Greenhouse Gas Emissions from Rewetted Extracted Peatlands in Sweden

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Cover: Aerial photograph of the Västkärr peatland (constructed lake and peat extraction area) in the lagg area of Skagerhultamossen
(Photo: Hasselfors Garden)

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Abstract

Peat extraction for horticultural purposes and energy production has a long tradition in Northern Europe. Related drainage activities directly affect the greenhouse gas (GHG) balance due to oxidative peat decomposition and denitrification, with concomitant emissions of carbon dioxide (CO_2) and nitrous oxide (N_2O). Rewetting, i.e. raising of the water table, is one after-use and restoration objective. Rewetting transforms an extracted peatland with aerobic soil conditions into a wetland with prevailing anaerobic conditions and can thus create suitable conditions for peat-forming plants, which could restore carbon (C) storage functions. Studies on GHG emissions from extracted peatlands after rewetting are limited. In general, peatland rewetting decreases emissions of CO_2 and N_2O, while methane (CH_4) emissions may increase. More data are needed on GHG emissions from extracted peatlands over longer periods after rewetting and from corresponding studies about constructed water bodies and their vegetated littoral zones, which have been identified as high CH_4 emitters.

Studies were investigated how typical peatland ecotopes and vegetation communities created after rewetting affected CO_2, CH_4 and N_2O emissions. Specific objectives were to determine the relationships between GHG emission fluxes and water conditions, soil/water temperature and vegetation cover. GHG emission fluxes were measured in two rewetted extracted peatlands in Sweden using manual opaque and automated transparent chambers for discontinuous and continuous measurements, respectively.

The overall climate impact of CH_4 emissions from the study areas did in general not exceed the impact of soil and plant respiration and neither the net CO_2 flux during summer. But, GHG emissions could vary between years and sites can shift from sinks to sources. In regards to management of extracted peatlands, the construction of shallow lakes showed great potential for lowering GHG fluxes to the atmosphere.

With continuous measurements a derivation of long-term gas balances can be achieved and short-term changes in environmental conditions influencing GHG exchanges can be detected more effectively as with discontinuous measurements, such as by vial sampling. But still, a correct indication of all GHG fluxes, e.g. for GHG upscaling purposes or national emission inventories, is strongly based on the correct estimation of all C fluxes including ebullition.

Keywords: ebullition, greenhouse gases, mire restoration, peat chemistry, peatland

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Dedication

Für meine Familie.

Für Silke.

Aus der Mühle schaut der Müller,
Der so gerne mahlen will.
Stiller wird der Wind und stiller,
Und die Mühle stehet still.

So geht's immer, wie ich finde,
Rief der Müller voller Zorn.
Hat man Korn, so fehlt's am Winde,
Hat man Wind, so fehlt das Korn.

Wilhelm Busch

Contents

List of Publications

This thesis is based on the work contained in the following papers, referred to by Roman numerals in the text:

I Lars Lundin, Torbjörn Nilsson, Sabine Jordan, Elve Lode, Monika Strömgren (2016). Impacts of rewetting on peat, hydrology and water chemical composition over 15 years in two finished peat extraction areas in Sweden (submitted manuscript).

II Sabine Jordan, Monika Strömgren, Jan Fiedler, Lars Lundin, Elve Lode & Torbjörn Nilsson (2016). Ecosystem respiration, methane and nitrous oxide fluxes from ecotopes in a rewetted extracted peatland in Sweden. *Mires and Peat* 17(7), 1-23.

III Sabine Jordan, Monika Strömgren, Elve Lode, Jan Fiedler, Lars Lundin & Torbjörn Nilsson (2016). Ecosystem respiration, methane and nitrous oxide emission fluxes along water level gradients in the littoral zones of constructed water bodies in a rewetted extracted peatland in Sweden (manuscript).

IV Sabine Jordan, Monika Strömgren, Jan Fiedler, Kristina Mjöfors, Elve Lode, Lars Lundin, Torbjörn Nilsson. Continuous carbon dioxide and methane flux measurements from the shore of a shallow lake in a rewetted extracted peatland in Sweden (manuscript).

Paper II is reproduced with the permission of the publisher.

The contribution of Sabine Jordan to the papers included in this thesis was as follows:

I Co-author. Contribution to field work; contribution to data analysis; contribution to writing of the paper.

II Main author. Field work with assistance from all co-authors; data analyses with assistance from the second and third author; writing of the paper with assistance from the co-authors.

III Main author. Field work with assistance from all co-authors; data analyses with assistance from the second, third and fourth author; writing of the paper with assistance from the co-authors.

IV Main author. Planning of the study together with the second author; field work together with the second and third author; contribution to data analyses; visual data inspection with assistance from the third author; writing of the paper with assistance from the co-authors.

Abbreviations

Ca_{tot}	Total calcium
C_{tot}	Total carbon
DOC	Dissolved organic carbon
$_{(g)}$	Gaseous
GHG	Greenhouse gas, greenhouse gases
IRGA	Infra-red gas analyser
K_{tot}	Total potassium
Mg_{tot}	Total magnesium
NH_4-N	Nitrogen bound in ammonium
NO_3-N	Nitrogen bound in nitrate
N_{tot}	Total nitrogen
P_{tot}	Total phosphorus
PVC	Polyvinylchloride
PO_4-P	Phosphorus bound in ortho-phosphate
S_{tot}	Total sulphur

1 Introduction

Peatlands are the most widespread of all wetland types globally. They account for approximately 3 % of the earth's land and freshwater surface (Succow & Joosten 2001, Joosten & Clarke 2002, Franzén 2006) but contain up to 30 % of the global soil carbon (C) due to sedentary peat accumulation (Bragazza *et al.* 2006, Limpens *et al.* 2008). To be defined as peat a soil has to consist of at least 30 % (dry mass) of dead plant (organic) material (Göttlich 1990).

Boreal and subarctic peatlands cover considerable areas in the northern hemisphere (Montanarella *et al.* 2006). For example, Sweden's total area of peatlands is about 10^6 ha (Päivänen & Hånell 2012), resulting in 95×10^{12} m^3 of peat resources (excluding protected areas). Of that amount, approximately 8×10^{12} m^3 of peat are potentially available for extraction because those peatlands are already drained (Hånell *et al.* 2008).

Peat extraction for horticultural purposes and energy production has a long tradition in Northern Europe and typically can take place in a peatland over a 20-year period. In Sweden, annual extraction for energy and horticultural peat was amounted to 2-3 $\times 10^6$ m^3 and 1-2 $\times 10^6$ m^3, respectively, between 1992 and 2012 (Päivänen & Hånell 2012).

When peat extraction ceases, one of several post-use alternatives of the area is rewetting (Quinty & Rochefort 1997, 2003; Blankenburg & Tonnis (eds.) 2004), i.e. raising the water level, in order to mitigate climate change, water pollution and loss of biodiversity (Hiraishi *et al. (eds.)* 2014). Rewetting transforms an extracted peatland with aerobic soil conditions into a wetland with prevailing anaerobic conditions and can thus create suitable conditions for peat-forming plants, which could restore C storage functions and biodiversity (Joosten & Clarke 2002, Rochefort *et al.* 2003, Strack & Zuback 2013) in a man-made landscape.

The changes in soil water conditions, and thus soil chemistry and vegetation development after rewetting, affect the greenhouse gas (GHG) balance. In

general, peatland restoration by rewetting decreases the CO_2 and N_2O fluxes to the atmosphere, while CH_4 fluxes to the atmosphere may increase (e.g. Couwenberg 2009, Wilson *et al.* 2016b). Some studies show that nutrient-poor peatlands could turn into GHG net sinks, but that nutrient-rich peatlands could still be net GHG emitters (Silvan *et al.* 2005, Glatzel *et al.* 2008, Höper *et al.* 2008), this means that rewetting does not necessarily result in lower GHG fluxes to the atmosphere (Joosten & Clarke 2002).

Numerous studies on various peatland types in the Nordic countries have characterised GHG fluxes to the atmosphere (Maljanen *et al.* 2010), including emissions from rewetted peatlands after peat extraction (e.g. Tuittila *et al.* 2000, Wilson *et al.* 2013 & 2016a, Strack *et al.* 2016). However, there has only been a limited amount of research on GHG fluxes from different ecotopes and vegetation communities such as constructed lakes, non-vegetated bare peat, newly established *Sphagnum* spp. and vegetated shore and littoral zones created by rewetting and in different time scales post-rewetting.

2 Objectives and structure of the thesis

In this thesis, studies were conducted in which GHG emission [1] fluxes (ISO 4225; IUPAC 2006) were measured in two rewetted extracted peatlands in Sweden. This was done using opaque manual chambers (CO_2, CH_4, N_2O; Papers II & III) and transparent automated chambers (CO_2, CH_4; Paper IV).

The main aim of the thesis was to investigate the effects of rewetting on GHG emissions. A secondary aim was to determine the impacts of rewetting impacts on hydrology and on soil and water chemistry in extracted peatlands. Therefore, studies on peat, hydrology and water chemical composition before and after rewetting were combined in Paper I. The resulting data on the physical geography of the study sites set the framework for this thesis (Figure 1).

Subsequent studies involving GHG measurements (Papers II-IV) investigated how typical peatland ecotopes and vegetation communities created after rewetting affected CO_2, CH_4 and N_2O emissions. Specific objectives were to determine the relationships between GHG emission fluxes and water conditions, soil/water temperature and vegetation cover, by:

- Quantifying CO_2, CH_4 and N_2O fluxes to the atmosphere in various ecotopes of a nutrient-poor peatland, such as vegetation-covered, bare peat and open water, and comparing these fluxes against those from a nearby open poor fen not subject to peat extraction (Paper II)
- Investigating and quantifying GHG emission fluxes along water level gradients from littoral zones of constructed shallow lakes with wetland and non-wetland type vegetation zones in a nutrient-rich peatland, in order to determine whether CO_2, CH_4 and N_2O emission fluxes differ from the

1. Emission in atmospheric chemistry is defined as discharge of substances into the atmosphere (ISO 4225, IUPAC 2006). Net emission expresses the balance of emission and uptake (deposition, photosynthesis).

rather dry vegetated shore with *Graminoids*, over the littoral zone with *Carex* spp. and *Typha latifolia* to the open water without vegetation; and how these fluxes change following short-term inundation events and seasonal droughts (Paper III)

- Investigating net CO_2 and CO_4 flux dynamics during summer and detecting short-term events such as ebullition (non-diffusive emissions) (Paper IV).

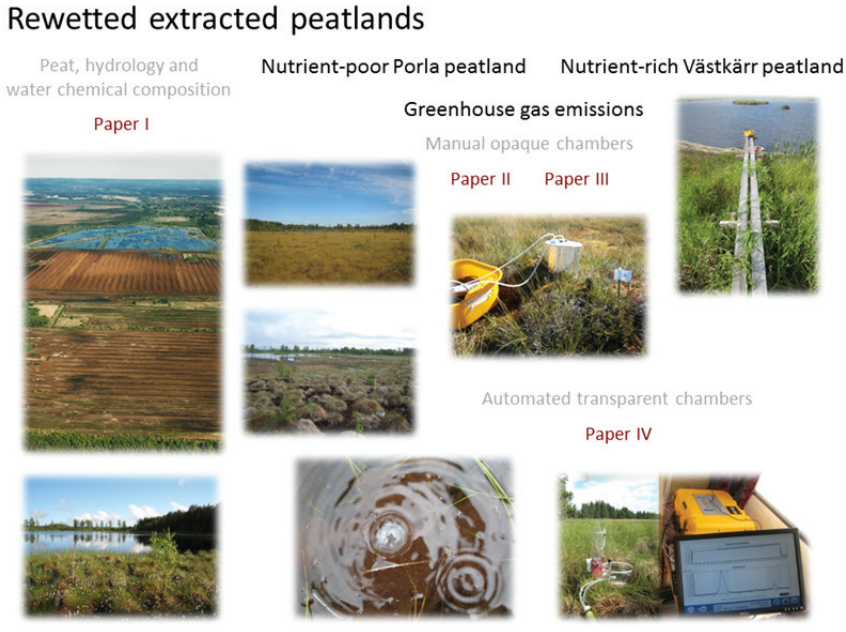


Figure 1. Illustration of the different study sites and measurement techniques included in the thesis (Photos: Sabine Jordan).

3 Background

3.1 Undisturbed mires

The requirements for a peat accumulation process on a site are peat-forming vegetation, such as *Sphagnum* spp., *Phragmites* spp. or *Carex* spp., a surplus of water, meaning a positive water balance, and a peat accumulation rate that outweighs peat decomposition, meaning a positive mass balance. A contributing factor in peat accumulation is low ambient temperature, where the speed of most physical, chemical and biological processes slows and there is no or very low activity of the microorganisms responsible for decomposition of plant material. In general, in reducing conditions (anoxic environment) peatlands act as accumulators, buffers, transformers and filters for nutrients and water. They are thus substance sinks in the landscape (Nykänen *et al.* 1995) and help in mitigating climate warming (Joosten & Clarke 2002, Strack (ed.) 2008) through the sequestration of atmospheric CO_2 as C in the peat matrix.

3.1.1 Carbon dioxide

The carbon cycling processes, pools and fluxes in sediments, wetlands and peatlands are outlined in many publications (Gorham 1991, Houghton 1997, Mitsch & Gosselink 2000, Blodau 2002, Whalen 2005, Blodau *et al.* 2007, Jahn *et al.* 2010).

Since photosynthesis and aerobic respiration take place in the aerobic soil horizons, those two processes are effective in terms of energy transfer between soil, water and atmosphere and thus in build-up and degradation of organic matter. Soil respiration can be divided into autotrophic respiration (by plants) and heterotrophic respiration (by microbes, microfauna and fungi). The latter is responsible for decomposition of soil organic matter and the conversion of soil C into CO_2 (Mäkilä & Goslar 2008).

3.1.2 Methane

In wetlands and in anaerobic horizons of peatlands, the degradation of organic matter is performed by anaerobic respiration by microbes in the form of fermentation and methanogenesis. Fermentation, also called glycolysis, can be carried out by either facultative or obligate anaerobes. Alcohols (e.g. ethanol), low-molecular-weight acids (such as lactic acid and acetate), DOC, CO_2 and H_2 are fermentation end products (Ferry 1992).

Methanogenesis, which is carried out by methanogenic *archaea* (methanogens), is the final step in the degradation of organic matter, after nitrogen (N), manganese (Mn), iron (Fe) and sulphur (S) have been reduced. To convert the organic matter itself into low-molecular-weight organic compounds and CO_2 into CH_4, a redox potential below -200 mV is required. Peatlands in natural conditions (mire) are characterised by a subsurface, anaerobic zone of CH_4 production by methanogenic bacteria and a superficial, aerobic zone of CH_4 oxidation by methanotrophic bacteria (Whalen 2005).

Acetic acid (acetate = derivate of acetic acid) is very common in the environment and can undergo a dismutation reaction to produce CH_4 and CO_2 simultaneously. In many studies (Whiticar *et al*. 1986, Ferry 1992, Conrad 1999, Keller & Bridgham 2007), the general claim is made that most CH_4 originates from the methyl group of acetate and only a smaller pool originates from the reduction of CO_2 with H_2. The process of aceticlastic methanogenesis, *i.e.* the conversion of acetate to CH_4 and CO_2, has not been studied as extensively as the reduction of CO_2 to CH_4 (autotrophic methanogenesis) (Ferry 1992, Keller & Bridgham 2007). Some studies report that aceticlastic methanogenesis is more important in fens (Keller & Bridgham 2007, Galand *et al.* 2010) whereas autotrophic methanogenesis is dominant in bogs (Duddleston *et al.* 2002, Horn *et al.* 2003). Keller & Bridgham (2007) attributed the dominance of aceticlastic methanogenesis during the growing season to high acetate concentrations due to the presence of root exudates, whereas the proportion of CH_4 production that originates from CO_2 reduction increases with depth as labile root exudates become less available. Liberation of CH_4 can occur from saturated peat horizons via diffusion, gas bubbles (ebullition) and by plant-mediated transport in vascular plants such as *Phragmites* or *Carex* (Bastviken *et al.* 2004, Lai 2009).

3.1.3 Nitrogen

Carbon mineralisation rate may be influenced by N concentrations in the litter of *Sphagnum* spp. (Blodau 2002). The main N inputs to peatland are deposition from atmosphere, N_2 fixation by bacteria and algae and N inflow through upland runoff or discharge. The main N pool in peatlands is in the peat material

itself, with a minor proportion stored in living organisms. Other pools are represented by vascular plants, *Sphagnum* spp. and other mosses, microbes and pore water (Limpens *et al.* 2006). The N content of the organic matter and its degradability strongly affect the availability of inorganic N for N_2O formation via nitrification and denitrification and thus the extent of N_2O emissions (von Arnold *et al.* 2005, Glatzel *et al.* 2008). *Sphagnum* spp. litter becomes more easily decomposed when N deposition and N content increase (Limpens & Berendse 2003, Bragazza *et al.* 2006). Limpens *et al.* (2008) concluded that N deposition at rates around 1-1.5 gm^{-2} yr^{-1} generally increases vascular plant cover, and thus increases CH_4 emissions through aerenchymatic tissue, while decreasing *Sphagnum* spp. cover.

The GHG balance of a mire or a peatland is controlled by the relative rates of net CO_2 uptake and efflux and CH_4 and N_2O emissions but, depending on climate, local and regional site characteristics, such as the position of the average water table (Alm *et al.* 2007), GHG emissions can vary between years with the function of the peatland changing from sink to source. From their initial formation, peatlands continuously take up and release GHGs. This fact should be borne in mind when discussing the status of mires and peatlands in atmospheric GHG balances (Strack (ed.) 2008).

3.2 Drained mires

Population pressure, need for food, fodder, fuel and raw material production let mire reclamation measures already start in the early middle ages (Heathwaite & Göttlich 1993). Before mires can be used for agriculture, forestry and peat extraction they have to be drained. But drainage is also the major cause of peatland disturbance. Drainage activities directly affect the GHG balance due to oxidative peat decomposition and denitrification, with concomitant emissions of CO_2 and N_2O to the atmosphere (Silvan *et al.* 2005). However, they also lower the biodiversity and change the water quality in runoff water in these areas (Lundin 1988, Sallantaus 1989). The CH_4 emissions from peatlands decrease after drainage, but those from drainage ditches may remain high (Minkkinen *et al.* 2008, Hiraishi *et al. (eds.)* 2014).

Furthermore, fundamental changes in the peat matrix, such as subsidence and shrinkage cracks, become visible (Jordan *et al.* 2007, Laiho 2008, Berglund *et al.* 2010). Porosity, water capacity, hydraulic conductivity and wettability of the peat decline and the peat body is characterised by structural damage (Zeitz & Velty 2002, Velty *et al.* 2006). In addition to the above-mentioned disturbances, ongoing peat mineralisation leads to transformation of organically bound phosphorus (P) into redox-sensitive inorganic

Fe(III)-oxyhydroxide-bound P (Gelbrecht & Koppisch 2001, Zak *et al.* 2004) driving P-leaching when the redox potential of P-binding chemical elements is changed, e.g. at rewetting (Jordan *et al.* 2007, Zak *et al.* 2008). However, this may only be a problem in peatlands used for agriculture and forestry with additional P fertilisation and therefore might be of less importance for extracted peatlands. It is widely reported that drained peatlands act as nutrient sources in the landscape in these conditions (Waddington *et al.* 2003, Maljanen *et al.* 2010) and therefore special account must be taken of the supplementary nutrient output potential to adjacent aquatic ecosystems when managing degraded peatlands.

3.3 Rewetted peatlands

For ecological purposes, peatlands drained for peat extraction have been subjected to rewetting actions (Vasander *et al.* 2003, Renou *et al.* 2006). The desired restoration goal with rewetting is re-establishment of the peatland's ecosystem functions (Mahmood & Strack 2011). How successfully this goal can be achieved depends on the existing environmental conditions, such as the used extraction method, the residual peat depth and peat type, the topography of the remaining peat surface together with its landscape situation, and the availability and quality of water resources (Lode 1999, Blankenburg & Tonnis (eds.) 2004, Jordan *et al.* 2009).

One problem in restoring extracted peatlands is re-vegetation. Natural colonisation by plants is a slow process, as seed or spore banks are often missing in the very old residual bare peat. In addition, the dry, powder-like and hydrophobic peat complicates germination and growth of plants (Vasander & Roderfeld 1996, Quinty & Rochefort 1997). Nevertheless, some studies report successful cultivation experiments with *Phalaris arundinacea* L. (e.g. Reinikainen *et al.* 2008, Shurpali *et al.* 2008, Mander *et al.* 2012) and herbs (e.g. Galambosi & Jokela 2008) on extracted peatlands. However, rewetting to construct new ecosystems for promoting the development of wetland vegetation (Tuittila *et al.* 2000), and thus climate mitigation, has greater value from an ecosystem rehabilitation point of view.

While CH_4 emissions may increase after peatland rewetting, it is also important to consider the transition time for emissions after rewetting (Vanselow-Algan *et al.* 2015). Augustin & Joosten (2007) describe high CH_4 emission fluxes as a transient phenomenon occurring in the initial stage directly after rewetting a fen type grassland. They attribute this temporary increase in flux to easily decomposable young biomass (*Phalaris arundinacea*) before inundation. Others have found persistently high CH_4 fluxes over

decades after rewetting in the presence of fresh organic material (Hahn-Schöfl *et al.* 2011, Vanselow-Algan *et al.* 2015). CO_2 emissions may be reduced in this period, but the recovery of the C sink function may take decades (Wilson *et al.* 2016a).

Furthermore, the magnitude of GHG fluxes from peatlands depends on various site parameters. Couwenberg *et al.* (2011) concluded that mean annual water level and vegetation are good proxies for GHG fluxes from peatlands in general. Vegetation not only reflects long-term water levels, but also directly affects GHG emissions due to assimilate supply (CO_2) and CH_4 transport to the atmosphere via the aerenchymatous system of plants (Tuittila *et al.* 2000, Couwenberg *et al.* 2011, Günther *et al.* 2014).

Studies on GHG emission fluxes from rewetted extracted peatlands have been undertaken in various time periods after rewetting (Tuittila *et al.* 2000, Marinier *et al.* 2004, Wilson *et al.* 2007, 2013 & 2016b, Waddington *et al.* 2010, Strack & Zuback 2013, Strack *et al.* 2014 & 2016). However, these results have to be complemented with information on how GHG fluxes change over space and time after rewetting. More data are needed on GHG emissions from extracted peatlands over longer time periods after rewetting (Strack *et al.* 2016) and from corresponding studies about constructed water bodies (e.g. shallow lakes), surrounded by morphologically vegetated littoral zones, which have been identified as high CH_4 emitters (Juutinen *et al.* 2003, Wilson *et al.* 2009), but also as low CH_4 emitters compared with open water (Franz *et al.* 2016).

3.4 GHG measurement techniques

The GHG exchange from different soil-plant-water systems to the atmosphere can be measured with manual or automated chambers or with tower- or aircraft-based micrometeorological methods such as eddy covariance (not developed for N_2O yet) or gradient techniques. The advantages and disadvantages of each measurement method are well summarised in Kutzbach *et al.* (2007) and Koskinen *et al.* (2014). For the detection of small-scale spatial variation in GHG fluxes, e.g. from tussocks-intertussocks or pool-hummock ecotopes in peatlands, use of chamber equipment is the preferred method, even though it poses a number of challenges (Koskinen *et al.* 2014). For example, there is no compulsory guideline regarding chamber design and measurement techniques, temporal resolution and flux estimation (Parkin & Venterea 2010, Sander & Wassmann 2014, ICOS 2016). Measurements with manual chambers are relatively inexpensive (but costs for operator(s) may be expensive), as the chamber can be self-constructed and a power supply in the field is not needed,

which is often the limiting factor for measurements in remote areas. Following some methodological standards for chamber measurements (Rochette & Eriksen-Hamel 2008, Parkin & Venterea 2010, ICOS 2016), e.g. to minimise the general impact (boundary layer effects) measurement imposes on the soil-atmosphere interface when 'sealing' a soil, water or vegetation surface with a chamber, the chambers can be inserted on any spot of interest.

Measuring ebullition in an accurate way is a difficult task (Yu *et al.* 2014). Ebullition is a non-diffusive emission process that can be divided into steady and episodic ebullition (Lai 2009, Green & Baird 2013). Steady ebullition can often be constant enough to be correctly measured with discontinuous sampling (e.g. with vials) of headspace air during a chamber enclosure period of less than 60 min. However, episodic ebullition, which is obviously non-constant over chamber enclosure, cannot easily be recorded by a vial sampling method (Duc *et al.* 2013). Yu *et al.* 2014 developed for dynamic chambers a graphical method for separating diffusion, steady ebullition and episodic ebullition fluxes from the total CH_4 flux. In particular, many studies reject data which are obviously disturbed by ebullition, because appropriate data handling is not ensured. However, ebullition is part of the CH_4 exchange from the soil/water to the atmosphere and should not be neglected in reporting of GHG emissions (Crawford *et al.* 2014).

The use of high temporal resolution measurement systems can provide continuous datasets with simultaneous detection of CH_4 and CO_2 and can also distinguish between biological and physical processes, such as ebullition (Pirk *et al.* 2016). Thus, high temporal resolution measurement of GHG fluxes is more reliable than other discontinuous sampling methods (e.g. with vials) for providing complete datasets to act as a basis for discussions about ecosystems and their GHG emissions impact in a changing climate, e.g. within national climate reporting work (Strack *et al*. 2005).

4 Material and methods

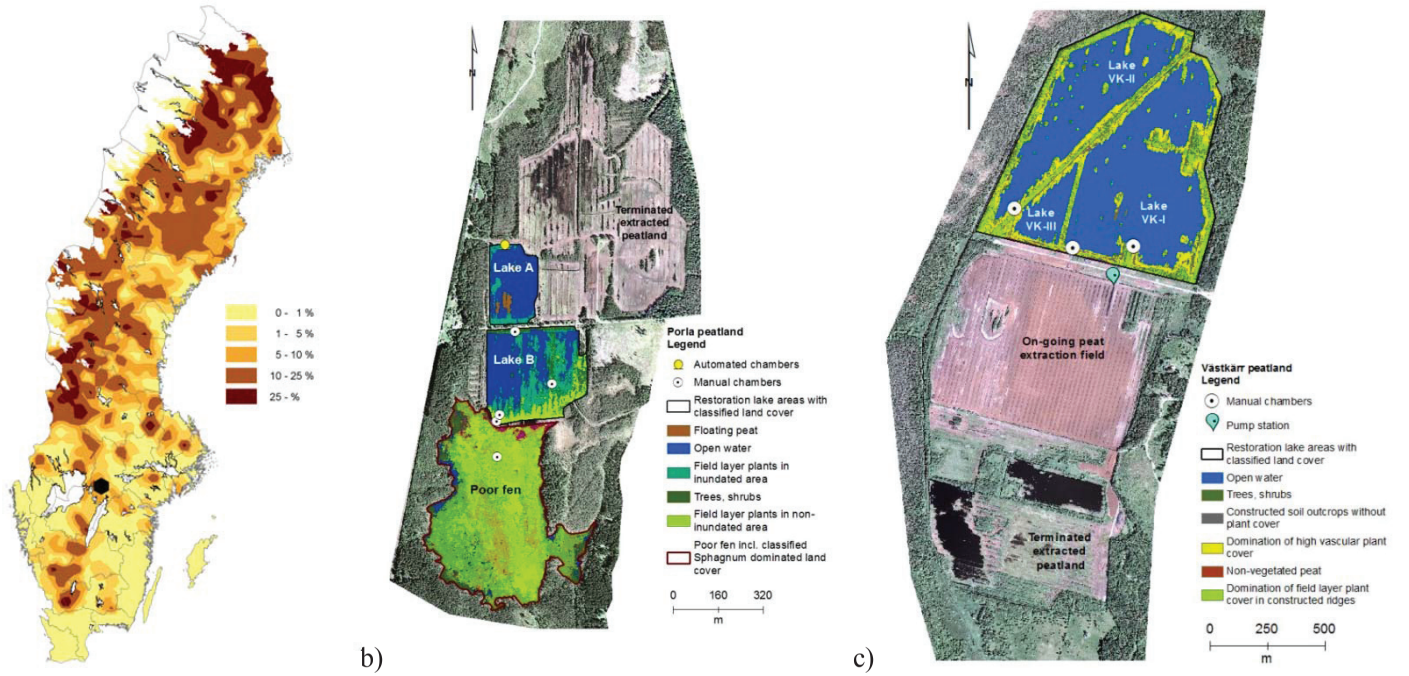
The work described in this thesis was carried out as part of the long-term project *'Restoration of terminated peat cuttings by rewetting'* (Jordan *et al.* 2009, Kozlov *et al.* 2016, Lundin *et al.* 2016) which started in 1997. Within the scope of the present work, investigations were carried out in two extracted peatlands, Porla and Västkärr, (Figure 2a) in sub-boreal central Sweden, where hydrology, peat and water chemical composition and vegetation development had been assessed two years before re-wetting in 1999 and over a 15 years post-rewetting period in Paper I. GHG measurements with manual opaque closed chambers started with pilot studies in 2008 and the first field campaigns took place in 2009. Monitoring was conducted to determine CO_2, CH_4 and N_2O fluxes to the atmosphere from several ecotopes and vegetation communities during different seasons over different years in these two rewetted peatlands after peat extraction (Papers II & III). Since measurements with manual chambers offer great spatial but low temporal resolution, a transparent automated chamber system for high frequency measurements of CO_2 and CH_4 was installed in 2013 and operated from then onwards (Paper IV).

Another study within this long-term project, but not included in this thesis, dealt with revegetation dynamics after 15 years of rewetting (Kozlov *et al.* 2016). The results of this vegetation monitoring were used as a basis for establishment of the GHG measurement positions in the present thesis and helped to interpret the results of these measurements in different wetland and peatland vegetation communities (Papers II-IV).

4.1 Study sites

4.1.1 The nutrient-poor Porla peatland (Papers I, II & IV)

The Porla peatland (place name: Porlamossen) is situated in Laxå municipality (59°01'N, 14°38'E; 74.1 ha, 85 m above sea level), 50 km south-west of the



a) b) c)

Figure 2. a) Undrained non-productive mire with peat thickness > 1 m in Sweden and proportion of total land area (%), the black dot marking the position of Porla and Västkärr study sites. Source: Hånell *et al.* (2008). Images of contemporary land cover in b) Porla and c) Västkärr peatlands, derived from orthophoto in 2014 (Sveriges Lantmäteriet) and the study sites. Images created as Supervised Classification (*ArcMap 10.2.1* tool); Cartography: Elve Lode.

city of Örebro in south-central Sweden, and includes a rewetted former peat extraction section and an open, nutrient-poor fen (Figure 2b). The climate is semi-humid and maritime (Köppen 1936) with a growing season length (temperature >5 °C for four days; Odin *et al.* 1983) of 200 days. Other climate conditions and peatland characteristics are summarised in Table 1.

Block peat extraction started in 1889 and continued until 1958. A second era of milled and sod peat extraction with mean annual volume of 30 000 m^3 started in 1980 and ended in 1999. A section of the peat extraction area was prepared for rewetting in 1999.

Rewetting established two shallow lakes (A and B, 5 and 12 ha, respectively) with a maximum water depth of 1.5 m. A device (outlet monk) was installed to control the water level from lake B. The Porla study site (Figure 2b) covers the southern part of the rewetted section, including lake B, a *Sphagnum* spp. dominated open poor fen (Paper II) and the northern part of lake A (Paper IV). The littoral zone of lake B is dominated by *Eriophorum vaginatum* tussocks and *Eriophorum angustifolium*. Large patches of bare peat form the inter-tussocks areas. The study site used in Paper IV was characterised by bare peat areas with *Eriophorum vaginatum* tussocks, as well as newly established *Sphagnum* spp.and *Eriophorum vaginatum* tussocks in the phreatic zone of the lake, both interspersed with small pools without vegetation.

4.1.2 The nutrient-rich Västkärr peatland (Papers I & III)

The Västkärr peatland is a rewetted peat extraction section in the lagg area of Skagerhultamossen, one of south-central Sweden's largest bogs (59°06'N, 14°45'E; 65 m above sea level). It is situated in Lekeberg municipality, 50 km south-west of Örebro (Figure 2a). Climate conditions are similar to those at Porla (Table 2). Prior to peat extraction, early drainage of the lagg area for hay production and cereal cultivation had already been conducted since the 18th century. In the 1970s, more intensive drainage for potato cultivation took place. Milled peat extraction for energy use started in 1987 and continued until 1997. Mean annual volume of extracted peat was 120 000 m^3 on 195 ha. The northern part of the peat extraction area in Västkärr (Figure 2c) was prepared for rewetting in 1998, with water storage starting in 1999. Rewetting established three shallow lakes with a total surface area of 80 ha and with mean water depth of around 1 m. These shallow lakes, which have many small constructed peaty islands, are valuable habitats for common and rare migration bird species.

The dominant mesotrophic and eutrophic wetland species in Västkärr are *Carex* spp., *Phragmites australis* and *Thypha latifolia*. Other helophytes and

Table 1. *Characteristics of the Porla study site.*

Geographic coordinates (WGS 84)	59°01'N, 14°38'E									
Mean annual temperature	1961-1990[a]: 5.7 °C 1997-2014[b]: 6.8 °C									
Mean annual precipitation	1961-1990[a]: 690 mm yr^{-1} 1997-2014[b]: 752 mm yr^{-1}									
Height above sea level	85-88 m									
Main peat types	*Sphagnum*, *Carex* and *Eriophorum* peat (often mixed with each other)									
Degree of peat humification	*H* 2-8									
Peat C/N ratio	45									
Peat pH	3.7-4.4									
Bulk density	0.02-0.12 g cm^{-3}									
Ecotopes	Mire	Vadose *Eriophorum vaginatum*	Bare peat	Phreatic *Eriophorum vaginatum*	*Eriophorum angustifolium*	Pond	Lake	Newly established *Sphagnum* spp.	Water between Eriophorum vaginatum tussocks	Pools
Opaque manual chambers with discontinuous sampling (vials and IRGA)	9	8	8	8	8	2	2	-	-	-
Opaque manual chambers with continuous sampling (high frequency measurements of CO_2 and CH_4)	9	8	8	8	8	-	-	4	-	-
8 transparent automated chambers[c] with continuous sampling (high frequency measurements of CO_2 and CH_4) and alternating placement during the years	-	+	+	+	-	-	-	+	+	+

[a] Raab & Vedin 1995; [b] 4x4 km gridded data; SMHI 2014; [c] different site as the manual chambers

Table 2. *Characteristics of the Västkärr study site.*

Geographic coordinates (WGS 84)	59°06'N, 14°45'E
Mean annual temperature	1961-1990[a]: 5.7 °C; 1997-2014[b]: 6.8 °C
Mean annual precipitation	1961-1990[a]: 690 mm yr^{-1}; 1997-2014[b]: 723 mm yr^{-1}
Height above sea level	65-68 m
Main peat types	*Carex* peat mixed with wood
Degree of peat humification	*H* 8 to *H* 10
Peat C/N ratio	21
Peat pH	5-6
Bulk density	0.2-0.3 g cm^{-3}
Sites	3 transects with a gradient from dry vegetated shore (above the water table), over the vegetated littoral zone (water level height 0-0.3 m) to the open water
Dominant plants	From land to water: *Graminoids*, *Scirpus* spp. and communities of both; *Thypha latifolia*, *Carex* spp. and communities of both
Opaque manual chambers with discontinuous sampling	3 × 8 chambers

[a] Raab & Vedin 1995; [b] 4x4 km gridded data; SMHI 2014

hydrophytes such as *Alisma plantago-aquatica*, *Butomus umbellatus*, *Hydrocharis morsus-ranae*, *Juncus* spp., *Lythrum salicaria*, *Scirpus* spp. and *Sparganium* spp. also occur. *Lemna minor* is ubiquitous between late June and October. Graminoids such as *Calamagrostis canescens*, *Poa trivialis* and *Phalaris arundinacea* dominate the dryer parts of the shore and the constructed ridges between the lakes. Vegetation composition changes somewhat in cover values (Kozlov *et al*. 2016) due to intense inner-annual and inter-annual water level fluctuations.

4.2 Peat and water sampling (Paper I)

Long-term rewetting investigations (Paper I) were carried out at the two peat extraction sites (Porla and Västkärr) between 1997 (before rewetting), in the initial years after wetland establishment and up to 15 years post-rewetting (2013). Field measurements are still ongoing. Discharge determinations were made in weirs with water level recorders. Routine fortnightly field observations of water depth and monthly water sampling at the Porla site were carried out by a local observer. In Västkärr, water was sampled 4-6 times per year. Control visits were made by the SLU research team 2-6 times per year.

Soil samples were taken using a steel auger from 0-10 cm and 10-20 cm in Västkärr and from 0-20 cm and 20-40 cm in Porla before rewetting (1997) and after rewetting (in 2003 and 2007) and again in 2009 and 2011 (not included here). These soil samples were placed in plastic bags, transported to the laboratory at the Department of Soil & Environment, SLU, Uppsala, and stored there under cool conditions until analysis. Soil chemical analyses (C_{tot}, Ca_{tot}, K_{tot}, Mg_{tot}, N_{tot}, NH_4-N, NO_3-N, P_{tot}, pH, and S_{tot}) were performed on field-fresh and air-dried samples. Water chemical analyses for pH, colour, electrical conductivity, alkalinity and various elements, such as DOC, N_{tot}, NH_4-N, NO_3-N, N_{org}, PO_4-P and P_{tot}, SO_4-S, base cations and metals, were carried out according to routine to methods (SLU 2016) at SLU's water laboratory in Uppsala within 1-2 days.

4.3 GHG measurements

In all field sites for GHG measurements (Paper II-IV) wooden boardwalks were installed to prevent disturbance to the peat by trampling. On every GHG flux measurement occasion, soil or water temperature at 10 cm depth was determined adjacent to each measurement position. *In situ* measurements of pH, redox potential, electrical conductivity and temperature were performed with a portable multi-parameter meter (Multi 1970i, WTW, Germany) from 2011 onwards.

4.3.1 Nutrient-poor Porla peatland (Papers II & IV)

Manual opaque closed chambers with non-continuous determination of CO_2, CH_4 and N_2O (Paper II)

In Porla, seven ecotopes with typical wetland and peatland characteristics (Table 1) were selected (Paper II). Within each of these ecotopes, sub-areas were chosen for investigations (Figure 2b). A total of 45 flux measurement positions were established in the seven sub-areas. The CO_2, CH_4 and N_2O fluxes to the atmosphere were measured on 10 occasions during daytime in the snow-free period (April to October) from 2009 to 2012. For determination of gas fluxes to the atmosphere, see Section 4.3.

Manual opaque chambers with continuous determination of CO_2 and CH_4 (Paper IV)

In addition to the non-continuous determination of CH_4, high-frequency measurements of CO_2 and CH_4 with manual chambers were carried out in the above named ecotopes (Figure 2b). A total of 45 flux measurement positions were established. The CO_2 and CH_4 fluxes to the atmosphere were measured

on six occasions during daytime in the snow-free period (April to October) in 2013. Continuous measurements for short-term detection of CH_4 and CO_2 fluxes were performed using the opaque chamber method as described by Jordan *et al.* (2016) but attached to an infrared laser absorption gas analyser (Model 915-0011, Los Gatos Research, USA) to measure CO_2 and CH_4 volume concentrations in headspace air (cf. Juszczak 2013). The gas analyser was additionally shielded with aluminium foil. Different ebullition patterns were determined by visual inspection of the continuous dataset for both CH_4 and CO_2.

Automated transparent chambers with continuous determination of CO_2 and CH_4 (Paper IV)

An automated system with eight transparent closed chambers was installed in five ecotopes (Table 1) along the northern shore of lake A (Figures 2b). During the snow-free period, continuous measurements of CH_4 and CO_2 fluxes were performed from 2013 to 2015. The measurements of CO_2 flux included autotrophic and heterotrophic respiration from the soil and release from the water, as well as respiration or uptake by photosynthesis from the biomass.

The automated closed chamber system consisted of eight transparent polycarbonate chambers (35 cm in diameter; Figure 3) resting on frames (frame and chamber height were dependent on the surface cover). Each chamber was equipped with a motor to open and close the chamber lid, an air mixing fan, a photosynthetically active radiation sensor, copper-constantan thermocouples for air temperature recording (every third minute) inside the chamber and a pressure vent. The chambers were connected to an infrared laser absorption gas analyser (Model 915-0011, Los Gatos Research, USA) via ~15 m long heated tubes. A pump circulated the air (4.5 ℓ min^{-1}) between the chamber and the gas analyser during each measurement. The gas analyser,

Figure 3. Automated transparent floating chambers (left) and automated transparent chamber (right) at Porla study site (Photos: Sabine Jordan).

pump, switching devices, multiplexer and solenoids were placed inside an insulated caravan.

Adjacent to each chamber, soil or water temperature was measured by copper-constantan thermocouples at a depth of 10 cm from the soil or water surface, also every third minute. At a central point between all chambers, a water level device with a data logger was installed (WT-HR, TruTrack, New Zealand). Water level was measured two times per hour and the value obtained was then used for headspace volume corrections in the chambers.

A similar automated system was used by Strömgren (2001), but in the present study a computer was used to control the system instead of a data logger. A measurement cycle for the eight chambers required 48 min. Each chamber was measured for three minutes. The first minute was used to flush the old air from the tubes and the analyser. Thus, the observations for the first minute were discarded because of the delay of the air transport through tubes and possible disturbance caused by the closure of the lid. Hence, the last two minutes of observations before the lid was opened were used to estimate the flux. CO_2, CH_4, $H_2O_{(g)}$ volume concentrations were observed every 10 s with start at chamber closure, but 1 s values were also stored. After a measurement and when the lid started to open, ambient air was flushed through the system.

The CO_2 and CH_4 fluxes were estimated by linear and non-linear regression according to Koskinen *et al.* (2014). As the change in air humidity was small during each measurement, the CO_2 and CH_4 volume concentration values were not corrected for water vapour dilution. Monthly arithmetic means of the fluxes were calculated.

Data for the synopsis of CO_2 and CH_4 emissions were chosen from June, July and August 2015 only. Different ebullition patterns were determined by visual inspection of the continuous dataset for both CH_4 and CO_2.

4.3.2 Nutrient-rich Västkärr peatland (Paper III)

Manual opaque closed chambers with non-continuous determination of CO_2, CH_4 and N_2O

In Västkärr, ecosystem respiration (CO_2), CH_4 and N_2O flux measurements were performed on 16 occasions during daytime in the snow-free period (April to October) from 2009 to 2014. A total of 24 GHG flux measurement positions, as frames or floating chambers were established in wetland and non-wetland vegetation zones along a water level gradient at three transects (Figure 2c): generally three frames in the vegetated soil (*Graminoids*, *Scirpus* spp. and communities of both) higher up the shore and above the water table; three in the vegetated part of the littoral zone (*Carex* spp., *Thypha latifolia* and

communities of both) with water depth 0-0.3 m; and two floating chambers in the (vegetation-free) water of the lake. These conditions were taken as 'normal' conditions and they changed to 'dry' or 'inundated' conditions from time to time, meaning that the whole transect was lying above or below the water level, respectively. Thus, it was also possible to investigate, how GHG emissions from typical wetland and non-wetland vegetation reacted to drought and inundation.

When seasonally flooded, floating chambers were used almost exclusively. When the water level became very low, frames were used instead of floating chambers. The frames in the littoral zone were adapted in height to the plant development and to water level height. Thus, young *Thypha latifolia* plants which fitted in the chamber without bending were measured in spring and early summer. *Thypha latifolia* can grow to heights up to 2.5 m. For that reason GHG flux measurements within the *Thypha latifolia* community were performed without including plants in the chamber between late June and late October. For determination of gas fluxes to the atmosphere, see Section 4.3.

4.3.3 Determination of gas fluxes to the atmosphere for manual opaque closed chambers with non-continuous determination of CO_2, CH_4 and N_2O (Papers II & III)

The CO_2, CH_4 and N_2O fluxes from soil-plant-water systems to the atmosphere were determined by time series measurements of the respective volume concentration in chambers placed on frames resting on the ground or floating on the water surface, followed by regression analyses to estimate the concentration change per unit time and to estimate the resulting flux.

The CO_2, CH_4 and N_2O measurements were made using the opaque closed chamber method (Parkin & Venterea 2010, Pumpanen *et al.* 2010) and, as such, only ecosystem respiration was measured for CO_2. For gas sampling on peat and *Sphagnum* spp., permanent annular polyvinylchloride (PVC) frames (inner base diameter 18.7 cm) were installed at each GHG flux measurement position (Figure 4). To avoid lateral gas exchange in the soil, the insertion depth of the frames varied due to different soil water conditions (cf. Hutchinson & Livingston 2001, Davidson *et al.* 2002). For flux measurements, a non-steady-state flow-through opaque chamber was attached to the frame and sealed with a rubber gasket (Figure 4). The chamber was made of PVC with 18.7 cm inner base diameter and 16 cm height and had an effective chamber air volume of 4.3 ℓ. Air samples were taken in crossflow through the chamber headspace with polysiloxane tubing. Frame air volume was determined on each GHG measurement occasion to obtain the headspace air volume (sum of chamber and frame air volumes). A chamber installed in the centre of a life

Figure 4. Frame (left), manual opaque chamber (middle) and manual opaque floating chamber (right) (Photos: Sabine Jordan).

buoy (i.e. floating chamber) identical in dimensions and with an effective headspace air volume of 3.0 ℓ was used for GHG concentration time series measurements on open water (Figure 4).

The CO_2 volume concentration in the headspace air was determined using a portable infra-red gas analyser (GMP343 and MI70, Vaisala, Finland) connected to the chamber. An external membrane pump (flow rate 0.4 ℓ min^{-1}) circulated the air between the chamber headspace and the CO_2 probe during the 4.5 min chamber closure. CO_2 concentration is measured as 10-s average at 25, 55, … , 265 s. For measurements of CH_4 and N_2O concentrations in headspace air, 20 mℓ air samples were collected in septum bottles (glass vials with 20 mm/3.0 mm butyl-polytetrafluorethylene septum in aluminium seal cap, Scantec Nordic, Sweden) at 10, 20, 30, 40 and 50 min after chamber closure (April 2009 to May 2012) or 10, 20 and 30 min after chamber closure (June 2012 to October 2013) or 1, 11, 21, 31 and 41 min after chamber closure (2014). Air was circulated with an external membrane pump (volume flow rate 0.4 ℓ min^{-1}) between chamber and vial over 20 s. The CH_4 and N_2O samples were analysed using a gas chromatograph (Clarus 500, Perkin Elmer, USA) equipped with a flame ionisation detector, an electron capture detector and an automatic vial headspace injector (Turbo Matrix 110, Perkin Elmer, USA). CH_4 standards of 2, 10, 20 and 350 ppmn and N_2O standards of 0.3, 1.7 and 4.7 ppmn (AGA, Sweden) were used for calibration. The standards' certified mixture compositions were given in amount fraction (ppmn), however, the volume concentration (ppmv) was used in the present study (1 ppmn ≙ 1 ppmv) (Calvert 1990, Möller 2003, IUPAC 2006).

CO_2, CH_4 and N_2O fluxes were estimated according to $F = f'(t_0) \times p \times V \div (A \times R \times T)$ where F is the molar flux to the atmosphere. The first functional derivative $f'(t_0)$ at the moment of chamber closure t_0 is estimated from the regression function $f(t) = y_{gas}(t)$ of the change in volume

concentration y_{gas} in headspace air over time t and given as concentration per unit time; p is the atmospheric pressure, V the headspace air volume, A the chamber base area, R the molar gas constant and T the sample air temperature. Before regression analysis for CO_2, all y_{gas} values within the first 50 s after chamber closure were rejected due to potential disturbances caused by chamber attachment (cf. Davidson *et al.* 2002, Kutzbach *et al.* 2007). To estimate CO_2 fluxes, linear, exponential and quadratic regressions were estimated and the monotone regression function with the least residual standard deviation was used (cf. Kutzbach *et al.* 2007, Koskinen *et al.* 2014). Linear regression was used to estimate CH_4 and N_2O fluxes (cf. Kutzbach *et al.* 2007, Parkin & Venterea 2010, de Klein & Harvey (eds.) 2012).

4.4 Statistical analyses

All variables studied in Paper I were not normally distributed. Therefore, before studying the effects of rewetting, all data were log-transformed to meet normality and homoscedasticity assumptions. A threshold of $P < 0.05$ was always used for significance. *SAS 9.4*® was used for statistical analyses.

A linear mixed effects analysis (cf. Bates 2010, Gries 2012, Winter 2013) was performed (Papers II & III) to investigate the relationships between GHG fluxes and some adjacent environmental conditions (e.g. soil temperature, vegetation community or ecotope, water level or soil water condition) based on the individual flux estimates of all measurement positions by means of the packages *lme4* and car in *R x64 3.2.2* (Fox & Weisberg 2011; Bates *et al.* 2015a, b; R Core Team 2015). To identify the best fitting model for a GHG flux, *P*-values and *Akaike's Information Criterion* were obtained by likelihood ratio tests comparing a model with the effect in question against the respective model without the effect in question (cf. Gries 2012, Winter 2013). The models were also checked and compared for good approaches to homoscedasticity and normality. CH_4 flux estimates were not included in the linear mixed effects analysis if disturbance by ebullition was plausible. N_2O was not included in the linear mixed effects analyses as the huge majority of flux values were below the detection limit.

In addition to the results in Paper II, differences between GHG fluxes from the various ecotopes on each measurement occasion were investigated using the *Kruskal-Wallis* test for multiple comparisons of independent samples and the *Conover* post-hoc test for pair-wise comparisons (*PMCMR 4.0* package in *R x64 3.2.3*; R Core Team 2015, Pohlert 2014, 2016). The intention with this was to develop an emission flux ranking sorted from high to low emitter ecotopes.

For illustrative purposes in this thesis, a two-tailed *Mann-Whitney U*-test was used for pair-wise comparison of the CO_2 and CH_4 emission fluxes (averages) from the natural *Sphagnum* mire (as a control site) with the fluxes from each of the other ecotopes, and thus to identify differences.

5 Results and discussions

5.1 From peat extraction areas to shallow lakes – after-use of industrial peat sites (Paper I)

Two functioning wetland ecosystems with stable hydrology have established during 17 years after rewetting of the former peat extraction sites. Both, fen vegetation and *Sphagnum* mosses have developed (Kozlov *et al*. 2016). This is an important prerequisite for new peat growth and a future C sink.

However, in the Porla peatland, some initial mishaps lowered the lake water level twice, in 2001 and 2002, to 0.1-0.2 m depth, but after that it stabilised at around 1 m depth in the central parts (Figure 5). In the Västkärr lakes, the water level remained fairly stable from the beginning of rewetting, with depth between 0.5 m and 1 m and occasional maximum depth of 2 m (Figure 5).

Compared with the pre-rewetting phase, the concentrations of DOC, base cations, NO_3-N, NH_4-N and N_{tot} were lower 15 years after rewetting, but concentrations also varied somewhat over time. Water pH at the Porla site remained fairly stable, but at the Västkärr site, after an initial dip, pH gradually increased to higher values than before rewetting. The P_{tot} concentration increased over time, especially in the nutrient-rich Västkärr site. After rewetting, P stored in sediments can easily be released when anoxic bottom conditions are reached and reductant-soluble or acid-soluble metals such as iron, aluminium and calcium as potential P binding partners are available (Zak *et al.* 2008). At the nutrient-rich Västkärr site, a significant increase in total P was observed after rewetting, while PO_4-P decreased after a small initial increase in the first two years of rewetting. This was probably related to biological uptake forming organically bound P. In addition, the remaining peat in Västkärr is highly decomposed (*H* 8 to *H* 10) and thus might have a high P release potential after rewetting (Jordan *et al*. 2007, Zak *et al.* 2008). After the initial pulse of P release, the P concentration decreased somewhat in the next

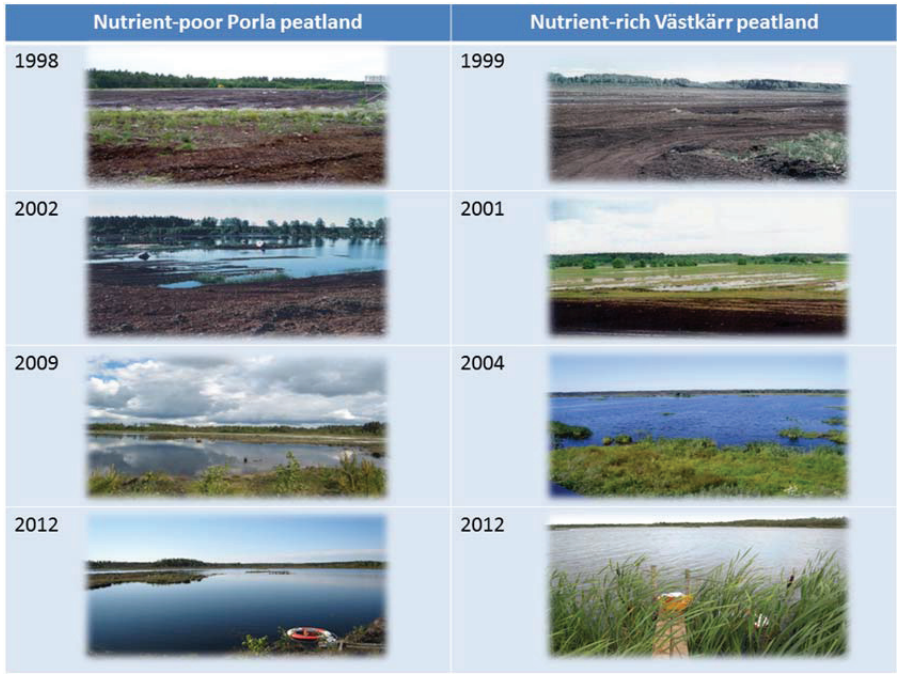


Figure 5. Rewetting stages at Porla and Västkärr (Photos: Lars Lundin, Sabine Jordan).

5-11 years after rewetting, but was still higher 12-15 years after rewetting than before rewetting. Thus, if oxidised metals remain after re-wetting, P may be released from these highly decomposed peat horizons at the bottom of newly established shallow lakes over many years. However, at the nutrient-poor Porla site there were no significant effects on total P or PO_4-P after rewetting, although there was a tendency for increased total P values. This is perhaps because the old bottom peat has not changed pedogenetically (*H* 2 to *H* 8) to the same extent as the peat at Västkärr and thus the P release after re-wetting might be rather small (Jordan *et al*. 2007, Zak *et al.* 2008).

5.2 Ecosystem respiration and CH_4 and N_2O fluxes to the atmosphere (Papers II & III)

Linear mixed effects analysis was performed for better generalisation between the environmental conditions and the CO_2 and CH_4 emission fluxes from the ecotopes and vegetation communities. The mixed models obtained allowed the influences of ecotope type and vegetation communities on GHG fluxes to be related to the influences of soil temperature, soil water condition and vitality of plants and, consequently, to aggregate the flux measurement results from a

number of one-day-observations, dependent on diurnal variations of temperature, to a more generalised view on temperature, weather and season.

5.2.1 The nutrient-poor Porla peatland (Paper II)

Ecotopes had a significant impact on CO_2 and CH_4 fluxes to the atmosphere. Ecosystem respiration and CH_4 emissions from the bare peat site, the constructed shallow lake and the open poor fen (mire) were low but were much higher from ecotopes with *Eriophorum vaginatum* tussocks and *Eriophorum angustifolium*. The N_2O emissions contributed little to total GHG fluxes from the soil-plant-water systems to the atmosphere as the site is nutrient poor (C:N ratio > 25).

CO_2

A combination of vascular plant cover and high soil temperatures enhanced ecosystem respiration (Figure 6). The linear mixed effects model also significantly distinguished between the CO_2 fluxes from the wet ecotopes and those from saturated and inundated ecotopes, which were highest from the vadose *Eriophorum vaginatum* ecotope and the bare peat under wet conditions (Figure 6). Since the measurement technique covered only ecosystem respiration, it was not possible to estimate the net CO_2 uptake. Therefore, all high CO_2-emitting sites such as the *Eriophorum* spp. ecotopes might accumulate more C (cf. Wilson *et al.* 2013, Strack *et al.* 2014) than non-vegetated sites, water bodies and the mire.

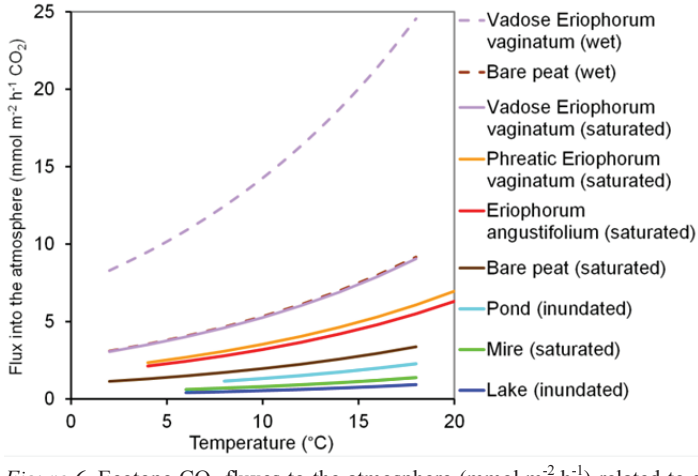


Figure 6. Ecotope CO_2 fluxes to the atmosphere (mmol m^{-2} h^{-1}) related to soil/water temperature (°C) and soil water conditions. Fluxes estimated with the linear mixed effects model (Table 5 in Paper II) based on measured temperature ranges.

CH_4

As has been found previously, water table position and the presence of aerenchymatous species are important controls on CH_4 fluxes from peatlands to the atmosphere (Bubier 1995, Le Mer & Roger 2001, Larmola *et al.* 2010, Mahmood & Strack 2011, Miller 2011). In this study a combination of vascular plant cover, high water table levels and high soil temperatures enhanced CH_4 emissions (Figure 7). Some authors (e.g. Juutinen *et al.* 2003, Wilson *et al.* 2009) have also observed a distinct contrast in CH_4 fluxes between the highly productive littoral zone (in Porla: the *Eriophorum angustifolium* ecotope) and the pelagic zone of a lake. Within the vegetated ecotopes at soil surface water level (mire, phreatic *Eriophorum vaginatum*, *Eriophorum angustifolium*), the mire emitted less CH_4 than the two *Eriophorum* spp. ecotopes. This can be attributed to minimised plant-mediated CH_4 transport in the mire due to the absence of vascular plants and to simultaneous CH_4 oxidation as a result of the methanotrophic bacteria living in the aerobic *Sphagnum* spp. lawn (Whalen 2005, Hornibrook *et al.* 2009, Fritz *et al.* 2011).

In contrast, sites with an oxic upper peat horizon (i.e. the bare peat ecotope in Porla) support CH_4 oxidation and thus low CH_4 fluxes to the atmosphere are observed. In addition to the increasing organic substrate supply after rewetting, which may lead to high CH_4 emissions, ecotopes with *Eriophorum vaginatum* tussocks can also serve as CH_4 catalysts. Due to their wide belowground network of roots and rhizomes, *Eriophorum vaginatum* can absorb CH_4 (Frenzel & Rudolph 1998) and can thus lead to somewhat higher CH_4 fluxes to the atmosphere than the surrounding bare peat (Tuittila *et al.* 2000).

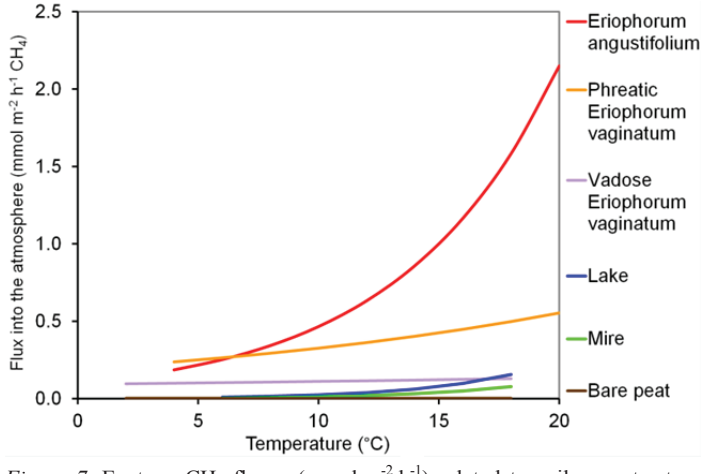


Figure 7. Ecotope CH_4 fluxes (mmol m^{-2} h^{-1}) related to soil or water temperature (°C). Fluxes estimated with the linear mixed effects model (Table 6 in paper II) based on measured temperature ranges.

Pair-wise comparison and emission ranking of ecotopes

A pair-wise comparison between the natural *Sphagnum* mire and each of the other ecotopes identified that the mire in general emitted less CO_2 (except the lake) and CH_4 (except the bare peat) than the other ecotopes (Figures 8 & 9) even though the differences were not always significant. These results are in line with the results achieved with the linear mixed effects analysis (Figures 6 & 7).

Ecotopes were ranked from the highest to the lowest CO_2 and CH_4 emission fluxes for each measurement occasion (Tables 3 and 4). The *Kruskal-Wallis* tests indicated that CO_2 emission fluxes were significantly different between the ecotopes, except for July 2010 and October 2011. The *Conover* post-hoc test indicated that the emission fluxes from the mire were significantly lower than those from the three *Eriophorum* spp. ecotopes for the majority of measurement occasions. The CH_4 emission fluxes were significantly different between the ecotopes, except for September 2012. The emission fluxes from the mire were significantly lower than those from the phreatic *Eriophorum vaginatum* and *Eriophorum angustifolium* ecotopes for the majority of measurement occasions, as indicated by the *Conover* post-hoc test results.

The non-parametric *Mann-Whitney* and *Kruskal-Wallis* tests and a simple calculation of arithmetic means and quartiles (Paper II; Figures 2 & 4) came in the end to the same result as the linear mixed effects analysis.

5.2.2 The nutrient-rich Västkärr peatland (Paper III)

Water level and different vegetation communities had a significant impact on CO_2 and CH_4 fluxes to the atmosphere in Västkärr. In general, ecosystem respiration was higher than CH_4 emissions (Figure 10). Since the same measurement technique as in Paper II was used in Paper III, only ecosystem respiration was covered and thus it was not possible to estimate the net CO_2 uptake. The CO_2 uptake by photosynthesis may decrease the net CO_2 fluxes to the atmosphere from the vegetated part of the shore zone. The N_2O emissions contributed little to total GHG fluxes from the soil-plant-water system to the atmosphere, even though the site is nutrient-rich (C:N ratio < 25).

Fluxes of CO_2 and CH_4 in 'normal' conditions

In 'normal' conditions, CO_2 fluxes decreased along the water level gradient from the dry vegetated shore towards the open water, whereas CH_4 fluxes were highest from the vascular plants in the littoral zone with standing water. These results are in line with general knowledge about vascular plants with aerenchymatous tissues being the dominant factor for CH_4 fluxes from

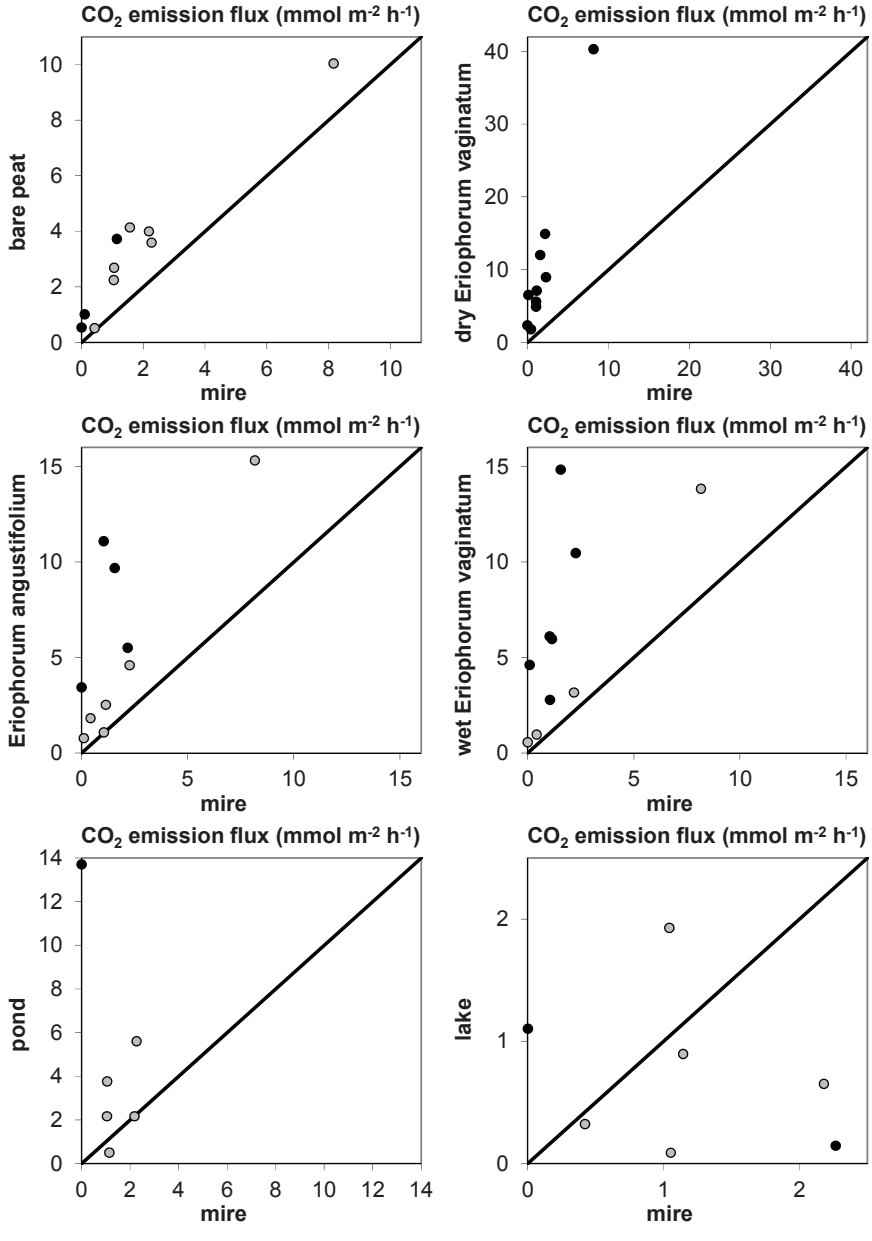


Figure 8. Pairwise comparison of the CO_2 emission fluxes (averages) from the natural *Sphagnum* mire (as a control site) and from each of the other ecotopes (two-tailed Mann-Whitney *U*-test); note different scales on x- and y-axes. Black dots = significant difference between the plot samples (two-tailed U-test, $P \leq 0.05$); grey dots = no significant difference between the plot samples (two-tailed U-test, $P > 0.05$); line = equal CO_2 emission fluxes.

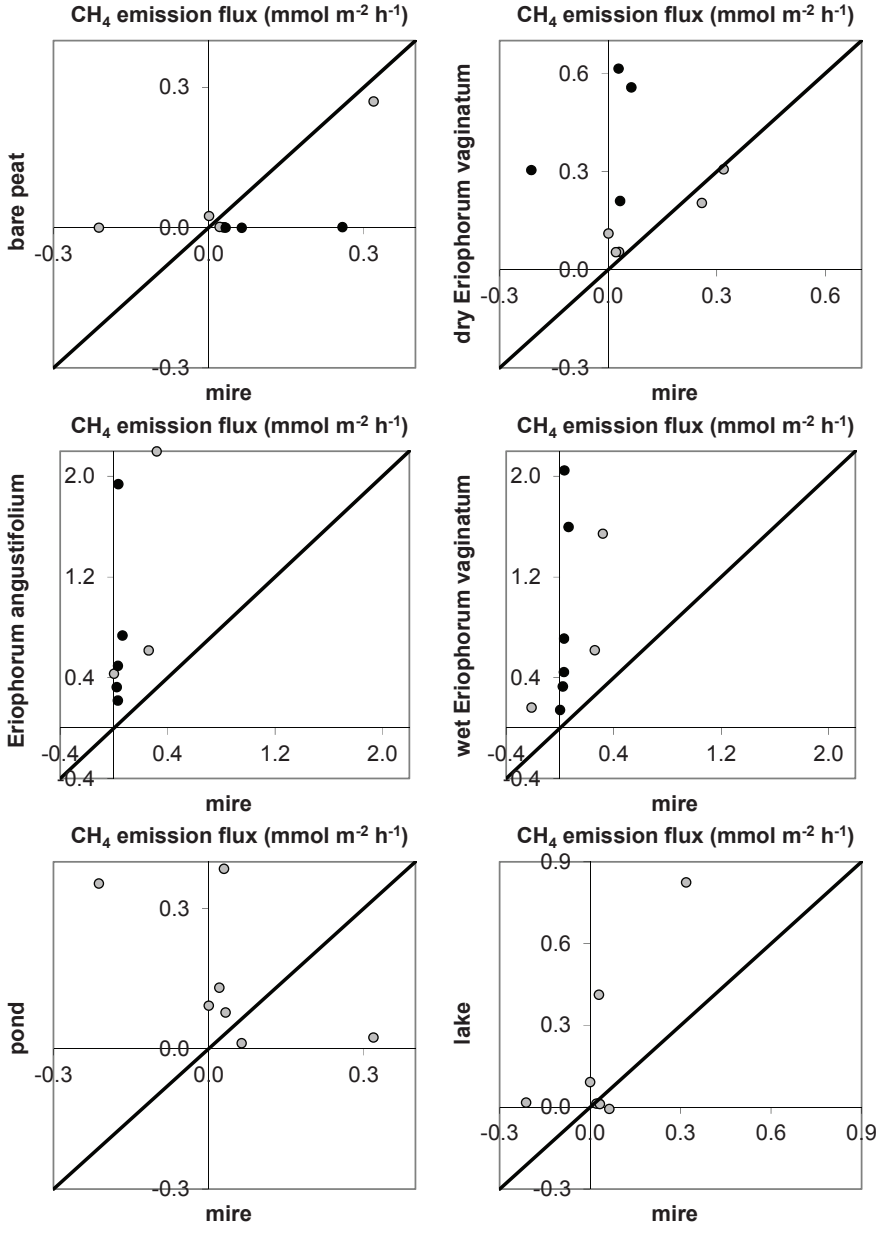


Figure 9. Pairwise comparison of the CH_4 emission fluxes (averages) from the natural *Sphagnum* mire (as a control site) with the fluxes from each of the other ecotopes (two-tailed Mann-Whitney *U*-test); note different scales on x- and y-axes. Black dots = significant difference between the plot samples (two-tailed U-test, $P \leq 0.05$); grey dots = no significant difference between the plot samples (two-tailed U-test, $P > 0.05$); line = equal CH_4 emission fluxes.

Table 3. *Arithmetic means of CO_2 emission fluxes [mmol m^{-2} h^{-1}], ranked by mean and sample size (in brackets) per ecotope and measurement occasion with results of Kruskal-Wallis and Conover post-hoc tests. Mire (as reference ecotope in bold) was not subjected to peat extraction. Ecotopes that are significantly different in fluxes from the mire are marked with *. P_{KW} = P values obtained from Kruskal-Wallis test; n_C = number of ecotope pairs with significantly different fluxes out of total number of pairs from Conover post-hoc test (P < 0.05); vad. = vadose; phr. = phreatic.*

	Measurement occasion									
date	16/04/09	15/05 & 16/05/12	06/06/09	26/06/12	05/07 & 06/07/10	15/07 & 16/07/09	04/09/12	19/10/09	24/10 & 25/10/11	24/10 & 25/10/12
P_{KW}	**0.002**	**0.003**	**0.0004**	**0.0003**	0.28	**0.0002**	**0.002**	**0.0001**	0.07	**0.009**
n_C	11 of 21	12 of 21	13 of 21	12 of 21	-	6 of 10	9 of 21	7 of 10	-	5 of 15
Rank										
1.	* Vad. *E. vag.* 7.1 (8)	* Vad. *E. vag.* 4.9 (6)	* Phr. *E. vag.* 10.5 (8)	* Vad. *E. vag.* 14.9 (6)	Vad. *E. vag.* 40.3 (5)	* Phr. *E. vag.* 14.8 (8)	* *E. ang.* 11.1 (6)	* Vad. *E. vag.* 6.5 (5)	*E. ang.* 1.8 (5)	* *E. ang.* 3.4 (8)
2.	* Phr. *E. vag.* 6.0 (8)	* Pond 3.8 (2)	* Vad. *E. vag.* 9.0 (5)	* *E. ang.* 5.5 (5)	*E. ang.* 15.3 (5)	* Vad. *E. vag.* 12.0 (5)	* Phr. *E. vag.* 6.1 (5)	* Phr. *E. vag.* 4.6 (8)	Vad. *E. vag.* 1.8 (5)	* Vad. *E. vag.* 2.3 (5)
3.	* Bare peat 3.7 (8)	* Phr. *E. vag.* 2.8 (5)	* Pond 5.6 (2)	* Bare peat 4.0 (8)	Phr. *E. vag.* 13.8 (8)	* *E. ang.* 9.7 (5)	* Vad. *E. vag.* 5.6 (5)	* Bare peat 1.0 (6)	Phr. *E. vag.* 9.7 (8)	* Lake 1.1 (2)
4.	*E. ang.* 2.5 (5)	* Bare peat 2.7 (8)	* *E. ang.* 4.6 (5)	Phr. *E. vag.* 3.2 (5)	Bare peat 10.0 (6)	Bare peat 4.1 (6)	Bare peat 2.2 (8)	*E. ang.* 0.8 (5)	Bare peat 0.5 (6)	Phr. *E. vag.* 0.6 (5)
5.	**Mire** 1.1 (7)	*E. ang.* 1.1 (5)	Bare peat 3.6 (6)	**Mire** 2.2 (8)	**Mire** 8.2 (8)	**Mire** 1.6 (8)	Pond 2.2 (2)	**Mire** 0.1 (8)	**Mire** 0.4 (8)	* Bare peat 0.5 (8)
6.	Lake 0.9 (2)	**Mire** 1.1 (8)	**Mire** 2.3 (8)	Pond 2.2 (2)	-	-	Lake 1.9 (2)	-	Lake 0.3 (1)	**Mire** 0.0 (2)
7.	Pond 0.5 (2)	Lake 0.1 (2)	Lake 0.1 (4)	Lake 0.7 (2)	-	-	**Mire** 1.0 (8)	-	-	-

Table 4. *Arithmetic means of CH_4 emission fluxes [mmol m^{-2} h^{-1}], ranked by mean and sample size (in round brackets) per ecotope and measurement occasion with results of Kruskal-Wallis and Conover post-hoc tests. Mire (as reference ecotope in bold) was not subjected to peat extraction. Ecotopes that are significantly different in fluxes from the mire are marked with *. Arithmetic means and sample size in curly brackets were compiled only from flux estimates being sufficiently free of disturbance by ebullition (no Kruskal-Wallis test for this compilation). P_{KW} = P values obtained from Kruskal-Wallis test; n_C = number of ecotope pairs with significantly different fluxes out of total number of pairs from Conover post-hoc test ($P < 0.05$); vad. = vadose; phr. = phreatic.*

	Measurement occasion								
date	16/04/09	15/05 & 16/05/12	06/06/09	26/06/12	05/07 & 06/07/10	15/07 & 16/07/09	04/09/12	19/10/09	24/10 & 25/10/12
P_{KW}	**0.02**	**0.0002**	**0.0004**	**0.002**	**0.006**	**0.0001**	0.20	**0.00002**	**0.01**
n_C	6 of 21	11 of 21	12 of 21	6 of 21	4 of 10	14 of 21	-	11 of 15	6 of 15
Rank									
1.	* *E. ang.* 0.43 (6) {-(0)}	* Phr. *E. vag.* 0.33 (7) {0.24 (2)}	* Phr. *E. vag.* 0.71 (7) {0.52 (1)}	* Phr. *E. vag.* 2.05 (7) {0.17 (2)}	Phr. *E. vag.* 0.62 (7) {0.03 (1)}	* Phr. *E. vag.* 1.60 (7) {0.33 (1)}	*E. ang.* 2.20 (6) {-(0)}	* Vad. *E. vag.* 0.61 (8) {0.59 (5)}	* *E. ang.* 3.22 (6) {-(0)}
2.	* Phr. *E. vag.* 0.14 (7) {0.06 (1)}	* *E. ang.* 0.33 (6) {-(0)}	* *E. ang.* 0.50 (6) {-(0)}	* *E. ang.* 1.94 (6) {-(0)}	*E. ang.* 0.62 (6) {-(0)}	* *E. ang.* 0.74 (6) {-(0)}	Phr. *E. vag.* 1.54 (7) {0.20 (2)}	* Phr. *E. vag.* 0.45 (7) {0.19 (1)}	* Vad. *E. vag.* 0.30 (7) {0.31 (5)}
3.	Vad. *E. vag.* 0.11 (8) {0.11 (8)}	* Pond 0.13 (2) {-(0)}	Lake 0.41 (4) {-(0)}	Vad. *E. vag.* 0.21 (7) {0.10 (5)}	**Mire** 0.26 (7) {0.06 (2)}	* Vad. *E. vag.* 0.56 (8) {0.43 (5)}	Lake 0.82 (1) {-(0)}	* *E. ang.* 0.22 (6) {-(0)}	* Phr. *E. vag.* 0.16 (7) {0.07 (2)}
4.	Pond 0.09 (2) {-(0)}	Vad. *E. vag.* 0.05 (7) {0.01 (5)}	* Pond 0.38 (2) {-(0)}	Pond 0.08 (2) {-(0)}	Vad. *E. vag.* 0.20 (8) {0.15 (5)}	**Mire** 0.06 (7) {0.00 (2)}	**Mire** 0.32 (7) {0.09 (2)}	**Mire** 0.03 (7) {0.01 (2)}	Lake 0.02 (2) {-(0)}
5.	* Lake 0.09 (2) {-(0)}	**Mire** 0.02 (7) {0.02 (2)}	Vad. *E. vag.* 0.05 (8) {0.03 (5)}	**Mire** 0.03 (7) {0.06 (2)}	* Bare peat 0.00 (8) {0.00 (8)}	Pond 0.01 (2) {-(0)}	Vad. *E. vag.* 0.31 (7) {0.11 (5)}	Lake 0.01 (2) {-(0)}	Bare peat 0.00 (8) {0.00 (8)}
6.	Bare peat 0.03 (8) {0.03 (8)}	Lake 0.01 (2) {-(0)}	**Mire** 0.03 (7) {0.01 (2)}	Lake 0.01 (1) {-(0)}	-	* Bare peat 0.00 (8) {0.00 (8)}	Bare peat 0.27 (8) {0.27 (8)}	* Bare peat 0.00 (8) {0.00 (8)}	**Mire** -0.21 (7) {0.04 (2)}
7.	**Mire** 0.00 (5) {0.00 (5)}	Bare peat 0.00 (8) {0.00 (8)}	* Bare peat 0.00 (8) {0.00 (8)}	* Bare peat 0.00 (8) {0.00 (8)}	-	Lake -0.01 (2) {-(0)}	Pond 0.02 (2) {-(0)}	-	-

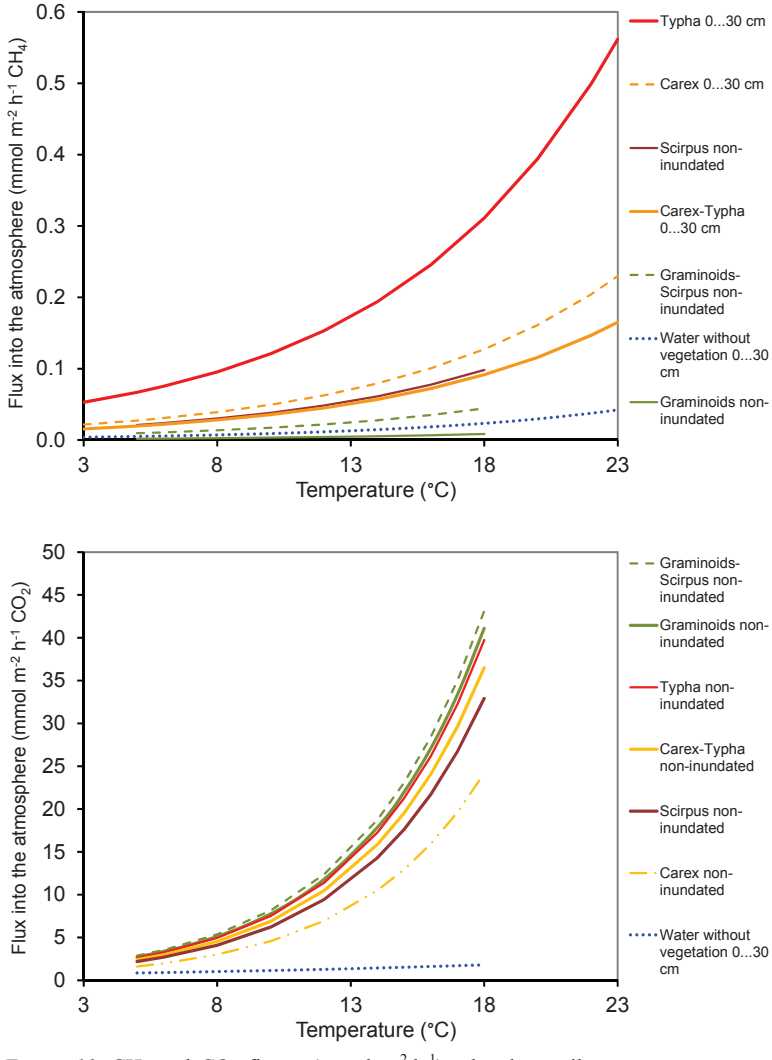


Figure 10. CH_4 and CO_2 fluxes (mmol m^{-2} h^{-1}) related to soil or water temperature (°C) under "normal" and "dry" conditions, respectively. Fluxes estimated with the linear mixed effects model (Tables 1 & 2 in Paper III) based on measured temperature ranges.

anaerobic wetland soil to the atmosphere (Frenzel & Rudolph 1998). Due to the CH_4 channelling through the tissues of these plants, CH_4 oxidation can be avoided because the pathway through the upper aerobic peat horizons is bypassed (Günther *et al.* 2013).

Fluxes of CO_2 and CH_4 in 'dry' conditions

In 'dry' conditions, sites with typical wetland plants started to emit more CO_2 and CH_4 than under inundated conditions. The increase in CO_2 fluxes from *Carex* spp. and *Thypha latifolia* during 'dry' conditions can be explained by the additional impact of the aerated soil and thus stimulated soil respiration that is otherwise hidden by the overlying water body. Furthermore, as the whole plant is above the surface water table under 'dry' conditions, the larger leaf area has an impact on plant-mediated CO_2 respiration due to a higher photosynthesis rate. Higher soil temperatures also stimulate CH_4 production (Schulz *et al.* 1997) and thus CH_4 emissions can be increased (Bergström *et al.* 2007). Furthermore, if the roots are still in contact with CH_4 depots in the anaerobic peat horizons, the plant-mediated CH_4 flux to the atmosphere is still active (Frenzel & Rudolph 1998) even when the uppermost peat horizon is aerated.

Fluxes of CO_2 and CH_4 in 'inundated' conditions

When all vegetation communities along the gradient were inundated, GHG fluxes to the atmosphere were reduced, even though CH_4 fluxes from the *Graminoids* and *Scirpus* spp. communities were somewhat higher due to available fresh organic material. Hahn-Schöfl *et al.* (2011) also found high CH_4 emissions from highly productive *Phalaris arundinacea* stands after rewetting a fen in Northern Germany. The frequent water level fluctuations in Västkärr can be seen as small rewetting events at the otherwise dry shore zone of a constructed shallow lake that may result in a change among vegetation communities under inundation, but also a re-change under drier conditions. All *Graminoids* at the Västkärr site would die off under longer lasting inundation, because they are not adapted for a life in standing water, and would simultaneously provide labile C for methanogenesis through input of labile C from the dying plant tissues (Hahn *et al.* 2015).

Ratio of CH_4 to CO_2

Once rewetting has created a new hydro-environment that may lead to a growing mire and to a C accumulation ecosystem, the GHG balance should be estimated. Expressed in CO_2 mol-equivalents, the ratio of CH_4 to CO_2 fluxes to the atmosphere at the Porla site (Paper II) and the Västkärr site (Paper III) was

less than 3.9 and 1.1, respectively, for 90 % of all flux measurement pairs, less than 2.0 and 0.6, respectively, for 80 % of pairs and less than 0.3 and 0.1, respectively, for 50 % of pairs, respectively. The overall climate impact of CH_4 fluxes to the atmosphere at both study sites was less than the impact of soil and plant respiration. Taking into consideration that Wilson *et al.* (2016a) found a negative GHG balance (net uptake) for *Eriophorum angustifolium* in a rewetted peatland, it can be speculated that CO_2 uptake by photosynthesis exceeds the GHG fluxes to the atmosphere in the *Eriophorum angustifolium* ecotope, which was the highest CH_4 emitter ecotope in Paper II. For restored peatlands in Canada, Strack *et al.* (2016) found a transition from sources to sinks of CO_2 with increasing vascular plant cover. Since the study site in Västkärr is characterised as fen without any bare peat surface, the vascular plant cover is much higher there than at Porla. This accounts for the higher ecosystem respiration (Figure 10) compared with the results in Paper II, but also suggests higher CO_2 uptake by photosynthesis. Due to the nutrient-rich status of the Västkärr peatland, higher CH_4 and N_2O fluxes could be expected from there compared with the nutrient-poor Porla peatland (Paper II). However, the fluxes from the different vegetation communities in Västkärr were of the same magnitude as from the lake, the mire, the bare peat and the vadose *Eriophorum vaginatum* in Porla (Figures 6 & 7). Even CH_4 fluxes from *Thypha latifolia* under 'normal' (inundated) conditions were smaller than from *Eriophorum angustifolium* and phreatic *Eriophorum vaginatum*.

As also found by other authors (Rõõm *et al.* 2014, Wilson *et al*. 2016b), the results from the studies in Porla and Västkärr indicated that rewetted peatlands are very sensitive to inner- and inter-annual changes in weather conditions and thus may be a GHG sink in one year and a source in the next, e.g. if exposed to drought.

Vial sampling strategy for CH_4 and N_2O

Taking the first headspace air sample into a vial 5 min, 10 min or even later after chamber closure is not unusual (e.g. Liu *et al.* 2014) and offers the possibility to sample quasi-simultaneously with several chambers. This leads to a long extrapolation distance between chamber closure and the first air sample that affects the quality of the CH_4 and N_2O flux estimates obtained because the centroid of the regression line is shifted to times where the assumption of an initial (quasi-)linear concentration increase is less valid. Thus, it is recommended to take the first air sample quickly after chamber closure (Rochette & Eriksen-Hamel 2008, Parkin & Venterea 2010, de Klein & Harvey (eds.) 2012). For that reason, the time schedule for vial sampling at the

Västkärr study site was improved in 2014, with the first headspace air sample taken at 1 min after chamber closure.

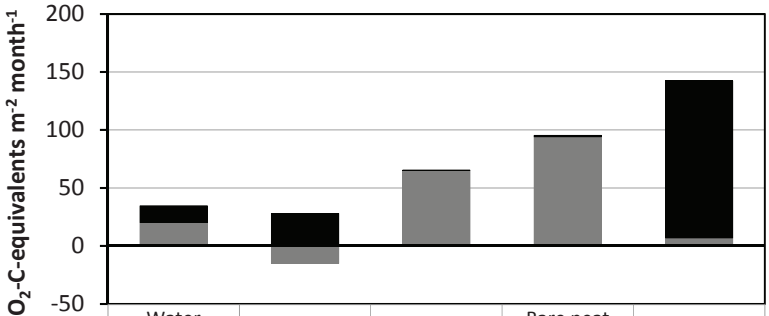
5.3 Automated transparent and manual opaque chambers with continuous determination of CO_2 and CH_4 (Paper IV)

The studies reported in Papers II & III provided an overview of the site-specific differences in GHG fluxes to the atmosphere between various ecotopes and vegetation communities, but they did not include diurnal patterns of the fluxes and uptake of CO_2 by photosynthesis. The specific objective in Paper IV was to investigate some CO_2 and CO_4 flux dynamics during one summer from different ecosystems created after peatland rewetting (Porla) and to detect short-term events such as ebullition. Flux measurements with the automated transparent chambers also included the uptake of CO_2 by photosynthesis.

5.3.1 CO_2 and CH_4 emissions from several ecotopes in the nutrient-poor Porla peatland during summer 2015

Several ecotopes along the shore of a constructed lake in a rewetted peatland (Porla) were investigated for GHG flux measurements during the summer 2015. The monthly net CO_2 fluxes from the different ecotopes showed a typical pattern: CO_2 emissions were highest from the 'bare peat' and the 'bare peat with *Eriophorum vaginatum* tussocks' ecotopes. Actually, when measuring with transparent chambers, lower CO_2 emissions were expected from the 'bare peat with *Eriophorum vaginatum* tussocks' than from the 'bare peat', due to higher CO_2 uptake by photosynthesis. However, this proved not to be the case, which might be explained by autotrophic and heterotrophic respiration from the peat around the '*Eriophorum vaginatum* tussocks'. Furthermore, the input of fresh organic matter, and thus its decomposition, can also lead to higher CO_2 emission rates (priming effect; Kuzyakov 2002). This might also explain the unexpectedly high CO_2 emissions from the 'water between *Eriophorum vaginatum* tussocks' in July and August. The '*Eriophorum vaginatum* tussocks in water' and 'fresh *Sphagnum* spp. in water' indicated CO_2 uptake due to photosynthesis: the *Sphagnum* spp. site was a clear CO_2 sink in all three months, whereas the '*Eriophorum vaginatum* tussocks in water' was a small CO_2 source in June (Figure 11) and July but a CO_2 sink in August. The CH_4 emissions were highest from the '*Eriophorum vaginatum* tussocks in water' due to the plant-mediated CH_4 transport.

In Paper IV, the ecotopes in the water were not as homogeneous as the ecotopes in Paper II. The 'fresh *Sphagnum* spp. in water' ecotope may contain some young single *Eriophorum vaginatum* leaves and may be influenced by



	Water between Eriophorum vaginatum tussocks	Fresh Sphagnum spp. in water	Bare peat	Bare peat with Eriophorum vaginatum tussocks	Eriophorum vaginatum tussocks in water
CH4	14	28	0	1	135
CO2	20	-15	65	94	7

Figure 11. Net emissions of CO_2 and CH_4 (g C-eq m^{-2} $month^{-1}$) from the five ecotopes in June 2015.

Eriophorum vaginatum roots as well. That would explain the higher CH_4 emissions compared with the pure *Sphagnum* spp. sites in the 'mire' in Paper II, where plant-mediated CH_4 transport was minimised due to the absence of vascular plants.

Only the 'bare peat' and the 'bare peat with *Eriophorum vaginatum* tussocks' consisted of a similar ecotope pattern in Papers II & IV and they were therefore comparable with each other. The 'fresh *Sphagnum* spp. in water' ecotope should not be compared with the (*Sphagnum*) 'mire' in Paper II due to their differences in formation: newly established *Sphagnum* spp. after rewetting compared with 'old' *Sphagnum* spp. in a growing mire. '*Eriophorum vaginatum* tussocks in water' were not found in such a dense cover in Paper II and thus neither was the 'water between *Eriophorum vaginatum* tussocks'. However, the average CH_4 fluxes (using the example of June 2015) for the ecotopes listed from left to right in Figure 11 would be 0.13, 0.26, 0.00, 0.14 and 1.27 mmol m^{-2} h^{-1}, respectively. They are thus within the range found in Paper II, even though nocturnal conditions were simulated when measuring with opaque chambers.

GHG emissions from the peatlands studied varied between years and their function ranged from sink to source and back. For the Porla study site, there was variation in CO_2 and CH_4 emissions between the three summer months, where the system was predominantly affected by net CO_2 flux in June and July, while CH_4 emissions dominated the fluxes in August. July received slightly

more precipitation than June and August and this might explain the decreased CO_2 emissions from the 'bare peat' and the 'bare peat with *Eriophorum vaginatum* tussocks' ecotopes. As long as large areas of bare peat exist, ecosystem respiration may govern the GHG fluxes in a rewetted peatland.

Seventeen years after re-wetting, the *Sphagnum* spp. cover has increased on the loose peat shores of Lake A and B in Porla, but the surface area covered with *Sphagnum* spp. is still relatively small compared with the bare peat and *Eriophorum vaginatum* areas, and thus the ecosystem is still a net GHG emitter. The first signs of development of a C accumulation ecosystem are already there and the terrestrialisation of the water body with wetland plants, such as *Eriophorum* spp. and *Sphagnum* spp., is progressing well. Creating a C source by drainage is a fast process, but creating a C sink by rewetting may take decades.

5.3.2 Visual inspection of short-term events: ebullition

Ebullition is a non-diffusive emission process that can be divided into steady and episodic ebullition (Lai 2009, Green & Baird 2013). Steady ebullition can often be constant enough to be correctly measured with discontinuous sampling (e.g. with vials) of headspace air during a chamber enclosure period < 60 min, but episodic ebullition, which is obviously non-constant over chamber enclosure, cannot easily be recorded by a vial sampling method. In Papers II & III, CH_4 was routinely determined by vial sampling at time intervals of 10 min. Thus, relatively short-term bubble emissions were integrated (masked) in relatively long-term averages of increasing headspace air concentration. Therefore, an initial CH_4 bubble probably generally resulted in strong underestimation of the real mean CH_4 emission flux, while an irregular series of single bubbles most likely resulted in different kinds of under- or over-estimation of the real mean CH_4 emission flux by the routine flux measurement method used. Observations of CH_4 headspace concentration, measured by infrared laser absorption with manual opaque chambers at the Porla study site, revealed that ebullition mostly occurred as a result of the shock caused by chamber closure on water-saturated soil and due to disturbances by the operator (Figure 12), or as an irregular series of single bubble emissions resulting in a step-like curvature of headspace concentration on a level high above the ambient concentration (Figures 13 & 14). In the Porla rewetted peatland, it was found that bubble ebullition often released CH_4 and CO_2 simultaneously to the atmosphere (Figures 12-14), as reported previously for rice paddies by Komiya *et al.* (2015). Furthermore, the most striking finding was the similar CO_2 and CH_4 emission patterns (Figure 14).

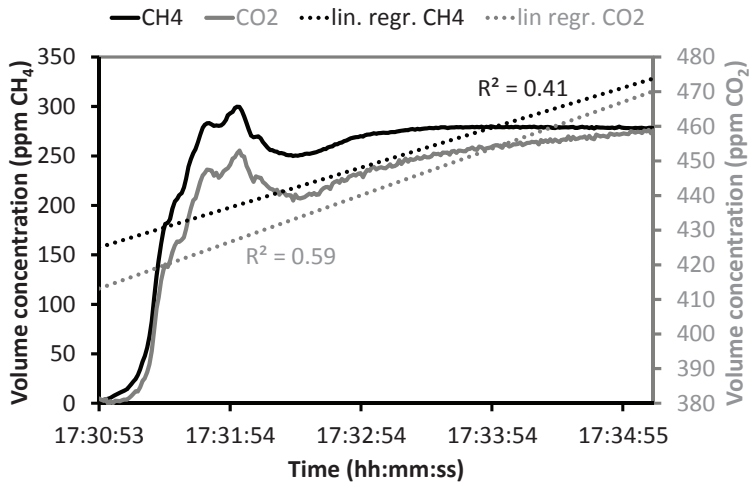


Figure 12. Changes in CH_4 and CO_2 concentrations (1-s values) in the headspace of a manual opaque chamber placed over *Eriophorum angustifolium* in the littoral zone of lake B on 27 August 2013.

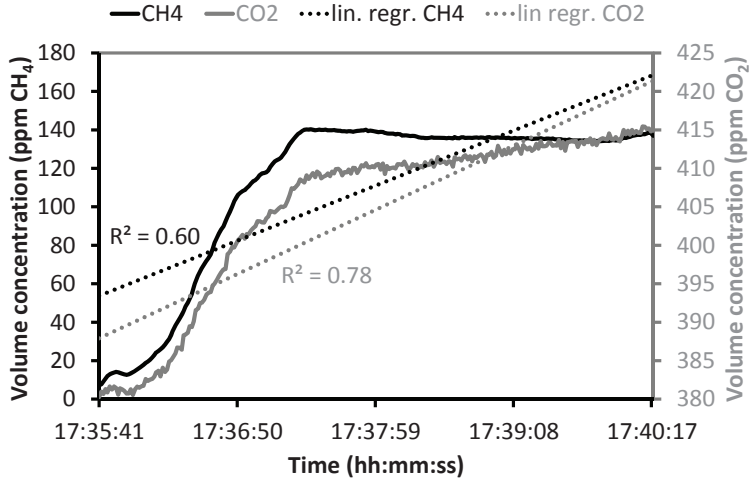


Figure 13. Changes in CH_4 and CO_2 concentrations (1-s values) in the headspace of a manual opaque chamber placed over *Eriophorum angustifolium* in the littoral zone of lake B on 27 August 2013.

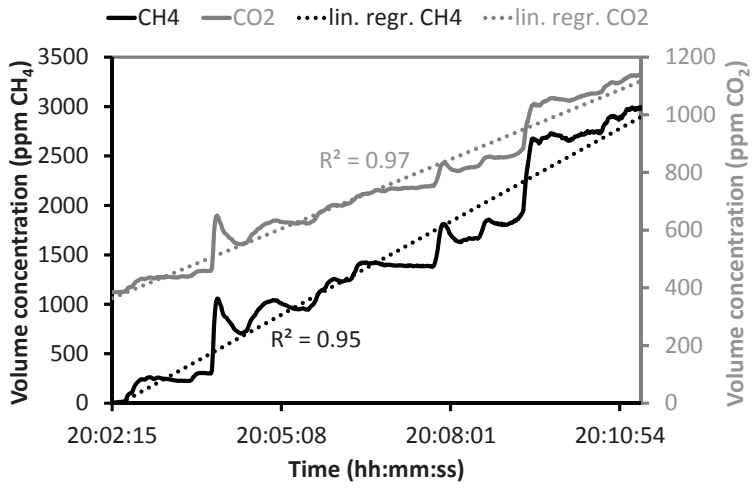


Figure 14. Example of changes in CH_4 and CO_2 concentrations (1-s values) in the headspace of a manual opaque chamber placed over *Eriophorum angustifolium* in the littoral zone of lake B on 28 June 2013.

Most often, vial samples are taken during the daytime. Thus, higher night time concentrations of both gases will be omitted. The next example illustrates the change in CH_4 and CO_2 concentration for 234 consecutive measurements from a single chamber over the observation period of eight days (Figure 15). The CH_4 and CO_2 concentrations followed a diurnal fluctuation cycle with higher concentrations for both gases occurring during night time and for CO_2 in the morning. The concentration of the two gases was noticeably high before and at the time of chamber closure. The higher concentrations during the nights could be explained by boundary layer effects in plants and chamber frames, microscale cold air layers, and katabatic winds. Koskinen *et al*. (2014) had a similar observation for CO_2. CH_4 (and CO_2) ebullition peaks occurred unsystematically.

Therefore, high temporal resolutions of GHG fluxes are needed for providing complete data sets as a basis for discussions about ecosystems and their GHG emission impact in a changing climate, e.g. within the framework of national climate reporting. More research on ebullition from different peatland ecosystems and water bodies is needed.

If CH_4 ebullition occurs, attention should also be given to CO_2 since it has been shown that bubble ebullition often releases CH_4 and CO_2 simultaneously. Furthermore, CO_2 and CH_4 emission patterns were often greatly similar and thus, similar criteria for ebullition detection might be used for both gases.

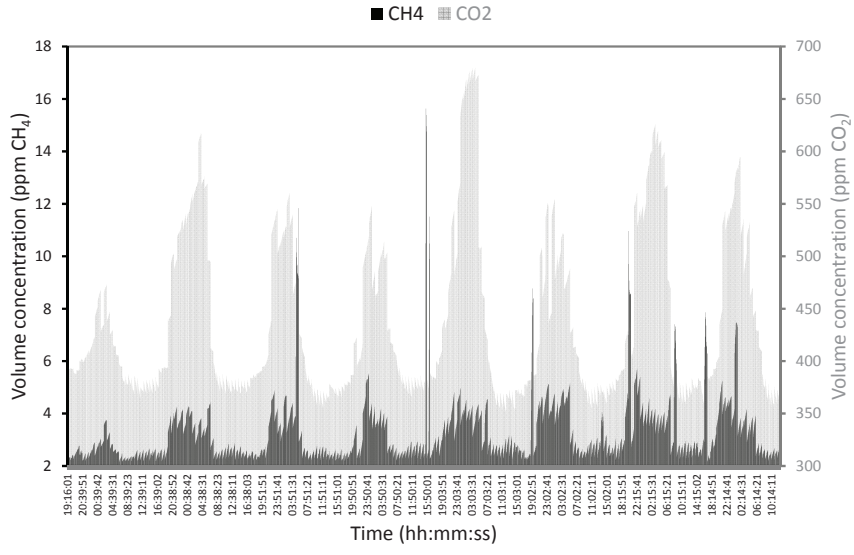


Figure 15. CH_4 concentration in 234 consecutive measurements (2807 measurement values) for one and the same chamber (2 min measuring time; 48 min break between each measurement); measured with a transparent automated chamber in an *Eriophorum vaginatum* inter-tussock (water) with some fresh *Sphagnum* spp. (3 to 11 July 2015). Note: Data for the first minute of chamber closure are not shown here.

If CO_2 and CH_4 are measured independently, e.g. with IRGA over a short chamber closure period and with vials over a longer chamber closure period, both measurement methods should be double-checked for ebullition. This could explain 'strange' emission patterns in each of the gases when combining their evaluation.

Episodic ebullition and how it is calculated are still insufficiently investigated, but it is likely that ebullition results in different forms of under- or over-estimation of the real mean CH_4 (and CO_2) emission flux. This should be kept in mind when modelling GHG scenarios.

6 Conclusions and recommendations

Wetland restoration at the Porla and Västkärr sites was successful, as two functioning wetland ecosystems with stable hydrology have established after 17 years since rewetting and characteristic peatland vegetation have developed. This is an important prerequisite for new peat growth and a future C sink. In particular, *Sphagnum* spp. cover started to establish on the loose peat shores of the two lakes in Porla. Simultaneously, *Eriophorum angustifolium* has vanished from nutrient-poor parts but has expanded in nutrient-richer parts (formed by blocked drainage ditches). Thus restoration is meeting the goal of *Sphagnum* spp. accumulation, a possible first step towards natural mire conditions.

Lake construction may eventually result in a CH_4 hotspot in littoral zones, but the transition from extracted peatland to wetland ecosystem may reduce the overall climate warming impact. Establishment of vascular and pioneer plants, such as *Eriophorum* spp., *Carex* spp., *Typha* spp. and *Phragmites* spp. in the littoral zone and along the lake shore is essential in paving the way for other peatland vegetation. If the shallow lake develops into bog or poor fen with *Sphagnum* spp. as the main vegetation form, CH_4 fluxes to the atmosphere from vascular plants directly after rewetting would be balanced by CO_2 uptake some years after rewetting, even though *Sphagnum* spp. lawns emit some CH_4. Furthermore, CO_2 fluxes to the atmosphere from ongoing peat extraction sites could be balanced by rewetting of abandoned sites. More data are now needed for water bodies in peatlands in general and for shallow lake ecosystems in rewetted peatlands in particular.

However, the overall climate impact of CH_4 emissions from the study areas did in general not exceed the impact of soil and plant respiration (paper II & III) and neither the net CO_2 flux in two of three months during summer 2015. But, GHG emissions could vary between years and sites can shift from sinks to sources. In regards to management of extracted peatlands, the construction of

shallow lakes showed great potential for lowering GHG fluxes to the atmosphere.

Anyway, rewetting should be seen as a balancing act between two faces: what is good for one environment, such as shaping new peatlands for C sinks, might be bad for another environment, such as peatland adjacent ecosystems that can suffer from (high) P leaching. The P eutrophication risk for associated aquatic ecosystems might be higher, especially in the first phase directly after rewetting, from rewetted former agricultural used peatlands as from extracted peatlands but this fact should keep in mind when planning to rewet a drained peatland for after-use issues. However, these impacts could change from an initial phase as the wetlands in the long-term perspective develop into mires.

Manual and automated closed chambers are a suitable method for measuring small-scale spatial variations of GHG fluxes to the atmosphere even though they have some impact to the soil-atmosphere-interface. Additionally, methods that to a lesser extent interfere with the measuring surface, such as open chambers, gradient methods or EC technique, could be employed. But here, other things, such as the loss of small-scale spatial variations or a power supply, have to be considered. With continuous measurements a derivation of long-term gas balances can be achieved and short-term changes in environmental conditions influencing GHG exchanges can be detected more effectively as with discontinuous measurements, such as by vial sampling. But still, a correct indication of all GHG fluxes, e.g. for GHG upscaling purposes or national emission inventories, is strongly based on the correct estimation of all C fluxes including ebullition.

7 Sammanfattning (Swedish summary)

Torvbruk för odlings- och energitorv har lång tradition i Nordeuropa och pågår på en täkt ofta under ca 20 år. Dränering genom dikning är oftast nödvändig och denna påverkar växthusgasbalansen genom genomluftning och ökad nedbrytning av torven och har även betydelse för denitrifikationen. Emissioner av koldioxid (CO_2) och lustgas (N_2O) ökar men metanemissionerna (CH_4) kan minska.

Efter avslutad torvtäkt ska området föras över till annan användning och skogsproduktion eller våtmark är de två vanligaste efterbehandlingarna. Restaurering av torvbildningsmiljön till ny våtmark kan bidra till förbättrad biodiversitet, bidra i kollagring till fördel för minskad klimatändring och möjligen till ändrad vattenkvalitet. Sådan restaurering åstadkoms genom återvätning av det dränerade området och målet är att nå en fungerande våtmarkshydrologi. I förlängningen kan den nya miljön återskapa de för torvmarker vanliga processerna under anaeroba förhållanden, bli en lämplig miljö för torvmarksbildande vegetation och bilda en kolsänka.

Undersökningar om effekter av återvätning på växthusgasbalanser är fortfarande begränsad men det finns en del studier av våtmarker tidigare nyttjade inom jordbruket. I huvudsak kan sägas att återvätning minskar CO_2 och N_2O emissioner medan CH_4 emissionerna ökar. Det behövs mer studier av växthusgasemissioner från återvätta avslutade torvtäkter och från grunda sjöar omgivna av vegetationsbeklädda strandzoner, som visat sig vara betydande miljöer för utsläpp av växthusgaser. Därför startade undersökningar av växthusgaser i två torvmarker återvätta efter torvutvinning.

Från genomförda undersökningar i dessa två våtmarker visas hur ecotoper med varierande egenskaper påverkar emissioner och balanser av växthusgaser. Betydelsen av vattenförhållanden, temperatur och vegetation studerades med manuella mätningar i mörka kamrar och i ett automatiserat system med transparenta kamrar där såväl fotosyntes som emissioner kunde bestämmas.

En stor variation mellan år noterades och det verkar som om CH_4 emissionerna inte är större än CO_2 flödena under sommaren. Tillskapandet av grunda sjöar visar på lägre utsläpp av växthusgaser till atmosfären. Vegetationstäckta vattenområden i strandzoner kan dock ha en stor betydelse med större emissioner. Nyttjandet av automatiska system med kontinuerliga mätningar adderar starkt till betydelsen av korttidsfluktuationer i inbindning och emissioner och har betydelse för balanserna även om den nuvarande rapporteringen av växthusgaser övergripande kan vara rimlig.

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