Roskilde University

MASTER'S THESIS

Northern Peatlands and their Feedback to Climate Change

A microcosm study to evaluate the effect of temperature increase and water-table drawdown on the emission of greenhouse gases from a Danish Raised Bog

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This is dedicated to my parents Md. Fakhrul Alam and Momtaz Shirin- the purest sources of my motivation all along

Abstract

Northern peatlands are considered crucial components in the global carbon (C) budget as well as in the global budget of greenhouse gases (GHGs) like carbon dioxide (CO_2) and methane (CH_4). These ecosystems have accumulated a large amount of atmospheric C over the Holocene and acted as a sink of atmospheric CO_2 on a long term but as a source of CH_4 on short time horizons. But the biogeochemical processes associated with the ecosystems' carbon and nitrogen (N) cycling can be significantly affected by the climatic factors- temperature and evapotranspiration. Consequently the function as CH_4 source and as CO_2 sink is also temperature and water level dependent. The increased air temperatures and evapotranspiration due to climate change, are therefore, frequently assumed as significant external climate forcings that can alter the GHG emission levels in northern bogs that are originally adapted to a more cold and humid climate. With a view to understand the impact of increased air temperatures and water level reductions on a northern bog's CH_4 and CO_2 emission levels, the current project conducts a laboratory experiment where 16 semi-intact peat core samples differentiated by high water and low water levels and three intact background core samples are incubated at three different temperature levels: $4°C$, $17°C$ and $25°C$. Soil samples are collected from a Danish raised bog called Holmegård and the experiment covers a total period of 28 days. The underlying assumption of the experiment is higher temperature will cause higher CH_4 and CO_2 fluxes but lower water table will cause lower CH_4 and higher CO_2 fluxes. At $4°C$ temperature, CH_4 emissions are considerably low in semi-intact cores than the intact-cores. At $17°C$ LW cores have produced more CH_4 and CO_2 emission than the HW cores. At $25°C$ both CH_4 and CO_2 fluxes have increase but the emission of LW cores are still higher than the HW cores. Compared to the expected range of emission at high temperatures, the CH_4 and CO_2 flux obtained have been low. Based on the flux data no significant trend of change in the CH_4 and CO_2 emissions with increased temperatures and reduced water levels can be derived due to the limited number of samples and length of measurement time. Finally, sectors of further studies are recommended based on the outcomes and limitations of the project.

Preface

The thesis is submitted as a partial requirement for the degree in Master's of Science in Environmental Risk under the department of Science and Environment at Roskilde University (RUC) in the year of 2018. The research has been conducted under the direct supervision of Lecturer Lauren Paige Seaby and co-supervised by Professor Gary Thomas Banta in the department of Science and Environment, RUC.

The choice of the thesis has been solely derived from my genuine passion for adding improved knowledge and data in the field of environmental and climatic research. Being a geographer by background, in this thesis, I have tried to present a holistic view of the prevailing and predicted interactions between peat ecosystem, man and climate components with a particular focus on greenhouse gases.

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Nazia Tabassum

Acknowledgement

I extend my sincere appreciation and gratitude to all people who made this work possible. First I would like to thank my supervisor Lauren Paige Seaby, Lecturer, Department of Science and Environment, Roskilde University whose intellectual vision and guideline have been a constant source of my encouragement and inspiration. She has guided me in formulating research plans, understanding thesis subtitles and presenting a comprehensive report. She was always insightful, available and looking out for the solutions to my problems. In addition, I want to thank Professor Gary Thomas Banta, Department of Science and Environment, RUC, whose deep insight and cordial assistance have greatly helped me throughout the experiment of the study. A very special gratitude of mine goes to Professor Niels H. Jensen, Department of Science and Environment, RUC who has been an immense support during the collection of samples from the Holmegård bog. He has also been a great adviser on the entire area of Holmegård as well as on the bog ecosystem. I would also like to thank all of highly skilled laboratory technicians specially Mr. Torben Brandt Knudsen who has assisted me a lot with the gas chromatography machine. Last but not the least I am especially grateful to my husband and family for giving me all their support and love all the time.

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Abbreviations

BD	=	Bulk Density
DMI	=	Danish Meteorological Institute
DOC	=	Dissolved Organic Carbon
EU	=	European Union
FDH	=	Formate Dehydrogenase
GHG	=	Greenhouse Gas
HW	=	High Water
ID	=	Inner Diameter
IR	=	Infrared Analyzer
IPCC	=	Intergovernmental Panel on Climate Change
IPS	=	International Peatland Society
LGM	=	Last Glacial Maximum
LOI	=	Loss on Ignition
LW	=	Low Water
OM	=	Organic Matter
PAS	=	Photo-Acoustic System
PVC	=	Polyvinyl Chloride
SD	=	Standard Deviation
WT	=	Water Table

Chapter 1

Introduction

1.1 Motivation of the Study

Peatlands- a substantial store and sink of terrestrial and atmospheric carbon (C), are often considered as crucial components in the global C cycle. According to the International Peatland Society (IPS)[1], peatlands are wetland ecosystems that are developed by the accumulation of partially decomposed organic matter in the form of a heterogeneous complex called peat. In these wetlands, environmental conditions such as water-saturation, high acidity (low pH) and prolonged absence of oxygen induce a slow and reduced mineralization of organic matter that eventually causes gradual sequestration of C in accumulating peat soils following the photosynthetic removal of carbon dioxide (CO_2) from the atmosphere (Frolking et al., 2006; Höll et al., 2009; Maltby, 2009; Clark et al., 2009). The regulation of biogeochemical processes, particularly in northern peatlands, relies on their thermal and hydrological regime that are originally supported by cold climate induced low temperatures and high water table conditions. Therefore, the C storage and cycling functions of these ecosystems are sensitive towards changes in these conditions i.e. elevated temperatures or lowering of the water table.

The characteristic cold and wet climate of the boreal and temperate ecosystems, being conducive to the limited decay of organic matter, have led to the occurrence of typical northern peatlands (Joosten and Clarke, 2002; Leifeld and Menichetti, 2018; Olefeldt et al., 2017). Northern peatlands together constitute the largest share (over 90% of global peatlands[1] and contribute to a total storage of nearly $550\ gigatons\ (Gt)$ of organic C (Essl et al., 2012; Charman et al.,

[1] International Peatland Society or IPS is an international organisation located in Finland. It promotes the wise use and management of peatlands and serves the people involved in the sector by organizing conferences and workshops as well as by publishing research papers on the key issues of climate change, biodiversity and the need for responsible use and restoration of peatlands. Website address: `http://www.peatsociety.org/`, accessed on June 14, 2018

2013). Most of these peatlands have accumulated C since the Last Glacial Maximum (LGM) and are found to be acting as small but constant sinks for atmospheric CO_2 (Roulet, 2000; Frolking et al., 2006; Charman et al., 2013) over the last $12,000$ years (Weedon et al., 2013). At the same time, due to water-filled anoxic soil pores or low redox potentials[2] these natural wetlands also release a considerable amount of methane (CH_4) and nitrous oxide (N_2O) into the atmosphere (Schaufler et al., 2010; Tokarz and Urban, 2015; Wang et al., 2017a). Any changes to the C and N balance of peatlands have impacts on the global budget of atmospheric greenhouse gases (GHGs) e.g. CO_2, CH_4 and N_2O (Elberling et al., 2011; Wang et al., 2017a; Kløve et al., 2017; Weedon et al., 2013) that are evidently potent to contribute to the warming of climate by increasing surface air temperatures (Nazarenko et al., 2015; Oreskes, 2018). Therefore the nature of feedback between peatland biogeochemical cycles and climate has significant implications in global warming as well as in future climate change.

So far, the Earth's climate or radiative budget has been affected by northern peatlands in a dual manner. As a sink of C, peatlands reduce the atmospheric CO_2 burden that provides the climate with a cooling effect by inducing the negative radiative forcing of climate[3] (Frolking et al., 2006; Wang et al., 2017a). They are also found to uptake about 12% of current human emissions (Moore, 2002) that again adds to the result of reduced climate warming. On the contrary, northern peatlands are persistent sources of CH_4 to the atmosphere that causes a warming effect or positive radiative forcing of climate (Frolking et al., 2006; Wang et al., 2017a). On a centennial scale, CH_4 has a 25-times greater global warming potential (Solomon et al., 2007; Askaer et al., 2011) than CO_2 but the half life of CH_4 in the atmosphere is lower than CO_2 (Olefeldt et al., 2017). Therefore, with the highest C accumulation in the early Holocene (Yu et al., 2010), the net result of the peatlands' two-way radiative forcings throughout the epoch (more than a 1000-year time horizon) has been negative with a long-term overall cooling effect on the climate (Frolking and Roulet, 2007; Charman et al., 2013; Gill et al., 2017). However, evaluating the current and future atmospheric burdens of CO_2 and CH_4 produced by the peatlands is not straightforward since there is a high variability in their CO_2 uptake and CH_4 emission levels on spatial and temporal scales (Roulet, 2000; Frolking et al., 2006; Wu and Roulet, 2014; Lees et al., 2018).

It is often hypothesised and in lot of cases found that perturbations in northern peatlands' bio-

[2]Redox potential or oxidation-reduction potential (Eh) indicates the measure of reduction of soil and the number of free electrons that is exchanged in the redox reactions. It is expressed in millivolts (mV) or volts (V) (Tokarz and Urban, 2015).

[3]Radiative forcings of climate refer to the externally forced perturbations in the Earth's radiative energy budget. Changes in the concentrations of radiatively active components like CO_2 or changes in the Earth's albedo can be considered as radiative forcings of climate (Ramaswamy et al., 2001)

geochemical processes as well as their functions as net sinks or sources of GHGs can be instigated by the differential practices of landuse as well as by the impacts of climate change, particularly, increases in temperature and evapotranspiration (Roulet, 2000; Strack et al., 2004; Frolking et al., 2006; Abdalla et al., 2016; Kløve et al., 2017). In the following 50 to 100 years, mean global temperatures have been estimated to increase by 1.0 to $3.5°C$ due to climate change that is essentially driven by GHG emissions (McCarthy et al., 2001; ACIA, 2004; Weedon et al., 2013).

Throughout the last century, northern boreal ecosystems have showed a rate of warming twice as fast as the global average (Bekryaev et al., 2010; Gill et al., 2017). As a consequence of this fact, it is frequently assumed that higher temperatures can potentially turn the peatlands into net emitters of CO_2 and induce losses of stored C into the atmosphere by accelerating the rate of microbial peat decay (Charman et al., 2013; Leifeld and Menichetti, 2018). By intensifying evapotranspiration, projected temperature increases may also lead to the lowering of water tables (WT)[4] in many of these wet systems (Strack et al., 2004). In a healthy and intact peatland, the WT usually lies very close to the ground surface so that the high water level can facilitate the supply of chemical substrate to the microorganisms and suppress the rates of decomposition by maintaining soil temperature and oxygen diffusion rate as low as possible (Schaufler et al., 2010; Rydin and Jeglum, 2013). The reduction in WT can induce aerated and warm soil conditions and increase the rate of oxic mineralisation that may accelerate the emission of CO_2 (Laine et al., 1996; Kløve et al., 2017). It is already suggested by a climate simulation study (if only CO_2 emission is considered) that due to climate change northern peatlands will be reduced in their capacity to absorb atmospheric CO_2 and thereby, contribute to global warming throughout the 21st century (Wu and Roulet, 2014). Similarly, landuse practices, particularly draining of peat for agriculture, are also found to increase net CO_2 emissions by lowering the water level and infusing oxygen through the exposed peat soil (Carlson et al., 2017).

The microbially-mediated CH_4 production (methanogenesis) rates in peatlands may also expedite with the increasing temperatures and concentrations of atmospheric CO_2 (Lai, 2009; Gill et al., 2017). However, peatland CH_4 emissions are mostly positively correlated with higher water levels indicating that CH_4 emissions may decrease with the lowering of WT (Lai, 2009; Serrano-Silva et al., 2014; Abdalla et al., 2016). If the reduction in WT is a result of drainage, the CH_4 fluxes from peatland may decline by both increasing levels of CH_4 oxidation in the upper layers as well as by limiting anoxic CH_4 production in the deeper layers (Kløve et al., 2017).

[4]Water Table (WT) indicates the point at which hydraulic pressure in the soil gets equal to atmospheric pressure or the level at which water will stand in a well connected to the groundwater reservoir (Gilman et al., 1994)

As far as the emission of N_2O is concerned, intact peatlands can function as a sink for N_2O and the emission of N_2O from these peatlands are mostly negligible (Martikainen et al., 1993; Kløve et al., 2017). Like methanogenesis and decomposition, microbial processes associated with N_2O production (nitrification and denitrification) are also temperature sensitive (Martikainen et al., 1993). The reduction in moisture content due to increased evapotranspiration or the lowering of WT due to drainage or prolonged lack of precipitation in summer, can diffuse oxygen in the peat soil and decrease the level of N_2O emissions by inhibiting anaerobic denitrification (Schaufler et al., 2010). However, if the peatland is nutrient rich i.e. fen, nitrification produces NO_3^- in the presence of O_2 that can be utilized by microbes in denitrification and as an intermediate product of both nitrification and denitrification, N_2O is produced (Kløve et al., 2017; Liimatainen et al., 2018). As a result of global warming and peat degradation, increasing levels of CO_2, CH_4 and N_2O fluxes from the northern peatlands may cause a positive feedback to climate warming leading to the global positive carbon cycle feedback (Charman et al., 2013; Wang et al., 2017a). However, the future of northern peatland GHG dynamics is still uncertain as well as difficult to predict since their responses in a changing climate are not well-understood (Baird et al., 2009; Charman et al., 2013).

This research project is an attempt to understand the potential emission responses of northern peatlands to climate change by evaluating the effects of variable water levels and temperatures on the emission of GHGs from peat soils. In the study, Holmegård Mose or Holmegaard bog, a cool and temperate raised peatland in Denmark, has been chosen as the main area of investigation. Although the majority of the active peatlands in Denmark are lost by human interventions (Chapman et al., 2003), the Holmegård peatland located at about 60 km southwest of Copenhagen on the Sjælland island, still have certain areas that constitute almost pristine bog conditions. To the best knowledge of this author, so far there is no study on this area regarding its emission of GHGs nor any work about its emission responses in future climate change scenarios predicted for world's northwest regions e.g. increased summer temperatures and decreased summer precipitations (Mauquoy and Yeloff, 2008).

1.2 Research Objectives

With the aim of understanding the effect of future climate change on the emission of GHG gases from northern peatlands, the following research objectives are defined for this project:

- To investigate experimentally how the increase in air temperature and the drawdown of the water table affects the fluxes of GHGs derived from the soil matrix of Holmegård bog

- To assess the GHG emissions of Holmegård bog in the context of emission scenarios estimated for other northern peat systems under changing climate and hydrological regimes

1.3 Research Approach

With a view to attain the objectives, the entire project has been broadly designed with two approaches- the review of relevant literature and the execution of a laboratory experiment. The main purpose of the literature review is to conceptualise the background of the research problem as well as to formulate and justify the methods of the study. The experimental approach involves the measurement of CO_2 and CH_4 fluxes from experimentally reconstructed semi-intact peat cores that are treated under laboratory conditions of elevated temperatures and reduced water levels within a defined period of time. Besides the flux measurements, soil properties are also analysed in order to estimate the background chemical status of the peat soils used in the treatment cores. Upon completion of the experiment, CH_4 and CO_2 flux outputs and their reliability are evaluated in terms of methodological aspects as well as in the light of published emission data on northern peatlands.

1.4 Scope and Limitation

In the broader sense, the scope of the research lies in the field of predicting ecosystem feedbacks to climate change in terms of potent GHG emissions from soils. As future climate projections indicate higher summer temperatures and lower summer precipitations for the northern hemisphere, emission responses of a climatically vulnerable northern bog have significant implications in the global budget of GHGs as well as in the management and restoration policies of wetlands. However, the scope of the study has been limited by several practical challenges. The limited resources has led to certain methodological compromises and adjustments that restricts the number of experimental outcomes within the defined time-frame and thereby limits the opportunity of conclusive evaluations based on the data obtained. As for example, the experiment has been adjusted to take measurements from semi-intact soil cores due to the lack of a proper peat sampler which would produce intact cores. Moreover, as the method is unique to this study, it has not been possible to have any methodologically identical work to compare with in terms of associated positives and negatives.

Due to limited collection opportunity in the field, the degraded section of the bog could not be included into experimental treatments. This has confined the scope to analyze how GHG fluxes from an original bog are impacted by changes in climate conditions and excluded the possibility

of analysing how the intact bog may differ from its degraded counterparts. As the study has solely focused on soil-atmosphere gaseous exchange, the evident influences of bog vegetation on regulating GHG emissions to the atmosphere has also been excluded. Besides this, it has been confined to CO_2 and CH_4 gases only while soil emission of another potent GHG, N_2O, could not be measured in the experiment due to the high probability of N_2O emissions being undetectable in the equipments available at RUC.

1.5 Thesis Structure

The entire thesis has been structured with three broad chapters. Chapter-2 includes a detailed literature review where the functional aspects and gas dynamics of northern peatlands are discussed with a view to better understand their potential role in global warming. A section of methodological review is also added to the end of Chapter-2 so that the experimental approach described in Chapter-3 can be better understood. A thorough overview of the study area and the methods is given in Chapter-3 to provide context for all the methodological choices and the their underlying reasons. In Chapter-4 results and discussions of the study are provided with necessary analytical representations.

Chapter 2

Literature Review

Literature review has been a vital component of this study since there has been a lot of research so far regarding peatlands across the world. The field of peatland science is not new as it was founded over 100 years ago through the pioneering work of Carl Albert Weber in 1902 which was a comprehensive account of a Lithuanian raised bog from eco-hydrological perspectives (Baird et al., 2009). Since then, studies within the field have mainly focused on the themes of Holocene climate reconstructions, peatland dynamics especially the hydraulics, rates of peat decomposition and balance of biogenic gases, C accumulation and loss and lastly, peatland conservation and management (Chambers and Charman, 2004; Baird et al., 2009). Most of the contemporary works are highly inclined towards estimating the C balance of peatlands (Chambers and Charman, 2004) and providing a clearer perception and prediction of the responses of soil processes in peatlands to global warming as well as to anthropogenic pressures like extraction of resources and land-use changes (Rezanezhad et al., 2016). From the current literatures, it is quite evident that both knowledge and data are growing rapidly regarding the potential feedbacks of decomposition rates and GHG emissions in peatlands to climate warming. Therefore, literature review for the current study, it has been carried out with a view to describe and explain the relevant background information and arguments in a comprehensive manner.

The review includes only those papers which are particularly based on the peatlands of northern hemisphere. The entire chapter consists of four major sections including peatland dynamics, GHG emission mechanisms and factors, impact of climate change and landuse on peatlands and flux measurement methods. In the initial section, a conceptual foundation has been set regarding the physical and chemical processes of natural peat bogs in general. As the GHG balance is strongly related to organic matter decay, the physicochemical properties of peat soil have been explained mostly in terms of decomposition. In the later section, natural CH_4, CO_2 and N_2O gas dynamics of peatlands are also described so that their changes with climate and

landuse factors can be well understood later. Recently published articles and their findings have been reviewed with greater priority while presenting the potential impacts of climate warming on northern bogs.

A brief overview of the soil flux measurement methods has been provided in the last section. However, particular emphasis is given to the prevailing laboratory incubation techniques which are implemented so far to measure and calculate the fluxes of GHGs from peat or soil cores.

2.1 Dynamics of Northern Peatland

As a unique organic landform, peatlands are the most extensive types of wetlands worldwide that yield crucial hydrological, ecological and biogeochemical functions by creating transitional environments between terrestrial and aquatic ecosystems (Joosten and Clarke, 2002; Chapman et al., 2003; Rezanezhad et al., 2016). From tropical to boreal and arctic climate zones as well as from sea levels to high alpine conditions, peat deposits are found to extend over large areas of northern Europe, western Siberia, northern North America, Indonesia and southeast asia (Joosten and Clarke, 2002; Rezanezhad et al., 2016). However, the term northern peatlands precisely refers to the peatland ecosystems that are mostly located between 40° and 70° N latitude and cover substantial parts of Canada, USA or especially Alaska, northern Russia, Scandinavia, Finland and the Baltic republics (Yavitt et al., 1997; Chapman et al., 2003).

In the northern regions, peatlands are found to occur as arctic and alpine tundra, boreal and sub-boreal bogs and fens (Joosten and Clarke, 2002). The terms often discussed within peatland classifications e.g. “mire”, “bog” and “fen” possess distinct meanings. Joosten and Clarke (2002) states the difference between peatland and mire where the former includes all the vegetated and non-vegetated areas with naturally accumulated peat layer at the surface and the later explains a type of peatland where peat is being formed or developed recently. However, bogs are significantly different from fens depending on their source of water and trophic status (Lai, 2009). In the most recent hydrogenetic classification of peatlands, both of the landforms are specified further as ombrotrophic bogs and minerotrophic fens (Joosten and Clarke, 2002).

As the word “Ombrotrophic” derives from the Greek word ombros which means “rain shower” or “relating to rain”, ombrotrophic bogs usually include those peat formations that depend entirely on precipitations as the input of water. In specific terms, bogs are also called as “raised bogs” or “raised ombrotrophic bogs” where the surface of the bog is often found to lie above the nearby lands or in isolation from the surrounding mineral soils (Nungesser, 2003). According

to Baird et al. (2008), central part of raised bogs can be 10 m higher above the surrounding mineral deposits and cover many square kilometers in the horizontal extent. Characteristically, ombrotrophic bogs have nutrient-poor and acidic environments whereas minerotrophic fens that form in depressions, have nutrient rich and alkaline conditions as their source of water lies in contact with mineral bedrock or substrate (Joosten and Clarke, 2002; Lai, 2009).

Many bogs have a microtopography consisting of hummocks that are the dry mounds above the surface of the bog and hollows that are the wet depressions lying within the spaces between hummocks (figure 2.1) (Nungesser, 2003; Barreto and Lindo, 2018). The hummocks are mostly populated by non-vascular mosses (bryophytes) e.g. *Sphagnum fuscum, S. capillifolium* and *S. magellanicum* (Van der Molen et al., 1994; Barreto and Lindo, 2018) as they are capable of supplying water to the capitulam[1] at comparatively low water levels (Robroek et al., 2007). Their ability to transport water through capillary network is supposedly related to their compact nature of growth, formation of carpet and porosity (Robroek et al., 2007). In case of hollows, the floating moss *S. cuspidatum*, *S. fallax* and *S. angustifolium* are considered mainly as pool species (Van der Molen et al., 1994; Mauquoy and Yeloff, 2008; Barreto and Lindo, 2018) along with *Eriophorum angustifolium* inhabiting around the hollow edges (Van der Molen et al., 1994).

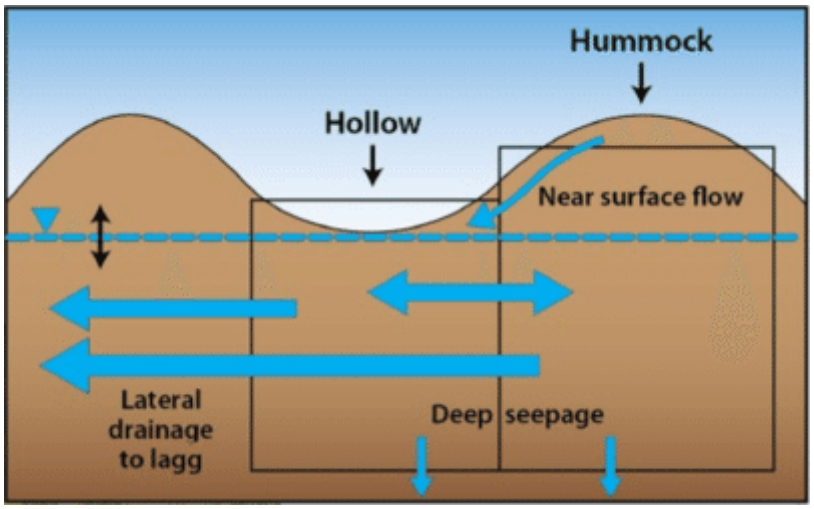

Figure 2.1: Hummock-hollow microtopography in raised bogs (Shi et al., 2015)

2.1.1 Major Functional Characteristics

The development of peatlands involves a long sedentary and autochthonous accumulation of partially decomposed plant biomass mostly originated from nonvascular and vascular plants as

[1]Capitulam is a structural component of *Sphagnum* moss that exists as a cluster of brunches at the end of *Sphagnum* Stems. The capitulam can hold large quantities of water because of their cells called “hyaline”.

well as microorganisms and microbially synthesized organic compounds (Yavitt et al., 1997; Joosten and Clarke, 2002; Chambers and Charman, 2004; Rezanezhad et al., 2016; Mustamo et al., 2016). During the process, organic matters of relatively high decomposability are lost early leaving the recalcitrant moieties to accumulate further to form peat (Leifeld et al., 2012). As identified by Stivrins et al. (2017), the rates of peat and carbon accumulation, particularly in northern bogs, are mainly controlled by factors of local vegetation, hydrology, oxygen exposure time of the surface peat layer, climate and human actions. In order to explain how these factors are associated with the consumption and production processes of biogenic GHG gases i.e. CH_4, CO_2 and N_2O and influenced by climate change, firstly, it is important to understand how a peat system actually functions in nature.

While describing peatland structure, ecologists and hydrologists often refer to a "diplotelmic" model of peat profile where the upper active part of highly permeable and less decomposed plant litter is called the acrotelm or aerobic layer and the underlying inert part of less permeable, highly decomposed and perpetually saturated peat is called the catotelm or anaerobic layer (Clymo, 1984; Holden and Burt, 2003; Baird et al., 2008) (figure 2.2). However, the C balance of peatland mostly relies on the function of acrotelm. According to Malmer and Wallén (2004), processes of acrotelm include both input of C in the form of litters and loss of C by decomposition. These two processes are essential in determining the net input of C to the catotelm as well as the net accumulation of C in a peatland (Input to the catotelm $>$ Integrated loss of C $\rightarrow$ Net C accumulation) (Malmer and Wallén, 2004). In addition to its role in peat formation, acrotelms found in northern bogs function as a buffer by protecting peat deposits from being exposed to oxidation and regulates the bog ecosystem in wet, cold and very acidic conditions (Nungesser, 2003).

Regarding the depth-wise extents of these layers, certain inconsistencies can be found in the literature. (Clymo, 1992), the most cited paper on structural layers of bogs, has defined the layers of an ombrotrophic bog on the basis of bulk density (BD) values. BD, that is calculated as the grams of dry weight per cm^3 of original wet volume of peat (gcm^{-3}), has a positive and linear relationship with decomposition (Lindsay et al., 2010; Rydin and Jeglum, 2013) as decomposition increases the mass of dry material per volume of peat (Rezanezhad et al., 2016). The relationship becomes significant when rates of decay increases with depth within the acrotelm and slows down upon reaching the catotelm. According to Clymo (1992), the values of bulk density rises from $0.03\ gcm^{-3}$ to $0.12\ gcm^{-3}$ (about 4 times) during the transition from actrotelm to catotelm. He has found that, based on BD, the acrotelm-catotelm boundary of a bog lies at 12 cm of depth as the BD values reach almost a steady state after this depth indicating no

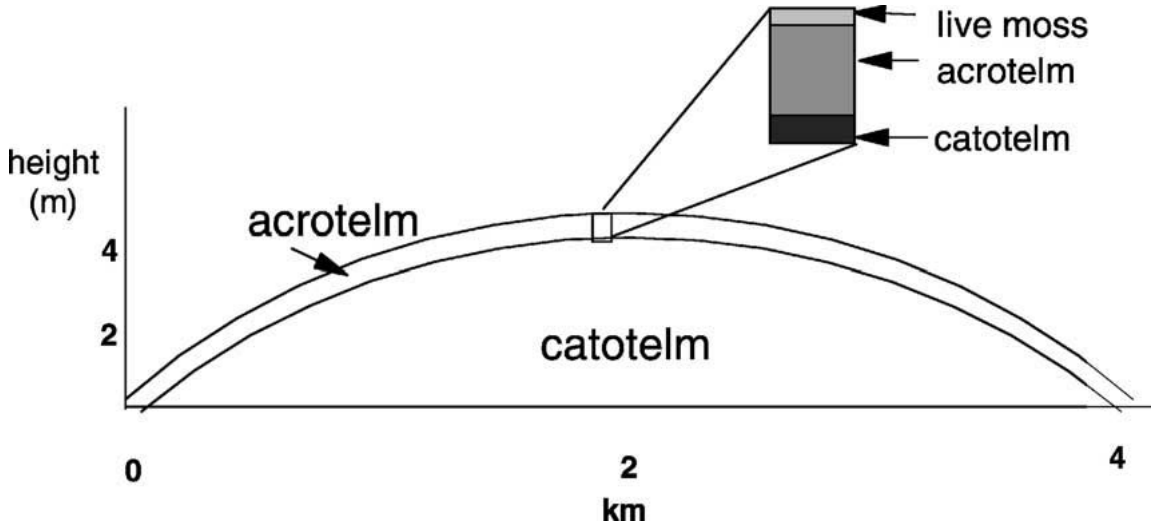

Figure 2.2: Theoretical representation of a raised bog structure where a thin acrotelm layer of living moss and aerated peat lies above the unaerated, deep and cold peat layer of catotelm (Nungesser, 2003)

further increase in decay similar to that of a catotelm.

On the basis of old Clymo-model, Lindsay et al. (2010) has currently provided a precise representation of the acrotelm-catotelm layer (figure 2.3) where a cotton grass plant, *Eriophorum angustifolium*, is shown above a *Sphagnum* bog carpet. However, here the acrotelm depth is higher up to 20 cm than the depth stated by (Clymo, 1992). As an explanation to the additional increase in acrotelm depth, Lindsay et al. (2010) mentions the intrusion of cotton grass roots into the catotelm that may induce aerobic conditions by the aerenchymatous diffusion of oxygen in the peat layer nearest to acrotelm and thus increase the vertical extent of aerobic acrotelm.

In terms of water table position, acrotelm lies above the lowest seasonal water table (Yu et al., 2001) or in other words, the lowest water table depth at a location is the base of acrotelm beneath which the saturated catotelm portion starts to prevail (Holden and Burt, 2003). By referring to Ivanov et al. (1981), Nungesser (2003), defines the acrotelm layer of 30-70 cm depth as the mean minimum water table level of the growing season while in Malmer and Wallén (1999, 2004), the depth of acrotelm has been variably defined as < 40 cm and < 30 cm and the depth of catotelm to be several meters and > 100 cm (Malmer and Wallén, 1999, 2004). However, it is also mentioned in previous studies that, since the seasonal changes in water table position make the boundary between acrotelm and catotelm unstable, such depth limits are eventually transitional but not distinct nor constant (Clymo, 1992; Malmer and Wallén, 2004; Lindsay et al., 2010).

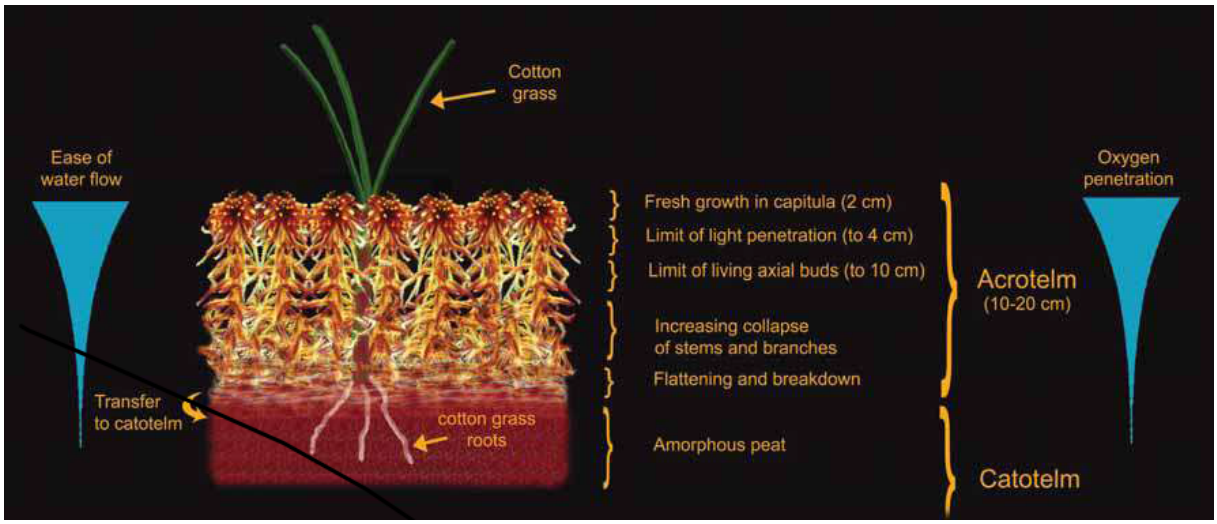

Figure 2.3: Structural characteristics of acrotelm and catotelm by Lindsay et al. (2010). Here, the rate of water flow (left) and the extent of air penetration (right) are represented as gradually decreasing from acrotelm to catotelm.

2.1.2 Major Hydrological Properties

Acrotelm is significantly different from the catotelm in terms of storage and conductivity of water. According to (Holden and Burt, 2003), acrotelm often has a fluctuating water table, variable water content and a high hydraulic conductivity while catotelm has a water content almost stable with time and a low hydraulic conductivity. In some cases, due to low conductivity and low lateral hydraulic gradient, lateral flow of water through the catotelm can be quite low and insignificant compared to that of acrotelm e.g. less than 1% of the total lateral discharge (Baird et al., 2008).

These contrasting hydraulic features are mainly associated with the level of decomposition that controls the size and interconnectivity of pore spaces (Belyea and Clymo, 2001; Rezanezhad et al., 2016). According to Rezanezhad et al. (2016), at higher depths both the amount of large pores and the inter-particle pore spaces decrease by decomposition as the plant debris get broken down into smaller fragments. Due to high fiber content undecomposed peats can have pore sizes more than 5 mm and thereby almost 80% of its saturated water can be released through drainage (Rezanezhad et al., 2016). However, as the acrotelm undergoes most of the hydrological changes as well as decay processes, the catotelm remains almost protected from the external factors and losses only 10% of its water to drainage (Lindsay et al., 2010). Therefore, changes in precipitation or evapotranspiration will presumably have a more profound impact on the acrotelm than the catotelm.

2.1.3 Major Chemical Properties

In terms of chemical composition, peats are typically formed by having at least 30% (dry mass) of dead organic material (Joosten and Clarke, 2002) and thus being rich in organic matter (OM), comprised of ≥ 20 mass percent organic carbon (C_{org}) (Rezanezhad et al., 2016). In northern bogs, partial decomposition of OM is sustained by a combination of factors including low seasonal temperatures, high water tables favouring anaerobes and the resistance of *Sphagnum* moss to decay (Barreto and Lindo, 2018). Bogs dominated by *Sphagnum* can have a partially decomposed peat layer deeper than 30 cm (Lai, 2009). According to Joosten and Clarke (2002), high levels of water can limit microbial decay and suppress microorganisms in two ways- first, by inducing lower than ambient temperatures as the heat capacity of water is high and second, by limiting the availability of oxygen as the diffusion rate of gases in water is low, about 10^{-4} that in air as per Clymo (1984).

The resistant nature of *Sphagnum*-derived debris is mostly related to the moss's chemical and structural constituents. *Sphagnum* structures contain polysaccharides but lacks lignin (Yavitt et al., 1997) and in terms of microbial decomposition, lignins are tough to break down while polysaccharides are readily decomposed through the C and nitrogen (N) cycling as their labile C forms can be utilised easily by microbes (Wang et al., 2017b). But, despite the predominance of polysaccharides, litter and organic substrates derived from the *Sphagnum* are naturally slow to decompose (Moore et al., 2007). As a reason to that resistance, Yavitt et al. (1997) considers other structural constituents of *Sphagnum* such as the lignin-like matrix of polyphenolic compounds, pectin-like "sphagnan" that inhibits microbial functions by binding nutrients and deactivating enzymatic biodegradation and high amounts of uronic acids that keeps the pH low to be responsible for the slower rates of decomposition in peat soils. Mastnỳ et al. (2018) has also referred to the *Sphagnum*-derived high molecular weight organic compounds e.g. phenolic and uronic acids and sphagnan while characterising the low biodegradability of DOC (Dissolved Organic Carbon) in peat pore waters.

Due to acidic conditions, peat bogs of northern Europe, West Siberia, the United States and Canada have low pH values typically ranging from 3.5 to 5 (Dedysh et al., 1998). In ombrotrophic bogs pH of the peat water can be lower ranging between 3 to 4.5 (Lai, 2009). However, even within a single bog it can vary e.g. the hummocks above the water table have lower pH than that of the hollows below the water table (Barreto and Lindo, 2018).

The status of decomposition varies significantly along the depths of a peat profile. The ratio of C to N can be a useful measure to indicate the degree of decomposition at different

soil depths where reducing C/N ratios are signs of increasing peat degradation with increasing depths (Malmer and Holm, 1984; Silamiķele et al., 2010; Batjes, 2014). By combining the findings from (Malmer and Nihlgård, 1980) and (Rosswall and Heal, 1976), Malmer and Holm (1984) has stated that, in a subarctic peat, the C:N quotient of top 2-4 cm lying just below the moss surface decreases to more than 2 times at the depth of 20-30 cm. This happens because fresh plant deposit on the top has a higher C:N but when the top peat mineralization occurs by oxygen exposure, level of N increases leading to the reduction in C:N (Bader et al., 2018). The relationship between C:N and decomposition has implication in the emission of soil N_20 as well. Kløve et al. (2017) states that usually water-logged soils lack nitrate NO_3^- but when they are drained, the presence of oxygen produces NO_3^- by nitrification. In denitrification, this nitrate can be used up by microbes and thus N_20 can be produced (Kløve et al., 2017). In lower CN ratio caused by N deposition, N_20 emission can also be increased by increasing availability of N to nitrifying and denitrifying bacteria (Wang et al., 2017a).

2.2 Mechanism of Biogenic Gases

It is now well accepted that the biogenic production of GHGs such as CO_2 and CH_4 is a significant part in the C-cycling of northern peatlands (Belyea and Clymo, 2001; Lai, 2009; Comas et al., 2014; Abdalla et al., 2016). In McLaughlin (2004), the main processes associated with the C sink and source function of peatlands have been precisely outlined as follows-

- Mass flux of organic and inorganic by means of hydrological influences and pathways
- Vegetative contribution of organic C to the peat in the forms of root exudates and fresh litters
- Microbially (aerobes and anaerobes) driven transformation of organic and inorganic C
- Uptake of atmospheric CO_2
- Transfer of CO_2 and CH_4 through a peat profile

Given that, the oxidation and reduction (redox) potentials of a peatland are greatly controlled by its hydrology and vegetation, it is mostly the anoxic or reduced condition that produces N_2O and CH_4 in the stable water phases i.e. catotelm (McLaughlin, 2004). However, in the phase of unstable water i.e. acrotelm, aeration promotes loss of C in the form of CO_2 by favoring more complete oxidation of organic matters and reducing the emission of N_2O and CH_4 (Clymo, 1984; McLaughlin, 2004).

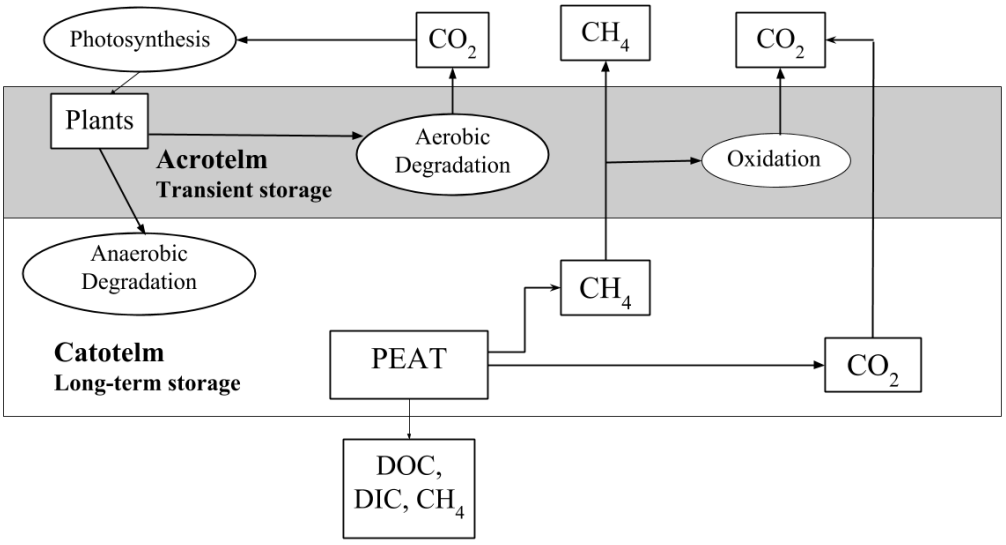

Figure 2.4: Major input, output and storage components of C-cycling in peatlands (McLaughlin, 2004)

2.2.1 Production and Consumption of CH_4 and CO_2

Both Lai (2009) and Serrano-Silva et al. (2014) have provided a detailed account of the mechanisms of CH_4 and CO_2 in soils known as methanogenesis and methanotrophy. Both of the mechanisms occur in several steps by a group of specialised microorganisms called methanogens and methanotrophs respectively. In Lai (2009), anaerobic production of CH_4 has been explained by certain steps of chemical transformations that mainly involve conversion of peat-derived carbohydrate polymers (polysaccharides) into monomers by hydrolysis, production of organic acids, fatty acids, alcohols, hydrogen (H_2) and CO_2 from the monomers by acidogenesis, production of acetate and CO_2 from the fatty acids and alcohols by acetogenesis and finally production of CH_4 from the acetate, $H_2 + CO_2$ or alcohols by methanogens. Among the methanogen types, hydrogenotrophs are found to produce CH_4 from CO_2 and H_2 while acetotrophs produce CH_4 out of acetates (Lai, 2009). As they function in lack of oxygen and at lower redox potentials i.e. $< -300\ mV$, deeper and saturated soils are often found to produce the largest share of CH_4 within a peat profile (Lai, 2009; Serrano-Silva et al., 2014)

However, this CH_4 can have two consequences- it can either get oxidised and consumed by the methanotrophs at the aerobic zone or it can be directly transported to the atmosphere via diffusion, ebullition and plant-mediated forms (Lai, 2009; Serrano-Silva et al., 2014). In the biological oxidation of CH_4, methanol, formaldehyde and format are found as intermediates where the end product, CO_2, derives particularly from the oxidation of format by a methanotrophic enzyme called Formate Dehydrogenase (FDH) (Serrano-Silva et al., 2014). Methanotrophs are

most active at the zone lying near to the mean water table or more precisely, at the boundary between oxic and anoxic layers (Dedysh, 2002; Lai, 2009). As oxygen availability decreases at depth, methanotrophs lack the amount of oxygen needed to start the oxidation reaction at greater depths while at the upper layers, although oxygen supply is adequate, methanotrophs lack enough CH_4 to consume and oxidize (Lai, 2009).

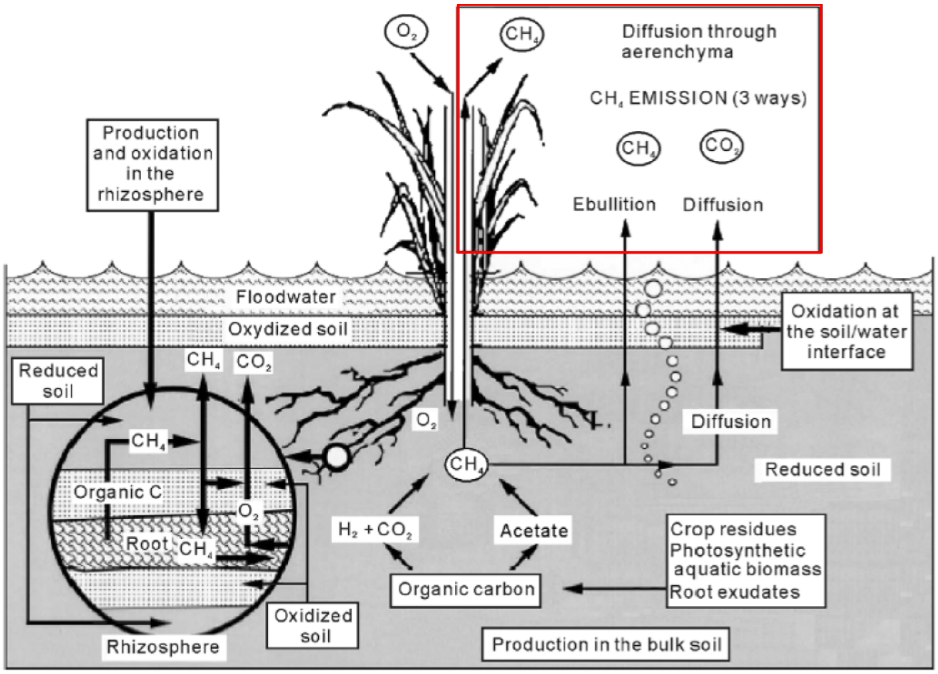

Figure 2.5: Production, consumption and emission processes of CH_4 in peatlands based on Le Mer and Roger (2001)

2.2.2 CH_4 Production and Consumption Rates

The rate of CH_4 oxidation can be significantly higher compared to the rate of CH_4 production which means methanotrophy can actually limit the release of CH_4 to the atmosphere by consuming most of the CH_4 produced at the deeper peats (Lai, 2009). In northern bogs, both temperature and water table depth are considered important in the production and oxidation rates of CH_4. The effect of temperature on the rates of microbially-mediated reactions can be explained by the equation of Arrhenius,

$$K = Ae^{-\frac{E_a}{RT}} or, lnK = lnA - \frac{E_a}{RT} \tag{2.1}$$

where, K is the reaction rate constant (CH_4 production or consumption rate), A is frequency factor that remains almost constant in small changes in temperature, E_a is the reaction activation energy ($J\ mol^{-1}$), R is the universal gas constant ($J\ k^{-1}\ mol^{-1}$), $e^{-\frac{E_a}{RT}}$ is the fraction of gas molecules that has reached or exceeded the activation energy and T is temperature (K). One

of the major implications of the equation is, when all other factors are constant, the $e^{-\frac{E_a}{RT}}$ value or the fraction of molecules having $\geq E_a$ or in other words the reaction rate increases upon rise in temperature e.g. by 10^0C. As the rate and temperature relationship is exponential a smaller rise in the temperature is expected to have a greater increase in reaction rates.

Underlying inferences of the Arrhenius law together with the Q_{10} value (factor by which rate of reaction changes for each 10^0C (Lai, 2009; Bader et al., 2018) have been often used in studies to explain the impact of temperature on the rates of CH_4 production and consumption. For instance, Dunfield et al. (1993) has found that, CH_4 production is much more temperature dependent than CH_4 consumption and requires higher activation energies, E_a ($123\ to\ 271\ KJmol^{-1}$) and Q_{10} value (5.3 to 16). In their experiment, production of CH_4 in 0 to 10^0C was almost insignificant while CH_4 consumption was noticeably high (13-38% of maximum) at that range.

Both Dunfield et al. (1993) and Metje and Frenzel (2005) have stated the optimum temperature for methanogenesis as 25^0C but significant microbial activity has also been found at 4^0C and the theoretical temperature limit for methanogenesis has been found even close to -5^0C (Metje and Frenzel, 2005). Therefore, it can be assumed that, although the activity of methanogens can exist between a range of low to high temperatures, increase in the rate of methanogenesis from a certain level needs higher temperatures than to increase the rate of CH_4 oxidation. In other words CH_4 production is more sensitive to warming than CO_2 production.

According to the study by Gill et al. (2017), deep layer peat warming has led to greater increase in CH_4 production in a boreal ombrotrophic bog. Northern bogs are often dominated by hydrogenotrophs (Lai, 2009; Gill et al., 2017) but Gill et al. (2017) has suggested that warming can increase the dominance of acetotrophs. The situation associated with that can be explained by the following reactions,

Hydrogenotrophic CH_4 production, $4H_2 + CO_2 \rightarrow CH_4 + 2H_2O$
Acetotrophic CH_4 production, $CH_3COOH \rightarrow CH_4 + CO_2$

The reactions suggest that if there is an increase in acetotrophic reaction than as a by product CO_2 may also increase and as there is a reduction in hydrogenotrophs with warming there may be less consumption of CO_2 to produce CH_4. Therefore from this point of view warming can actually increase the production of CH_4 and CO_2 at deep layers.

In case of water table depth, differences in findings can be found. According to Lai (2009),

most of the non-flooded northern bogs represent a negative relationship where decreases in the mean water table depth increases the production of CH_4. He explains it further by the lower thickness of CH_4 oxidation and higher thickness of production zone associated with higher water tables or lower water table depth. Studies have also found that in a *Sphagnum* peatland, water table depth and average potential CH_4 oxidation rate are significantly positively correlated to each other (Sundh et al., 1995; Lai, 2009) which means a less production of CH_4 with high levels of CH_4-derived CO_2. However, contrasting findings also exist where higher water table depths actually led to greater CH_4 productions. As for example, Treat et al. (2007) reported an increase in CH_4 emission from a temperate fen as water table dropped over time, which might due to the overriding effects of increased CH_4 production and ebullition arising from higher peat temperature and reduced hydrostatic pressure respectively on CH_4 flux. Depth wise cont...

2.3 Peatland and Climate Change

It has been evidently established by the scientific community that the process of global warming is mostly caused by the human-induced increase of GHG concentrations in the atmosphere (Cubasch et al., 2013; Oreskes, 2018). Ever since the late 19th century, intensive human activities like landuse change, fossil fuel burning, deforestation and animal husbandry have considerably elevated the concentrations of GHGs, mainly CO_2, CH_4 and N_2O in the atmosphere (Ehlers and Krafft, 2006) leading to a notable rise of $0.6°C$ in the global average surface temperature during the 20th century (Folland et al., 2001; Graßl, 2006). Over the northern latitude land masses, this average warming is highly likely to continue even during the 21st century and expedite the effects of climate change if the global emission of GHGs was not reduced to a reasonable level (Collins et al., 2013). In order to control and mitigate further emissions of GHG into the atmosphere, it is essential to accurately quantify the prevailing GHG sources and sinks in terms of their relative contributions to the net global emission.

A reliable and realistic global GHG budget, not only includes direct human emissions but also incorporates the GHG releases from soil- the largest terrestrial pool of C and N (Oertel et al., 2016). For instance, on a global scale, the annual net soil emission is estimated to be $\geq 350\ Pg\ CO_2\ e$ with a conservative average of $300\ mg\ CO_2\ em^{-2}h^{-1}$, that is quite significant compared to the annual net emission of $33.4\ Pg\ CO_2$ from fossil fuel combustion and cement industry (Oertel et al., 2016). However, soil emissions greatly vary with the types of land covers such as forests, wetlands, crop fields, barren lands etc.

It is suggested that among all the land cover types, wetlands represent the highest average abso-

lute emission rates (Oertel et al., 2016). Usually landforms like peatlands or bogs, fens, shallow lakes, swamps, marshes, floodplains and flooded rice paddy fields are considered as wetlands that together cover only 5-8% of the total land surface but emit about 30% of the global CH_4 emissions (Chamberlain et al., 2018). Depending on the climatic origin, distribution, management and use of these wetland types, wetland GHG emission issues may vary across the world. As for example, in the countries of India, China and Vietnam where rice production is relatively high, the problem of CH_4 fluxes from rice paddy fields is considered more prominent than that of peatlands (Carlson et al., 2017). However, in the regions of Europe, degradation of natural peatlands, their current and future emission potentials are mostly emphasized (Carlson et al., 2017). This emphasis can be justified in several ways.

Firstly, these peatlands have been so far acted as constant sinks of atmospheric CO_2. Excluding the long term decay, the annual CO_2 uptake rate of peatlands is $0.088\ Gt\ C\ yr^{-1}$ that is little in amount but found to be persistent (Yu et al., 2010). Despite the fact that ocean outgassing driven by lowering of atmospheric CO_2 and air-sea CO_2 partial pressure can reduce the true effect of peatlands' CO_2 uptake on the atmosphere, it is suggested that on a millennial scale, changes in the size of peatland C sinks have affected global atmospheric CO_2 concentrations significantly (Charman et al., 2013).

Secondly, peatlands are great sources of CH_4 and northern peatlands contribute to more than 12% of the global total CH_4 emissions (Wuebbles and Hayhoe, 2002; Askaer et al., 2011) which is really important from the global warming perspective. CH_4 is more effective in absorbing infrared radiation than CO_2 and on a 100-year time scale the warming potential of CH_4 is about 25 times greater than CO_2 (Askaer et al., 2011; Smith et al., 2018). However, in the cropped peatlands of Europe, drainage activities have been found to increase the net CO_2 emission of bogs by uncovering the peat C to the oxygen (Carlson et al., 2017). For example, draining of peatlands is estimated to be responsible for about $630\ \pm\ 90\ Tg\ CO_2\ e\ yr^{-1}$ rates of emissions, in which, about 90% is CO_2 and only 1% is CH_4 (Carlson et al., 2017). Those rates of CO_2 emissions are of complete contradiction to the CH_4 dominating emission characteristic of natural bogs and indeed, add high levels of variability in future peatland GHG estimates.

Finally, being adapted to low temperatures and humid soil conditions, northern peatlands can be really vulnerable to the impacts of climate change. In case of Europe, although future precipitation estimates may not be obvious but it is highly likely that upcoming decades are going to experience increase in mean temperatures and net evapotranspirations (Collins et al., 2013).

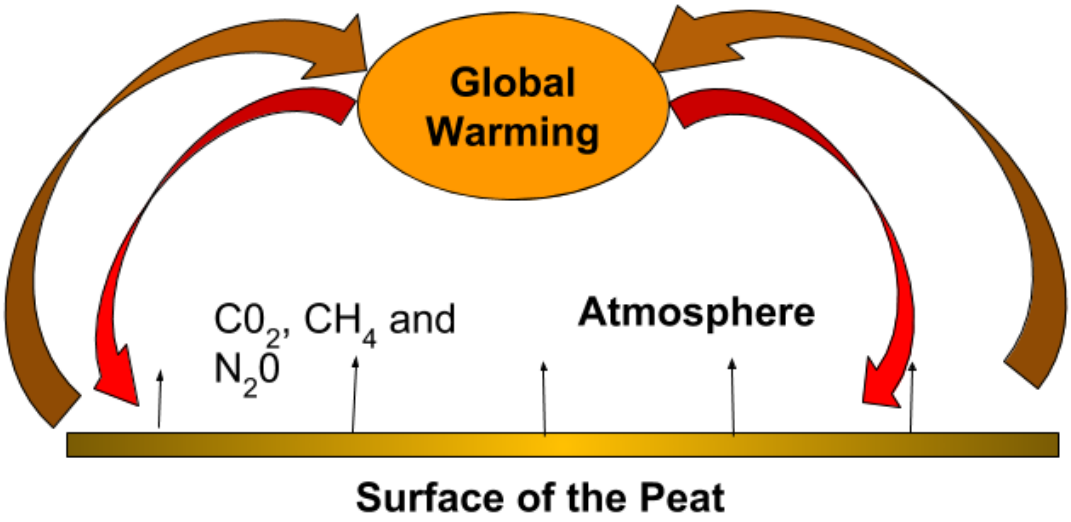

Figure 2.6: Relationship between global warming and peatland GHG emissions (prepared by this author)

Future temperature increases associated with climate change will change chemical hydrological dynamics of soil. In terrestrial ecosystems, higher temperatures mean higher evaporation rates, potentially resulting in reductions in soil water content. For northern peatlands which are generally waterlogged (anaerobic), this could lead to higher decomposition rates of organic carbon as soil conditions turn aerobic. This would result in increased GHG emissions as a byproduct of the decomposition processes.

Given that climate change is a result of the greenhouse effect, for certain ecosystems, this would create a positive feedback loop with GHG emissions, global warming, and soil chemical hydrological dynamics (figure 2.6).

Chapter 3

Methodology

The study has emphasized and implemented specific methodological approaches in order to accomplish its objectives. Therefore, this chapter is designed to describe and explain the methods and techniques of the current research in a systematic and comprehensive manner. It evolves through the description of the study area where relevant features of the Holmegård raised bog have been described on the basis of background literature. In the study area section, Danish Nature Agency's investigation report on Holmegård bog in connection with the EU LIFE - LIFE14 (Aaby and Riis, 2016) has been studied and referred extensively for defining the current areal extent as well as for describing the developmental and structural properties of the peatland.

The later sections are based on the overall design of the study which extends from the field procedures of collecting peat soil samples to the laboratory experiment of incubating peat soil cores at different temperatures and water levels. In order to determine the ultimate method for field sampling and laboratory analysis, a number of test trials has been performed. Until now studies focusing on soil gaseous flux measurements in ex-situ conditions have represented a variety of methods, however, those methods have their own challenges, scopes and limitations. The purpose of the test trials was therefore to experience those methodological challenges, assess their positives and negatives in the context of the study and eventually to define a specific and acceptable strategy to address the research problem on the basis of available technical and literary resources.

3.1 The Study Area- Holmegård Mose

Holmegårds Mose is a Danish raised bog that has so far been studied in multiple fields as in anthropology to explore the earlier human settlements, in paleoecology to analyze vegetational history and most recently in Danish wetland management plannings and policies. However, the

site has never been considered for analyzing the probable impacts of future warming on raised bogs and their GHG emissions although both natural and degraded raised bogs in Europe are considered highly vulnerable to the changes in climate. Therefore, the justification of selecting Holmegård as the study area is explained here by certain key points.

The site distinctly represents the scenario of peat degradation in Denmark which started with the establishment of Holmegård Glass Factory in 1825. Excavation of peat for fuels used in glass production had been extensive for more than 100 years until the beginning of oil consumption in glassworks in 1945 (Schlüter, 1988). Despite being the largest raised bog in Eastern Denmark, the total bog area has been reduced to about 420 ha from its earlier extent of 506 ha (Mogensen et al., 2000). It is suggested that currently only two areas of 10.4 ha and 16.5 ha in its western part remain unexcavated or as being not entirely dugged up (Aaby and Riis, 2016). However, although commercial peat excavation was over long ago, the "active raised bog" parts in this area are still considered highly vulnerable to the problems of fragmentation, draining and drying of the peat layer and also to increased atmospheric N depositions which can potentially reduce its characteristic *Sphagnum* rich vegetation and cause tree and shrub growth in the long term (Hviid, 2014; Mauquoy and Yeloff, 2008).

The bog can be highly relevant in the context of peatland GHG emissions and climate change. It has been found that positive climate forcings can reduce the ability of raised bogs to survive in increased summer temperatures and limited summer precipitations (Mauquoy and Yeloff, 2008) and Holmegård Mose may not be an exception to that. Being an ombrotrophic bog, the peatland is essentially dependent on precipitation to fulfill its water demand. Reduced summer precipitation or prolonged periods of high evapotranspiration or drought conditions can make this bog water deficit and thus enhance aerations leading to the change in relative depths of oxic and anoxic zones.

From the above discussions, it is well-apprehensible that the dynamic ecosystem of Holmegård Mose can be significant in terms of both peat degradation and climate change.

3.1.1 Location and Significance

Being spatially connected with the "Suså" river, Holmegård mose covers parts of large low lying areas adjacent to the river in the north of Næstved Municipality of southern Sjælland or Zealand (figure 3.1b). The area stretching from the west of Rønnede to Næstved along Suså, includes, Holmegård bog and Porsemose moss meadow on its east side and "Tystrup-Bavelse" lake area on its west (figure 3.2). Together with the bog of "Slagmosen" that lies in the south

of Næstved, these areas are designated as European Union's (EU) "Natura 2000" in order to protect and conserve the bird and habitat diversity existing therein (source[1]).

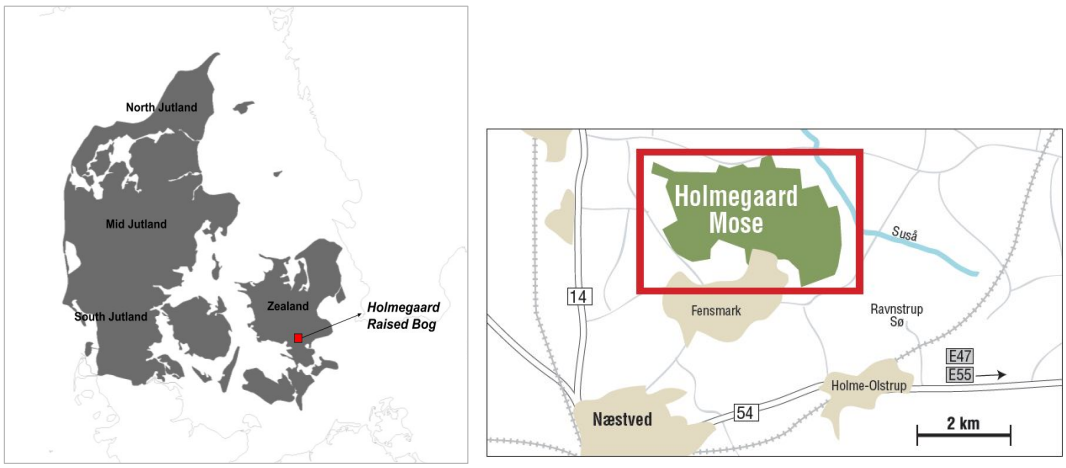

Figure 3.1: (a) Location of the Holmegård in Denmark (b) location of Holmegård bog in relation to Næstved Municipality and Suså river (source: The Danish Nature Agency)

The raised bog of Holmegård has been particularly emphasized in the sector of natural wetland restorations in Denmark. Raised bog restoration methods involve the development of certain environmental conditions that facilitate nutrient-poor and acidic soils and most importantly, high levels of water so that higher turf surfaces of the bog can be protected from lowering of water tables or droughts (Aaby and Riis, 2016). Natural restoration has been found to improve the conditions of the degraded areas of Holmegård bog. The EU-life project 2010-2013, coordinated by the Danish Nature Agency, has hydrologically restored 70 ha of the bog and found that such restorations have actually improved the bogs *Sphagnum* growth similar to the original active raised bogs.

3.1.2 Formation and Structure of the Raised Bog

The development of topographic landforms in Denmark has been found to be greatly influenced by the climate-driven environmental changes and the incidences of glacier advances and retreats of the Middle and Late Pleistocene Ice age (Svendsen et al., 2004). At Holmegård, formation of the elevated bog took place over the layers of glacial formations from the Late Pleistocene Ice age, or, more precisely, from the Last Ice Age or Weichsel Ice Age. A detailed account of the area's geological development has been given in the Nature Agency's 2016 investigation report

[1]The Danish Nature Agency. Website- `https://eng.naturstyrelsen.dk/`

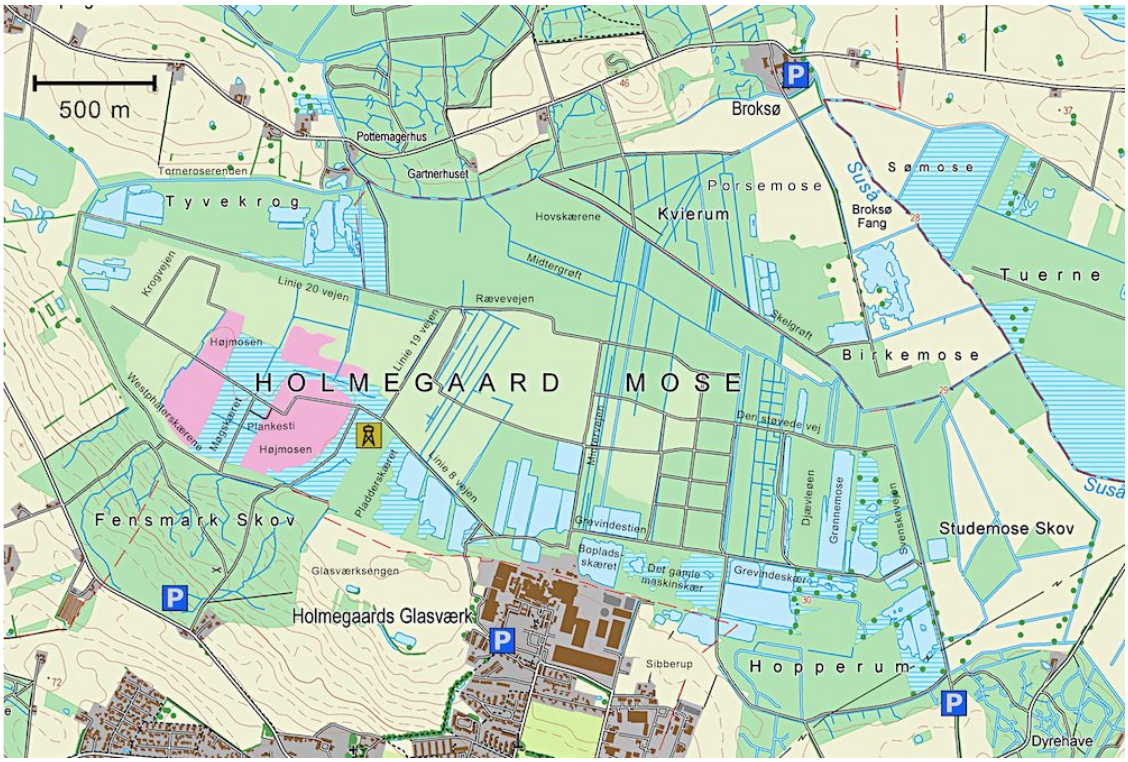

Figure 3.2: Location of Holmegård raised bog or højmosen (presented in pink color) in relation to Fensmark and Studemose skovs (forests), to Porsmose and Birkmose bogs (source: The Danish Nature Agency)

on Holmegård bog (Aaby and Riis, 2016). According to that report, vertical cross-section of the area from the high bog surface to the base layers presents sequences of sedimentary successions that mainly include different formations of turf and types of depositions (figure 3.3).

The foundation layer contains moraine hills and dead ice deposits from the glacial retreat that occurred following the period of LGM. LGM refers to the final period of the Last Ice age when the ice sheets reached their highest extent, however, the end of LGM is characterized by the start of rapid and disproportionate receding of ice sheets and thereby gradual deglaciation (Svendsen et al., 2004). The retreat and melting of Baltic ice sheets (young Baltic ice) has been found to create dead ice formations and lakes at the area of Holmegård (figure 3.4). Later during the younger dryas time, in a peaceful sedimentation environment, these lakes were deeply buried under clay. Entering into the Holocene Epoch, there was a significant growth of turf vegetations covering the lake shores. Until the Atlantic period, the lake mostly received depositions of calcium-gyttja followed by a layer of detritus or algaeal gyttja[2](figure 3.4).

However, development towards a nutrient- poor raised bog environment occurred when turf

[2]Gyttja- sedimentary peat having fecal pellets

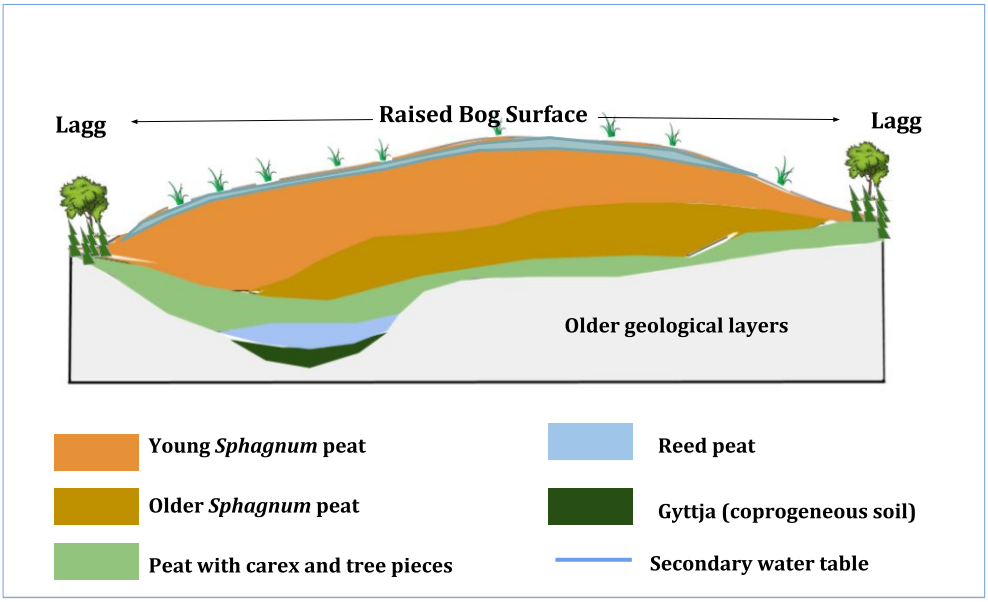

Figure 3.3: Structural layers of the bog and their compositions (Source: The Danish Nature Agency)

vegetation covered the entire lake and ultimately created a bog. This bog formation is estimated to take place at around 7.5 ka years BP. At that time, the bog had swamp turfs at its wet centre with high density of trees growing closely at its drier edges or near-shore places. But, as the bog had no streams flowing through it, very low to no movement of nutrient-rich groundwater could occur there resulting in a very nutrient poor environment over time. Rainwater also caused the dilution of nutrients into the groundwater and thus, expedited the process of nutrient loss at the bog. At the central part, loss of nutrients was the greatest where the bog started to create a highly nutrient-poor and acidic environment conducive only for the survival of raised bog plant species.

3.2 Methods and Techniques

With a view to examine the effect of increasing temperatures and water-table manipulations on the emission of GHGs from Holmegård bog (section 1.2), a laboratory experiment was designed on the first place following a thorough review of the methodologies and techniques implemented by other relevant studies. The experimental design has evolved around three major components:

- one physical component that is the peat soil of Holmegård bog
- one climatic component or variable that is the air temperature

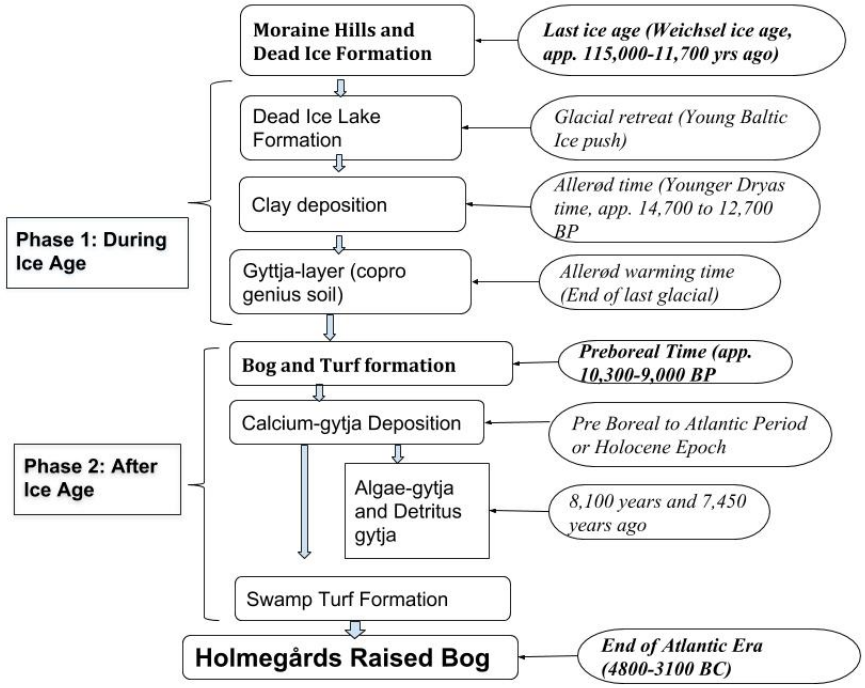

Figure 3.4: Chronological development of Holmegård raised bog (based on Aaby and Riis (2016))

- one climatically induced physical variable that is the soil moisture content or peatland water table

In the presence of all these components, the purpose of the experiment has involved the preparation and maintenance of an ex-situ setup where the peat soil samples divided by high and low water levels will be exposed to higher temperatures for a specific period of time and both the CH_4 and CO_2 fluxes from peat samples will be estimated by measuring the gases' respective concentrations at each of the warming levels. In order to execute the experiment and obtain emission values, the study has implemented a number of methods and techniques while conducting the field and laboratory procedures outlined in figure 3.5.

3.2.1 Field Sampling Design

The field procedures are designed essentially to collect soil samples from the Holmegård raised peat bog and to transport those samples back to the lab at Roskilde University for temperature and water level treatments. The sampling design had to be adjusted twice during the study. The initial peat sampling design is based on the optimal methods of soil sampling particularly relevant for laboratory experiments on soil GHG emissions. Wang et al. (2017b) has been followed mostly for preparing the design. While the plan adapted for the final sampling has

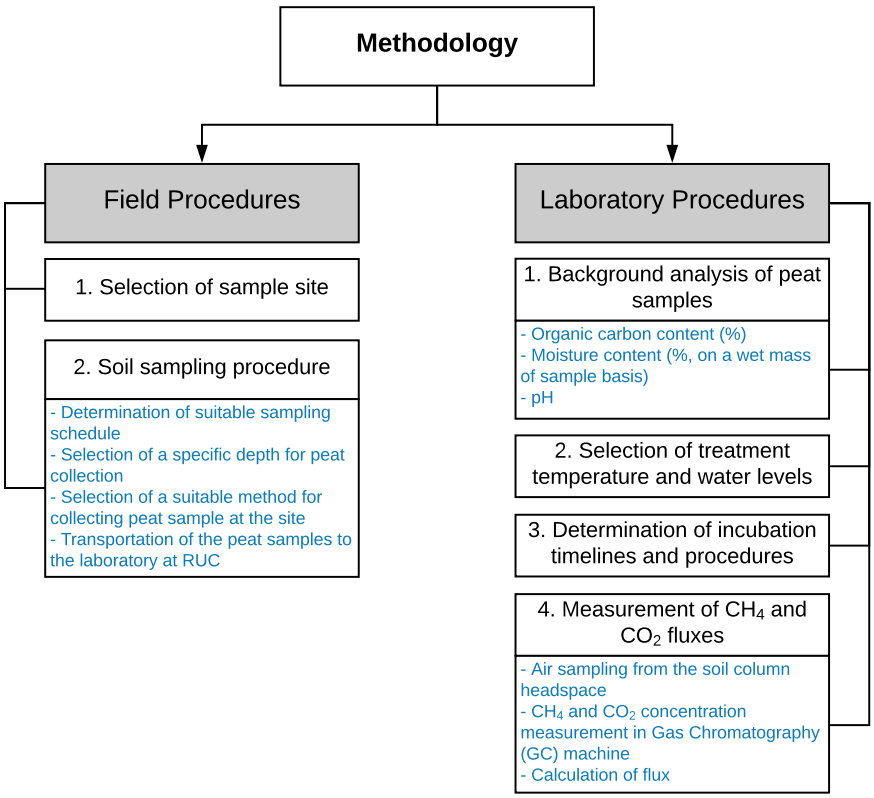

Figure 3.5: Classification of the major field and laboratory procedures conducted during the study

been more of an adjusted version of the initial plan considering the practicality of the initial design in terms of limited time and resources. Therefore, before presenting the final procedure, a brief discussion of the initial experimental design and its adjustments are given below:

Initial Soil Sampling Design

The initial soil sampling design includes the collection of a total number of 50 intact peat core samples from hollows at the pristine and drained peat sections of the bog. Ideally, intact peat soil cores are preferred for laboratory incubation experiments. Disturbed soil samples constitute more structural and microbial modifications than the intact ones (Schaufler et al., 2010; Van der Weerden et al., 2010; Petersen et al., 2013; Oertel et al., 2016) and thus may not represent the actual gaseous response of the natural soil during the experiment. Regarding the choice of sampling location, between the hollows and hummocks of the bog, hollows are chosen over the hummocks as sampling locations so that the chance of spatial variability within the replicate cores of the experiment can be minimized. Sampling peat cores from both the natural and degraded sites at the bog would allow for the investigation of how the different conditions of the sites respond to the increases in temperature and reduction in water level in terms of GHG fluxes.

The design has proposed the selection of at least five representative sampling points at each of the sites and the collection of five intact soil cores from each of the sampling points (figure 3.6). In order to decrease the effects of heterogeneity caused by intact soil samples, larger sample sizes are recommended particularly when the effects of a single parameter i.e. temperature on soil emissions are investigated (Oertel et al., 2016; Gritsch et al., 2015). It is also desirable to have multiple replicates in a sampling design so that statistical inferences i.e. statistical significance testing or confidence intervals can be used to determine the validity of the hypothesis.

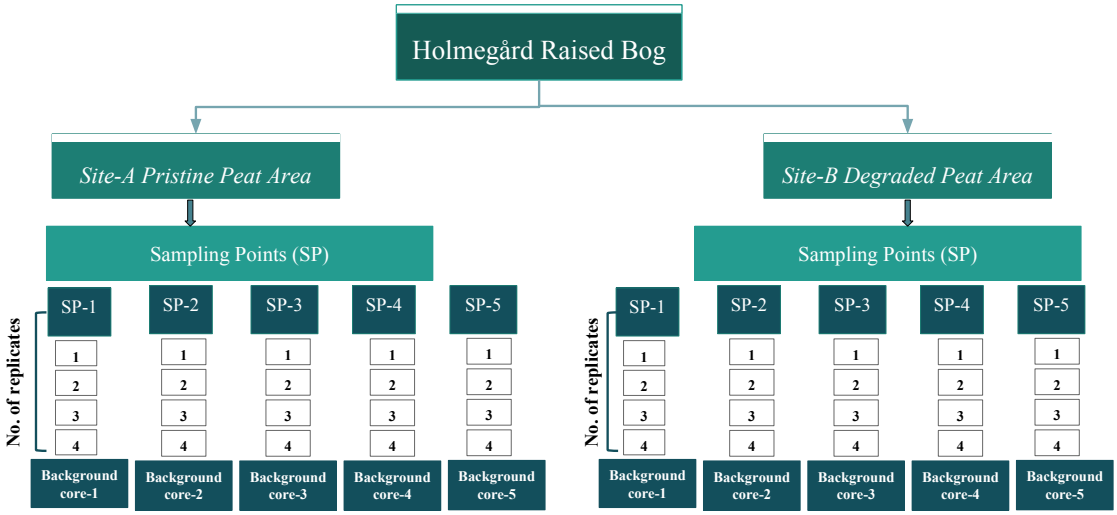

Figure 3.6: The initial design of sampling peat soils at the Holmegård raised bog

According to the design, sample peat cores are taken from the surface to about 50 cm of depth at the hollows by a peat sampler having a minimum length (L) and inner diameter (ID) of 55 cm and 5 cm respectively. The depth of 50 cm is significant here since most of the northern peatlands are found to have peat layers of more than 30 cm deep (Lai, 2009). After sampling, the intact cores are shifted into transparent PVC tubes where the dimensions of the peat columns will be 50 cm (L) $\times$ 5 cm (ID) with headspaces of 5 cm (L_h) $\times$ 5 cm (ID_h) above the peat surface. Disturbed soil samples will also be collected from the sampling points to analyse the background chemical properties of the soil such as organic C content, moisture content, pH, and C:N. In the laboratory, five samples from each site are used as background, or control cores, maintained at field temperatures and water levels throughout the experiment (figure 3.6). The remaining 40 samples are used for the temperature and water table treatments, with four replicates per experiment.

Final Soil Sampling Design

The initial design is ideal and methodologically sound, however, its practical implementation has been found to be highly inefficient, while considered in terms of the time and resources granted for the study. During the field trial of sampling, certain practical limitations were discovered. Namely, lack of peat core sampler for collecting the required number of intact core samples and lack of relevant background work on the bog. Unfavourable weather conditions during the scheduled time of sampling resulted in frozen water in hollows which made it impossible to sample at the intended depth and remove intact core. With the restricted time frame of the project, sampling could not wait for warmer time, therefore modifications are made to the initial design to make sampling feasible. The final plan of peat sampling has been therefore developed as a result of major adjustments to the initial design that can be outlined as follows:

1. In the new sampling design, semi-intact or partially disturbed peat core samples are used instead of the intact ones. Unlike the collection of intact peat cores at the field, the design has included the laboratory preparation of semi-intact cores where only the depth-wise peat types are kept similar to that of the field but the constituent peat soils used are mostly disturbed in terms of structure.

2. The number of sampling sites has been changed considering the time limitation of the study. For the final sampling, only the most natural section of the bog has been selected as the sampling site.

3. The total number of samples has been reduced to 16 from 50 as there is only one sampling site in the final design. However, weather conditions during the time of sampling has also been a major obstacle in the planning of multiple sampling schedules as well as in the collection of an optimum amount of peat soil samples from the bog.

4. The sample depth limit of 50 cm has also been changed to 30 cm. Based on the literature review, it is assumed that the depth of 30 cm from the surface should sufficiently represent the bog's distinct layers of less decomposed upper acrotelm and more decomposed lower catotelm (see section 2.1.1)

5. During the test trials 3 intact cores have been possible to collect. The cores have been used in the final experiment as background cores with no temperature and water table manipulations.

Based on the above changes, the final sampling procedure has been conducted by three main steps- selection of the most unaffected and natural site of the bog, collection of peat samples at

the hollows from two consecutive depths within the top 30 cm and transportation of the samples to RUC in order to make a total of 16 semi-intact peat cores at the lab (figure 3.7).

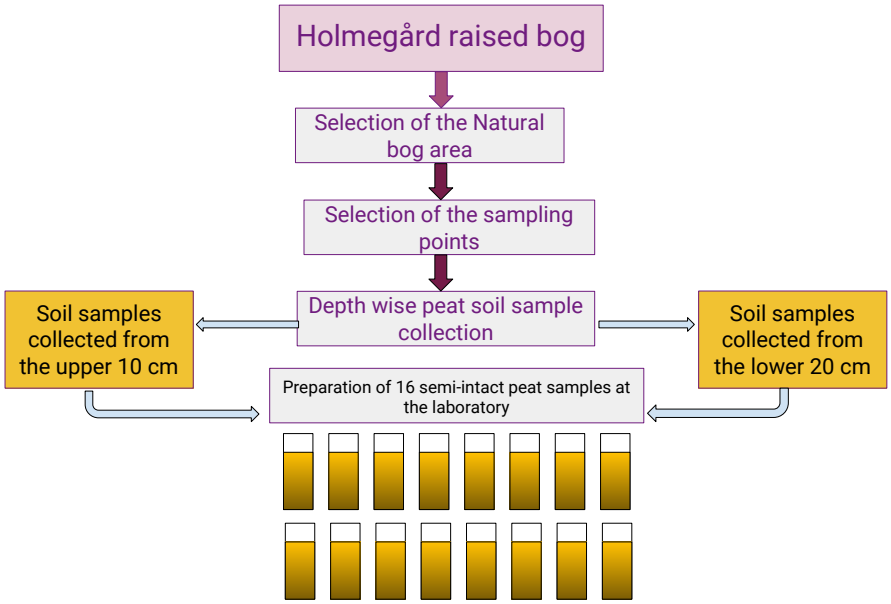

Figure 3.7: The final procedure of collecting peat samples at the Holmegård raised bog

3.2.2 Selection of the Site

As the first step of sampling peat soils from the Holmegård bog, the most undisturbed section of the bog has been selected as the site of sampling. The basis for assuming the condition of peat at different parts of a peatland may involve consideration of multiple factors e.g. intrusion of nutrient rich groundwater from the forest area nearby, local position of the water table and also the growth of trees or large vascular plants growing by the edge. For this study, decreased thickness of the raised peat has been considered as the main indicator of peat degradation. In order to identify locations having the highest and lowest levels of peat degradation, the Danish Nature Agency's EU LIFE project - LIFE14 project report (Aaby and Riis, 2016) is selected as the background literature. The report provides peat thickness data for specific drilling locations across the entire bog and defines three sections in a map where the largest elevated central part has the highest peat thickness (outlined in purple, figure 3.8) and therefore, represents the most unaffected raised peat formation in the bog. Peat samples for the current study have been collected from a randomly selected location within the area of highest peat thickness (outlined in black, figure 3.8). The absolute location of the sampling area was $55°17'35.16''N$, $11°48'06.40''E$

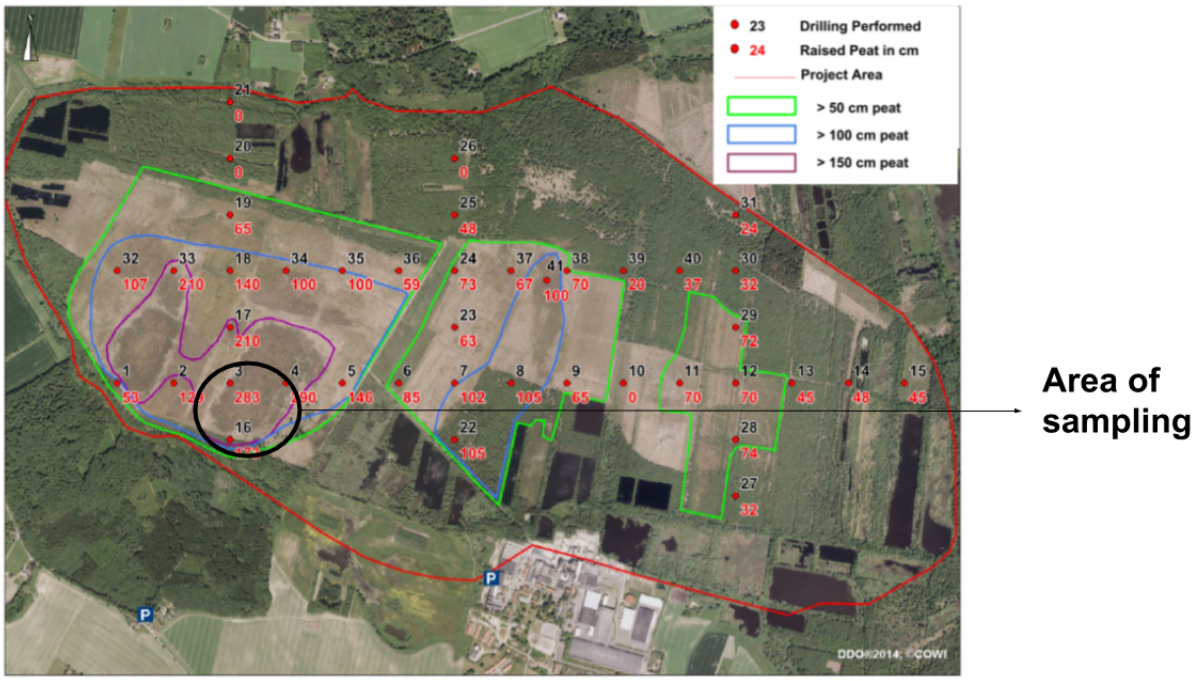

Figure 3.8: The area of sample collection within the boundary of highest peat thickness. The background image is taken from Aaby and Riis (2016) where layer thickness of peat is shown on a background orthophoto DDO® 2014 (scale 1:20,000 with height contour intervals of 0.5m)

3.2.3 Soil Sampling Procedure

Schedule of Sampling

For the current study, the ideal choice of time for sampling would be summer, which usually prevails in Denmark from June to August. Experimental studies that assessed the sensitivity of northern peatlands towards increased temperatures, have been found to choose summer (Wang et al., 2017b; Weedon et al., 2013) for collecting samples from the field which can be associated with the fact that climate change is increasing summer temperatures and evapotranspiration in the northern regions and is predicted to intensify the effect in the future as well.

However, during the short duration of the study it has not been possible to schedule the sampling in summer. The collection of samples has been conducted in winter by the cold end of February in 2018. According to the Danish Meteorological Institute (DMI)[3], it was the coldest February in Denmark since 2010 and had a higher number of frost days than the normal (1961-1990) of 19 days. During the month, Næstved municipality had an average temperature of $-1°C$ that was lower than the monthly average of $-0.8°C$ for the region of Southern Sjælland. Besides this, the municipality had an average precipitation (rainfall and snowfall) of 14.1 mm which was higher than the month's regional monthly precipitation sum of 6 mm. At the time of sampling, the

temperature was below $0°C$ (about $-7°C$) with low to moderate levels of snowfall intensities. Within the hollows the depth of water varied within the range of 20 to 30 cm above the peat surface and due to freezing temperatures the uppermost surface of the hollows were mostly frozen.

Depth of Sample Collection

Peat samples have been collected from the soil layer lying underneath the hollow water where the top and bottom parts refer to the peat sections of 0-10 cm and 10-30 cm depths respectively (figure 3.9). On the basis of Clymo-model (see section 2.1.1), it is assumed that both the acrotelm and catotelm layers of this ombrotrophic bog are present within these two depths.

Procedure of Soil Collection

As the first step of sampling, vegetation cover lying over the hollows are cleared carefully using a large shear. While doing that, the plants are not uprooted rather cut from the bottom so that the acrotelm layer of the peat can be kept undisturbed. At the next step, peat samples have been collected with the help of a "Edelman" soil auger that is specialized for sampling clayey, boggy and root-bound soils. The augur has three parts- the Edelman sampler at the bottom, extension rod attached with the sampler to penetrate deeper into the soil and a T-shaped handle to twist, push and pull the auger.

Initially, the frozen upper layers of the hollows ruptured by slowly striking the surface with the bottom part of the auger. Then the auger is inserted into the hollows as vertical as possible until the required depth is reached. Upon reaching the soil, the auger is marked with a meter stick to measure the water level above the soil surface. Then the auger is pushed further 10 cm and twisted clockwise for at least three to four times so that required peat soils are captured by the sampler. Afterwards the auger is pulled out slowly and the sample is transferred to a 0-10 cm marked plastic bucket with the help of a knife. For 10-30 cm samples, similar process is repeated at the same hollow, except the sampler is pushed until 30 cm depth, and transferred to a 10-30 cm marked plastic bucket.

Based on the total number and dimension of the semi-intact cores to be prepared at lab and the amount of soil to be used in the background analysis, an estimated amount of 8000g (17lb) has been collected from each depth. At the end of the sampling, the buckets are covered with thick black polythenes so that the soil can be kept devoid of any light penetration. Finally, the buckets

[3]DMI Website- `http://www.dmi.dk/vejr/`

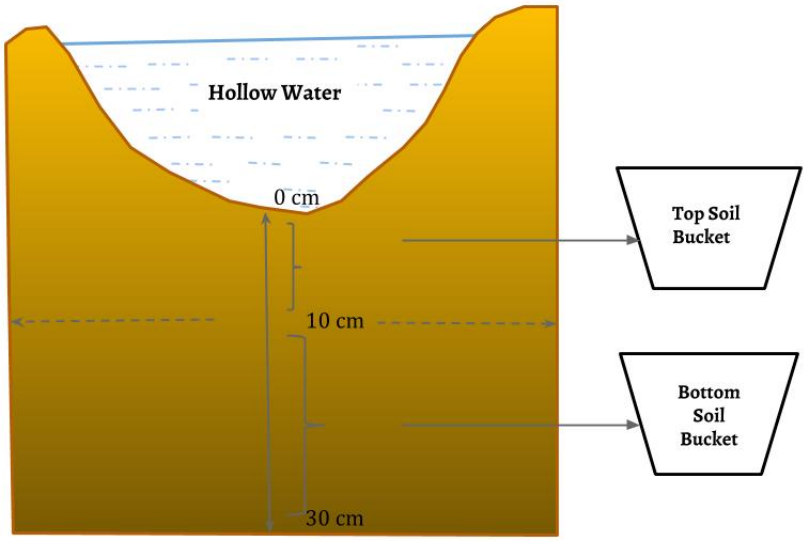

Figure 3.9: Peat sample collection across the depth of a peat profile lying underneath the hollows

are transported to the $4°C$ climate room at RUC for the preparation of semi-intact cores as well as for the background analysis.

3.2.4 Background Analysis

For background analysis, organic carbon content (C_{org},%), water content (on a wet mass of sample basis) and pH of the peat soils have been calculated. In order to obtain Corg values, the dry ashing technique or the Loss on Ignition (LOI) method has been implemented on a total of 20 peat samples (10 from the 0-10 cm depth and 10 from the 10-30 cm depth soils) each having approximately 12 grams of peat. In porcelain crucibles, the wet samples are oven dried for about 48 hours at $105°C$ to determine the moisture content (W_w, g/g). At this stage, the dried mass of samples (W_{ds}) have both organic matter and ash content, hence, to estimate the C_{org}, all the samples are combusted in a muffle or furnace oven at $550°C$ for 5 hours. Upon combustion, the weight loss on ignition and the weight of ash content (W_{ash}) are measured to calculate the dry weight (g of C/g of dry soil) and percent C_{org} values as follows,

$$C_{org} = \frac{1}{1.724}[(W_{ds} - W_{ash})/W_{ds}] \tag{3.1}$$

Here, $1/1.724$ is a conversion factor that is used while converting organic matter to organic carbon content as the mass loss during the (Agus et al., 2010). The pH values of top and bottom

peat are obtained separately. For the measurement, samples have been prepared by adding 20 ml of deionised water to 20 grams of air-dried peat soil (1:1) in a plastic beaker each time. To make a suspension, the mixture is shaken by a shake-machine for 40 minutes and then kept in rest for 20 minutes before inserting the pH electrode probe. The pH meter has been calibrated each time to the maintain the accuracy of measurement.

3.3 Experimental Design

3.3.1 Treatment Temperatures and Water Levels

The experimental part of the study has been designed to measure the fluxes of CH_4 and CO_2 gases from semi-intact peat cores at increased temperatures and reduced water levels. Here, the term "treatment temperatures" refer to the specific temperature levels to which the sample peat cores will be exposed for a defined period of time. In order to observe the effect of climate-change induced temperature increase on the GHG emissions of peat samples from Holmegård, it is reasonable to have treatment temperatures that represent the future warming levels predicted particularly for Denmark. According to the IPCC report on Danish climate change scenario, by the end of this century (2081-2100), the temperature is estimated to increase around $1.2°C$ in both summer and winter for the mild scenario-RCP2.6[4] compared to the reference period of 1986-2005 (Olesen et al., 2014). For the high scenario-RCP8.5, the heating is expected to be $4.0°C$ and $3.7°C$ in summer and winter respectively (Olesen et al., 2014).

As the sampling is scheduled in winter, one method considered for determining the treatment temperatures was to select two temperature levels:

1. Winter mild temperature (current average winter temperature $+1.2°C$)
2. Winter high temperature (current average winter temperature $+3.7°C$)

For instance, if the average winter temperature of Næstved municipality is $-1°C$, the treatment temperatures would be (-1 + 1.2) or $0.2°C$ and (-1 + 3.7) or $2.7°C$ based on the estimated increase of winter temperature for mild scenario-RCP2.6 and high scenario-RCP8.5 respectively.

[4]RCP scenario: Representative Concentration Pathways or RCPs refer to a set of scenario that contains GHG emission, concentration and land-use trajectories adopted by IPCC. On the basis of the estimated levels of GHG emissions in future, four possible climate scenarios in future are described by RCP2.6, RCP4.5, RCP6, and RCP8.5 where the number indicate four possible ranges of radiative forcing values in 2100 relative to the pre-industrial values, +2.6, +4.5, +6.0, and +8.5 W/m^2 (Van Vuuren et al., 2011).

While this approach would replicate a realistic increase of temperatures, such small increments of change or increase may not reflect any significant changes among the samples in terms of GHG emission. As per the discussions in section 2.2.2 that increase may not be enough to eventually make an impact on the rates of underlying biochemical reactions responsible for CH_4 production and consumption. Even if the fluxes change, there remains a high possibility that the changes are caused by measurement error or by the heterogeneity among the samples. The outcomes from such little intervals of temperature may not also provide a trend to conclude the direction of change e.g. whether it is positive or negative. Therefore, larger temperature intervals are considered between the treatments. As per Q_{10} values, the rates of microbially driven reactions in soils may change by a rise in temperature of $10°C$ (Q_{10}) (section 2.2.2). Assuming that for the current study as well and considering the availability of climate rooms at RUC, the temperature treatments have been finally performed with three temperatures $4°C$, $17°C$ and $25°C$

Two different water levels have been maintained in the peat cores during the experiment. Depending on the level of water, the 30 cm long peat cores have been divided into following categories:

- Peat cores with high water level or HW cores (water table lies above soil surface)
- Peat cores with low water level or LW cores (water table lies 10 cm below the surface)

Here, it is assumed that the HW cores represent the natural conditions of hollow where the top peat is submerged under water year long and the LW cores represent the situation of drought where water level may potentially decrease to a depth of 10 cm and expose the top peat layer to aeration.

3.3.2 Soil Core Preparation

The semi-intact cores have been prepared in transparent plastic tubes. Each of the tubes has an inner diameter of 5 cm and length of 35 cm with two ends open. A total number of 16 peat cores has been prepared by filling the tubes with the top (0-10 cm) and bottom (10-30 cm) peat soil samples where each time about 5 cm length above the top part is kept as headspace (figure 3.10). In order to maintain the water levels, required amount of tap water has been added to the cores on a weekly basis. Because of the compressive nature of soil, the headspace has been changed throughout the experiment. On an average, the length of the headspace has mostly varied within a range of 4 to 6 cm. For closing the top and bottom ends, truncated cone-shaped rubber stoppers are used for the core tubes. However, the top rubber caps have been modified according to the method of flux measurement (section 3.3.5).

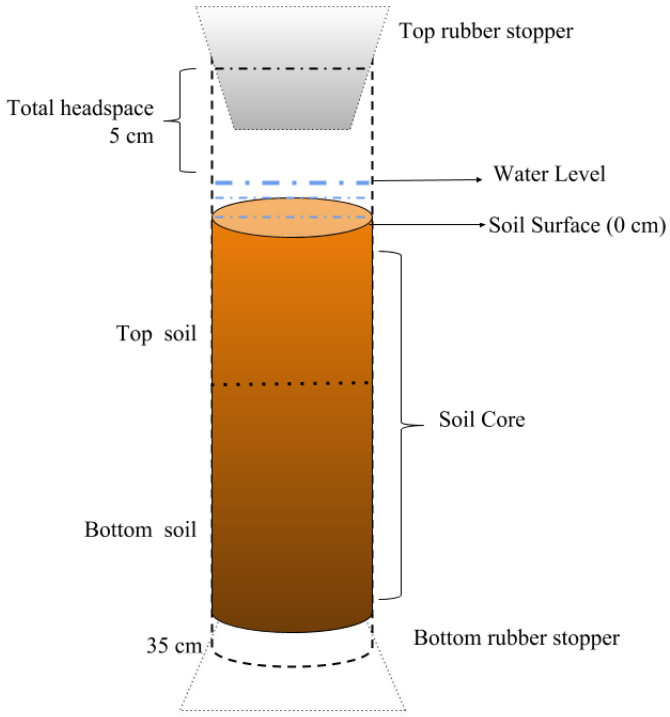

Figure 3.10: The structure of a HW type semi-intact peat core

3.3.3 Treatment Procedure

The treatment procedure has included a total number of 22 cores- 16 semi-intact peat cores, 3 intact background peat cores and 3 blank cores without soil. Initially all these cores are kept in $4°C$ climate room for 7 days. After that, 8 of the semi-intact cores (4 HW cores and 4 LW cores) and a blank core have been transferred to the climate room of $17°C$ while rest of the semi-intact cores with another blank core have been transferred to the climate room of $25°C$ The background cores are kept untreated throughout the experiment.

3.3.4 Flux Measurement Timelines

At $4°C$climate room, all the sample peat cores- intact, semi-intact and blank, are incubated for 7 days. During the period, GC measurements of CH_4 and CO_2 concentrations in the headspace air samples are performed twice:

- At the end of the first 24 hours (24-hour measurements)
- At the end of 7 days (1-week measurements)

At $17°C$ and $25°C$, GC measurements of CH_4 concentrations at the semi-intact and blank samples are performed at the following time intervals:

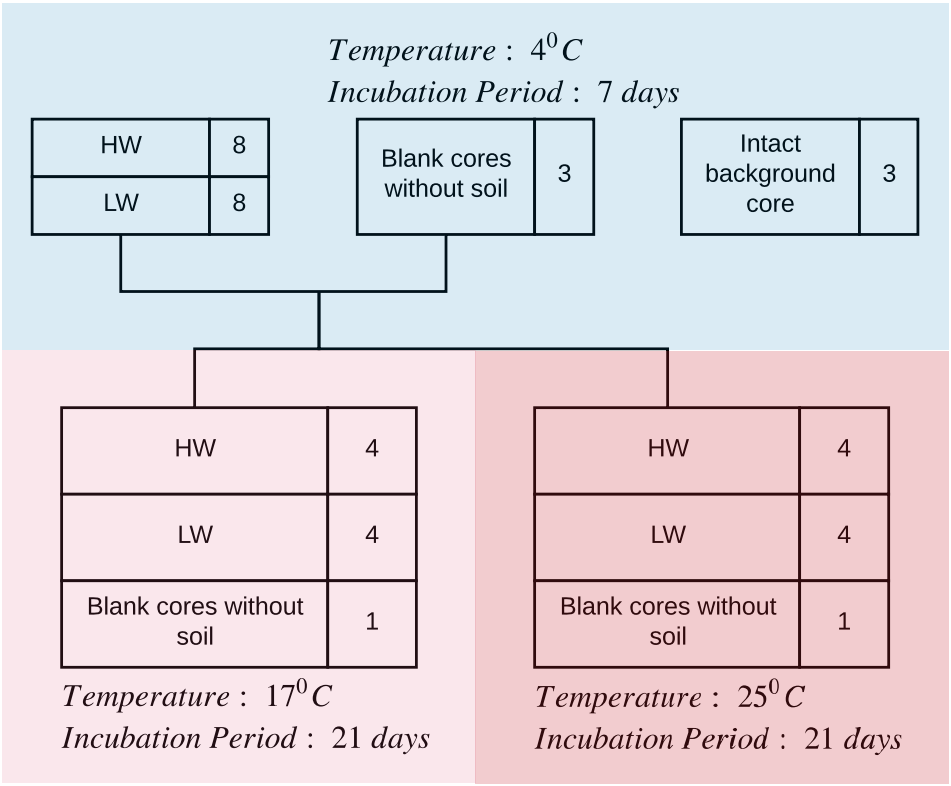

Figure 3.11: Components and duration of the treatment procedure

- At the end of the first 24 hours (24-hour measurements)
- At the end of 7-days (7-day measurements)
- At the end of 14 days (14-day measurements)
- At the end of 21days (21-day measurements)

At $17°C$ and $25°C$, GC measurements of CO_2 concentrations at the semi-intact and blank samples are performed only once at the end of 21-days over the total period of incubation. The limited measurements of CO_2 is not an experimental choice rather an adjustment that is made due to the erroneously low concentrations found by the CO_2 probe during the first 2-weeks of incubation.

3.3.5 Headspace Air Sampling

CH_4 and CO_2 fluxes from the soil cores have been measured by sampling gas samples from the headspaces of above the surface of the soil column. In order to prepare the air-tight lids, bottom-penetrated eppendorf tubes are tightly inserted through a 9 mm hole in the rubber stoppers. Then the openings of eppendorfs are closed by septa ensuring minimal leakage of air.

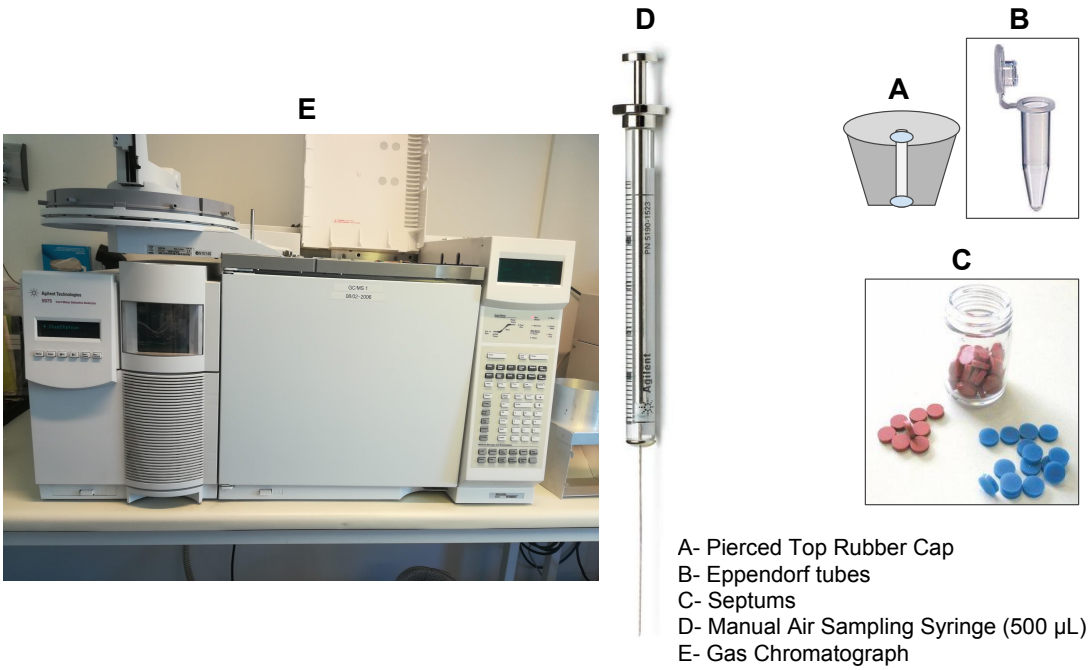

Figure 3.12: Equipment used in the measurement of gas concentrations

The caps of eppendorfs are kept closed for 24-hours before gas sampling. During sampling the caps are opened and the gas sampling syringe is inserted carefully through the septum to the headspace (figure 3.12).

For measuring CH_4 and CO_2 concentrations, $500\ \mu L$ and $100\ \mu L$ air samples have been collected by gas sampling syringe respectively (figure 3.12). In order to avoid leakage of air, gas samples are not taken into the vials rather injected directly into the GC machine carefully. The concentration of standard CH_4 and CO_2 were 10 ppm. Each time before gas sampling, the syringe has been flushed by N_2 gas so that contamination of gases can be avoided. After each 15 measurements, the GC septum has been changed as frequent injections through the GC septum can cause leakage of gas samples during measurement.

Chapter 4

Results and Discussions

The results and analysis section begins with the results that are obtained from the soil property analysis of peat samples of two consecutive depths at the bog of Holmegård. Based on the respective depths of the samples, the soil analysis results i.e. C_{org} $(\%)$, pH and moisture content (wet weight (g/g) basis) are presented by mean +/- standard deviation (SD) values per depth.

The estimated results of CH_4 and CO_2 fluxes from the sample peat cores at varying temperatures and water levels are presented with relevant analysis later in this section. Instead of describing the flux data derived from each of the experimental cores, the section provides summarized CH_4 and CO_2 flux data as mean +/- SD values in molar units i.e. $\mu mol\ CH_4\ m^{-2}h^{-1}$ as well as in mass units i.e. $g\ CH_4 - C\ m^{-2}y^{-1}$ (See Appendix A for the detail of CH_4 and CO_2 flux results and their associated calculations and unit conversion procedures). Following the emission results, analysis of the differences in flux outcomes are also presented in this section based on the underlying assumptions and limitations of the study. Instead of providing an overall estimation of change (increase or decrease) in peat soil CH_4 and CO_2 emissions with increase in temperature and decrease in water level, the analysis attempts to explain the trajectory of CO_2 and CH_4 emissions in the experimental cores at different levels of temperature and water.

The discussion section presents explanations for the magnitude and pattern of CH_4 and CO_2 flux outputs on the basis of the experimental factors. It provides final interpretations of the experimental emission values in the context of global warming and climate change. The section ends with a thorough discussion on the implication and significance of the current study and its outcomes with reference to other relevant studies and their findings on northern peatlands' GHG emission levels at high temperatures and low water table conditions.

4.1 Soil Property Analysis

The values of water content, pH and organic C content of the top soil samples have minimal difference with the values obtained from the bottom soil samples (table 4.1). The mean pH values lie within the range of 3.5 - 5 that is typical for northern peatlands (see section 2.1.3). As a raised ombrotrophic bog, the pH results are also consistent with the pH found at other northern raised bogs i.e. pH 3.7 (Roulet et al., 2007) and pH 4.2 (Mander et al., 2012) have been found at a Canadian and Estonian raised bog respectively. Until the depth of 30 cm, the organic C content (C_{org}) values are considerably higher than the minimum percentage of C_{org} (20%) suggested for northern peatlands (Group et al., 1998; Rezanezhad et al., 2016). In table 4.1 the water content values are given on a wet mass of sample basis, however, on a percent basis the moisture contents have exceeded 100% (Appendix A.2). This is because organic materials in peat soils can hold water several times higher than their dry weight (O'Kelly and Sivakumar, 2014).

Table 4.1: Soil characteristics of the top and bottom peat samples from the Holmegård bog

Soil depth (cm)	**Water content by weight, W_w, g/g**		**pH**		**Organic C content (%)**	
	Mean	*SD*	*Mean*	*SD*	*Mean*	*SD*
Top soil (0-10 cm)	8.67	0.56	3.92	0.26	54.19	1.19
Bottom soil (10-30 cm)	9.24	0.44	3.28	0.16	53.93	0.49

4.2 CH_4 and CO_2 Emission from the Peat Cores at $4°C$ Temperature

4.2.1 CH_4 Flux Results

At $4°C$ temperature, CH_4 concentrations in the headspace air samples of all the semi-intact peat cores (HW and LW samples) and within the blank cores have been found to be below the detectable limit in the first 24-hour and 7-day measurements. However, intact background cores have produced measurable levels of CH_4 at $4°C$ temperature which have been detected not only in the first 24-hour measurements but also in the 7-day measurements. The rates of increases in CH_4 concentrations at the 3 intact cores have varied from 41.67 $ppmh^{-1}$ to 37.50 $ppmh^{-1}$ in the first 24-hour measurements leading to the estimated flux of 49.57+/- 9.08 $\mu mol\ CH_4\ m^{-2}h^{-1}$ and annual flux of 5.21+/-0.96 $g\ CH_4 - C\ m^{-2}y^{-1}$.

In the 7-day measurements, CH_4 concentrations at the respective intact cores have mostly showed lower rates of increase compared to that of the 24-hour measurements. The flux result derived from the 7-day measurements is 37.96+/-8.47 $\mu mol\ CH_4\ m^{-2}h^{-1}$ or, on annual basis, 3.99+/-0.89 $g\ CH_4 - C\ m^{-2}y^{-1}$.

4.2.2 CO_2 Flux Results

The concentrations of CO_2 have been detected in the air samples of both intact and semi-intact soil cores at the temperature of $4°C$. In the blank cores without soil, CO_2 concentrations remained almost constant at approximately 475 ppm during the first 24-hour and 7-day measurements as expected.

Assuming the blank value of 475 ppm as the initial CO_2 concentrations for all of the sample peat cores, the intact cores CO_2 concentrations have been found to increase at rates of 1.46 $ppmh^{-1}$, 2.96 $ppmh^{-1}$ and 2.71 $ppmh^{-1}$ during the first 24-hour measurements leading to the flux of 2.95+/-0.89 $\mu mol\ CO_2\ m^{-2}h^{-1}$ or 0.31+/-0.09 $g\ CO_2 - C\ m^{-2}y^{-1}$. After a week, CO_2 concentrations in the samples cores have increased at higher rates and produced higher levels of CO_2 flux that is 11.51+/-1.35 $\mu mol\ CO_2\ m^{-2}h^{-1}$ or 1.21+/-0.14 $g\ CO_2 - C\ m^{-2}y^{-1}$. However, compared to the CH_4 fluxes of intact cores at $4°C$ temperature, the intact core CO_2 fluxes are really low.

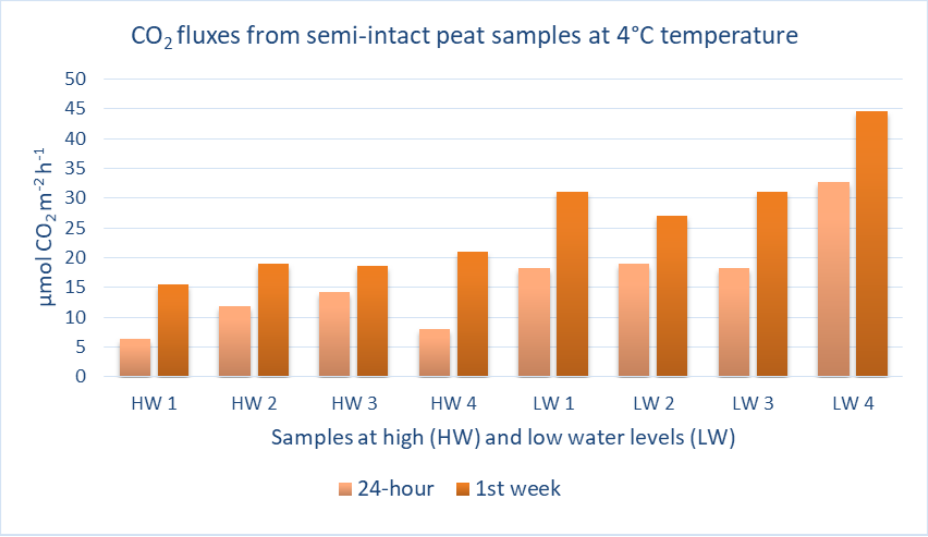

Figure 4.1: Emission of CO_2 gas at $4°C$ temperature from the semi-intact peat samples differentiated by high and low water levels

Both 24-hour and 7-day measurements have showed that the CO_2 fluxes derived from the low water level semi-intact cores (LW samples) are higher than the CO_2 fluxes from the high water level semi-intact cores (HW samples) (figure 4.1) indicating a positive relationship between

lower water levels and CO_2 emissions.

During the 24-hour measurements, CO_2 fluxes generated by HW peat cores have varied between a range of 6 to 15 $\mu mol\ CO_2\ m^{-2}h^{-1}$ whereas the CO_2 fluxes from LW peat cores have varied within a wide range of 18 to 33 $\mu mol\ CO_2\ m^{-2}h^{-1}$ with 3 cores having almost similar values of emission i.e. 18.25, 18.97 and 18.26 $\mu mol\ CO_2\ m^{-2}h^{-1}$ and 1 core having much higher value of emission that is 32.75 $\mu mol\ CO_2\ m^{-2}h^{-1}$. On a yearly basis, the estimated mean CO_2 fluxes of the HW and LW samples are 1.07+/-0.38 $g\ CO_2 - C\ m^{-2}y^{-1}$ and 2.32+/-0.75 $g\ CO_2 - C\ m^{-2}y^{-1}$ respectively. It is important to note that, all these fluxes are higher in comparison with the 24-hour intact core CO_2 fluxes stated above.

In the 7-day measurements, an overall increase of CO_2 fluxes can be found in almost all the semi-intact samples though the amount of increase has been highly variable within the HW and LW samples as well as between the HW and LW samples. CO_2 emissions from HW samples have varied between a range of 15 to 22 $\mu mol\ CO_2\ m^{-2}h^{-1}$ while CO_2 emissions from LW samples have varied within a range of higher values, from 30 to 45 $\mu mol\ CO_2\ m^{-2}h^{-1}$. On annual basis, mean CO_2 flux is estimated to be 1.94+/-0.24 $g\ CO_2 - C\ m^{-2}y^{-1}$ for the HW peat cores and 3.52+/-0.81 $g\ CO_2 - C\ m^{-2}y^{-1}$ for the LW peat cores which are again noticeably higher than the 7-day CO_2 emissions of intact cores.

Based on the above results, the following key findings can be outlined for $4°C$,

1. Intact cores emit more CH_4 than CO_2
2. Semi-intact cores emit more CO_2 than the intact cores
3. Low water level cores emit more CO_2 than the high water level cores

4.3 CH_4 Emissions from the Peat Cores at $17°C$ Temperature

In the first 24-hour measurements, CH_4 concentrations in the headspace air samples of all the semi-intact and blank cores have been below the detection limit. Therefore, considering the changes in CH_4 concentrations as 0 $ppmh^{-1}$, the CH_4 fluxes estimated for all the cores at 24-hour have been 0 $\mu mol\ CH_4\ m^{-2}h^{-1}$. In the following two sections, CH_4 fluxes derived from the high water and low water level peat cores at 7-day, 14-day and 21-day measurements are presented separately.

4.3.1 CH_4 Flux Results of HW Samples

In the 7-day measurements, the semi-intact peat cores have started to emit measurable amounts of CH_4 except two of the HW samples and the blank core. Among the HW samples, only 2 cores (sample 3 and 4) have found to produce CH_4 fluxes while the other two have showed no changes in their concentration (0 flux) (figure 4.2). For the samples, the situation remained same even in the 14-day measurements where the increase in CH_4 concentrations have taken place only at sample 3 and 4 inducing fluxes of 2.48 and 2.63 $\mu mol\ CH_4\ m^{-2}h^{-1}$ respectively and no detectable change in CH_4 concentrations at sample 1 and 2 (figure 4.2).

However, in the 21-day measurements, sample 2 has started to emit CH_4 with sample 1 still lacking any measurable emission of CH_4. Compared to the 14-day measurements, there has been no increase in CH_4 fluxes at other two samples rather sample 3 showing a decline in its CH_4 emission (figure 4.2). As per the 21-day measurements, the HW samples have emitted an average flux of 1.26+/-1.08 $\mu mol\ CH_4\ m^{-2}h^{-1}$ or 0.1325+/-0.11 $g\ CH_4 - C\ m^{-2}y^{-1}$.

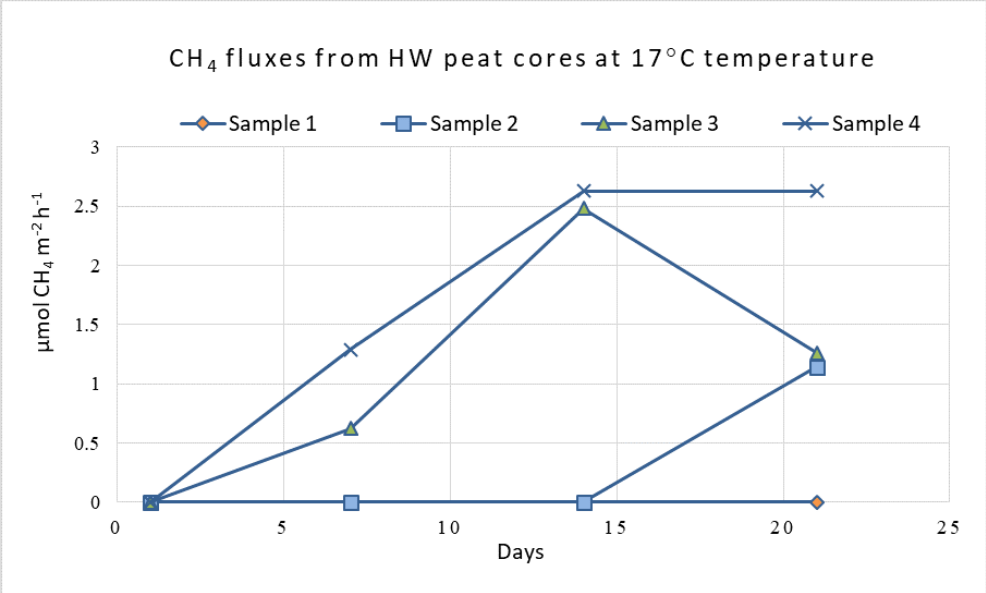

Figure 4.2: Emission of CH_4 gas at $17°C$ temperature from the semi-intact peat samples with high water levels (HW)

4.3.2 CH_4 Flux Results of LW Samples

Unlike the HW samples, all the semi-intact cores with low water levels (LW) have produced CH_4 fluxes during the 7-day measurements at $17°C$ temperature. The CH_4 concentrations in the headspace air samples of LW cores have been found to generate average CH_4 flux of 1.23+/-0.85 $\mu mol\ CH_4\ m^{-2}h^{-1}$ (0.13+/-0.09 $g\ CH_4 - C\ m^{-2}y^{-1}$) that is higher than the average

flux of HW samples at the end of the first week of incubation. After 2 weeks, each of the LW samples has showed a substantial increase in their headspace CH_4 concentrations (figure 4.3). The resulting flux of 4.28+/-1.43 $\mu mol\ CH_4\ m^{-2}h^{-1}$, have also been noticeably higher than the 14-day CH_4 emission of HW samples. In the 21-day measurements, CH_4 fluxes derived from LW cores have been found to both increase and decrease in a rather gradual manner compared to the abrupt fluctuations of the fluxes present in the 21-day measurements of HW samples. However, even at $17°C$ temperature, the fluxes derived from the HW and LW samples have not reached the CH_4 fluxes of $4°C$ intact cores.

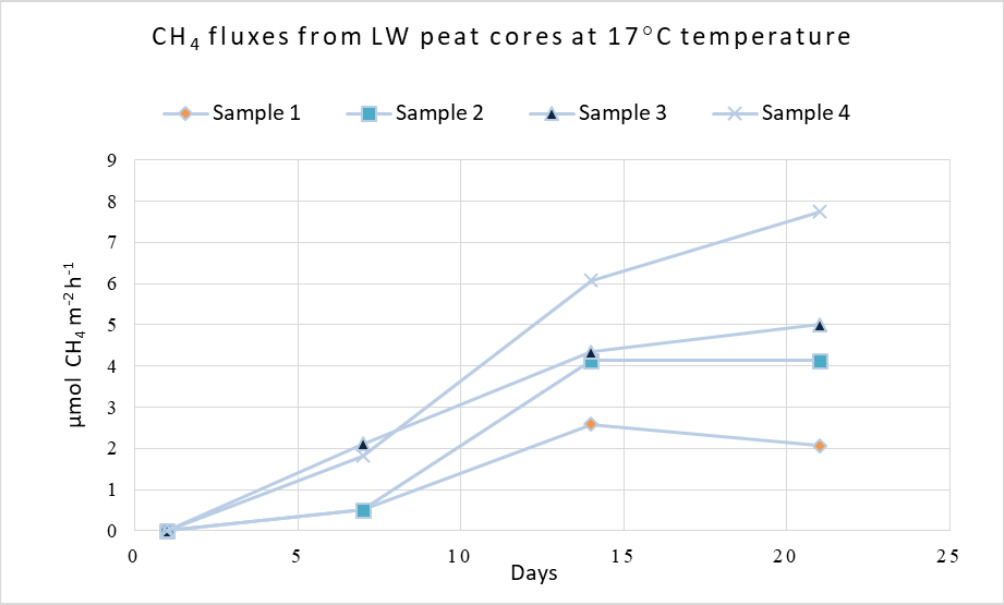

Figure 4.3: Emission of CH_4 gas at $17°C$ temperature from the semi-intact peat samples with low water levels (LW)

Based on the above results, the key finding has been as follows-

1. LW samples emit more CH_4 than the HW samples indicating that reduced water levels cause more CH_4 emission with the rise in air temperature

4.4 CH_4 Emissions from the Peat Cores at $25°C$ Temperature

CH_4 concentrations in the headspace air samples of all the semi-intact and blank cores have been below the detection limit in the first 24-hour measurements. Assuming the changes in CH_4 concentrations as 0 $ppmh^{-1}$, the CH_4 fluxes estimated for all the cores at 24-hour have been 0 $\mu mol\ CH_4\ m^{-2}h^{-1}$. In the following two sections, CH_4 fluxes derived from the HW and LW peat cores at 7-day, 14-day and 21-day measurements are presented separately.

4.4.1 CH_4 Flux Results from HW Samples

The semi-intact peat cores with high water levels have produced higher amounts of CH_4 fluxes in the 7-day measurements if compared with the CH_4 fluxes produced by $17°C$ HW soil cores. Although there still remains sample 1 that has not been able to show detectable CH_4 levels even at this highest temperature of $25°C$ (figure 4.4). After 14 days of incubation, situation has remained the same for sample 1 and other samples have presented lower values of fluxes than that of 7-day measurements as well as 14-day measurements of $17°C$ HW cores.

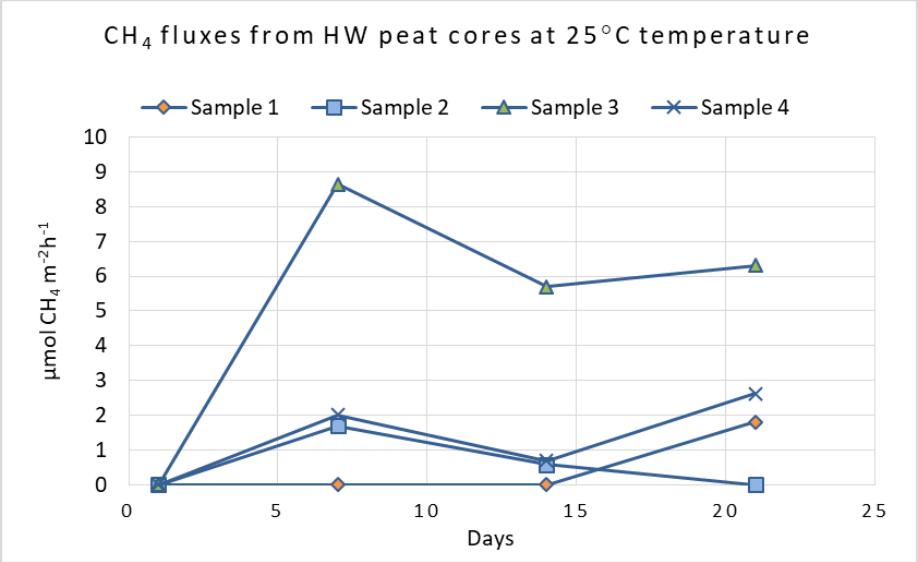

Figure 4.4: Emission of CH_4 gas at $25°C$ temperature from the semi-intact peat samples with high water levels (HW)

In the 21-day measurements, CH_4 concentrations have increased at higher rates in all the HW samples except sample 2. It seems that throughout the 21-days incubation at $25°C$ the high water cores have been unstable with some of the measurements showing very high flux values e.g. 8.64 $\mu mol\ CH_4\ m^{-2}h^{-1}$ for sample 3 at day 14. Due to the frequent high, low and even no values, it is difficult to assert whether the cores are moving towards higher CH_4 emissions or towards lower CH_4 emissions as a result of their exposure to an increase of $21°C$. The 7-day, 14-day and 21-day measurements have given mean CH_4 flux results of 3.09+/-3.80, 1.745+/-2.65 and 2.68+/-2.65 $\mu mol\ CH_4\ m^{-2}h^{-1}$ respectively. The extreme high values of standard deviation also indicate the prevailing high variability within the flux data of HW samples at $25°C$ temperature.

4.4.2 CH_4 Flux Results from LW Samples

Similar to the LW cores at $17°C$ all the LW samples at $25°C$ have emitted CH_4 in the 7-day measurements. The rate of CH_4 production has been quite higher in the LW cores than the HW ones

indicating that lowering of water table has been more conducive to CH_4 emissions at higher temperatures. The LW samples have been found to emit flux of 14.55+/-11.97 $\mu mol\ CH_4\ m^{-2}h^{-1}$ with a low and extreme high value of 6.48 and 31.38 $\mu mol\ CH_4\ m^{-2}h^{-1}$ respectively. The high value is significantly distant from the other observations and can be considered as an outlier that usually derives from measurement or experimental errors.

In the 14-day measurements, CH_4 fluxes have substantially dropped in all of the LW cores causing lower emission of 2.44+/-1.85 $\mu mol\ CH_4\ m^{-2}h^{-1}$ similar to that of the $17°C$ LW cores. However, the flux reductions in $25°C$ LW cores from day 7 to 14 have been steeper than the flux reductions in $17°C$ LW cores from day 7 to 14. But that is mainly because of the presence of an outlier in the 7-day measurements of the LW samples.

After the end of 2nd week, a smaller increase in CH_4 flux has been found in each of the LW cores which are again higher than that of the $17°C$ LW cores (figure 4.5).

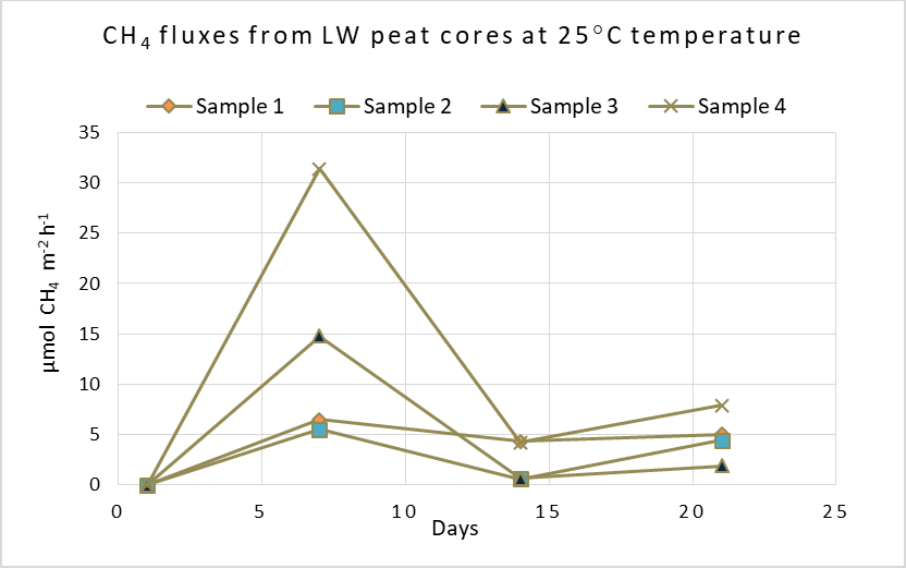

Figure 4.5: Emission of CH_4 gas at $25°C$ temperature from the semi-intact peat samples with low water levels (LW)

Based on the above results, the following key findings can be outlined for $25°C$,

1. CH_4 emission is higher in reduced water level samples but still lower than that of the intact cores of $4°C$
2. Both HW and LW CH_4 flux data are highly variable and unstable over the period of measurement

4.5 CO_2 emissions at $17°C$ and $25°C$ temperature

Higher CO_2 fluxes have been observed in the LW samples at higher temperatures (figure 4.6) indicating that temperature increase can induce more CO_2 emission in unsaturated peat soils than that of the saturated ones.

With the shifts to $17°C$ and $25°C$ temperatures from $4°C$, the variability within CO_2 flux data of HW samples has been found to increase. The higher standard deviations found in the mean CO_2 values of HW cores (27.6+/-15.59 $\mu mol\ CO_2\ m^{-2}h^{-1}$ at $17°C$ and 82.46+/-51.66 $\mu mol\ CO_2\ m^{-2}h^{-1}$ at $25°C$) are also indicative of the considerable variability. In case of both HW and LW flux data, highest variability has been observed at the $25°C$ emission values (similar to the CH_4 fluxes at $25°C$ temperature).

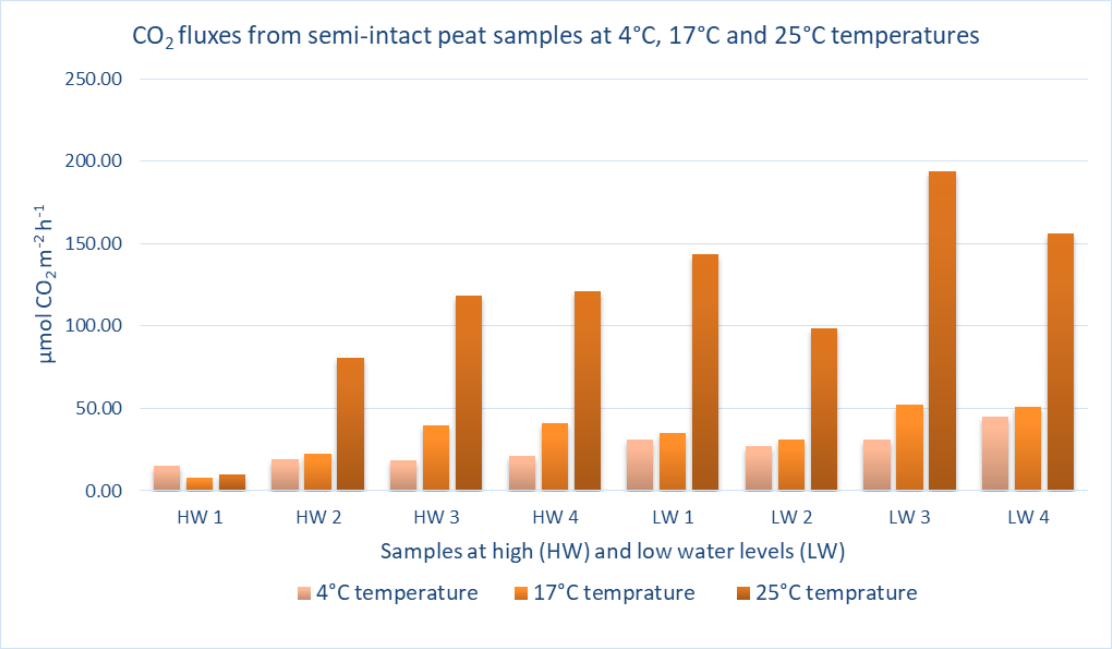

Figure 4.6: Emission of CO_2 gas at $4°C$, $17°C$ and $25°C$ temperature from the HW and LW semi-intact peat samples

4.6 Discussions

4.6.1 Sample Cores at $4°C$ Temperature

On the basis of the results given in section 4.2, it is apparent that the emission outcomes of intact and semi-intact cores at $4°C$ temperature have been significantly different from each other. CH_4 emissions derived from the intact cores have been considerably higher than that of the

semi-intact cores while CO_2 emissions from intact cores have been lower than that of the semi-intact peat cores over the time of measurement indicating that the intact cores of peat soil can be different from their semi-intact counter parts. Even within the same category of semi-intact cores e.g. HW level samples, varying levels of fluxes are found both in case of CO_2 and CH_4. Such differences contradict with the basic assumption of the experiment which considers all the semi-intact cores to be homogeneous and differentiated only by the level of water.

The difference can be attributed to certain experimental factors such as the manipulation of soil samples during the preparation of semi-intact cores. Soil structure has been found to influence production and transport of gases through soil besides temperature and water-filled pore spaces as any changes to the structure can also influence microbial activity within soil (Van Veen and Kuikman, 1990; Smith et al., 2003; Ball, 2013). According to Van Veen and Kuikman (1990), within soil, organic matter is physically protected by mechanisms like adsorption of organics on clay surfaces and entrapment of materials in places i.e. as aggregates where microbes have minimal access (Van Veen and Kuikman, 1990). Manipulation of the peat soil structure, as performed in this study, may have disturbed the physical protection to an extent where the experimental cores have not only become heterogeneous in terms of their physical structure but also in their respective status and rate of microbial processes at the temperature of $4°C$ and beyond.

Increasing air permeability in the soil as a result of disturbance can cause significant changes in the CH_4 and CO_2 emissions (Smith et al., 2003). During the preparation of semi-intact cores at the lab, no mechanisms have been followed to make the cores entirely anaerobic, hence there remains a reasonable possibility that the HW cores have not been anoxic or possibly at a pre-stabilization stage where the artificial addition of water has not been yet started to increase the anoxic CH_4 production at the deeper parts of the column. Aeration is also significant as it can make the soil act more as a CH_4 sink rather than as a CH_4 source by converting atmospheric CH_4 into CO_2 (Smith et al., 2003; Ball, 2013). This can happen to the semi-intact cores at $4°C$ since at the same temperature intact cores have showed sufficient emission of CH_4 but lower emissions of CO_2.

The literatures have varied statements regarding the depth-wise delineation of oxic-anoxic zones or between the depth of acrotelm and catotelm zones (section 2.1.1) indicating that such zonation can vary from one peatland to another. If aeration is high in soil the oxic zone will be increased leading to higher CO_2 and lower CH_4 fluxes (Abdalla et al., 2016; Wang et al., 2017a). Therefore, the higher rates of CO_2 production in the semi-intact cores can be resulted from two

reasons-

- The disturbance of peat soil during core preparation leading to higher infusion of oxygen
- The assumption of acrotelm-catotelm boundary being situated at 10 cm depth which may not be appropriate in case of Holmegrd bog

Regarding the later cause it is important note that, it is still unexplored at what depth the oxic and anoxic boundary lies seasonally at Holmegård bog and how much CH_4 and CO_2 emission can possibly be emitted from the bog with the changes in that boundary. Therefore, in contrast to the study's assumption, it is likely that in reality, the bog's layer of high decomposition (acrotelm) lies beyond the depth of 10 cm. If so, then the experimental cores of 30 cm depth may have greater oxic zones compared to the anoxic zones leading to higher CO_2 results.

As explained in section 2.2.2 methanogenesis can occur in a wide range of low to high temperatures and hence it is not unlikely to have CH_4 emissions at lower temperatures like $4°C$. Alm et al. (1999), by using closed chamber and snowpack diffusion method, have found mean CH_4 emission of 1.0 $g\ CH_4 - C\ m^{-2}d^{-1}$ from a Finnish natural bog even during January when the mean temperature was $-11.9°C$. While investigating how temperature affects the rates and pathways of CH_4 production in the acidic peat of a mire in northern Scandinavia, Metje and Frenzel (2005) have found considerable rates of methanogenesis (0.25 $\mu mol\ CH_4\ g$ per dry weight per day even at $4°C$ temperature. Hence, despite the different rates of CH_4 emission, average CH_4 flux results derived from the intact cores at $4°C$ can be considered as consistent with those studies. The undetectable CH_4 emission of semi-intact cores, on the contrary, can be explained as a consequence of physical disturbances that may have increased the activity of CH_4-oxidising methanotrophs in the presence of oxygen (Lai, 2009) and the emission of CO_2.

The output of higher and lower CO_2 fluxes from LW and HW semi-intact samples respectively (figure 4.1) can be explained by their relative levels of water. In each of the LW cores, the extent of unsaturated zone has been higher which have led higher rates organic matter decay and CO_2 than that of the HW cores (Aurela et al., 2007; Webster et al., 2013; Wang et al., 2017a). Moreover, as the water table is above the soil surface in HW cores, the diffusion of CO_2 from the soil column to the atmosphere can be slowed down, leading to lower CO_2 fluxes than the LW cores (Fang and Moncrieff, 1999; Wang et al., 2017b).

4.6.2 Sample Cores at $17°C$ and $25°C$ Temperature

From the measurements at $17°C$, it is visible that CH_4 fluxes derived from the samples with HW have changed in a very fluctuating and unstable manner over the entire incubation period of 21

days (figure 4.2). The fluxes are also very low compared to the $4°C$ CH_4 fluxes of intact cores indicating that an increase of $13°C$ temperature may have only triggered the microbial production of CH_4 in certain HW cores but failed to induce higher rates of emission within the time of incubation. The emission behavior of HW samples becomes more unusual when considered in terms of water level. Theoretically, high water level should induce more anoxic condition suitable for anaerobic CH_4 production (Wang et al., 2017a) and a considerable increase in temperature i.e. $10°C$ should result into two-times higher rate of microbial CH_4 production (Lai, 2009) and thereby significantly higher CH_4 fluxes than that of the $4°C$ temperature. The flux results of the HW samples exposed to a substantial temperature shift of $13°C$ have contradicted with the theoretical assumption as the estimated HW CH_4 fluxes are considerably low if compared to the intact core CH_4 emissions at $4°C$ temperature.

After the shift from $4°C$ to $25°C$ temperature, although the emission has increased, the variability within data has been quite high meaning that not all of the HW cores are responding to the temperature increase similarly. The 1 or 2 extreme high or extreme low values or simply outliers have affected the mean flux, mainly the HW fluxes at higher temperature. Such extreme values are likely to arise from experimental or measurement errors and can only be considered as noises in the experimental dataset that cannot be explained. In the LW flux data, variability is relatively low but the CH_4 emission is high compared to that of the HW samples. The increase in LW samples' CH_4 emissions contradicts with the theoretical assumption of decreasing CH_4 with reducing water table or anoxic zone (Sundh et al., 1995; Strack et al., 2004; Lai, 2009; Abdalla et al., 2016).

The contradictions outlined above are not unlikely as some of the previous studies have found such results as well. For instance, increase in water level above the peat surface in flooded peatlands often causes lower CH_4 emissions due to the oxygenated water column that oxidizes CH_4 (Bubier, 1995; Lai, 2009). This is relatable to the lower CH_4 emissions from HW cores even at higher temperatures as the increase in temperature will not only increase microbial CH_4 production but also the microbial oxidation of CH_4 in the water column above peat surface (van Winden et al., 2012). Again, lower water tables may not always decrease CH_4 emissions as assumed. A higher seasonal CH_4 flux with a lower mean water table has been found by Bellisario et al. (1999) in a Canadian peat complex. Moreover, investigation by Treat et al. (2007) has showed positive relationship between CH_4 emission and water table reduction over time in a temperate fen. The study explains the higher rates of CH_4 by higher peat temperatures that facilitated both CH_4 production and its transport via ebullition (see section 2.2.1 and figure 2.5).

The increase in CO_2 fluxes in LW samples with increasing temperature is consistent with the assumption of higher temperatures inducing higher rates of organic matter decomposition in unsaturated peats (Aurela et al., 2007; Charman et al., 2013; Webster et al., 2013; Rezanezhad et al., 2016; Wang et al., 2017a). However the annual fluxes of CO_2 estimated for the $17°C$ and $25°C$ samples (see A.1.2) have been both lower and higher than the fluxes reported by the other studies. For example, Silvola et al. (1996) has found 60 to 200 $g\ CO_2 - C\ m^{-2}y^{-1}$ of emissions from a Finnish natural ombrotrophic bog at the temperature of $12°C$. It further estimates that a lowering of water table by 1 cm increases the CO_2 flux by an average of 9.5 $g\ CO_2 - C\ m^{-2}y^{-1}$ (Silvola et al., 1996). With only water table being changed, Dinsmore et al. (2009) has found mean CO_2 value of 54.7 $\mu g\ m^{-2}h^{-1}$and higher fluxes from lower water tables in peatlands. The value is substantially lower than the CO_2 fluxes obtained in the current experiment. Therefore, it can be stated that, although both water table draw-down and temperature can independently increase CO_2 fluxes, the increase in rate of emission is the highest when both water table is lowered and temperature is increased.

Based on the discussion, it is not unlikely to have higher CH_4 fluxes from LW samples, lower CH_4 fluxes from HW samples and higher CO_2 fluxes from LW samples. However, what is important is that whether there is any statistically significant change across the samples' emission data at different temperatures over the time of measurement.

For wetlands, the expected range of change in flux rates can vary within a wide range i.e. 0.1-6950 $\mu mol\ CH_4\ m^{-2}h^{-1}$ (Oertel et al., 2016). For northern bogs, annual CH_4 emission varies in a narrower range i.e. 0-53.7 $g\ CH_4 - C\ m^{-2}y^{-1}$ (Abdalla et al., 2016) at mean annual temperatures. If only raised bogs (with hummock-hollow CH_4 emissions) are considered, the annual fluxes are found to vary mostly within 0.9-9 $g\ CH_4 - C\ m^{-2}y^{-1}$ at mean annual temperatures that are mostly below the experimental temperatures of $17°C$ and $25°C$ (Abdalla et al., 2016). In other words, when the temperature is higher i.e. $17°C$ or $25°C$, the expected range of CH_4 emissions from northern raised bogs will be greater than 0.9-9 $g\ CH_4 - C\ m^{-2}y^{-1}$. However the annual CH_4 fluxes estimated for the LW and HW cores at $17°C$ and $25°C$ have been mostly within the above range (see results in Appendix A.1.2) indicating that the changes in the experimental flux data (except the few high values of LW samples) with increasing temperatures are below the expectation level.

Since the flux measurements have not been performed over a time period that is sufficient enough to obtain stable results, the high values outside the usual range cannot be interpreted as the initial point of a trend or a signal of increase in CH_4 and CO_2 emissions at the temperature

and water table treatments. Due to the limited number of samples and length of measurement time, rest of the experimental flux data set have also been insufficient to conclude any significant trend of change in the CH_4 emissions at Holmegård bog with higher temperatures and lower water levels. Therefore, although the mean+/-sd CH_4 and CO_2 flux values at different temperatures can be compared to other findings, the variability in the LW and HW samples' flux results cannot be interpreted to indicate any signal of positive (increased GHG emission, positive feedback to climate change) or negative change (decreased GHG emission, negative feedback to climate change) with the increase in air temperature and lowering of water table at northern bog in future.

Chapter 5

Conclusion

The study was motivated by the northern peatland systems and their role in atmospheric GHG budget as well as in climate change. Both temperature and water table are considered as dominant factors in boreal peatland CH_4 and CO_2 emissions. Higher water table induced anoxic condition is fundamental to CH_4 emission whereas higher temperature is key to the decomposition of organic matter and thus to the emission of CO_2. Again higher temperature can be attributed to increase the microbially-mediated CH_4 production rate whereas lower water table can cause higher fluxes of CO_2. The effect of temperature increase and water table draw-down on a natural peatland GHG emission can be highly variable but highly important in the context of climate change as higher emissions lead higher levels of climatic warming or increased levels of surface air temperature.

In the current study, due to lack of data points and inconsistencies among flux values, it has not been possible to obtain any statistically reliable positive or negative correlation between the increase in temperature and the levels of CH_4 and CO_2 fluxes within high and low water level peat cores. However, based on the study, a longer time of measurement and higher number of sample points can be recommended in case of laboratory experiments on soil GHG emissions. As there are differences in the emission of intact and semi-intact cores, further research can be suggested on the role of different sample types i.e. intact, semi-intact and totally disturbed on the estimated levels of soil GHG fluxes. Assuming that the fluctuating flux results derived in the study within the incubation period of 21 days are representing instability within the semi-intact cores, a longer time of incubation is recommended so that the stable point of emission for each of the cores can be observed. Moreover, regarding the temperature treatments, it is ideal to increase the temperature of the cores gradually if the purpose of the experimental temperature rise is to replicate more of a natural change.

The raised bog of Holmegård bog represents areas that are differentiated by varying level of disturbance i.e. natural, disturbed and restored sections at a time. Further studies on the net GHG-budget of the bog is highly recommended since all those sections can be highly variable in terms of their relative contributions as GHG emitters. However, before that, production of a detailed database on the bog's net C-stock and seasonal water table is highly recommended. As atmospheric N-deposition is quite high in this area, the nutrient status of the bog should also be assessed as higher nutrient status can increase the emission of another GHG, N_2O.

Restoration methods have been found to improve the bog's degraded conditions at some parts, however, whether that is also restoring the natural emission characteristics of the bog or not, is still not investigated. The restoration outcomes can also be assessed on the basis of relative changes in CO_2, CH_4 and N_2O fluxes which can be of significant importance in developing improved wetland management and restorations policies. Based on the current emission status of the bog at field level, its GHG emission potentials in increased temperatures and lowered water-table conditions can also be better understood.

Bibliography

Aaby, B., Riis, N., 2016. Mosegeologiske undersøgelser i holmegårds mose i forbindelse med eu-life naturgenopretningsprojekt. Tech. rep., Naturstyrelsen, Storstrm, Denmark.

Abdalla, M., Hastings, A., Truu, J., Espenberg, M., Mander, Ü., Smith, P., 2016. Emissions of methane from northern peatlands: a review of management impacts and implications for future management options. Ecology and evolution 6 (19), 7080–7102.

ACIA, A. C. I., 2004. Impacts of a warming arctic-arctic climate impact assessment. Impacts of a Warming Arctic-Arctic Climate Impact Assessment, by Arctic Climate Impact Assessment, pp. 144. ISBN 0521617782. Cambridge, UK: Cambridge University Press, December 2004., 144.

Agus, F., Hairiah, K., Mulyani, A., 2010. Measuring carbon stock in peat soils: practical guidelines. World Agroforestry Centre.

Alm, J., Saarnio, S., Nykänen, H., Silvola, J., Martikainen, P., 1999. Winter co 2, ch 4 and n 2 o fluxes on some natural and drained boreal peatlands. Biogeochemistry 44 (2), 163–186.

Askaer, L., Elberling, B., Friborg, T., Jørgensen, C. J., Hansen, B. U., 2011. Plant-mediated ch_4 transport and c gas dynamics quantified in-situ in a phalaris arundinacea-dominant wetland. Plant and Soil 343 (1-2), 287–301.

Aurela, M., Riutta, T., Laurila, T., Tuovinen, J.-P., Vesala, T., Tuittila, E.-S., Rinne, J., Haapanala, S., Laine, J., 2007. Co2 exchange of a sedge fen in southern finland-the impact of a drought period. Tellus B: Chemical and Physical Meteorology 59 (5), 826–837.

Bader, C., Müller, M., Schulin, R., Leifeld, J., 2018. Peat decomposability in managed organic soils in relation to land use, organic matter composition and temperature. Biogeosciences 15 (3), 703.

Baird, A. J., Comas, X., Slater, L. D., Belyea, L. R., Reeve, A., 2009. Understanding carbon cycling in northern peatlands: recent developments and future prospects. Carbon cycling in northern peatlands, 1–3.

Baird, A. J., Eades, P. A., Surridge, B. W., 2008. The hydraulic structure of a raised bog and its implications for ecohydrological modelling of bog development. Ecohydrology 1 (4), 289–298.

Ball, B., 2013. Soil structure and greenhouse gas emissions: a synthesis of 20 years of experimentation. European Journal of Soil Science 64 (3), 357–373.

Barreto, C., Lindo, Z., 2018. Drivers of decomposition and the detrital invertebrate community differ across a hummock-hollow microtopology in boreal peatlands. Écoscience 25 (1), 39–48.

Batjes, N., 2014. Total carbon and nitrogen in the soils of the world. European Journal of Soil Science 65 (1), 10–21.

Bekryaev, R. V., Polyakov, I. V., Alexeev, V. A., 2010. Role of polar amplification in long-term surface air temperature variations and modern arctic warming. Journal of Climate 23 (14), 3888–3906.

Bellisario, L., Bubier, J., Moore, T., Chanton, J., 1999. Controls on ch4 emissions from a northern peatland. Global Biogeochemical Cycles 13 (1), 81–91.

Belyea, L. R., Clymo, R., 2001. Feedback control of the rate of peat formation. Proceedings of the Royal Society of London B: Biological Sciences 268 (1473), 1315–1321.

Bubier, J. L., 1995. The relationship of vegetation to methane emission and hydrochemical gradients in northern peatlands. Journal of Ecology, 403–420.

Carlson, K. M., Gerber, J. S., Mueller, N. D., Herrero, M., MacDonald, G. K., Brauman, K. A., Havlik, P., OConnell, C. S., Johnson, J. A., Saatchi, S., et al., 2017. Greenhouse gas emissions intensity of global croplands. Nature Climate Change 7 (1), 63.

Chamberlain, S. D., Anthony, T. L., Silver, W. L., Eichelmann, E., Hemes, K. S., Oikawa, P. Y., Sturtevant, C., Szutu, D. J., Verfaillie, J. G., Baldocchi, D. D., 2018. Soil properties and sediment accretion modulate methane fluxes from restored wetlands. Global change biology.

Chambers, F. M., Charman, D. J., 2004. Holocene environmental change: contributions from the peatland archive. The Holocene 14 (1), 1–6.

Chapman, S., Buttler, A., Francez, A.-J., Laggoun-Défarge, F., Vasander, H., Schloter, M., Combe, J., Grosvernier, P., Harms, H., Epron, D., et al., 2003. Exploitation of northern peatlands and biodiversity maintenance: a conflict between economy and ecology. Frontiers in Ecology and the Environment 1 (10), 525–532.

Charman, D. J., Beilman, D. W., Jackson, S., Korhola, A., Mauquoy, D., Mitchell, F., Prentice, I., van der Linden, M., De Vleeschouwer, F., Yu, Z., et al., 2013. Climate-related changes in peatland carbon accumulation during the last millennium. Biogeosciences.

Clark, J. M., Ashley, D., Wagner, M., Chapman, P., Lane, S., Evans, C., Heathwaite, A. L., 2009. Increased temperature sensitivity of net doc production from ombrotrophic peat due to water table draw-down. Global Change Biology 15 (4), 794–807.

Clymo, R., 1984. The limits to peat bog growth. Phil. Trans. R. Soc. Lond. B 303 (1117), 605–654.

Clymo, R., 1992. Models of peat growth. Suo 43 (4-5), 127–136.

Collins, M., Knutti, R., Arblaster, J., Dufresne, J.-L., Fichefet, T., Friedlingstein, P., Gao, X., Gutowski, W., Johns, T., Krinner, G., et al., 2013. Long-term climate change: projections, commitments and irreversibility.

Comas, X., Kettridge, N., Binley, A., Slater, L., Parsekian, A., Baird, A. J., Strack, M., Waddington, J. M., 2014. The effect of peat structure on the spatial distribution of biogenic gases within bogs. Hydrological processes 28 (22), 5483–5494.

Cubasch, U., Wuebbles, D., Chen, D., Facchini, M., Frame, D., Mahowald, N., Winther, J., 2013. Climate change 2013: the physical science basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change Cambridge, United Kingdom and New York, NY, USA, 119–158.

Dedysh, S., 2002. Methanotrophic bacteria of acidic sphagnum peat bogs. Microbiology 71 (6), 638–650.

Dedysh, S. N., Panikov, N. S., Liesack, W., Großkopf, R., Zhou, J., Tiedje, J. M., 1998. Isolation of acidophilic methane-oxidizing bacteria from northern peat wetlands. Science 282 (5387), 281–284.

Dinsmore, K. J., Skiba, U. M., Billett, M. F., Rees, R. M., 2009. Effect of water table on greenhouse gas emissions from peatland mesocosms. Plant and Soil 318 (1-2), 229.

Dunfield, P., Dumont, R., Moore, T. R., et al., 1993. Methane production and consumption in temperate and subarctic peat soils: response to temperature and ph. Soil Biology and Biochemistry 25 (3), 321–326.

Ehlers, E., Krafft, T., 2006. Managing global change: earth system science in the anthropocene. In: Earth System Science in the Anthropocene. Springer, pp. 5–12.

Elberling, B., Askaer, L., Jørgensen, C. J., Joensen, H. P., Kuhl, M., Glud, R. N., Lauritsen, F. R., 2011. Linking soil o2, co2, and ch4 concentrations in a wetland soil: implications for co2 and ch4 fluxes. Environmental science & technology 45 (8), 3393–3399.

Essl, F., Dullinger, S., Moser, D., Rabitsch, W., Kleinbauer, I., 2012. Vulnerability of mires under climate change: implications for nature conservation and climate change adaptation. Biodiversity and Conservation 21 (3), 655–669.

Fang, C., Moncrieff, J. B., 1999. A model for soil co2 production and transport 1:: Model development. Agricultural and Forest Meteorology 95 (4), 225–236.

Folland, C., Karl, T., Christy, J., Clarke, R., Gruza, G., Jouzel, J., Mann, M., Oerlermans, J., Salinger, M., Wang, S., 2001. Climate change 2001: the scientific basis. Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change (eds Houghton JT et al.), Cambridge University Press, Cambridge, UK.

Frolking, S., Roulet, N., Fuglestvedt, J., 2006. How northern peatlands influence the earth's radiative budget: Sustained methane emission versus sustained carbon sequestration. Journal of Geophysical Research: Biogeosciences 111 (G1).

Frolking, S., Roulet, N. T., 2007. Holocene radiative forcing impact of northern peatland carbon accumulation and methane emissions. Global Change Biology 13 (5), 1079–1088.

Gill, A. L., Giasson, M.-A., Yu, R., Finzi, A. C., 2017. Deep peat warming increases surface methane and carbon dioxide emissions in a black spruce dominated ombrotrophic bog. Global change biology.

Gilman, K., et al., 1994. Hydrology and wetland conservation. John Wiley & Sons.

Graßl, H., 2006. Climate change, new weather extremes and climate policy. In: Earth System Science in the Anthropocene. Springer, pp. 41–50.

Gritsch, C., Zimmermann, M., Zechmeister-Boltenstern, S., 2015. Interdependencies between temperature and moisture sensitivities of co 2 emissions in european land ecosystems. Biogeosciences 12 (20), 5981–5993.

Group, S. C. W., et al., 1998. The canadian system of soil classification. Agriculture and agri-food Canada publication 1646, 187.

Holden, J., Burt, T., 2003. Hydrological studies on blanket peat: the significance of the acrotelm-catotelm model. Journal of Ecology 91 (1), 86–102.

Höll, B. S., Fiedler, S., Jungkunst, H. F., Kalbitz, K., Freibauer, A., Drösler, M., Stahr, K., 2009. Characteristics of dissolved organic matter following 20 years of peatland restoration. Science of the Total Environment 408 (1), 78–83.

Hviid, T., 2014. After-life conservation plan. Tech. rep., Danish Forest & Nature Agency.

Ivanov, K. E., et al., 1981. Water movement in mirelands. Academic Press Inc.(London) Ltd.

Joosten, H., Clarke, D., 2002. Wise use of mires and peatlands. International Mire Conservation Group and International Peat Society 304.

Kløve, B., Berglund, K., Berglund, Ö., Weldon, S., Maljanen, M., 2017. Future options for cultivated nordic peat soils: Can land management and rewetting control greenhouse gas emissions? Environmental Science & Policy 69, 85–93.

Lai, D., 2009. Methane dynamics in northern peatlands: a review. Pedosphere 19 (4), 409–421.

Laine, J., Silvola, J., Tolonen, K., Alm, J., Nykänen, H., Vasander, H., Sallantaus, T., Savolainen, I., Sinisalo, J., Martikainen, P. J., 1996. Effect of water-level drawdown on global climatic warming: Northern peatlands. Ambio, 179–184.

Le Mer, J., Roger, P., 2001. Production, oxidation, emission and consumption of methane by soils: a review. European Journal of Soil Biology 37 (1), 25–50.

Lees, K., Quaife, T., Artz, R., Khomik, M., Clark, J., 2018. Potential for using remote sensing to estimate carbon fluxes across northern peatlands–a review. Science of The Total Environment 615, 857–874.

Leifeld, J., Menichetti, L., 2018. The underappreciated potential of peatlands in global climate change mitigation strategies. Nature communications 9 (1), 1071.

Leifeld, J., Steffens, M., Galego-Sala, A., 2012. Sensitivity of peatland carbon loss to organic matter quality. Geophysical Research Letters 39 (14).

Liimatainen, M., Voigt, C., Martikainen, P. J., Hytönen, J., Regina, K., Óskarsson, H., Maljanen, M., 2018. Factors controlling nitrous oxide emissions from managed northern peat soils with low carbon to nitrogen ratio. Soil Biology and Biochemistry 122, 186–195.

Lindsay, R., Campus, S., Lane, W., 2010. Peatbogs and carbon: a critical synthesis to inform policy development in oceanic peat bog conservation and restoration in the context of climate change. RSPB Scotland 315.

Malmer, N., Holm, E., 1984. Variation in the c/n-quotient of peat in relation to decomposition rate and age determination with 210 pb. Oikos, 171–182.

Malmer, N., Nihlgård, B., 1980. Supply and transport of mineral nutrients in a subarctic mire. Ecological Bulletins, 63–95.

Malmer, N., Wallén, B., 1999. The dynamics of peat accumulation on bogs: mass balance of hummocks and hollows and its variation throughout a millennium. Ecography 22 (6), 736–750.

Malmer, N., Wallén, B., 2004. Input rates, decay losses and accumulation rates of carbon in bogs during the last millennium: internal processes and environmental changes. The Holocene 14 (1), 111–117.

Maltby, E., 2009. Functional assessment of wetlands: towards evaluation of ecosystem services. Elsevier.

Mander, Ü., Järveoja, J., Maddison, M., Soosaar, K., Aavola, R., Ostonen, I., Salm, J.-O., 2012. Reed canary grass cultivation mitigates greenhouse gas emissions from abandoned peat extraction areas. GCB Bioenergy 4 (4), 462–474.

Martikainen, P. J., Nykänen, H., Crill, P., Silvola, J., 1993. Effect of a lowered water table on nitrous oxide fluxes from northern peatlands. Nature 366 (6450), 51.

Mastnỳ, J., Kaštovská, E., Bárta, J., Chroňáková, A., Borovec, J., Šantrčková, H., Urbanová, Z., Edwards, R., Picek, T., 2018. Quality of doc produced during litter decomposition of peatland plant dominants. Soil Biology and Biochemistry 121, 221–230.

Mauquoy, D., Yeloff, D., 2008. Raised peat bog development and possible responses to environmental changes during the mid-to late-holocene. can the palaeoecological record be used to predict the nature and response of raised peat bogs to future climate change? Biodiversity and Conservation 17 (9), 2139–2151.

McCarthy, J. J., Canziani, O. F., Leary, N. A., Dokken, D. J., White, K. S., 2001. Climate change 2001: impacts, adaptation, and vulnerability: contribution of Working Group II to the third assessment report of the Intergovernmental Panel on Climate Change. Vol. 2. Cambridge University Press.

McLaughlin, J. W., 2004. Carbon assessment in boreal wetlands of Ontario. Sault Ste. Marie: Ontario Forest Research Institute.

Metje, M., Frenzel, P., 2005. Effect of temperature on anaerobic ethanol oxidation and methanogenesis in acidic peat from a northern wetland. Applied and environmental microbiology 71 (12), 8191–8200.

Mogensen, A., Fougt, H., Hansen, C., 2000. Geologiske interesseområder. Tech. rep., Storstrms Amt, Teknik og Miljforvaltningen.

Moore, P. D., 2002. The future of cool temperate bogs. Environmental Conservation 29 (1), 3–20.

Moore, T. R., Bubier, J. L., Bledzki, L., 2007. Litter decomposition in temperate peatland ecosystems: the effect of substrate and site. Ecosystems 10 (6), 949–963.

Mustamo, P., Hyvärinen, M., Ronkanen, A.-K., Kløve, B., 2016. Physical properties of peat soils under different land use options. Soil Use and Management 32 (3), 400–410.

Nazarenko, L., Schmidt, G., Miller, R., Tausnev, N., Kelley, M., Ruedy, R., Russell, G., Aleinov, I., Bauer, M., Bauer, S., et al., 2015. Future climate change under rcp emission scenarios with giss modele2. Journal of Advances in Modeling Earth Systems 7 (1), 244–267.

Nungesser, M. K., 2003. Modelling microtopography in boreal peatlands: hummocks and hollows. Ecological Modelling 165 (2-3), 175–207.

Oertel, C., Matschullat, J., Zurba, K., Zimmermann, F., Erasmi, S., 2016. Greenhouse gas emissions from soilsa review. Chemie der Erde-Geochemistry 76 (3), 327–352.

O'Kelly, B. C., Sivakumar, V., 2014. Water content determinations for peat and other organic soils using the oven-drying method. Drying Technology 32 (6), 631–643.

Olefeldt, D., Euskirchen, E. S., Harden, J., Kane, E., McGuire, A. D., Waldrop, M. P., Turetsky, M. R., 2017. A decade of boreal rich fen greenhouse gas fluxes in response to natural and experimental water table variability. Global change biology 23 (6), 2428–2440.

Olesen, M., Christensen, T., Christensen, O., et al., 2014. Fremtidige klimaforandringer i danmark: Danmarks klimacenter rapport 12-04. danmarks meteorologiske institut.

Oreskes, N., 2018. The scientific consensus on climate change: How do we know were not wrong? In: Climate Modelling. Springer, pp. 31–64.

Petersen, S. O., Ambus, P., Elsgaard, L., Schjønning, P., Olesen, J. E., 2013. Long-term effects of cropping system on n2o emission potential. Soil Biology and Biochemistry 57, 706–712.

Ramaswamy, V., Boucher, O., Haigh, J., Hauglustine, D., Haywood, J., Myhre, G., Nakajima, T., Shi, G., Solomon, S., 2001. Radiative forcing of climate. Climate change 349.

Rezanezhad, F., Price, J. S., Quinton, W. L., Lennartz, B., Milojevic, T., Van Cappellen, P., 2016. Structure of peat soils and implications for water storage, flow and solute transport: A review update for geochemists. Chemical Geology 429, 75–84.

Robroek, B., Limpens, J., Breeuwer, A., Crushell, P., Schouten, M., 2007. Interspecific competition between sphagnum mosses at different water tables. Functional Ecology 21 (4), 805–812.

Rosswall, T., Heal, O., 1976. Structure and function of tundra ecosystems. Soil Science 121 (6), 374.

Roulet, N. T., 2000. Peatlands, carbon storage, greenhouse gases, and the kyoto protocol: Prospects and significance for canada. Wetlands 20 (4), 605–615.

Roulet, N. T., Lafleur, P. M., Richard, P. J., Moore, T. R., Humphreys, E. R., Bubier, J., 2007. Contemporary carbon balance and late holocene carbon accumulation in a northern peatland. Global Change Biology 13 (2), 397–411.

Rydin, H., Jeglum, J. K., 2013. The biology of peatlands, 2e. Oxford university press.

Schaufler, G., Kitzler, B., Schindlbacher, A., Skiba, U., Sutton, M., Zechmeister-Boltenstern, S., 2010. Greenhouse gas emissions from european soils under different land use: effects of soil moisture and temperature. European Journal of Soil Science 61 (5), 683–696.

Schlüter, M., 1988. The use of peat in danish glassworks, 1825–1945. Journal of Glass Studies, 94–101.

Serrano-Silva, N., Sarria-Guzmán, Y., Dendooven, L., Luna-Guido, M., 2014. Methanogenesis and methanotrophy in soil: a review. Pedosphere 24 (3), 291–307.

Shi, X., Thornton, P. E., Ricciuto, D. M., Hanson, P. J., Mao, J., Sebestyen, S. D., Griffiths, N. A., Bisht, G., 2015. Representing northern peatland microtopography and hydrology within the community land model. Biogeosciences Discussions (Online) 12 (4).

Silamiķele, I., Nikodemus, O., Kalniņa, L., Purmalis, O., Kļaviņš, M., 2010. Peat humification character in two ombrotrophic bogs depending on peat properties. In: Proceedings of the

Latvian Academy of Sciences. Section B. Natural, Exact, and Applied Sciences. Vol. 64. Versita, pp. 159–166.

Silvola, J., Alm, J., Ahlholm, U., Nykanen, H., Martikainen, P. J., 1996. Co_2 fluxes from peat in boreal mires under varying temperature and moisture conditions. Journal of ecology, 219–228.

Smith, K., Ball, T., Conen, F., Dobbie, K., Massheder, J., Rey, A., 2003. Exchange of greenhouse gases between soil and atmosphere: interactions of soil physical factors and biological processes. European Journal of Soil Science 54 (4), 779–791.

Smith, K., Ball, T., Conen, F., Dobbie, K., Massheder, J., Rey, A., 2018. Exchange of greenhouse gases between soil and atmosphere: interactions of soil physical factors and biological processes. European Journal of Soil Science 69 (1), 10–20.

Solomon, S., Qin, D., Manning, M., Averyt, K., Marquis, M., 2007. Climate change 2007-the physical science basis: Working group I contribution to the fourth assessment report of the IPCC. Vol. 4. Cambridge university press.

Stivrins, N., Ozola, I., Gałka, M., Kuske, E., Alliksaar, T., Andersen, T. J., Lamentowicz, M., Wulf, S., Reitalu, T., 2017. Drivers of peat accumulation rate in a raised bog: impact of drainage, climate, and local vegetation composition. Mires & Peat 19.

Strack, M., Waddington, J., Tuittila, E.-S., 2004. Effect of water table drawdown on northern peatland methane dynamics: Implications for climate change. Global Biogeochemical Cycles 18 (4).

Sundh, I., Mikkelä, C., Nilsson, M., Svensson, B. H., 1995. Potential aerobic methane oxidation in a sphagnum-dominated peatlandcontrolling factors and relation to methane emission. Soil Biology and Biochemistry 27 (6), 829–837.

Svendsen, J. I., Alexanderson, H., Astakhov, V. I., Demidov, I., Dowdeswell, J. A., Funder, S., Gataullin, V., Henriksen, M., Hjort, C., Houmark-Nielsen, M., et al., 2004. Late quaternary ice sheet history of northern eurasia. Quaternary Science Reviews 23 (11-13), 1229–1271.

Tokarz, E., Urban, D., 2015. Soil redox potential and its impact on microorganisms and plants of wetlands. Journal of Ecological Engineering 16 (3), 20–30.

Treat, C. C., Bubier, J. L., Varner, R. K., Crill, P. M., 2007. Timescale dependence of environmental and plant-mediated controls on ch4 flux in a temperate fen. Journal of Geophysical Research: Biogeosciences 112 (G1).

Van der Molen, P., Schalkoort, M., Smit, R., 1994. Vegetation and ecology of hummock-hollow complexes on an irish raised bog. In: Biology and Environment: Proceedings of the Royal Irish Academy. JSTOR, pp. 145–175.

Van der Weerden, T., de Klein, C., Kelliher, F., 2010. Influence of pore size distribution and soil water content on n2o response curves. In: 19th World Congress of Soil Science, Soil Solutions for a Changing World. pp. 1–6.

Van Veen, J., Kuikman, P., 1990. Soil structural aspects of decomposition of organic matter by micro-organisms. Biogeochemistry 11 (3), 213–233.

Van Vuuren, D. P., Edmonds, J., Kainuma, M., Riahi, K., Thomson, A., Hibbard, K., Hurtt, G. C., Kram, T., Krey, V., Lamarque, J.-F., et al., 2011. The representative concentration pathways: an overview. Climatic change 109 (1-2), 5.

van Winden, J. F., Reichart, G.-J., McNamara, N. P., Benthien, A., Damsté, J. S. S., 2012. Temperature-induced increase in methane release from peat bogs: a mesocosm experiment. PLoS one 7 (6), e39614.

Wang, H., Yu, L., Zhang, Z., Liu, W., Chen, L., Cao, G., Yue, H., Zhou, J., Yang, Y., Tang, Y., et al., 2017a. Molecular mechanisms of water table lowering and nitrogen deposition in affecting greenhouse gas emissions from a tibetan alpine wetland. Global change biology 23 (2), 815–829.

Wang, X., Siciliano, S., Helgason, B., Bedard-Haughn, A., 2017b. Responses of a mountain peatland to increasing temperature: A microcosm study of greenhouse gas emissions and microbial community dynamics. Soil Biology and Biochemistry 110, 22–33.

Webster, K., McLaughlin, J., Kim, Y., Packalen, M., Li, C., 2013. Modelling carbon dynamics and response to environmental change along a boreal fen nutrient gradient. Ecological modelling 248, 148–164.

Weedon, J. T., Aerts, R., Kowalchuk, G. A., van Logtestijn, R., Andringa, D., van Bodegom, P. M., 2013. Temperature sensitivity of peatland c and n cycling: Does substrate supply play a role? Soil Biology and Biochemistry 61, 109–120.

Wu, J., Roulet, N. T., 2014. Climate change reduces the capacity of northern peatlands to absorb the atmospheric carbon dioxide: The different responses of bogs and fens. Global Biogeochemical Cycles 28 (10), 1005–1024.

Wuebbles, D. J., Hayhoe, K., 2002. Atmospheric methane and global change. Earth-Science Reviews 57 (3-4), 177–210.

Yavitt, J. B., Williams, C. J., Wieder, R. K., 1997. Production of methane and carbon dioxide in peatland ecosystems across north america: effects of temperature, aeration, and organic chemistry of peat. Geomicrobiology Journal 14 (4), 299–316.

Yu, Z., Loisel, J., Brosseau, D. P., Beilman, D. W., Hunt, S. J., 2010. Global peatland dynamics since the last glacial maximum. Geophysical Research Letters 37 (13).

Yu, Z., Turetsky, M., Campbell, I., Vitt, D., 2001. Modelling long-term peatland dynamics. ii. processes and rates as inferred from litter and peat-core data. Ecological Modelling 145 (2-3), 159–173.

Appendix A

Appendix

A.1 Flux Calculation Method

In order to calculate CH_4 and CO_2 flux, the following equation has been used,

$$F_X = \frac{b \times V_{CH} \times MW}{A_{CH} \times MV_{corr}} \tag{A.1}$$

Where, X refers to CH_4 or CO_2 gas
$F_X = Flux\ of\ X, \mu g\ X\ m^{-2}h^{-1}$
$A_{CH} = Basal\ area\ of\ the\ measuring\ chamber\ or\ headspace\ (m^2)$
$b = Increase\ in\ concentration\ of\ X\ (ppmh^{-1})$
$MV_{corr} = Temperature\ corrected\ molecular\ volume\ (m^3 mole^{-1})$
$V_{CH} = Volume\ of\ the\ measuring\ chamber\ (m^3)$
$MW_X = Molecular\ weight\ of\ X\ (gmol^{-1})$

The MV_{corr} is again calculated as follows,

$$MV_{corr} = 0.02241 \times (\frac{273.15 + T}{273.15})\ m^3 mol^{-1} \tag{A.2}$$

Where, T is the air temperature during the measurement, $0.02241\ m^3$ is the molar volume of an ideal gas at 1 atmospheric pressure and $273.15\ K$ temperature.

A.1.1 Sample Calculation

24 hour flux measurement of intact core-1 (IC-1) at $4°C$ temperature has been done as follows,

$Initial\ CH_4\ concentration\ in\ the\ headspace\ air\ sample\ = 0\ ppm$

$24\ hour\ CH_4\ concentration\ in\ the\ headspace\ air\ sample\ = 1000\ ppm$

$Increase\ in\ CH_4\ concentration,\ b\ = (1000 - 0)/24\ ppmh^{-1}$

$Headspace\ height,\ h\ = 6.5\ cm$

$Radius\ of\ the\ headspace\ surface,\ r\ = 2.5\ cm$

$Volume\ of\ the\ measuring\ headspace,\ V_{CH}\ = \pi r^2 h\ = 1.28 \times\ 10^{-4}\ m^3$

$Molecular\ weight\ of\ CH_4\ = 16.05\ gmol^{-1}$

$Basal\ area\ of\ headspace,\ A_{CH}\ = \pi r^2\ = 0.0019635\ m^2$

$MV_{corr}\ for\ air\ temperature\ of\ 14C\ or\ 287.15K\ = 0.04596\ m^3 mol^{-1}$

Therefore, the CH_4 flux for IC-1 is,

$F_{CH_4}\ = (41.67 \times 1.28 \times 10^{-4} \times 16.05)/(0.0019635 \times 0.04596)\ = 948.636 \mu g\ CH_4 m^{-2} h^{-1}$

Also in molar unit 59.11 $\mu mol\ CH_4 m^{-2} h^{-1}$

A.1.2 CO_2 and CH_4 Emission Data

Table A.1: Constants and variables associated with the calculation of CH_4 and CO_2 gas fluxes

Gas parameter	Molecular Weight of CH_4, MW, $gmol^{-1}$	16.05
	Molecular Weight of CH_4-C, MW, $gmol^{-1}$	12
	Molecular Weight of CO_2, MW, $gmol^{-1}$	44.01
	Molecular Weight of CO_2-C, MW, $gmol^{-1}$	12
Headspace	Headspace radius, r cm	2.5
	π	3.14
	Soil Surface Area within tubes, A_H, m^2	0.0019635
Temperature correction	T18 room temperature in Kelvin	291.15
	Temperature Corrected Molecular Volume for T18, MV_{corr}, m^3mol^{-1}	0.046296771
	T16 room temperature in Kelvin	289.15
	Temperature Corrected Molecular Volume for T16, MV_{corr}, m^3mol^{-1}	0.046132685
	T14 room temperature in Kelvin	287.15
	Temperature Corrected Molecular Volume for T14, MV_{corr}, m^3mol^{-1}	0.0459686
GC measurement	CH_4 sample	500 μ
	CO_2 sample	100 μ
	Conc. of CH_4 standard	10 ppm
	Conc. of CO_2 standard	10 ppm

Table A.2: CH_4 flux calculation at $4°C$ temperature from intact soil cores (24-hour and 7-day measurements)

Time of measurement	**Sample**	**Concentration**	**Concentration increase, b**	**Headspace Volume**	**Estimated flux**	**Estimated Flux**
		ppm	$ppmh^{-1}$	V_H, m^3	$\mu mol\ CH_4\ m^{-2}h^{-1}$	$g\ CH_4 - C\ m^{-2}y^{-1}$
24-hour	IC-1	1000	41.67	1.28E-04	58.94	6.2
	IC-2	900	37.5	1.18E-04	48.97	5.15
	IC-3	900	37.5	9.82E-05	40.81	4.29
7-day	IC-1	700	29.17	9.43E-05	30.47	3.2
	IC-2	800	33.33	9.82E-05	36.27	3.81
	IC-3	1000	41.67	1.02E-04	47.15	4.96

Table A.3: CH_4 flux calculation at $17°C$ temperature from semi intact peat cores (7-day measurements)

Time of measurement	Sample	Concentration	Concentration increase, b	Headspace Volume	Estimated flux	Estimated Flux
		ppm	$ppmh^{-1}$	V_H, m^3	$\mu mol\ CH_4\ m^{-2}h^{-1}$	$g\ CH_4 - C\ m^{-2}y^{-1}$
7-Day ($17°C$)	HW-1	ND	ND	9.63E-05	ND	ND
	HW-2	ND	ND	8.458E-05	ND	ND
	HW-3	100	0.60	9.43E-05	0.62	0.056
	HW-4	200	1.19	9.82E-05	1.29	0.14
	LW-1	100	0.60	7.66E-05	0.50	0.05
	LW-2	100	0.60	7.86E-05	0.51	0.05
	LW-3	400	2.38	8.05E-05	2.11	0.22
	LW-4	300	1.79	9.23E-05	1.81	0.19

Table A.4: CH_4 flux calculation at $17°C$ temperature from semi intact peat cores (14-day measurements)

Time of measurement	Sample	Concentration	Concentration increase, b	Headspace Volume	Estimated flux	Estimated Flux
		ppm	$ppmh^{-1}$	V_H, m^3	$\mu mol\ CH_4\ m^{-2}h^{-1}$	$g\ CH_4 - C\ m^{-2}y^{-1}$
14-Day ($17°C$)	HW-1	ND	ND	9.63E-05	ND	ND
	HW-2	ND	ND	8.648E-05	ND	ND
	HW-3	400	2.38	9.43E-05	2.48	0.26
	HW-4	400	2.38	1.00E-04	2.63	0.28
	LW-1	500	2.98	7.86E-05	2.58	0.27
	LW-2	800	4.76	7.86E-05	4.13	0.43
	LW-3	800	4.76	8.25E-05	4.34	0.46
	LW-4	1000	5.95	9.23E-05	6.07	0.64

Table A.5: CH_4 flux calculation at $17°C$ temperature from semi intact peat cores (21-day measurements)

Time of measurement	Sample	Concentration	Concentration increase, b	Headspace Volume	Estimated flux	Estimated Flux
		ppm	$ppmh^{-1}$	V_H, m^3	$\mu mol\ CH_4\ m^{-2}h^{-1}$	$g\ CH_4 - C\ m^{-2}y^{-1}$
21-Day ($17°C$)	HW-1	ND	ND	9.82E-05	ND	ND
	HW-2	200	1.19	8.648E-05	1.14	0.12
	HW-3	200	1.19	9.63E-05	1.26	0.13
	HW-4	400	2.38	1.00E-04	2.63	0.28
	LW-1	400	2.38	7.86E-05	2.01	0.22
	LW-2	800	4.76	7.86E-05	4.13	0.43
	LW-3	900	5.36	8.45E-05	5.00	0.53
	LW-4	1200	7.14	9.82E-05	7.74	0.81

Table A.6: CH_4 flux calculation at $25°C$ temperature from semi intact peat cores (7-day measurements)

Time of measurement	Sample	Concentration	Concentration increase, b	Headspace Volume	Estimated flux	Estimated Flux
		ppm	$ppmh^{-1}$	V_H, m^3	$\mu mol\ CH_4\ m^{-2}h^{-1}$	$g\ CH_4 - C\ m^{-2}y^{-1}$
7-Day ($25°C$)	HW-1	ND	ND	9.63E-05	ND	ND
	HW-2	300	1.79	8.648E-05	1.70	0.18
	HW-3	1400	8.33	9.43E-05	8.64	0.91
	HW-4	300	1.79	1.02E-04	2.01	0.21
	LW-1	900	5.36	1.10E-04	6.48	0.68
	LW-2	1000	5.95	8.45E-05	5.53	0.58
	LW-3	2500	14.88	9.04E-05	14.79	1.55
	LW-4	6100	36.31	7.86E-05	31.38	3.30

Table A.7: CH_4 flux calculation at $25°C$ temperature from semi intact peat cores (14-day measurements)

Time of measurement	Sample	Concentration	Concentration increase, b	Headspace Volume	Estimated flux	Estimated Flux
		ppm	$ppmh^{-1}$	V_H, m^3	$\mu mol\ CH_4\ m^{-2}h^{-1}$	$g\ CH_4 - C\ m^{-2}y^{-1}$
14-Day ($25°C$)	HW-1	ND	ND	9.82E-05	ND	ND
	HW-2	100	0.60	9.04E-05	0.59	0.06
	HW-3	900	5.36	9.63E-05	5.69	0.60
	HW-4	100	0.60	1.06E-04	0.70	0.07
	LW-1	600	3.57	1.10E-04	4.34	0.46
	LW-2	100	0.60	8.45E-05	0.56	0.06
	LW-3	100	0.60	9.63E-05	0.63	0.07
	LW-4	800	4.76	8.05E-05	4.23	0.45

Table A.8: CH_4 flux calculation at $25°C$ temperature from semi intact peat cores (21-day measurements)

Time of measurement	Sample	Concentration	Concentration increase, b	Headspace Volume	Estimated flux	Estimated Flux
		ppm	$ppmh^{-1}$	V_H, m^3	$\mu mol\ CH_4\ m^{-2}h^{-1}$	$g\ CH_4 - C\ m^{-2}y^{-1}$
21-Day ($25°C$)	HW-1	300	1.79	9.23E-05	1.81	0.19
	HW-2	ND	ND	8.64E-05	ND	ND
	HW-3	1000	5.95	9.63E-05	6.30	0.66
	HW-4	400	2.38	1.00E-04	2.62	0.28
	LW-1	700	4.17	1.10E-04	5.04	0.53
	LW-2	800	4.76	8.45E-05	4.42	0.47
	LW-3	300	1.79	9.63E-05	1.89	0.20
	LW-4	1500	8.93	8.05E-05	7.91	0.83

Table A.9: CO_2 flux calculation at $4°C$ temperature from intact soil cores (24-hour and 7-day measurements)

Time of measurement	Sample	Concentration increase	Concentration increase, b	Headspace Volume	Estimated flux	Estimated Flux
		ppm	$ppmh^{-1}$	V_H, m^3	$\mu mol\ CH_4\ m^{-2}h^{-1}$	$g\ CH_4 - C\ m^{-2}y^{-1}$
24-Hour ($4°C$)	IC-1	35	1.46	1.28E-04	2.06	0.22
	IC-2	71	2.96	1.18E-04	3.85	0.40
	IC-3	65	2.71	9.82E-05	2.94	0.31
7-Day (4C)	IC-1	230	9.58	9.43E-05	9.98	1.05
	IC-2	278	11.58	9.82E-05	12.56	1.32
	IC-3	255	10.63	1.02E-04	11.98	1.26

Table A.10: CO_2 flux calculation at $4°C$ temperature from semi intact peat cores (24-hour measurements, blank core concentration 475 ppm)

Time of measurement	Sample	Concentration increase	Concentration increase, b	Headspace Volume	Estimated flux	Estimated Flux
		ppm	$ppmh^{-1}$	V_H, m^3	$\mu mol\ CH_4\ m^{-2}h^{-1}$	$g\ CH_4 - C\ m^{-2}y^{-1}$
24-Hour ($4°C$)	HW-1	142	5.92	9.82E-05	6.42	0.67
	HW-2	300	12.5	8.64E-05	11.93	1.25
	HW-3	323	13.46	9.63E-05	14.30	1.50
	HW-4	175	7.29	1.00E-04	8.06	0.85
	LW-1	505	21.04	7.86E-05	18.25	1.92
	LW-2	525	21.88	7.86E-05	18.97	1.99
	LW-3	470	19.58	8.45E-05	18.26	1.92
	LW-4	725	30.21	9.82E-05	32.75	3.44

Table A.11: CO_2 flux calculation at $4°C$ temperature from semi intact peat cores (7-day measurements, blank core concentration 475 ppm)

Time of measurement	Sample	Concentration increase	Concentration increase, b	Headspace Volume	Estimated flux	Estimated Flux
		ppm	$ppmh^{-1}$	V_H, m^3	$\mu mol\ CH_4\ m^{-2}h^{-1}$	$g\ CH_4 - C\ m^{-2}y^{-1}$
7-Day ($4°C$)	HW-1	342	14.25	9.82E-05	15.45	1.62
	HW-2	478	19.92	8.64E-05	19.00	2.00
	HW-3	420	17.50	9.63E-05	18.60	1.95
	HW-4	456	19.00	1.00E-04	21.01	2.10
	LW-1	860	35.83	7.86E-05	31.08	3.27
	LW-2	750	31.25	7.86E-05	27.11	2.85
	LW-3	800	33.33	8.45E-05	31.08	3.27
	LW-4	988	41.17	9.82E-05	44.64	4.69

Table A.12: CO_2 flux calculation at $17°C$ temperature from semi intact peat cores (21-day measurements, blank core concentration 475 ppm)

Time of measurement	Sample	Concentration increase	Concentration increase, b	Headspace Volume	Estimated flux	Estimated Flux
		ppm	$ppmh^{-1}$	V_H, m^3	$\mu mol\ CH_4\ m^{-2}h^{-1}$	$g\ CH_4 - C\ m^{-2}y^{-1}$
21-Day ($17°C$)	HW-1	1220.68	7.27	9.82E-05	7.88	0.83
	HW-2	3936.46	23.43	8.64E-05	22.36	2.35
	HW-3	6234.57	37.11	9.63E-05	39.43	4.15
	HW-4	6188.11	36.83	1.00E-04	40.74	4.28
	LW-1	6755.09	40.21	7.86E-05	34.88	3.67
	LW-2	6052.44	36.03	7.86E-05	41.22	3.28
	LW-3	9450.17	56.25	8.45E-05	52.45	5.51
	LW-4	7905.91	47.06	9.82E-05	51.02	5.36

Table A.13: CO_2 flux calculation at $25°C$ temperature from semi intact peat cores (21-day measurements, blank core concentration 50 ppm)

Time of measurement	Sample	Concentration increase	Concentration increase, b	Headspace Volume	Estimated flux	Estimated Flux
		ppm	$ppmh^{-1}$	V_H, m^3	$\mu mol\ CH_4\ m^{-2}h^{-1}$	$g\ CH_4 - C\ m^{-2}y^{-1}$
21-Day ($25°C$)	HW-1	1637.49	9.75	9.23E-05	9.93	1.04
	HW-2	14255.63	84.85	8.64E-05	80.96	8.51
	HW-3	18675.15	111.16	9.63E-05	118.12	12.42
	HW-4	18354.34	109.25	1.00E-04	120.83	12.70
	LW-1	19841.73	118.11	1.10E-04	143.42	15.08
	LW-2	17790.77	105.90	8.45E-05	98.75	10.38
	LW-3	30687.99	182.67	9.63E-05	194.10	20.40
	LW-4	29544.62	175.86	8.05E-05	156.36	16.44

A.2 Detailed Soil Property Analysis Result

Table A.14: Organic C(%) and water content(%) calculation in LOI method

Soil Type	Sample	Weight of Crucible	Weight of crucible with wet soil	Weight of wet soil sample	Weight of crucible with dry soil (over dried at $105°C$)	Weight of dry soil
Units		W_c,g	g	W_{ws}, g	g	W_{ds}, g
Top soil 0-10 cm	S1.0	34.38	45.39	11.01	35.6057	1.2257
	S2.0	30.35	44.06	13.71	31.6696	1.3196
	S3.0	32.46	45.04	12.58	33.7111	1.2511
	S4.0	31.66	44.08	12.42	32.9726	1.3126
	S5.0	29.15	42.57	13.42	30.5695	1.4195
Bottom soil 10-30 cm	S1.10	29.25	41.94	12.69	30.5024	1.2524
	S2.10	35.15	47.13	11.98	36.3922	1.2422
	S3.10	18.98	31.15	12.17	20.1642	1.1842
	S4.10	20.09	32.29	12.2	21.2126	1.1226
	S5.10	21.96	34.99	13.03	23.2288	1.2688

Table A.15: Cont'd organic C(%) and water content(%) calculation in LOI method

Soil Type	Sample	Water content by weight	Moisture content	Weight of crucible with combusted dry soil	Weight of ash content	Loss on ignition (LOI)	Organic C content
Units		W_w, g/g	%	g	W_{ash}, g	g	%
Top soil 0-10 cm	S1.0	7.98	651.06	34.4883	0.1083	1.1174	52.88
	S2.0	9.39	711.58	30.4448	0.0948	1.2248	53.84
	S3.0	9.06	724.16	32.5592	0.0992	1.1519	53.41
	S4.0	8.46	644.52	31.7109	0.0509	1.2617	55.76
	S5.0	8.45	595.24	29.2221	0.0721	1.3474	55.06
Bottom soil 10-30 cm	S1.10	9.13	729	29.3451	0.0951	1.1573	53.6
	S2.10	8.64	695.54	35.2247	0.0747	1.1675	54.52
	S3.10	9.28	783.65	19.0532	0.0732	1.111	54.42
	S4.10	9.87	879.21	20.1752	0.0852	1.0374	53.6
	S5.10	9.27	730.61	22.0583	0.0983	1.1705	53.51