Spatial variations in GHG emissions from drained organic forest soils

– implications for Swedish UNFCCC reporting

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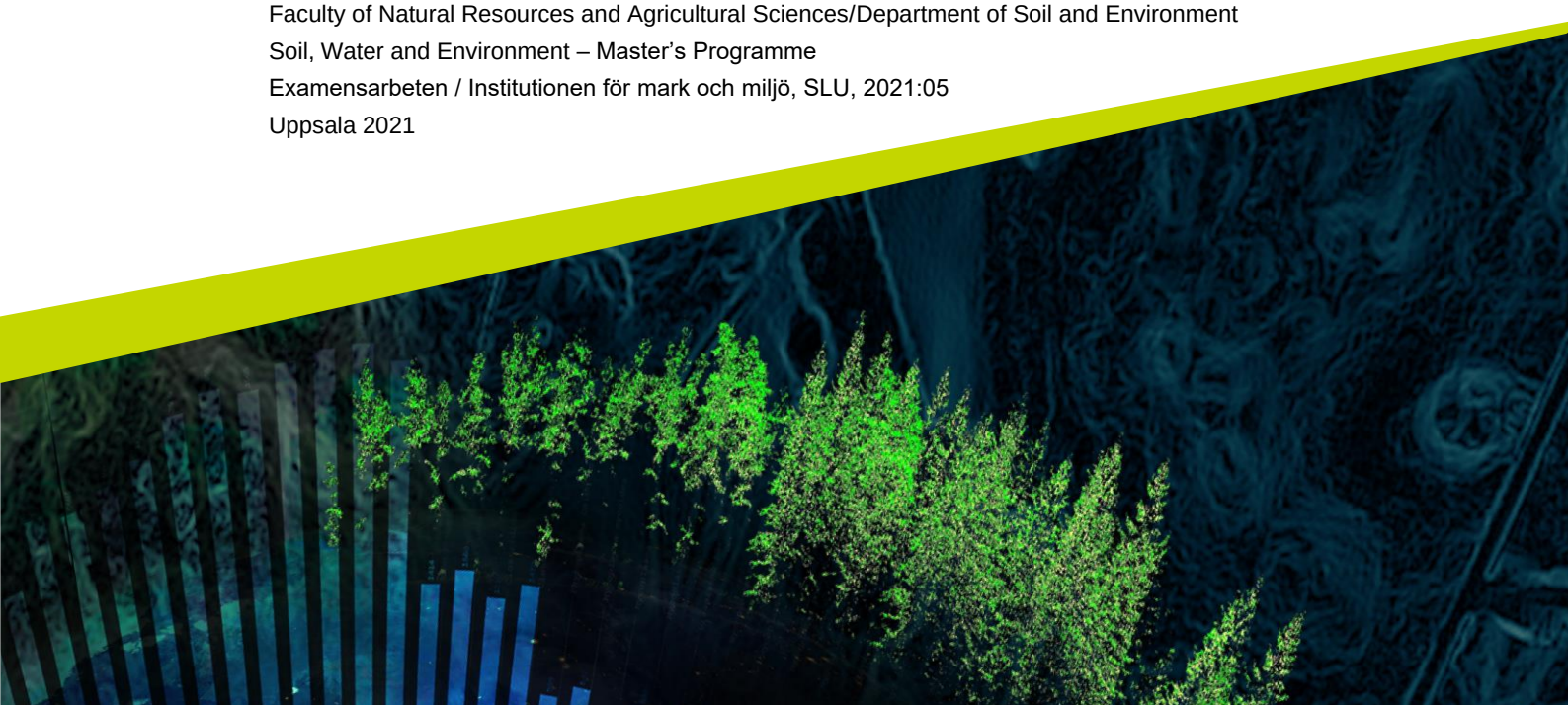

Spatial variation in GHG emissions from drained organic forest soils – implications for Swedish UNFCCC reporting

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Abstract

Drainage of organic soils for forestry generally increases soil CO_2 and N_2O emissions but decreases CH_4 emissions. Under the United Nations Framework Convention on Climate Change (UNFCCC) countries provide annual national inventories of anthropogenic greenhouse gas (GHG) emissions and removals, which includes emissions of GHGs from drained organic forest soil. In the Swedish GHG inventory these emissions are estimated by multiplying default emission factors (EFs) provided by the IPCC by the area of drained organic forest soil. A soil is considered drained if there is a functioning ditch within 25 m from the centre of a plot in the Swedish National Forest Inventory (NFI), however, this corresponds to only half of the total area of productive forest on organic soils.

The aim of this study was to assess GHG emissions from drained organic forest soils, and to suggest additional parameters with which emission factors could be adjusted. To assess the default EFs, a literature review was carried out in which emission data from original peer-reviewed studies were compiled and compared to the default values. To investigate the effect of distance to a ditch on GHG emissions, CO_2, CH_4 and N_2O fluxes were measured along 52.5 m long transects from a ditch in a pine forest stand on drained organic peat soil. Water table depth (WTD) and soil moisture were also measured.

The default EFs and those developed based on the literature review in this study differed markedly and emission estimates varied substantially within climate and nutrient subtypes. Using the emission factors found in the literature review in this study would result in total GHG emissions of 6.5 Mt CO_2-eq., compared to 6.1 Mt CO_2-eq. using the IPCC emission factors.

The GHG measurements showed that average fluxes of CO_2 and CH_4 differed significantly between <25 m and >25 m from the ditch. CO_2 fluxes at >25 m corresponded to 65% of fluxes at <25 m. CH_4 fluxes were 20% higher at >25 m. No significant difference was found in N_2O. CO_2 was linearly correlated with both WTD and soil moisture, while CH_4 showed a quadratic correlation with WTD and soil moisture. No correlation with either parameter was found in the N_2O emissions.

The results show that using data published within the last ten years results in EFs that differ considerably to the EFs of the IPCC, however, to develop new and robust national EFs for Sweden would require a more extensive compilation of emission data than was possible within this study. The results also show that not including drained organic forest soils farther than 25 m from a ditch may lead to substantial underestimation of GHG emissions. Using WTD or soil moisture as predictors of GHG emissions is possible. However, to do so in the inventory would require large national datasets of both WTD and GHG emissions. This requires more studies of GHG emissions from different peatland and forest types in Sweden.

Keywords: greenhouse gas, carbon dioxide, methane, nitrous oxide, peat, drainage, ditch, GHG inventory

Popular science summary

Drainage of an organic soil for forestry generally increases emissions of greenhouse gases (GHGs) such as carbon dioxide (CO_2) and nitrous oxide (N_2O). On the other hand, the emission of methane (CH_4) decreases when the soil is drained. As a part of the effort to combat climate change, countries that have signed the United Nations Framework Convention of Climate Change (UNFCCC) must report all sources and sinks of GHGs caused by human activities. This includes emissions from drained organic forest soils. In the Swedish inventory, the estimation of these emissions is based on a simple calculation: an emission factor multiplied by the total area of drained organic forest soil. A soil is considered drained if there is a ditch within 25 m. However, only half of the total area of forest land on organic soils is located within 25 m from a ditch, but may still be affected by drainage. This means that potentially, a large part of the soil emissions are overlooked. One way that the estimation may be improved is by possibly relating the emissions to drainage parameters that can be measured or modelled, such as ground water table depth, or moisture content in the soil.

This study aimed to answer three questions: How do GHG emissions found in other scientific studies compare with the emission factors used in the Swedish GHG inventory? How are GHG emissions affected by the distance from a ditch? And can water table depth or soil moisture be used to estimate GHG emissions?

The first question was answered through a review of scientific papers. It was discovered that the data in these studies were very different from the emission factors used in the Swedish inventory, and would lead to very varied emission estimates. The second and third questions were answered through a field experiment. CO_2, CH_4 and N_2O emissions were measured on an organic soil in a 125-year-old pine forest. The emissions were measured at points along a transect starting at the ditch and stretching across the 25 m boundary. It was discovered that CO_2 emission was 35% lower and CH_4 fluxes were 20% higher beyond 25 m from the ditch, but that N_2O emissions were not affected. Water table depth and soil moisture could be linked to CO_2 and CH_4, but not to N_2O. Overall, the results showed that not including all forested organic soils in the inventory could lead to an underestimation of GHG emissions. They also showed that water table depth or soil moisture could be used to estimate GHG emissions, but that more national data on both drainage parameters and GHG emissions is needed.

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Abbreviations

EF	Emission factor
FAO	Food and Agriculture Organization of the United Nations
GHG	Greenhouse gas
IPCC	Intergovernmental Panel on Climate Change
LULUCF	Land Use, Land Use Change and Forestry
MDF	Minimum detectable flux
NFI	National Forest Inventory
NIR	National Inventory Report
UNFCCC	United Nations Framework Convention on Climate Change
WL GL	2013 Supplement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories: Wetlands
WTD	Water table depth

1. Introduction

1.1. Emission of greenhouse gases from drained organic forest soils

Globally, drained peatlands are a major point of interest in regards to climate change and emissions and removals of the greenhouse gases (GHG) carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) (IPCC 2014). It has been estimated that the global peatland coverage is 4.23 million km^2 or 2.84% of the world's land area (Xu *et al.* 2018). More than 500 000 km^2 of peat is estimated to have been drained for purposes of agriculture and forestry (Joosten 2009).

The drainage of peat for forestry has a serious impact on peat carbon and nitrogen dynamics. While some studies have found drained organic forest soils to be a net sink of CO_2 (eg. Ojanen *et al.* 2010, 2013; Lohila *et al.* 2011; Minkkinen *et al.* 2018), they are generally considered a net CO_2 source (IPCC 2014; Wüst-Galley *et al.* 2016). The increased availability of oxygen associated with drainage results in increased rates of aerobic decomposition, thus increasing CO_2 release.

For CH_4, the situation is the opposite. The formation of CH_4 by microbial decomposition occurs in anaerobic conditions, and thus the emission of CH_4 decreases when the soil is aerated through drainage (Christiansen *et al.* 2012; Mustamo *et al.* 2016). Drainage may even result in a small sink of CH_4 when aerobic conditions in the topsoil favour microbial oxidisation of CH_4 (Christiansen *et al.* 2012). Emissions of CH_4 from areas of drained organic forest soil occur mainly from the ditches themselves (Schrier-Uijl *et al.* 2011).

N_2O is formed as a by-product in both nitrification and denitrification by soil microorganisms (Butterbach-Bahl *et al.* 2013). The emission of N_2O has been shown to be larger in drained peatlands than in pristine peatlands (Mustamo *et al.* 2016; Pärn *et al.* 2018).

Under the United Nations Framework Convention on Climate Change (UNFCCC), as well as the Kyoto Protocol and Paris Agreement, signatory countries commit to the reduction of anthropogenic GHG emissions. A vital part of the different climate frameworks is the estimation and reporting of anthropogenic GHG

emissions and removals on a national basis. This includes emissions and removals from drained organic forest soils and is further detailed below (section 1.2).

Drainage of organic soils for forestry has been a long-time and widespread practice in Sweden. Although the drainage of soils is currently subject to authorisation or even banned in large parts of Sweden, large areas of drained organic soil remain. Roughly 1.0 Mha of organic forest soils are considered drained in Sweden (Swedish Environmental Protection Agency 2020a). In 2018, 5.4 Mt CO_2 was released from drained organic forest soil, as well as 9.07 kt CH_4 and 3.59 kt N_2O (Swedish Environmental Protection Agency 2020a). This equals a total of 6.6 Mt CO_2-eq. GHG emitted from drained organic forest soil in that year.

1.1.1. Factors influencing GHG emissions from drained organic soils

Many factors, such as drainage parameters, nutrient status, land use and climate, can influence the magnitude of GHG emissions from drained organic forest soils.

Fluctuations in water table depth (WTD) causes varying oxic or anoxic conditions and thus lead to a shift in GHG production. Couwenberg *et al.* (2011) found WTD to be a good proxy for GHG fluxes in a meta-analysis of data from European temperate peatlands. However, the strength and nature of the relationship of GHG emissions and WTD differ between CO_2, CH_4 and N_2O.

Several studies have found a positive relationship between WTD and CO_2 emissions. Ojanen & Minkkinen (2019) found that net soil CO_2 emissions increased linearly with WTD down to 60 cm in boreal sites drained for forestry in Finland. Karu *et al.* (2014) also found that emissions of CO_2 were strongly dependent on WTD. However, there was a high variability of emissions in drained sites, due to large fluctuations in WTD. The relationship between emissions and WTD may also vary with depth. Kritzler *et al.* (2016) found a more variable relationship at WTD between 15 and 20 cm.

There is no clear evidence on how WTD affects CH_4 emissions. Some studies have found a negative correlation between WTD and CH_4 emissions in boreal ombrotrophic bogs (Munir & Strack 2014), temperate mixotrophic bogs (Salm *et al.* 2012) and temperate ombrotrophic bogs (Karu *et al.* 2014).

Other studies have found none or weak relationships between WTD and CH_4. Olson *et al.* (2013) found only a weak relationship between WTD and CH_4 emissions in a temperate nutrient-poor fen. Rinne *et al.* (2018) found no significant relationship between WTD and CH_4 at WTD between -40–20 cm. Some studies have found non-linear relationships between CH_4 emissions and WTD. For example, Brown *et al.* (2014) found that CH_4 emissions were not the largest at WTD 0 cm, but rather at WTD 40–55 cm in an ombrotrophic bog.

The link between WTD and N_2O emission is unclear, with most studies finding little to no relation between the two. Mustamo *et al.* (2016) found higher emissions

of N_2O in sites with larger WTD, levelling off at about 60 cm. Laine *et al.* (2019) found N_2O emissions to be higher in drained than undrained sites, however WTD did not explain N_2O emissions. Pärn *et al.* (2018) found that drainage increases N_2O emissions due to fluctuations around an intermediate soil moisture of 50%.

Two major factors which have been found to influence emissions of GHG from drained organic soils are climate and nutrient status. This is emphasized by the fact that the IPCC in its 2013 Wetlands Guidelines (WL GL), the guideline used in the GHG inventory, uses these parameters to stratify the emission factors rather than drainage status parameters such as WTD or soil moisture (IPCC 2014). The primary factor related to climate, which affects emissions, is soil temperature. Uri *et al.* (2017) found soil temperature to be the dominant driver of CO_2 emission from drained organic forest soils. Soil temperature has also been identified as a more important factor than WTD influencing the size of CH_4 emissions (Olson *et al.* 2013).

For N_2O, nutrient status seems to be a highly important factor determining the size of emissions. One of the parameters that can be used to indicate nutrient status is C:N ratio. Klemedtsson *et al.* (2005) found that a low C:N ratio strongly favours N_2O emissions, but only below C:N 25. Above 25, the emissions were nearly non-existent. Mu *et al.* (2014) found a similar exponential relationship as Klemedtsson *et al*. (2005) between N_2O and C:N ratio, with high emissions at low C:N.

The time that has passed since drainage also affects emissions. Munir & Strack (2014) found that CH_4 emissions were reduced by 50% three years after drainage and by 76% thirteen years after drainage. The time after drainage is also indirectly linked to GHG emissions through long-term changes in vegetation community composition (Straková *et al.* 2012).

Both tree species composition (Christiansen & Gundersen 2011) and field layer composition (Acosta *et al.* 2017; Creevy *et al.* 2020) have been found to affect GHG emissions. For CH_4, vegetation plays an especially important role. The presence of aerenchymatous plants which can transport CH_4 directly into the atmosphere, thereby bypassing the soil body where CH_4 is often oxidised to CO_2, increases the emission of CH_4 (Rinne *et al.* 2018). Vegetation has also been suggested as a proxy for estimating GHG emissions from drained organic soils (Couwenberg *et al.* 2011).

1.2. GHG inventories under the UNFCCC and the EU

The UNFCCC was adopted in 1992 with the ultimate objective to "achieve […] stabilization of greenhouse gas concentrations in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system" (UNFCCC article 2). Although the convention contained no legal commitment for countries to reduce GHG emissions, in 1997 it was followed by the Kyoto Protocol,

which introduced legally binding reduction targets. The first commitment period of the Kyoto Protocol lasted between 2008–2012, and contained reduction targets for 37 signatory parties. The Kyoto Protocol was extended for a second commitment period, from 2012 to 2020, through the Doha Amendment in 2012. However, only 138 parties out of the required 144 have ratified the amendment as of May 27, 2020, and as such the amendment is currently not in force (UNFCCC 2020).

The most recent international climate treaty, the Paris Agreement of 2015, replaces the Kyoto Protocol from 2020 onward. The goal of the Agreement is to keep the global temperature rise to below 2°C and to strive to keep it below 1.5°C (Article 2.1a).

Countries within the EU are party to these climate treaties both as individual countries, and as the EU as a whole. As such, each country has its own individual mitigation targets, as well as a common EU goal of 40% reduction in GHG emissions by 2030 compared to 1990. The Effort Sharing Regulation (Regulation EU 2018/842) contains reduction targets for individual member countries for the period 2021–2031, in the sectors Energy, Industrial Processes and Product Use, Agriculture and Waste. Emissions and removals from the sector Land Use, Land Use Change and Forestry (LULUCF) are regulated separately through Regulation (EU) 2018/841, where the goal for each member state is for accounted emissions to not exceed accounted removals, as estimated using activity specific accounting rules on reported emissions and removals.

Under both the UNFCCC and the Kyoto Protocol – and in the future, the Paris Agreement – signatory nations are required to provide annual inventories of anthropogenic GHG emissions and removals. Annex I parties (developed countries) are required to do so through an annual National Inventory Report (NIR), with a certain level of quality. Seven direct greenhouse gases – CO_2, CH_4, N_2O, HFC, PFC, SF_6 and NF_3 – and four indirect gases – NO_x, CO, NMVOC and SO_2 – are reported in five sectors: Energy, Industrial Processes and Product Use, Agriculture, LULUCF, and Waste. All sectors and gases are reported as time series starting in 1990.

Drained organic soils are included in the LULUCF sector. The land use categories within LULUCF are Forest land, Cropland, Grassland, Settlements and Wetlands, where peat extraction is included. These categories are further subdivided based on soil type, specific land use and greenhouse gas. CO_2, CH_4 and N_2O emissions from drained organic forest soils are included in the Forest land category. In the Swedish GHG inventory, net CO_2 emissions from drained organic soils are reported under the subcategory “Forest land”, while CH_4 and N_2O are reported under the subcategory “Drained organic soils” (Swedish Environmental Protection Agency 2020a).

1.2.1. The Swedish UNFCCC reporting

The latest Swedish NIR from 2020 contains sources and sinks of GHGs up until the year 2018 (Swedish Environmental Protection Agency 2020a). While the Swedish LULUCF sector as a whole acts as a net sink for GHGs, removing 42 Mt CO_2-eq. in 2018, drained organic forest soils are one of the biggest sources in the sector, contributing with emissions of 6.6 Mt CO_2-eq. (Swedish Environmental Protection Agency 2020a).

In the NIR, soils are considered organic in accordance with the FAO classification "Histosol", i.e. if the organic horizon is >40 cm (Swedish Environmental Protection Agency 2020a). Forest is defined according to the FAO definition, i.e. a crown cover of 10% and a height of 5 m at maturity. Exceptions include forest roads, which are not considered forest, and there is no minimum width requirement (Swedish Environmental Protection Agency 2020a).

In the UNFCCC reporting, the methodologies with which emissions and removals are estimated are classified into three "Tiers" of increasing complexity (IPCC 2006). The current Swedish inventory of GHG emissions from drained organic forest soil follows the simplest Tier 1 methodology, where activity data, i.e. the total area of a particular land use category, is multiplied by an emission factor (EF) (Swedish Environmental Protection Agency 2020a). CO_2 and N_2O are both calculated in this manner (Eq. 1 and 2), whereas CH_4 emissions also include the fraction of ditches multiplied by a specific emission factor for the ditches (Eq. 3).

$$CO_2 = \sum_{c,n} A_{c,n} \cdot EF_{c,n} \cdot 44/12 \quad (1)$$

$$N_2O = \sum_{c,n} A_{c,n} \cdot EF_{c,n} \cdot 44/28 \quad (2)$$

$$CH_4 = \sum_{c,n} A_{c,n}((1 - Frac) \cdot EF_{land_{c,n}} + Frac \cdot EF_{ditch_{c,n}}) \quad (3)$$

In the Tier 1 methodology, these emission factors are default values provided in the 2013 IPCC Wetlands Guidelines (WL GL) (IPCC 2014). The emission factors are stratified into classes based on climate and nutrient status (Table 1). In eq. 1 to 3, *c* and *n* denote climate and nutrient status, of which there are four possible combinations. The factors 44/12 (Eq. 1) and 44/28 (Eq. 2) convert the values from CO_2-C to CO_2 and N_2O-N to N_2O. The emission factor for methane is already in the form of CH_4. The emission factors for CO_2, reflecting the carbon balance of the soil, are calculated as the net change of soil organic carbon and below-ground litter carbon (IPCC 2014). In addition to the emissions from the soil, dissolved organic carbon (DOC) is also estimated for drained organic soils using a default emission factor for all soils of 0.12 t CO_2-C ha^{-1}.

Table 1. Emission factors for drained organic forest soil used within the Swedish GHG inventory (Swedish Environmental Protection Agency 2020b).

Climate	Nutrient status	CO_2-C (t ha^{-1} yr^{-1})	CH_4 (kg ha^{-1} yr^{-1})	CH_4 ditch[a] (kg ha^{-1} yr^{-1})	N_2O-N (kg ha^{-1} yr^{-1})
Boreal	Rich	0.93	2.0	5.4	3.2
	Poor	0.25	7.0	5.4	0.22
Temperate	Rich	2.6	2.5	5.4	2.8
	Poor	2.6	2.5	5.4	2.8

[a]The factor for CH_4 ditch is based on the WL GL emission factor for forest ditches of 217 kg CH_4 ha^{-1} yr^{-1} (EF_{ditch}, eq. 3), multiplied by the indicative fraction of ditches of 0.025 (*Frac*, eq. 3), also given by the WL GL (IPCC 2014).

The climate classes in Sweden consist of boreal in the north (i.e. north of the River Dalälven) and temperate in the south. The nutrient status is classified as either rich or poor. Within the IPCC guidelines, ombrogenic peat is considered nutrient-poor and minerogenic peat is considered nutrient-rich (IPCC 2006). In Sweden, the classification of soils by nutrient status is based on information in the Swedish National Forest Inventory (NFI). However, the NFI does not carry information about whether the peat is oligotrophic or minerotrophic and as such, vegetation is used as a proxy (Table 2) (Swedish Environmental Protection Agency 2020b). The C:N ratio has also been suggested as a possible parameter by which to define nutrient status, however not all NFI plots are assessed for C:N ratio (Lindgren & Lundblad 2014).

Table 2. Nutrient categorization of the vegetation classes within the NFI (Swedish Environmental Protection Agency, 2020)

Nutrient-rich	Nutrient-poor
01 – Tall herbs without shrubs	10 – Tall *Carex*
02 – Tall herbs with shrubs/blueberry	11 – Low *Carex*
03 – Tall herbs with shrubs/lingonberry	14 – Lingonberry
04 – Low herbs without shrubs	15 – Crowberry/*Calluna*
05 – Low herbs with shrubs/blueberry	16 – Poor shrubs
06 – Low herbs with shrubs/lingonberry	
07 – Without field layer (no plants, just mosses)	
08 – Broad grasses	
09 – Narrow grasses	
12 – Horsetail	
13 – Blueberry	

To improve the GHG inventories, countries are encouraged to use higher tier (Tier 2 and 3) methods when calculating emissions. The Tier 2 method uses the same type of calculation as in Tier 1, but with national emission factors developed specifically for the country itself. Tier 3 methodologies are the most complex and

are based on modelling (IPCC 2006). Lindgren & Lundblad (2014) assessed the possible adoption of national emission factors based on both the studies used within the WL GL, as well as other studies deemed reflective of Swedish conditions, but ultimately recommended the WL GL emission factors to be used for the GHG inventory, as they were more robust.

Countries are further obliged to define so-called *key categories* of sinks and sources. Key categories are categories that contribute significantly to a country's emissions or removals, based on absolute levels, trends or uncertainty. It is recommended to use higher tier methods (Tier 2 or 3) for most, but not all, key categories, including emissions from drained organic forest soils (IPCC 2006). Within the Swedish GHG inventory, both CO_2 from forest land remaining forest land, and CH_4 and N_2O emissions from drained organic soil are considered key categories (Swedish Environmental Protection Agency 2020a). As mentioned above, the current GHG inventory uses a Tier 1 methodology for all emissions from drained organic forest soils.

Further complicating the effort to achieve accurate GHG estimations is the fact that the current method of classification of soils as drained is rather vague. In the NFI, a soil is assumed to be drained within 25 m distance from a ditch (SLU 2020). Under this definition, roughly 1.0 Mha of organic soil is estimated as drained for forestry (Swedish Environmental Protection Agency 2020). However, this corresponds to only roughly half of the total area of productive forest on organic soil in Sweden, meaning that a much larger area is likely affected by drainage, either by a functioning ditch farther than 25 m from the inventory plot, or through tree transpiration (Lundblad *et al.* 2016). Figure 1 illustrates the discrepancy between the area that is considered drained in the GHG inventory (i.e. within 25 m of a functioning ditch) and the total area of productive forest on organic soil in Sweden.

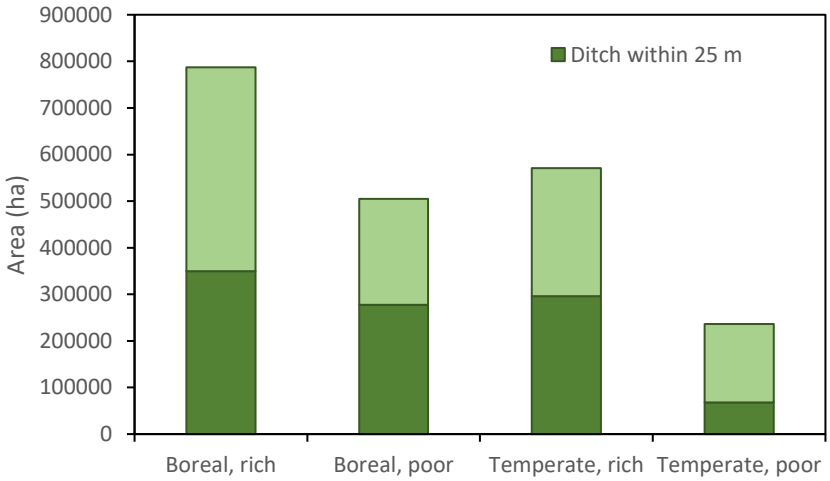

Figure 1. Total areas of drained organic forest soils in Sweden, categorised by climate and nutrient status. Based on data from Lundblad et al. (2016).

An alternative estimation of drainage status could be based on soil moisture classes within the NFI (Lindgren & Lundblad 2014). There are five moisture classes within the NFI: wet, moist, mesic-moist, mesic and dry, of which the latter three are considered well drained (Lindgren & Lundblad 2014). The assignation of moisture classes is based on an estimate of average WTD during the growing season (fig. 1). Several indicators are used to assess the average WTD, including observations of landscape morphology, vegetation or soil type (SLU 2020).

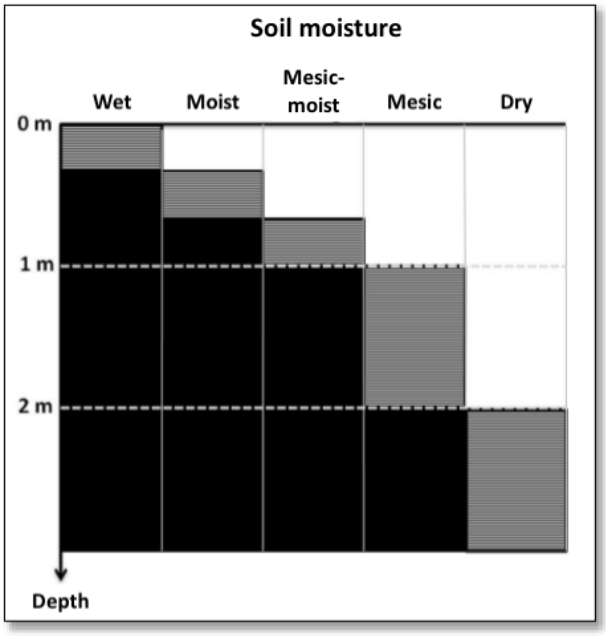

Figure 2. Schematic representation of soil moisture classes in the NFI (SLU 2019). The hatched areas represent expected average WTD intervals. Translated from Swedish.

These estimations of drainage status – distance from ditch or by moisture class – are likely to result in uncertainties in the estimations of GHG emissions (Lindgren & Lundblad 2014). It is estimated that the uncertainty in the area estimation of drained organic forest soil is 25%, and further, that the uncertainty in the emission factors is 40% for CO_2 and over 100% for CH_4 and N_2O (Swedish Environmental Protection Agency 2020a). Due to the steadily rising importance of accurate accounting of anthropogenic GHG emissions and removals, it is vital to improve both the accuracy of the area estimation, as well as the methodology with which emissions are calculated.

1.3. Aim

The aim of this project is to assess GHG emissions from drained organic forest soils in relation to its drainage status and to suggest a method for adjusting the emission factors with additional parameters than climate and nutrient status. The specific research questions are:

- How does GHG emission data from recently published literature compare with the default emission factors in the WL GL?
- How are GHG emissions effected by the distance from a ditch?
- Can drainage status parameters such as WTD or soil moisture be used to estimate GHG emissions?

The first question will be answered through a literature review and the second and third question will be answered through a field experiment.

2. Methods and materials

2.1. Literature review

Default emission factors in the WL GL were compared to values found in recent publications, i.e. published after 2010. Original peer-reviewed studies on GHG emissions from drained organic forest soils were searched for using Web of Science, Scopus, Google Scholar and the SLU library database. The specific search terms used are shown in Table 3. Studies were selected based on the following criteria:

- The study is published within the last ten years and not used to develop the WL GL default emission factors.
- The study site is located in a boreal or cool temperate (hemi-boreal) climate (i.e. reasonably similar to Sweden).
- The nutrient status of the site is clearly stated in the study, or the C:N ratio is given. If only C:N ratio is available, C:N <25 is assumed to be nutrient-rich and C:N > 25 nutrient-poor (Ernfors *et al.* 2007).
- The emission data have already been annualised.

Table 3. Search terms used in this review. The asterisk indicates a truncated search term.

peat*	“greenhouse gas*”	drain*	boreal
“organic soil*”	GHG	ditch*	temperate
forest*	“carbon dioxide”		
	CO2		
	methane		
	CH4		
	“nitrous oxide”		
	N2O		

Annual values found in the studies were averaged to single average emission values for this review. If a study contained more than one site, values for all sites were included in the final average. This is different from the IPCC approach in the WL GL, where all emission values are averaged per study and only one value per study is included in the final emission factor.

The default emission factors in the WL GL represent heterotrophic respiration of CO_2 only and do not include autotrophic respiration. If a study reported heterotrophic respiration, that value was chosen. If a study reported only forest floor or ecosystem respiration, an assumption that 50% of the total respiration was due to heterotrophic respiration (von Arnold *et al.* 2005) was used in the calculation of the emission factors.

In addition to detracting autotrophic respiration, the WL GL emission factors for CO_2 are adjusted for inputs of carbon from litterfall and root mortality (IPCC 2014). In this study, values for litterfall were taken from each study and subtracted from the heterotrophic respiration. If a study did not report litterfall, an average value of litterfall from the studies of the same nutrient status and climate was applied. Average annual root mortality values were estimated based on a limited number of studies for both boreal (Finér et al. 2011; Gaudinski et al. 2010; Minkkinen et al. 2018) and temperate (Finér et al. 2011; Gaudinski et al. 2010; Riley et al. 2009; Uri et al. 2018) forests and detracted from the respiration as well, resulting in the final emission factors.

2.2. Field experiment

2.2.1. Study site

The study site is located in the Norunda forest (60°05’43” N, 17°29’01” E), ca. 30 km north of Uppsala, Sweden. The average annual temperature and precipitation are 5.6 °C and 544 mm respectively (ICOS Sweden 2020). The site is situated within the source area of the Norunda forest Ecosystem and Atmosphere station, an International Carbon Observation System (ICOS) station which was established in 1994. Measurements at the station include GHG, energy and water exchange, both on the ground and at different heights from a 102 m tall tower. The study site is located on the northernmost edge of the 1 km radius of the tower.

The site was drained for forestry 125 years ago and is crosscut by a large ditch. The main tree species are Scots pine (*Pinus sylvestris* L.), with some Norway spruce (*Picea abies* (L.) H. Karst.) and birch. The vegetation on the south side of the large ditch is dominated by dwarf shrubs (mainly *Vaccinium myrtillus)*. On the north side, the vegetation is mainly comprised of marsh Labrador tea (*Rhododendron tomentosum*) and *V. myrtillus*.

Transects were established perpendicularly on both sides of the large ditch, two transects (A and B) on the south side and one long transect (C) on the north side between the large ditch and a smaller, poorly maintained ditch (fig. 3). The transects were designed to stretch from the ditch and beyond the 25 m cut-off used in the GHG inventory to classify a soil as drained. Transects A and B were 52.5 m long with six measurement plots each, and transect C was 105 m long with 11 plots.

The southern edge of the large ditch, as well as along the small northerly ditch, has a "barrier" of Norway spruce (*Picea abies* (L.) H. Karst.), affecting the first two points of transects A and B (A1, A2, B1 and B2) and the last two points of transect C (C10 and C11).

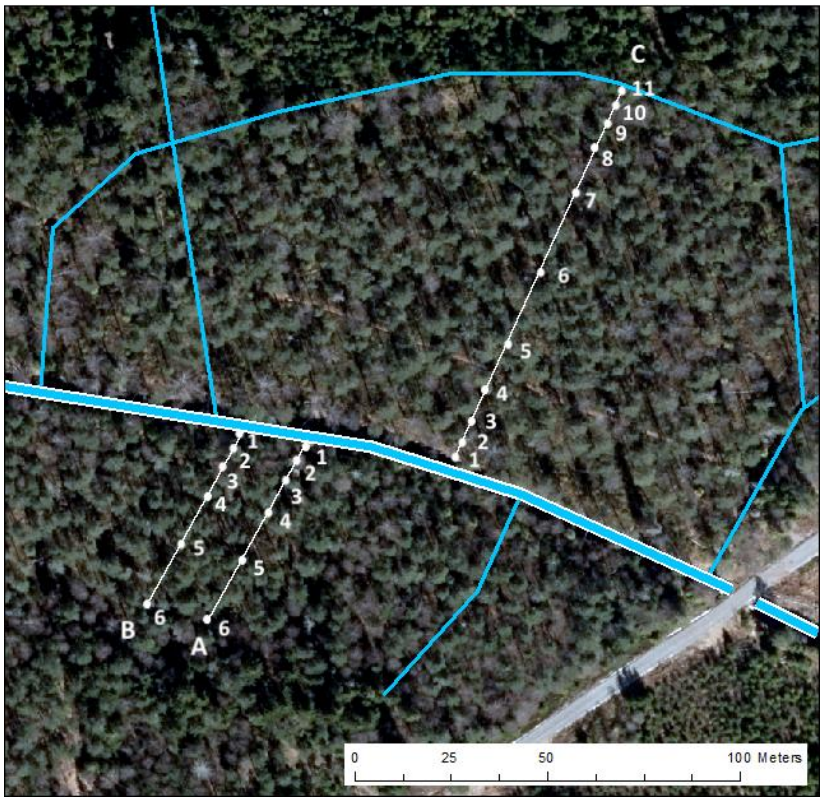

Figure 3. Overview of the site with transects marked in white and ditches in blue. Orthophoto © The Swedish Mapping, Cadastral and Land Registration Authority.

Soil sampling to determine soil characteristics was carried out horizon-wise. Since the peat type and degree of peat decomposition were rather similar over the entire study site, only a general description of the soil characteristics is given. Below the litter horizon (3–5 cm thick), earthified peat (H10 in the von Post scale) was found between 0–10 cm. The deeper peat horizons were identified as mixed peat consisting of low to medium decomposed fine *Carex* peat with *Menyanthes*

trifoliata, *Bryales* and *Sphagnum* peat, *Eriophorum* peat with some *Equisetum* and wood peat (*Betula* and *Alnus*).

Peat depth varies between 0.7 and >4 m, with greater depths in the middle of the transects and smaller depths closest to both ditches and at the end of transects A and B (fig. 4). The average depth in the entire site is 2.5 m. The plots closest to the ditch on both sides (A1, B1 and C1) are at a substantially lower elevation than the rest of the transect plots due to the slope of the ditch banks.

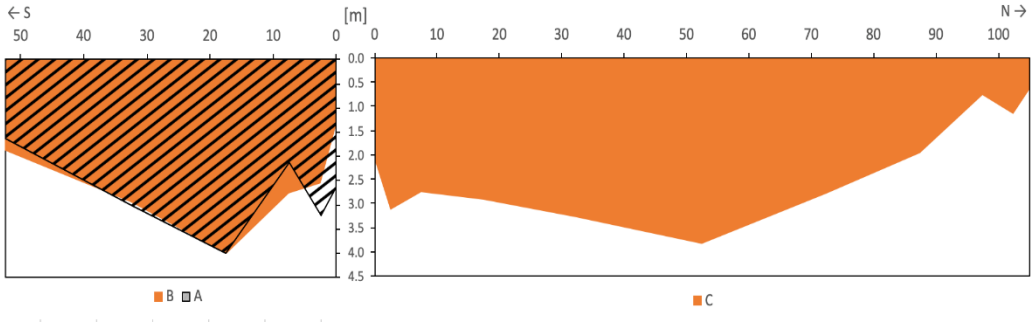

Figure 4. Peat depth across all three transects.

The soil pH, organic content (determined via loss on ignition, LOI%) and C:N ratio measured at each plot can be seen in Table 4. pH was measured in deionised water (volume to volume ratio of 1:5) and total C and N determined measured through dry combustion in a C:N-analyser (TruMac CN, LECO Corp., USA). The average soil pH of the entire site is 4.1 in the top 0–10 cm and 3.9 at 30–40 cm. The C:N ratio of the site is indicative of nutrient-poor conditions at both 0–10 cm and 30–40 cm. The organic matter content varies between plots, with the highest values in the middle and end of transects A and B, and middle of transect C. The first two plots counting from the ditch of transect B stand out with much lower organic matter content, plot B2 especially only having 6.0% of organic matter in the top 0–10 cm. This is presumably due to mineral material being mixed with the peat and dumped on top of the soil surface along the ditch during the ditching.

Table 4. Soil pH, organic matter content (LOI%) and C:N ratio at 0–10 cm and 30–40 cm depth.

Transect		0–10 cm				30–40 cm			
		Mean	Median	Min	Max	Mean	Median	Min	Max
A	pH	4.14	4.16	4.00	4.21	3.94	3.99	3.74	4.02
	LOI%	86.9	95.6	52.3	96.8	94.6	95.6	90.2	97.7
	C:N	29.3	29.0	22.3	36.6	41.8	42.6	31.8	49.9
B	pH	4.27	4.15	4.00	4.78	3.87	3.88	3.62	4.11
	LOI%	72.8	96.9	6.0	97.7	92.2	98.3	63.5	98.9
	C:N	30.2	31.0	22.7	40.7	45.0	45.4	28.6	58.0
C	pH	4.07	4.12	3.71	4.30	3.86	3.88	3.69	4.07
	LOI%	92.5	93.4	79.0	97.7	93.1	96.0	73.5	98.1

	C:N	29.2	29.2	24.9	34.2	36.0	34.2	28.1	49.5
Site	pH	4.14	4.14	3.71	4.78	3.89	3.62	4.11	3.89
	LOI%	85.9	95.7	6.0	97.7	93.2	96.0	63.5	98.9
	C:N	29.5	29.6	22.3	40.7	39.9	39.5	28.1	58.0

2.2.2. GHG measurements

Fluxes of CO_2, CH_4 and N_2O were measured using dark manual flux chambers. Permanent cylindrical PVC collars (d=18.7 cm) were inserted into the soil at distances of 0, 2.5, 7.5, 17.5, 32.5 and 52.5 m from the ditch along the transects (2.5, 5, 10, 15 and 20 m between plots), see figure 3. Vegetation was not removed from the plots.

Forest floor respiration (CO_2) was measured using a manual opaque PVC chamber with a volume of 4.3 L and equipped with a fan. The CO_2 concentration was measured with a portable infrared CO_2-analyser (EGM-4, PP Systems Inc., USA) over a measurement period of 180 s. Weekly flux measurements were conducted from mid-March to the end of April 2020. In total, CO_2 was measured during six measurement occasions.

CH_4 and N_2O samples were taken from the chamber headspace in vials via cross-flow with an external membrane pump (volume flow rate 0.4 L min^{-1}) (Jordan *et al.* 2020). Weekly measurements were conducted on four occasions in April 2020. Gas samples were extracted 0, 10, 20 and 30 min after chamber closure on the south side (A & B), and after 0, 13, 26 and 39 min on the north side (C). Additional measurements were performed in the large ditch using a chamber fitted with a buoy at the start of each transect. The gas samples were then analysed in the lab using gas chromatography.

2.2.3. WTD & soil moisture

Dip wells consisting of perforated PVC tubes (d=40 mm) were installed at each measurement point. Water table depth (WTD) was measured manually using a water level sounder at the time of each GHG measurement. Volumetric soil moisture content in the topsoil was also measured during each measurement occasion with a HH2 Moisture Meter and ThetaProbe type ML2 (Delta-T Devices Ltd., UK).

2.2.4. GHG flux calculations and statistical analysis

Fluxes of CO_2, CH_4 and N_2O were calculated using equation 4. Linear regression was used to estimate the change in concentration with time in the chamber headspace (*f*).

$$F = f \cdot \frac{P \cdot V}{R \cdot T} \cdot \frac{1}{A} \cdot m \quad (4)$$

F is the gas flux. *P* is the air pressure inside the chamber headspace, *V* is the headspace volume, *R* is the ideal gas constant and *T* is the mean temperature inside the chamber during the measurement period. The flux was estimated per area *A* (soil surface area inside the collar) and converted to units of mass through *m* (molecular weight). All CO_2 measurement data were included in the calculations except the sixth and final CO_2 measurement, as the measurements were disturbed due to a faulty fan.

The minimum detectable fluxes (MDF) of CH_4 were 24 µg CH_4 m^{-2} h^{-1} on the south side of the ditch and 18 µg CH_4 m^{-2} h^{-1} on the north side. MDF of N_2O were 15 µg N_2O m^{-2} h^{-1} on the south side of the ditch and 12 µg N_2O m^{-2} h^{-1} on the north side. These MDF were estimated for linear least-square regression of a chamber air concentration time series determined with GC analysis, and for a confidence of 95%. No measured CH_4 fluxes fell below the MDF and all measurement data were therefore included in further statistical analyses. About 10% of the measured N_2O data fell below the MDF and were excluded from further analysis.

In order to determine if there was a significant difference between gas fluxes <25 m from the ditch and gas fluxes >25 m from the ditch, a two-sample unequal variances t-test (Welch's t-test) was carried out for all three gases. Measurements at 0, 2.5, 7.5 and 17.5 were grouped together as <25 m, and measurements at 32.5 and 52.5 m were grouped as >25 m. For the sake of this analysis, transect C was considered as two transects with starting point 0 m at each of the two ditches and ending at the same point in the middle.

3. Results

3.1. Literature review

CO_2

The emission factors for CO_2 found in this review differ from those in the WL GL (Table 5). The emission factor for boreal+rich (2.5 t CO_2-C ha^{-1} yr^{-1}) is remarkably high in comparison to the default emission factor (0.93 t CO_2-C ha^{-1} yr^{-1}), and also fall far outside the 95% confidence interval of the WL GL. Boreal+poor has a negative value (–0.2 t CO_2-C ha^{-1} yr^{-1}) but does fall within the WL GL 95% confidence interval. Two different values for CO_2 emissions under temperate conditions were found in this review, in contrast with the one common emission factor in the WL GL. The value for temperate+rich (1.7 t CO_2-C ha^{-1} yr^{-1}) is lower than in the WL GL, but falls does fall within its 95% confidence interval. The value for temperate+poor (0.1 t CO_2-C ha^{-1} yr^{-1}) is much lower than the WL GL value.

Table 5. EF for CO_2 (t CO_2-C ha^{-1} yr^{-1}) found in this study and from the WL GL (IPCC 2014). n = number of sites.

Climate	Nutrient status	This study		WL GL	
		EF[a]	n	EF[b]	n
Boreal	Rich	2.5 (-0.3 – 4.0)[c]	4	0.93 (0.54 – 1.3)	62
	Poor	-0.2 (-2.8 – 2.4)[d]	2	0.25 (-0.23 – 0.73)	59
Temperate	Rich	1.7 (0.7 – 2.4)[e]	3	2.6 (2.0 – 3.3)	8
	Poor	0.1[f]	6	2.6 (2.0 – 3.3)	8

[a] (range).

[b] (95% confidence interval).

[c] Väisänen *et al*. 2013, Moilanen *et al*. 2012.

[d] Väisänen *et al*. 2013, Minkkinen *et al*. 2018.

[e] He *et al*. 2016, Uri *et al*. 2017, Kasimir *et al*. 2018.

[f] Salm *et al*. 2012.

CH_4

The emission factors for boreal+rich (18 kg CH_4 ha^{-1} yr^{-1}) and boreal+poor (16 kg CH_4 ha^{-1} yr^{-1}) are much higher than the corresponding emission factors within the WL GL (2.0 and 7.0 kg CH_4 ha^{-1} yr^{-1} respectively) (Table 6). However, the range of average emission values found in the literature is also very large, in both types. For boreal+poor, the highest and lowest value found differ by two orders of magnitude. For temperate+rich only negative emissions, i.e. consumption, were found resulting in an average value of -2.6 kg CH_4 ha^{-1} yr^{-1}. The temperate+poor value was remarkably high, 32 kg CH_4 ha^{-1} yr^{-1}, it is however based on only one study. Same as with CO_2, the WL GL provides only one emission factor for CH_4 for both temperate+rich and temperate+poor soils.

Table 6. EF for CH_4 (kg CH_4 ha^{-1} yr^{-1}) found in this study and from the WL GL (IPCC 2014). n = number of sites.

Climate	Nutrient status	This study		WL GL	
		EF[a]	n	EF[b]	n
Boreal	Rich	18 (-2.2 – 63)[c]	8	2.0 (-1.6 – 5.5)	62
	Poor	16 (0.30 – 36)[d]	3	7.0 (2.9 – 11)	59
Temperate	Rich	-2.6 (-4.4 – -0.71)[e]	2	2.5 (-0.60 – 5.7)	8
	Poor	32[f]	6	2.5 (-0.60 – 5.7)	8

[a] (range).

[b] (95% confidence interval).

[c] Väisänen *et al*. 2013, Maljanen *et al*. 2014, Korkiakoski *et al*. 2017.

[d] Väisänen *et al*. 2013, Maljanen *et al*. 2014, Mustamo *et al*. 2016.

[e] Meyer *et al*. 2013, Kasimir *et al*. 2018.

[f] Salm *et al*. 2012.

N_2O

The emission factor for boreal+poor (3.9 kg N_2O-N ha^{-1} yr^{-1}) is much higher than that for boreal+rich (0.4 kg N_2O-N ha^{-1} yr^{-1}), which is the opposite to the default emission values of 0.22 and 3.2 kg N_2O-N ha^{-1} yr^{-1} respectively (Table 7).

The emission factor for temperate+rich (5.1 kg N_2O-N ha^{-1} yr^{-1}) is higher than the default emission factor but still within the 95% confidence interval. For temperate+poor, the emission factor (-0.01 kg N_2O-N ha^{-1} yr^{-1}) is far below that of the WL GL (2.8 kg N_2O-N ha^{-1} yr^{-1}) but is, similarly to both other gases, based on only one study. Again, there is only one emission factor for both temperate+rich and temperate+poor in the WL GL.

Table 7. EF for N_2O (kg N_2O-N ha^{-1} yr^{-1}) found in this study and from the WL GL (IPCC 2014). n = number of sites.

Climate	Nutrient status	This study		WL GL	
		EF[a]	n	EF[b]	n
Boreal	Rich	0.44 (-1.7 – 3.1)[c]	7	3.2 (1.9 – 4.5)	75
	Poor	3.9 (0.28 – 11)[d]	3	0.22 (0.15 – 0.28)	43
Temperate	Rich	5.1 (0.93 – 9.4)[e]	5	2.8 (-0.57 – 6.1)	13
	Poor	-0.01[f]	6	2.8 (-0.57 – 6.1)	13

[a] (range).

[b] (95% confidence interval).

[c] Väisänen *et al*. 2013, Maljanen *et al*. 2014.

[d] Väisänen *et al*. 2013, Maljanen *et al*. 2014, Mustamo *et al*. 2016.

[e] Meyer *et al*. 2013, Eickenscheidt *et al*. 2014, He *et al*. 2016, Kasimir *et al*. 2018.

[f] Salm *et al*. 2012.

3.2. Field experiment

3.2.1. Gas fluxes

CO_2

The average CO_2 flux in the entire site was 340±120 mg CO_2 m^{-2} h^{-1} (mean±SD). The CO_2 fluxes peaked at 7.5 m and then declined until 32.5 m from the ditch (fig. 3). Welch's t-test resulted in significantly ($p < 0.05$) lower average flux in plots >25 m from the ditch compared to in plots <25 m from the ditch. The average CO_2 flux over 25 m from the ditch was 65% of that closer than 25 m to the ditch (245±76 vs 377±114 mg CO_2 m^{-2} h^{-1}).

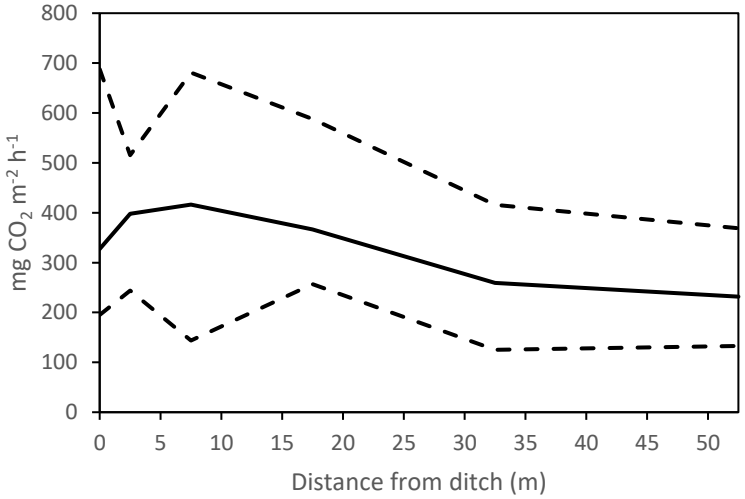

Figure 5. Average (solid), min and max (dashed) CO_2 flux in the site.

The average fluxes of CO_2 in the individual transects A and B were 327±122 and 373±141 mg CO_2 m^{-2} h^{-1}, respectively. The average fluxes in transect C were 288±78 mg CO_2 m^{-2} h^{-1} in the southern half (large ditch to mid-point) and 366±106 mg CO_2 m^{-2} h^{-1} in the northern half (small ditch to mid-point). There was a large variation in flux between plots in all transects, with average fluxes ranging between 125–498 mg CO_2 m^{-2} h^{-1} in transect A, 133–688 mg CO_2 m^{-2} h^{-1} in transect B and 144–613 mg CO_2 m^{-2} h^{-1} in transect C (fig. 6).

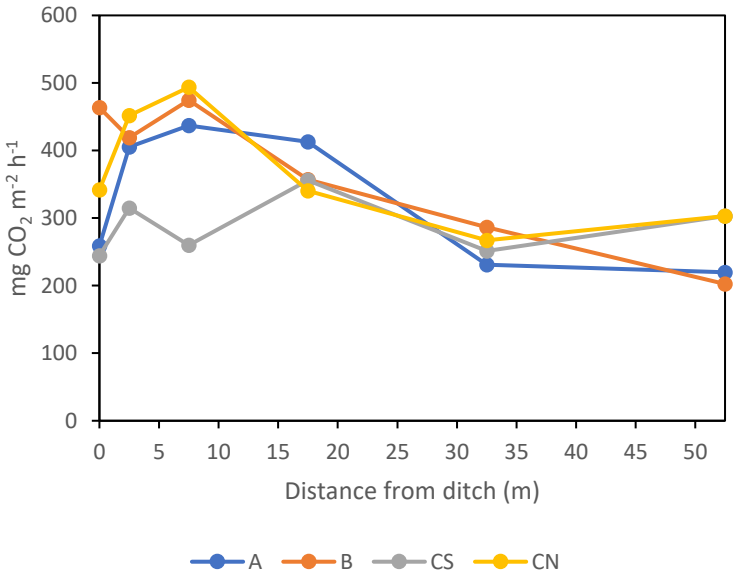

Figure 6. Average CO_2 flux in transects A, B and C. Transect C is treated as two transects, where CS = southern half of transect C (from the large ditch to the midpoint) and CN = northern half of transect C (from the small ditch to the midpoint).

While all transects experienced higher fluxes near the ditches, the response varied between transects. Transect A experienced an increase in flux in the first 0–7.5 m, a slight decrease between 7.5 and 17.5 m, a more substantial decrease in CO_2 flux between 17.5 and 32.5 m, before levelling out at above 220 mg CO_2 m^{-2} h^{-1}. Transect B showed a more continuous decrease along the entire transect, with the highest values at 0 and 7.5 m. The fluxes in transect C differed close to the two ditches. Fluxes near the large ditch varied but showed a general increase up to 17.5 m. Near the smaller ditch there was a much clearer increase and peak at 7.5 m from the ditch, similarly to transect A.

CH_4

Soil and water (ditch) CH_4 fluxes are shown in figure 7. The average rate of CH_4 flux from the soil was –543±271 µg CH_4 m^{-2} h^{-1}, indicating a consumption of CH_4. The average flux from the water (ditch) was 883±373 µg CH_4 m^{-2} h^{-1}. T-test analysis indicated a significant ($p < 0.05$) difference between the flux rates within 25 m from the ditch and flux rates over 25 m from the ditch. The average CH_4 consumption was 20% lower at >25 m than at <25 m.

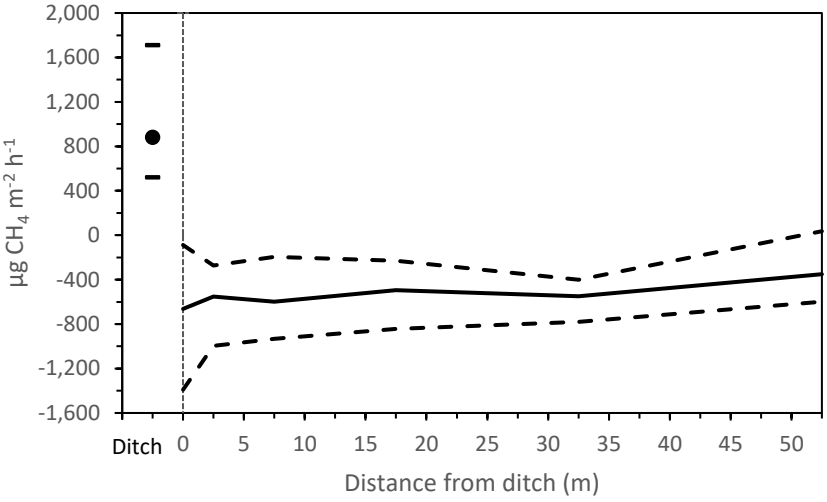

Figure 7. Average CH_4 flux from the soil (solid line) and from the water in the ditch (dot). Dashed lines indicate maximum and minimum fluxes.

CH_4 flux from the soil was negative in all transects (fig. 8). Average flux rates were -774±298 µg CH_4 m^{-2} h^{-1} and -484±262 µg CH_4 m^{-2} h^{-1} in transects A and B respectively. In transect C, the average soil flux was -359±119 µg CH_4 m^{-2} h^{-1} in the southern half and -529±168 µg CH_4 m^{-2} h^{-1} in the northern half. Average fluxes in each plot varied between -1390 and -436 µg CH_4 m^{-2} h^{-1} in transect A, –931 and 36 µg CH_4 m^{-2} h^{-1} in transect B, and -921 and -87 µg CH_4 m^{-2} h^{-1} in transect C. Similarly to the CO_2-measurements, the plots closest to the ditch seem to differ most from the rest of the transects. Transect A experienced a drastic decrease in CH_4 consumption between 0–7.5 m from the ditch, while the rates in transects B and C are more variable. Only the southern half of transect C experienced lower rates of consumption closest to the ditch.

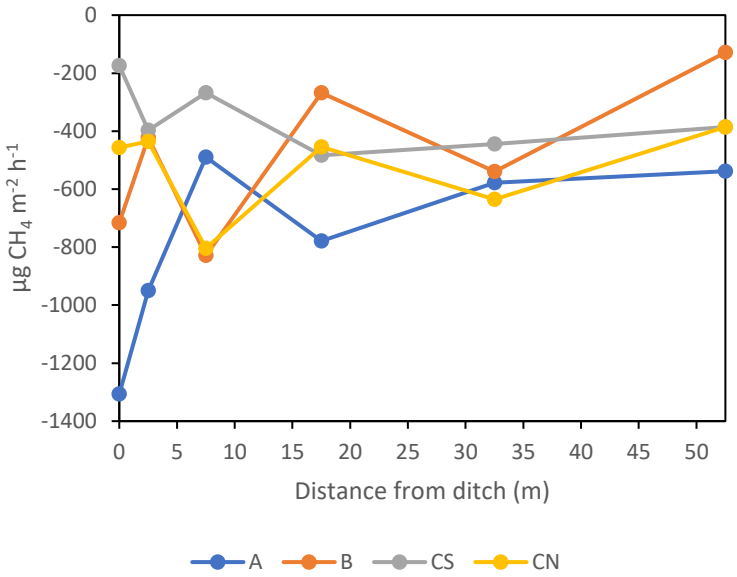

Figure 8. Average soil CH_4 flux in transects A, B and C. Transect C is treated as two transects, where CS = southern half of transect C (from the large ditch to the midpoint) and CN = northern half of transect C (from the small ditch to the midpoint).

N_2O

Overall fluxes of N_2O were very small. The average flux rates were 52±286 µg N_2O m^{-2} h^{-1} from the soil and 78±170 mg N_2O m^{-2} h^{-1} in the ditch. The average rate of N_2O flux varied little across the entire 52.5 m from the ditch (fig. 9). T-test analysis revealed no significant difference between fluxes <25 m from the ditch and >25 m from the ditch.

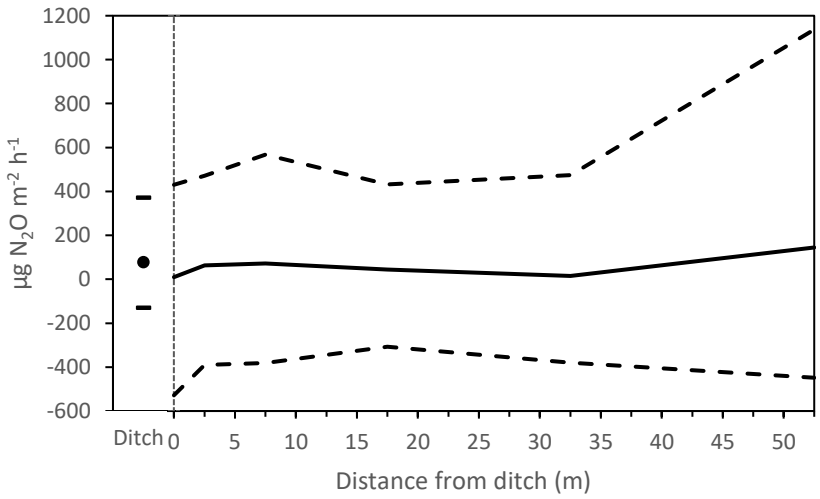

Figure 9. Average N_2O flux from the soil (solid line) and from the water in the ditch (dot). Dashed lines indicate minimum and maximum rates of fluxes.

Average rates of soil N_2O flux were 131±354 µg N_2O m^{-2} h^{-1} in transect A and 81±282 µg N_2O m^{-2} h^{-1} in transect B. In the southern half of transect C the average soil flux was 13±211 µg N_2O m^{-2} h^{-1} and in the northern half of transect C the average soil flux was -13±265 µg N_2O m^{-2} h^{-1}. The fluxes ranged from positive to negative (i.e. consumption), with average rates between -381–1137 µg N_2O m^{-2} h^{-1} in A, -528–567 µg N_2O m^{-2} h^{-1} in B and -464–518 µg N_2O m^{-2} h^{-1} in C. The individual transects show no clear pattern in fluxes with regards to distance from the ditch (fig. 10).

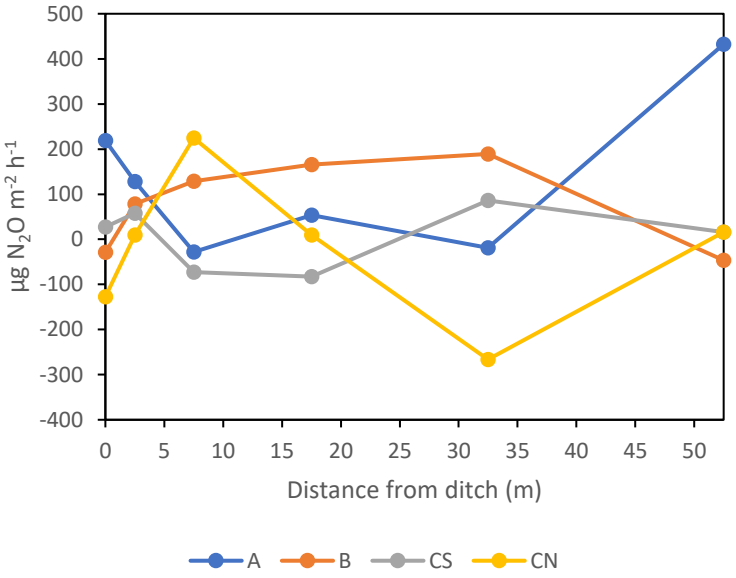

Figure 10. Average soil N_2O fluxes in transects A, B and C. Transect C is treated as two transects, where CS = southern half of transect C (from the large ditch to the midpoint) and CN = northern half of transect C (from the small ditch to the midpoint).

3.2.2. GHG fluxes in relation to drainage status

The average WTD across all transects was 50 cm, although this is largely influenced by the low WTD (86 cm) at 2.5 m from the ditch (fig. 11a). In the first point (0 m), WTD was shallower due to the lower elevation of the ditch banks, where the plots were located. In the rest of the site, the average WTD slightly decreased with distance from the ditch, from 46 to 39 cm. Overall, WTD decreased consistently from the start to the end of the measurement period by an average of 16 cm.

The soil moisture content followed a similar pattern as the WTD, with a high value close to the ditches, followed by a sharp decrease at 2.5 m and a slight increase from 7.5 m and onward (fig. 11b). The soil moisture varied between 6.0% and 69.1%, with an overall average of 34.6% in the entire site.

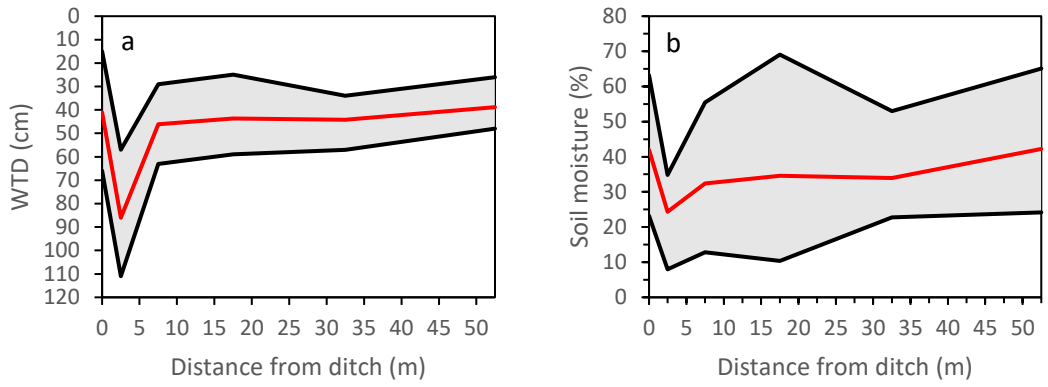

Figure 11. a) Range of WTD across all transects. b) Range of soil moisture content across all transects. Average values are indicated by the red lines.

Fluxes of all three GHGs are plotted against WTD in figures 12a-c. The emission of CO_2 shows a positive correlation with WTD. However, the correlation is very weak ($r^2 = 0.13$, $p < 0.05$). The CH_4 consumption shows a non-linear relationship with WTD ($r^2 = 0.09$, $p < 0.05$), with the highest rate of consumption around WTD 74 cm. No significant correlation between N_2O emissions and WTD could be found.

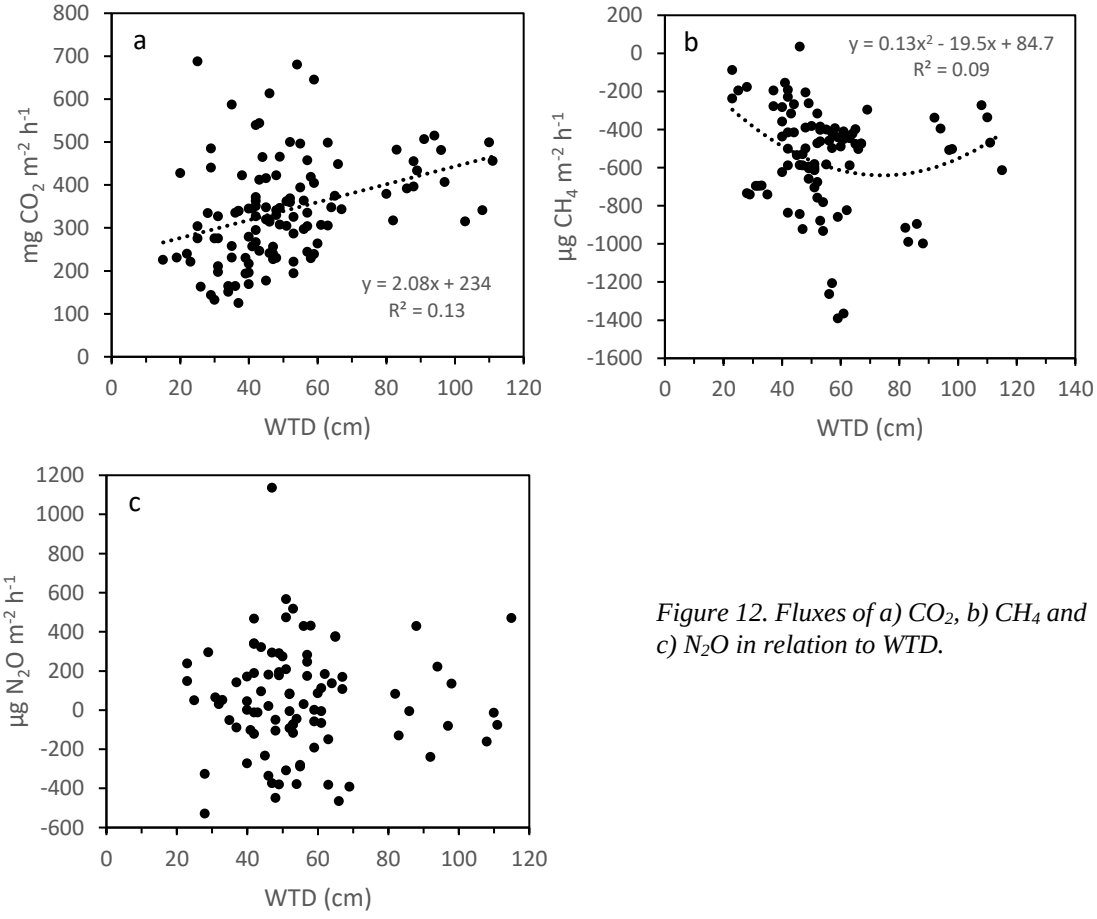

Figure 12. Fluxes of a) CO_2, b) CH_4 and c) N_2O in relation to WTD.

Gas fluxes are plotted against soil moisture content in figures 13a-c. The CO_2 emissions are weakly negatively correlated ($r^2 = 0.10$, $p < 0.05$) with soil moisture content. The rate of CH_4 consumption displays a non-linear relationship with soil moisture ($p < 0.05$, $r^2 = 0.17$), similar to WTD, with the highest (most negative) rates around 30% soil moisture content. As with WTD, no correlation could be found between N_2O emissions and soil moisture.

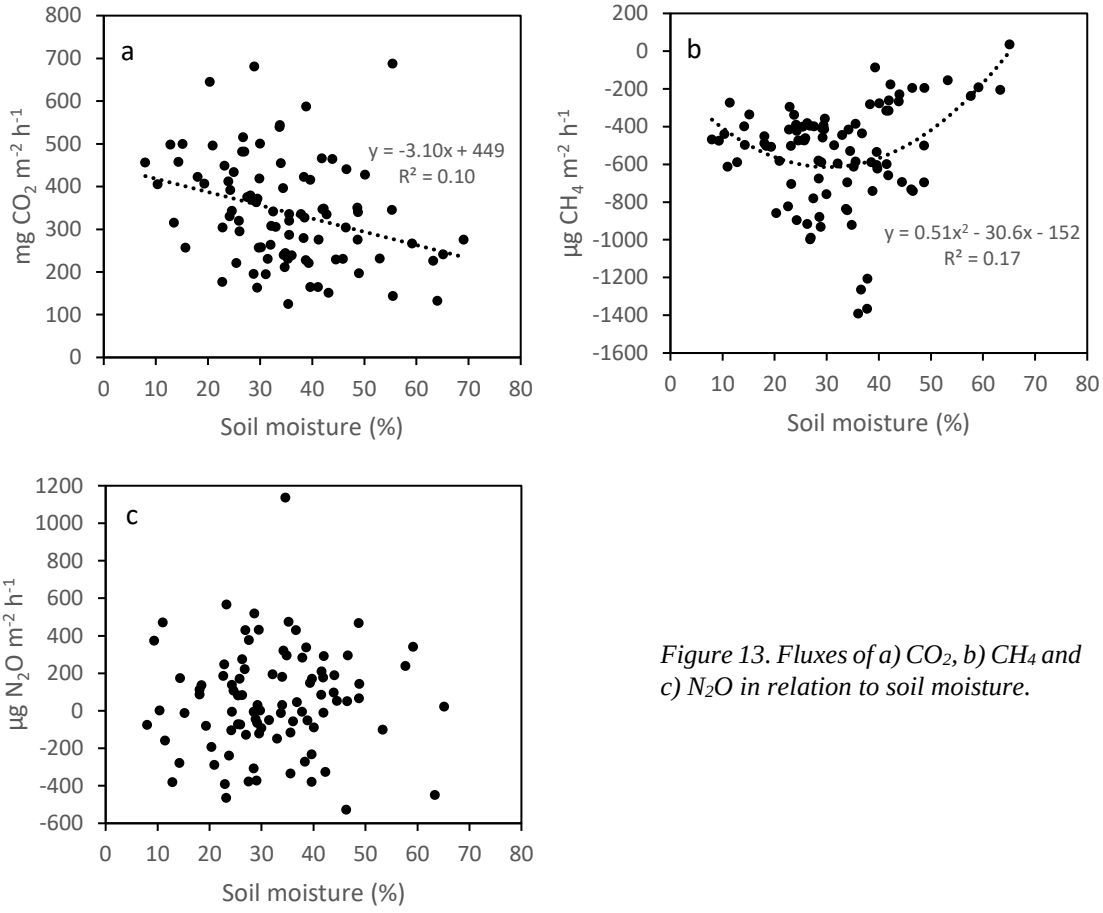

Figure 13. Fluxes of a) CO_2, b) CH_4 and c) N_2O in relation to soil moisture.

4. Discussion

4.1. Emission factors

The evaluation of emission data in recent publications shows a noticeable difference to the WL GL emission factors in most cases. Especially the EF for nutrient rich boreal soil is remarkably high in comparison with the WL GL. However, the values found in this study are more in line with the EFs used in the Finnish UNFCCC reporting. Finland uses nation-specific EFs for CO_2 that range between 1.8–4.3 t CO_2-C ha^{-1} yr^{-1} and for N_2O that range between 0.18–2.1 kg N_2O-N ha^{-1} yr^{-1} (Statistics Finland 2020). Interestingly, in the Finnish inventory, the EFs for CH_4 are divided into well drained and poorly or recently drained, with values of –2.8 and 11.6 kg CH_4 ha^{-1} yr^{-1} respectively (Statistics Finland 2020).

To better illustrate the difference in using the emission factors from the WL GL and the ones found in this study, emissions of CO_2, CH_4 and N_2O were estimated using Eq. 1, 2 and 3. Emissions of CH_4 from ditches were included using the EF of 5.4 kg CH_4 ha^{-1} yr^{-1} from the WL GL in both estimates. Area data of drained organic forest soils in Sweden from Lundblad *et al*. (2016) were used in the estimate. Total GHG emissions in CO_2-equivalents were also estimated using the global warming potential over 100 years of 28 for CH_4 and 265 for N_2O (IPCC 2013).

On land within 25 m of a functioning ditch (the criterion used in the current Swedish GHG inventory), the emission factors developed in this study would yield only slightly lower CO_2 emissions than if the WL GL emission factors are used (4.86 Mt CO_2 and 4.92 Mt CO_2 respectively) (fig. 14). The estimated CH_4 emissions would be 2.0 times as large, and estimated N_2O emissions 1.2 times as large. Total GHG emissions would be 6.5 Mt CO_2-eq. when using the EFs from this study compared to 6.1 Mt CO_2-eq. when using the WL GL EFs.

As soil within 25 m of a ditch only correspond to half of the total area of productive forest on organic soil in Sweden, emission estimates would be much larger if the GHG-balance of these areas were also assumed to be affected by drainage and included in the emission estimates. In figure 14, total emissions from both types of soils are shown. In this calculation, emission factors for CO_2 and CH_4 for areas outside of the 25 m boundary are based on the relation that was found

between <25m and >25m in the field experiment, i.e. the EF for CO_2 is 35% lower and EF for CH_4 is 20% higher. As no significant difference was found for N_2O, the original emission factors were used for both categories. Even with the default EFs, the emissions of all three GHGs would roughly double if all productive forest land on organic soil was included in the inventory and assumed to contribute according to the findings here. CH_4 emissions are especially high using the EFs found in the literature study. This shows that the omission of areas that lie beyond the 25 m cut-off but are still potentially drainage-affected may cause a substantial underestimation of GHG fluxes, regardless of which emission factors are used. It should though be noted that the calculation is based on the single ratio for drained or not drained soils developed in this study only. It is likely that there are large differences in the ratio with distance to a ditch for different organic forest soils as well as it is likely that organic forest soils without a ditch also emit GHG.

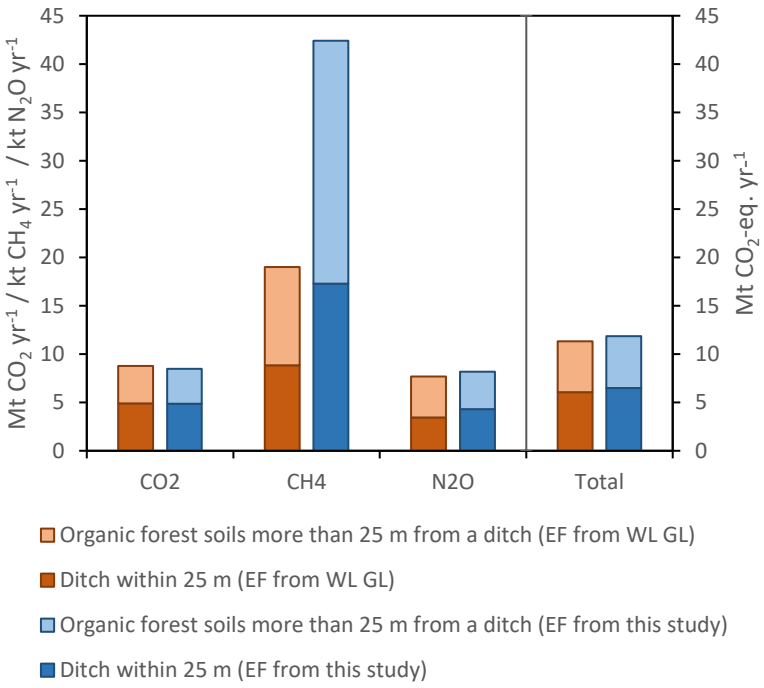

Figure 14. Estimated emissions of CO_2, CH_4, N_2O and total estimated GHG emissions in CO_2-eq. from organic forest soils drained by a ditch within 25 m and from all areas of organic soils with productive forest, using EFs from WL GL and EFs from this study respectively.

It is also important to note that while the estimated CO_2 emissions within 25 of a ditch differ only by 0.06 Mt between the EFs of this study and the WL GL, there is a large variation between the two within each of the individual climate-nutrient types. For example, the estimation for boreal+rich soil is 2.6 higher using the EFs from this study, and the estimation for temperate+poor soil is almost 40 times higher using the WL GL EFs (fig. 15a). Similarly, the emission estimates of N_2O

within each climate-nutrient type are vastly different depending on which EFs are used (fig. 15c), even though the overall emission estimates are nearly equal (fig. 14). Estimated CH_4 emissions also vary between subtypes, with the largest difference in boreal+poor soil (fig. 15b). However, as the emission estimates are higher for all climate-nutrient except temperate-rich using the EFs developed in this study, this difference is also reflected in the overall emission estimate (fig. 14). On the other hand, the similarity between the estimated CO_2 and N_2O emissions of is due to these differences balancing out between the climate-nutrient types, rather than any similarity in the emission factors themselves.

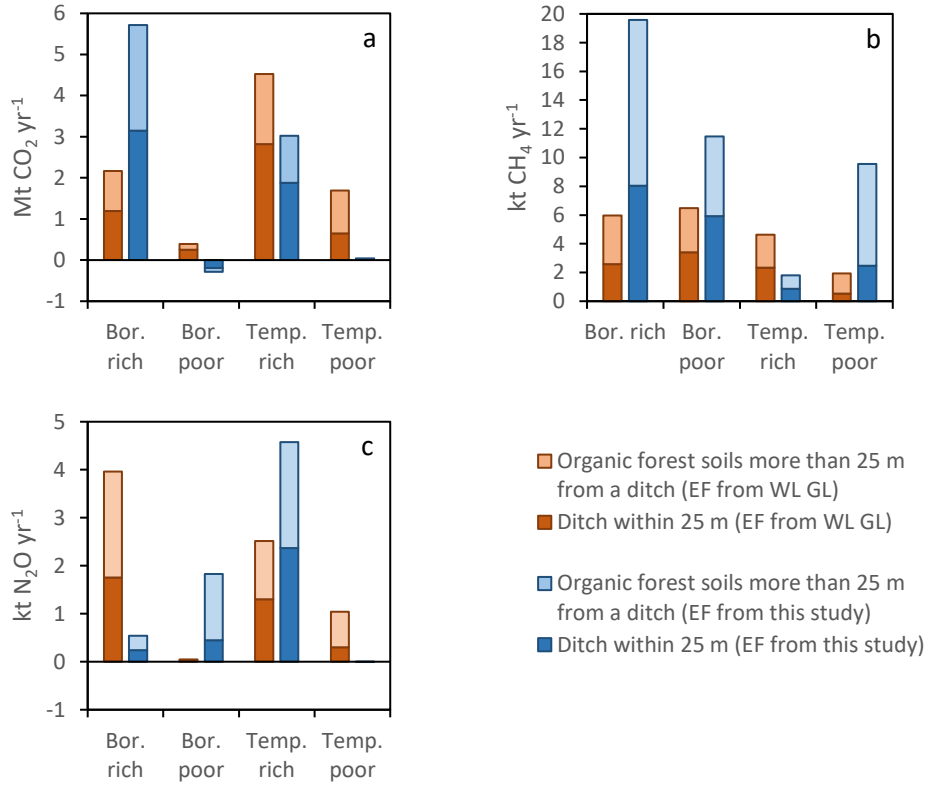

Figure 15. Estimated emissions of a) CO_2, b) CH_4 and c) N_2O from organic forest soils within each climate-nutrient type, on land drained by a ditch within 25 m and from all areas of organic soils with productive forest, using EFs from WL GL and EFs from this study respectively.

The reasons for the difference between the EFs in this study and in the WL GL may be several. A lower number of sites were included in this review than in the WL GL. Several studies that were otherwise relevant had to be rejected for this review because they did not report annualized values of GHG emissions. While annualization of seasonal emissions is possible, it was not within the scope of this study. In addition to this, several otherwise relevant studies had to be excluded from this review based on a lack of information regarding the nutrient status of the study

sites. However, with both the current emission factors, as well as the ones found in this study, it is clear that the division of sites into types based on climate and nutrient status holds up. It is interesting to note that while the WL GL only provides one emission factor for each gas for temperate climates, the values found in this review differed significantly between nutrient-rich and nutrient-poor sites.

In the literature study, a factor of 0.5 was applied to soil respiration data to account for only heterotrophic respiration, unless the authors clearly stated the actual portioning in their study. While 50% seems to be a commonly accepted apportioning of heterotrophic respiration (von Arnold *et al.* 2005), studies have found heterotrophic respiration both higher and lower. Uri *et al.* (2017) found heterotrophic respiration to be between 60–70% of the total soil respiration in temperate nutrient-rich peat. Jovani-Sancho *et al.* (2018) found values of heterotrophic respiration as low as 31–41% of the total respiration.

It is stated in the IPCC guidelines for national GHG inventories (IPCC 2006) to be *good practice* to use a higher tier methodology for key categories, which in the Swedish inventory includes GHG emissions from drained organic forest soils. This necessitates the development of at least a Tier 2 method using emission factors developed on a national basis. A similar process of compiling emission data from studies in or in locations representative of Sweden could be a viable option. A more extensive review could also include the annualisation of emission data, something which was not feasible within the time frame of this study. There were a number of potentially relevant studies that were found during the literature search wherein the emission data had not been annualised, e.g. Aguilos *et al.* 2013; Munir *et al.* 2015; Koskinen *et al.* 2016; Korkiakoski *et al.* 2019. These may be used to develop simple averaged EFs. However, there are potentially other issues surrounding the development of more complex EFs. Jauhiainen *et al.* (2019) identified a number of concerns with regards to developing more dynamic EFs, including lack of background and environmental data, cold season emission data, long-term studies of GHG emissions in relation to WTD, and others. Such concerns may make a more complex literature-based approach difficult.

4.2. Gas fluxes

Compared with studies on nutrient-poor sites located in other cool temperate regions, the average CO_2 emission in this study (340 mg CO_2 m^{-2} h^{-1}) is high. For example, Yamulki *et al*. (2013) recorded a rate of 189 mg CO_2 m^{-2} h^{-1} and Salm *et al.* (2012) a rate of 125 mg CO_2 m^{-2} h^{-1}. However, these values are based on whole-year monitoring of fluxes, whereas the values in this study were measured for one and a half month in spring. CO_2 fluxes from drained organic forest soils have been shown to have substantial interannual variability. Yamulki *et al*. (2016) found the emission of CO_2 to be 4–5 times larger in summer than in winter, and Mustamo *et*

al. (2016) found CO_2 emissions during the growing season in a boreal site to be about one order of magnitude larger than in the winter period.

As no vegetation was removed and no trenching of roots occurred in this study, the CO_2 emissions represent forest floor emissions, i.e. both heterotrophic and autotrophic respiration (Högberg *et al.* 2001). Depending on the partitioning of the respiration, this may affect the actual emission rates between plots. While 50% is a common assumption, this does not always hold true (e.g. Uri et al. 2017), and as the field layer vegetation differed between plots, this is likely the case.

Consumption of CH_4 as in this study has been recorded in other drained organic forest soils (e.g. Meyer *et al.* 2013; Koskinen *et al.* 2016; Korkiakoski *et al.* 2019). In contrast to these studies, the rate of consumption recorded here (–543 µg CH_4 m^{-2} h^{-1}) is high. However, as for CO_2 the time frame of these studies is much longer than the four weeks that CH_4 was measured in this study. Interestingly, the studies of Yamulki *et al*. (2013) and Salm *et al.* (2012), which are also both set in cool temperate, nutrient-poor sites, report emissions of CH_4 rather than consumption.

Most studies of CH_4 emissions from drained organic forest soils do not include emissions from the drainage ditches. However, the ditch emission of 883 µg CH_4 m^{-2} h^{-1} in this study is higher than that of a few other studies of ditch emissions. In boreal forestry-drained sites, rates of 327 µg CH_4 m^{-2} h^{-1} (Koskinen *et al*. 2016) and 248 µg CH_4 m^{-2} h^{-1} (Minkkinen *et al.* 2018) have been recorded. In both studies, ditch emissions exceeded those of the soil, which is consistent with the results of this study.

The average emission of N_2O (52 µg N_2O m^{-2} h^{-1}) is higher than in other comparable cool temperate, nutrient-poor sites. Salm *et al.* 2012 measured an average rate of 2.2 µg N_2O m^{-2} h^{-1} and Yamulki *et al*. (2013) measured an average rate of 9.1 µg N_2O m^{-2} h^{-1}. N_2O has been found to have a strong negative relationship with C:N ratio (Klemedtsson *et al.* 2005), meaning that low N_2O fluxes are not unexpected for this nutrient-poor site. The emission is much lower than that of another cool temperate, nutrient-rich site with an emission rate of 127 µg N_2O m^{-2} h^{-1} (Meyer *et al.* 2013).

4.3. Effect of distance to ditch and drainage status on GHG fluxes

The parts of the site that fall within 25 m of the ditch would be considered drained by the current Swedish classification system. By the NFI soil moisture classification, the WTD at over 25 m from the ditch (39–46 cm) corresponds to a "moist" soil (SLU 2020), which could be considered poorly drained (Lindgren & Lundblad, 2014). It is important to note that the NFI classification is based on

average WTD during the entire growing season, and as the measurements took place mid-spring, the WTD is likely to continue to increase during the season.

Both CO_2 and CH_4 experienced significantly different average fluxes within 25 m from the ditch, compared to emissions over 25 m from the ditch. However, the emission of CO_2 at >25 m still corresponds to 65% of that within 25 m. Likewise, the consumption of CH_4 was only 20% lower farther (>25 m) from the ditch. While there was no significant difference in N_2O emissions, meaning that the impact of excluding anything beyond 25 m is potentially even higher than for CO_2 and CH_4, the emissions were also rather low throughout the entire site, including from the ditch. See also figure 14.

Similar studies of emissions with regards to distance to the ditch are few. For example, Koskinen *et al*. (2016) measured CH_4 fluxes along transects perpendicular to a ditch similar to the transects in this study but saw no significant difference between rates at 2 m from the ditch and those farther away (although, the exact distance of the farthest points along the transects was not specified). Another study found effects on water table drawdown up to 250–320 m and effects on the vegetation up to 400 m from a ditch (Paal *et al.* 2016). While this does not mean that the same must be true for GHG emissions, they have as previously mentioned been linked to both WTD and vegetation.

The positive correlation between CO_2 and WTD is in line with results of other studies of drained organic forest soils (e.g. Karu *et al.* 2014; Ojanen & Minkkinen 2019).

A non-linear correlation was found between CH_4 and WTD, with the highest rates of consumption around WTD 74 cm. Non-linear relationships of CH_4 and WTD have been observed in several studies and can be explained by the presence of an optimum for redox conditions and substrate availability in part of the profile (Brown *et al*. 2014).

WTD has been suggested as a proxy for predicting GHG emissions from drained organic forest soils. Ojanen & Minkkinen (2019) developed linear regression models for WTD and net soil CO_2 emission, based on data from 76 forestry-drained sites in Finland. For the German UNFCCC reporting, non-linear response functions for CO_2 and CH_4 have been developed based on national datasets of GHG emissions and WTD (Tiemeyer *et al*. 2020). Much like in this study, Tiemeyer *et al*. (2020) were also unable to find a robust relationship between N_2O and WTD, and instead developed their own EFs for N_2O based on the available emission data. This type of response function method may also be an option for improving the Swedish GHG inventory in the future, however, it would require large datasets of WTD and GHG emission measurements that may not be currently available. For an accurate representation of GHG emissions under Swedish conditions, this requires studies of GHG fluxes on different peatland and forest types in both climate zones in Sweden.

5. Conclusion

The default emission factors currently used in the Swedish GHG inventory differed a great deal from the emission factors found in the literature review in this study. Overall emission estimates were roughly the same using both sets of EFs, although this was due to the difference averaging out across the different climate-nutrient subtypes. Within each subtype, the emission estimates varied substantially between the two sets of EFs. However, developing new robust EFs for the Swedish UNFCCC reporting would require a far more comprehensive compilation of emission data than was possible in this study.

While there was a significant difference between emissions of CO_2 and CH_4 at <25 m and >25 m, the flux rates were still relatively substantial beyond 25 m. To exclude potentially drainage-affected organic soils from the GHG inventory may lead to a severe underestimation of GHG emissions. Inclusion of drainage-affected areas outside of the 25 m boundary, currently used in the GHG inventory, perhaps also with their own EFs, may lead to a more accurate estimate of total emissions.

Both CO_2 and CH_4 could be correlated to WTD and soil moisture. It may be possible to develop similar response functions as other countries have started to develop, but this would require large amounts of data that may not be available in Sweden. More studies of GHG emissions from different peatland and forest types in Sweden are necessary.

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