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The effects of land use, temperature and water level fluctuations on the emission of nitrous oxide (N_2O), carbon dioxide (CO_2) and methane (CH_4) from organic soil cores in Iceland

by

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Abstract

Agricultural practices can affect soil microbial production and emission of the major greenhouse gases (GHG's) nitrous oxide (N_2O), carbon dioxide (CO_2) and methane (CH_4). The purpose of this study was to gain insight into the influence of temperature, water level and water level fluctuations on the GHG emission of soils representing the three major types of landuse in Iceland: undisturbed peatland, drained uncultivated peatland and hayfield on drained peatland. Twenty-four soil cores, from three different areas were set up in controlled laboratory conditions and subjected to varying temperatures and water table levels. Vegetation was removed on eighteen of the twenty-four soil cores. On six soil cores the vegetation was kept undisturbed to gain information on the GHG emission effect of vegetation. Gas samples were collected by a common methodology based on a static chamber technique for N_2O and CH_4 and a dynamic chamber method for CO_2. The results show that landuse type significantly affects soil GHG's production ($P=0.000$). Raised temperatures increased the emission of CO_2 ($P<0.001$) and CH_4 ($P<0.001$) significantly but not the emission of N_2O ($P=0.458$). Water level fluctuations (WL), which were only conducted for the drained peatland and hayfield soil cores, had a strong influence on soil N_2O emission. The N_2O emission increased significantly ($P=0.000$) with water level fluctuations for both drained peatland soil cores and hayfield soil cores. However, the magnitude of the hayfield on drained peatland emission was 3-fold compared to drained soil cores. Water fluctuations increased the hayfield soil CO_2 emission significantly ($P=0.002$), but no significant change was detected in the emission of the drained peatland soil cores ($P=0.086$) nor between the landuse groups (drained and hayfield) ($P>0.005$). No CH_4 emission change was recorded with soil water fluctuations, which may be due to an insufficiently long incubation time.

These studies show that draining of peatland increases emission of N_2O and CO_2 and decreases emission of CH_4. Temperature had significant effect on the emission of CH_4 and CO_2 but not on N_2O emission. Water level fluctuations significantly affected N_2O production in both drained and hayfield soil cores but CO_2 production was only significantly affected in the hayfield soil cores. Vegetation had significant effect on the production of N_2O and CO_2.

Here, the highest emission calculated in CO_2 equivalents was measured from the undrained peatland. The reason for this was the very high methane emission at the highest temperatures, 13°C and 18°C. However, if the results are evaluated for each temperature compared to average annual temperature in Iceland, the hayfield soil cores emitted most GHG's. From the WL study the highest GHG emission was measured from the hayfield soils and secondly from the drained peatland soils.

Keywords: Landuse, wetland draining, peat, nitrous oxide, methane, carbon dioxide, GHG emission, temperature, water level fluctuations

Samantekt

Landbúnaður og önnur landnotkun getur haft áhrif á myndun og losun gróððuhúsalofttegunda (N_2O, CO_2, CH_4) úr jarðvegi. Í þessari rannsókn er borin saman losun gróðurhúsalofttegunda úr lífrænum jarðvegskjörnum teknum úr þrennskonar landi, þ.e. ósnertu votlendi, framræstu votlendi og túni á framræstu votlendi. Markmið rannsóknarinnar var að skoða áhrif hitastigs, mismunandi grunnvatnstöðu og vatnstöðusveiflna á losun gróðurhúsalofttegunda úr lífrænum jarðvegi með mismunandi landnýtingu á Íslandi. Gassýnum var safnað með hefðbundnum aðferðum byggðum á “static chamber method” til að mæla N_2O og CH_4 og “dynamic chamber model” til að mæla CO_2.

Niðurstöður úr hitastigshluta rannsóknarinnar sýndu að tölfræðileg marktækur munur er á losun gróðurhúsalofttegunda úr jarðvegi milli landnýtingarflokka (P=0,000). Hækkandi hitastig hafði marktæk áhrif á losun CO_2 (P< 0,001) og CH_4 (P<0,001) en ekki á losun N_2O (P=0,458).

Áhrif vatnssveiflna á losun lofttegunda voru metin fyrir framræst votlendi og tún á framræstu votlendi (kjörnum úr óröskuðu votlendi sleppt). Engin metan losun var mælanleg og var það sennilega vegna of stutts aðlögunartíma. Marktækt aukin losun á CO_2 mældist úr túnkjörnunum (P=0,002) en ekki fyrir framræstu kjarnana (P=0,086). Losun N_2O jókst marktækt (P=0,000) með vatnssveiflum, bæði úr túnkjörnunum og úr framræstu kjörnunum. Losun úr túnkjörnum reyndist þrefalt hærri en úr framræstu kjörnunum.

Þessi rannsókn sýnir að framræsla votlendis dregur úr CH_4 losun en eykur losun N_2O og CO_2. Rannsóknin sýnir einnig að við aukinn hita eykst losun CH_4 og CO_2 marktækt en losun N_2O ekki. Ennfremur undirstrika niðurstöður ransóknarinnar áhrif vatnstöðusveiflna í myndun og losum N_2O úr framræstu votlenndi. Vatnssveiflur leiddu einnig til aukinnar losunar á CO_2 úr túnkjörnunum. Þá sýndi rannsóknin og fram á það að gróður hefur marktæk áhrif á losun N_2O og CO_2.

Með fyrirvara um forsendur verkefnisins (rannsókn á kjörnum í tilraunastofu en ekki við náttúrlegar aðstæður) var samanlögð losun reiknuð í CO_2 jafngildi. Samkvæmt því reyndist losunin gróðurhúsalofttegunda í hitastigshluta rannsóknarinnar vera mest frá kjörnum úr ósnortnu votlendi. Ástæða þessa er mjög há metanlosun við hitastig 13°C og 18°C. Ef þessar niðurstöður eru metnar með tilliti til ársmeðalhita á Íslandi reynist losun mest úr túnunum. Túnin losuðu einnig mest í CO_2 jafngildi í þeim hluta rannsóknarinnar sem sneri að áhrifum vatnsstöðusveiflna. Því má álykta að við þær aðstæður sem ríkja hér á landi (tíðar rigningar og lágur ársmeðalhiti) séu tún á framræstu votlendi að skila frá sér mestu magni gróðurhúsalofttegunda.

Lykilorð:
Landnýting, votlendi, framræsla, hlátursgas, metan, koltvístýringur, losun GHG, hitastig, vatnstöðusveiflur

1. Introduction

Climate warming, resulting from human induced increase in the concentration of heat-trapping gases in the atmosphere, is a serious threat to the livelihood of the denizens of this planet. The increased concentration of these gases, termed greenhouse gases (GHG), is the result of numerous inherently different processes associated with our daily lives. According to the Intergovernmental Panel on Climate Change (IPCC) the main contributors of anthropogenic greenhouse gases are the following six sectors: Energy use, Industrial Processes; Solvent Use; Waste generation; Agriculture; and Landuse, Landuse change and forestry (LULUCF) (IPCC, 2006).

The agriculture and landuse sectors play an important role in the emission of the three main greenhouse gases, i.e. CO_2, CH_4 and N_2O. On a global scale, agriculture is a major contributor to, and the principal anthropogenic source of N_2O emission, where agricultural soils are the major GHG emission factor (Smith & Conen, 2004; Prather *et al.*, 2001). Agricultural land consists of cropland, managed grassland, and permanent crops which include agro-forestry and bio-energy crops (Smith *et al.*, 2007). Anthropogenic activity in the form of landuse management can affect naturally occurring soil bacteria in ways that will increase/decrease the rate of decomposition of organic matter and hence affect the production of the greenhouse gases (Freney, 2002; Brady & Weil, 1999).

At northern latitudes peat soils are commonly used for agricultural purposes. Peat is the collective name for organic deposits formed in wet areas where rate of decomposition of plant material is slower than plant growth rate (Smeck & Burras, 2002). These organic deposits accumulate over time and can be several meters thick (Smeck & Burras, 2002; Brady & Weil, 1999). If peat soils contain more than 20% organic matter by weight they are considered organic soils and are classified as Histosol (Brady & Weil, 1999; FAO, 1998). Peatlands cover approximately 3% of the Earth's land surface (Strack, 2008). The greatest extent of peatlands is found in the northern hemisphere or 350 million ha out of 400 million ha in total (Strack, 2008). Northern peatlands store approximately 10-30% of the global soil carbon stock and hence play an important role in the global carbon cycle (Strack, 2008). Peat acts as a sink for carbon dioxide and a source of atmospheric methane (Strack *et al*., 2008). The major causes of peatland disturbance are agriculture, forestry and peat extraction for fuel. Draining of peatland alters the hydrology and results in oxidation of peat and loss of stored carbon when the balance between primary production and decomposition is altered (Strack, 2008; Strack *et al*., 2008). The draining of peatlands for agricultural purposes increases the emission of N_2O and CO_2 but decrease the emission of CH_4 (Smith *et al*, 2003; Smith & Conen, 2004).

Within Iceland peatlands have been extensively drained for agricultural purposes (Thorhallsdottir *et al*., 1998; Oskarsson, 1998). Landuse on the drained areas is for most parts twofold, i.e. hay fields for the production of winter feed for livestock and rangeland for grazing animals. The drainage has been shown to have considerable effects on the greenhouse gas budget of peatlands, effectively changing Icelandic peatlands from an overall sink for atmospheric greenhouse gases to a source (Oskarsson, 1998). It is not, however, well understood how different types of landuse, i.e. hay field and rangeland, affect the fluxes of greenhouse gases of these drained peatlands. Additionally, it is not clear to what extent varying soil hydrology and soil temperature influence gas fluxes from drained Icelandic peat soils. Due to Iceland's commitment to the Kyoto Protocol, where decrease and mitigation of greenhouse gas emission is the main target, it is important to recognize and quantify the factors affecting these sources.

Here, I report the results of a study on the greenhouse gas fluxes of drained and un-drained Icelandic peat soils. The objectives of this study are twofold, firstly, to obtain data on the release of greenhouse gases from organic soil cores collected from sites representing the three common landuses on Icelandic peatlands (i.e. undisturbed peatlands, drained peatlands used for grazing, and drained peatlands turned to hay fields) and secondly to examine the effects of varying temperature and water table on the gas fluxes of the soil cores.

The null hypotheses (H_0) are as follows:

1. The GHG emission from soils of the three different land use managements is equal
2. Soil GHG emission is equal at different temperatures
3. Soil GHG emission is equal at different water table levels / fluctuations

The main goal of the study is to quantify the effects of temperature, water levels and water level fluctuations on soil GHG emission under different landuse. The emphasis of the study is on soil N_2O emission, as data on this greenhouse gas are sorely lacking. A rough estimate of the overall GHG emission from all the different land use groups calculated in CO_2 equivalents is also presented.

2. Background

Climate change

The term *Climate change* refers to a change in the state of the climate (Verbruggen, 2007). It can be statistically identified by changes in the mean and/or the variability's of its properties. The changed climatic state persists for an extended period, typically decades or longer. Climatic change can arise from natural internal processes, external forcing or from anthropogenic activities that affects the atmospheric gas composition (Verbruggen, 2007). Global warming is a gradual increase of surface temperature, which can be observed or projected. An increase in the concentration of the major greenhouse gases (GHG's) in the atmosphere is considered one of the main factors causing the warming (IPCC, 2006). The GHG's are the natural and anthropogenic induced gaseous constituents of the atmosphere, which absorb and emit radiation at specific wavelengths within the spectrum of infrared radiation emitted by the Earth's surface, the atmosphere and the clouds. In this way the GHG's trap heat within the surface-troposphere system and this is what is known as the greenhouse effect (Verbruggen, 2007).

CO_2 is the principal anthropogenic greenhouse gas that affects Earth's radiative balance. It is used as the reference gas against other greenhouse gases and is defined as having a Global Warming Potential (GWP) of 1. The Global Warming Potential (GWP) is an index for the radiative properties of well mixed greenhouse gases. The index measures the radiative forcing of a unit mass of a given well mixed greenhouse gas in today's atmosphere integrated over a chosen time horizon, relative to that of CO_2 (Smith & Conen, 2004; Verbruggen, 2007).

Nitrous oxide (N_2O) is one of the most important of the GHG's contributing to global warming additionally to carbon dioxide (CO_2) and methane (CH_4) (excluding water vapor and Ozone) accounting for approximately 6% of total GWP (Minami, 2002). Per unit of weight, N_2O has more than 300 times the effect of a CO_2 molecule for producing global warming in an approximately 100 year time horizon, and CH_4 has 23 times the effect of CO_2 in the same time horizon (Minami, 2002; Houghton *et al.,* 2001). The N_2O molecule is long-lived, approximately 120-150 years, and the acceleration rate for the increased atmospheric concentration has been 0.8 ppbv per year for several decades (Minami, 2002). The increased GHG concentration in the atmosphere during the industrial era is significant. The atmospheric N_2O abundance has increased from 270 ppb pre-industrial level (in 1750) to 314 ppb in 1998 and to 319 ppb in 2005. Ice core data on the N_2O, CO_2, and CH_4 concentrations 2000 years back in time show little change in mixing ratios. The last 200 years exhibit a relative rapid increase of these gases in the atmosphere (Forster *et al.,* 2007). N_2O is also the major source of ozone depleting nitric oxide (NO) and nitrogen dioxide (NO_2) in the stratosphere. An estimate has predicted that a doubling of the N_2O concentration in the atmosphere could result in a 10% decrease of the ozone layer which would increase the ultraviolet radiation reaching the earth by 20% (Crutzen & Ehhalt, 1977).

International agreements such as the Kyoto Protocol (IPCC, 1998) to the United Nations Framework Convention on Climate Change (UNFCCC) have the reduction of human-induced greenhouse gases as their main target. The group of countries included in the Annex 1 to the UNFCCC had the aim of returning their GHG emission to their 1990 level before the year 2000 (Verbruggen, 2007). To be able to reduce and alter the emission trend of the human induced GHG's all participants of the agreement need to work collectively and collect and report accurate knowledge of the GHG emission in their region (IPCC, 1998).

Soil microbial activity

Nitrous oxide (N_2O), carbon dioxide (CO_2) and methane (CH_4) are natural gases produced by soil microbial organisms under different circumstances (Prather *et al.*, 2001). CO_2 is a product of aerobic respiration from a vast majority of soil organisms in the process of decomposition of organic material. CH_4 is a product of an integral part of the metabolism of a large number of Euryarchaeota bacteria. These organisms are called *methanogens* and they are physiologically strictly anaerobic (Madigan, Martinko & Parker, 2000). Microorganisms do not produce N_2O as a main product, rather as a by-product in *nitrification* (aerobe process) and *denitrification* (anaerobe process).

Denitrification is the respiratory reduction of nitrogen oxides (N-oxides) and a vital part of the global nitrogen cycle whereby molecular nitrogen is emitted back to the atmosphere. Denitrification is performed by a vast range of phylogenetically distant bacteria which are able to use N as the final electron receiver when oxygen tension is low (Zumft, 1997).

Organically bound nitrogen goes first through the process of mineralization (decay) which converts the organically bound N into inorganic mineral forms (organic N $\rightarrow$ NH_4^+ and NO_3^-). The nitrogen bound in amino groups (as proteins or in humic compounds) is attacked by microbes and simple amino compounds are formed. They are hydrolyzed and the nitrogen is released as ammonium ion (NH_4^+) which can be oxidized into the nitrate form (NO_3^-) via *nitrification* ($NH_4^+ \rightarrow NO_2^- \rightarrow NO_3^-$). Plant roots mainly take up nitrogen in the form of NH_4^+ or NO_3^- ions. *Denitrification* occurs when oxidized forms of nitrogen such as NO_3^- and NO_2^- are converted to nitrogen (N_2). In this process some amounts of nitrous oxide gas (N_2O) can be released as a by product by the following sequence: $NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2$ (Brady & Weil, 1999; Minami, 2002).

Little is known about the direct influence of external factors on the dynamics of denitrification in complex systems. Scientists have almost exclusively focused on the roles of nitrate availability, oxygen and pH as regulators of denitrification rates (Davidson & Swank, 1986). However, these factors act through the denitrifying communities influenced and structured by long-term environmental drivers. Moisture conditions, temperature, competition, substrate availability and disturbance have long-lasting effect of the structural development of denitrifying community (Wallenstein *et al.,* 2006).

Nitrogen in soil

Soil nitrogen occurs mostly as a part of organic molecules. Soil organic matter contains approximately 5% nitrogen and soil nitrogen content is therefore correlated with soil organic matter content. Despite the low content of soil N compared to the atmospheric content (79% gaseous nitrogen, N_2), the soil contains 10 to 20 times more N than standing vegetation in forested or cultivated areas (including roots) due to decomposition and loss of C through CO_2 release, which lowers the C/N ratio (Brady & Weil, 1999). Soils contain most of the N found in terrestrial systems and account for approximately 65-70% of the global N_2O emission (Smith & Conen; Conrad, 1996).

Soil nitrogen occur mostly (95-99%) as part of organic molecules. The nitrogenous organic compounds are often in association with certain silicate clays or resistant humic acids that decrease the possibility for rapid microbiological breakdown. Approximately 2 -3 % of the nitrogen in soil organic matter is released annually as inorganic nitrogen. Inorganic nitrogen accounts for approximately 1-2 % of the total N in the soil. It is quite soluble in water and can be lost from the soil by leaching or volatilization (Brady & Weil, 1999).

Icelandic peatlands

Icelandic peatlands are made up of Histosols and Histic Andosols with high content of organic matter (Arnalds, 2004). Mires in Iceland are roughly divided into sloping mires, flatland mires and seasonally flooded mires depending on their location in the landscape. The peat thickness ranges from 1-6 meters (Steindórsson, 1975; Einarsson, 1975). Compared to mires in other countries, the Icelandic mires are higher in nutrients due to some endemic factors. Most of the mires receive water from surrounding mountains or hills in addition to rainwater. This makes them not truly ombrotropic or rainfed. They receive a constant source of nutrient inflow from adjacent areas (Steindórsson, 1975; Einarsson, 1975). The erosion in the highland interior also adds minerals to the mires when large amounts of windborne material are deposited in the lowland. Volcanic activity is also an important factor adding mineral tephra periodically. These two environmental factors affect the amount of soil organic carbon which is diluted in soil if the deposition of eolian and tephra materials is rapid. However, Histosols are not located nearby eoilan sources or within the active volcanic zone (Arnalds, 2004). The color of the Histosol soil is usually dark due to the amount of organic matter and the parent black basaltic glass material but reddish colors are not uncommon due to the amount of ferrihydrite (Arnalds, 2004). The amount of organic –C available is high in cultivated organic soils which might increase the N_2O emission from agricultural soil (Sahrawat & Keeney, 1986). Agricultural soils originating from drained wetlands are high in organic carbon.

Land use management

Land use management must be seen in a broad perspective. The net effects of activities related to land use, land use change and forestry (LULUCF) depend on all activities involved (Sampson & Scholes, 2000). On a global scale, the N_2O and CH_4 emission from agricultural land use has increased nearly 17% from 1990 to 2005 (Smith *et al.*, 2007). Five regions composing the Non-Annex countries were by 2005 responsible for about three-quarters of total agricultural emission. A collective decrease of 12% was shown by the other five regions, mostly Annex I countries (Smith *et al.*, 2007). Compaction is one of many factors affecting N_2O emission in agriculture. It reduces porosity and increases denitrification (Smith & Conen, 2004). Abandoned organic cropland has been shown to release both CO_2 and N_2O 20-30 years after cultivation has ceased (Maljanen *et al.*, 2007b).

Landuse practices in Iceland

Large areas of Icelandic wetlands have been drained, mostly during 1942-1988, for agricultural purposes. An area of 8-10,000 km^2 has been estimated as the original extent of wetlands before approximately 32,000 km of open ditches were excavated which effectively drained large areas and substantially altered the hydrology (Magnusson & Fridriksson, 1989). Estimates indicate that the hydrology of between 60-80% of the lowland peatland has been altered through drainage (Oskarsson, 1998). Recent numbers from the NYTJALAND database, used in the National Inventory Report (2008) show that an area of 3,636.2 km^2 of wetland has been converted to grassland by drainage. Wetlands remaining undisturbed (referred to as peatland in this study) were estimated 6,675.9 km^2 (The Environment Agency of Iceland, 2008). In other parts of Europe, the minority of farmed areas are on former wetlands. However, in Iceland drained lowland mires are one of the main ecosystems converted to agricultural land. Some of the drained areas have only been used as rangeland and were not been cultivated further. Ditches that were dug were not always successful, resulting in ineffective draining of the wetland which also can have large soil seed banks from wetland species a decade after draining (Thorhallsdottir, 1998). Drained peat for agricultural purpose increases the GHG emission roughly by 1 t CO_2 equivalents/ha/yr (Kasimir-Klemedtsson *et al.,* 1997).

In the Icelandic National Inventory Report (see section on Land Use, Land-Use Change and Forestry (LULUCF), four types of land conversion are reported i.e. grassland converted to forested land, wetland converted to grassland, grassland converted to wetland and other land converted to wetland. The extent of drainage of peatlands is estimated based on the length of ditches. Land reported as being converted applies to forest plantations up to 20 years old, drained wetlands and flooded land (reservoirs - hydropower plants) (The Environment Agency of Iceland, 2008). According to the report, 13% of the planted forest since 1990 in Iceland were planted on wetland (3%) or drained wetland (10%). Preparation of the area for plantation often requires ploughing which might cause increased N_2O emission. Calculations on default values from IPCC, estimates that approximately 76% of N_2O emission in Iceland is due to agriculture and agricultural soils are responsible for the bulk of the emission. Between the years 1990 to 2006, a decrease in the animal livestock and the termination of fertilizer production in Iceland resulted in an overall N_2O emission decrease by 9% (The Environment Agency of Iceland, 2008).

Borgarfjörður region

The organic soil in the mires of the Borgarfjörður region has on average lower pH than in other areas in Iceland (Thorsteinsson, 1995; Johannesson, 1988). The two main factors impacting the soil pH are on one hand the additional volcanic ash and the eoilan deposits (from highland erosion etc.) which raise the pH and on the other hand the precipitation which lowers the pH. In Borgarfjörður region the mineral content is rather low compared to the rest of Iceland due to the distance from active volcanoes. The dominating south-west towards north-east wind pattern transports volcanic ash and eolian material into other areas. Precipitation is quite high in the area with an average of 890 mm (1961-1990) and the mean annual temperature is 3.2°C (1961-1990) (Meteorological Office of

Iceland, 2008). Most of the precipitation falls as rain and high precipitation rates also lower the average pH (Thorsteinsson, 1995; Johannesson, 1988). In the eastern part of Borgarfjörður area 91% of the mires have been drained or altered, mostly sloping mires. Approximately 7% of the sloping mires remain intact and 27% of the flatland mires (Oskarson, 1998).

Variables affecting greenhouse gas emission from organic soil

Many factors affect the N_2O emission from soil and the processes are not all well understood. Firestone and Davidson (1989) suggest that the production of NO and N_2O can be compared with pipes from where the gases are leaking. Inorganic nitrogen flows through the pipes which are the microbial processes of nitrification and denitrification. In this process some nitrogen compounds escape through "leaks" in the pipe as NO and N_2O (Figure 1).

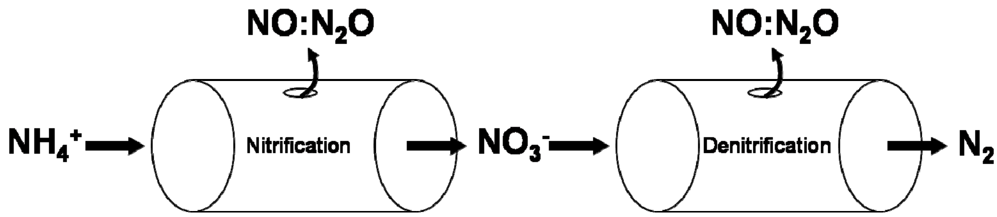

Figure 1. The "Hole in the Pipe" conceptual model of the microbial processes of nitrification and denitrification (Firestone & Davidson, 1989).

Many variables have shown to be important variables for N_2O emission from soil for example soil type, vegetation, availability of carbon, oxygen concentration, fertilization rate, tillage practice, land use management, temperature, pH and water holding capacity of the soil, C/N ratio (Freney, 1997). Because the N_2O is formed through microbial processes its production is dependent on all the factors controlling and affecting microbial growth (Sahrawat & Keeney, 1986). To be able to predict N_2O production (and consumption) it is necessary to identify soil factors that regulate the rate of microbial production and consumption. Additionally to microbial denitrification and nitrification,

N_2O can also be produced by chemodenitrification which refers to abiotic reactions resulting in N_2O, NO and N_2 formation. These reactions are highly pH-dependent (Davidson & Schimel, 1995; Minami, 2002). Davidson and Schimel (1995) divide the N_2O producing organisms into functional groups rather than gas production by species. They suggest that it might be of more informative to identify the factors controlling rates of production by knowing which functional group is responsible for the gaseous production.

A study done by Dobbie and Smith (2002) on gleysol (Inceptisols – U.S. *Soil Taxonomy*), show a large emission response to temperature, which was greater from grassland soils than from arable soil. By measuring the water filled pore space (WFPS) the arable soil increased the emission 30-fold when WFPS increased from 60% to 80% and grassland soil emission increased by 12-fold. They concluded that the N_2O emission was mostly due to denitrifying processes and that WFPS and temperature developed anaerobic zones within the soil (More on soil water content in Chapter 2). Koponen *et al.* (2006a) had the same conclusion studying N_2O emission due to low temperatures. In their experiment, the $NO:N_2O$ ratio was <1 below 0°C, indicating that the N_2O emission was most likely due to denitrification processes. Other studies indicate similar results. In a literature review by Sahrawat and Keeney (1986) they found that denitrification was the major process contributing to N_2O emission. Firestone and Davidson (1989) found that N_2O produced through nitrification is controlled primarily by O_2 and NH_4^+ availability and that the denitrification process is affected by O_2, NO_3^- and availability of organic - C. They call these *cellular controllers* which are affected by numerous physical, chemical and biological properties of the ecosystem (or ecosystems). Soil acidity has shown to be one of the many important factors controlling N_2O emission. Firestone, Firestone and Tiedje (1980) found that soil acidity interacted with nitrate and increased the ratio of N_2O to molecular nitrogen. By adding 10 parts per million (ppm) of NO_3^--N the N_2O production increased more at pH 4.9 than at pH 6.5. In anoxic conditions N_2O production initially increased but then the pattern changed to consumption of N_2O. The authors conclude that the consumption was due to sequential synthesis of nitrogenous oxide reductases. Concerning pH, Davidson and Schimel (1995) stressed the importance of different scale;

the bulk soil acidity does not need to approximate the pH optimum for microbial processes to occur. Rather, only the microsites need to be within the pH tolerance level for the organisms to survive and grow. The microsite pH might be quite different from the bulk soil pH which indicates that the pH of a soil sample might be a poor estimator for N_2O production (Davidson & Schimel, 1995).

Methodology

Emission measurements of N_2O and CH_4 are commonly performed by using a static chamber method. Healy *et al.* (1996) suggest that the method used to measure the emission might affect the results due to physical factors such as the distortion of the soil gas concentration gradient and the slow transport rate of diffusion relative to mixing in the chamber. By using analytical and numerical solutions to the soil-gas diffusion equation they found that the gas flux density to a static chamber based on the soil surface was less in magnitude than the ambient exchange rate in the absence of the chamber. They also suggest that nonlinear models were superior to the linear regression model for estimating flux densities. Mosier *et al.* (1996) measured N_2O emission by using several flux measurement techniques. They concluded that the different measurement techniques was not the major factor for the uncertainty in N_2O flux values found in the literature, rather the diverse combination of all the physical and biological factors which control gas fluxes.

Vegetation

The effect of vegetation on N_2O and N_2 emission from soil is not well understood but has been shown to have a major effect (Maljanen *et al.,* 2003; Regina *et al.,* 2004; Davidson & Swank, 1986). The N_2O emission in a field study from drained organic soil in Finland was found to be higher from soils without plants (kept bare by regular cutting or tilling) compared to emission from cultivated soil (Maljanen *et al.,* 2003). The tilling effect and plant residues might increase the availability of organic carbon for denitrifying organisms. Other studies show that fallow soil plots to emit less than barley plots, but grassland emitted the least (Regina *et al.,* 2004). Measurements performed by Regina *et al.* (2004) from the field and laboratory showed highest emission from sites with patches of fescue (*Festuca* sp.) or from fescue sites which where apparently residuals from grassland conversion.

When the presence of plants increases emission it has been suggested that the plant effect on the denitrification is two-fold; first by increasing the demand for oxygen and secondly by supplying easily decomposable organic matter. Nitrate availability might be higher in soils with plants and the denitrifying bacteria derive easily decomposable organic compounds to enhance their activity (Stefanson, 1972).

The effect of plant roots on soil denitrification has also been studied. The denitrifying activity has shown to be greater closest to the roots (in the rhizophere) in anaerobic conditions. In aerobic conditions the roots have no significant effect on denitrification. Increased NO_3^- concentration increases the ration $N_2O/(N_2+N_2O)$. Low NO_3^- concentration may lower the denitrification in the rhizophere (Smith & Tiedje, 1979). The emission of methane has also been shown to be vegetation related, with different emissions are found in different plant communities (Komulainen *et al,* 1998; Shannon & White, 1994). Well developed root system can act as gas conduits and enhance GHG transported from the soil to the atmosphere. Additionally, the effect of easily degradable carbon compounds which are released from the roots, are sources of substrates for production of CH_4 as well as the other GHG (Joabsson & Christensen, 2002).

Fertilizer

Soil scientist and agronomists have studied N_2O emission as a mechanism of losing fertilized- N from the soil. Increased N_2O emission due to agricultural use of N-fertilizers has been reported (Freney, 1997; Smith *et al*., 2004; Peterson *et al.,* 2006). The pattern of loss of N_2O –N from soils is affected by different types of fertilizers and combination of fertilizers as well as the crop management. However, Bouwman (1996) calculated N_2O emitted to be approximately 1.25% of the N applied (kg N/ha) despite the fertilizer type and the wide range of different environmental and managements conditions. The lost N from the fertilization is an extra cost for the farmer. Better management aimed at increasing the efficiency of the applied N-fertilizer used by the crop should be an incentive for higher returns for the farmer and less impact on the environment (Freney, 1997). Clayton *et al*. (1996) studied the N_2O –N loss effect of different fertilizers on the N_2O emission in clay loam grassland. They found that high emission fluxes in summer were associated with high water filled pore space (WFPS) values. The highest N_2O –N loss was from supplemented slurry and urea treatment and the lowest from the ammonium sulphate treatment. However, the lowest emission factor 0.8% was obtained after the application of urea in the spring and ammonium nitrate twice in the summer of the 2nd year. Sahrawat & Keeney (1986) and Petersen *et al.* (206) got markedly exceeded N_2O emission from plots receiving urea.

Temperature

Koponen *et al.* (2004) studied the effect of low temperatures on the N_2O emission. They found gaseous production down to -0,6°C and a peak at -0,1°C in organic, clay and silt soil and a peak at +1,6°C in the loam soils. They also found that the N_2O emission increased close to 0°C when temperature was lowered stepwise from +15°C. Clayton *et al.* (1997) found that emission rose steeply with temperature. A study performed on Canadian agricultural soils showed larger emission of N_2O attributed to a trend in higher daily temperature (Smith *et al.,* 2004). Correlation with peat temperatures and increased methane emission has also been found in field studies (Komulainen *et al.,* 1998; Shannon & White, 1994). A study performed during winter showed increased N_2O emission from plots without snow cover and a soil temperature up to 15°C lower than those with snow cover during soil freeze and thaw. The soil thermal conditions were lower due to the removed snow which isolated the soil from the low air temperatures. Minor changes were found in CH_4 and CO_2 emission after removal of snow during the cold period (Malajanen *et al.,* 2007a). Short-term variation effect in N_2O, CH_4 and CO_2 emission can affect the calculations of annual emission. This was studied in Finland were CO_2 emission was found to be 14-23% higher during daytime than the mean daily fluxes. The N_2O emission had high diurnal variation fluxes. During daytime the emission was up to 1.3 fold higher than the daily mean flux and as much as 5-fold more during daytime compared to nighttime emission (Maljanen *et al.,* 2002).

Thermal sensitivity of the GHG flux can be described by calculating a Q_{10} value. It describes how much the flux rate changes over a 10°C interval (Raich & Sclesinger, 1992). Q_{10} values of CO_2 flux of Icelandic peat soil has been reported to be 2.05 on average (Oskarsson, 1998) and a global review reports values of 1.3-3.3 and a median of 2.4 (Raich & Sclesinger, 1992). Oskarsson (1998) also reports Q_{10} value of methane flux to be 4.2 and Q_{10} value of nitrous oxide was reported to be 1.9 to 3.5.

Soil moisture

All microbes are dependent on water. The availability of water in soil is a controlling factor in gas production and consumption in soil. The soil water content is affected by the relative proportion between H_2O and O_2 and vise versa (Davidson & Schimel, 1995). Soil water content is important for gas diffusion into and out of the soil. Diffusion through water is approximately 10,000 slower than through air. O_2 is one of the key controlling factors in nitrification, denitrification, CH_4 oxidation and methanogenesis where nitrification requires O_2 but inhibits denitrification. The margin for optimal N_2O production is very complex (Firestone & Davidson, 1989; Davidson & Schimel, 1995). When increased soil moisture content results in anaerobic conditions, methanogenic bacteria produce more CH_4 than is consumed and the soil becomes a source of methane flux to the atmosphere. The opposite applies when the soil is aerated and methanotrophic bacteria mineralize CH_4 to carbon dioxide which results in a flux of CO_2 to the atmosphere from the soil and the soil becomes a sink for methane (Madigan, Martinko & Parker, 2000; McNamara *et al.*, 2006).

Lowering the water table by drainage drastically changes the hydrology of the peatland. This will affect the above- and belowground ecosystem. Both the drainage (aerobic conditions) and the fluctuations in the water table level (WL) affect the ecosystem and the GHG's emission from the soil microorganisms. Not only may drainage lower the water table level, climatic change with dryer summers can lower the water table significantly during summer in the boreal zone and change the production of the GHG's (Manabe & Wetherald, 1986). Field studies on N_2O emission often show flux peaks after rain events even though many studies fail to show strong correlations in the regression analysis for different reasons (Clayton *et al.*, 1996).

During freeze-thaw cycles (FTC), N_2O accumulated in the soil might be released to the atmosphere. The soil water content and O_2 in soil available for microorganisms is affected by FTC. Soil microorganisms are affected by the FTC both by temperature fluctuations and the availability of unfrozen water at microsites as well as the anaerobic conditions within the microsites enclosed in frozen water (Teepe *et al.*, 2004).

Priemé and Christensen (2001) investigated the drying-wetting and freezing-thawing effect from organic soil cores from Germany, Sweden and Finland. The N_2O emission increased up to 1,000-fold from the German and Swedish soil cores during the first week following wetting or thawing. The grassland sites emitted more than the arable sites. The CO_2 emission increased up to 5-fold. CH_4 emission rates were very small and often too small to detect. No significant release was found in the soil cores from the Finnish site which might be due to a prolonged drought before the soil cores were collected. They conclude that a single wetting or thawing event may account for a large proportion of the N_2O emission from farmed organic soil (Priemé & Christensen, 2001).

Teepe *et al.* (2004) suggest that increasing O_2 concentration in thawing soil has the effect of slowing down denitrification rather than accelerating nitrification. The water filled pore space (WFPS) parameter gives an indication of water content in the soil. Koponen *et al.* (2006a) found the highest N_2O emission from abandoned soil with a high WFPS or 70.6% ±13.6%. The contrary was found in the study performed by Teepe *et al.* (2004) on sandy, silty and loamy soils. The N_2O emission was less at 76% WFPS than at 55% WFPS which the authors suggest might be due to an increased ratio of N_2/N_2O in the very moist condition. Similar emission was found for CO_2 with highest emission at low WFPS (42%). Teepe *et al.* (2001) studied FTC and N_2O emission and concludes that the N_2O emission during thawing was due to the physical release of trapped (by ice barriers) N_2O and/or denitrification during thawing.

Koponen, Flöjt and Martikainen (2004) investigated the effect of FTC and soil moisture on actual N_2O production, not only the release of accumulated gas. When the soil cores thawed from -2°C to +4°C the emission begun to increase close to 0°C and the organic soil cores increased the N_2O emission up to + 3°C. They conclude that the soil cores had a clear N_2O emission maximum both during soil freezing and soil thawing and that N_2O production was intact down to -6°C. However, they could not conclude whether the emission originated solely from N_2O production or also from liberation of entrapped N_2O in frozen soil. Similar results were found by Priemé and Christense (2001) when N_2O emission started within a few hours after transition from frozen soil to +4°C. They also suggest that it is either a physical release of gases or the N_2O producing microorganisms that are active within the frozen soil. Koponen and Martikainen (2004) found that both the severity of the frost and soil moisture level affect the N_2O burst after thawing. They suggest that it might be the destruction of microbial cells during FTC that releases energy which is a key factor controlling N_2O production at low temperatures. Teepe *et al.* (2004) found no effect on the N_2O emission resulting from differences in soil texture (sandy, silty or loamy soils) but the duration of freezing and soil water content (64% WFPS) gave highest N_2O emission and increased the emission from all three soil types.

From a study on freeze –thaw cycles including boreal peat soil and loamy sand, Koponen *et al.* (2006b) found that N_2O emission in peat soil was extremely low and suggest it might be due to low soil water content. The CO_2 emission during soil thawing was higher in the peat soil than the loamy sand. FTC increased the anaerobic respiration especially in peat soil and did not affect the microbial community structure and biomass. They conclude that freezing and thawing did not have a strong effect on microbial biomass or community structure (Koponen *et al.,* 2006b). These results are to some extent contrary to other studies where peatlands drained for agricultural purpose have been considered to be a source of N_2O emission (Smith & Conen, 2004). However, too high soil water content leads to conditions where anaerobic conditions prevail and a decrease in the N_2O production. The reduction is likely due to the water slowing down the N_2O gas diffusion rate and N_2O being reduced to N_2 (Clayton *et al.,* 1997).

Methane production occurs in anaerobic conditions with a redox potential of below about -180 mV. Unlike denitrification which can be initiated rapidly, methane production needs longer anaerobic conditions to establish. The soil has to be flooded for several days for such conditions to establish (Smith & Conen, 2004; Smith *et al.,* 2003).

Methane can be released from the soil to the atmosphere by diffusion, bubble ebullition or be transported through vascular plants (Joabsson & Christensen, 2002). In the soil surface layer, CH_4 is subjected to methanotrophic oxidation and reduced to CO_2. However, bubbles are quite stabile and begin to form when the production rates exceed the capacity of diffusion loss (Joabsson & Christensen, 2002). When diffusion is the main pathway of methane emission, more or less of the produced CH_4 is consumed in the soil profile and is never emitted to the atmosphere. The emission pathway is therefore affecting the rate of CH_4 emission (Joabsson & Christensen, 2002).

Komulainen *et al.* (1998) reported that methane emission started to rise quite soon after a rewetting treatment of a previously drained fen and a bog site. However, the emission level was low compared to pristine mires. After one year they found a significant increase in the mean methane emission and the highest emission was observed in the group with the highest water level (Komulainen *et al.,* 1998). Shannon and White also found that a longer period (> one year) was needed to get high CH_4 production (Shannon & White, 1994).

Most literature on N_2O production concludes that there are many unknown factors controlling gaseous emission and that much remains to be understood before a cause-and-effect relationship can be established between anthropogenic activity and the disorder of the earth's chemical content and reactions in the atmosphere and stratosphere (Firestone, Firestone & Tiedje, 1980).

To be able to predict the magnitude and effects of the anthropogenic impacts through different agricultural land use management, it is necessary to collect data on microbial respiration and the resulting soil greenhouse gas emission. Data on the emission factors for the relevant gases and land use management systems will improve the quality of information from Icelandic agricultural land use management to the UNFCCC on greenhouse gas release.

3. Methods and materials

A soil core approach was chosen for studying the effect of different landuse on the greenhouse gas fluxes of drained peat soils. This approach has two major advantages over a field study: 1) standardized environmental conditions where gas fluxes of soils under different landuse can be compared, and 2) the possibility of manipulating key environmental factors influencing soil gas emission, i.e. soil gas flux can be studied under varying temperature and soil water level conditions.

Study region

Twenty four soil cores were collected (2006) from nine study sites located in Borgarfjörður region, in West Iceland. Borgarfjörður area covers 4,926 km^2 and has 3,700 inhabitants (Borgarbyggd, 2008). The summer (May-September) mean temperature is 8.1°C ranging from 5.5°C in May and up to 10.3 in July (1961-1990). The mean winter temperature is -0.3°C in the period October-April, ranging from -2.5 in January up to 3.0 in October. The mean annual precipitation for the area is 890 mm with most of the precipitation falling as rain in the winter period (Meteorological Office of Iceland, 2008).

Study sites

The nine study sites where the soil cores were collected from represent the three main management types currently practiced on peatlands of the Borgarfjörður region: 1) undisturbed peatland, 2) drained peatland and 3) fertilized hay-field on drained peatland. The nine sites were evenly distributed between the management categories. The sites were located on the adjacent farms Mávahlíð, Hestur, Kvígstaðir, Heggstaðir and Vatnshamar, all located within an approximately 7 km^2 radius (Figure 2, Table 1). Mountains and rock outcrops surround all these mires. The drained sites had been drained around 1965-75 and then converted to hayfields approximately 10-12 years later.

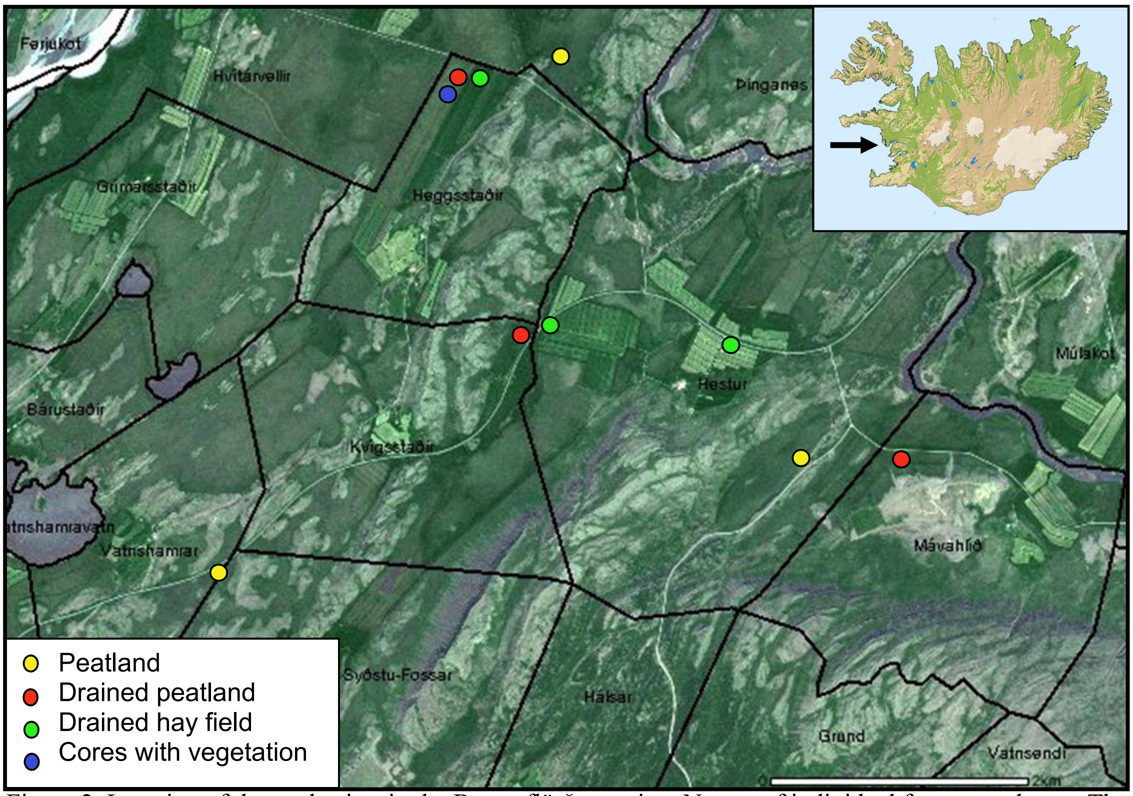

Figure 2. Location of the study sites in the Borgarfjörður region. Names of individual farms are shown. The location of Borgarfjörður region is shown on the inserted map of Iceland (Icelandic Farmland database, 2006).

The undisturbed peatland sites were dominated by *Carex* spp. and *Eriophorum angustifolium*. The drained sites still had a noticeable cover of *Carex* spp. and *Eriophorum angustifolium*, but various graminoids and bryophytes were also common. The dominating plant species of the hay field sites were *Deschampsia caespitosa* (snarrótarpuntur), *Poa pratensis* (vallarsveifgras), *Phleum pratense* (vallarfoxgras) and *Alopecurus pratensis* **(**háliðagras) (Kristinsson, 1989).

Table 1. Distribution of study sites between the farms.

	Landuse category		
Farm	**Undisturbed**	**Drained**	**Hay field**
Mávahíð		X	
Hestur	Hestmýri		X
Kvígstaðir		X	X
Heggstaðir	Skjólholtsmýri	X	X
Vatnshamar	Stráksmýri		

Soil cores

Two soil cores were collected from each site making up a total of 18 soil cores, six for each of the three respective landuse groups. The cores were 10 cm wide and 50 cm long and were taken from the depth of -5 cm down to -55 cm. Additional six soil cores, with intact vegetation, were collected from the Heggstaðir drained landuse site (named "G-drained vegetated"). These six additional cores were collected for the purpose of understanding the influence vegetation has on in-soil production of greenhouse gases. Soil cores collected in the category undisturbed peatland were collected from the sloping mires Stráksmýri, Skjólholtmýri and Hestmýri (Table 1; Figure 2).

Description of soil properties

Soil samples from the soil cores were analyzed for the following properties: pH (H_2O), percent C content and C/N in the Keldnaholt soil laboratory of the Agricultural University. The undisturbed peatland sites were not included in these evaluations.

Laboratory setup

The core gas flux measurements were performed in controlled laboratory conditions. All 24 cores were setup in a temperature controlled room at the Keldnaholt research facility of the Agricultural University of Iceland (Plates 1, 2 and 3). The soil cores were placed in 50 cm long and 10 cm diameter PVC pipes (after Regina *et al.,* 1999). Additionally, the top 8 cm of each pipe served the purpose of an air filled gas collection chamber (headspace). A static chamber technique (Maljanen *et al.,* 2007c; Kasmir-Klemedtsson *et al.,* 1997) was used to collect the gas and samples were collected with a syringe from the headspace on the top of the soil core pipe. Gas samples (2 ml) were collected from each soil

Plate 1. Monitoring soil core CO_2 flux with an LI-6200 analyser

core at regular intervals and the concentration of N_2O and CH_4 were determined with a Varian CP-3800 gas chromatograph (Varian, Inc.). Methane was detected by a flame ionization detector (FID) and the nitrous oxide was detected by a ^{63}Ni electron capture detector (ECD) (Cleemput & Boeckx, 2002). CO_2 was measured with a dynamic chamber technique using the LI-6200 CO_2 analyzer (Li-COR Inc.). For each sampling instance four gas samples were collected to determine the change in gas concentration over time. The time intervals used for N_2O and CH_4 sampling were time 0, 20 minutes, 40 minutes and 60 minutes. The CO_2 analyzer sampled the CO_2 concentration within the dynamic chamber every minute over a four minute interval.

Plate 2. N_2O and CH_4 gas collection

There are two basic parts to the study: On one hand I studied the effect of temperature on GHG fluxes of soil cores representing different landuse (Study 1) and on the other hand the effect of water level and water level fluctuation on soil core gas fluxes (Study 2). The laboratory setup differed somewhat between the two parts of the study:

<u>Study 1</u>.

For estimating the effects of temperature on soil gas flux, the soil cores were incubated at temperatures 3, 8, 13 and 18°C under controlled laboratory conditions. The soil cores were adjusted to each temperature for one month before measurements were performed except for the last temperature (18°C) which had roughly a two months incubation time. The hay field and the drained peat soil cores were standardized to the same water level, 30 cm below surface. The water level of the undisturbed peatland cores was set at 2 cm below surface. The water level was adjusted by a pipe attached in the bottom of the soil cores. All the soil cores from the same land use category were connected to a collective water filled tube where the water level was manually determined. After incubating for a minimum of four weeks at a given temperature each soil core was sampled at four different time intervals, time 0, 20, 40 and 60 minutes. All in all this yielded 384 syringe gas samples that were analyzed for N_2O and CH_4 in a Varian gas chromatograph (see previous chapter). Soil CO_2 flux was estimated with an LI-6200 CO_2 analyzer, as previously described, with readings taken every minute over a four minute period (480 readings in all).

Study 2.

The main objective of this part of the study was to gain information about the variation in emission of N_2O, CH_4 and CO_2 associated with different water table levels (WL). As a part of the study special attention was paid to changes in gas emission in the days after the WL was reset back to base level (-35cm) and water withdrew slowly from the soil, as after a rain event.

The soil cores were adjusted to a steady temperature of 10°C for two months before the WL experiment was performed. Eighteen of the twenty four soil cores used for the land use and temperature study were used to measure the GHG emission at five different water table levels, i.e. -35 cm from surface, -25 cm, -15cm, -5cm and 0 cm respectively. Similar water levels were used by Jungkunst *et* al. (2008) in a comparable study. The water level was adjusted by a pipe attached in the bottom of the soil cores (Plate 3). The same static chamber technique was used to collect gas samples as performed and described in study 1. The soil cores from the undisturbed peatlands were not included in this part of the study. Additionally, variation in gas emission due to WL fluctuations was measured. Once gas sampling was completed at a given WL, the water table was lowered back to basic WL (-35 cm). N_2O and CH_4 emission was measured for 1-6 days for each WL manipulation. The CO_2 emission was only measured at each WL level, not the days following the lowering of the WL. Approximately 9 days were allowed to pass between the different water table level manipulations. The different water table levels were

Plate 3. The water level of the soil cores was adjusted with the aid of an outside cylinder connected to the cores with a pipe.

adjusted through tubes connecting the soil core pipes to an external water filled canister. All six soil cores from each landuse group were collectively attached to common canister. The number of days (1-6) of continued sampling following lowering of the WL was determined after examination of soil water data collected by an automatic logger at one of the drained field sites. It showed a clear reduction and stabilization of the water content by day 6 after rain events.

Data analysis

Regression models were used to determine the significance of all estimated gas fluxes where solely regression models showing an r^2 greater than 0.85 were accepted as giving significant increase in gas concentration over the sampled time interval. For statistical analysis of the data sets the statistical software Minitab was used (Minitab Inc. 14.20, 2005). The data were not normally distributed and hence the non-parametric statistical Kruskal-Wallis test was used to evaluate the statistical significance of the results (Fowler, Cohen & Jarvis, 1998).

4. Results

Descriptive soil properties

The results of the soil analyses were as follows (see table 2). The pH was fairly similar in all the soil cores despite landuse category. The average pH (H_2O) was 4.84 with a range of 4.44-5.12 and the median was 4.89 which are within the typical pH range (4-5.5) for Icelandic Histosol soil at 0-50 cm depth (Arnalds, 2004). The highest pH (5.12) was recorded in soil cores from a Heggstaðir hayfield and in soil cores from the Mávahlíð drained area (5.05). The lowest pH (4.44) was recorded in soil cores from the Kvígstaðir hayfield. The N content was on average 1.66 % for the hayfield soil cores and 1.79 % for the drained soil cores. The C % content was on average 24.6 % for the hayfield soil cores and 27.9 % for the drained soil cores. The vegetated "G" soil cores had 28.7 % C content and 1.9 % N content and a C/N ration of 17.1 respectively. The average C/N ratios from the hayfields were 16.4 and 17.4 for the drained soil cores. These results are similar to other studies in this region (Oskarsson, 1998).

Table 2. Results for N%, C%, C/N ratio and pH of the soil analysis.

Landuse group	N%	C%	C/N ratio	pH
Hayfield	1,62	25,0	16,8	5,1
Hayfield	1,77	26,3	16,5	4,4
Hayfield	1,57	22,4	15,8	4,7
Drained peatland	1,67	25,8	17,1	4,8
Drained peatland	2,03	31,2	17,2	5,0
Drained peatland	1,68	26,7	17,9	5,1
Vegetated drained (G)	1,86	28,7	17,1	4,9
Average hayfield	***1,66***	***24,6***	***16,4***	***4,7***
Average drained	***1,79***	***27,9***	***17,4***	***4,9***

Study 1 - The effect of temperature on the GHG gas flux from organic soil cores representing different landuse regimes

This study examined the effects of varying temperature on the emission of green house gases from soil cores from peatlands under three different landuses. Additionally, the effect of vegetation on soil GHG flux was estimated. The results for each greenhouse gas are treated separately.

Nitrous oxide - N_2O

The results for the emission of nitrous oxide at different temperature and landuse are shown in Figures 3 and 4 (see also Appendix 1).

The results showed no discernable trend in N_2O flux with increasing/decreasing temperatures. The N_2O emission was also not statistically significant between different temperatures (P=0.458). The lowest emission was measured from the peatland soil cores, ranging from no emission (0.00 mg N_2O m^{-2}/day^{-1} (3°C)) to 0.06 mg N_2O m^{-2}/day^{-1} (13°C) (see Appendix 1, and Figures 2 and 3). The emission from drained peatland ranged from 3.73 mg N_2O m^{-2}/day^{-1} (3°C) up to 17.77 mg N_2O m^{-2}/day^{-1} (8°C). At 13°C and 18°C, the emission decreased from the emission at 8°C down to 2.02 mg N_2O m^{-2}/day^{-1} (13°C) and 7.93 mg N_2O m^{-2}/day^{-1} (18°C). The hayfield soil cores had the highest emission ranging from 3.37 mg N_2O m^{-2}/day^{-1} (3°C) to 18.32 mg N_2O m^{-2}/day^{-1} at 18°C. At 8°C they emitted 14.34 mg N_2O m^{-2}/day^{-1} and 8.51 mg N_2O m^{-2}/day^{-1} at 13°C.

The emission pattern was similar for the drained and hayfield soil cores, with an increase in emission from 3°C to 8°C,,a decrease at 13°C and then an increase again at 18°C (Figure 3). The undisturbed peatland was not a source for N_2O in this study.

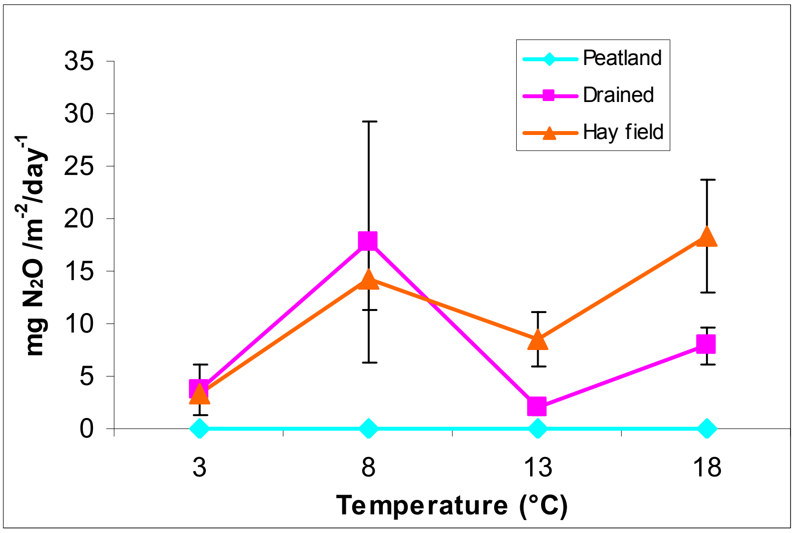

Figure 3. Effects of different temperature treatments on N_2O emission of soil cores representing 3 different landuse regimes (peatland, drained peatland and hayfield) in Borgarfjörður, W Iceland. Bars represent standard error.

The average N_2O emission from all four temperature treatments varied ($P<0.001$) among landuse group (Figure 4). There was a significant difference between peatland and hayfield ($P=0.000$) as well as between drained peatland and hayfield ($P=0.038$).

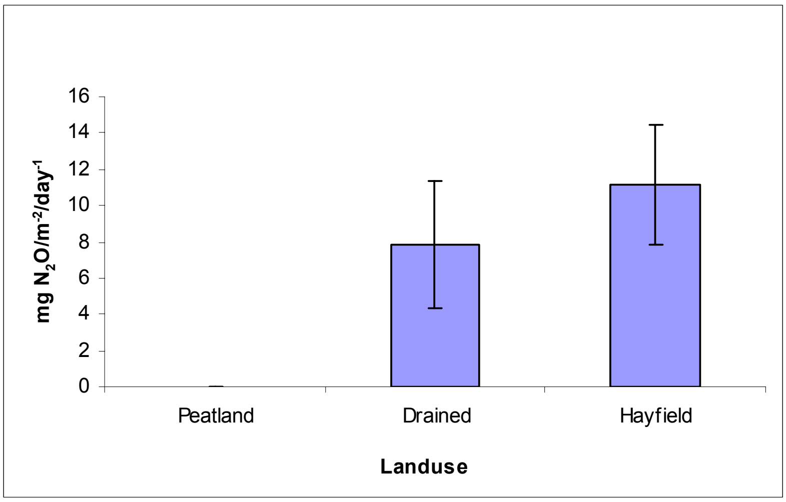

Figure 4. N_2O emission in soil cores representing three different landuse regimes (peatland, drained peatland and hayfield) in Borgarfjörður, W Iceland, averaged for all temperature treatments. Bars represent standard error.

Most N_2O-N was released from the hayfield with an average of 25 kg N_2O-N/ha^{-1}/day^{-1} ranging from 3 to 43 kg N_2O-N/ha^{-1}/day^{-1} between temperature treatments (Appendix 2).

The drained soil cores released on average 17 kg N_2O-N/ha^{-1}/day^{-1} and ranged from 3 to 41 kg N_2O-N/ha^{-1}/day^{-1}.

Methane - CH_4

The undisturbed peatland soil core emissions ranged from 1.06 mg/m^{-2}/day^{-1} at 3°C up to 1,818.54 mg/m^{-2}/day^{-1} at 18°C. At 8°C the emission was 7.63 mg/m^{-2}/day^{-1} and increased to 647.76 mg/m^{-2}/day^{-1} at 13°C. The methane emission increased greatly at the highest temperatures. Very high emission was measured from the undisturbed peatland when the temperature was increased from 8°C to 13°C and 18°C. This result indicates that some of the peatland emission is due to bubble release. By excluding the data suspected to be bubble release, the peatland soil core emission is reduced to 1,302.47 mg/m^{-2}/day^{-1} at 18°C and the drained soil core emission is reduced from 55.40 mg/m^{-2}/day^{-1} to 17.46 mg/m^{-2}/day^{-1}. The other landuse groups were not a source for methane emission (Figures 5 and 6 and Appendix 1).

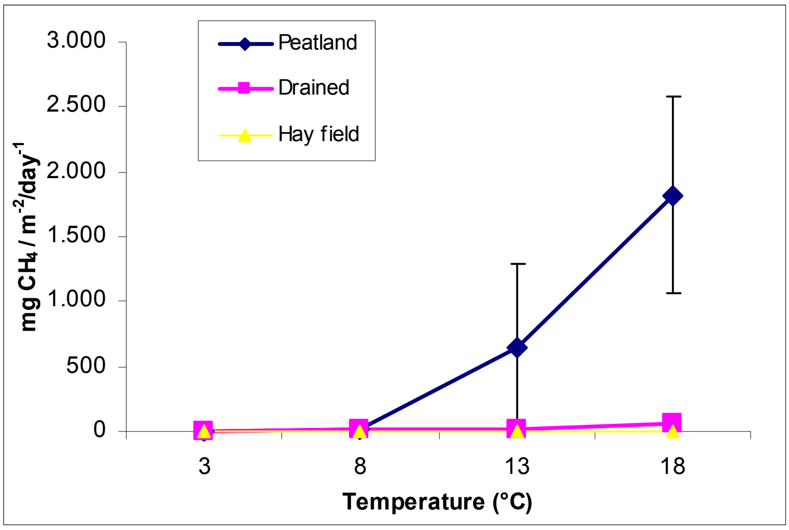

Figure 5. Effects of different temperature treatments on CH_4 emission of soil cores representing 3 different landuse regimes (peatland, drained peatland and hayfield) in Borgarfjörður, W Iceland. Bars represent standard error.

The peatland soil cores were the largest source for methane emission in this study (Figure 6). The drained soil cores produced some quantities of methane especially at the highest temperature (18°C). The hayfield soil cores were not a source of methane.

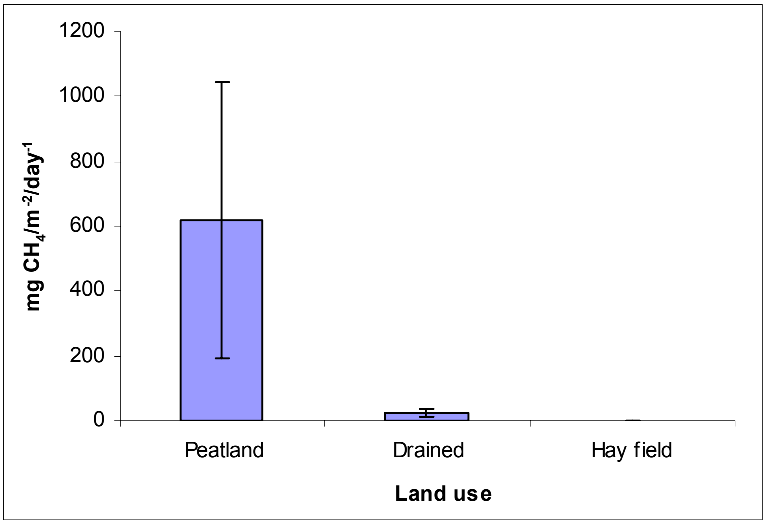

Figure 6. CH_4 emission in soil cores representing three different landuse regimes (peatland, drained peatland and hayfield) in Borgarfjörður, W Iceland, averaged for all temperature treatments. Bars represent standard error.

The methane emission was statistically significant ($P<0.05$) between peatland and drained (no emission from hayfield) and for different temperatures ($P<0.05$).

Carbon dioxide - CO_2

The CO_2 emission increased with increased temperature from 8°C and 13°C to the maximum emission which was achieved at 18°C for all land use groups.

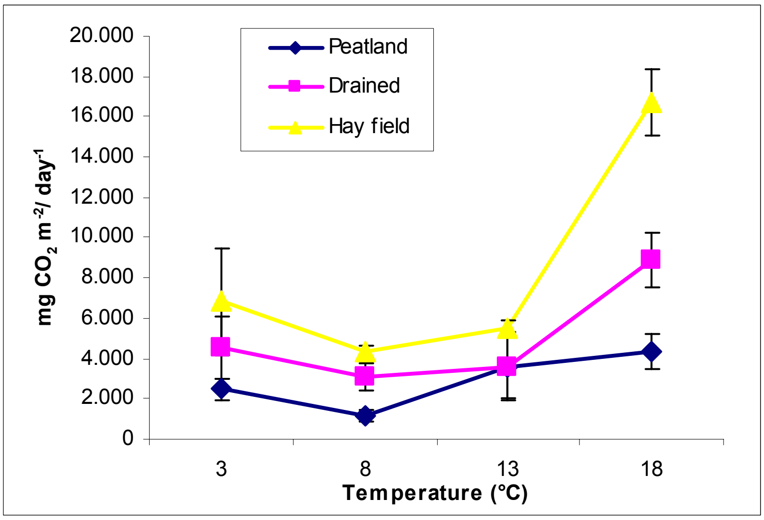

Figure 7. Effects of different temperature treatments on CO_2 emission of soil cores representing 3 different landuse regimes (peatland, drained peatland and hayfield) in Borgarfjörður, W Iceland. Bars represent standard error.

The carbon dioxide emission response to varying temperature was similar for the three landuse groups (Figure 7, Appendix 1). Minor decrease in emission was seen from 3°C to 8°C for all landuse groups. Emission remained level from 8°C to 13°C in the drained soil cores, the hayfields exhibited a minor increase but the peatland soil cores showed the greatest increase in emission. From 13°C to 18°C the emission of the hayfield soil cores increases greatly (11,168 mg CO_2 /m^{-2}/ day^{-1}), the drained cores exhibit a minor emission increase (5,265 mg CO_2 /m^{-2}/ day^{-1}) and there were only small changes in the emission from the peatland cores (709 mg CO_2 /m^{-2}/ day^{-1}) (Figure 7).

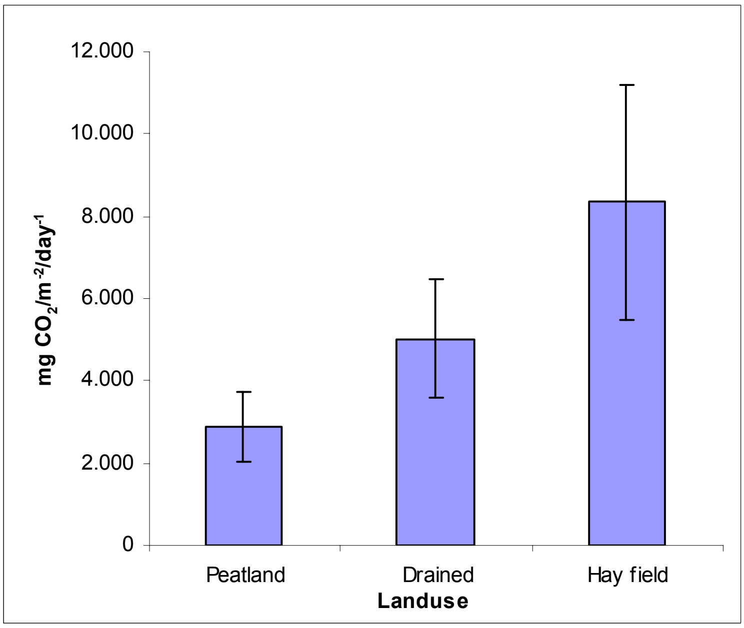

Figure 8. CO_2 emission in soil cores representing three different landuse regimes (peatland, drained peatland and hayfield) in Borgarfjörður, W Iceland, averaged for all temperature treatments. Bars represent standard error.

The emission of CO_2 differed between landuse groups (Figure 8) was statistically significant for peatland versus hayfield soil cores ($P<0.024$) and between peatland and drained peatland ($P=0.001$) but not between drained and hayfield soil cores ($P=0.419$). The average emission due to different temperature ($P<0.001$) and the interaction between land use and temperature ($P<0.001$) were significant. The peatland soil cores ranged from 1,152 mg CO_2 m^{2-1}/day^{-1} (at 8°C) to 4,330 mg CO_2 m^{2-1}/day^{-1} (at 18°C). The emission from the drained soil was higher at 3°C (4,540 mg CO_2 m^{2-1}/day^{-1}) than at 8°C and 13°C (3,076 and 3,607 mg CO_2 m^{2-1}/day^{-1} respectively). A large increase in emission was measured at 18°C (8,872 mg CO_2 m^{2-1}/day^{-1}).

The hayfield soil cores emitted less CO_2 at 8°C (4,311 mg CO_2 m^{2-1}/day^{-1}) than at 3°C (6,847 mg CO_2 m^{2-1}/day^{-1}). At 13°C they emitted 5,526 mg CO_2 m^{2-1}/day^{-1} and increased the emission more than 2-fold at 18°C (16,694 mg CO_2 m^{2-1}/day^{-1}).

Soil cores with vegetation (G)

The gas emission of the six cores with intact vegetation (see previous chapter) was also studied under varying temperatures. These soil cores were set up in the same fashion as the other 18 soil cores except that they were subjected to a light regime of 12 hours of darkness and 12 hours of light with 400 watt bulbs during the whole experiment to allow for vegetation photosynthesis. GHG emission was measured simultaneously from these soil cores and the undisturbed, drained and hayfield soil cores. The results are shown in Table 3.

Table 3. The average emission of N_2O, CH_4 and CO_2 of the vegetated drained soil cores (from Heggstaðir) at different temperatures

	3°C	8°C	13°C	18°C
mg N_2O /m^{-2} /day^{-1}	19.10	16.33	19.80	13.54
mg CH_4 / m^{-2}/day^{-1}	0	0.11	0.035	1.502
mg CO_2 m^{-2} / day^{-1}	4,088	3,815	7,202	19,809

Nitrous oxide - N_2O

The highest emission was measured from the G soil cores at 13°C (19.80 mg N_2O m^{-2}/day^{-1}) almost the same as at 3°C (19.10 mg N_2O m^{-2}/day^{-1}). The loss of N was higher from the vegetated soil cores (31 kg N_2O-N/ha^{-1}/day^{-1}) than the drained soil cores without vegetation (17 kg N_2O-N/ha^{-1}/day^{-1}). A comparison of the N_2O emission of vegetated and non-vegetated soil cores from the same drained site is shown in Figure 9. The vegetated soil cores emit 2-fold the N_2O emission of the soil cores without vegetation and was significantly different between vegetated and not vegetated ($P<0.05$).

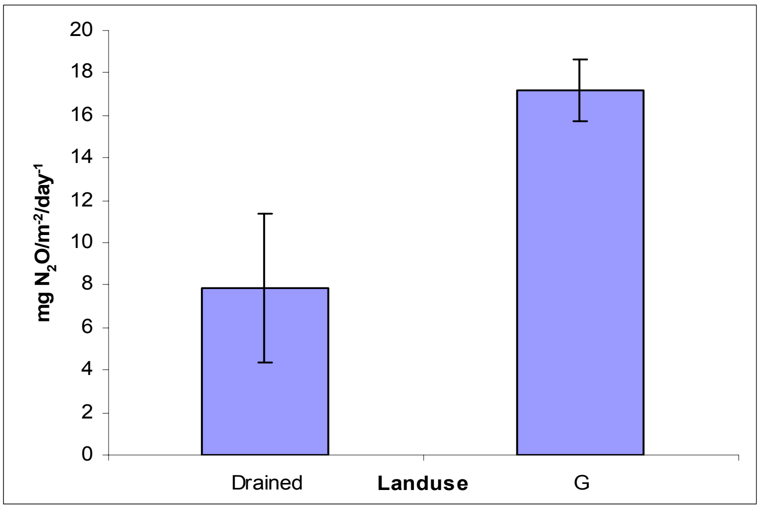

Figure 9. N_2O emission in soil cores representing the same landuse group (drained peatland) with (G) and without (Drained) vegetation (in Borgarfjörður, W Iceland) averaged for all temperature treatments. Bars represent standard error.

Carbon dioxide - CO_2

The same comparison between vegetated and non-vegetated soil cores was made for carbon dioxide with similar results as for nitrous oxide. The vegetated soil cores emitted twice as much carbon dioxide as the cores without vegetation, but was not statistically significant ($P>0.05$) (Figure 10).

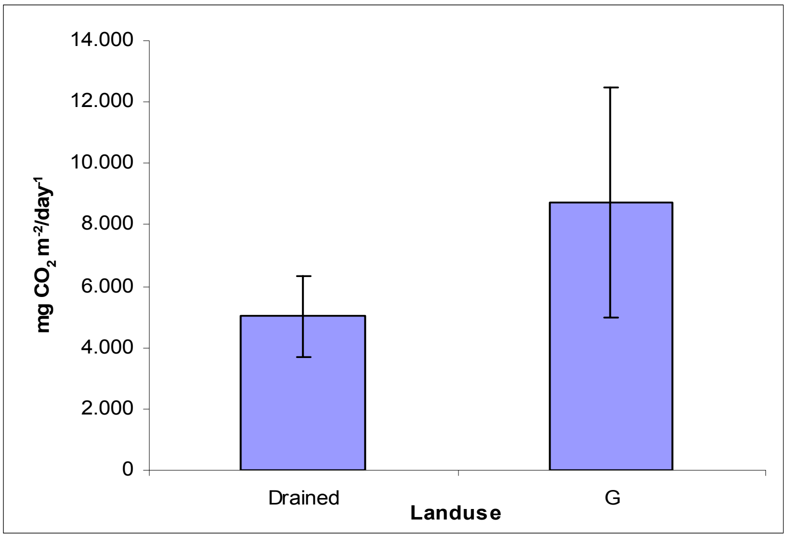

Figure 10. CO_2 emission in soil cores representing the same landuse group (drained peatland) with (G) and without (Drained) vegetation (in Borgarfjörður, W Iceland) averaged for all temperature treatments. Bars represent standard error.

Methane - CH_4

A comparison between the methane emission of vegetated and non-vegetated soil cores showed different emission response for methane than for the other two gases. The drained without vegetation soil cores exhibited large emission and there was almost no emission from the vegetated cores (Figure 11). The emission difference was statistically different between landuse groups (P<0.05).

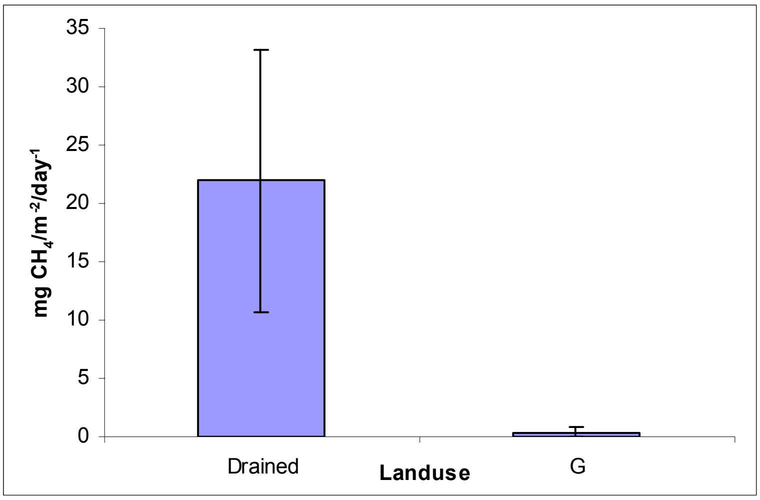

Figure 11. CH_4 emission in soil cores representing the same landuse group (drained peatland) with (G) and without (Drained) vegetation (in Borgarfjörður, W Iceland) averaged for all temperature treatments. Bars represent standard error.

Q_{10} values

Q_{10} values were not calculated for the nitrous oxide emission which was not statistically significant for temperature. The calculated Q_{10} values for the CO_2 and the CH_4 emission are shown in table 4.

Table 4. Calculated Q_{10} values for methane and carbon dioxide emission.

	CH_4	CO_2
Peatland	114,7	1,5
Drained	1,9	1,6
Hayfield	0,0	1,8
Vegetated drained	9,1	2,9

Study 2 – The effect of water table level and water table fluctuations on the GHG gas flux from organic soil cores representing different landuse

No CH_4 emission was detected for the different water table levels (WL) or for the fluctuations, which might reflect that the incubation time (24 hours) was too short for the methan producing microorganisms, and will therefore not be discussed further. The results for the emission of CO_2 and N_2O are shown in Figures 12 through 17, in Table 3 and in Appendix 2.

Carbon dioxide - CO_2

CO_2 was only measured at the different water table levels and not for the days (1-6) of drawdown. The CO_2 emission was significantly different between different WL for the hayfield soil cores (P=0.002) but not for the drained peatland soil cores (P>0.005), and not between landuse groups (drained versus hayfield) (P>0.005) (Figure 12).

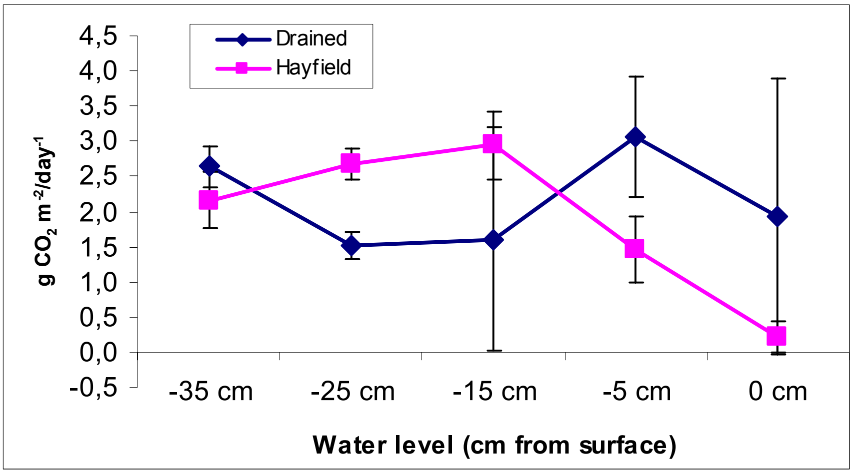

Figure 12. Effects of different water level treatments on CO_2 emission of soil cores representing 2 different landuse regimes (drained peatland and hayfield) in Borgarfjörður, W Iceland. Bars represent standard error.

The emission from the hayfield increased slowly until WL -15 when the emission decreased (Figure 12). The drained soil cores showed slightly different patterns than the hayfield soil cores, they emitted more at -35 cm, decreased at -25 cm and then they stayed steady. An increase was seen at -5 cm and then the emission decreased again at 0 cm. The average emission for *all* the water levels was slightly higher from the drained peatland soil cores compared to emission from the hayfield soil cores; 2.15 CO_2 m^{-2}/day^{-1} and 1.89 CO_2 m^{-2}/day^{-1} respectively.

Nitrous oxide - N_2O

N_2O was measured both at different water levels and at day 1, 2, 3 and 6 after drawdown to simulate the water level fluctuation after a rain event.

The N_2O emission pattern was similar for the drained and hayfield soil cores (Figure 13). The magnitude of the emission however was different. The highest emission rate was seen on the first day of draining and decreased until day 6 when the emission was almost back to initial levels (at day 0).

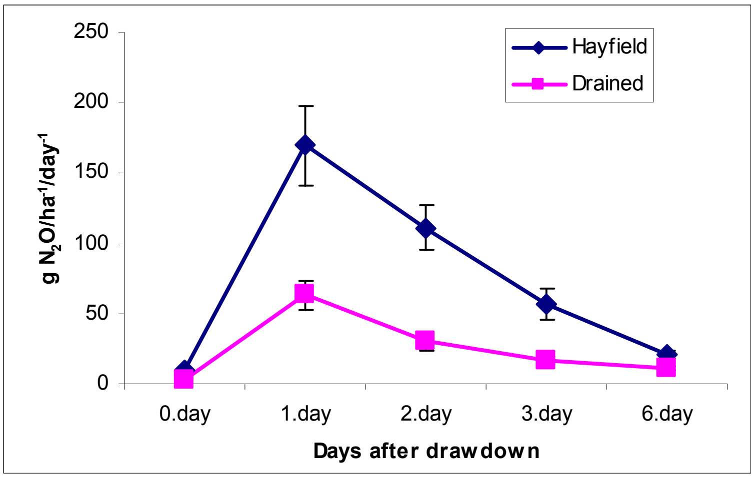

Figure 13. Effects time elapsed from water table drawdown following a previously raised water table (averaged over all water table levels) on N_2O emission of soil cores representing 2 different landuse regimes (drained peatland and hayfield) in Borgarfjörður, W Iceland. Bars represent standard error.

The first day of drawdown is clearly emitting most N_2O from drained peatland without vegetation for all the different water table levels except from -15 cm below surface where very low emission was measured (Figure 14).

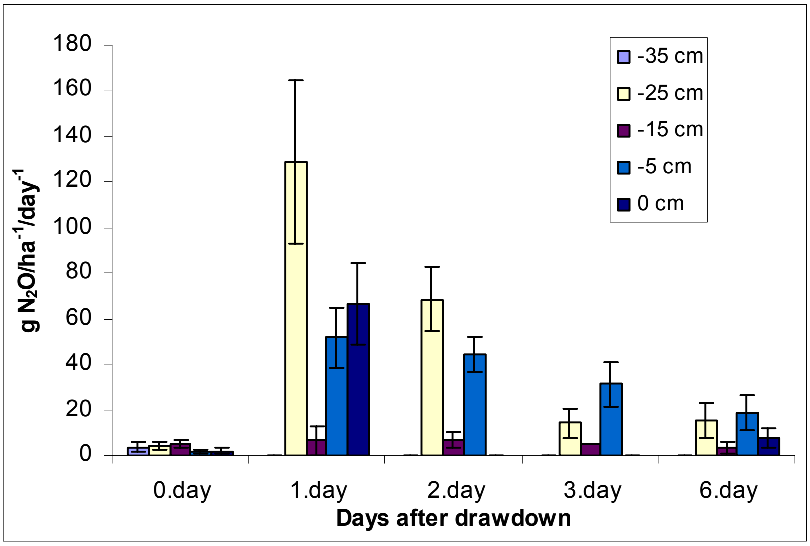

Figure 14. Average N_2O emission from drained peatland soil cores in Borgarfjörður, W Iceland in the days following water table drawdown from four different water levels (0, -5, -15 and -25 cm below surface). Bars represent standard error.

The results showed a clear effect on the N_2O emission from different WL and from the water level fluctuations for the drained peatland (Figure 14). Day 0 is the measurement on the WL; -35 cm, -15 cm, -5 cm and 0 cm respectively. Day 1st is the first day (24 hours) after water has been drained out (or is draining out) back to initial level (-35 cm). Day 2nd is the second day after water was drained out etc. Measurements at day 6th show that the emission is still a little higher than before the fluctuation but decreasing rapidly.

The first day after the WL has been lowered down to -35 cm has the highest emission, particularly after the WL was raised to -25 cm. The lowest emission was measured after WL -15 cm and is suspected to be due to the effect of the high emission at the prior water level (-25 cm). The non-parametric Kruskal-Wallis test (Minitab 14 Inc., 2008) showed that the N_2O emission was significantly different between drained and hayfield ($P<0.000$), WL ($P<0.000$) and the WL fluctuations ("days") ($P<0.000$) the following 1-6 days while water was draining out of the soil both for the drained soil cores and the hayfield soil cores (see Figures 14 and 15).

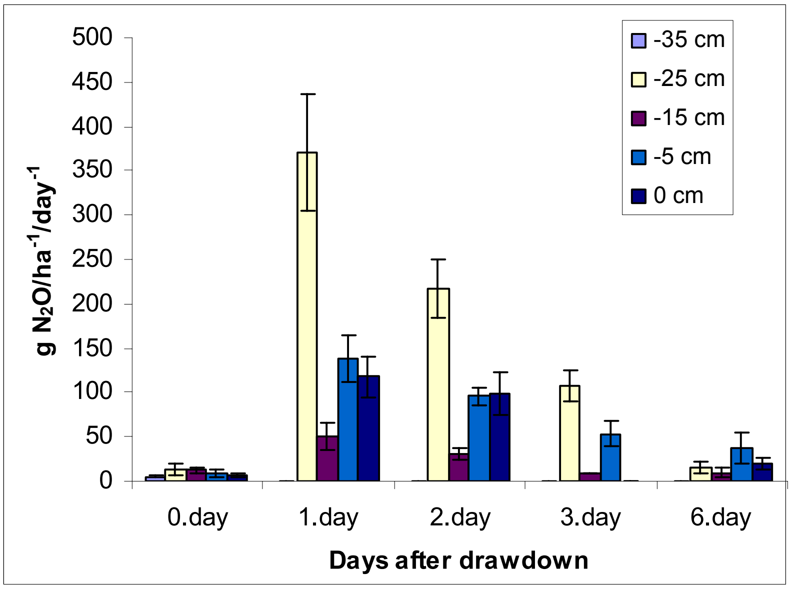

Figure 15. Average N_2O emission from hayfield soil cores in Borgarfjörður, W Iceland in the days following water table drawdown from four different water levels (0, -5, -15 and -25 cm below surface). Bars represent standard error.

The behavior of the N_2O emission was similar for the drained soil cores and the hay field on drained soil cores during the water level fluctuations. However, the magnitude of the emission was on average 2.9-fold larger in the hayfield averaged for all treatment water levels and days.

Soil cores with vegetation (G)

Soil cores with vegetation (group G) had intact vegetation whereas the other group (drained and hayfield) had the vegetation removed. Aside from the difference of having intact vegetation the G-soil cores were collected in the same fashion as the other cores.

Carbon dioxide - CO_2

Carbon dioxide was only measured with different WL and not sequentially for the days following water drawdown. The CO_2 emission was not significantly different between different WL for the G soil cores (P=0.774). The emission pattern was similar for the drained peatland and G-soil cores. However, the magnitude between the two drained groups, differing only in being with or without vegetation, was significantly different (P=0.000) with the emission being on average 3.8-fold higher from the G-soil cores.

The drained soil cores (both with and without vegetation) showed little different pattern than the hayfield soil cores, they emitted more at -35 cm, decreased at -25 cm and then the G-soil cores increased their emission, whereas the non-vegetated stayed steady (Figure 16). The G-soil cores decreased their emission from -5 -0 cm. Average emissions for all the different water levels were higher for the G-soil cores than for the drained, vegetation-less cores and the hayfield soil cores.

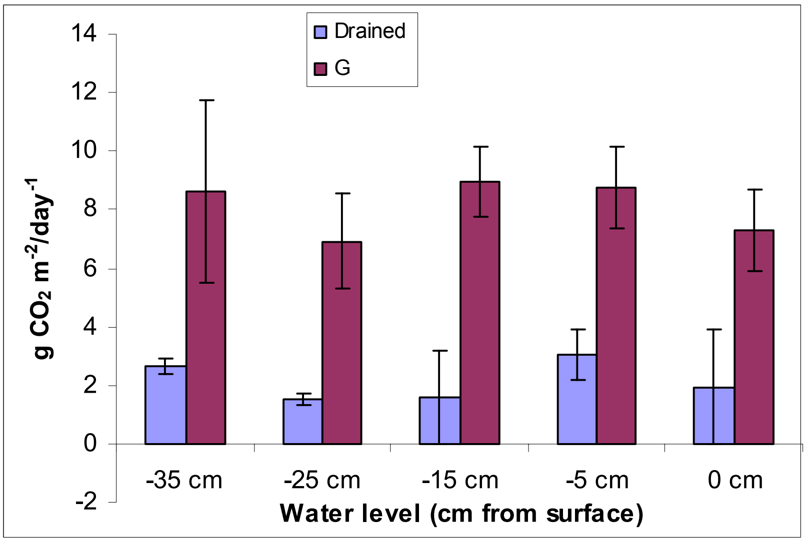

Figure 16. Average CO_2 emission from drained peatland soil cores in Borgarfjörður, W Iceland with (G) and without (Drained) vegetation at different water table levels. Bars represent standard error.

Nitrous oxide - N_2O

In the water level fluctuation study, N_2O emissions of G-soil cores and drained cores without vegetation were similar (Figure 17).

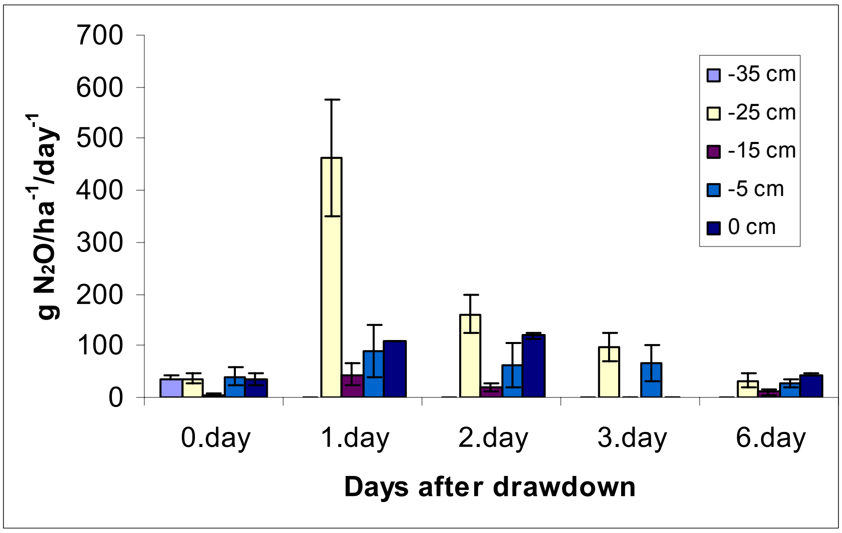

Figure 17. Average N_2O emission from vegetated drained peatland (G) soil cores in Borgarfjörður, W Iceland in the days following water table drawdown from four different water levels (0, -5, -15 and -25 cm below surface). Bars represent standard error.

However, the emission magnitude was higher from the G-soil cores than from the soil cores without vegetation (Table 3). The emission was significantly different ($P<0.05$) between soil core with (G) and without vegetation.

Table 5. Average emission for all the days (1-6) from drained peatland with (*vegetated G*) and without vegetation (*not vegetated*) at different water levels (g N_2O ha^{-1}/day^{-1}).

WL	Vegetated (G)	Not vegetated
-35 cm	34,1	3,8
-25 cm	158,2	46,2
-15 cm	20,2	5,5
-5 cm	57,6	29,6
0 cm	76,7	18,9

The G soil cores emit 4,4-fold more N_2O during WL fluctuations than the soil cores with removed vegetation (Table 5). The greatest differences were recorded at WL -35 below surface where the vegetated soil cores emitted 9-fold more than the non-vegetated. The highest emission during WL fluctuations was the first day of drainage from WL -25 cm. The lowest emission for both groups was after draining from -15 cm.

5. Discussion

Land use

This experimental study indicates that greenhouse gas emission differs significantly among major landuse regimes in Iceland, with N_2O (Figure 4) and CO_2 (Figure 8) emission increasing but CH_4 emission (Figure 6) decreasing when land is converted from pristine peatland to drained peatland and hay field. Large regions of Icelandic peatlands have been drained. Some areas have been converted to hay fields, others are used for grazing and some are abandoned and not longer utilized for agriculture (Oskarsson, 1998). The results of this study indicate that drained areas emit N_2O in high quantities along with a doubling of CO_2 emission compared to originally undisturbed peatland which release no N_2O and half of CO_2 compared to the drained soil cores (Appendix 1).

CO_2 sequestration by afforestation is a reversible process where carbon is bound in wood for a period of time until the material is decomposed and CO_2 is released again. N_2O emission is not a reversible process (Smith & Conen, 2004) and considering these facts, mitigations of GHG emission from the agricultural sector in Iceland would most likely benefit more from wetland restoration than afforestation.

Temperature

The response of CO_2 to different temperatures (Figure 7; Appendix 1) is similar compared to other studies (Maljanen *et al.*, 2002). The carbon dioxide production is higher from the fertilized hayfield where there are less limiting factors for soil microorganism (Houghton *et al.*, 2001; Clayton *et al.*, 1996). The Q_{10} values for the CO_2 flux (see table 4) are comparable to values reported by other studies (Oskarsson, 1998; Raich & Schlesinger, 1992). The highest value is from the vegetated drained soil cores (2.9) and can be explained by the enhanced emission derived from the substrate released from the roots and the more open gas transport pathway system in the soil with roots. The unrealistically high Q_{10} value for the methane flux from the peatland (114) is expected to be due to bubble release at 18°C. These results also indicate that the sampling methodology is suspected to be affecting the release of large quantities of methane bubbles. However, a large increase in the bacterial community and thereby production of methane is realistic, but there might be methane gas in some soil cores that are not released which also migh be one reason for the large SE and variation in the production measured.

The exponential response of the methane production from the peatland soil cores (Figure 5) in this study is larger than found in other studies for the higher temperatures (13°C and 18°C) (Komulainen, *et al.,* 1998; Smith & Conen, 2004) but can be explained to some extent by bubble release. Additionally, longer incubation time before measurements at this temperature compared to the other temperatures can also affect the high production measured. However, the incubation time is the same for all the landuse groups and does not affect the production of methane for other landuse groups than the undisturbed peatland. Drained soil cores produce significant amounts of methane as well (Figure 5 and 6) which can be produced in lower anaerobic soil layers. It can be speculated that the methanotropic bacteria in the Icelandic peatlands are highly productive and respond quickly to increased temperature. However, the fact that the incubation time for 18°C measurements was almost two-fold the incubation time for the other temperatures (1 month versus almost 2 months for 18°C) may explain the steep increase in the methane production even though the other gases did not respond similarly.

The N_2O response to temperature was neither linear nor exponential (Figure 3). The emission was low at 3°C but then increased both for drained and hayfield at 8°C. Then both land use categories (drained and hayfield) reduced the emission at 13°C. One of the factors affecting the drop in the emission was speculated to be due lack of raw material (N-compounds) for N_2O production, i.e. most of it being completed during the high emission at 8°C. The N_2O flux from soil is highly variable both temporarily and spatially which affects the estimates given by the flux chamber measurements (Kasmir-Klemedtsson *et al.,* 1997). The production of nitrous oxide is not as a predictable as production of CO_2 and CH_4. Laboratory studies performed in Finland found that N_2O

emission increased when temperature was lowered from +15°C to 0°C (Koponen *et al.,* 2004). However, the N_2O emission results at temperature 3°C for the drained and hayfield soil cores are comparable to the results of a field study by Maljanen *et al.* (2003). They got a mean N_2O flux during snow free period of 2.4-7.8 mg N_2O m^{-2}/day^{-1}. Smith *et al.* (2003) and Maag and Vinther (1999) showed an exponential increase in N_2O production with increased temperature which is contrary to this study's findings.

Different gas production can also be dependent on the differences between experimental designs. In this study the temperature was increased from 3 up to 18°C. Other studies start on the highest temperature and lower it in a stepwise manner. This can be one of the reasons why the results are contrary from this studies findings concerning N_2O. Other reasons can include different temperature optima for bacterial populations. For example, the Icelandic bacterial communities may be adapted to low temperatures and exhibit maximum productivity at temperature close to their natural environment, which is highest at 8°C for the drained soil cores. The hayfield soil cores produced most at the highest temperature, and that can be explained by the larger nitrogen resource with additions through fertilization. Limiting environmental factors can affect the production of N_2O and dilute the effects of emission due to temperature which is therefore not detected (Davidson & Schimel, 1995).

The results give important information for future scenarios with increased temperatures due to climate change. Temperatures above 8-10°C increase the methane emission sharply. Future studies should calculate the annual emission from days with temperatures above before mentioned temperatures. It is also necessary to investigate the length of the incubation time the methanotropic bacteria needs in Icelandic peatlands to respond with increased methane production. Summers in Iceland can have many warm days and the number of these warm days might increase with predicted increased temperatures which can increase the GHG production from wetland significantly in the future.

Water level and water level fluctuations

No CH_4 emission was recorded in this study which might be explained by the short incubation time at each WL level (24 hours) followed by 6 days of drawdown back to base level (-35 cm). Smith and Conen (2004) suggest that soils have to be completely waterlogged for several days without interruption for methanogenesis to occur (Smith & Conen, 2004). Longer waterlogged periods may have been needed especially in view of the fact that the soil cores were wetted from a pipe in the bottom of the soil cores and therefore might have become dry on the surface.

Different water levels and water level fluctuations have been shown to increase N_2O emission (Regina *et al.,* 1999; Jungkunst *et al.*, 2008; Kasimir-Klemedtsson *et al.*, 1997) comparable to the results of this study (Figure 13). Regina *et al.* (1999) drained soil cores from waterlogged peatlands and got an increased N_2O emission after only 14 weeks. Raising the water table again caused a cessation of the N_2O production immediately and the authors concluded that the water table did not need to be lowered for a long time to change a peatland from a sink to a source of N_2O production.

However, in this study, the water level fluctuations had a greater affect on the N_2O emission than did different water table levels, though both variables affected the emission significantly (Figures 13, 14 and 15). The N_2O emission was highest the days following the reduction of the water table from -25cm to -35cm and the emission was lowest the days following the reduction from -15 to - 35cm. High concentration of NH_4^+ / NO_3^- can be produced within the soil when water fluctuation cause alternating anaerobic and aerobic conditions. This can explain the large N_2O emission difference between the 1-6 days after the -25cm WL and the -15cm WL. The high emission on the -25 cm cycle and then the very low emission on the -15 cm cycle could be explained by a large quantity of raw material (N-compounds) available for N_2O production which had been produced in denitrifying processes and was then released the first three days after draining the water out of the -25 cm cycle. The next cycle (-15 cm) N_2O emission is low and the reason is speculated to be due to little raw material (N-compounds) left from the high emission only approximately 3-8 days earlier. A reduction in emission seen in study 1 at temperature 13°C might also be explained by a similar reason (Figure 3).

In evaluating these results, it must be taken into account that the experimental water levels in this study are not comparable to natural water levels in the field which are usually much lower for hayfields and drained rangeland. For technical reasons the WL in this study was higher than in the field, but comparable to other laboratory studies on water level effect on GHG emission (Jungkunst *et al.,* 2008).

The largest change in N_2O production and emission of drained wetlands however, is not only associated with lowered water level, but also the affect of water level fluctuations during and after rain events as this study indicates. Wetlands are more stabile in terms of the water table level and there are smaller fluctuations due to rain events etc. Water table levels in drained peatland can fluctuate greatly due to heavy rainfall or when the soil and snow thaw after the winter. However, dry summers with little precipitation can increase the WL fluctuations also in the undisturbed peatland. Regina *et al.* (1999) suggest that short dry periods (i.e. one summer) can cause significant changes in N_2O production in an undisturbed peatland. This will most probably differ between wetlands sites due to differences in size, thickness of peat, ground water level and other site specific environmental factors. Emission of GHG from cultivated organic soils is known to continue for at least 20-30 years after cultivation activities has ceased (Maljanen *et al.,* 2007a).

Vegetation

Vegetation had a significant effect on the GHG emission in this study. The magnitude of gas production from the drained soil cores with vegetation was much higher for N_2O (Figure 9 and Table 3) and CO_2 (Figure 10 and 16) but the emission was lower for methane (Figure 11) compared to the drained soil cores without vegetation.
Other studies also show increased emission from vegetated soils compared to non-vegetated (Maljanen *et al.*, 2003; Ineson, Coward & Hartwig, 1998; Stefanson, 1972; Dobbie & Smith, 2002). This is ascribed to enhanced root-derived available soil C acting

as an energy source for microbial denitrification (Ineson, Coward & Hartwig 1998). However, contrary results have also emerged from other studies where N_2O emission from cropland under barley and fallow soil plots was higher than the emission from cropland under grass (Maljanen *et al.,* 2007b). Maljanen *et al.* (2007a) found the CO_2 emission to be lower in the plots with vegetation compared to the plots under fallow soil.

This part of the study gained information on the effect of plants on the production of the three major GHG's. The results indicate that vegetation has a major effect on GHG emission. However, further investigations are needed to quantify the sources affecting these results.

CO_2 equivalents

The different behaviour, or response, of the three GHG's to varying temperature and water level makes comparison between the three landuse groups difficult. The use of CO_2-equivalents, where the results for CH_4 and N_2O are converted into CO_2-equivalent values by use of the GWP constants for these gases, allows for comparison between landuse groups. The two studies are however not totally comparable due to differences in study performances. All the results in CO_2 equivalents g/m^{-2}/day^{-1} are shown in Appendix 4.

A rough estimate of the results of different landuse groups and different temperatures (study 1) calculated in CO_2 equivalents shows the highest emission from the undisturbed peatland, with 18.55 g/m^{-2}/day^{-1} CO_2 EQ from 13°C and 46.16 g/m^{-2}/day^{-1} from 18°C (Appendix 1). The drained soil cores emitted 4.52 g/m^{-2}/day^{-1} at 13°C) and 12.60 g/m^{-2}/day^{-1} at 18°C and the hayfield soil cores emitted 8.16 g/m^{-2}/day^{-1} at 13°C) and 22.39 g/m^{-2}/day^{-1} at 18°C respectively. However, at the lower temperature treatments (3 and 8°C), the highest emission was similar for the drained and the hayfield at 8°C, 8.89 g/m^{-2}/day^{-1} and 8.74 g/m^{-2}/day^{-1} respectively. Drained soil cores emitted less than the hayfield at 3°C (5.84 g/m^{-2}/day^{-1} and 7.90 g/m^{-2}/day^{-1} respectively). The undisturbed peatland had the lowest emission at the lower temperatures, 2.49 g/m^{-2}/day^{-1} at 3°C and 1.33 g/m^{-2}/day^{-1} at 8°C respectively (Figure 18).

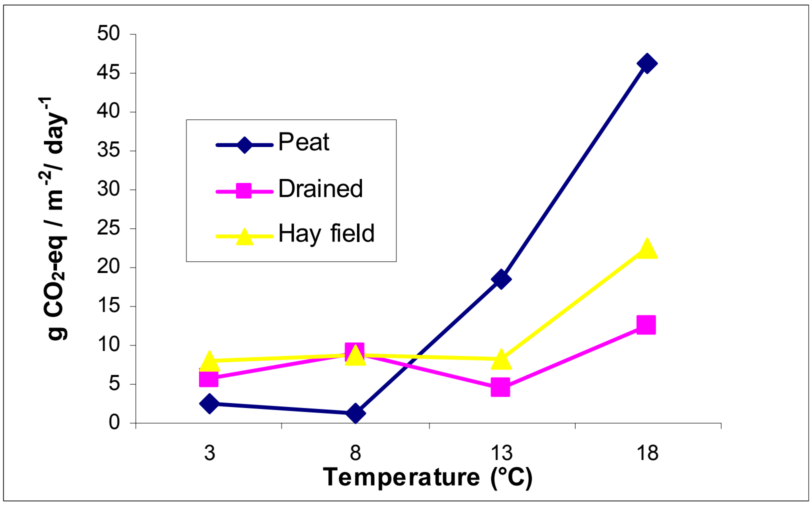

Figure 18. Sum of the GHG emissions of soil cores in Borgarfjörður, W Iceland at different temperatures given in CO_2 equivalents for the three landuse groups. Calculations made from emission results of study 1.

It must be taken into account that study 1- temperature is measuring emission from 3°C-18°C. This temperature only occurs for a few months of the year in Iceland. The annual average temperature for this area is 3.2°C with a mean summer temperature of 8.1°C. This means that the annual emission must be measured also for the colder periods of the year and temperatures at 18°C are rarely experienced for longer periods at the time in Iceland. The highest temperatures are not common for longer periods at the time, but occasionally summer days can be even warmer than this study monitor.

The water fluctuation (WL) treatments for different landuse groups (study 2) were not repeated on the undisturbed soil cores. Water level fluctuations due to precipitation and the following drawdown of water table level are a common occurrence in this region. It would be suspected to be a highly pronounced environmental factor which on an annual basis might be producing a large amount of the annual GHG in this region. The results for study 2 calculated in CO_2 equivalents are shown in Appendix 4.

6. Conclusions

Both soil temperature and water levels fluctuate frequently in natural ecosystems, affecting the GHG production in soil. Anthropogenic modification of ecosystems due to changed landuse management is an additional cause of changes in GHG production. The results from these studies indicate that draining of wetlands increases N_2O and CO_2 emission and almost extinguish soil CH_4 emission. High soil temperature ($\geq$13°C) increased the CO_2 emission from drained soil cores and increased the CH_4 emission from the peatland cores drastically. Short-term water level fluctuations markedly increased N_2O emission of drained soil cores with the highest emission recorded during the first day after initiation of water level drawdown. The number of precipitation events and the associated in-soil water level fluctuation will affect the annual N_2O emission of drained peatlands. More frequent precipitation events during winter will likely increase the annual N_2O emission whereas warmer summers might increase the methane emission in an almost exponential manner as demonstrated in this study. Further studies on in-field methane emission and its response to short periods of high temperature intervals are needed to validate these results. Further field studies during the winter period and of water level fluctuations during low temperatures are necessary to predict the annual emission of drained and undrained peatlands alike. Additionally, important information was gained on the effect of vegetation (G soil cores) on soil GHG production which was a major factor both in study 1 and study 2. To be able to quantify the GHG emission from different landuse categories, more studies are needed to investigate further the effects of different plant species and plant communities on the GHG production.

Natural wetlands are recognized as being sources of CH_4 emission to the atmosphere and draining reduces the CH_4 emission but increases, on the other hand, the emission of N_2O and CO_2. According to the calculated CO_2 EQ, methane emission from the undisturbed peatlands in the temperature study was higher than the total emission from the temperature *and* the water level study for all the gases aggregated. Considering the premises of these studies, it could be concluded that draining reduces the total GHG emission, calculated in CO_2 equivalents, under high temperature conditions. However, other factors such as biological diversity and wetland habitat for all organisms living there are issues important for everyone concerned about healthy landuse management practices. Additionally, it must be remembered that the annual temperature in Iceland and Borgarfjörður area is not as high as in study-1 and that one month with 13°C and 18°C is not realistic. Further information is needed to find out how short the temperature interval must be to accelerate the methane emission in peatlands.

In Iceland, water level fluctuations following precipitation events are common and therefore likely to contribute substantially to the annual N_2O emission. That, along with the fact that N_2O emissions can be quite high during low temperatures in winter (Koponen *et al,* 2004; Maljanen[a] *et al,* 2007), implies that the winter period is of particular concern regarding overall N_2O emission of drained peatlands. Considering that N_2O is 310 times as powerful a greenhouse gas as CO_2, the overall result of draining on an annual basis (considering low average annual temperature in Iceland) is an increase in the total GHG emission.

A landuse management plan, aimed at minimizing GHG emission, must have an annual scope and include all three GHG's included in the calculation to find the best solution. Mitigation measures towards a reduced GHG emission policy in Iceland could include the restoration of wetlands by filling in ditches of drained areas that are no longer in agricultural use. Iceland's most used CO_2 sequestration measure, as a part of the mitigation plan towards reduced GHG emission, is afforestation (The Environmental Agency of Iceland, 2008). Restoration of wetlands should be considered as a realistic tool as well as afforestation and land reclamation. In addition it can be mentioned, although not within the scope of this study, that other benefits are likely to be gained by wetland restoration, such as water regulation, nutrient retention and wetland species habitat.

If the future climate change scenario will be increased temperature with more rainfall during winter and warmer and dryer summers, it could result in more methane emission from wetlands due to higher temperatures (although counteracted by drier summers) and more water level fluctuations increasing the N_2O emission during winter. However, it must be remembered that the results from this study indicate that the temperature needs to be higher than ca 10-13°C to increase the methane production from the wetlands and it is (as yet) not fully know how methanogenic bacteria will respond to short-periods of high temperatures. It is unrealistic to expect being able to have an affect on the future increased temperature, but we can have an effect on soil water table level fluctuations by filling up ditches. More stabile water table levels in undisturbed or restored wetlands might decrease the N_2O winter emission associated with rainfall events and hence reduce the annual emission from the agricultural landuse sector in Iceland.

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Appendix 1 Gas emission from study 1 – temperature

The average gas emission quantities of N_2O at different temperatures given in mg N_2O m^{-2}/day^{-1}

mg N_2O /m^{-2} /day^{-1}	3°C	8°C	13°C	18°C
Peat land	0,00	0,00	0,06	0,00
Drained	3,73	17,77	2,02	7,93
Hay field	3,37	14,34	8,51	18,32

The average gas emission quantities of CH_4 at different temperatures given in mg CH_4 m^{-2}/day^{-1}

mg CH_4 /m^{-2}/day^{-1}	3°C	8°C	13°C	18°C
Peat land	1,06	7,63	647,76	1.818,54
Drained	6,62	13,02	12,85	55,40
Hay field	0	0	0	0,85

The average gas emission quantities of CO_2 at different temperatures given in mg CO_2 m^{-2}/day^{-1}

mg CO_2 m^{-2} /day^{-1}	3°C	8°C	13°C	18°C
Peat land	2.464	1.152	3.621	4.330
Drained	4.540	3.076	3.607	8.872
Hay field	6.847	4.311	5.526	16.694

The average gas emission quantities of all the three GHG's at different temperatures for the vegetated drained soil cores.

	3°C	8°C	13°C	18°C
mg **N_2O** /m^{-2} /day^{-1}	19,10	16,33	19,80	13,54
mg **CH_4** /m^{-2}/day^{-1}	0	0,11	0,035	1,502
mg **CO_2** m^{-2} /day^{-1}	4.088	3.815	7.202	19.809

Appendix 2 Gas emission from study 2 – water level fluctuations

g N2O /ha/d	WL								
Drained	**-35**	**SE**	**-25**	**SE**	**-15**	**SE**	**-5**	**SE**	**0**	**SE**
0.day	3,79	1,90	4,20	1,70	4,91	1,70	1,54	0,62	1,96	1,36
1.day	*	*	128,74	35,97	6,55	6,54	51,70	12,92	66,20	17,99
2.day	*	*	68,53	14,30	7,14	3,37	44,40	7,94	0	0
3.day	*	*	14,13	6,43	4,99	*	31,40	9,66	*	*
6.day	*	*	15,42	7,59	3,77	2,56	18,83	7,53	7,50	4,05

hayfield	**-35**	**SE**	**-25**	**SE**	**-15**	**SE**	**-5**	**SE**	**0**	**SE**
0.day	4,30	2,03	12,93	7,03	12,34	2,76	9,06	4,63	6,72	3,05
1.day	*	*	370,50	65,96	50,83	14,86	138,41	26,31	117,37	23,48
2.day	*	*	217,54	33,11	30,80	6,13	95,51	10,59	98,28	23,71
3.day	*	*	107,42	16,97	9,01	*	53,30	14,85	*	*
6.day	*	*	15,31	6,87	9,68	4,59	36,69	17,23	20,01	7,32

G	**-35**	**SE**	**-25**	**SE**	**-15**	**SE**	**-5**	**SE**	**0**	**SE**
0.day	34,13	9,613	36,64	9,24	5,50	3,5	40,51	18,66	34,09	11,11
1.day	*	*	462,99	113,46	44,00	22,6	89,62	48,79	108,63	1,52
2.day	*	*	160,46	36,43	19,98	8,96	62,80	42,38	119,46	6,84
3.day	*	*	98,06	26,17	*	*	67,87	34,94	*	*
6.day	*	*	32,81	15,19	11,19	5,85	27,32	8,02	44,43	0,94

Appendix 3 N_2O-N

N_2O-N kg/ha^{-1}/yr^{-1}

N2O-N kg/ha-1/yr-1					
TEMP°C	**3**	**8**	**13**	**18**	**Average LU**
Drained	3	41	5	18	17
Hay field	3	33	20	43	25
G	15	38	50	21	31
Peatland	0	0	0	0	0
average°C	7	38	25	27	

WL	**-35**	**-25**	**-15**	**-5**	**0**	**Average LU**
Drained	7,9	10,7	1,3	8,1	4,4	**6,5**
Hayfield	0,9	31,9	6,0	15,5	14,1	**13,7**
G	1,0	36,5	4,7	13,4	17,8	**14,7**
Average WL	**3,3**	**26,4**	**4,0**	**12,3**	**12,1**	

Average-all land uses

WL-days	**-35**	**-25**	**-15**	**-5**	**0**	**Average row**
0.day	3,3	3,7	1,8	4,0	3,3	**3,2**
1.day	*	71,6	7,9	21,7	22,6	**30,9**
2.day	*	34,6	4,5	17,7	16,9	**18,4**
3.day	*	17,0	*	11,8	*	**14,4**
6.day	*	4,9	1,9	6,4	5,6	**4,7**
Average column	**3,3**	**26,4**	**4,0**	**12,3**	**12,1**	

Appendix 4 CO_2 equivalents

CO_2 Equivalents calculated from study 1- temperature and study 2-WL

°C				
N2O	**CO_2 equivalents mg / m^{-2} /d^{-1}**			
	3	8	13	18
Peat	0	0	31	0
Drained	1147	5518	620	2449
Hay field	1054	4433	2635	5673
CH4				
	3	8	13	18
Peat	25,3	174,8	14899,4	41825,5
Drained	151,8	299	296,7	1274,2
Hay field	0	0	0	20,7
CO2				
	3	8	13	18
Peat	2464	1152	3621	4330
Drained	4540	3076	3607	8872
Hay field	6847	4311	5526	16694
Total CO_2 equivalents g / m^{-2} /d^{-1}				
	3	8	13	18
Peat	2,49	1,33	18,55	46,16
Drained	5,84	8,89	4,52	12,60
Hay field	7,90	8,74	8,16	22,39

CO_2 equivalents calculated from study 1 – temperature.

G	3	8	13	18
CH4	0,0000	0,0025	0,0008	0,0345
N2O	5,921	5,063	6,139	4,198
CO2	4,088	3,815	7,202	19,809
Total for G	10,01	8,88	13,34	24,04

CO_2 equivalents calculated from study 1 – temperature for the drained vegetated soil cores.

WL					
CO_2-equivalent	g /m^{-2}/day^{-1}				
N2O					
Drained	**-35**	**-25**	**-15**	**-5**	**0**
0.day	0,117	0,130	0,152	0,048	0,061
1.day	*	3,991	0,203	1,603	2,052
2.day	*	2,124	0,221	1,376	0,000
3.day	*	0,438	0,155	0,973	*
6.day	*	0,478	0,117	0,584	0,232
N2O					
Hayfield	**-35**	**-25**	**-15**	**-5**	**0**
0.day	0,133	0,401	0,382	0,281	0,208
1.day	*	11,486	1,576	4,291	3,639
2.day	*	6,744	0,955	2,961	3,047
3.day	*	3,330	0,279	1,652	*
6.day	*	0,474	0,300	1,137	0,620
g /m^{-2}/day^{-1}					
CO_2	**-35**	**-25**	**-15**	**-5**	**0**
Drained	2,64	1,51	1,61	3,07	1,93
Hayfield	2,16	2,67	2,95	1,45	0,22

Measurements of N_2O and CO_2 calculated in CO_2-equivalents for the drained and hayfield soil cores for study 2- water level fluctuation.

CO_2-equivalent	g /m^{-2}/day^{-1}				
WL	**-35**	**-25**	**-15**	**-5**	**0**
Drained	2,755	2,946	1,775	3,984	2,515
Hay field	2,29	7,154	3,645	3,516	2,100

Average emission from each water level.

CO_2-equivalent	g /m^{-2}/day^{-1}				
DAYS	0.day	1.day	2.day	3.day	6.day
Drained	0,102	1,962	0,931	0,522	0,353
Hay field	0,281	5,248	3,426	1,754	0,633

Average emission between days after drawdown.

Vegetated drained soil cores:

CO_2-equivalent g /m^{-2}/day^{-1}

N2O						
G		**-35**	**-25**	**-15**	**-5**	**0**
0.day		1,058	1,136	0,171	1,256	1,057
1.day	*		14,353	1,364	2,778	3,368
2.day	*		4,974	0,619	1,947	3,703
3.day	*		3,040 *		2,104 *	
6.day	*		1,017	0,347	0,847	1,377
G						
CO_2		8,63	6,91	8,97	8,78	7,29

CO_2-equivalent g /m^{-2}/day^{-1}

WL	**-35**	**-25**	**-15**	**-5**	**0**
Drained	2,755	2,946	1,775	3,984	2,515
G	9,690	11,814	9,599	10,563	9,662

CO_2-equivalent g /m^{-2}/day^{-1}

DAYS	0.day	1.day	2.day	3.day	6.day
Drained	0,102	1,962	0,931	0,522	0,353
G	0,935	5,466	2,811	2,572	0,897

Comparison of the emission from drained soil cores with and without vegetation