also vertical and lateral losses of dissolved gases through water and the different GWP's were taken into account. Fluxes of carbon and CH_4 through water were relatively small (<5% of the full GHG balance), due the flat topography, compact soils and low discharge (Hendriks et al., 2007).

For a complete terrestrial C balance (seen from the field point-of-view) manure inputs and biomass removal have to be included (see for boundaries of the considered system in this study Appendix A). 0n the basis of a C content of 26.4 ± 1.8 g C kg^{-1} slurry manure (mean ±1 SD) and a specific weight of the slurry manure of 1.08 ± 0.01 kg l^{-1} (determined from the first slurry application) the average annual remittal of C through manure into the field

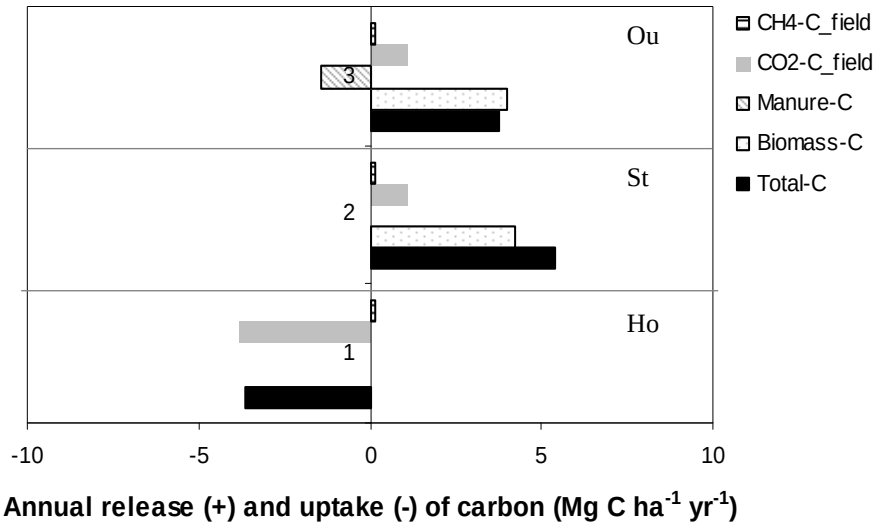


Fig 7.6. *Summary of carbon fluxes considered in the research areas Horstermeer (Ho), Stein (St) and Oukoop (Ou) averaged over 2005, 2006, 2007 and 2008. The annual carbon balance is presented in Mg C* ha^{-1} yr^{-1}*, (+) is release and (-) is uptake, and consists of fluxes due to GHG emissions (field-NEE* CO_2 *and field-NEE* CH_4*) and fluxes due to management (manure application and biomass removal).*

was estimated at 142 g C m^{-2} yr^{-1} in Oukoop (*Table 7.8.*). The final calculation of the average C source strength was based on the years 2005-2007; in 2008 the farmer disposed of his stored manure on his fields because he moved to another farm. In 2006-2008 very little manure was directly deposited by cattle in the field because there were only a few days of grazing. So, whereas an additional amount of 14 g C m^{-2} yr-1 from grazing cattle was included in 2005 by cattle grazing in 2005 (Veenendaal et al., 2007) in 2006-2008 no carbon flux through direct manure deposition was taken into account. In Stein and Horstermeer no fertilisers were applied.

Table 7.8*. Cow manure and fertiliser application and in Oukoop*

Oukoop	manure	manure	manure	fertilizer
	m^3 ha^{-1}	kg N ha^{-1}	ton C ha^{-1}	kg N ha^{-1}
2005	56	NA[1]	1.42	NA[1]
2006	55	253	1.43	100
2007	57	262	1.48	95
2008	90	414	2.34	70

[1]*NA means not available*

Removal of biomass in Oukoop was estimated at 9.0 t dry matter ha^{-1} for 2005, 6.2 t dry matter ha^{-1} for 2006 and 9.0 t dry matter ha^{-1} for 2008, which is an average loss of 400 g C m^{-2} yr^{-1}. In Stein, carbon loss by biomass removal was estimated at 420 g C m^{-2} yr^{-1} (Veenendaal et al., 2007). *Fig 7.6.* shows the carbon balance taking into account the field GHG emissions (arrows 5, 6, 7 and 8 Appendix A), manure application and biomass export (arrows 15 and 10 Appendix A). The total terrestrial C-release in Oukoop and Stein is 3.8 and 5.4 Mg C ha^{-1} yr^{-1}, respectively; the total C-uptake in Horstermeer is 4.4 Mg C ha^{-1} yr^{-1}.

7.3.4. Nitrous oxide balances

Measured cumulative NEE N_2O over a three-year period in the intensively managed site (Oukoop) were separated in fertiliser related emissions or event emissions and background emissions. Nitrous oxide emission values are evaluated in Kroon et al. (2010) and are given in *Table 7.9.* N_2O emissions in the extensively managed site (Stein) and unmanaged peat marsh (Horstermeer) are taken from the literature (Velthof et al., 1997).

Table 7.9. *Nitrous oxide flux estimates (kg N_2O ha^{-1} yr^{-1}) and their uncertainties (u) for the intensively managed site (Oukoop), extensively managed site (Stein) and the unmanaged peatland (Horstermeer).*

Site	Source	N_2O emissions (kg N_2O ha^{-1} yr^{-1})	Reference	Total emission (kg N_2O ha^{-1} yr^{-1})
Oukoop	Background emission [1]	17 (±17%)	Kroon et al., 2010	24 (±28%)
	Emissions due to fertilisers[2]	5 (±50%)	Kroon et al., 2010	
	Indirect emission due to leaching and run-off	1 (±50%)	Kroon et al., 2010; IPCC, 2006	
	Indirect emission due to deposition	1 (±50%)	Kroon et al., 2010; IPCC, 2006	
Stein	Total emission	8 (±100%)	Velthof et al., 1997; IPCC, 2006	8 (± 100%)
Horstermeer	Total emission	8 (±100%)	Velthof et al., 1997; IPCC, 2006	8 (± 100%)

[1] *Background emissions were determined by a multivariate regression model based on EC flux data excluding EC fluxes measured around a management event.*

[2] *Emissions due to fertilizer application have been determined by subtraction the background emission from the total measured N_2O emission around fertiliser application. The IPCC default value of 1% is used for the missing fertilising events.*

In Oukoop farm-based N_2O emissions from manure storages must also be added to the total N_2O balance. Using the estimates of Hensen et al. (2006) and a GWP of 298 for N_2O we estimated N_2O emissions from the manure storage in Oukoop at 1.5 10^{-2} g N_2O m^{-2} yr^{-1}, which did not contribute significantly to the total N_2O balance. From *Table 7.9*. it can be seen that the total measured N_2O emission for Oukoop was 21 kg N2O ha^{-1} yr^{-1}, whereas for Stein and Horstermeer the estimated N_2O emissions were the same: 8 kg N_2O ha^{-1} yr^{-1}.

7.3.5. Total GHG balance

All incoming and outgoing GHG fluxes shown in *Fig 7.1*. were available for the three sites for the period 2006 – 2008. Leaching to groundwater and runoff were not measured in Oukoop and Stein but release of N_2O through leaching was estimated for Oukoop in Kroon et al. (2010). Significant differences in GHG emissions had been found earlier between the sites and within the sites between landscape elements: emissions from drainage ditches and waterlogged soil edges along ditches were significantly higher than the fluxes from the relatively dry land (Schrier et al., 2010a; Hendriks et al., 2007). When calculating fluxes on landscape scale, both the proportion of each landscape element in the landscape and the farm-based emissions were taken into account. The CH_4 component in the GHG balance in the ecosystems studied consists of outgoing fluxes only and N_2O emission from the intensively managed site consists of emissions originating from fertiliser events and from background emission. *Fig 7.7.* shows the total GHG balance of the three sites in terms of warming potential.

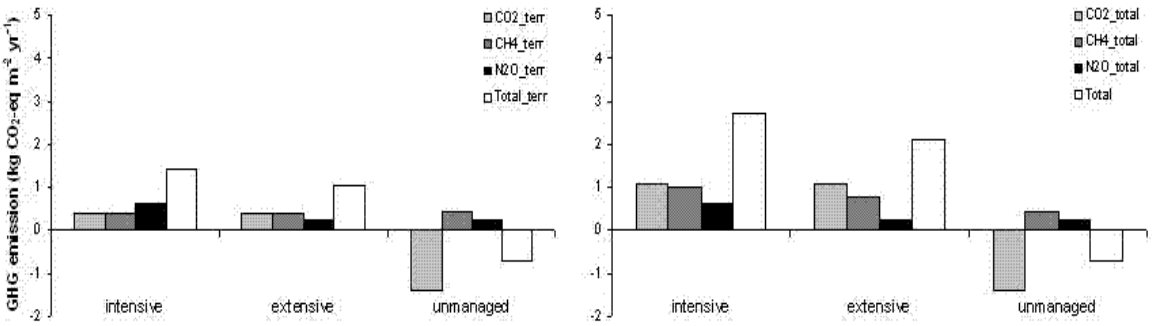


Fig 7.7. *The GHG balances including CO_2, CH_4 and N_2O for the three sites: intensive (Oukoop), extensive (Stein) and unmanaged (Horstermeer). On the left, excluding farm-based CH_4 and CO_2 emissions and on the right, including farm-based CH_4 and CO_2 emissions, averaged over 2006, 2007 and 2008 (fluxes are given in warming potentials, kg CO_2-equivalents m^{-2} yr^{-1}).*

The managed peatlands acted as terrestrial GHG sources of 1.4 and 1.0 kg CO_2-eq m^{-2} yr^{-1}, respectively for Oukoop and Stein and the unmanaged site acted as a GHG sink of 0.8 kg CO_2-eq m^{-2} yr^{-1}. Nitrous oxide emissions were dominant in the intensively managed peatland when no farm based emissions were accounted for. Carbon dioxide and CH_4 dominated the terrestrial GHG balance in the extensively managed peatland. In the unmanaged peatland CO_2 was the most contributing GHG. Accounting for the farm-based CH_4 and CO_2 emissions decreased the relative importance of N_2O in the total GHG balance of the intensively managed peatland. The difference in total source strength between the intensively managed peatland and the extensively managed peatland was mainly attributable to the higher N_2O emission and the higher farm-based CH_4 emissions from the intensively managed site.

7.3.6. Upscaling

In the Netherlands, extensive peatlands developed in the Holocene. The western and northeastern peatlands are threatened by human disturbance and oxidation due to drainage. Currently, 270,000 ha of the Netherlands, mainly in the western part of the country, consist of peatland, but the area is decreasing because of degradation (Kempen et al., 2009; Stouthamer et al., 2008). In the western peat area, 68% is intensively managed grassland, 8% is extensively managed grassland or unmanaged grassland, 6% is water (*Table 7.5.* and *Table 7.10.*) and the remaining part is road, farm or has other land use. Using the emission values found in this study for intensively and extensively managed peatland and the total area for both of these land uses under the assumption that the sites measured in this study

were representative for the western peat area, emissions were estimates for the total intensively managed grassland and extensively managed/unmanaged grasslands in the western peatland. The total terrestrial emission, not taking into account farm-based emissions, estimated using a time-horizon of 100 years (GWP CH_4 = 25 and N_2O = 298) from the western peatland is approximately 1210 Gg CO_2-eq (=kton CO_2-eq) yr^{-1}.

Table 7.10. Estimated area and annual GHG release for the area of intensively managed and extensively managed (mown only) or unmanaged grasslands on peat within the total western peatland region of the Netherlands. Farm-based emissions are not included.

Ecosystem type	Area in western peatland		Total N_2O emission	Total CH_4 emission	Total CO_2 emission
	(ha)	(% of total)	10^3 kg N_2O yr^{-1}	10^3 kg CH_4 yr^{-1}	10^3 kg CO_2 yr^{-1}
Intensively managed grassland	78,375	68%	1881	12853	313498
Extensively managed/ unmanaged grassland	8,786	8%	70	1577	35145
Shallow water bodies	87	6%	unknown	unknown	33583[1]

[1]*An annual emission of 0.5 kg CO_2 m^{-2} yr^{-1} was assumed (Table 7.6.).*

7.3.7. Future scenario for an increase in water storage

An increase in the percentage of surface water in the landscape will increase the total terrestrial CH_4 emissions to the atmosphere. In addition to the current situation (scenario 1), two other simple scenarios with different percentages of open water and dry land in the landscape were investigated for the Stein site (the effect for Oukoop is comparable) and the Horstermeer site (*Table 7.11.*), under the simple assumption that the processes that determine the emissions do not change within the land elements or in time. In Stein, scenario 1 represents 16% water in the landscape, which is the actual situation, scenario 2 represents 26% water in the landscape and scenario 3 represents 36% water in the landscape. In Horstermeer, the percentage of relatively dry land has been changed from 60% (scenario 1, current situation) to 55% (scenario 2) and 45% (scenario 3) and the contributions of open water and waterlogged land were changed (*Table 7.11.*). The emissions of CH_4 were spatially weighted for the contribution of each landform (dry and wet fields, ditches and waterlogged field) in the landscape. The estimated CO_2 emissions are based on the assumption that the NEE from ditches is around 0. *Fig 7.8.* shows that in scenarios 2 and 3 for Stein, CH_4 will be the dominant GHG in terms of warming potential.

Table 7.11. *Mitigation scenario's used for the calculations in Fig 7.8. Scenario 1 is the current situation in the Stein site and the Horstermeer site. Scenario's 2 and 3 describe possible situations if water levels are raised and ditches are enlarged.*

		Scenario 1 (%)	Scenario 2 (%)	Scenario 3 (%)
Stein	Dry field	79	69	59
	Waterlogged edge	5	5	5
	Open water	16	26	36
Horstermeer	Dry field	60	55	45
	Wet field	25	25	30
	Waterlogged field	10	10	15
	Open water	5	10	10

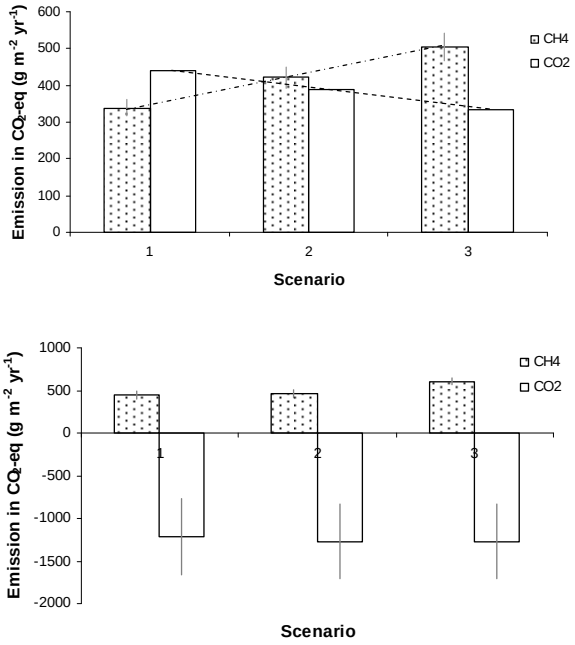


Fig 7.8. *Scenarios 1, 2 and 3 for Stein (left-hand figure) and Horstermeer (right-hand figure) with the emissions of CH_4 (dotted bars) and CO_2 (white bars) in CO_2-eq (g m^{-2} yr^{-1}).*

7.4. Discussion

7.4.1. Balances

Long-term emission values of the GHGs and carbon fluxes were compared for peatlands under different management: a drained intensively managed peatland with application of fertiliser and biomass export, a drained extensively managed peatland with biomass export only and a shallow drained former agricultural peatland. Significant differences in GHG emissions have previously been reported between landscape elements within the three areas: emissions from drainage ditches and waterlogged soil were significantly higher than the fluxes from the relatively dry land (Schrier et al., 2010a; Hendriks et al., 2007). Fluxes originating from the operating farm in the intensively managed peat were found to be important contributors to the GHG balance and to the carbon balance (Schrier-Uijl et al., 2010a; Veenendaal et al., 2007) and therefore, terrestrial as well as farm-based emissions have been included as is shown in *Fig 7.1*. Fluxes through machinery and import through cow feed except grass is not accounted for (Appendix A). Strictly speaking, external C sources from external cow feeds account for roughly 15% of C input of the total amount of cow feed (van Raamsdonk et al., 2007).

Both managed experimental grass-on-peat-areas, Oukoop and Stein, acted as CO_2 sources of on average 0.4 kg CO_2 m^{-2} yr^{-1} (*Table 7.6.*). The variation in NEE CO_2 in Oukoop and Stein was mainly a result of management. Oukoop has the most variable NEE which is a result of the very high frequency of mowing, grazing and manure application (*Fig 7.2.*). Sharp decreases in NEE are a direct result of mowing events (eg. days 853 and 090 in 2007) in Oukoop. In Stein management, and therefore variability in NEE, showed less variability with the first biomass removal on 15 June and the second biomass removal in September of each year. Both areas had a common history in terms of management, and only during the past 20 years has the Stein site gradually become a meadow bird reserve. This change has not resulted in a significant difference of annual NEE compared to the intensive site. The unmanaged site, Horstermeer, acted as a CO_2 sink of on average -1.4 kg CO_2 m^{-2} (*Table 7.6.*). The cumulative NEE shows a stable pattern, with high uptake rates in spring and summer. The Horstermeer site is still under restoration and despite agriculture being abandoned here 15 years ago, soil conditions have remained eutrophic. The continuing nutrient-rich conditions generate high plant productivity and microbial activity, resulting in high carbon fluxes (both uptake and emissions) and more peat accumulating than is oxidised (Hendriks et al., submitted to ecohydrology). Under eutrophic conditions vegetation succession may move this system towards a more natural peat vegetation such as an alder carr forest. Sink strength may remain for a long time but at present little is known about the carbon balance under these conditions. If however nutients are removed from the system by e.g. mowing or top soil removal the system may develop a nutrient poor Spaghnum bog and NEE will decrease.

All three sites, Oukoop, Stein and Horstermeer, acted as sources for CH_4 (terrestrial source strengths, averaged over four years of measurements, of 16.4, 16.7 and 19.2 g CH_4 m^{-2} yr^{-1}). Temporal variability was high at all three sites and CH_4 emissions in the Oukoop en Stein

site varied between days, between seasons and between years, but no diurnal cycles were observed after correction for temperature (Schrier-Uijl et al., 2010b). In the Horstermeer site clear diurnal cycles have been found during all seasons, synchronous with incoming radiation (Hendriks et al. 2009). Hendriks and collegues attribute this to the stomatal opening and/or pressurized convective gas transport through the vascular plants with highest emissions in the late afternoon and lowest emissions during the night, as is also reported for other swamp areas (e.g. Whiting and Chanton, 1993; Hirota et al., 2004). Differences in terrestrial CH_4 emissions between the three sites were not significant. Farm practices in Oukoop and Stein caused estimated additional CH_4 emissions of 26 and 17 g CH_4 m^{-2}. The total source strength (terrestrial + farm-based emissions) decreases when management intensity decreases.

The Horstermeer is a polder under restoration, but the end stage of restoration will depend on the management intensity of the surrounding area affecting groundwater supply and nutrient input. Even if the influx of nutrient-rich water from surrounding areas and the atmospheric nutrient deposition stop it may even be necessary to remove the very eutrophic top layer of soil to allow succession to reach a near-natural system as studied by Van den Pol-van Dasselaar et al. (1999). Van den Pol-van Dasselaar et al. (1999) studied the emission of CH_4 over two years in three near-natural peatlands in a Dutch nature preserve with narrow plots of grasslands (mown once a year), reed beds, fields and ditches and a water table at a depth of 18 cm. The soils were comparable to the soils in this study. The average field-CH_4 fluxes were 13.3, 20.4 and 7.9 g CH_4 m^{-2} yr^{-1} and the ditch fluxes were 11.3 g m^{-2} yr^{-1} on average. After weighting the contributions of the water and land, CH_4 emissions were on average lower than the emissions measured in this study. The reported CH_4 emissions from undisturbed peatlands are highly variable. For example a natural peatland in Quebec, Canada showed CH_4 emissions of 9.8 g m^{-2} yr^{-1} (Moore and Knowles, 1990), and Nykänen et al (1995) reported CH_4 emissions of 34.7 g CH_4 m^{-2} yr^{-1} for Scandinavian undisturbed peat lands.

Comparing ‘polder emissions’ with emissions from large shallow fresh water lakes reveals that water bodies contribute importantly to the CH_4 balance. The calculation of the total CH_4 polder balances for the three sites is based on the current classification of the landscape with 16% open water in Oukoop and Stein and 5% in Horstermeer. Changing the contribution of water and/or waterlogged land in the landscape by reclassification will cause large changes in the CH_4 balance because these landscape elements together are responsible for over 50% of the total flux. In conclusion, large scale spatial differences in CH_4 emission depend on the combination of management and water table and the presence or absence of water bodies. Drainage ditches, large shallow lakes and waterlogged land are CH_4 hotspots and therefore spatial differences greatly depend on the proportion of these landscape elements in the landscape. The temporal variability within sites was largely driven by temperature.

The terrestrial carbon balance (from the field point-of-view) considered in this study consisted of field CO_2-C, field CH_4-C, biomass removal and manure and fertiliser application. The two managed sites acted as C-sources and the unmanaged site acted as a C-sink. In the two managed sites, the CO_2 emission (farm-based + terrestrial) and the biomass removal accounted for the largest part of the C-release. Because in Oukoop carbon was added through manure and fertilizer application, the total C-release was smaller than from to Stein. In the unmanaged site Horstermeer, the C-balance was dominated by the uptake of CO_2-C. Except for the small release of C through CH_4, no other C-sources or sinks were involved in this undisturbed system. Release of C from ditch water was minor (Hendriks et al., 2007). Measurements at the three contrasting sites show that an intensively managed fen meadow area can change from a carbon source towards a carbon sink when the water table is raised to close to the surface and when management is reduced to zero**.**

Measurements of N_2O emission were performed at the Oukoop site for three years, but in the Stein and Horstermeer sites the emissions were too small to be detected by the methods applied in this study. The background emissions in Stein and Horstermeer are estimated at 8 ± 8 kg N_2O ha^{-1} yr^{-1} by Velthof et al. (2007), who defined these background values for Dutch unfertilised but mown grassland sites. The N_2O in these types of ecosystems is produced during the nitrification and/or denitrification of NO_3^-. This nitrate is produced by oxidation of ammonium. Leaching is considered to be negligible, but in this study this is not measured. Continuous measurements in Oukoop showed a typical pattern of long periods with low emissions (background emissions) followed by short periods of high emissions (peak-emissions) around manure application (Kroon et al., 2010). Emissions due to manure application accounted for 25% of the total annual N_2O emissions in Oukoop. The average annual terrestrial N_2O emission in Oukoop was 24 kg N_2O ha^{-1} yr^{-1} which was comparable to N_2O emission values of 17 kg N ha^{-1} yr^{-1} found in an earlier study in non-fertilized managed Dutch peatland (Berendse et al., 1993). The probable reason why background emission was higher in Oukoop than in Stein is the accumulation of easily decomposable organic materials in the soil as a result of manure application 5 times a year. In Stein, this application ceased almost 20 years ago. Hensen et al. (2006) show that manure based emissions from storages around the farm can cause an additional emission of 14.8 mg N_2O m^3 manure d^{-1}. With the 700 m^{-3} slurry stored on the Oukoop farm this would result in an extra (marginal) emission of 3.8 kg N_2O yr^{-1} over 50 ha or 0.08 kg N_2O ha^{-1} yr^{-1}. Around 30% of the temporal variability in N_2O background emission could be explained by a non-linear relationship with temperature and mean wind velocity. Emission factors around manure application were calculated by subtracting the background emission from the total emissions measured by the EC system. The emission factor ranged from 1.2 to 2.8% which is larger than the IPCC default emission factor of 1% (Kroon et al., 2010).

Combining all incoming and outgoing GHG fluxes shows that Oukoop is the largest GHG source in terms of warming potential. In Oukoop, N_2O dominated the emissions and CO_2 and CH_4 contributed equally. In Stein, N_2O was the least contributing GHG and the total emission was lower than in Oukoop. The Horstermeer appeared to be a GHG sink with a release of CH_4 and N_2O from the system, but uptake of CO_2 was large. Changing the management from intensive to extensive and then to unmanaged changes the total GHG balance from release to uptake. When summer emissions of CO_2 and CH_4 were investigated, it was found that water bodies were large contributors to the greenhouse gas balance (Schrier-Uijl et al., 2010a,c).

7.4.2. Potential ways for mitigation

Agricultural peatlands in the Netherlands are mostly fen meadow ecosystems. The mitigation of CO_2, CH_4 and N_2O of these areas is of concern for two reasons: 1) to maintain this fen meadow landscape, it is important to stop peat soils degrading to mineral soils and 2) GHG emissions from peatlands have to be reduced because the international community is committed to the reduction of GHG emissions. Strategies to reduce GHG emissions from these areas and to increase carbon uptake may be rewetting intensively cultivated peatlands, reducing farm-based fluxes and decreasing management intensity.

In this study we found that the rewetting of agricultural peatland can turn areas of carbon release into areas of carbon uptake and changes the GHG balance from GHG sources into GHG sinks. Peat soils without clay layers are extremely vulnerable to oxidation (Schothorst, 1977) and also strongly vulnerable to subsidence. Therefore, on these soils, intensive management practices are not sustainable. With dynamic water tables in the extensively managed polder (high water tables in winter and low water tables in summer), only a small reduction in GHG emission is attained. The lower total emission is mainly due to a decrease in farm-based CH_4 emissions and a reduction in N_2O emissions because no fertiliser is applied. High water tables in summer will likely reduce emissions of CO_2 from extensively managed areas and reduce subsidence although the direct effects (other than reduced intensity of farming) remain uncertain (e.g. Lloyd et al., 2006; Parmentier et al., 2008). Inverse drainage systems may for the purpose of obtaining high water levels in summer be applied. The sink strength in the unmanaged polder may decline in the long term (timescale of centuries) due to a decrease in nutrient availability (Limpens et al., 2008) or remain under nutrient rich conditions (e.g. alder carr forest).

We did not study farm-based emissions separately. Sommer et al. (2009) studied farm-based emissions in Sweden, Denmark, France and Italy and found that shortening the on-farm manure storage and lowering the storage temperatures reduced GHG emissions from manure by 0-40% depending on current management and climatic conditions. Large GHG

reductions were obtained when slurry was separated into a liquid component and a solid, organic component and the liquid fraction was applied earlier than the solid fraction.

Looking at the sum of CO_2 and CH_4 emissions, the summer emission from large shallow lakes are higher than the emissions from the intensively and extensively managed polders when considering the sum of CO_2 and CH_4 emissions, but lower than the emissions from drainage ditches within the polders. Reducing the inputs of organic material and nutrients from the surroundings will probably reduce emissions from these water bodies (Schrier-Uijl et al., 2010c). This suggests there is a strong link between emissions from ditches and the intensity of the management in the polders within the catchment area.

7.4.3. Gaps in knowledge

Little is known about the emissions from open water bodies, as this has scarcely been studied and so is poorly understood. It has been shown that summer emissions of methane from water bodies can contribute significantly to the summer release in the fen meadow area (Schrier-Uijl et al., 2010a, c; Kankaala et al., 2007) and they may play an important role in the total annual GHG balance of the Dutch fen meadow area (*Fig 7.5.*). When establishing emission factors for the peatlands these landscape elements should be included in the inventories.

The estimates in this study refer to the landscape scale. Factors that play a major role in the control of emissions have been indentified. Local emissions, from differently contributing landscape elements have been extrapolated to landscape-scale emissions. Large variability in time has been captured by using recently developed continuous measurement techniques. Although measurement techniques for quantifying gas fluxes have improved considerably, large uncertainties remain in the national and regional budgets. This study shows that because of the large temporal variability (daily, seasonal, annual and inter-annual), continuous measurement systems are needed to capture this temporal variability and to study the underlying processes that drive the emissions.

The effect of a changing climate in the near future in terms of temperature rise and changes in rainfall patterns has not been studied. The result reported here, that all emissions were temperature related (the higher the temperature, the higher the emission) implies that the predicted future temperature rise will be accompanied by increased peat oxidation. Peat subsidence will therefore increase, and so it is likely that the peatlands will be more vulnerable to the anticipated more frequent extreme rainfall events in terms of flooding risks. However, the GHG balance for a scenario of increasing the area of open water in peatlands, which is both a mitigation and an adaptation measure, was estimated (*Table 7.11.*and *Fig 7.8.*). An increase in the area of open water increased the CH_4 emissions,

decreased the CO_2 emissions and had in total a negative effect on the GHG balance. Differences in soil conditions (e.g. peat origin and depth, C content of the peat) and past land management (e.g. drainage, nutrient input) may influence present mitigation. For instance differences in organic content between the sites may have lead to a conservative estimate of GHG reduction for the management option of the Horstermeer, which has a higher C content in the topsoil and therefore presumeably a high GHG potential.

7.4.4. Conclusions

The global community has agreed on the need to reduce GHG emissions, including CO_2 CH_4 and N_2O. In this study, attention focused on the quantification of emissions from peatlands under different management, the assessment of various GHG mitigation strategies and on finding effective options to reduce emissions.

Table 7.12. gives an overview of the expected effects of rewetting and management reduction on the GHG balance.

Table 7.12*. Overview of the expected effects of different mitigation strategies on the total GHG balance (including emissions due to management). The effect on the GHG balance has been determined for the three research sites Oukoop (intensively managed), Stein (extensively managed) and Horstermeer (unmanaged) and does not include the expected future temperature rise. (-) = decrease in emission, (+) = increase in emission, (0) = neutral effect, (?) = effect unknown, (x) = not relevant.*

	Rewetting+management reduction			Management reduction towards extensively managed			Increase in % open water		
	CO_2	CH_4	N_2O	CO_2	CH_4	N_2O	CO_2	CH_4	N_2O
Intensive management	–	–	–	0	–	–	–	+	–
Extensive management	–	–	–	x	x	x	–	+	?
Unmanaged	x	x	x	x	x	x	–	+	?
Open water	–	–	–	–	–	–	x	x	x

This study provides evidence that intensively managed agricultural peat soils are large sources of GHG and carbon. However, if appropriate measures are taken such as rewetting the peat soils and reducing the management intensity, these peat soils can be turned back into GHG sinks and carbon sinks again. In addition, when implementing mitigation strategies to reduce emissions from one source, GHG emissions from other sources might also be reduced. For example, reducing the input of nutrient-rich water or groundwater in lakes and drainage ditches by reducing management in the surrounding catchments, will probably also reduce emissions from water bodies. A change in landuse from intensive

agriculture to nature restoration indeed influences a number of interconnected processes and leads to large changes in the greenhouse gas balance of fen meadow areas.

7.5. Acknowledgements

This research project was funded by Wageningen University, The Province of North Holland, CarboEurope IP (contract number: GOCE-CT-2003-505572) and the Dutch National Research Programme Climate Changes Spatial Planning (www.klimaatvoorruimte.nl). Many thanks are due to the field assistants for their help in the field and in the lab and to the van Eijk and van Eeuwijk families for their logistical support. We are grateful to the State Forestry Service for access to the site and to Nico de Bruin for his collaboration. J. Burrough advised on the English.

7.6. References

Berendse, F., Oomes, M.J.M., Altena, H.J., Visser, W. (1994). A Comparative Study of Nitrogen Flows in Two Similar Meadows Affected by Different Groundwater Levels. Journ. Of Applied Ecol. 31, pp. 40-48.

Burgerhart, N. (2001) in Dutch: Mogelijkheden voor koolstofopslag in Nederlandse ecosystemen. Dutch National Inventory Report for LULUCF sector, category 5D1.

Christensen, T. R., Ekberg, A., Strom, L., Mastepanov, M., Panikov, N., Mats, O., Svensson, B. H., Nykanen, H., Martikainen, P. J., and Oskarsson, H. (2003): Factors controlling large scale variations in methane emissions from wetlands, Geophysical Research Letters, 30.

Dalal, R.C., Allen, D.E., Livesley, S.J. (2007): Magnitude and biophysical regulators of methane emission and consumption in the Australian Agricultural, forest, and submerged landscapes: a review. Plant and Soil, doi:10.1007/s11104-007-9446-7.

Falge, E., Baldocchi, D., Olson, R., Anthoni, P., Aubinet, M., Bernhofer, C., Burba, G., Ceulemans, R., Clement, R., Dolman, H., Granier, A., Gross, P., Grunwald, T., Hollinger, D., Jensen, N. O., Katul, G., Keronen, P., Kowalski, A., Lai, C. T., Law, B. E., Meyers, T., Moncrieff, H., Moors, E., Munger, J. W., Pilegaard, K., Rannik, U., Rebmann, C., Suyker, A., Tenhunen, J., Tu, K., Verma, S., Vesala, T., Wilson, K., and Wofsy, S. (2001): Gap filling strategies for defensible annual sums of net ecosystem exchange, Agr. Forest Meteorol., 107, pp. 43–69.

Guinand-Flment, F., Delamaire, E., Lamberton, P., Peyraud, J.L. (2007): Adaptations of mammary uptake and nutrient use to once-daily milking and feed restriction in dairy cows, J. Dairy Sci. 90, pp 5062-5072.

Hendriks, D. M. D., van Huissteden, J., Dolman, A. J., and van der Molen, M. K. (2007): The full greenhouse gas balance of an abandoned peat meadow, Biogeosciences, 4, 411-424, 2007.

Hendriks, D.M.D., van Huissteden, J., Dolman, A.J. (2009). Multi-technique assessment of spatial and temporal variability of mehtane fluxes in a peat meadow. Ag. and Forest Meteorol., doi:10.1016/j.agrformet.2009.06.017.

Hendriks 2009b, thesis: Integrated observations of greenhouse gas budgets at the ecosystem level – changing environment and management practives in peat meadows.

Hensen, A., Groot, T.T., van den Bulk, W.C.M., et al. (2006): Dairy farm emissions, from one square meter to the full farm scale. Agriculture, Ecosystems and Environment 112: Issues 2-3, 146-152.

Hirota, M., Tang, Y.H., Hu, Q.W., Hirata, S., Kato, T., Mo, W.H., Cao, G.M., Mariko, S. (2004): Methane emissions from different vegetation zones in a Qinghai-Tibetan Plateau wetland. Soil Biology and Biochemistry 36, pp737-748.

Howden, S.M.and Reyenga, P.J. (1999): Methane emissions from Australian livestock: implications of the Kyoto Protocol. Australian Journ. Of Agr. Res. 50, pp. 1285-1291.

Iepema, G., Baars, T., Bolk, L. (2005) in Dutch: Afgewenteld grondgebruik op melkveebedrijven: externe hectares, Rapport 10 Bioveem.

IPCC, 2006: Guidelines for National Greenhouse Gas Inventories. Prepared by the National Greenhouse Gas Inventories Programme, Eggleston, H.S., Buendia, L., Miwa, K., Ngara T. & Tanabe, K., (eds), IGES, Japan.

IPCC 2007. Climate change 2007.The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Janssens, I. A., Medlyn, B., Gielen, B., Laureysens, I., Jach, M. E., Van Hove, D. & Ceulemans, R. (2005): Carbon budget of Pinus sylvestris saplings after four years of exposure to elevated atmospheric carbon dioxide concentration. Tree Physiology 25: 325-337.

Joosten, H and Clarke, D. (2002): Wise use of mires and peatlands – Background and principles including a framework for decision-making. International Mire Conservation Group / International Peat Society, 304 pp.

Kankaala P, Taipale S, Nykanen H et al (2007): Oxidation, efflux, and isotopic fractionation of methane during autumnal turnover in a polyhumic, boreal lake. J Geoph Res-Biogeosc. 112 G2 G02003.

Kempen, B., Brus, D.J., Heuvelink, G.B.M., Stoorvogel, J.J. (2009): Updating the 1:50,000 Dutch soil map using legacy soil data: A multinomial logistic regression approach. Geoderma 151, pp. 311-326.

Kormann, R., Meixner, F.X., 2001. An analytical footprint model for neutral stratification. Boudary-Layer Meteorology: 99, 207-224.

Kroon, P.S., Hensen, A., Jonker, H.J.J., Zahniser, M.S., van 't Veen, W.H.,Vermeulen, A.T. (2007): Suitability of quantum cascade laser spectroscopy for CH_4 and N_2O eddy covariance flux measurements, BG, 4, 715-728.

Kroon, P.S., Hensen, A., Jonker, H.J.J., Zahniser, M.S., Van het Veen, W.H. & Vermeulen, A.T. (2007): Suitability of quantum cascade laser spectroscopy for CH_4 and N_2O eddy covariance flux measurements. Biogeosciences, 4, 715-728.

Kroon, P.S., Hensen, A., Jonker, H.J.J., Ouwersloot, H.G. Vermeulen, A.T. & Bosveld, F.C. (2009): Uncertainties in eddy covariance flux measurements assessed from CH_4 and N_2O observations. Agric. For. Meteorol., doi:10.1016/j.agrformet. 2009.08.008.

Kroon, P.S., Schrier-Uijl, A.P., Hensen, A., Veenendaal., E.M., Jonker, H.J.J., (2010a): Annual balances of CH_4 and N_2O from a managed fen meadow using eddy covariance flux measurements. Eur. J. Soil Sci.

Kroon 2010b, Thesis: Eddy covariance observations of methane and nitrous oxide emissions - towards more accurate estimates from ecosystems.

Langeveld, C.A., Segers, R., Dirks, B.O.M., Van den Pol-van Dasselaar, A., Velthof, G. C. & Hensen, A. (1997): Emissions of CO_2, CH_4 and N_2O from pasture on drained peat soils in the Netherlands. Eur. J. Agron., 7(1-3), 25-42.

Limpens, J., Berendse, F., Blodau, C., Canadell, J.G., Freeman, C., Holdens, Roulet, N., Rydin, H., and Scheapman-Strub, G. (2008): Peatlands and the carbon cycle: from local processes to global implications – a synthesis. Biogeosciences, 5, pp. 1475–1491.

Lloyd, J., Taylor, J.A. (1994): On the temperature dependency of soils. Functional Ecology: 8, 315-323.

Lloyd, C.R. (2006): Annual carbon balance of a managed wetland meadow in the Somerset Levels, UK. Agricultural and Forest Meteorology 138, pp. 168-179.

Maas, C.W.M., Coenen, P.W.H.G., Ruyssenaars, P.G., Vreuls, H.H.J., Brandes, L.J., Baas, K., Van den Berghe, G., Van den Born, G. J., et al., (2008): Greenhouse gas emissions in the Netherlands 1990-2006, National inventory report 2008. MNP report 500080009, Netherlands Environmental Assessment Agency (MNP).

Moore, T.R and Knowles, R (1990): Methane emissions from fen, bog and swamp peatlands in Quebec. BIOGEOCHEMISTRY 11, pp. 45-61.

Nieveen, J.P., Campbell, D.I., Schipper, L.A., Blair, I.J. (2005): Carbon exchange of grazed pasture on a drained peat soil, Global Change Biology 11, pp 607-618.

Nol, L., Verburg, P.H., Heuvelink, G.B.M., Molenaar, K., (2008): Effect of land cover data on nitrous oxide inventory in fen meadows. J. of Envir. Qual.: 37, pp. 1209-1219.

Martikanen, P.J., Nykanen, H., Alm, J., Silvola, J (1995): Cahnge in fluxes of carbon-dioxide, methane and nitrous oxide due to forest drainage of mire sites of different trophy. Plant and Soil 168, pp. 571-577.

Parmentier, F.J.W., van der Molen, M.K., de Jeu, R.A.M., Hednriks, D.M.D., Dolman, A.J. (2008): CO_2 fluxes and evaporation on a peatland in the Netherlands appear not

affected by water table fluctuations. Agricultural and Forest Meteorology, 149, pp. 1201-1208.

Raamsdonk, L.W.D., Kan, C.A., Meijer, G.A.L., Kemme, P.A. (2007) in Dutch: Kengetallen van enkele landbouwhuisdieren en hun consumptiepatronen, Rapport 2007.010, RIKILT – Instituut voor voedselveiligheid.

Ruiter, J.F., Van Pul, W.A.J., Van Jaarsveld, J.A., & Buijsman, E., 2006: Zuur- en stikstofdepositie in Nederland in de periode 1981-2002. RIVM rapport 500037005/2006 (In Dutch).

Schrier-Uijl, A.P., Veenendaal, E.M., Leffelaar, P.A., van Huissteden, J.C., Berendse, F. (2010a): Methane emissions in two drained peat agro-ecosystems with high and low agricultural intensity. Plant Soil, doi:10.1007/s11104-009-0180-1.

Schrier-Uijl, A.P., Kroon, P.S., Hensen, A., Leffelaar, P.A., Berendse, F. & Veenendaal, E.M. (2010b): Comparison of chamber and eddy covariance based CO_2 and CH_4 emission estimates in a heterogeneous grass ecosystem on peat, Agric. For. Meteorol., doi:10.1016/j.agrformet.2009.11.007.

Schrier-Uijl, A.P., Veraart, A.J., Leffelaar, P.A., Berendse, F., Veenendaal, E.M (2010c): Release of CO_2 and CH_4 from lakes and drainage ditches in temperate wetlands. Biogeochemistry, doi:10.1007/s10533-010-9440-7.

Schulp, C.J.E., Nabuurs, G.J., Verburg, P.H (2008): Future carbon sequestration in Europe – Effects of land use change. Ecosyst. And Enivronm. 127, pp. 251-264.

Sneath, W, Beline F, Hilhorst A, Peu P (2006): Monitoring GHG from manure stores on organic and conventional dairy farms, Agric. Ecosyst. Environ. 112, pp. 122–128.

Sommer, S.G., Olesen, J.E., Petersen, S.O., Weisberg, M.R., Vallis, L., Rodhe, L., Beline, F. (2009): Region-specific assessment of greenhouse gas mitigation with different manure management strategies in four agro ecological zones. Global Change Biology 15, pp. 2825-2837.

Soussana, J.F., Loiseau, P., Vuichard, N., Ceschia, E., Balesdent, J., Chevallier, T., Arrouays, D. (2004): Carbon cycling and sequestration opportunities in temperate grasslands, Soil Use and Management 20, pp. 219-230.

Soussana, J,F., Maire, V., Gross, N., Reinhold, T., Dahring, H., Pontes, L.S., Picon-Cochard, C., Fontaine, S., Pages, L., Wirth, C. (2008): Modeling the relationships between the diversity and the functioning of pasture swards with a complex floristic composition, Fourrages 195, pp. 259-274.

Stouthamer, E. and ten Brinke, W. (2008), in Dutch: Fluvial sedimentology, Geologie en mijnbouw 86, pp. 1-1.

TDN, 2006. TOP10vector Product information (in Dutch): Topografische Dienst Kadaster Emmen the Netherlands. Available at http://www.tdn.nl/ (verified 6 Dec 2007).

Van Amstel A., Kroeze C., van Eerdt M., Dumont M., Both D. (2003): National emissions of methane and nitrous oxide from agriculture in The Netherlands: current estimates and future trends, Workshop on Inventories and projections of greenhouse gas and

ammonia emissions from agriculture, 27–28 February 2003, European Environmental Agency, Copenhagen.

Van den Pol-van Dasselaar, A., Van Beusichem, M.L., Oenema, O. (1999): Effects of nitrogen input and grazing on methane fluxes of extensively and intensively managed grasslands in the Netherlands. Biol. Fertil. Soils, 29, pp. 24-30.

Veenendaal, E.M., Kolle, O., Leffelaar, P.S. Schrier-Uijl, A.P., Van Huissteden, J., Van Walsem, J., Möller, F. & Berends, F., 2007: CO_2 exchange and the carbon balance in two grassland sites on eutrophic drained peat soils. Biogeosciences, 4, pp. 1027-1040.

Velthof, G.L. and Oenema, O., (1997): Nitrous oxide emission from dairy farming systems in the Netherlands. Netherlands Journal of Agricultural Science, 45, pp. 347-360.

Whiting, G.J. and Chanton, J.P. (1993): Primary Production Control of Methane Emission from wetlands. Nature. 364, pp. 794-795.

Chapter 8

Summary

,

8.1. Introduction.

Peatlands, which cover 3% (4,000,000 km^2) of the world's land area and between 6 and 7% (273,000 ha) of the Dutch land area, are the most carbon-dense ecosystems of the terrestrial biosphere. Worldwide they store about 550 Gton of carbon. This huge stock of carbon can exist only under very wet conditions. When peatlands are drained, e.g. for agricultural use, large amounts of carbon are released as CO_2. The degradation of peatlands is a major and growing source of anthropogenic greenhouse gas (GHG) emissions, and small changes in the management of peatlands can lead to drastic changes in GHG emissions and changes in carbon storage and in the past have indeed done so. GHG emissions from peatlands and the subsidence of peat soils can both probably be decreased by rewetting peatland and by restoring peatland by reducing the intensity of agricultural land use on peat soils. It might even be possible for these agricultural peatlands to revert to being sinks of GHGs.

To test whether agricultural peat areas can be turned into GHG sinks and carbon sinks if they are restored and whether GHG emissions can at least be reduced by reducing management intensity and rewetting, a landscape-scale experiment measuring GHG emissions and carbon releases was started in 2005. Field experiments were set up and data was collected in peat areas under different management regimes in the west of the Netherlands. In an additional experiment, emissions from drainage ditches and lakes were studied in detail. The research objectives were:

- to quantify the emission of greenhouse gases (CO_2, CH_4, N_2O) that are produced at small-scale (1mx1m), plot scale (ha scale) and landscape scale (whole polder scale) in Dutch fen meadows;
- to determine the spatial and temporal variability of fluxes at different scales;
- to develop a system for upscaling GHG emission from a small-scale to landscape scale and to provide predictions based on regression models relating emissions with measured explanatory variables;
- to answer the question: can restoration of fen meadows turn these areas into sinks when the greenhouse gases CO_2, CH_4 and N_2O are included in the emission balance?

8.2. Summary of research

The research focused on quantifying the full GHG balances (including CO_2, CH_4 and N_2O) of two agricultural peat areas in the Netherlands: an intensively managed dairy farm peatland with application of manure and fertiliser and an intensive mowing regime (Oukoop polder) and a peatland managed extensively (only mowing): Stein polder. The GHG balances and carbon balances of the two agricultural sites (Oukoop and Stein) were

compared with those of an abandoned former agricultural peatland under restoration, which was studied in detail by Hendriks (2009b). To this end, all GHG fluxes (uptake and release) and possible explanatory variables were measured on the experimental sites.

Fluxes of the major GHGs vary spatially as well as temporally and therefore there are large uncertainties in regional and local GHG budgets and the emission of non-CO_2 GHGs is particularly poorly understood. The research project therefore had to use an integrated approach which entailed using different measurement techniques for different spatial and/or temporal scales. The spatial and temporal variability of fluxes at small scale was studied in detail by performing small-scale chamber measurements in fields (on dry, wet and saturated parts), drainage ditches and lakes. Spatial variability at large scale (between ecosystems) was studied by using all data available and by comparing the total balances of the GHGs (CO_2, CH_4 and N_2O). Temporal variability was studied by using chamber-based data for seasonal to inter-annual variation and continuous data, collected by eddy covariance systems, for daily to inter-annual variation. The chamber-based method typically enables measurements at the scale of a few dm^2 and is the approach most commonly applied to capture small-scale variability of CO_2, CH_4 and N_2O fluxes. The eddy covariance technique is commonly used for assessing GHG fluxes over areas of a few ha and is the approach most commonly applied to capture temporal variability of CO_2 fluxes. The eddy covariance technique has recently become available for CH_4 and N_2O; the data used in this thesis were from a new system which has been evaluated in detail by Kroon (2010b).

Two different measurement techniques i.e. chamber-based and eddy covariance-based, were used to upscale fluxes temporally and spatially. Driving variables (e.g. temperature, water table and wind velocity) that may explain GHG fluxes were measured for periods of up to four years and their relationships with emissions quantified by using multiple linear or nonlinear regression analyses. Temporal upscaling was done by using the statistically significant regression models. The continuous time series of the independent driving variables could then be used to estimate complete balances. Chamber-based data was most suitable for spatially upscaling from fluxes within sites. The experimental set-up of the chamber experiment was designed so that all contributing landscape elements (dry/wet fields, saturated ditch edges and drainage ditches) were proportionally sampled for GHGs and other variables. Because the drainage ditches in the experimental peatlands appeared to be emission hot-spots, in the summer of 2009 an additional experiment was performed on shallow lakes and drainage ditches in peat areas in the west and east of the Netherlands, to obtain a first impression of emissions from these landscape elements.

Different flux measurement techniques should give similar results even if they measure at different scales. The comparability of the chamber technique and the eddy covariance technique was tested by comparing year-round eddy covariance data for CO_2 and a three-month period of eddy covariance data for CH_4 with statistical regression models for CO_2

and CH_4 based on chamber measurements and explanatory variables. Additionally, a direct comparison experiment of simultaneous measurements for CH_4 was performed on a large shallow lake in the west of the Netherlands. The chamber data were collected within the footprint area of the eddy covariance systems; small-scale heterogeneities were accounted for by using weighting factors for the areas of the landscape elements that contributed to the GHG balance.

In summary, full GHG balances and carbon balances were quantified in two agricultural peatlands with high and low agricultural intensity and these were compared with the full GHG balance and the carbon balance of an abandoned former agricultural peatland. Spatial variability was accounted for by taking into account the different landscape elements (dry/wet field, saturated soil and drainage ditches) within a peatland area, whereas temporal variability was captured by using regression models for upscaling and by using eddy covariance techniques. To quantify the total GHG balances and carbon balances of the two agricultural systems, fluxes related to management, such as the application of manure and fertiliser, mowing and grazing and farm-related emissions such as emissions from manure storages and enteric fermentation by dairy cattle were also included. In this study these management-related fluxes were not measured but were derived from research by others. Because open water bodies form part of the total peat area in the Netherlands, an additional experiment was performed in which emissions were measured on 6 large lakes and 12 drainage ditches in the western and eastern peat areas.

8.3. Conclusions

Chapter three of this thesis reports on the influence of reduced agricultural activity on the CH_4 balance of the two peat areas (managed intensively and extensively) peat areas. Chamber measurements were performed in the different landscape elements and potential explanatory variables were monitored during three years. The main conclusions were:

- Temporal variability of CH_4 emissions was high, with high fluxes concentrated in summer and spring, partly associated with applications of slurry.
- Spatial variability of CH_4 emissions was high; drainage ditches and adjacent ditch edges were emission hot spots in the landscape, accounting for over 60% of the total annual CH_4 emission in both peatlands.
- A non-linear, upscaling approach based on exponential regressions proved more reliable than a linear interpolation approach, since in the linear interpolation approach exceptionally high emissions led to an overestimation of annual emissions.
- With the exception of temperature, no measured variables contributed significantly to the predictive power of the chamber-based non-linear regressions.

- Both areas were a net terrestrial source of CH_4 (range 146 to 203 kg ha^{-1} for intensively managed soils and 156 to 180 kg ha^{-1} for extensively managed soils) and there was no significant difference between the intensively and extensively managed peatlands.
- Taking into account the farm-based CH_4 emissions, the source strength increased to 427 kg ha^{-1} for the intensively managed area and 339 kg ha^{-1} for the extensively managed area.

In order to establish an independent validation of CO_2 and CH_4 balances, data from two large-scale eddy covariance measurement set-ups was compared with data from small-scale chamber measurement set-ups (chapter 4). The chamber-based data was upscaled in a heterogeneous landscape that comprised fields, ditch edges and ditches. Two alternative ways of upscaling were tested: the first considered only the field, and in the second, fluxes were weighted with fixed coverage fractions for the three main landscape elements within the average footprint area of the mast. The results show that:

- In the procedure of spatial upscaling it is very important to determine the fluxes associated with all relevant landscape elements since each contributes differently to the CH_4 balance. When only fields were included, the difference between eddy covariance and upscaled chamber-based cumulative emissions was 31% for CO_2 (higher fluxes measured by chambers) and 55% for CH_4 (higher fluxes measured by eddy covariance). When all landscape elements were taken into account, cumulative emissions agreed: the difference was 16.5% for CO_2 and 13% for CH_4, and the emission values obtained from the chamber-based method were higher than those obtained from the eddy covariance method.
- The temporal variation of CO_2 and CH_4 fluxes as measured by eddy covariance was large, whereas the emission rates estimated by the empirical model were smooth. The models based on chamber measurement could not be used to capture daily variation in emission.
- Small-scale chamber measurements can be used to estimate fluxes of CO_2 and CH_4 at landscape scale if measurements are performed in the quantitatively significantly contributing landscape elements.
- The results demonstrate that chamber measurements and eddy covariance measurements can be used individually or in conjunction to assess terrestrial GHG emissions.

The fifth chapter deals with the GHG balance of the intensively managed agricultural peat area. Measurements were performed for almost three years with a very recently validated eddy covariance system for CH_4 and N_2O. The main findings include:

- The average annual CH_4 emission was 165 kg CH_4 ha^{-1} yr^{-1} and the average annual N_2O emission during the study period was 24 kg N_2O ha^{-1} yr^{-1}.
- The agricultural peatland was an important source of all three gases, contributing 30% CO_2, 25% CH_4 and 45% N_2O in the total balance.
- The estimates of annual CH_4 emission measured by eddy covariance were less uncertain than the annual CH_4 emission estimates obtained from chamber measurements.
- Emissions due to fertiliser events amounted up to 30% of the total annual N_2O emission (background emission plus event emission due to manure application).

In the sixth chapter, the magnitude of CH_4 and CO_2 fluxes from 12 water bodies in Dutch peatlands in the summer of 2009 is determined. We studied the factors that might regulate emissions from these lakes and drainage ditches. The main conclusions from this short study were:

- Lakes and drainage ditches acted as sources of CO_2 and CH_4.
- Per unit of area, the fluxes from drainage ditches were significantly larger than those from large shallow lakes.
- A higher trophic status was positively correlated with summer emissions of CO_2 and CH_4, whereas the depth of the water and the pH were inversely correlated with CO_2 emission.
- Reducing the nutrient loads and input of organic substrates to ditches and lakes will probably reduce summer emissions of CO_2 and CH_4 from these water bodies.

Chapter 7 provides an overview of all data collected in the period 2005 – 2009 in the peatland landscape-scale experiment in order to test the hypothesis that agricultural peatlands can be turned into sinks of GHGs and carbon. Three peatlands with different management were compared: 1) an intensively managed agricultural peatland, 2) an extensively managed agricultural peatland and 3) a peatland where as part of restoration management the agricultural exploitation ceased 15 years ago (around 1995). We integrated the effects of reducing the agricultural management practices and allowing the water table to rise in the currently and formerly intensively managed agricultural peat areas on the GHG balance and the carbon balance of lowland peat areas. The main conclusions were:

- Both managed sites acted as sources for CO_2: the source strength was 0.4 kg CO_2 m^{-2} yr^{-1} averaged over four years of eddy covariance measurements.
- The unmanaged site acted as a sink for CO_2: the sink strength was 1.4 kg CO_2 m^{-2} yr^{-1} averaged over four years of eddy covariance measurements.
- Variation in net ecosystem exchange of CO_2 in the agricultural peatlands was mainly a result of management; farm-based emissions were estimated at 0.7 kg CO_2 m^{-2} yr^{-1} for both sites.

- Net ecosystem exchange of CO_2 in the unmanaged site showed high uptakes in spring and summer.
- The two managed sites and the unmanaged site all acted as sources of CH_4, with source strengths of 16.4, 16.7 and 19.2 g CH_4 m^{-2} yr^{-1}, averaged over three years of measurements.
- Changing the proportion of water and/or waterlogged land in the landscape will probably result in large changes in CH_4 emission since these landscapes together contribute over 60% of the total annual CH_4 emission.
- Large-scale spatial differences in CH_4 emission depend on the combination of management and water table and the presence or absence of water bodies and saturated land.
- The two managed sites acted as carbon sources: the source strength was 3.8 ton C ha^{-1} yr^{-1} for the intensively managed peatland and 5.4 ton C ha^{-1} yr^{-1} for the extensively managed peatland.
- The unmanaged peatland acted as carbon sink: the sink strength was 4.4 kg C ha^{-1} yr^{-1}.
- Around 30% of the temporal variability of background emissions of N_2O was attributable to a non-linear relationship with temperature and wind velocity.
- Farm-based N_2O emissions from manure storage can result in an extra 1.5 10^{-2} g N_2O m^{-2} yr^{-1} (based on Hensen et al., 2006).
- Combining all incoming and outgoing GHG fluxes showed that the intensively managed peatland is the largest source in terms of warming potential.
- Release of N_2O dominated the GHG emissions in the intensively managed peatland, release of CO_2 and CH_4 dominated the emissions from the extensively managed peatland, and uptake of CO_2 dominated in the unmanaged peatland.
- The intensively managed peatland is a large GHG source and a large carbon source, but, if appropriate measures are taken, these peat soils can be turned back into GHG sinks and carbon sinks within 15 years of abandonment. The exact length of time needed to switch from a GHG source to a GHG sink is still uncertain.
- The sink strength in the unmanaged polder may decline in the long term (timescale of centuries) due to a decrease in nutrient availability and development towards a Spaghnum bog ecosystem, or remain under nutrient rich conditions and development to e.g. a alder carr forest.

The main conclusion is that spatial variability and temporal variability of GHG emissions in fen meadow ecosystems is high. Uncertainties in current emission inventories can be reduced by using a combination of small-scale and large-scale measurement methods. By using a well-designed experimental set-up and including all significantly contributing landscape elements, small-scale emission measurements can be upscaled spatially by exploiting robust predictive relationships with the significant explanatory variables. Intensively managed peatlands are large sources of GHGs and carbon, but, if appropriate

measures are taken, these peat soils can be turned back into GHG sinks and carbon sinks within 15 years of abandonment, by rewetting and reducing management.

8.4. Recommendations and potential ways for mitigation

We concluded that chamber-based measurement and eddy covariance are both reliable methods to quantify landscape-scale GHG budgets. However, we recommend the use of both methods for estimating landscape-scale emissions. On the one hand it is very important for management strategies to understand the relative small-scale spatial variability and to upscale fluxes to larger scales and also outside the eddy covariance mast's footprint. Eddy covariance systems measure emissions over areas of a few ha, and thus small-scale spatial variability is averaged out. On the other hand, the temporal variability of emissions is high, and is not captured fully by chamber measurements. Chamber-based systems are mostly manually operated and therefore is it impossible to cover the temporal scales of eddy covariance systems (30-minute time resolution) which means that from a practical point of view, temporal variability can be measured only at lower temporal resolutions (e.g. weeks). It would be helpful to use automated flux chamber set-ups for observations over longer periods and to capture both temporal and spatial (also daily) variability of emissions.

The hydrological situation and the average temperature in Dutch fen meadow ecosystems will change in the future because of climate change. Given the conclusion in this thesis that all emissions (CO_2, CH_4 and N_2O) in the fen meadow ecosystems studied increase with temperature, it is also important to further investigate the effect of the expected 1–2^{o}C temperature rise by 2050 in combination with the expected increase in extreme rainfall events (ref climate scenario's developed by KNMI) on the GHG balance of these ecosystems. It has been shown in this study that oxidation of Dutch peat soil is likely to occur when temperatures rise and thus the low land peat soils in the Netherlands will be more vulnerable to the expected extreme rainfall events because of the risks of flooding.

We concluded that lakes and drainage ditches in peatlands are CH_4 hotpots and that they contribute significantly to the total GHG balance. Freshwater bodies should therefore be included in national inventories. We recommend year-round combined eddy covariance and chamber-based measurements on lakes and ditches. Further work is also needed to elucidate the processes underlying the formation, transport and emissions of GHGs, in order to be able to improve the models simulating the mechanisms of GHGs from these water bodies. Although we suggest that decreasing nutrient loads and organic matter inputs of the surrounding catchments will reduce emissions, this hypothesis has to be tested.

This research evaluated the effect of rewetting and reducing the management on the total GHG balance and the carbon balance of an intensively managed agricultural peatland. We concluded that the GHG balance as well as the carbon balance can switch from positive (release) to negative (uptake) if these agricultural peatlands are rewetted and intensive agricultural practices have been reduced. However, the restored peatland investigated in this study was abandoned 15 years ago and was still eutrophic. In the longer term, the effect of rewetting and reduced management might decrease when the ecosystem evolves into a peatland with more ”natural” conditions and with decreased internal nutrient status. Therefore, we recommend further comparison studies on near-natural peat marshes where the input of nutrient-rich groundwater and runoff has been reduced and alder carr forests where the input of nutrient-rich groundwater remains.

Mitigation of the emissions of CO_2, CH_4 and N_2O of agricultural peatlands is a priority for two reasons: 1) it is important to stop peat soils degrading to mineral soils and 2) GHG emissions from peat areas have to be reduced because the international community is committed to reducing GHG emissions. Strategies to reduce emissions from these areas and to increase carbon uptake may be oriented towards rewetting of intensively cultivated peat areas, reducing farm-based emissions and decreasing management intensity.

- This study has shown that rewetting and decreasing the management intensity could lead to an emission reduction of 24.8 tons CO_2 equivalents per hectare per year if farm-based emissions are included.
- The dynamic water tables in the extensively managed site with high water tables in winter and low water tables in summer did not lead to a significant reduction in terrestrial emissions compared with the intensively managed site. Higher water tables in summer will probably reduce emissions from intensively and extensively managed fen meadow areas, although the effects remain uncertain.
- Decreasing the nutrient loads and organic matter inputs in drainage ditches and large shallow lakes will probably decrease emissions from these water bodies in the long term. This suggests there is a strong link between emissions from water bodies and the intensity of the management in the polders within the catchment area. However, it is uncertain how long the effect of reduced management on the emissions will last.
- Shallow water bodies contribute significantly to the CH_4 balance and therefore these landscape elements should be included in national emission inventories. Increasing the area of open water and/or saturated land in all three experimental peatlands will increase CH_4 emissions and will have reduce the total GHG balance. It will have only a marginal effect on the carbon balance.

Table 8.1 gives an overview of the expected effects of different mitigation strategies on the total GHG balance of the three experimental sites.

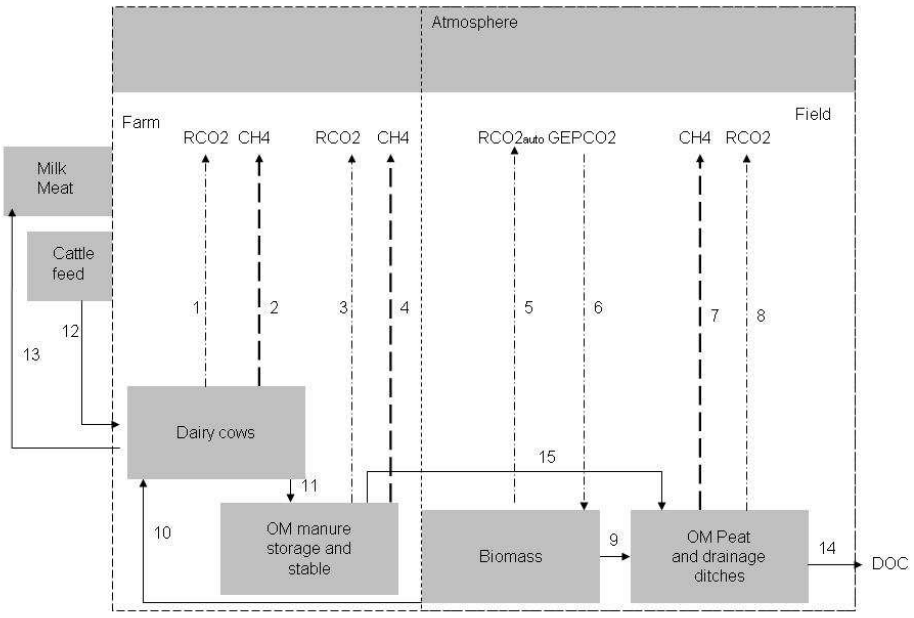
Table 8.1. *Overview of the expected effects of different mitigation strategies on the total GHG balance (including emissions due to management) based on this research. The effect on the GHG balance has been determined for the three research sites Oukoop (intensively managed), Stein (extensively managed) and Horstermeer (unmanaged) and does not include the expected future temperature rise. (-) = decrease in emission, (+) = increase in emission, (0) = neutral effect, (?) = effect unknown, (x) = not relevant.*

	Rewetting+reduced management			Management reduced (towards extensively managed)			Increase in % open water		
	CO_2	CH_4	N_2O	CO_2	CH_4	N_2O	CO_2	CH_4	N_2O
Intensive management	–	–	–	0	–	–	–	+	–
Extensive management	–	–	–	x	x	x	–	+	?
Unmanaged	x	x	x	x	x	x	–	+	?
Open water	–	–	–	–	–	–	x	x	x

Appendix A

Supporting information Chapter 7

An overview of carbon flows in the intensively managed peat area Oukoop. The dashed box represents the boundary of the total polder system, the dotted, small box on the right-hand of the figure represents the boundary of the considered system (from the field point-of-view).



Black arrows are C flows, thick dashed arrows are CH_4 flows and dashed arrows are CO_2 flows (autotrophic and heterotrophic respiration; $R_{CO2auto}$ and R_{CO2}, respectively and photosynthesis(GEP_{CO2})).

Samenvatting

Veengebieden bedekken ongeveer 3% van het terrestrische gedeelte van de aarde en tussen de 6% en 7% van Nederland (273.000 ha). In Nederland worden de meeste veengebieden veenweiden genoemd, omdat ze over het algemeen worden gebruikt voor grasproductie. Venen zijn de meest koolstofrijke ecosystemen van de terrestrische biosfeer en wereldwijd bevatten ze ongeveer 550 Gton koolstof. Deze grote koolstofvoorraad kan alleen blijven bestaan onder natte condities. Echter, wanneer veen wordt ontwaterd, bijvoorbeeld om het geschikt te maken voor landbouw, zal het veen oxideren en zullen grote hoeveelheden koolstof door oxidatie als koolstofdioxide (CO_2) de lucht ingaan met als gevolg dat het veen verdwijnt en de totale broeikasgasemissie mogelijk toeneemt. Momenteel is de oxidatie van venen een belangrijke en ook groeiende bron van broeikasgassen met antropogene oorzaak. Kleine veranderingen in het management van veengebieden kunnen leiden tot grote gevolgen voor de broeikasgasbalans en de koolstof balans. Verlies van veen en emissie van broeikasgassen vanuit veen kan mogelijk worden gereduceerd door het veen weer te vernatten en de management intensiteit te verlagen. Het is misschien zelfs mogelijk om emitterende veengebieden (sources) te veranderen in broeikasgas- en koolstofputten (sinks) omdat de CO_2 uitstoot waarschijnlijk zal afnemen. Kennis over hoe andere belangrijke broeikasgassen (methaan (CH_4) en lachgas (N_2O)) zullen reageren op vernatting van intensief gebruikte veenweiden ontbreekt echter grotendeels. Het vermoeden is dat met name de CH_4 emissies zullen toenemen.

Binnen het BSIK ME-1-project is een landschapsschaal(vergelijkings)experiment uitgevoerd, waarbij is onderzocht of gereduceerd management en/of vernatting kan leiden tot een afname van broeikasgasemissie en van koolstofverliezen uit veenweiden. Er zijn hiervoor gebieden geselecteerd die representatief zijn voor de westelijke Nederlandse veenweiden en die verschillen in managementintensiteit en in ontwateringsdiepte. In een aanvullend experiment is ook onderzocht hoe de broeikasgasemissie van waterlichamen (meren en drainagesloten in het veenweidelandschap) zich verhoudt tot de emissie vanuit het terrestrische deel van het veenweidelandschap.

De specifieke doelen van mijn onderzoek waren:

- Het kwantificeren van broeikasgasfluxen (CO_2, CH_4 en N_2O) op kleine schaal (m^2-schaal), veldschaal (ha-schaal) en landschapsschaal (polderschaal) in Nederlandse veenweidegebieden.
- Het bepalen van de ruimtelijke en temporele variabiliteit van fluxen op verschillende schaalniveaus.
- Het ontwikkelen van een systeem om fluxen gemeten op kleine schaal op te schalen naar landschapsschaal. Hiervoor werden emissievoorspellingen gedaan, gebaseerd op empirische regressiemodellen die sturende variabelen koppelen aan emissies.

- Het beantwoorden van de vraag of herstel van intensief beheerde veenweide ertoe kan leiden dat broeikasgasbronnen in broeikasgasputten worden veranderd als CO_2, CH_4 en N_2O worden meegenomen in de broeikasgasbalans.

Mijn onderzoek richt zich onder andere op het kwantificeren van de totale broeikasgasbalans van twee Nederlandse veenweidegebieden met een verschillend management: (1) een intensief beheerd veenweidegebied gebruikt voor melkveehouderij, waaraan dierlijke mest en kunstmest worden toegediend en waarin een intensief maairegime geldt (polder Oukoop) en (2) een veenweide met extensief management in de vorm van weidevogelbeheer, waar alleen gemaaid wordt en waar de grondwaterstand iets hoger is dan in de intensief beheerde polder (polder Stein).

De broeikasgasbalansen van deze twee veengebieden werden vergeleken met de broeikasgasbalans van een voormalig intensief gebruikt veenweidegebied dat sinds 15 jaar uit productie is genomen en waar de grondwaterstand 0-20 cm onder het maaiveld is. Dit laatste gebied is in detail bestudeerd door Hendriks (2009b). Alle broeikasgasfluxen (zowel opname als verlies) en mogelijk verklarende variabelen zijn gemeten in de drie experimentele gebieden. Overige koolstofstromen zijn bepaald aan de hand van gegevens uit de literatuur en volgens eenvoudige berekeningsmethoden op basis van boerderijgrootte.

Doordat omstandigheden in bodem, water, vegetatie en klimaat sterk wisselen, zowel in ruimte als in tijd, fluctueren ook de uitstoot en opname van broeikasgassen sterk. Fluxen van CO_2, CH_4 en N_2O zijn daarom moeilijk te meten. Gevolg is dat in de huidige emissieschattingen nog grote onzekerheden bestaan op regionale en lokale schaal. Ook is het door het gebrek aan inzicht in deze variabiliteit lastig om iets te kunnen zeggen over wat er gebeurt met de broeikasgasbalans als het landgebruik verandert. Er is tot nu toe slechts een beperkt begrip over de ordegrootte van emissie van de niet-CO_2-broeikasgassen. In dit onderzoek is een geïntegreerde benadering gevolgd waarbij de drie belangrijke broeikasgassen zijn meegenomen in de berekeningen en waarbij verschillende meettechnieken zijn gebruikt om de balansen op verschillende ruimtelijke en temporele schalen in kaart te brengen. De variabiliteit op kleine schaal is in detail bestudeerd door gebruik te maken van kamermetingen in velden (droge en natte gedeelten), drainagesloten en meren. Ruimtelijke variabiliteit op grote schaal (tussen ecosystemen) is bestudeerd door gebruik te maken van alle beschikbare meetdata van de drie sites en door de totale broeikasgasbalansen te vergelijken. Temporele variabiliteit op de schaal van jaren tot seizoenen is bepaald door gebruik te maken van meetkamers en op de schaal van dagen tot jaren door gebruik te maken van eddy-covariantiesystemen. De meetkamer meetmethode wordt meestal gebruikt voor het in kaart brengen van variatie op kleine schaal (aantal dm^2), terwijl de eddy-covariantiemeetmethode emissies integreert over een groot gebied (aantal ha). De eddy-covariantietechniek is de meestgebruikte techniek om vrijwel continue (per half uur) CO_2-fluxen te kwantificeren. De eddy-covariantiemeetmethode is recent ook

beschikbaar gekomen voor het meten van CH_4 en N_2O; de CH_4- en N_2O-fluxen die continu zijn gemeten in deze studie zijn gemeten met een nieuw systeem dat uitgebreid is geëvalueerd door Kroon (2010b).

De twee onafhankelijk meetmethoden zijn beide gebruikt om fluxen op te schalen, zowel in de ruimte als in de tijd. Omgevings variabelen (bijvoorbeeld temperatuur, waterstand, windsnelheid) die mogelijk (variatie in) broeikasgasemissies bepalen, zijn gemeten over een periode van ongeveer 4 jaar. De relaties tussen variabelen en emissies zijn bestudeerd door gebruik te maken van multiple lineaire regressie of niet-lineaire regressie. Temporele opschaling is gedaan door in de statistisch significante regressiemodellen de verklarende variabelen op halfuursbasis in te vullen. Voor ruimtelijke opschaling van fluxen in de veenweiden is gebruik gemaakt van kamerdata. De experimentele set-up is zo ontworpen dat alle landschapselementen die significant verschillen in bijdragen aan de broeikasgasbalans zijn bestudeerd. In deze landschapselementen zijn emissies en verklarende variabelen proportioneel bemonsterd en ook proportioneel opgeschaald. Omdat binnen het polderlandschap de sloten emissiehotspots voor CH_4 bleken te zijn is er een aanvullend experiment uitgevoerd waarin emissies van waterlichamen (meren en sloten) zijn bestudeerd in westelijke en oostelijke Nederlandse veenweidegebieden.

Het is belangrijk dat verschillende emissiemeettechnieken dezelfde uitkomsten genereren op verschillende schaalniveaus. Daarom is in dit onderzoek de vergelijkbaarheid van de meetkamertechniek en de eddy-covariantietechniek getest in een heterogeen veenweidelandschap. De vergelijkbaarheid van deze onafhankelijke meetmethoden is gecrossvalideerd door langetermijn-eddy-covariantie-CO_2-data (van één jaar) te vergelijken met binnen de footprint van de eddy-covariantiemasten opgeschaalde kamerdata in datzelfde jaar. Hetzelfde is gedaan over een periode van drie maanden om de vergelijkbaarheid van CH_4-metingen te bestuderen. De statistisch relevante regressiemodellen werden gebruikt om temporeel op te schalen. In het experiment waar metingen werden verricht op meren is een directe vergelijking gemaakt met kamermetingen binnen de footprint van het eddy-covariantiesysteem. Voor alle experimenten geldt: er is rekening gehouden met variabiliteit op kleine schaal door gebruik te maken van weegfactoren voor elk landschapselement in het bestudeerde landschap dat de broeikasgasbalans significant anders beïnvloedt. Om de totale broeikasgasbalans en de totale koolstofbalans in kaart te brengen werden in de geëxploiteerde gebieden ook de aan de boerderij gerelateerde emissies meegenomen. Het gaat om emissies afkomstig van mestopslagen rondom de boerderij en directe emissies van koeien. Deze fluxen werden niet gemeten, maar werden geschat door gebruik te maken van onderzoek uitgevoerd door anderen.

Hoofdstuk 1 en hoofdstuk 2 van dit proefschrift beschrijven de probleemstelling, de hypothese en de wetenschappelijke achtergrond van dit onderzoek. Hoofdstuk 3 beschrijft

de invloed van verminderd management en verhoging van de gemiddelde grondwaterstand op de CH_4-balans van twee veengebieden (intensief en extensief gemanaged) op basis van kamermetingen. Het experiment is zo opgezet dat de verschillende landschapselementen worden meegenomen in de berekening en dat ook potentieel sturende variabelen zijn gemeten. Zowel variabiliteit van emissies in de ruimte als de variatie in de tijd bleek groot te zijn. De hoogste CH_4-emissies werden gemeten in het voorjaar en in de zomer; de waarden waren gedeeltelijk gekoppeld aan mesttoediening en significant gerelateerd aan bodemtemperatuur. De drainagesloten en hun aangrenzende (vaak waterverzadigde) slootkanten bleken emissiehotspots te zijn in het veenweidelandschap. In een periode van één jaar komt in beide gebieden meer dan 60% van de CH_4-emissie uit de sloten en de slootkanten. Behalve bodemtemperatuur was er geen andere gemeten variabele die een significant effect toevoegde aan de niet-lineaire regressies. Beide gebieden waren CH_4-bronnen (range 146 tot 203 kg ha^{-1} jr^{-1} voor het intensief beheerde gebied en 156 tot 180 kg ha^{-1} jr^{-1} voor het extensief beheerde gebied) en er was qua brongrootte geen significant verschil tussen de twee gebieden. Als emissies gerelateerd aan de boerderij (bijvoorbeeld mestopslag, koegerelateerde emissies) werden meegenomen in de berekening nam de bronsterkte toe tot 427 kg ha^{-1} jr^{-1} voor de intensieve polder en tot 339 kg ha^{-1} jr^{-1} voor de extensieve polder.

Om van de twee onafhankelijke meetmethoden die zijn gebruikt in dit onderzoek de vergelijkbaarheid te testen is een cross-validatie-experiment uitgevoerd dat is beschreven in hoofdstuk 4. Hierbij zijn de data van twee verschillende eddy-covariantiesystemen (een voor CO_2 en een voor CH_4) vergeleken met de data van de kleineschaalmeetkamermethode. De kamer data zijn opgeschaald in het heterogene landschap (velden, sloten en slootkanten) binnen de footprint van de meetmasten. Hierbij zijn twee manieren van opschalen getest: wel of niet rekening houdend met de heterogeniteit van het landschap. De resultaten laten zien dat het bij de opschalingsprocedure heel belangrijk is om de emissies van de verschillende landschapselementen mee te nemen. Als alleen velden werden meegenomen in de opschaling was het verschil tussen de eddy-covariantiedata en de kamerdata 31% voor CO_2 (hogere fluxen voor de kamermeetmethode) en 55% voor CH_4 (hogere fluxen gemeten met het eddy-covariantiesysteem). Als proportioneel alle landschapselementen werden meegeteld in de berekening werden de verschillen in emissies gereduceerd tot respectievelijk 16.5% en 13% voor CO_2 en CH_4 met hogere emissies voor de meetkamermethode. Temporele variabiliteit op dagschaal werd niet goed meegenomen in de empirische kamergebaseerde regressiemodellen, omdat deze modellen alleen zijn gebaseerd op bodemtemperatuur. De resultaten laten zien dat het belangrijk is om een combinatie van beide methoden te gebruiken om zowel temporele als ruimtelijke variabiliteit goed in kaart te brengen en dat de vergelijkbaarheid van de onafhankelijke meetsystemen goed is, mits de kamermetingen op de goede manier zijn opgeschaald.

In hoofdstuk 5 zijn de terrestrische CH_4- en N_2O-balansen beschreven van de intensief beheerde polder Oukoop, zoals gemeten door een recent ontwikkeld eddy-covariantiesysteem. De onzekerheid in de schattingen van de CH_4- en N_2O-balansen is gereduceerd ten opzichte van de schattingen gebaseerd op kamermetingen. Over een periode van drie jaar is de gemiddelde CH_4-emissie geschat op 165 kg $ha^{-1}jr^{1}$ en de gemiddelde N_2O-emissie op 24 kg ha^{-1} jr^{-1}. De totale broeikasgasbalans van de intensief beheerde site is beschreven door gebruik te maken van gegevens over de CO_2-balans beschreven door Veenendaal et al (2007). De intensieve polder blijkt een belangrijke bron voor alle drie de broeikasgassen. Uitgedrukt in opwarmingspotentiëlen is het aandeel van CO_2, CH_4 en N_2O in de terrestrische broeikasgasbalans geschat op respectievelijk 30%, 25% en 45%.

Uit de resultaten van hoofdstuk 1 blijkt dat drainagesloten belangrijke hotspots zijn voor CH_4-emissie. Deze emissie uit waterlichamen wordt nog niet meegenomen in de nationale emissieschattingen. Daarom is er een kortdurend experiment uitgevoerd in de zomer van 2009 waarbij de CO_2- en CH_4- fluxen en hun mogelijk verklarende variabelen zijn gemeten in 12 waterlichamen (drainagesloten en meren) in het oostelijke en westelijke veenweidegebied. Dit experiment is beschreven in hoofdstuk 6. De resultaten laten zien dat zowel meren als sloten CO_2 en CH_4 emitteren en dat per oppervlakte-eenheid sloten significant meer emitteerden dan meren. De graad van eutrofie correleerde positief met de grootte van de CO_2- en CH_4-emissies, terwijl de diepte van het waterlichaam en pH van het water negatief correleerden met de grootte van de emissies. Dit zou betekenen dat een afname van de toevoer van nutriënten en organische stof vanuit de omgeving indirect ook de emissies van CH_4 en CO_2 naar beneden brengt.

Hoofdstuk 7 is een synthesehoofdstuk waarin alle data verzameld binnen het ME-1-project in de periode 2005-2009 door de verschillende groepen (ECN, VU, WUR) zijn samengevoegd. De hypothese dat intensief beheerde en bemeste veenweiden weer teruggebracht kunnen worden tot broeikasgas-sinks in plaats van broeikasgas-sources is getest. Drie veenweidepolders met vergelijkbare landgebruikshistorie (intensief beheer) zijn bestudeerd, elk van de polders had een eigen huidige gebruiksintensiteit en grondwaterstand: de gebruiksintensiteit variërt van intensief (polder Oukoop) tot extensief (polder Stein, voormalig intensief beheerde polder) tot geen management (polder Horstermeer, voormalig intensief beheerde polder). De grondwaterstand variërend van laag (polder Oukoop) tot dynamisch met hoge waterstanden in de winter en lage waterstanden in de zomer (polder Stein) tot hoog (polder Horstermeer). Beide beheerde gebieden fungeren (intensief en extensief) als bronnen voor CO_2 (beide een bronsterkte van 0.4 kg CO_2 m^{-2} jr^{-1}) terwijl de niet-beheerde polder Horstermeer fungeert als een opslag voor CO_2 (opnamesterkte 1.4 kg CO_2 m^{-2} jr^{-1}). De hoge temporele variatie in netto ecosysteemuitwisseling van CO_2 is over het algemeen een gevolg van beheer (maaien en afvoer van biomassa). De drie polders fungeren als CH_4-bronnen met emissies van 16.4,

16.7 en 19.2 g CH_4 m^{-2} jr^{-1}, gemiddeld over drie jaar gemeten. Ruimtelijke verschillen op grote schaal ontstaan door een combinatie van beheer en waterstand. Het aandeel van CH_4 in de totale broeikasgasbalans is daarbij sterk afhankelijk van de hoeveelheid open water in het bestudeerde gebied. Een verandering in het aandeel water in het landschap impliceert dan ook een sterke verandering in het aandeel CH_4 in de broeikasgasbalans. Voor de totale koolstofbalans geldt dat beide beheerde gebieden koolstof verliezen op jaarbasis en het onbeheerde gebied koolstof opslaat. Voor het terrestrische gedeelte van het veenweidegebied blijkt dat N_2O het belangrijkste broeikasgas is (uitgedrukt in potentiële opwarmingssterkte) in het intensief beheerde veenweidegebied Oukoop, terwijl in Stein CO_2 en CH_4 de belangrijkste twee broeikasgassen zijn. In de Horstermeer domineert de opname van CO_2 de totale broeikasgasbalans. Ongeveer 30% van de temporele variabiliteit in N_2O-emissie in het intensief beheerde gebied wordt verklaard door de variabelen bodemtemperatuur en windsnelheid. Als we de totale broeikasgasbalansen (uitgedrukt in CO_2-equivalenten) van de drie bestudeerde gebieden vergelijken kunnen we concluderen dan het intensief beheerde gebied de grootste broeikasgasbron is. Een intensief beheerd gebied kan echter weer teruggebracht worden (binnen 15 jaar) naar een gebied met broeikasgasopname als de juiste maatregelen worden genomen (verhoging grondwaterstand en reductie van de beheersintensiteit). Het precieze tijdsbestek waarin een broeikasgas-source kan switchen naar een broeikasgas-sink is nog onbekend. Ook is onbekend hoe een gebied als de Horstermeer zich zal ontwikkelen in de toekomst: zal het eutroof blijven omdat er een continue toevoer van nutriënten is vanuit de omgeving? En zal het zich dus ontwikkelen naar bijvoorbeeld een elzenbroekbos? Of zal het systeem langzaam voedselarmer worden en zal het zich ontwikkelen naar een hoogveen? Het is daarom ook onbekend of de huidige broeikasgasopname in dit gebied in dezelfde orde van grootte zal blijven.

Afsluitend zijn in hoofdstuk 8 conclusies besproken en zijn toekomstperspectieven gegeven voor het veenweidegebied, gebaseerd op de bevindingen die beschreven zijn in dit proefschrift. Er is besproken waar nog kennis ontbreekt en waar verder onderzoek naar gedaan zou moeten worden. In een overzichtstabel is per broeikasgas weergegeven wat het gevolg is van verschillende mitigatiestrategieën.

Hendriks (2009), thesis: Integrated observations of greenhouse gas budgets at the ecosystem level – changing environment and management practives in peat meadows.

Kroon (2010), thesis: Eddy covariance observations of methane and nitrous oxide emissions - towards more accurate estimates from ecosystems.

Veenendaal, E.M., Kolle, O., Leffelaar, P.S. Schrier-Uijl, A.P., Van Huissteden, J., Van Walsem, J., Möller, F. & Berends, F. (2007): CO_2 exchange and the carbon balance in two grassland sites on eutrophic drained peat soils. Biogeosciences, 4, pp. 1027-1040.

About the author

Curriculum Vitae

Arina Uijl was born in Poortlviet on November 4th, 1974. After she completed secondary school (VWO) in 1993 at the Voetius Scholengemeenschap in Goes, she started a study as X-ray technician at the Fontys Hogeschool Eindhoven. In the mean time she worked as X-ray technician in Lievensberg Hospital in Bergen op Zoom three years as a student and one year as a graduated X-ray technician. In 1997 she started the study 'Soil, Hydrology and Meteorology' at the Wageningen University and for 1.5 day per week she worked as a X-ray technician in Hospital Gelderse Vallei from 1998 untill 2002. For her first MSc thesis she studied the effect of soil biota on the formation of micro- and macroaggregates in soils under different management. For her second MSc thesis and practical period she moved to Canada and worked for 13 months at the University of Saskatchewan in Saskatoon on the carbon distribution and sediment redistribution in pothole landscapes. After her graduation in 2003 she worked on various projects and in 2003 she started working at the Miliedienst Zuid Oost Utrecht in Zeist. In September 2005, she started her PhD for 4 days a week at the chair group 'Nature Conservation and Plant Ecology' and worked on greenhouse gas emissions from fen meadow landscapes. Next to doing science, Arina is active in sports, is an enthousiastic photographer and is mother of two children.

Journal publications

Uyl, A., Didden, W., Marinissen, J. (2002). Earthworm activity and decomposition of C-14-labelled grass root systems. Biology and Fertility of Soil, 36, pp. 447-455.

Pulleman, M.M., Six, J., Uyl, A, et al. (2005). Earthworms and management affect organic matter incorporation and microaggregate formation in agricultural soils. Applied Soil Ecology 29, 1, pp. 1-15.

Bedard-Haughn, A., Jongbloed, F., Akkerman, J., Uijl, A., et al. (2006). The effects of erosional and management history on soil organic carbon stores in ephemeral wetlands of hummocky agricultural landscapes. Geoderma 135, pp. 296-306.

Veenendaal, E.M., Kolle, O., Leffelaar, P.S. Schrier-Uijl, A.P., Van Huissteden, J., Van Walsem, J., Möller, F. & Berends, F., (2007). CO_2 exchange and carbon balance in two grassland sites on eutrophic drained peat soils. Biogeosciences, 4, pp. 1027-1040.

Jacobs, C.M.J., Jacobs, F.C., Bosveld, F.C., Hendriks, D.M.D., Hensen, A., Kroon, P.S., Moors, E.M., Nol, L., Schrier-Uijl, A.P. et al. (2007). Variability of annual CO_2 exchange from Dutch grasslands. Biogeosciences, 4, pp. 803–816.

Schrier-Uijl, A.P., Veenendaal, E.M., Leffelaar, P.A., van Huissteden, J.C., Berendse, F. (2010a). Methane emissions in two drained peat agro-ecosystems with high and low agricultural intensity. Plant Soil, doi:10.1007/s11104-009-0180-1.

Schrier-Uijl, A.P., Kroon, P.S., Hensen, A., Leffelaar, P.A., Berendse, F. & Veenendaal, E.M. (2010b). Comparison of chamber and eddy covariance based CO_2 and CH_4 emission estimates in a heterogeneous grass ecosystem on peat, Agric. For. Meteorol., doi:10.1016/j.agrformet.2009.11.007.

Schrier-Uijl, A.P., Veraart, A.J., Leffelaar, P.A., Berendse, F., Veenendaal, E.M (2010c). Release of CO_2 and CH_4 from lakes and drainage ditches in temperate wetlands. Biogeochemistry, doi:10.1007/s10533-010-9440-7.

Schrier-Uijl, A.P., (2010d, in prep): Agricultural peatlands; towards a greenhouse gas sink.

Kroon, P.S., Schrier-Uijl, A.P., Hensen, A., Veenendaal., E.M., Jonker, H.J.J., (2010). Annual balances of CH_4 and N_2O from a managed fen meadow using eddy covariance flux measurements. Eur. J. Soil Sci.

Stolk, P.C., Hendriks, R.F.A., Jacobs, C.M.J., Moors, E.J., van Groenigen, J.W., Kroon, P.S., Schrier-Uijl, A.P., et al. (submitted). Simulation of daily N_2O emissions from managed peat soils.

Other publications

A.P. Schrier-Uijl, P.S. Kroon, D.M.D. Hendriks, P. A. Leffelaar, F. Berendse and E.M. Veenendaal (2009): How the methane balance changes if agricultural peatlands are transformed into wetland nature and how this transformation influences the total carbon balance – contribution to Cost Action ES0804. In: Water in a Changing Climate, 6th international Scientific Conference on the Global Energy and Water Cycle and 2nd Integrated Land Ecocystem – Atmosphere Processes Study (iLEAPS) Science Conference. Australia, Melbourne.

Kroon, P.S., Schrier-Uijl, A.P., Stolk, P.C., van Evert, F.K., Kuikman, P.J., Hensen, A.H., Veenendaal, E.M: Kunnen we sturen op landgebonden broeikasgas emissies? naar een klimaat neutrale(re) inrichting van het landelijk gebied, Landschap 27/2: 99-109.

Schulp, C.J.E., Jacobs, C.M.J., Duyzer, J.H., van Beek, C.L., Bosveld, F.C., Dias, A., Jans, W.W.P., Schrier-Uijl, A.P., Vermaat, J.E.: Variabiliteit in ruimte en tijd ontrafeld. Broeikasgasemissies uit Nederlandse landschappen. Landschap 27/2: 67-79.

Contributions in: 'Waarheen met het Veen', ISBN 9789077824108 by Martin Woestenburg.

Conference proceedings

Schrier-Uijl, A.P, E.M. Veenendaal, P.A. Leffelaar, 2005: Greenhouse gas balances of Dutch fen meadows and their management potential for emission reduction. 3rd Annual CarboEurope-IP Project Meeting on Assessment of the European Terrestrial Carbon Balance, 14-18 November 2005, Levi, Finland.

Schrier, A, 2006: Controlling factors of GHG emissions in fen meadow ecosystems and the possibility of using water level manipulation as a tool for emission reduction. Netherlands Scientific Symposium on Soil and Water, 14-15 June 2006, Zeist, Netherlands.

Kroon, P.S, A.P. Schrier-Uijl, A.P, D. Hendriks, 2006: Inter-comparison of chamber and eddy-correlation flux measurements. Open Science Conference on the GHG cycle in the Northern Hemisphere, November 14-18 2006, Sissi-Lassithi, Crete, Greece.

Veenendaal, E, Schrier, A, Leffelaar, P, van Huissteden, k, Berendse, F, 2006. Land use dependent GHG exchange and the Carbon balance in two grassland sites on eutrophic drained peat soils. Open science conference on the GHG cycle in the Northern Hemisphere, November 14-18 2006, Sissi-Lassithi, Crete, Greece.

Schrier-Uijl, A.P, P.S. Kroon, D. Hendriks, E.M. Veenendaal, P.A. Leffelaar, F. Berendse, 2007: The full Greenhouse Gas Balance and Spatial Variability in two peat meadows in the Netherlands. International symposium on Carbon in Peatlands, 15-18 April 2007, Wageningen, Netherlands.

Hendriks, D. M. D, A. Schrier, P. S. Kroon, 2007: The effects of vegetation and soil on methane emissions in a natural fen meadow in the Netherlands. International symposium on Carbon in Peatlands, 15-18 April 2007, Wageningen, Netherlands.

Kroon, P. S, D.M D. Hendriks, A. Schrier, A. Hensen, W. H. Van 't Veen, 2007: Micrometeorological observations of CH_4 and N_2O at a managed fen meadow in the Netherlands. International symposium on Carbon in Peatlands, 15-18 April 2007, Wageningen, Netherlands.

Schrier-Uijl, A.P and D. Hendriks, 2007b: carbon dioxide and methane in Dutch Peat Meadows, 2007. 5th CarboEurope-IP Integrated Project Meeting on Wetlands, 7-12 October, 2007, Poznań, Poland.

Schrier-Uijl, A.P, P. Leffelaar, F. Berendse, E. Veenendaal, 2008a: Methane emissions in fen meadows in the Netherlands. Necov Winter Symposium, 7-8 February 2008, Antwerpen, Belgium.

Schrier-Uijl, A. P, P. S. Kroon, E. M. Veenendaal, P. A. Leffelaar, F. Berendse, 2008: Comparison of chamber based flux estimates and fluxes measured by EC for CO_2 and CH_4 in a managed peat meadow in the Netherlands. 6th CarboEurope-IP Integrated Project Meeting, 29 September - 2 October 2008, Jena, Germany.

Veenendaal, E, A. Schrier-Uijl, D. Hendriks, K. van Huissteden, P. Kroon, P. Leffelaar, A. Hensen, H. Dolman, F. Berendse, 2008. Carbon balance and greenhouse gas fluxes in intensive and extensive managed grasslands on peat. Netherlands Ecological Research Network Annual Meeting, 12-13 February 2008, Lunteren, Netherlands.

Schrier-Uijl, A.P, 2009. Spatial- and seasonal variability of greenhouse gases in two managed peat meadows in the Netherlands and the comparison of small scale and large

scale measurement techniques Netherlands. Netherlands Ecological Research Network Annual Meeting 2009, 10-11 February 2009, Lunteren, Netherlands.

Schrier-Uijl, A., A. Hensen, P. Kroon, E. Veenendaal, 2009: Greenhouse gas emissions: spatial and seasonal variability in two managed peat meadows in the Netherlands and the comparison of small scale and large scale measurement techniques. 5th International Symposium on non-CO_2 greenhouse gasses, Reduction Policy and Implementation, June 30-July 3 2009, Wageningen, Netherlands.

Schrier-Uijl, A. P., P. S. Kroon, D. M. D. Hendriks, E. M. Veenendaal, P. A. Leffelaar, F. Berendse, 2009: How the methane balance changes if agricultural peatlands are transformed into wetland nature and how this transformation influences the total carbon balance. GEWEX/iLEAPS young scientists workshop, 20-22 August, 2009, Melbourne, Australia.

Kroon, P. S, A. Hensen, H. J. J. Jonker, A. P. Schrier-Uijl, M. J. Tummers, F. C. Bosveld, 2009: Micrometeorolgical observations of CH_4 and N_2O at a managed fen meadow in the Netherlands. GEWEX/iLEAPS young scientists workshop, 20-22 August, 2009, Melbourne, Australia.

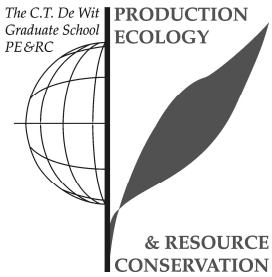
Schrier-Uijl, A.P., P.S. Kroon, D. Hendriks, P. A. Leffelaar, F. Berendse and E.M. Veenendaal, 2009. How the methane balance changes if agricultural peatlands are transformed into wetland nature and how this transformation influences the total carbon balance – contribution to Cost Action ES0804, 6th International Scientific Conference on the Global Energy and Water Cycle and the 2nd Integrated Land Ecosystem-Atmosphere Study (iLEAPS), 22-28 August 2009, Melbourne, Australia.

Kroon, P. S., A. Hensen, H. J. J. Jonker, A. P. Schrier-Uijl, M. J. Tummers, F. C. Bosveld, 2009: Micrometeorological observations of CH_4 and N_2O at a managed fen meadow in the Netherlands. 6th International Scientific Conference on the Global Energy and Water Cycle and the 2nd Integrated Land Ecosystem-Atmosphere Study (iLEAPS), 22-28 August 2009, Melbourne, Australia.

Schrier-Uijl, A.P, 2010. Release of CH_4 and CO_2 from lakes and ditches in temperate wetlands. Netherlands Ecological Research Network Annual Meeting 2010, 9-10 February 2010, Lunteren, Netherlands.

Schrier-Uijl, A.P **(invited expert**), 2010. Towards a total GHG balance for agricultural peatlands in the Netherlands; calculation methodology and upscaling. Expert workshop on Establishment of emission factors for Danish organic soils – SINKS project 7 Program, 4-5 March 2010, Kolding, Denmark.

PE&RC PhD Education Certificate



With the educational activities listed below the PhD candidate has complied with the educational requirements set by the C.T. de Wit Graduate School for Production Ecology and Resource Conservation (PE&RC) which comprises of a minimum total of 32 ECTS (= 22 weeks of activities)

*Review of literature (**5 ECTS**)*

- Integrated observations and modelling of greenhouse gas budgets at the ecosystem level in the Netherlands-controlling factors of greenhouse gas emission in fen meadow ecosystems and the possibility of manipulation of water level and management as a tool for emission reduction – a review (2005) (this literature review is included in the project proposal); presented at the ME-1 project meeting, Amsterdam, the Netherlands (2006) and at the CarboEurope-IP Project Meeting, Levi, Finland (2005)

*Writing of project proposal (**4.5 ECTS**)*

- Integrated observations and modelling of greenhouse gas budgets at the ecosystem level in the Netherlands-controlling factors of greenhouse gas emission in fen meadow ecosystem and the possibility of manipulation of water level and management as a tool for emission reduction (including review of literature)

*Post-graduate courses (**6.5 ECTS**)*

- Art of modelling (2006)
- Crossing the frontier between below-and above-ground; Soil Ecology (2007)
- Advanced statistics (2007/2008)

*Deficiency, refresh, brush-up courses (**4.9 ECTS**)*

- Basic statistics (2007)
- In-house training PEATLAND model; VU (2007)
- Workshop: statistical methods for spatial data analysis and modelling; BSIK, Arina Schrier; Wageningen (2007)
- Climate changes spatial planning; Ecohydrology, funded by Business Centre for Hydrology and the Bsik program (2007)

*Competence strengthening / skills courses (**1.2 ECTS**)*

- Scientific writing (2007)

*PE&RC Annual meetings, seminars and the PE&RC weekend (**2.3 ECTS**)*

- Environmental research in context; PE&RC (2006)
- PE&RC 10th Anniversary (2007)

*Discussion groups / local seminars / other scientific meetings (**11.3 ECTS**)*

- Netherland Scientific Symposium on Soil and Water; Zeist (2006)
- International symposium on Carbon in Peatlands; Wageningen (2007)
- PE&RC Discussion group Climate change & Soil-water-atmosphere Interactions (CSI); Wageningen (2007-2010)
- Netherlands Ecological Research Network Annual Meeting; Lunteren (2008/2009/2010)
- Symposium on non-CO_2 greenhouse gases, Reduction Policy and Implementation; Wageningen (2009)

*International symposia, workshops and conferences (**15 ECTS**)*

- 3rd Annual CarboEurope-IP Project Meeting on Assessment of the European Terrestrial Carbon Balance; Levi, Finland (2005)
- Open Science Conference on the GHG cycle in the Northern Hemispere; Sissi-Lassithi, Crete, Greece (2006)
- 5th CarboEurope-IP Integrated Project Meeting on Wetlands; Poznán, Poland (2007)
- Necov Winter Symposium; Antwerpen, Belgium (2008)
- 6th CarboEurope-IP Integrate Meeting; Jena, Germany (2008)
- GEWEX/iLEAPS Young scientists workshop; Melbourne, Australia (2009)
- Contribution to Cost Action ES0804, 6th International Scientific Conference on the Global Energy and Water Cycle and the 2nd Integrated Land Ecosystem-Atmosphere Study (iLEAPS); Melbourne, Australia (2009)
- Expert workshop on Establishment of emission factors for Danish organic soils-SINKS project 7 Program; Kolding, Denmark (2010)

*Lecturing /supervision of practical's / tutorials (**9 ECTS**)*

- Vegetation Science and System Ecology; 6 days (2006)
- Ecology II; 4 days (2006/2007)
- Ecology I; 12 days (2006/200720/08)
- Trends; 2 days (2007/2008)
- Bos en natuurbeheer II; 6 days (2008)

Supervision of (MSc) students (30 days)

- Carbon dioxide and methane fluxes from peatlands: modelling the effects of water table management on emissions; Clif R. Sabajo (2009/2010)
- The main controlling factors of greenhouse gas emission from Dutch fen meadow ecosystems; Larenstein students: Du Bing Zhen and Wang Yafei (2007)
- The influence of temperature, water table depth and agricultural practices on the emissions of CO_2, CH_4 and N_2O; Larenstein student Liu Fei (2008)