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Land-use and greenhouse gas balances of peatlands in the Nordic countries – present knowledge and gaps

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Abstract

This article provides an overview of the effects of land-use on the fluxes of methane (CH_4), nitrous oxide (N_2O) and carbon dioxide (CO_2) of peatlands in the Nordic countries presented in about 100 studies. In addition, the article identifies the gaps in the present knowledge on the greenhouse gas (GHG) balances associated with the land-use of these northern ecosystems. Northern peatlands have accumulated, as peat, a vast amount of carbon from the atmosphere since the last glaciation. However, past land-use and the present climate have evidently changed their GHG balance. The mean annual GHG balances of undisturbed ombrotrophic and minerotrophic peatlands were surprisingly positive (net sources) of 140 and 380 g CO_2 eq m^{-2}, respectively, even if the former was a sink of 63 g CO_2 eq m^{-2} when only the CO_2 balance was considered. Drainage of such peatlands for agriculture resulted in the most disadvantageous land-use option for the atmosphere, with the net annual GHG balance increasing to 2190, 2280 and 3140 g CO_2 eq m^{-2} for areas drained for grass swards, cereals or those left fallow, respectively. Even after ceasing of the cultivation practices, N_2O emissions remained high and together with the other GHGs resulted in net emissions of 1570 and 500 g CO_2 eq m^{-2}, in abandoned and afforested peatlands, respectively. Peat extraction sites were also net sources, 730 g CO_2 eq m^{-2}. The cultivation of reed canary grass turned the site to net sink of −330 g CO_2 eq m^{-2} but restoration did not (source of 470 g CO_2 eq m^{-2}). Data for afforested extraction sites is lacking. Peat soils originally drained for forestry may act as net sinks of 780 g CO_2 eq m^{-2}, and when those sites were restored the sink was 190 g CO_2 eq m^{-2}. However, more data is needed to confirm this point. Peat soils submerged under water reservoirs had a mean annual emission of 240 g CO_2 eq m^{-2}. In general, there is a lack of studies where all three GHGs have been measured at an ecosystem level, especially in the forested peatlands.

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1 Introduction

1.1 Carbon dioxide, methane and nitrous oxide as greenhouse gases

Carbon dioxide (CO_2) is a greenhouse gas which transmits visible light but absorbs strongly infrared and near-infrared light, thereby trapping heat in the troposphere and heating the soil surface (Solomon et al., 2007). The other important greenhouse gases which are emitted or produced by soil are nitrous oxide (N_2O) and methane (CH_4). N_2O is a more efficient greenhouse gas than CO_2. With a 100-year time horizon its global warming potential (GWP) is 298 times that of CO_2 (Solomon et al., 2007). N_2O is produced in soils mainly by microbial activities, nitrification and denitrification being the key processes (Priemé and Christensen, 2001). Methane (CH_4) is a greenhouse gas with 25 times the GWP of CO_2 in a 100-year time horizon (Solomon et al., 2007). CH_4 is formed in soil in anaerobic processes by methanogenic microbes (e.g. Le Mer and Roger, 2001). Concentrations of CO_2, CH_4 and N_2O in the atmosphere have increased in the atmosphere since pre-industrial times due to anthropogenic activities, agriculture being the most important one. Concentrations of CO_2, CH_4 and N_2O in the atmosphere were 37%, 156% and 19% above the pre-industrial levels in 2007 (WMO, 2008).

1.2 Peatlands and their use in Nordic countries

Peatlands are large carbon (C) reservoirs containing 6% of the global terrestrial C-stock as aboveground biomass and belowground organic matter (OM), mainly as peat (Schlesinger, 1997; Janzen, 2004). The OM accumulates in the anaerobic, water-logged soil and becomes peat because the decay rate of OM is slower than biomass accumulation. About 350 million ha of the peatlands are at high latitudes (Starck, 2008; Turunen et al., 2002). During the postglacial period the boreal and subarctic peatlands have accumulated about 455 Pg C (Gorham, 1991), which means that 1.670 Pg CO_2 have been removed from the atmosphere during this period. This can be compared to

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the total CO_2 stored in the atmosphere at any given time, which has been estimated to be ca. 750 Pg (Solomon, 2007).

There has been a major decrease in the area of pristine peatlands in all the Nordic countries. Peat soils have been drained for forestry, croplands, pastures and rangelands, peat mining, and for infrastructure such as roads and settlements (Table 1). The present fractions of undrained peatlands in the Nordic countries range from 9% (Denmark) to 80% (Norway) of the original peatland area (Table 1). Drainage has therefore affected the overall carbon and nitrogen cycles of peatlands differently in the countries under consideration. However, there are no accurate estimates for the coverage of all the land-use options, including e.g. the various after-use options of abandoned peat extraction sites and abandoned croplands (Table 1).

In Finland and Sweden, peatlands have been drained intensively for forestry, about 55% and 15% of the original peatland areas in Finland and Sweden, respectively (Paavilainen and Päivänen, 1995). In these countries, 5 to 15% of the drained peatlands have been used for agriculture (Myllys and Sinkkonen, 2004; Berglund and Berglund, 2008b). In Iceland, more than half of the peatlands have been drained, but less than 20% of those are used for cultivation (The Environment Agency of Iceland 2009). There, the majority of drained peatlands is either abandoned (as uncultivated) or used for livestock grazing. The origin of forests on peatlands in Sweden is poorly known. Probably, more than one third of the present peatland forests in Sweden have originally been agricultural land (von Arnold et al., 2005a). In Denmark, Iceland and Norway only a minor part of the peatlands are used for forestry, agriculture being the main use of drained peatlands (Table 1). In general, the area of former peatlands used for agriculture is difficult to identity based on the present soil properties, since with time the cultivated peat soils are turned to mull (OM content from 20 to 39.9%) or mineral soils (Myllys and Sinkkonen, 2004). For example, in Finland there are at present 850 km^2 of agricultural soils classified as peat soils but 2140 km^2 ha drained organic soils are classified as mull soils with OM content from 20 to 39% (Myllys and Sinkkonen, 2004). Also, the classification of peatlands differs between countries. In Sweden

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and Denmark, the organic soils are classified as shallow (<50 cm) and deep peat soils (>50 cm). In Sweden, all soils with OM content >20% are peat soils and also "gyttja" soils with OM content between 6 and 20% (Berglund and Berglund, 2008b), whereas in Finland only soils with >40% OM are classified as peat.

In Finland and Sweden, drainage of new areas for forestry and agriculture is no longer a common practice. In Iceland, some new areas are still being drained for agriculture (Guðmundsson, J. personal communication). In Finland, the maintenance of the existing drained forested sites requires cleaning of the ditches (Paavilainen and Päivänen, 1995). Forestry and agriculture are not the only anthropogenic activities changing the function of peatlands. Peatlands have been used as livestock pastures or rangelands, to construct roads and other infrastructures. Also, artificial lakes have been constructed on peatlands (Turunen, 2008; Berglund and Berglund, 2008a; Óskarsson and Guðmundsson, 2008a). Historically, peat extraction for energy and other purposes has been carried out in all Nordic countries. At present, only Finland and Sweden have large-scale peat extraction activities. In Finland and Sweden the area used for peat extraction is 0.6% and 0.1% of the total peatland area, respectively (Turunen, 2008; SCB, 2004, 2008). In these countries, large areas of the extraction sites are annually removed from the extraction, and new extraction areas are established on natural peatlands or on previously drained sites, e.g. on forestry drained peatlands. Until now, a small area of the abandoned peat extraction sites have been afforested in Finland but this land-use option is increasing (Paavilainen and Päivänen, 1995). In Iceland peat was extracted for cooking, house heating and as a construction material over many centuries. This usage stopped mostly in the early 20th century and no peat extraction has been carried out in Iceland since 1950 (NN, 1965). However, in recent years the excavation of peat due to the expansion of settlement has increased in Iceland. The excavated peat is often used on areas denuded by erosion or in landscape construction (The Environment Agency of Iceland, 2008).

The area of peat soils used for croplands is decreasing in Finland and Sweden. Some peat soils are annually abandoned and left out from crop production. Some of

the peat soil will turn to mull or mineral soils in the course of time (Myllys and Sinkkonen, 2004). In Sweden about 2500 km^2 of 6970 km^2 ha drained for agriculture are still used for agricultural purposes (Hjertestedt, 1946; Berglund and Berglund, 2008a, b). In Finland, more peat soils have been removed from agriculture with only 850 km^2 of the original 5000 km^2 presently under agricultural use (Myllys and Sinkkonen, 2004). This trend continues and arable peat soils are withdrawn from farming e.g. for forestry (Paavilainen and Päivänen, 1995). In Iceland, by contrast, cultivation of peatland is increasing in many areas due to enlargement of dairy farms and increased barley cultivation as the climate has become warmer in the past two decades (Björnsson et al., 2008). This increase takes place on both previously drained peatlands and new drainages. The large portion of the drained peatlands in Iceland with low or no utilization at all has lead to the inclusion of wetland restoration in climatic policy (Ministry for the Environment, 2007; UNFCCC, 2008).

Afforestation of abandoned croplands has been carried out in Finland, where more than 20 km^2 of drained peatlands are afforested annually (Hytönen, J., personal communication). The most common after use option after peat extraction for biomass production has also been afforestation, but the cultivation of reed canary grass (*Phalaris arundinacea*) for bioenergy is increasing in Finland (Shurpali et al., 2009). In Iceland, the total afforestation areas cover 300 km^2 and afforested peatlands cover 12% of that, 36 km^2 (The Environment Agency of Iceland, 2009).

Restoration of peatlands to their natural state by raising their water table level is also an after-use option. Restoration of drained croplands has been carried out in a small scale in Iceland, Norway and Denmark.

1.3 Greenhouse gas fluxes in natural and drained peatlands

Since the last glaciation period, natural peatlands have on average acted as a sink for atmospheric CO_2 (Turunen et al., 2002). However, during years with a dry growing season they can lose stored carbon as CO_2 (Alm et al., 1999a). Hydrological conditions have a great impact on the carbon balance of natural peatlands and the CO_2 balances

of various peatlands differ in their sensitivity towards climatic variability (Alm et al., 1997; 1999a; Sagerfors et al., 2008). In addition to CO_2, N_2O and CH_4 are the two other greenhouse gases which determine the atmospheric impact of peatlands. The dynamics of these two gases are also highly dependent on peatland hydrology.

Drainage of natural peat soils, e.g. for forestry, agriculture or peat extraction, greatly changes their GHG dynamics (e.g. Martikainen et al., 1993; Hargreaves et al., 2003). After drainage, the decomposition of organic matter increases and the sites may turn from a sink of CO_2 to net sources of CO_2. The GHG fluxes of drained peat soils depend on soil properties (Klemedtsson et al., 2005), ground water level (e.g. Óskarsson, 1998; Martikainen et al., 1993) and management practices, e.g. fertilization and ploughing (e.g. Regina et al., 1998; Maljanen et al., 2003a, b).

Water saturated peat soils emit C as methane (Saarnio et al., 2007). When the availability of oxygen in peat increases after drainage, there is a decrease in CH_4 emissions resulting from a decrease in CH_4 production and an increase in oxidation of CH_4 by methane oxidizing microbes (Saarnio et al., 2007; Le Mer and Roger, 2001). Drained peat soils are often only minor sources or sinks of CH_4 (Maljanen et al., 2007a; Mäkiranta et al., 2007).

Natural peatlands generally have negligible emissions of N_2O and they can even act as a sink for this gas (e.g. Regina et al., 1996; Martikainen et al., 1993). However, some nutrient rich natural peatlands, i.e. herb-rich spruce mires and alder forests emit N_2O, (Huttunen et al., 2003; von Arnold et al., 2005c). A recent study by Repo et al. (2009) reveals that arctic peatlands have bare surfaces that also emit N_2O at high rates. These high-emitting bare surfaces are created by cryogenic processes which do not exist in boreal peatlands, but these surfaces may exist in northern Scandinavia and in Iceland.

After drainage of a peatland, the availability of oxygen and mineral nitrogen increases, which favours N_2O production (e.g. Martikainen et al., 1993). Drained organic soils with high N_2O emissions have great importance in the atmospheric N_2O load. For example, organic soils drained for agriculture are responsible for 25% of the anthro-

pogenic N_2O emissions in Finland (Kasimir Klemedtsson et al., 1997), even though they cover less than 10% of the total arable land area.

1.4 GHG emissions from water reservoirs

Reservoirs (i.e., man-made lakes) have been constructed to store water for different purposes, such as electrical power generation, water supply and flood control. In northern regions, such as in Finland, this has also meant flooding of natural peatland ecosystems (e.g. Turunen, 2008). The reservoirs have been estimated to represent a significant world-wide source of atmospheric CO_2 and CH_4 (St. Louis et al., 2000). Data on the exchange of N_2O, the third most important GHG in reservoirs, are sparse. However, based on the present results, the importance of N_2O in the total GHG emissions from reservoirs should be small (Huttunen et al., 2002b; Huttunen and Martikainen, 2005a; Óskarsson and Gudmundsson, 2008a, b).

CO_2 and N_2O are released from reservoirs to the atmosphere mainly by diffusion (and by turbulence) at the water-air interface, while CH_4 can be transported by diffusion, ebullition (bubbling) and/or through aerenchymatous tissues of aquatic macrophytes. In northern, seasonally ice-covered reservoirs, GHGs produced during wintertime accumulate in the water column under the ice. It has been suggested that these gas stores are largely released to the atmosphere during spring overturn after the ice melt (Huttunen et al., 2002; Duchemin et al., 2006). Dissolved GHGs can also exit reservoirs in outflowing water, causing possible "degassing emissions" downstream of the dam, within turbine/spillway constructions and/or in lower river reaches (Roehm and Tremblay, 2006; Guerin et al., 2006).

At present, most of the data on reservoir GHG emissions are from tropical countries such as Brazil and French Guiana, and from the boreal zone in Canada (e.g., Soumis et al., 2004). In the Nordic countries, there are published data on the reservoir CO_2, CH_4 and N_2O fluxes from Finland, CO_2 fluxes from Sweden (Bergström et al., 2004; Åberg et al., 2004) and on CH_4 and N_2O (Óskarsson and Gudmundsson, 2008a) and CO_2 (Óskarsson and Guðmundsson, 2008b) fluxes from Iceland. Some preliminary

results have been presented for GHG exchange in small reservoirs in Norway (Harby et al., 2006).

Measured fluxes of GHG from reservoirs can generally not be interpreted as connected to specific type of land impounded e.g. peatland, since dissolved gasses from different submerged land types are mixed into the water body.

1.5 Aims

There is an urgent need to develop mitigation strategies to reduce GHG emissions. This was highlighted by the ratification of the Kyoto Protocol in 2004. At present, afforestation, forest management, cropland management and grazing land management are valid mitigation options for drained peatlands according to the Kyoto protocol, and it is being discussed whether wetland restoration and bioenergy production could also be accepted as valid mitigation options in the next commitment period in 2013–2017.

There are large uncertainties regarding the true potential of different land-use options to mitigate GHG emissions. In this review we summarize the present knowledge in the Nordic countries on how the various land-use options affect the GHG fluxes of peatlands. We used mostly published experimental data, but some unpublished data is also included. The most serious gaps in present knowledge regarding the GHG fluxes of managed peatlands were identified. The land use categories discussed in this review were 1) unmanaged peatlands, 2) peatlands drained for forestry, 3) peatlands drained for agriculture, 4) peat extraction sites and 5) flooding of peatlands for water reservoirs. The various land-use options (e.g. rewetting, afforestation, energy crops) for peat soils withdrawn from current land-use were compared. The data used was mainly from studies in Denmark, Finland, Iceland and Sweden, from which countries research groups participated in a large Nordic project entitled: *The Nordic Centre for Studies of Ecosystem Carbon Exchange and its Interactions with the Climate System* (NECC). NECC was funded between 2003–2008 by the Nordic Centre of Excellence (NCoE) Pilot Programme.

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2 Data sources from various managed peatlands

2.1 Measurement techniques for terrestrial CH_4, CO_2 and N_2O fluxes

Fluxes of CH_4, CO_2 and N_2O are measured using various chamber techniques (e.g. Nykänen et al., 1995); micrometeorological methods (e.g. Lohila et al., 2004) or the fluxes can be calculated from the gas concentration gradients in snow or soil using the diffusion approach (Alm et al. 1999b; Maljanen et al., 2003c). Static chamber techniques determine the gas fluxes from the change in gas concentration in the headspace of the chamber with time. Dynamic chamber techniques determine the flux rate based on the equilibrium in the gas concentration in the chamber. The gas concentrations are measured e.g. by gas chromatograph, infrared analyzer or photo-acoustic analyzer, recently also by laser analyzers. The most commonly used micrometeorological method is the eddy correlation (EC) method, where the gas flux is determined from the high-frequency gas concentration, wind and scalar atmospheric data series. Gas concentrations are measured with a high-frequency infrared analyzer or tunable diode laser absorption spectrometer. This method provides continuous, spatially integrated fluxes from large areas (e.g. Baldocci et al., 1988; Lohila et al., 2003, 2007a, b). It can be used to measure the net ecosystem exchange of CO_2 (e.g. over forest canopy) over long periods. The EC method has also been applied for the CH_4 and N_2O fluxes (Clement et al., 1995; Pihlatie et al., 2005).

2.2 Annual greenhouse gas fluxes of soil ecosystems

A total of about 100 published articles or manuscripts reporting GHG fluxes from different land-use categories of peatlands are summarized for the Nordic countries. The available annual data on the fluxes is listed in Tables 2, 3, 4 and 5. In this summary, the minus sign indicates uptake of gas and the plus sign shows loss of gas from the ecosystem. If a site has flux data for several years, the average flux of these years is shown. If the flux data only covers the growing season, then the annual fluxes

were calculated based on the published general gas dynamics in various seasons. For methane, the winter (=outside growing season) fluxes are estimated to be 15% of the growing season fluxes (Saarnio et al. 2007). Winter time respiration is assumed to be 15% of the growing season respiration or 210 g CO_2 yr (if respiration data is not available) for drained sites (Lohila et al., 2007; Shurpali et al., 2008) and 110 g CO_2 yr^{-1} for the undrained sites (Saarnio et al., 2007). The winter time N_2O emissions from drained peat soils may account for 1 to 99% of the annual ones, on average 40% (Maljanen et al., 2004, 2007a). Therefore, for the drained sites nitrous oxide emissions outside the growing season were estimated to be 50% of the growing season emissions, which is likely to be a conservative estimate. If emissions are reported for growing seasons longer than six months, the fluxes are normalized to six months and winter is assumed to cover the rest of the year. This approach has to be taken to complete the data for the winter if this data was lacking. The data filling method applied certainly caused some inaccuracies in the estimates.

2.3 Greenhouse gas emissions from aquatic systems

The measurement methods for GHG fluxes in aquatic systems differ only slightly from those used in terrestrial sites (see above). Both the eddy covariance (EC) and chamber methods are used in the reservoir GHG flux studies. CH_4 ebullition (bubbling) should be detected parallel to diffusive (turbulent) CH_4 exchange. The diffusive/turbulent gas transfer rates can be calculated from the surface water GHG concentration gradient and local wind speed using the so-called "thin boundary layer method". The springtime GHG emissions, caused by the mixing of the water column after the spring ice melt, can be approximated from excess GHG storages accumulated under the ice during the ice-covered season, since reaching the sites at the times of ice melt is difficult (e.g. Huttunen et al., 2002b).

3 Synthesis of the data for the annual greenhouse gas balances in the Nordic countries

3.1 Unmanaged peatlands

Greenhouse gas fluxes of natural peatlands have been studied intensively in the Nordic countries. These data are more complete than that for managed peatlands. However, there are no GHG data on undrained but grazed peatlands in Iceland, which is the most common land use there. There is a recent review by Saarnio et al. (2007) on the CO_2 and CH_4 fluxes of the natural boreal peatlands, and the data of natural peatlands divided to ombrotrophic (nutrient poor) and minerotrophic (nutrient rich) ones is used here. The N_2O fluxes of natural peat soils (together 24 sites) are taken from studies by Martikainen et al. (1993), Nykänen et al. (1995), Regina et al. (1996), Huttunen et al. (2002a) and von Arnold et al. (2005b). This data and additional data by Saarnio et al. (2007) are summarized in Fig. 1. The spruce swamps (Huttunen et al., 2002a) and alder sites (von Arnold et al., 2005c) form a special group among the natural peatlands. These are excluded when determining the mean N_2O fluxes of natural peatlands with a high water table. Some recent CO_2 data (Aurela et al., 2007; Bäckstrand et al., 2009; Sagerfors et al. 2008; Lund et al., 2007; Nilsson et al., 2008; Soini et al., 2009) from natural peatlands, not included in the review by Saarnio et al. (2007), are also used in the calculations.

Carbon gas fluxes (CH_4 and CO_2) of natural peatlands vary greatly depending on weather conditions and the geographical location of the site. According to the review by Saarnio et al. (2007), the average net CO_2 flux for the Nordic ombrotrophic peatlands is $55\pm190\,g\,CO_2\,m^{-2}\,yr^{-1}$ and that for the minerotrophic peatlands is $-55\pm230\,g\,CO_2\,m^{-2}\,yr^{-1}$. Thus, according to the existing data the ombrotrophic peatlands, in contrast to the minerotrophic ones, are net sources of CO_2 to the atmosphere. Mean methane emissions are 6.7 ± 5.3 and $17.3\pm13.3\,g\,CH_4\,m^{-2}\,yr^{-1}$ from ombrotrophic and minerotrophic peatlands, respectively (Saarnio et al., 2007). The

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total carbon gas fluxes (CO_2+CH_4) of the ombrotrophic peatlands indicate a lower C accumulation rate for the ombrotrophic peatlands than for the minerotrophic peatlands. Some recent results have shown even higher CO_2 uptake rates for boreal minerotrophic mires than the data reviewed by Saarnio et al. (2007). This differs from the long term carbon accumulation rate (Turunen et al., 2002), showing higher C accumulation in the ombrotrophic than in the minerotrophic peatlands. This contradiction may be a result of spatial and temporal variation in the gas fluxes not covered completely by the flux data available.

The average net CO_2 uptake of Swedish nutrient poor peatland sites has been found to range between $-78\,g\,CO_2\,m^{-2}\,yr^{-1}$ (Lund et al., 2007) and $202\,g\,m^{-2}\,yr^{-1}$ (Sagerfors et al., 2008). In a Swedish minerotrophic site, the mean CO_2 uptake was $-189\,g\,m^{-2}\,yr^{-1}$ (Nilsson et al., 2008). Bäckstrand et al. (2009) reported an annual CO_2 uptake of $-9.5\,g\,m^{-2}$ for a subarctic mire in northern Sweden. Haapala et al. (2009) measured a seasonal CO_2 exchange of -158 to $-136\,g\,m^{-2}$ from April to December for a flark surface of a fen in northern Finland and Soini et al. (2009) reported seasonal (May to September) CO_2 exchange of $-390\,g\,m^{-2}$ for a fen in southern Finland. For a minerotrophic peatland in northern Finland the annual net fluxes were from -188 to $-219\,g\,CO_2\,m^{-2}$ (Aurela et al., 2007). If these results are also taken into account, the average annual net CO_2 exchange rate for the minerotrophic peatlands in Finland and Sweden is $-63\,g\,CO_2\,m^{-2}\,yr^{-1}$ and that for the ombrotrophic peatlands $9\,g\,CO_2\,m^{-2}\,yr^{-1}$. This is again in contrast to the long term C accumulation rates for ombrotrophic and minerotrophic peatlands (Turunen et al., 2002).

The water table level affects N_2O emissions from peatlands (e.g. Martikainen et al., 1993). Emissions can be high with a low water table level but water saturated peat can even consume atmospheric N_2O in the absence of O_2 (Huttunen et al., 2002a; Martikainen et al., 1993). The mean N_2O emissions from undrained peatlands are $<0.001\,g\,N_2O\,m^{-2}$ (n=7) for the ombrotrophic sites and $0.018 \pm 0.019\,g\,N_2O\,m^{-2}$ (n=17) for the minerotrophic sites (Regina et al., 1996; Martikainen et al., 1993; Huttunen et al., 2002a; Nykänen et al., 1995; von Arnold et al., 2005b). Exceptions are

the natural herb-grass spruce mires (Huttunen et al., 2002a) and alder forests (von Arnold et al., 2005b) with a lower ground water table (WT) level. The annual N_2O emissions were 0.25 and 0.10 g m^{-2}, respectively. Especially in the spruce mires, the N_2O emissions can be similar or even higher than those from peatlands drained for forestry (see later). It has to be noted that these natural sites with high emission rates are not included in the estimate for the mean N_2O emission from natural minerotrophic sites (Fig. 1). When using the GWP approach with 100 years reference period (CO_2eq) to calculate the atmospheric effect of the three greenhouse gases in the unmanaged peatlands, mainly CH_4 causes the positive GWP for the minerotrophic peatlands and CO_2 with CH_4 for the ombrotrophic peatlands (Fig. 1).

3.2 Drained peat soils

3.2.1 Peatlands drained for forestry

Drainage of peatlands for forestry usually differs from drainage for agriculture in the Nordic countries; the ditches are open and draining intensity is partly achieved by excavating secondary ditches connected to the main ditches (Paavilainen and Päivänen, 1995). For the forestry drained peatlands there are GHG flux data in Finland and Sweden, where large peatland areas have been drained to improve forest productivity (Table 2). In addition, one study has been carried out in Denmark and none in Norway and Iceland. Results from 65 study sites are included in Table 2, where the forestry drained peatlands are divided into ombrotrophic (nutrient poor) and minerotrophic (nutrient rich) sites.

Fluxes of N_2O and CH_4 have been measured with chamber methods from the forest floor. These fluxes have been intensively studied in Finland and Sweden. However, few data are available from the same sites for the net ecosystem CO_2 exchange, which requires the EC technique.

The depth of WT after drainage largely determines the CH_4 flux rates. In general, the drainage changes the CH_4 fluxes less in the ombrotrophic than in the minerotrophic

peatlands because drainage lowers WT less in the ombrotrophic sites (Nykänen et al., 1998). Drainage decreases the CH_4 emissions on average by 50%, whereas many of the high-emitting minerotrophic peratlands show CH_4 uptake after drainage (Nykänen et al., 1998). Annual mean CH_4 emissions from ombrotrophic and minerotrophic peatlands drained for forestry are $0.90\pm1.35\,g\,m^{-2}$ (n=9) and $0.57\pm1.28\,g\,m^{-2}$ (n=31), respectively.

The N_2O fluxes in ombrotrophic peatlands are less affected by drainage than those in minerotrophic peatlands. In addition to the WT, the N_2O emissions are highly regulated by the C/N ratio of peat. The high emissions require a low C/N ratio, between 15 to 25, in peat (Klemedtsson et al., 2005; Mäkiranta et al., 2007). A low C/N ratio evidently favors net nitrogen mineralization and associated nitrification serving nitrate for denitrification. The annual mean N_2O emission from forestry drained ombrotrophic peatlands is lower, $0.01\pm0.01\,g\,m^{-2}$ (n=11), than that in minerotrophic sites $0.43\pm0.77\,g\,m^{-2}$ (n=33). There are only few measured winter data from boreal peatland forests. According to recent results these emissions can be high, up to $3\,g\,m^{-2}$, and they may even account for 80% of annual emissions, resulting in underestimation of annual values for forested peatlands (Maljanen et al., this issue + unpubl. data).

Drainage of peatlands for forestry changes the microbial processes in soil and the associated greenhouse gas fluxes. However, the ditches can have different gas dynamics than the strips. Even though the strips show CH_4 uptake, the ditches can still emit CH_4 as shown in a Finnish study (Minkkinen et al., 1997). Minkkinen et al. (1997) estimated that the CH_4 emission from ditches was 4.5% of the total CH_4 emissions from peatland drained for forestry during summer. Ditches would also affect the N_2O release from drained peatlands. GHG emissions associated with ditches include not only the direct emissions but also transport in the ditch water. There is no data on the importance of the later aspect.

The net annual CO_2 exchange measured with the EC method has been reported in Finland only for one minerotrophic site with Scots pine (*Pinus sylvestris*) stand. This site has an annual uptake of $900\,g\,CO_2\,m^{-2}$ (Laurila et al., 2007). Klemedtsson et al.

(2007) modeled a similar CO_2 annual uptake rate, from 540 to 1100 g CO_2 m^{-2} for a forested peatland in Sweden. Pihlatie et al. (2009) measured CO_2 fluxes above a canopy in a drained peatland from April to June and the growing season uptake was 800 g CO_2 m^{-2}. The annual CO_2 uptake in a forested peatlands in Scotland has been higher than that in the Finnish and Swedish studies, as high as 1830 g CO_2 m^{-2} (Hargreaves et al., 2003).

The net CO_2 ecosystem exchange (NEE) in forestry drained peatlands a highly dependent on the age of the tree stand as well as the weather conditions. The site fertility probably has an impact on CO_2 balance, but the existing data based on the EC measurements are too limited to conclude the importance of fertility in the CO_2 balance. An important open question is the fate of soil carbon in forested peatlands. There is evidence that the development of soil carbon in peatland forests varies in peatlands with different fertility (e.g. Minkkinen and Laine, 1998). Hargreaves et al. (2003) reported that in Scotland during the first years after draining, peat soils were sources of CO_2 but after 4 to 8 years from planting of trees the sites turned to sinks for CO_2.

To estimate the impact of forestry drainage on the GHG balances the whole stand rotation has to be included (Minkkinen et al., 2001). There is little information how on the carbon balance or N_2O fluxes are affected by clear cutting and what is the balace during the second rotation period. Clear cutting changes e.g. hydrology and thus also the oxygen status of soil which would affect the decomposition rate of organic matter and emissions of CO_2, CH_4 and N_2O. The effect of clear cutting has been studied in two forested peatland sites in Finland. In the study by Saari et al. (2009a), peatland forest turned from a small N_2O sink to a N_2O source after clear cutting. In the study by Huttunen et al. (2003) N_2O emissions increased and CH_4 uptake rates decreased following clear cutting (Huttunen et al., 2003). No data exists on the net CO_2 exchange after clear cutting and how soil preparation for the second rotation would change the CO_2 dynamics.

Buffer zones are used in peatland forestry to remove P and N from the discharge. The sedimentation ponds in these areas with high N load can be potential hot spots for

N_2O emissions, up to 82 $\mu g\, m^{-2}\, h^{-1}$ (Saari et al., 2009b). However, the areas covered by buffer zones are rather small and therefore they are not significant GHG sources (Saari et al. 2009b).

The mean greenhouse gas flux rates of forested Nordic peatlands are compared with those measured in upland forests in Table 6. Data for net CO_2 exchange is available from 16 upland forests; most of them are sinks for CO_2. However, some have been sources of CO_2 (Lindroth et al., 1998, 2007). One of these mineral soil sites with exceptionally high net CO_2 emissions also had a high organic C content in the soil profile (Lindroth et al., 1998). Mineral soil with fast growing plants can also act as a significant C sink, as with willow plantation in the study by Grelle et al. (2007) (not included in Table 6). In general, peatland forest soils emit more N_2O than upland forest soils (Table 6). Peatland forest soils can be either source or sink for CH_4 depending on the drainage status, but upland forest soils most often are sinks for atmospheric CH_4 (Table 6).

3.2.2 Peatlands drained for agriculture

Peat soils with annual or perennial crops

GHG dynamics of peat soils used for agriculture are intensively studied in Finland, Sweden, Iceland and Norway. There is data on soils growing perennial grasses (15 sites), barley (six sites) and for one site with potato and one with carrot. No published data are available on GHG emissions directly from the drainage ditches of croplands in the Nordic countries.

CH_4 and N_2O fluxes are measured with chamber methods, CO_2 balance with chambers at 10 sites and with EC methods at four sites (Table 3). The croplands have been divided into grass swards and croplands with cereals or potato. The grasslands studied in Finland have been non-permanent grassland which is ploughed and re-sown with three to four year intervals. In addition to the actively cultivated peatlands, fallow soils without vegetation (Maljanen et al., 2003a, b, 2004; Regina et al., 2004), abandoned

croplands in Finland (Maljanen et al., 2007a) and drained but not ploughed sites used for grazing in Iceland (Table 3) have been also studied.

The GHG emissions from peatlands used for agriculture can differ in magnitude resulting from differences in cultivation methods, crops and weather conditions. Peatlands used for croplands are significant sources of CO_2 and N_2O. Depending on the WT level, croplands on peat soils are small sinks or sources of CH_4. The average emissions with standard deviation from peat soils with perennial grass swards is $0.21\pm0.53\,g\,CH_4\,m^{-2}$ (n=10), $1.28\pm1.56\,g\,N_2O\,m^{-2}$ (n=14) and $1800\pm1180\,g\,CO_2\,m^{-2}$ (n=4).

Regularly ploughed and fertilized cropland soils (cereals) have been larger sources of CO_2 and N_2O than grass leys but because of their better aeration they are larger sinks for atmospheric CH_4 than the grass swards. Peat soils planted with barley emit $-0.03\pm0.18\,g\,CH_4\,m^{-2}$ (n=5), $1.71\pm0.51\,g\,N_2O\,m^{-2}$ (n=5) and $1770\pm1160\,g\,CO_2\,m^{-2}$ (n=3). Annual N_2O emission is also available for a potato field, $1.57\,g\,m^{-2}$ (Regina et al., 2004).

Peat fields have always shown a net loss of CO_2 in contrast to cropland on mineral soils (Table 6). However, Tagesson and Lindroth (2007) reported a high annual CO_2 loss, from 1390 to $2410\,g\,CO_2\,m^{-2}$, from a meadow on mineral soil, but the soil had a thick (28 cm) humus layer, therefore this site was excluded from the summary Table 6. Peat soils drained for agriculture also have higher N_2O emissions than mineral soils (Table 6). Some peat fields emit CH_4 depending on the WT level, whereas the croplands on mineral soils have all been sinks for atmospheric CH_4 (Table 6). In agricultural peat soils, CO_2 is responsible for 80% of the total GWP and the rest is mostly from the N_2O emissions. Methane makes a negligible contribution to the total GWP on these well drained soils.

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Fallow soils (agricultural soils without vegetation)

Nitrous oxide emissions from fallow soils without vegetation have been studied in five sites, CH_4 fluxes in three sites and CO_2 balance in three sites in Finland. These sites have been kept clear from vegetation during one season with regular ploughing (Maljanen et al., 2001a, 2003a, b, 2004, Nykänen et al., 1995; Regina et al., 2004). In the absence of plants, some of these sites emit more CO_2 and N_2O than vegetated agricultural soils. The mean N_2O and CO_2 annual emissions from fallow sites, $2.13\pm1.62\,g\,N_2O\,m^{-2}$ and $2507\pm968\,g\,CO_2\,m^{-2}$, were slightly higher than those from soils with grass or cereals. These well drained fallow sites have been annually small sinks for CH_4 ($-0.10\pm0.14\,g\,CH_4\,m^{-2}$).

Abandoned croplands

Annual balances of CH_4, N_2O and CO_2 on abandoned croplands have been reported for five sites in Finland (Maljanen et al., 2007a). We could expect that when soil management activities for cultivation cease, the emissions of N_2O and CO_2 would decrease. However, the annual N_2O and CO_2 emissions from abandoned croplands have been close to the emissions from actively managed croplands. The mean emissions from abandoned sites have been $0.865\pm0.780\,g\,m^{-2}$ (n=5) for N_2O and $1320\pm1120\,g\,m^{-2}$ (n=5) for CO_2. The mean methane uptake rate, $0.219\pm0.180\,g\,CH_4\,m^{-2}$ (n=5) is, however, higher than that for the regularly ploughed croplands. Some GHG measurements have also been performed on drained but not ploughed abandoned sites in Iceland (Jón Guðmundsson, personal communication). Because these sites are drained but never ploughed, they have a different history compared to the abandoned cropland sites in Finland. The estimated average annual N_2O emission from the abandoned sites in Iceland (n=3) is $0.032\pm0.020\,g\,N_2O\,m^{-2}$ is lower than that from the formerly ploughed and fertilized soils in Finland.

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Afforested croplands

Methane and N_2O balances of afforested agricultural soils have been studied intensively (Table 3). Annual CH_4 and N_2O fluxes measured for the forest floor of afforested croplands with static chambers have been reported for 10 sites in Finland (Mäkiranta et al., 2007; Maljanen et al., 2001b) and two sites in Sweden (von Arnold et al., 2005c; Weslien et al., 2009). However, the annual net CO_2 exchange (EC method), including the canopy, has been reported only for one site in Finland (Lohila et al., 2007).

Afforestation of organic croplands has been expected to reduce the GHG emissions. However, in most cases this is not true for the N_2O emissions. The mean emission, $1.48 \pm 1.31\,g\,N_2O\,m^{-2}$, is similar or even higher than that from cultivated croplands on peatlands, from 1.2 to $1.7\,g\,N_2O\,m^{-2}$. Depending on the WT level, the afforested sites act as either small sources or sinks for CH_4. The mean CH_4 emission is $0.23 \pm 0.69\,g\,m^{-2}$. The data on the net CO_2 exchange after afforestation of agricultural peatlands is limited to one site in Finland (Lohila et al., 2007). This site had an annual loss of $50\,g\,CO_2\,m^{-2}$ which is lower than the CO_2 loss from cultivated croplands. Afforestation of organic croplands would thus reduce CO_2 emissions, but N_2O emissions may remain high. However, drained peat soils can act annually either as sinks or sources of CO_2, since CO_2 balance depends strongly on the climatic conditions including temperature and precipitation (e.g. Shurpali et al., 2009).

3.2.3 Peatlands drained for peat extraction

Peatlands under active peat extraction

Peat extraction for energy purposes is carried out in Finland and Sweden. Both pristine and forestry-drained peatlands have been taken to peat extraction. Traditionally, the peat extraction sites are drained and the vegetation cover removed before the extraction. The draining phase lasts from one to five years before extraction. Peat extraction on these open areas is carried out for several years depending on the peat depth.

A new method called "biomass dryer" has recently been developed for peat extraction. There peat is extracted without any drainage phase of the site and the extracted moist peat is dried outside the extraction site on an asphalted field (VAPO, 2009). The GHG fluxes associated with this method are at present under study.

GHG emissions have been measured with chamber methods from the "traditional" peat extraction in Finland and Sweden (Table 4). Drained peat extraction sites are always sources of CO_2 as a result of WT draw down and removal of vegetation. The sites are also sources for N_2O and CH_4. The annual emissions have been measured in five studies (Alm et al., 2007; Shurpali et al., 2008; Hyvönen et al., 2009; Nykänen et al., 1996; Tuittila et al., 2000). The annual emissions for the other sites were calculated from the growing season values as described earlier. The mean annual emissions from peat extraction sites, excluding the emissions from ditches, are $1.39 \pm 2.23\,g\,CH_4\,m^{-2}$ (n=16), $0.092 \pm 0.10\,g\,N_2O\,m^{-2}$ (n=7) and $664 \pm 251\,g\,m^{-2}$ for CO_2 (n=16).

The CO_2 emissions from peat extraction areas depend e.g. on the quality of the peat and the time since drainage, as well as the climate (Nykänen et al., 1996; Waddington et al., 2002). Waddington et al. (2002) reported higher seasonal (from May to August) CO_2 emissions for a dry year (1300 to 1500 g CO_2) than for a wet year (320 to 430 g CO_2) from peat extraction sites in Canada, Québeck. Emissions were slightly higher from the sites seven to eight years after peat extraction than from the sites from two to three years after peat extraction (Waddington et al., 2002).

The ditches contribute to the CH_4 emissions from peat extraction sites. Methane emissions from drainage ditches have been measured in Sweden (Sundh et al., 2000) and Finland (Nykänen et al., 1996). The annual emissions from the ditches are from 0.3 to $140\,g\,CH_4\,m^{-2}\,yr^{-1}$ (Nykänen et al., 1996; Sundh et al., 2000), thus CH_4 emission from the ditch network has importance in the total emissions from peat extraction sites. However, the importance of ditches in the CO_2 emissions is negligible (Sundh et al., 2000; Waddington and Day, 2007; Nykänen et al., 1996). The annual mean emissions from peat extraction sites shown in Fig. 1 do not include the emissions from ditches, since in most studies these emissions are not measured.

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The wetlands constructed to purify runoff waters of peat extraction areas can also be sources of greenhouse gases (Liikanen et al., 2006). The mean emission from these wetlands can change with time, associated with the development of vegetation. In the study by Liikanen et al. (2006), CH_4 emissions increased from 140 to 400 mg CH_4 m^{-2} d^{-1} and the ecosystem respiration increased from 7270 to 13 600 mg CH_4 m^{-2} d^{-1} from 5 to 15 years in operation, whereas N_2O emissions remained similar, from 340 to 450 µg N_2O m^{-2} d^{-1} (Liikanen et al. 2006).

When the atmospheric impact of peat extraction is considered, the total gas balances including the site specific ones, the logistical ones and the use of peat e.g. for energy production have to be taken into account. Life cycle analysis of peat as fuel has been carried out in Sweden and Finland (Kirkinen et al., 2007; Nilsson and Nilsson, 2004). These studies show that use of pristine peatlands for energy production has a more negative atmospheric impact than if existing drained sites are utilized. In addition, a life cycle analysis of nonfuel peat extraction has been performed in Canada (Cleary et al., 2005). However, the lack of proper data on emissions associated with the various phases of peat utilization causes some bias in the results.

Abandoned peat extraction sites

Annually, about 20 km^2 of extraction sites are abandoned both in Finland and Sweden. Therefore, the after use of peat extraction sites also has importance in the national GHG budgets. Some GHG measurements have been performed on abandoned peat extraction sites with no active after use (Tuittila et al., 2000; Tuittila and Komulainen, 1995; Tuittila et al., 2004; Nykänen et al., 1996). Abandoned sites emitted 740±92 g CO_2 m^{-2} (n=3), 0.25±0.01 g CH_4 m^{-2} (n=2) and 0.08 g N_2O m^{-2} (n=1) (Table 4).

Afforestation and restoration are the most common after-use options. Other options include e.g. energy crop cultivation. Results for the greenhouse gas balances of an abandoned peat extraction site growing perennial grass (reed canary grass) for energy

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are available in Finland (Table 4). In Finland, afforestation has been the most common after-use option for abandoned peat extraction sites. However, the GHG emission data from various after-use options is sparse. Methane and N_2O results are reported for five afforested subsites from a former peat extraction area (Mäkiranta et al., 2007), but the data on the CO_2 balance is lacking (Table 4).

Cultivation of bioenergy crops on abandoned peat extraction sites

GHG balances of a cut-away peatland used for cultivation of a bioenergy crop (reed canary grass, *Phalaris arundinacea*) have been studied in Finland over four years. In the cultivation of perennial reed canary grass soil is not ploughed after the first year since vegetation is established. During the rotation period of 10–15 years there is no ploughing but fertilizers are applied annually and sometimes lime. Therefore, there are fewer soil disturbances than in the production of annual crops, which would reduce the GHG emissions. The first results from Finland show a mean annual CO_2 uptake of 365 g m^{-2}, mean annual CH_4 emission of 0.38 g m^{-2} and N_2O emissions of 0.09 g m^{-2} (Hyvönen et al., 2009; Shurpali et al., 2009). According to these results, the site has been a net sink for C, there is evidence that some carbon is allocated into the soil (Shurpali et al., 2009; C. Biasi, personal communication). The N_2O emissions were much lower than those from croplands in general. However, it has to be pointed out that the data is available only for one site and the sites with variable peat characteristics have not been studied.

Afforestation of peat extraction sites

Annual CH_4 and N_2O balances have been studied for five sites within one peat extraction area in southern Finland (Mäkiranta et al., 2007), but there are no data on net CO_2 exchange from any site. As a result of low WT level, all the sites were small sinks for CH_4 and sources of N_2O. The mean annual CH_4 uptake rate was 0.053±0.02 g m^{-2} and the mean annual N_2O emission was 0.38±0.31 g m^{-2}. Because there was no data

on the net CO_2 exchange, the GWP including all gases are not shown for afforested peat extraction sites in Fig. 1.

Restored drained petlands

Interest in restoring formerly drained peatlands is increasing in the Nordic countries (e.g. UNFCCC, 2008; Petersen et al., 2009). In Denmark, for example, there are plans to restore nutrient rich fens drained for croplands. In Iceland, there are large areas of unproductive drained peatlands and the government has decided to include restoration of these wetlands as an action to decrease the national GHG emissions (Ministry for the Environment, 2007). At present, published data on the GHG emissions from restored peatlands in the Nordic countries are available only in Finland, where restoration of peatlands drained for forestry has been carried out in a small scale. Additionally, unpublished data exists from a case study in Iceland that compared GHG emissions from intact, drained and restored peatlands (Óskarsson, unpublished).

When peatlands are re-wetted, the WT level is elevated close to the soil surface in order to recreate a water logged situation, slow down the decomposition of OM and gain carbon to the system as peat. Increasing nutrient uptake by vegetation (e.g. *Eriophorium vaginatum*) also decreases N_2O emissions (Silvan et al., 2005). A drawback, however, is the increase in CH_4 emissions resulting from the anaerobic conditions in peat after elevation of the WT. Methane emissions have been reported from two restored originally forestry drained sites and from five peat extraction sites. N_2O emissions have been reported from one restored forestry drained site and from two restored peat extraction sites. Net CO_2 exchange is reported for two restored forest sites and five restored peat extraction sites (Tables 2, 4).

The restored forest sites emitted methane with an average rate of $3.35 \pm 1.77\,g\,CH_4\,m^{-2}$. The methane emissions are highly dependent on the age of restoration. Therefore, emissions change with time after restoration (Waddington and Day, 2007; Höper et al., 2008). N_2O emissions were only studied at one site. There, the annual N_2O emission without fertilization was $0.55\,g\,N_2O\,m^{-2}$ and with

nitrate (45 kg NO_3^--N ha^{-1} yr^{-1}) fertilization 2.0 g N_2O m^{-2} (Silvan et al., 2002). These emissions were high compared to those from undrained peatlands having only minor N_2O emissions (see Sect. 3.1). The net CO_2 exchange during the growing season from the restored forestry drained ombrotrophic site was lower (−275 g CO_2 m^{-1}) than that from the minerotrophic site (−816 g CO_2 m^{-1}). The mean annual net CO_2 exchange of a restored originally forestry drained site including winter respiration (see Sect. 2.2) was −440±377 g CO_2 m^{-1}, indicating that these ecosystems act as carbon sinks after restoration.

The CO_2 balance of restored peat extraction sites in Finland has been reported in five publications. Ylipetäys et al. (2007) described annual emissions from an abandoned peat extraction site with natural develoment of peatland vegetation. The other studies have been carried out within the same restored site where re-vegetation was actively supported. However, in this case measurements were only taken during the growing season (Soini et al., 2009; Kivimäki et al., 2008; Tuittila et al., 1999, 2000, 2004) and the winter emissions given by Ylipetäys et al. (2007) were used to calculate the annual emissions (Table 4). The studied restored peat extraction sites have been sinks or sources for atmospheric CO_2, depending on the time since restoration and vegetation cover. During the growing season there has been uptake of CO_2, but in winter respiration CO_2 loss has in some cases exceeded the uptake during summer. On average, there has been an annual uptake of CO_2, 74.5±202 g m^{-2}, but there is a large variation between the years (Table 4). Petrone et al. (2003) reported a net CO_2 loss of about 1800 g m^{-2} from a vacuum harvested restored peatland two years after restoration in Québeck, Canada. However, these results are not comparable with the earlier ones, due to the different peat harvesting method (milling).

The two restored omrotrophic peat extraction sites studied did not emit N_2O. The mean annual CH_4 emission from two restored ombrotrophic sites three years after rewetting was from 0.62 to 1.46 g m^{-2} (Tuittila et al., 2000). However, in four subsites in another area about 50 years after extraction and with natural development of vegetation, emissions were higher, from 18.7 from 45.2 g m^{-2} (Yli-Petäys et al., 2007), which

is much higher than the average CH_4 emission from natural ombrotrophic peatlands (see above). The average annual CH_4 emission of restored peat extraction sites was thus 21.9±18.3 g m^{-2}. Waddington and Day (2007) reported seasonal (May–October) methane emissions of 4.2 g m^{-2} for a restored peat extraction site in Canada (Québec) three years after restoration, which is close to the results by Tuittila et al. (1999). The site in Canada did not emit CH_4 prior to restoration. Similarly to the peat extraction sites, ditches and ponds can act as hot spots for the CH_4 emissions in restored sites (Waddington and Day, 2007), but this data is not available.

3.3 Relations between soil characteristics and CH_4 and N_2O fluxes on drained peat soils

The data in Fig. 2 includes all the annual emissions from peat soils drained for forestry, agricultural soils, afforested agricultural soils and peat extraction sites, except rewetted sites, where information about soil pH and C/N ratios were available. The C/N ratio in most of the sites with high N_2O emissions was between 15 and 22, similar to that presented by Klemedtsson et al. (2005). However, there was a high variation in annual flux rates within that C/N range. In general, soils with a C/N ratio lower than 15 or higher than 22 had low N_2O emissions. The two exceptions with high C/N ratio and high N_2O emission were agricultural soils where e.g. N-fertilization and ploughing may overcome the C/N dependence, reflecting that the availability of mineral nitrogen for microbial processes (nitrification, denitrification) is important for N_2O production.

Methane emissions seemed to have an increasing trend with increasing C/N ratio, however the regression was not significant. Most of the sites having a C/N ratio in the range from 15 to 22 had close to zero CH_4 fluxes.

Most of the studied drained soils had a pH in the range from 3 to 6 (Tables 2, 3, 4). The N_2O emissions were very variable within this range. The sites with a pH less than 4 or over 6 seemed to have low N_2O emissions. However, there was no significant correlation between soil pH and N_2O emission in the present study. This finding was in contrast to Weslien et al. (2009), who found a relationship between pH and N_2O

emission in afforested agricultural soil. In managed peat soils fertilization and liming may alter the soil pH as can been seen in Fig. 2, where the mean pH was higher in agricultural soils (5.5) than in peat soils drained for forestry (3.8). The highest CH_4 emissions occurred from soils with a low pH, which is in agreement with findings by Weslien et al. (2009). The sites with a low pH were forestry drained sites, where a higher WT level in old drainage systems could explain the higher CH_4 emission.

Soil CH_4 and N_2O fluxes had a nonlinear relationship (Fig. 3). CH_4 emissions occurred only when N_2O emissions were low, except in two cases. These exceptions were a peat extraction site and forestry drained site, where similar N_2O and CH_4 emissions occurred. Surprisingly, Weslien et al. (2009) found the opposite relationship, increasing N_2O emissions with increasing CH_4 emissions. When the several different sites are compared, the WT depth may be the key factor. However, soil WT data were available only from few sites and this comparison was therefore not possible.

3.4 Water reservoirs and artificial lakes

One of the land use options affecting the GHG balance of peatland is flooding of the soil, as when water reservoirs are established. The land impounded is in most cases a mixture of both mineral and organic soils. Measurements on the GHG fluxes are accordingly indirect regarding the peatland, as gases are dissolved in the reservoir before being emitted.

3.4.1 Studied sites and their GHGs emissions

In Finland, the fluxes of all three GHGs have been measured in two large hydroelectric reservoirs in northern Finland (Lokka and Porttipahta) (Huttunen et al., 2002b). These reservoirs flooded large areas of natural peatland within the river Kemijoki watercourse, 76% and 45% of their maximum area is on previous peatlands, respectively. Measurements included flux determinations with floating chambers, while CH_4 ebullition was measured continuously using bubble traps (Table 5). Because the chambers some-

times recorded higher or similar CH_4 emissions than the bubble traps, even chamber results were considered to represent "total CH_4 emissions" including the possible contribution of small bubbles in the CH_4 flux to the chambers (Huttunen et al., 2002b).

Emissions of CO_2 were calculated with the thin boundary layer (TBL) approach from one reservoir flooding an area of 20 km^2 in northern Sweden, of which 30% was previously peatland, mesotrophic Skinnmuddselet within the River Gideälven (Åberg et al., 2004, Table 5). The partial pressure of CO_2 in water (pCO_2) was measured in a subset of hydropower reservoirs with 49–354 km^2 flooded area along seven regulated rivers in northern Sweden and then modeled for the total flooded area based on the relationship between DOC and dissolved CO_2 in lake and reservoir. The CO_2 exchange at the water-air interface was then calculated using the TBL method (Bergström et al., 2004). These reservoirs were oligotrophic and flooded mostly mineral soils and previous water bodies in alpine forest areas.

Fluxes of CH_4, CO_2 and N_2O were measured on the Gilsárlón hydropower reservoir in northern Iceland (area 4.3 km^2, average depth 9 m, constructed in 1990; Óskarsson and Guðmundsson, 2008, 2009). Of the area impounded, 67% was peatland. The CH_4 and N_2O diffusive fluxes were measured with floating chambers and CH_4 ebullition with bubble traps. The CO_2 emission was measured with EC and floating chambers.

The data published to date from the Nordic reservoirs are summarized in Table 5. Degassing emissions at the spring ice melt or those from the outflowing waters downstream of the dam are not included in the data. There are also preliminary data on the CH_4 and CO_2 emissions from one reservoir in the Follsjø area in mid-Norway 6.5 km^2, constructed in 1968, with 8.6% of the impounded area as peat. The CH_4 release greatly differed between the two reservoirs studied in Finland, both of them flooding extensive areas of former peatlands. The chamber measurements indicated up to ten times higher CH_4 release from Lokka compared with Porttipahta, while in Lokka the measurement of ebullition had the seasonal mean up to 26 g CH_4 m^{-2} yr^{-1} and in Porttipahta CH_4 ebullition was almost negligible. The high CH_4 emissions from Lokka have been attributed to its relatively high trophic state and anoxic conditions in the sedi-

ment, although the water column was effectively mixed and aerated by wind during the open water season (Huttunen et al., 2002b). Moreover, the persistence of the high CH_4 release in this 26–27 years old reservoir was suggested to be mainly fuelled by relatively fresh organic carbon, based on the relatively high modern carbon content of CH_4 in the bubbles released from its sediment (Huttunen et al., 2002b). The erosion caused by wind and wave action has been identified as an important source of internal nutrient loading in this shallow eutrophic reservoir (Hellsten et al., 1993), which was further reflected in its relatively high authochthonous primary production (Huttunen et al., 2002b). The data from the Icelandic site shows similarly high CH_4 release from the younger Gilsárlón reservoir, also constructed mainly on previous peatland, releasing annually about 9–11 g CH_4 m^{-2}, of which most was due to CH_4 ebullition (Óskarsson and Guðmundsson, 2009). In Follsjø reservoir in Norway, where peatland contributed less than 10% to the flooded area, CH_4 release was observed during the spring in association with a major water level drawdown of >25 m, but during the open-water season with higher water levels, emissions were at the detection limit (Harby et al., 2006).

The CH_4 emissions from the Nordic sites presented above correspond to the emissions from close to zero to 17 g CH_4 m^{-2} yr^{-1}, calculated from the daily emissions (assuming an active period of 150 d) reported from 34 reservoirs during open water conditions in British Columbia, Manitoba/Ontario and Quebec, Canada (Tremblay et al., 2005). The Nordic data suggests that age does not have a major role in explaining the long-term emission rates between the reservoirs, although CH_4 emissions are known to decrease within some year after a phase of rapid decomposition of flooded fresh organic matter following the flooding (e.g. Bodaly et al., 2004). The shallow water column in Lokka (mean depth 2.3–5.0 m) and Gilsárlón (mean depth 9 m) would have favoured the release of CH_4, since ebullition is likely to occur at shallow water depths, from 1 to 10 m (Keller and Stallard, 1994). This CH_4 escaping from the sediments via bubbles also bypasses possible CH_4 oxidation in the water column. Duchemin et al. (1995) also reported higher CH_4 emissions from shallow (<3 m) than deeper (>7 m) stations

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in LaGrande-2 and Laforge-1 reservoirs in Quebec, and suggested that CH_4 oxidation was important in controlling the CH_4 fluxes from sediments to the atmosphere.

The two Finnish lowland reservoirs and the Icelandic reservoir also showed higher CO_2 release during the open water season than the oligotophic Swedish sites (Table 5) and Follsjø in Norway (estimated annual emission 139 g m^{-2} yr^{-1}) (Harby et al., 2006). However, all these emissions ranged from 8 to 280 g m^{-2} yr^{-1} and were in the range of −155 to 860 g CO_2 m^{-2} yr^{-1} (150 d active period assumed) measured for the CO_2 fluxes from 55 reservoir sites in British Columbia, Manitoba/Ontario, Quebec and New Foundland, Canada, in campaigns during the open-water season (Tremblay et al., 2005). Due to the high ranges of the CO_2 emissions from various boreal reservoirs, the sources of CO_2, whether being within a reservoir or catchment area, need to be identified to allow estimation of potential CO_2 emissions from any planned reservoir in the boreal region.

3.4.2 Emissions at spring ice melt

The springtime mixing of the water-column can make an important contribution to annual GHG emissions from northern reservoirs. At the deepest site in Porttipahta, Finland, GHG emissions were measured during the mixing period after the ice melt with floating chambers and the changes in the GHG profiles in the water were simultaneously determined (Huttunen et al., 2002b). The springtime emissions accounted for most of the annual CH_4 release at that site, while the CO_2 and N_2O emissions during the overturn corresponded to the CO_2 and N_2O emissions during the rest of the season. In Follsjø in Norway, the annual CH_4 release, resulting only from the springtime degassing emission, was estimated at 7.7 g CH_4 m^{-2} yr^{-1} (Harby et al., 2006). At the Swedish sites studied by Bergström et al. (2004), emptying of the reservoirs in January–May was estimated to result in CO_2 emissions which were 2.6–7.4% of the values presented for the open water season (Table 5). The emptying of the reservoir Skinnmuddsselet in Sweden was estimated to result in the emission of 13.9 g CO_2 m^{-2},

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which accounted for 14% of the annual CO_2 emission (Åberg et al., 2004). The study of Duchemin et al. (2006) suggests that the episodic springtime CH_4 and CO_2 emissions from Canadian reservoirs at spring ice melt are a small but non-negligible component of the annual GHG budgets of these sites.

3.4.3 Importance of turbines and downstream river

There is only one experimental study on the importance of outgassing in turbines/spillways and downstream river in the total CO_2 emissions in boreal reservoirs and none on the degassing of CH_4 and N_2O. Roehm and Tremblay (2006) concluded that the degassing of CO_2 from the turbines accounted for only <1% to 7% of the CO_2 emissions occurring through the surface in LaGrande-2 and LaGrande-3 reservoirs in Quebec, Canada. It should be stressed here that the degassing emissions downstream of the dam can play an important role in the CH_4 emissions from some tropical reservoirs (Kemenes et al., 2007).

4 Discussion

4.1 General

A large number of GHG measurements have been carried out in the Nordic countries for managed peatlands. However, in some land use options, measurements have been limited to only few sites with a short measuring period. This reduces the reliability of the estimates on the GHG emissions because they have high spatial and temporal variation, emissions depend on several factors e.g. on weather conditions, drainage intensity (i.e. WT depth), the type of original peatland drained, and the land use history. For some land-use options, the data is totally lacking or scarce. In particular, the carbon balance of forested sites, i.e. peatland forests, afforested peat extraction sites and afforested croplands is poorly studied. In addition, life cycle analyses for not only

peat extraction but also for other land-use options, e.g. forested or cultivated sites are required.

An accurate up scaling of the GHG fluxes of each land-use class requires enough experimental sites to cover the internal variation within the groups. There are certainly risks that this variation is not fully covered with the available data. There is also a lack in the background information regarding the studied sites. For example, peat characteristics (trophy of the site, origin of the peat, pH, C/N ratio, WT depth) and the land use history are rarely reported, which makes comparison between various land-use options difficult. The relationships between the peat characteristics and emissions could have been more informative if this background information were available from all sites (Figs. 2, 3).

4.2 Flux measurement methods

The GHG flux data from different land-use categories has been obtained with variable methods and measuring periods. Most of the N_2O and CH_4 flux studies have been carried out with static chamber techniques, whereas CO_2 balance is studied using either EC or chamber methods. The results obtained with these methods may not be totally comparable (e.g. Clement et al., 1995; Lohila et al., 2007b, 2008; Pihlatie et al., 2009). These methods have both advantages and disadvantages. Chambers cover only a small area of the soil surface and installation of collars for the chamber measurement may disturb the root systems of the plants. The measuring frequency is generally low (e.g. weekly-biweekly) if an automatic chamber system is not applied. The installation of the chambers for the measurements may disturb the gas concentration in soil, causing an extra release of gas from soil. On the other hand, non- linear changes in the gas concentration in chambers, especially with a long incubation time, causes deviation in the real flux rate if a linear approach is used to calculate the flux rate from the change in gas concentration with time (Kroon et al., 2008; Kutzbach et al., 2007; Rochette and Eriksen-Hamel, 2008). A benefit with chambers is that various functional surfaces with a small area can be measured whereas the EC method integrates flux for

a large area without separating the functional surfaces in the footprint area (Alm et al., 2007). EC does not damage the vegetation or disturb soil/vegetation and can be used for tall vegetation like forested ecosystems. The EC method can provide continuous data when the gap filling of missing data is used (e.g. Lohila et al., 2007b).

4.3 Measurement periods

The length of the study periods applied varies from a few months (growing season) to several years. In many studies, winter measurements are scarce or totally missing. In some publications the emissions measured just during a growing season are reported as annual emissions. There the annual emissions are underestimated if the winter emissions are assumed to be zero. The emissions during winter can vary but in most cases the winter emissions contribute significantly to the annual gas budget (e.g. Alm et al., 1997, 1999b; Lohila et al., 2007b; Regina et al., 2004). In particular N_2O emissions during the period with snow and soil frost can contribute even more than 80% of the annual N_2O emissions from e.g. agricultural soils or forested peat soils (Mäkiranta et al., 2007; Maljanen et al., 2004; Maljanen et al., this issue). N_2O produced in frozen soil continuously can accumulate in soil if an ice layer on top of the soil acts as a diffusion barrier (Maljanen et al., 2007b; Maljanen et al., this issue; Koponen et al., 2004). High amounts of accumulated N_2O can be then released during a short thawing period, in addition to new N_2O produced in denitrification (Wagner-Riddle et al., 2008). Depending on the soil properties, high N_2O bursts may also occur during soil freezing (Maljanen et al., 2007b; Maljanen et al. this issue). If these high emission periods are missed then the annual emission can be greatly underestimated.

4.4 Gaps in the knowledge of GHG emissions from peat soils

The gas dynamics of agricultural soils have been studied most intensively, probably because the gas fluxes of these soils have been identified as having great atmospheric importance. When considering organic agricultural soils this is true for all the gases;

CH_4, CO_2 and N_2O. These soils are also methodologically easier to measure than soils with tall vegetation. Winter emissions from agricultural soils are also well documented. However, the long term use of peat soils can induce C-loss from soil, which reduces peat depth. The effect of this change on GHG emissions with time is not known.

In addition, emissions from the drainage ditches have not been reported for any agricultural soils, in contrast to the forestry drained peat soils or peat extraction sites. Emissions from the drainage ditches can have importance in the total emissions of drained sites (Sundh et al., 2000; Minkkinen et al., 1997).

The area of afforested croplands is increasing, and therefore more information from this land-use option is needed. GHG emissions from afforested peat soils can depend on the age of afforestation and tree species. The whole rotation period, including harvesting, should be considered when comparing afforestation and other possible land-use options of drained peat soils. GHG emissions from soils with ground vegetation of peatlands drained for forestry have been studied. However, for forested or afforested sites the data on CO_2 balance including the canopy is sparse. Due to the lack of measured data, the C balance has been estimated using changes in the soil carbon pool and estimating C sequestration in tree stands (Minkkinen et al., 2001; Klemedtsson et al., 2008).

The gas dynamics of the restoration of forestry drained peatlands have been studied in Finland in one geographical location. The restoration of drained peat soils has been carried out in a small scale in the Nordic countries. However, the area of restored peatlands may increase in the future (UNFCCC, 2008; Ministry for the Environment, 2007). In Iceland, large areas of drained organic soils are outside cultivated or afforested land and only used for grazing, often with low intensity. Restoring these areas seems to be a promising option to reduce the present GHG emission. More studies are required to evaluate the effects of the restoration involving different land use history and soil properties. Annually, large areas (mainly in Finland) of peat extraction sites are abandoned and therefore more information on GHG emissions related to various after-use options of peat extraction sites is required. Crop production (energy crops such as reed canary

grass) could be a promising option to mitigate CO_2 losses from drained peat soil, but there is a need for more studies covering sites with variable peat characteristics and hydrological conditions.

4.5 Net emissions from water reservoirs

The present day GHG emissions from reservoirs can be called "gross emissions" (post-flood), conventionally meaning on a smaller scale those GHG emissions occurring though the reservoir surface and sometimes from turbine structures in the vicinity of dams. On a larger scale, these emissions may have included additional emissions from the downstream river directly attributable to the existence of a reservoir upstream. However, the knowledge of how the reservoirs potentially affect the original (preflood) GHG balance of a flooded landscape has highlighted the need to introduce a term "net emissions". Net emissions mean real changes in the watershed-scale GHG (and carbon) balances (and possibly stocks) caused by a reservoir. To estimate this, the large-scale preflood GHG of both the reservoir area and the whole watershed from upstream trough the river down to estuary need to be known, and the "net" is defined as the difference between the postflood and preflood situation. To complete the picture, possible changes in the ecosystem C storages (vegetation, soils) need to be evaluated.

Determination of the net emissions, even based on inventorying of flooded land and GHG flux data from similar ecosystems, could result in unexpected results. In Finnish reservoirs, Lokka and Porttipahta, pre- and postflood fluxes have been calculated for CH_4 and N_2O (Huttunen and Martikainen, 2005a, b). Surprisingly, those reservoirs have lowered rather than increased the CH_4 release in their area, although the CH_4 release from Lokka was mostly high. This was because the minerotrophic peatlands flooded by these reservoirs originally had high CH_4 emissions. For N_2O, changes in the landscape-scale fluxes were not remarkable, since the fluxes both before and after the reservoirs were constructed were low. In Swedish reservoirs, the change of CO_2 fluxes of the landscape was relatively small and the landscape was alpine upland forest. The Icelandic reservoir established on peatland and upland mineral soils was

a considerable source of CO_2, whereas the impounded ecosystems were close to zero or a small source (Óskarsson and Guðmundsson, 2008)

An interesting question is what happens to the carbon stocks in flooded landscapes. Finnish reservoirs have been estimated to have drowned tens Tg of carbon. Based on the measurements at Lokka, the contribution of fresh carbon in the GHG production is most probably high and the fate of flooded peat deposits unknown (Huttunen et al., 2002b). However, a recent tracer study suggested that the new sedimentation in a boreal reservoir is dominated by autochthonous organic carbon or by carbon eroded in other parts of the reservoirs (Houel et al., 2006). These finding may indicate that the recent production of fresh organic matter fuels the GHG production within the systems. The continuing GHG release from a reservoir does not then necessarily lead to major losses from the flooded C deposits. Much needs to be done to be able to identify all sources of GHG production in the reservoirs and their environment, before real net emissions can be assessed.

4.6 Conclusions

Agriculture is climatically the most unfavorable land-use option for peat soils when considering GHG fluxes from biological processes. Drainage, ploughing and fertilization cause a high decomposition rate of peat leading to high CO_2 and N_2O emissions. After drainage, CH_4 emissions decrease and CH_4 has hence been considered to be a minor part of the total GWP of peat soils used for agriculture. Soils with perennial grasses seems to emit less CO_2 and N_2O than soils which are regularly ploughed, e.g. for cultivation of cereals. The mitigation of GHG emissions from peat soils used for agriculture is challenging. Afforestation could lead to lower CO_2 losses but N_2O emissions may remain similar to those from cultivated peat soils for decades. However, there are only few results regarding the CO_2 balance of afforested agricultural soils and more data about the CO_2 balance after afforestation is required to make any conclusions about the atmospheric impact of this land-use option. Evaluation of whole rotation periods and the inclusion of the GHG effect of harvested wood products are needed to better

understand the long term effects of afforestation.

In addition to agricultural soils, peat extraction sites are net GHG emitters. Peat extraction for fuel is a very drastic land use option. After peat extraction has ended, restoration, afforestation or growing of crops may take place, but the GHG emissions from these land use options are still poorly known. If the new biomass dryer method is used in the future it may lower GHG emissions from peat extraction because no pre-drying phase is needed. However, there is no research evidence regarding the after use of such peat extraction site.

Peat soils drained for forestry and reed canary grass cultivation on abandoned peat extraction sites seemed to be climatically more favorable than those used for agriculture. In these cases, carbon loss from the drained peat soil is compensated by the accumulation of C in aboveground and belowground biomass and in soil (Shurpali et al., 2009; Laurila et al., 2007). However, the biomass C is mostly returned back into the atmosphere when these crops are harvested. There is very little experimental data to show the possible atmospheric benefits of perennial crops on organic soils. To estimate the GHG impact of e.g. peat or cultivation of bioenergy crops, life cycle analysis is required (e.g. Kirkinen et al., 2007; Kirkinen et al. 2008; Nilsson and Nilsson, 2004). Furthermore, the fate of the native peat carbon is poorly known. Some ecosystems may lose soil carbon even though there is an increase in the biomass carbon.

Peat soils drained for agriculture have been studied more intensively than the other land-use options. CH_4 and N_2O fluxes from forested peatlands are also studied intensively in Finland and Sweden, where forestry on peat soils has been a common practice. However, measured CO_2 balances are scarce. More data on the net CO_2 exchange is also required for the other afforested peat soils, especially for afforested peat extraction sites where no CO_2 data is available. In addition, the fate of soil carbon is poorly known.

Even if drainage reduces CH_4 emissions, ditches may remain as sources of CH_4 (Minkkinen et al., 1997; Nykänen et al., 1996; Sundh et al., 2001). The emissions from drainage ditches are not known and they are therefore not included in the emissions

from drained sites at present. Ditch emissions can potentially be a significant part of the total GHG budget.

Large emissions of GHG have been detected from hydro reservoirs where peatlands have been flooded. Conversely, the net effects of flooding can, in some cases, decrease CH_4 emission compared to previous land use. Studies on the origin of the emitted GHG indicate that the emission may be mostly driven by fresh carbon rather than the impounded C-stocks.

Restoring of drained wetlands is believed to be an effective measure to sequester carbon as peat, but there is a need for on that issue to determine the long term effect on the balance of all three GHG's. If the present natural state of the undisturbed peatlands is not a sink, as seems to be the case with the Nordic ombrotrophic peatlands, the restoration measures are unlikely to lead to net C sequestration. A natural baseline is therefore always needed when the restoration measures are evaluated. Greatly decreasing the GHG emissions, but still being a small source, may indeed indicate the most natural situation at present and hence the most successful restoration.

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Table 1. Peatland areas in the Nordic countries (km^2).

Soil use option	Finland	Sweden	Norway	Denmark	Iceland
1. Original	104 000[a]	103 000[e,f]	25 000[h]	10 000[j]	9000[l]
2. Undrained	40 000[a]	40 000[e,f]	20 000[p]	910[j]	4000
3. Forestry drained	57 000[a]	14 100[e]	4200[i]	nd	25–37[l,n]
4. Agriculture	850[b]	3015[f]	850[p]–1500[h]	600[j]–1520[q]	3750[o]
4.1 Abandoned	nd	4470[f]	nd	nd	nd
3-4.1 Restored	<100	12[f]	0	109[q,r]	8[m]
4.2 Afforested	850[c]	nd[1]	30[h]	nd	9[l]
5. Peat extraction	630[a]	102–400[g]	0	9[q]	0
5.1. Afforested	52	nd[2]	0	nd	0
5.2.1 Energy crops	28[d]	0	0	nd	0
5.2.2 Other crops	54[d]	0	0	nd	0
5.3. Restored	<66[a]	0	0	nd	0
6. Water reservoirs	600[a]	12[f,3]	nd	nd	5[l]

[a] Turunen (2008)
[b] Myllys and Sinkkonen (2004), not including mull soils, 2140 km^2
[c] Hytönen, personal communication
[d] Wall and Heiskanen (1998)
[e] Hånell (1990)
[f] Berglund and Berglund (2008a, b)
[g] SCB (2007, 2008)
[h] Grønlund et al. (2006)
[i] Paavilainen and Päivänen (1995)
[j] Djurhuus et al. (2005)
[l] Gudmundsson, personal communication (including drained but abandoned after drainage, crop production, 650 km^2)
[m] Votlendisnefnd Landbúnaðarráðuneytisins, 2006 (including lakes and ponds 5 km^2)
[n] Sigurdsson, B. D., personal communication
[o] The Environment Agency of Iceland (2008)
[p] Norwegian Pollution Control Authority (2009)
[q] Nielsen et al. (2009) including annual crops in rotation and set-a-side, grass in rotation and permanent grass accounted for 54%, 11%, 16%, and 18%, respectively. 38% of the organic soils are according to the Danish soil classification are deep organic soils
[r] Includes both restored wetlands (68 km^2, assumed to be only from agriculture and wetlands with elevated water table (41 km^2) in 2005. These wetlands are only reported to have been restored or rewetted on 27 and 13 km^2 organic soils, respectively; the remainder was established on mineral soil.
[1] Most of the abandoned peat fields are forested
[2] Extraction area in 2002, the total area with permission to peat extraction is about 400 km^2
[3] Restored wetlands includes artificial lakes
nd=no data available.

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Table 2. Annual balances of CH_4 ($g\,m^{-2}\,yr^{-1}$), N_2O ($g\,m^{-2}\,yr^{-1}$) and net ecosystem CO_2 exchange ($g\,m^{-2}\,yr^{-1}$, R=only soil respiration) from forestry drained peatlands and restored (REF) forested sites in the Nordic countries (DK=Denmark, FI=Finland, SE=Sweden). Peatland types Om=ombrotrophic peatland or Mi=minerotrophic peatland. Tree species were spruce (SP), beech (BE), alder (AL), pine (PI) or birch (BI). A negative value indicates uptake, a positive value indicates emission. Methods are CH=chamber, GC=gas chromatograph, IR=infra-red analyzer and EC=Eddy correlation method. ^{X} indicates that annual emission is estimated based on seasonal results.

Type, trees	Method	Location		C:N	pH	CH_4	N_2O	CO_2	Reference
Mi, SP	CH+GC	FI	60°21′ N, 25°03′ E	ND	ND	0.02^X	0.02^X	ND	Huttunen et al. (2003)
Mi, SP	CH+GC	FI	61°23′ N, 25°03′ E	ND	ND	0.19^X	0.08^X	ND	Huttunen et al. (2003)
Mi	CH+GC	FI	61°47′ N, 24°18′ E	ND	3.7	3.03	ND	ND	Nykänen et al. (1998)
Mi	CH+GC	FI	61°47′ N, 24°18′ E	ND	3.8	2.69	ND	ND	Nykänen et al. (1998)
Mi	CH+GC	FI	61°47′ N, 24°18′ E	ND	3.8	3.47	ND	ND	Nykänen et al. (1998)
Mi	CH+GC	FI	62°46′ N, 29°50′ E	ND	3.8	0.99	ND	ND	Nykänen et al. (1998)
Mi	CH+GC	FI	61°47′ N, 24°18′ E	ND	4.3	1.1	ND	ND	Nykänen et al. (1998)
Mi	CH+GC	FI	62°51′ N, 30°53′ E	ND	ND	−0.87	ND	ND	Martikainen et al. (1995b)
Mi	CH+GC	FI	61°47′ N, 24°19′ E	ND	ND	0.05	0.15	1320^R	Martikainen et al. (1995a)
Mi	CH+GC	FI	61°48′ N, 24°17′ E	ND	ND	0.1	ND	ND	Komulainen et al. (1998)
Mi	CH+GC	FI	61°48′ N, 24°19′ E	ND	ND	<0.01	ND	ND	Laine et al. (1996)
Mi, PI	EC	FI	60°43′ N, 24°27′ E	ND	ND	ND	0.10	−900	Laurila et al. (2007)
Mi	CH+GC	FI	61°47′ N, 24°18′ E	ND	ND	0.09^X	ND	ND	Martikainen et al. (1992)
Mi, PI	CH+GC	FI	62°51′ N, 30°53′ E	ND	4.3	ND	0.21	ND	Martikainen et al. (1993)
Mi, BI	CH+GC	FI	61°47′ N, 24°18′ E	ND	4.6	ND	0.10	ND	Martikainen et al. (1993)
Mi, PI	CH+GC	FI	61°47′ N, 24°18′ E	ND	4.6	ND	0.15	ND	Martikainen et al. (1993)
Mi, BIPI	CH+GC	FI	63°55′ N, 23°58′ E	21	3.7	−0.20	4.10	ND	Maljanen et al. (2009b)
Mi, BI	CH+GC	FI	62°31′ N, 29°23′ E	20	4.5	−0.39	0.66	ND	Maljanen et al. (2003a, b)
Mi, PI	CH+GC	FI	64°51′ N, 26°33′ E	ND	3.7	−0.30	1.05	ND	Maljanen et al. (2006a)
Mi	CH+GC	FI	64°88′ N, 26°13′ E	ND	3.5	0.77	0.01	ND	Maljanen et al. (2006a)
Mi	CH+GC	FI	64°86′ N, 26°11′ E	ND	4.1	0.06	0.01	ND	Maljanen et al. (2006a)
Mi	CH+GC	FI	64°85′ N, 26°07′ E	ND	3.9	4.76	0.19	ND	Maljanen et al. (2006a)
Mi, PI	CH+IR	FI	58°59′ N, 25°27′ E	ND	ND	ND	ND	1380^R	Minkkinen et al. (2007a)
Mi, PI	CH+IR	FI	61°22′ N, 25°07′ E	ND	ND	ND	ND	909^R	Minkkinen et al. (2007a)
Mi, PI	CH+IR	FI	66°21′ N, 26°37′ E	ND	ND	ND	ND	1890^R	Minkkinen et al. (2007a)
Mi	CH+GC	FI	60°39′ N, 24°22′ E	ND	ND	−0.30	ND	ND	Minkkinen et al. (2007b)

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Table 2. Continued.

Type, trees	Method	Location		C:N	pH	CH_4	N_2O	CO_2	Reference
Mi, SP	CH+GC	FI	61°22′ N, 25°07′ E	ND	ND	−0.69	ND	ND	Minkkinen et al. (2007b)
Mi, BI	CH+GC	FI	61°24′ N, 25°02′ E	ND	ND	−0.44	ND	ND	Minkkinen et al. (2007b)
Mi	CH+GC	FI	66°21′ N, 26°37′ E	ND	ND	−0.03	ND	ND	Minkkinen et al. (2007b)
Mi	CH+GC	FI	66°28′ N, 25°51′ E	ND	ND	0.65	ND	ND	Minkkinen et al. (2007b)
Mi	CH+GC	FI	61°47′ N, 21°18′ E	ND	4.6	ND	0.16^X	ND	Regina et al. (1996)
Mi, PI	CH+GC	FI	61°47′ N, 21°18′ E	ND	4.0	ND	0.05^X	ND	Regina et al. (1996)
Mi, PIBI	CH+GC	FI	62°46′ N, 30°58′ E	ND	4.5	ND	0.22^X	ND	Regina et al. (1996)
Mi	CH+GC	FI	62°46′ N, 30°58′ E	ND	ND	ND	$<0.01^X$	ND	Regina et al. (1996)
Mi, BIPISP	CH+GC	FI	62°46′ N, 30°58′ E	ND	4.5	ND	0.82	ND	Regina et al. (1998)
Mi, BIPISP[b]	CH+GC	FI	62°46′ N, 30°58′ E	ND	4.5	ND	0.93	ND	Regina et al. (1998)
Mi, BIPISP[c]	CH+GC	FI	62°46′ N, 30°58′ E	ND	4.3	ND	1.5	ND	Regina et al. (1998)
Mi, BIPISP[d]	CH+GC	FI	62°46′ N, 30°58′ E	ND	4.5	ND	1.2	ND	Regina et al. (1998)
Mi,BE	CH+GC	DK	55°31′ N, 11°46′ E	ND	6.8	ND	0.10	ND	Struwe and Kjøller (1994)
Mi,BE	CH+GC	DK	55°15′ N, 14°45′ E	ND	5.4	ND	0.30	ND	Struwe and Kjøller (1994)
Mi,BE	CH+GC	DK	55°15′ N, 11°55′ E	ND	5.2	ND	0.14	ND	Struwe and Kjøller (1994)
Mi,BE	CH+GC	DK	55°57′ N, 12°17′ E	ND	4.4	ND	0.15	ND	Struwe and Kjøller (1994)
Mi, SPyoung	CH+GC	SE	57°08′ N, 14°45′ E	26	3.3	0.03	0.08	1430^R	VonArnold et al. (2005b)
Mi, SPold	CH+GC	SE	57°08′ N, 14°45′ E	28	3.2	0.30	0.05	3800^R	VonArnold et al. (2005b)
Mi, PI	CH+GC	SE	57°08′ N, 14°45′ E	40	2.7	1.07	0.04	1470^R	VonArnold et al. (2005b)
Mi, BI	CH+GC	SE	57°08′ N, 14°45′ E	22	3.4	0.9	0.20	1900^R	VonArnold et al. (2005c)
Mi, AL	CH+GC	SE	57°08′ N, 14°45′ E	16	4.5	0.9	0.90	1700^R	VonArnold et al. (2005c)
Mi, SP[a]	CH+GC	FI	60°21′ N, 25°03′ E	ND	ND	0.02^X	0.02^X	ND	Huttunen et al. (2003)
Mi, SP[a]	CH+GC	FI	61°23′ N, 25°03′ E	ND	ND	0.19^X	0.08^X	ND	Huttunen et al. (2003)
Mi SPPI[a]	CH+GC	FI	63°39′ N, 29°29′ E	ND	5.3	0.08	0.02	ND	Saari et al. (2009)
Mi SP	CH+GC	SE	58°23′ N, 12°09′ E	23	4.5	−0.45	0.41	ND	Sikström et al. (2009)
Om, PI	CH+GC	SE	57°15′ N, 13°35′ E	34	4.9	0.89	<0.01	ND	Sikström et al. (2009)
Om, PI	CH+GC	FI	61°47′ N, 24°18′ E	ND	4.1	ND	<0.01	ND	Martikainen et al. (1993)
Om, PI	CH+GC	FI	61°47′ N, 24°18′ E	ND	4.0	ND	<0.01	ND	Martikainen et al. (1993)
Om	CH+GC	FI	61°47′ N, 24°19′ E	ND	ND	4.33	<0.02	880^R	Martikainen et al. (1995a)
Om	CH+GC	FI	61°47′ N, 24°18′ E	ND	ND	0.83^X	ND	ND	Martikainen et al. (1992)
Om	CH+GC	FI	61°51′ N, 24°14′ E	ND	ND	0.7	ND	ND	Komulainen et al. (1998)
Om	CH+GC	FI	61°47′ N, 24°18′ E	ND	3.8	−0.02	ND	ND	Nykänen et al. (1998)
Om	CH+GC	FI	62°46′ N, 29°50′ E	ND	4.0	0.12	ND	ND	Nykänen et al. (1998)
Om	CH+GC	FI	62°46′ N, 29°50′ E	ND	4.5	−0.14	ND	ND	Nykänen et al. (1998)
Om	CH+GC	FI	62°46′ N, 29°50′ E	ND	ND	0.87	ND	ND	Nykänen et al. (1998)
Om	CH+GC	FI	64°49′ N, 26°26′ E	ND	4.3	0.34	0.02	ND	Maljanen et al. (2006a)

Table 2. Continued.

Type, trees	Method	Location		C:N	pH	CH_4	N_2O	CO_2	Reference
Om, Pl	CH+GC	FI	61°47′ N, 21°18′ E	ND	3.8	ND	0.01[X]	ND	Regina et al. (1996)
Om, Pl	CH+GC	FI	61°47′ N, 21°18′ E	ND	4.1	ND	<0.01[X]	ND	Regina et al. (1996)
Om, Pl	CH+GC	FI	61°47′ N, 21°18′ E	ND	3.8	ND	<0.01[X]	ND	Regina et al. (1996)
Om, Pl	CH+GC	FI	61°47′ N, 21°18′ E	ND	4.3	ND	0.01[X]	ND	Regina et al. (1996)
Om	CH+GC	FI	61°47′ N, 21°18′ E	ND	4.0	ND	<0.01[X]	ND	Regina et al. (1996)
Om, Pl	CH+GC	FI	61°47′ N, 21°18′ E	ND	ND	ND	<0.01[X]	ND	Regina et al. (1996)
REF, Mi	CH+IR	FI	61°48′ N, 24°17′ E	29	ND	ND	ND	−705	Komulainen et al. (1999)
REF, Mi	CH+GC	FI	61°48′ N, 24°17′ E	ND	4.3	ND	0.55	ND	Silvan et al. (2002, 2005)
REF, Mi[e]	CH+GC	FI	61°48′ N, 24°17′ E	ND	4.3	ND	2.0	ND	Silvan et al. (2002, 2005)
REF, Mi	CH+GC	FI	61°48′ N, 24°17′ E	29	ND	2.1	ND	ND	Komulainen et al. (1998)
REF, Om	CH+GC	FI	61°51′ N, 24°14′ E	42	ND	4.6	ND	ND	Komulainen et al. (1998)
REF, Om	CH+IR	FI	61°51′ N, 24°14′ E	42	ND	ND	ND	−174	Komulainen et al. (1999)

[a] After clear cut
[b] KNO_3 fertilization
[c] NH_4Cl fertilization
[d] Urea fertilization
[e] NO_3^- fertilization.

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Table 3. Annual emissions of CH_4 ($g\,m^{-2}\,yr^{-1}$), N_2O ($g\,m^{-2}\,yr^{-1}$) and net ecosystem CO_2 exchange ($g\,m^{-2}\,yr^{-1}$, R=only soil respiration) from drained peatlands used for agriculture (GR=grass, BR=barley, PO=potato, CA=carrot, AB=abandoned with vegetation, FA=fallow without vegetation, AF=afforested with pine (PI) birch (BI) or alder (AL)), in the Nordic countries (DK=Denmark, FI=Finland, IS=Iceland, NO=Norway, SE=Sweden). A negative value indicates uptake, a positive value indicates emission. Methods are CH=chambers, GC=gas chromatograph, IR=infra-red analyzer and EC=Eddy correlation method. ^{X} indicates that annual emission is estimated based on seasonal results.

Site	Method		Location	C:N	pH	CH_4	N_2O	CO_2	Reference
GR	CH+GC	FI	62°40′ N, 30°50′ E	19	5.3	0.13	1.3	2200^{R}	Nykänen et al. (1995)
GR	EC	FI	60°53′ N, 23°30′ E	20	5.8	ND	ND	290	Lohila et al. (2004)
GR	CH+GC	FI	62°31′ N, 29°23′ E	17	6.1	−0.08	1.70	ND	Maljanen et al. (2003a, b)
GR	CH+GC	FI	62°46′ N, 30°58′ E	ND	5.3	ND	1.74^{X}	ND	Regina et al. (1996)
GR	CH+IR	FI	62°31′ N, 29°23′ E	17	6.1	ND	ND	2800	Maljanen et al. (2001)
GR	CH+GC+IR	FI	65°55′ N, 23°51′ E	33	4.4	−0.10	0.4	1500	Maljanen et al. (2004)
GR	CH+GC	FI	63°54′ N, 23°56′ E	18	5.0	−0.17	3.7	ND	Maljanen et al. (2009a)
GR	CH+GC	FI	63°09′ N, 27°20′ E	13	5.8	−0.12	0.9	ND	Maljanen et al. (2009a)
GR	CH+GC	FI	63°54′ N, 23°56′ E	18	5.0	0.04	5.5	ND	Maljanen et al. (2009b)
GR	CH+GC	FI	60°49′ N, 23°30′ E	21	5.8	−0.03	ND	ND	Regina et al. (2007)
GR	CH+GC	FI	66°35′ N, 26°01′ E	18	5.6	0.64	ND	ND	Regina et al. (2007)
GR	CH+GC	FI	60°49′ N, 23°30′ E	21	5.8	ND	1.2	ND	Regina et al. (2004)
GR	CH+GC	FI	66°35′ N, 26°01′ E	18	5.6	ND	0.63	ND	Regina et al. (2004)
GR	CH+GC	SE	58°20′ N, 13°30′ E	10	7.4	0.12	0.20	ND	Kasimir, K. et al. (2009)
GR	CH+GC	NO	67°17′ N, 14°28′ E	17	5.2	1.6	0.69	4800^{R}	Grønlund et al. (2006)
GR	CH+GC	NO	67°17′ N, 14°28′ E	17	5.2	ND	ND	2700	Grønlund et al. (2008)
GR	CH+GC	IS	64°34′ N, 21°46′ E	16	4.7	ND	$0.05^{X,a}$	ND	Guðmundsson (unpubl.)
PO	CH+GC	FI	60°49′ N, 23°30′ E	21	5.8	ND	1.57	ND	Regina et al. (2004)
BA	EC	FI	60°53′ N, 23°30′ E	20	5.8	ND	ND	771	Lohila et al. (2004)
BA	CH+GC	FI	62°31′ N, 29°23′ E	17	6.1	−0.24	1.3	ND	Maljanen et al. (2003a, b)
BA	CH+IR	FI	62°31′ N, 29°23′ E	17	6.1	ND	ND	1500	Maljanen et al. (2001a)
BA	CH+GC+IR	FI	65°55′ N, 23°51′ E	31	5.0	−0.1	1.3	3000	Maljanen et al. (2004)
BA	CH+GC	FI	60°49′ N, 23°30′ E	21	5.8	−0.03	ND	ND	Regina et al. (2007)
BA	CH+GC	FI	66°35′ N, 26°01′ E	18	5.6	0.24	ND	ND	Regina et al. (2007)
BA	CH+GC	FI	60°49′ N, 23°30′ E	21	5.8	ND	2.36	ND	Regina et al. (2004)
BA	CH+GC	FI	66°35′ N, 26°01′ E	18	5.6	ND	2.04	ND	Regina et al. (2004)
BA	CH+GC	SE	58°20′ N, 13°30′ E	10	7.2	−0.01	1.51	ND	Kasimir, K. et al. (2009)

Table 3. Continued.

Site	Method		Location	C:N	pH	CH_4	N_2O	CO_2	Reference
AB[c]	CH+GC	IS	64°34′ N, 21°46′ E	17	5.0	ND	0.07[X]	ND	Guðmundsson (unpubl.)
AB	CH+GC	FI	63°54′ N, 23°56′ E	19	4.9	−0.22[b]	0.32	2980	Maljanen et al. (2007a)
AB	CH+GC	FI	63°54′ N, 23°56′ E	19	5.0	−0.22[b]	0.64	417	Maljanen et al. (2007a)
AB	CH+GC	FI	63°54′ N, 23°56′ E	18	5.9	−0.22[b]	0.48	287	Maljanen et al. (2007a)
AB	CH+GC	FI	63°54′ N, 23°56′ E	16	4.5	−0.22[b]	0.64	1870	Maljanen et al. (2007a)
AB	CH+GC	FI	63°54′ N, 23°56′ E	19	4.3	−0.22[b]	2.24	1060	Maljanen et al. (2007a)
FA	CH+GC	FI	60°49′ N, 23°30′ E	21	5.8	ND	0.69	ND	Regina et al. (2004)
FA	CH+GC	FI	66°35′ N, 26°01′ E	18	5.6	ND	3.96	ND	Regina et al. (2004)
FA	CH+GC	FI	62°31′ N, 29°23′ E	17	6.1	−0.26	1.70	3360	Maljanen et al. (2003a, b)
FA	CH+GC	FI	65°55′ N, 23°51′ E	31	5.0	−0.04	3.70	2710	Maljanen et al. (2004)
FA	CH+GC	FI	62°40′ N, 30°50′ E	19	5.3	−0.01	0.58	1450	Nykänen et al. (1995)
AF PI	EC	FI	62°12′ N, 22°42′ E	ND	ND	ND	0.94	ND	Pihlatie et al. (2004)
AF BI	CH+GC	SE	58°20′ N, 13°30′ E	13	3.0	−0.8	3.05	ND	Weslien et al. (2009)
AF PI	EC	FI	62°10′ N, 22°47′ E	22	4.8	ND	ND	50	Lohila et al. (2007)
AF BI	CH+GC	FI	63°54′ N, 23°56′ E	13	4.7	−0.15[b]	3.5	ND	Mäkiranta et al. (2007)
AF BI	CH+GC	FI	63°54′ N, 23°56′ E	18	4.6	−0.15[b]	3.4	ND	Mäkiranta et al. (2007)
AF BI	CH+GC	FI	63°54′ N, 23°56′ E	19	4.2	−0.15[b]	0.73	ND	Mäkiranta et al. (2007)
AF PI	CH+GC	FI	63°54′ N, 23°56′ E	17	4.1	−0.15[b]	3.1	ND	Mäkiranta et al. (2007)
AF BI	CH+GC	FI	63°54′ N, 23°56′ E	19	4.1	−0.15[b]	0.17	ND	Mäkiranta et al. (2007)
AF PI	CH+GC	FI	64°06′ N, 24°21′ E	18	4.7	−0.15[b]	1.8	ND	Mäkiranta et al. (2007)
AF BI	CH+GC	FI	64°06′ N, 24°21′ E	19	5.4	−0.15[b]	0.11	ND	Mäkiranta et al. (2007)
AF BI	CH+GC	FI	64°06′ N, 24°21′ E	19	5.3	1.72[X]	0.43[X]	ND	Maljanen et al. (2001b)
AF PI	CH+GC	FI	64°06′ N, 24°21′ E	ND	4.9	1.40[X]	0.22[X]	ND	Maljanen et al. (2001b)
AF BI	CH+GC	FI	64°06′ N, 24°21′ E	19	4.7	−0.10[X]	0.97[X]	ND	Maljanen et al. (2001b)
AF AL	CH+GC	SE	57°08′ N, 14°45′ E	21	4.2	0.90	0.90	ND	VonArnold et al. (2005c)

[a] average of preliminary results from three sites

[b] average of five sites (Maljanen et al., 2007a) or average of seven sites (Mäkiranta et al., 2007)

[c] average of preliminary results from three sites, drained but not ploughed, used for grazing occasionally

Table 4. Annual balances of CH_4 ($g\,m^{-2}\,yr^{-1}$), N_2O ($g\,m^{-2}\,yr^{-1}$) and net ecosystem CO_2 exchange ($g\,m^{-2}\,yr^{-1}$) from peat extraction sites (PE=active site, ABE=abandoned, AFE=afforested with pine (PI) or birch (BI), REE=restored, RCG=abandoned extraction site cultivated with reed canary grass) in Finland (FI) and Sweden (SE). Also, emissions from the ditches (DI) are shown. A negative value indicates uptake, a positive value indicates emission. Methods used for the measurements are CH+GC=chamber method+analysis by gas chromatograph, IF+GC=inverted funnel for gas collection from ditches, CH+IR=chamber method+infra-red analyzer and EC=Eddy correlation method. [X] indicates that annual emission is estimated based on seasonal results.

Site	Method		Location	C:N	pH	CH_4	N_2O	CO_2	Reference
PE	CH+IR	FI	62°46′ N, 30°58′ E	ND	ND	ND	ND	870	Ahlholm et al. (1990)
PE	CH+GC	FI	62°30′ N, 30°30′ E	42	4.3	0.99	0.004	ND	Hyvönen et al. (2009)
PE	CH+IR	FI	62°30′ N, 30°30′ E	42	4.3	ND	ND	381	Shurpali et al. (2008)
PE	CH+GC	FI	62°46′ N, 30°58′ E	ND	4.6	ND	0.02[X]	ND	Regina et al. (1996)
PE	CH+GC	FI	62°47′ N, 24°18′ E	ND	ND	0.32	ND	880	Nykänen et al. (1996)
PE, new	CH+GC	FI	62°40′ N, 30°55′ E	ND	ND	0.24[a]	0.08[a]	ND	Nykänen et al. (1996)
PE	CH+GC	FI	62°40′ N, 30°55′ E	ND	ND	0.24[a]	0.08[b]	ND	Nykänen et al. (1996)
PE	CH+GC	SE	64°05′ N, 18°10′ E	ND	ND	4.97[X]	ND	631[X]	Sundh et al. (2000)
PE	CH+GC	SE	64°55′ N, 17°08′ E	ND	ND	4.97[X]	ND	900[X]	Sundh et al. (2000)
PE	CH+GC	SE	64°05′ N, 20°30′ E	ND	ND	0.11[X]	ND	780[X]	Sundh et al. (2000)
PE	CH+GC	SE	64°05′ N, 20°30′ E	ND	ND	0.11[X]	ND	303[X]	Sundh et al. (2000)
PE	CH+GC	SE	60°20′ N, 16°50′ E	ND	ND	0.05[X]	ND	566[X]	Sundh et al. (2000)
PE	CH+GC	SE	60°03′ N, 16°51′ E	ND	ND	1.14[X]	ND	427[X]	Sundh et al. (2000)
PE, new	CH+GC	SE	50°42′ N, 16°17′ E	ND	ND	1.09[X]	ND	1130[X]	Sundh et al. (2000)
PE, old	CH+GC	SE	50°42′ N, 16°17′ E	ND	ND	−0.05[X]	ND	517[X]	Sundh et al. (2000)
PE	CH+IR	FI	Several sites	ND	ND	7.23	0.31	980	Alm et al. (2007)
ABE	CH+GC	FI	62°40′ N, 30°55′ E	ND	ND	0.24[a]	0.08[a]	ND	Nykänen et al. (1996)
ABE	CH+GC	FI	62°12′ N, 23°18′ E	ND	ND	ND	ND	661[X]	Tuittila et al. (1995)
ABE	CH+GC	FI	62°12′ N, 23°18′ E	ND	ND	0.25	ND	ND	Tuittila et al. (2000)
ABE[g]	CH+IR	FI	62°12′ N, 23°18′ E	ND	ND	ND	ND	717[X]	Tuittila et al. (2004)
ABE[h]	CH+IR	FI	62°12′ N, 23°18′ E	ND	ND	ND	ND	841[X]	Tuittila et al. (2004)
REE[b]	CH+IR	FI	62°12′ N, 23°18′ E	ND	ND	ND	ND	−90[X]	Kivimäki et al. (2008)
REE[c]	CH+IR	FI	62°12′ N, 23°18′ E	ND	ND	ND	ND	−101[X]	Kivimäki et al. (2008)
REE[d]	CH+IR	FI	62°12′ N, 23°18′ E	ND	ND	ND	ND	−355[X]	Kivimäki et al. (2008)
REE[e]	CH+IR	FI	62°12′ N, 23°18′ E	ND	ND	ND	ND	−317[X]	Kivimäki et al. (2008)
REE[f]	CH+IR	FI	62°12′ N, 23°18′ E	ND	ND	22.0	ND	−21.2	Yli-Petäys et al. (2007)
REE[f]	CH+IR	FI	62°12′ N, 23°18′ E	ND	ND	36.7	ND	−3.92	Yli-Petäys et al. (2007)
REE[f]	CH+IR	FI	62°12′ N, 23°18′ E	ND	ND	48.0	ND	−12.6	Yli-Petäys et al. (2007)
REE[f]	CH+IR	FI	62°12′ N, 23°18′ E	ND	ND	61.1	ND	−29.9	Yli-Petäys et al. (2007)

Table 4. Continued.

Site	Method		Location	C:N	pH	CH_4	N_2O	CO_2	Reference
REE	CH+IR	FI	62°12′ N, 23°18′ E	ND	ND	ND	ND	−75.1[X]	Tuittila et al. (1999)
REE	CH+IR	FI	62°12′ N, 23°18′ E	ND	ND	ND	ND	300[X]	Tuittila et al. (1999)
REE[g]	CH+IR	FI	62°12′ N, 23°18′ E	ND	ND	ND	ND	−37.0[X]	Tuittila et al. (2004)
REE[h]	CH+IR	FI	62°12′ N, 23°18′ E	ND	ND	ND	ND	343[X]	Tuittila et al. (2004)
REE	CH+GC	FI	62°12′ N, 23°18′ E	ND	ND	0.57	ND	ND	Tuittila et al. (2000)
REE	CH+IR	FI	62°12′ N, 23°18′ E	ND	4.1	ND	ND	−338[X]	Soini et al. (2009)
AFE, BI	CH+GC	FI	62°12′ N, 23°18′ E	19	4.4	−0.05[a]	0.18	ND	Mäkiranta et al. (2007)
AFE, BI	CH+GC	FI	62°12′ N, 23°18′ E	19	4.0	−0.05[a]	0.17	ND	Mäkiranta et al. (2007)
AFE, BI	CH+GC	FI	62°12′ N, 23°18′ E	18	3.9	−0.05[a]	0.68	ND	Mäkiranta et al. (2007)
AFE, PI	CH+GC	FI	62°12′ N, 23°18′ E	16	4.1	−0.05[a]	0.13	ND	Mäkiranta et al. (2007)
AFE, BI	CH+GC	FI	62°12′ N, 23°18′ E	19	4.6	−0.05[a]	0.75	ND	Mäkiranta et al. (2007)
RCG	EC	FI	62°30′ N, 30°30′ E	42	5.3	ND	ND	−365	Shurpali et al. (2009)
RCG	CH+GC	FI	62°30′ N, 30°30′ E	42	5.3	0.38	0.09	ND	Hyvönen et al. (2009)
DI	CH+GC	SE	64°05′ N, 18°10′ E	ND	ND	15.4[X]	ND	547[X]	Sundh et al. (2000)
DI	CH+GC	SE	64°55′ N, 17°08′ E	ND	ND	5.46[X]	ND	209[X]	Sundh et al. (2000)
DI	CH+GC	SE	64°05′ N, 20°30′ E	ND	ND	6.01[X]	ND	542[X]	Sundh et al. (2000)
DI	CH+GC	SE	64°05′ N, 20°30′ E	ND	ND	11.4[X]	ND	−556[X]	Sundh et al. (2000)
DI	CH+GC	SE	60°20′ N, 16°50′ E	ND	ND	15.9[X]	ND	800[X]	Sundh et al. (2000)
DI	CH+GC	SE	60°03′ N, 16°51′ E	ND	ND	79.5[X]	ND	502[X]	Sundh et al. (2000)
DI, new	CH+GC	SE	50°42′ N, 16°17′ E	ND	ND	124[X]	ND	466[X]	Sundh et al. (2000)
DI, old	CH+GC	SE	50°42′ N, 16°17′ E	ND	ND	94.4[X]	ND	−159[X]	Sundh et al. (2000)
DI, old	IF+GC	FI	62°47′ N, 24°18′ E	ND	ND	0.23	ND	227	Nykänen et al. (1996)
DI, new	IF+GC	FI	62°47′ N, 24°18′ E	ND	ND	137	ND	807	Nykänen et al. (1996)

[a] Average values from three sites (Nykänen et al., 1996) or from five sites (Mäkiranta et al., 2007).
[b] *Eriophorium*
[c] *Carex*
[d] *Eriophorium+Sphagnum*
[e] *Carex+Sphagnum*
[f] *Sphagnum*
[g] *Sphagnum, wet*
[h] *Sphagnum, dry.*

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Table 5. Gross GHG emissions ($g\,m^{-2}\,yr^{-1}$) from hydroelectric reservoirs in the Nordic countries (FI=Finland, SE=Sweden, IC=Iceland). Methods are FC=gas sampling of floating chamber+GC analyses; EB=recording of ebullition by bubble traps+GC analyses (EBf=ice free, EBi=ice); TBL=water sampling for pCO_2 and the flux calculation by the TBL method. Only the GHG fluxes during open-water season are presented here.

Site	Meth.	Location		Flooded land	CH_4	N_2O	CO_2
Lokka	FC	FI	67°48′ N, 27°01′ E	Forest 21% Mire 75.7% Water 3.3%	1.8–4.7	0.002–0.01	152–280[a]
	EB	FI	67°48′ N, 27°01′ E		6.6–26		
Porttipahta	FC	FI	68°02′ N, 26°46′ E	Forest 44.7% Mire 54.6% Water 0.4%	0.53	0.02	232[a]
	EB				0.11		
Skinnmuddselet	TBL	SE	63°59′ N,18°26′ E	Forest 55% Mire 33% Water 15%	ND	ND	87.6[b]
R. Ljusnan/Voxnan[f]	TBL	SE	63–67° N, 13–19° E	[g]	ND	ND	101[c]
R. Ljungan[f]	TBL	SE	63–67° N, 13–19° E	[g]	ND	ND	59.7[c]
R. Indal[f]	TBL	SE	63–67° N, 13–19° E	[g]	ND	ND	61.6[c]
R. Ångerman[f]	TBL	SE	63–67° N, 13–19° E	[g]	ND	ND	41.1[c]
R. Ume[f]	TBL	SE	63–67° N, 13–19° E	[g]	ND	ND	21.2[c]
R. Skellefte[f]	TBL	SE	63–67° N, 13–19° E	[g]	ND	ND	12.6[c]
R. Lule[f]	TBL	SE	63–67° N, 13–19° E	[g]	ND	ND	8.0[c]
Gilsárlón	FC	IC	62°57′ N, 09°07′ E	Heath+dwarf Shrubs 33% Peatland 67%	0.41–0.57	0	ND[d]
	EBf	IC	62°57′ N, 09°07′ E		8.6–10.2	ND	ND[d]
	EBi	IC	62°57′ N, 09°07′ E		0.4	ND	ND[d]
	EC	IC	62°57′ N, 09°07′ E		ND	ND	180[e]

[a] Huttunen et al. (2002b)
[b] Åberg et al. (2004)
[c] Bergström et al. (2004)
[d] Óskarsson and Guðmundsson (2009)
[e] Óskarsson and Guðmundsson (2008)
[f] Data from several reservoirs within each dammed river.
[g] The sites represent dammed regulated rivers located on alpine forest areas, the preflood water surface areas accounted for 57–92% of the postflood surface area in the different rivers.

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Table 6. Annual GHG emissions from peat soils vs. mineral soils used for forestry or agriculture based on studies in the Nordic countries. Negative values indicate net uptake and positive values net emission from the ecosystem (Mean=average of all studies, Min=lowest value, Max=highest value, *n*=number of studies).

Gas ($g\,m^{-2}\,yr^{-1}$)	Peat soils (agriculture)[a]				Mineral soils (agriculture)[c]			
	Mean	Min	Max	*n*	Mean	Min	Max	*n*
Net CO_2 exchange	1790	290	3040	7	−741	–	–	1
CH_4 flux	0.123	−0.24	1.56	15	−0.053	−0.26	0.033	16
N_2O flux	1.39	0.046	5.50	19	0.342	0.050	0.64	26
Total GWP as CO_2 eq	2200				−640			
	Peat soils (forests)[b]				Mineral soils (forests)[d]			
	Mean	Min	Max	*n*	Mean	Min	Max	*n*
Net CO_2 exchange	−900	–	–	1	−480	−1100	760	16
CH_4 flux	0.647	−0.869	4.76	40	−0.262	−0.760	−0.025	11
N_2O flux	0.324	>0.001	4.10	44	0.035	−0.009	0.108	16
Total GWP as CO_2 eq	−790				−480			

[a] Data from Table 3. excluding fallow and abandoned soils.

[b] Data from Table 2. excluding clear cut and restored sites.

[c] Data from Ambus (1998); Ambus et al. (2001); Chatskikh and Olesen (2007); Flechard et al. (2007); Kanerva et al. (2007); Kasimir Klemedtsson (unpubl.); Maljanen et al. (2007b), Virkajärvi et al. (2009); Perälä et al. (2006); Petersen (1999); Regina et al. (2007); Syväsalo et al. (2004, 2006); Sousana et al. (2007).

[d] Data from Ambus et al. (2001); Kasimir Klemedtsson and Klemedtsson (1997); Klemedtsson et al. (1997); Lagergren et al. (2008); Lindroth et al. (1998, 2007); Maljanen et al. (2006a, b, 2009); Markkanen et al. (2001); Pihlatie et al. (2005, 2007); Pilegaard et al. (2003, 2006); Saari et al. (1998, 2004, 2005); Suni et al. (2003); Valentini et al. (2000); Yu et al. (2008); Zha et al. (2007).

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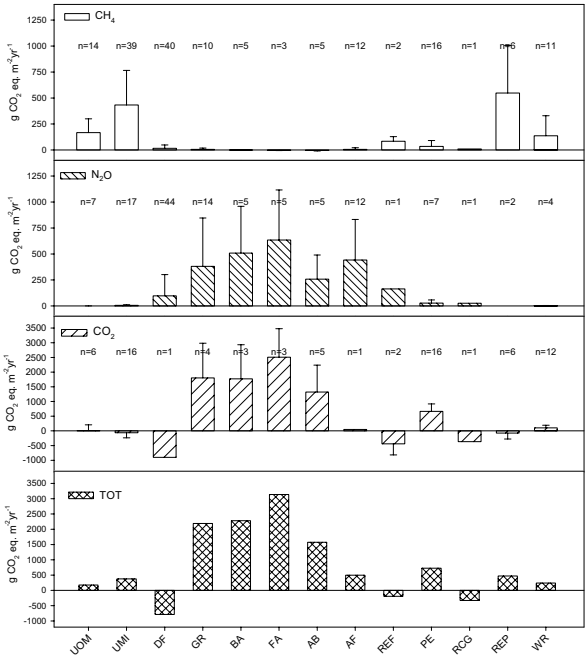

Fig. 1. The mean values of reported annual net fluxes of CH_4, N_2O and net CO_2 exchange and the net effect of these gases as $g\,CO_2\,eq\,m^{-2}$ (100-year time horizon) from different peatland categories in the Nordic countries (UOM=undrained ombrotrophic peatlands, UMI=undrained minerotrophic peatlands, DF=ombrotrophic and minerotrophic sites drained for forestry, GR=drained for agriculture – grass, BA=agriculture – barley, FA=agriculture – fallow, AF=agriculture – afforested, AB=agriculture – abandoned, REF=forestry – restored, PE=drained for peat extraction, RCG=abandoned peat extraction – cultivation of reed canary grass, REP=drained for peat extraction – restored, WR=water reservoirs). A negative value indicates uptake by the ecosystem and a positive value net emission, *n*=number of sites, error bars indicate standard deviation between sites.

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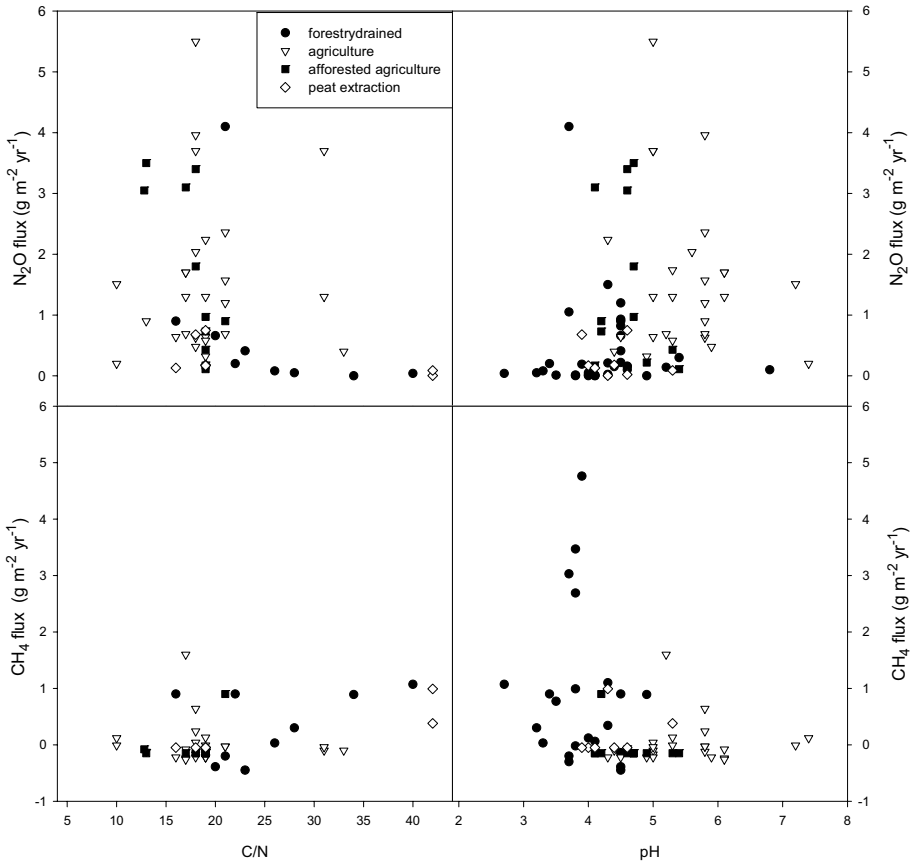

Fig. 2. Relationships between annual fluxes of N_2O and CH_4 and soil C/N ratio and soil pH on drained peat soils (black circle=forestry drained, open triangle=agriculture, black square=afforested agricultural soils, open diamond=peat extraction sites).

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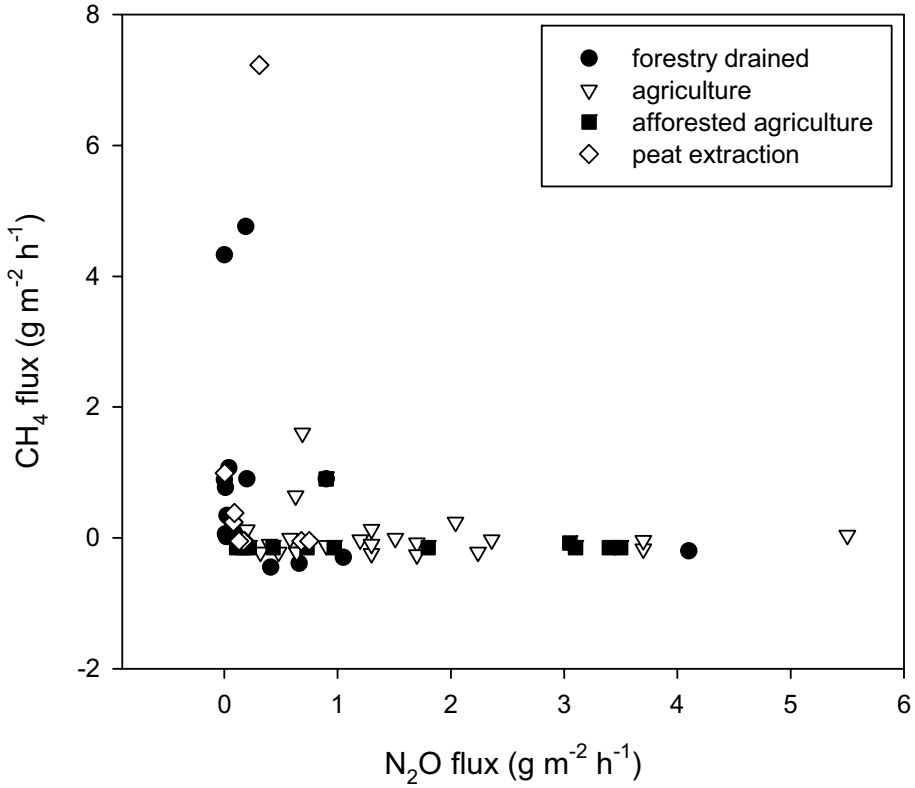

Fig. 3. Relationships between annual fluxes of N_2O and CH_4 on drained peat soils (black circle=forestry drained, open square=agriculture, black circle=afforested agricultural soils, open triangle=peat extraction sites).

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