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Land use effects on nitrous oxide emission from drained organic soils

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Abstract

Since industrialisation, the atmospheric concentration of greenhouse gases (GHG) has increased significantly. The most important anthropogenic GHGs are carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O). N_2O is of major concern for two reasons; i) it is a potent GHG responsible for 6.2% of the observed anthropogenic radiative forcing and ii) it plays a major role in the destruction of stratospheric ozone. Soil processes are the largest contributor to the atmospheric N_2O, with agriculture as the largest anthropogenic source, accounting for 65% to 80% of total emission. The N_2O emissions are largely influenced by the land use and, due to the long half life time of N_2O in the atmosphere, is it important to find mitigation options that will reduce N_2O from agriculture. This is particularly important for drained organic soils, which are emitting large amounts of N_2O and CO_2. The main processes that produce N_2O in soil are nitrification and denitrification. Several interacting factors control these processes and the magnitude of N_2O emission, such as site fertility, ground water level, pH and competition from vegetation. Thus, because of the complexity of the driving variables, it is difficult to predict N_2O emission from environmental factors.

The articles in this thesis deals with measurements of N_2O, CH_4 and CO_2 fluxes, although I in my summary of these, focus on mitigation options for N_2O emissions from organic soils, as these are largely increased after drainage and have a large impact on the national GHG budget for Sweden.

Main findings of my work are that cereals, row sown crops (e.g. vegetables) and forest emit larger amounts of N_2O than pastures. However, due to the many controlling factors, the different soil properties as well as measurement periods it is difficult to generalise the findings for all peat soils. Soil pH was found to be a driving factor for N_2O emission from the forest site, with enhanced N_2O emission from low pH. Increased soil pH due to wood ash addition was found to reduce N_2O emission at a spruce forest site, which might provide a mitigation option for organic forest soils.

Based on the results from the thesis, a suggested mitigation strategy for N_2O emission from drained agricultural peat soils is to avoid cereal and vegetable cultivation in favour of permanent meadows or pastures. Increasing pH by wood ash or lime can decrease N_2O emission from forest soils. Although not a result in this thesis, liming of agricultural drained peat soils could also be an option.

Sammanfattning

Koncentration av växthusgaser i atmosfären har sedan industrialiseringen ökat kraftigt. De viktigaste antropogena växthusgaserna är koldioxid (CO_2), metan (CH_4) och lustgas (N_2O). Lustgas är av stor betydelse av två skäl, dels för att det är en potent växthusgas som uppskattas stå för 6,2% av den antropogena växthuseffekten och dels för att det har en avgörande roll vid nedbrytningen av stratosfärisk ozon. Markprocesser är den största källan till atmosfärisk N_2O, varav jordbruket är den största antropogena källan och står för 65% till 80% av de totala utsläppen.

Då markanvändning i hög grad påverkar N_2O-emissionerna är det viktigt att hitta begränsningsalternativ som kan minska lustgasutsläppen. Särskilt viktigt är det hitta metoder för att minska växthusgasutsläppen från åker och skogsmark på näringsrika dränerade torvjordar då dessa är stora punktkällor av såväl CO_2 som N_2O.

De viktigaste processerna som producerar N_2O i jord är nitrifikation och denitrifikation. Flera samverkande faktorer såsom som bördighet, grundvattennivå, pH och konkurrens från vegetation påverkar båda processerna och därmed storleken på lustgasemissionerna. Det är således svårt att förutsäga storleken på lustgasemissionerna på grund av komplexiteten av de drivande variablerna.

Trots att artiklarna i denna avhandling beskriver mätningar av både metan, koldioxid och lustgas, har jag i sammanställningen av dessa valt att fokusera på att hitta alternativ för att minska N_2O-utsläppen från dränerade torvjordar, eftersom dessa har en stor inverkan på den nationella växthusgasbudgeten för Sverige.

Viktigaste resultaten från mina undersökningar är att spannmål, radsådda grödor, såsom morötter och potatis, samt skog släpper ut mer N_2O än betesvall på dränerade näringsrika torvjordar. Eftersom det finns många faktorer som styr emissionerna och att markegenskaperna och mätperioderna var olika för mätytorna, kan jag inte dra slutsatsen att detta gäller för alla dränerade torvjordar. pH visade sig vara en drivande variabel för lustgasemissionerna från björkskogen, med ökade lustgasemissioner vid lägre pH. Vidare fann vi att askåterföring till dränerad organogen granskog minskade lustgasemissionerna samtidigt som mark-pH ökade.

Baserat på resultaten från undersökningarna i denna avhandling, föreslår jag att en strategi för att minska N_2O utsläppen från organogen dränerad jordbruksmark skulle vara att undvika spannmål och grönsaker till förmån för permanent ängsmark eller betesmark. Emissionerna från dränerad organogen skogsmark kan minskas med askåterföring. Huruvida kalkning av organogen jordbruksmark kan minska lustgasemissionerna är inte ett resultat av mina undersökningar, men det skulle kunna vara en möjlighet.

Till Mirjam

LIST OF PAPERS

This thesis is based on the following papers, which are referred to as **Paper I-V** in the text:

I. **Weslien, P**., Kasimir Klemedtsson, Å. Börjesson, G. & Klemedtsson, L. 2009. Strong pH influence on N_2O and CH_4 fluxes from forested organic soils. *European Journal of Soil Science*, **60**,311-320

II. Kasimir Klemedtsson, Å., **Weslien, P.** & Klemedtsson, L. 2009. Methane and nitrous oxide fluxes from a farmed Swedish Histosol, *European Journal of Soil Science,* **60**,321-331

III. **Weslien, P.**, Rütting, T., Kasimir Klemedtsson, Å., & Klemedtsson, L. 2010. Carrot crops on Organic Soil are hotspots for Greenhouse gas emissions. *Manuscript*

IV. Klemedtsson, L, von Arnold, K., **Weslien, P.** & Gundersen, P. 2005. Soil CN ratio as a scalar parameter to predict nitrous oxide emissions. *Global Change Biology* ***11*** *1142-1147*

V. Klemedtsson, L., Ernfors, M., Björk, R., **Weslien, P.** Rütting, T., Crill, P. & Sikström, U. 2010. Reduction of greenhouse gas emissions by wood ash application to a *Picea abies* forest on a drained organic soil. *European Journal of Soil Science,(accepted)*

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Introduction

Since industrialisation, the atmospheric concentration of so called greenhouse gases (GHG) has increased significantly. These GHG absorb the outgoing long wave radiation from the earth surface which results in an increase of the Earth's surface temperature. The most important gases that are thought to contribute to the anthropogenic global warming are carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) (IPCC, 2007). The terrestrial emissions of GHG are affected by land use (Smith & Conen, 2004) and there is a tight connection between the nitrogen (N) and carbon (C) cycle (Galloway *et al.,* 2008) affecting the emissions of N_2O. The contributing papers in this thesis deals with measurements of N_2O and CH_4 fluxes (**Paper I, II, III** and **V**), but also with CO_2 fluxes (**Paper V**). However, in my thesis I will focus on mitigation options in order to reduce the N_2O emissions from organic soils, as these are largely increased after drainage (Kasimir Klemedtsson *et al.,* 1997; Maljanen *et al.,* 2009) and have a large impact on the national GHG budget for Sweden (Ernfors *et al.,* 2007).

The concentration of N_2O in the atmosphere is rising since the 1750's and this increase of atmospheric N_2O is of major concern for two reasons. First, N_2O is a potent GHG with a 298 times higher global warming potential compared to CO_2 (on a 100 year basis) and responsible for 6.2% of the observed anthropogenic radiative forcing (WMO, 2009). Second, due to its role in atmospheric chemistry, N_2O is predicted to be the single most cause for stratospheric ozone depletion during the 21^{st} century (Ravishankara *et al.,* 2009). Soil processes are the largest contributor to the atmospheric N_2O (Bouwman, 1990). Agriculture is the largest anthropogenic source of N_2O, accounting for 65% to 80% of total emission (Crutzen *et al.,* 2008; IPCC, 2007). The emissions are largely influenced by the land use (Smith & Conen, 2004) and, due to the long half life time of nitrous oxide in the atmosphere (114 years), is it important to find mitigation options that, by altered management, will reduce nitrous oxide from agriculture. This in particular important for drained organic soils as these are high emitting (Regina *et al.,* 1997; Maljanen *et al.,* 2003a; Kasimir Klemedtsson *et al.,* 2009; Maljanen *et al.,* 2009) and are direct effects of land use change (Kasimir Klemedtsson *et al.,* 1997). Furthermore, for Sweden the N_2O fluxes from forests on drained peatlands were shown to contribute to 15% of the total anthropogenic N_2O emissions from Sweden (Ernfors *et al.*, 2007).

Pristine, undrained wetlands accumulate C in the peat, but are also a source for CH_4 (Laine *et al.,* 1996). The flux pattern is, however, dependent on the nutrient status of the peatland; minerotrophic peatlands have larger CO_2 uptake rates and emit more CH_4 than ombrotrophic peatlands, which could also be a source for CO_2 (Sarnio *et al.,* 2007; Maljanen *et al.,* 2009). Undrained peatlands are in general low emitters or has uptake of N_2O (Martikainen *et al.,* 1993; Nykänen *et al.,* 1995; von Arnold *et al.,* 2005a), with the exceptions of fertile forest swamps (von Arnold *et al.,* 2005b; Maljanen *et al.,* 2009) that may have significant emissions of N_2O. When the peat soil is drained it results in a subsidence of the peat layer. This is due to a degradation, oxidation, of the organic matter (Eggelsman, 1972; Kasimir Klemedtsson *et al.*, 1997). The subsidence is also due to physical changes in the peat (Berglund, 1996). The management will affect the subsidence rate of the peat as the subsidence rate that have been found to increase with cropping intensity, with an annual loss of 5 mm for a grassed ley, 10-20 mm for a barley crop and 20 to 30 mm for potatoes and carrots (Berglund, 1996; Berglund & Berglund, 2010). Drainage will thus change the emission pattern of the peatlands; the system becomes a weak emitter or a net sink for CH_4 (Maljanen *et al.*, 2007; Mäkiranta *et al.*, 2007) and a net emitter of N_2O (Kasimir Klemedtsson *et al.*, 1997; Maljanen *et al.*, 2009). The changes in CH_4 emission are a result of a decreased CH_4 production and increased CH_4 oxidation due to the increased oxygen status in the peat (Saarnio *et al.*, 2007). The increased aeration of the peat also enhances decomposition of organic matter and, thus, N mineralisation (Martikainen *et al.*, 1993). This can then lead to higher N_2O emissions. The most intensive period of drainage for forestry in Sweden was between 1920 and World War II, with a short period of remedial drainage during late 1970s and 1980s (Hånell, 1990). Of the original 103 000 km^2 pristine peatland in Sweden about 14 100 km^2 are drained and used for forestry (Hånell, 1990). In the 1940s 7485 km^2 drained peat soils were used for agriculture, but the number declined thereafter to 3015 km^2 in 2003 (Berglund & Berglund, 2010). Most of the abandoned agricultural land is most likely covered with forest today (Ernfors *et al.*, 2007), which is included in the estimates by Hånell (1990). However, a large uncertainty exists in the national database on previous land use history (Ernfors *et al.*, 2007). Only 12 km^2 of the abandoned land has been restored as wetlands (Berglund & Berglund, 2010).

For many ecosystems N is the major limiting factor for plant growth. Plants take up N in the form of ammonium (NH_4^+) or nitrate (NO_3^-) and to some extent organic N via mycorrhiza (Högberg, 2004). For non fertilised ecosystems the plants are dependent on organically bound N in plant residues and soil organic matter. In the absence of mycorrhiza, organic N has to be mineralised by microorganisms to NH_4^+, which may then be immobilised by microorganisms or taken up by plants. On the other hand, NH_4^+ can also be oxidised to NO_3^- by nitrification, which is a two step process: first ammonia is oxidized to nitrite (NO_2^-) by a two groups of organisms, ammonia oxidising bacteria and archaea (Treusch *et al.*, 2005). The second step, oxidation of NO_2^- to NO_3^-, is a bacterial process, mainly performed by nitrobacter species (Prosser, 1989). The produced NO_3^- can subsequently been taken up by plants, leached from the soil, but can also be the substrate for denitrification. This process occurs under anoxic conditions and is mainly performed by heterotrophic bacteria (Robertson & Groffman, 2007). The end product of denitrification is nitrogen gas (N_2), with NO_2^-, nitric oxide (NO) and N_2O as intermediate products. A second, heterotrophic pathway of nitrification in soil is the oxidation of organic N to NO_3^- by heterotrophic microorganisms (Killham, 1990), which, however, might also oxidise NH_4^+ to NO_3^- (de Boer & Kowalchuk, 2001).

Usually, autotrophic nitrification and respiratory denitrification are assumed to be the main pathways of N_2O production in soil, which is conceptualised in the Hole-in-the-pipe model (Firestone & Davidson, 1989). This conceptual model describes the production of N_2O as dependent on the overall flow of N through the considered process, whereas environmental factors may alter the fraction of N loss as N_2O. The highest N_2O emission occurs at oxygen limited, but not totally anaerobic conditions in the soil, because nitrification is an aerobic process requiring oxygen and denitrification, an anaerobic process, leads to N_2 rather than N_2O production under absence of oxygen. The soil oxygen content is governed by the diffusion (affected by pore space) and the oxygen consumption, driven by the C availability and temperature (Granli & Bøckman, 1994). Thus, the emissions can be described as a function of the water filled pore space (WFPS), where the nitrification related N_2O emission appears in at lower WFPS than denitrification (Davidson, 1991). Both nitrification and denitrification are affected by several chemical and physical factors in the soil, such as pH and temperature (Firestone & Davidson, 1989). Denitrifiers are pH sensitive and a decreased pH tends to decrease denitrification rates but increases the N_2O/N_2 ratio (Simek & Cooper, 2002). Due to the complex process regulation of nitrification and denitrification, the field emissions of N_2O are highly variable over time and space as well as show a large inter-annual variation driven by climate (Kaiser *et al.*, 1998). Moreover, the Hole-in-the-pipe concept may be an oversimplification, as several other soil processes are known to also produce N_2O, including heterotrophic nitrification (Anderson & Levine, 1986), fungal co-denitrification (Laughlin & Stevens, 2002), and nitrifier-denitrification (Wrage *et al.*, 2001).

Aims

My aim of this work was to find ways to mitigate the emissions of GHG and in particular of N_2O from drained organic soils by altered land use and management.

Material and methods

Sites

The Falköping research site is located in the southwest of Sweden (58°20´N; 13°30´E), with an annual average temperature of 5 °C and precipitation of 620 mm. At the site the organic soils are currently managed with several different land uses (Table 1). The area of the arable fields was drained in the 1920s and used mainly for grass production, either as silage or for cattle-grazing. The drainage system was improved in 1974 when a tile drainage system was installed at approximately 1m depth. The soil is classified as Histosol with a depth between 0.9 m (barley field) and 2 m (pasture) deep, covering a clay mineral soil (**Paper II**). The soil at the barley and pasture was rich in $CaCO_3$, originating from mollusc shells, which explained the relatively high pH of 6.8-7.4 (Table 1, Fig. 1).

Table 1. Soil properties of the fields at the Falköping research site and at the experimental site at Skogaryd

	BD kg m^{-3}	OM %	pH	C/N 0-10 cm	GWL m
Falköping					
Pasture	400-430	32-36	6.8-7.4	9-10	0.10-0.73
Barley	270-450	35-55	7.3-7.4	9.4-9.5	0.14-1.03
Forest	194	53	3.6-5.9	12.8	0.12-0.96
Carrot	280	45	5.8	nd	nd
Skogaryd					
Control	195	79	4.5	23	
Low	195	79	4.8	23	
High	195	79	5.1	23	

BD-bulk density, OM-organic matter, GWL-ground water level, nd – not determined

Figure 1. Arial photo of the Falköping site showing the locations for the pasture, barley, forest, carrot and the bare soil sites (photo from Google maps, 2010).

The forest area was drained in the 1920s for extensive peat extraction and grazing, and in the 1960s the area was colonised by silver birch (*Betula pendula* Roth). The average peat depth was 2.2 m (**Paper I**).

All measurements were done within the GEFOS project. Soil chambers were installed in March 1994 on the pasture and barley sites. The forest site was added in August 1994 to test the hypothesis that forestation of agriculture peat soil would reduce the expected high N_2O fluxes from the arable sites. In late spring 1997, we got the opportunity to investigate the GHG fluxes from a carrot field 1 km west of the other sites and chambers were installed after sowing in June.

The Skogaryd site is a well drained former fen in the southwest of Sweden (58°23'N, 12°09'E). It was drained in the 1870s and used for tilled agriculture until 1951 when it was planted with Norway spruce (*Picea abies* (L.) Karst.). The long-term mean (1961-1990) annual temperature at Vänersborg, 12 km from the site, is 6.4ºC and the mean annual precipitation is 709 mm. The experiment was designed as a 3 × 3 random block experiment. The three treatments were no addition (control,), 3.3 tonnes (dry matter) crushed ash ha^{-1} and 6.6 tonnes (dry matter) crushed ash ha^{-1} (**Paper V**).

Measurements

All flux measurements were done using dark stainless steel chambers with an area of 0.225 m^2 as described in Weslien *et al.* (1998) and sampling to 22 ml headspace bottles was done according to von Arnold *et al.* (2005b) with the difference that we at the Falköping sites sampled at 0 and 20 minutes after closure. At the barley site, chamber height was increased by extensions as the crop height increased. Measurements of abiotic variables and soil nitrogen content are described in the **Papers I-III** and **V**

Results and discussion

There were large differences in the N_2O fluxes between the four sub-sites at Falköping, which were low from the pasture (0.20 g N_2O m^{-2} year^{-1}), intermediate from the barley (0.99 g m^{-2} year^{-1}), high from the forest (1.96 g m^{-2} year^{-1}) and very high from the carrot field (4.1 g m^{-2} year^{-1}) (**Papers I-III**). The measured annual N_2O flux from the spruce forest at Skogaryd (control) was 0.38 g N_2O m^{-2} year^{-1} and was decreased in the ash treated plots (**Paper V**). The measured fluxes will be discussed more thoroughly in the following paragraphs in context with the controlling factors, although it sometimes is necessary to discuss the effect of other controlling factors than mentioned in the headings.

Temperature and winter fluxes

A significant part of the annual flux at the Falköping site was emitted when the soil was frozen, with a proportion of 20, 34 and 15% for the pasture, barley and forest, respectively (**Paper I-II**). The largest contribution to the annual flux occurred, however, during summer on all four sites.

The significant difference in the fluxes at the Skogaryd site between the treatments was largely due to differences during winter. The contributions of the winter fluxes were 46, 45 and 30% for the control, low dose and high dose, respectively (the difference will be further discussed in the section on pH). Maljanen *et al.* (2004) reported that 99% of the annual flux from a peat soil with barley was emitted during winter. Other studies have reported winter flux contributions of 25-73 % (Flessa & Dörsch, 1995; Röver *et al.*, 1998; Papen & Butterbach-Bahl, 1999; Teepe *et al.*, 2000; Maljanen *et al.*, 2003b; Regina *et al.*, 2004).

Several factors favouring N_2O production are prevailing outside the cropping season since inorganic N accumulates, the soil is wet and freezing events are specially promoting N_2O production (Ruser *et al.*, 2006). During the wet seasons with frost, up to 50-70% of the yearly N_2O emission can be emitted during a period of a few days (Kaiser & Ruser, 2000; Wagner-Riddle & Thurtell, 1998). Freeze-thawing events were experimentally examined by Teepe *et al.* (2000, 2001), showing that freezing results in increased emission but that the highest emission occurred after thawing. Similar experiments were also performed by Priemé and Christensen. (2001) on organic soils, collected in Sweden, Finland and Germany, who found N_2O emission rate to increase 1000 fold after thawing. The average proportions found for the Falköping and Skogaryd sites are hence in the same range as reported by other studies. This stresses the need for continuous measurements over the whole year in order to generate accurate annual mean values. This was not conducted in the bare plots (presented below), but the site was a part of the barely field, which was measured the whole year. Thus, the winter emissions could be estimated. However, the estimation of the annual emissions from the carrot field is more uncertain (**Paper III**).

Agriculture sites and management

The emission from the pasture at Falköping was low compared to other studied grasslands (Table 2), despite the fact that the pasture was the only of our studied fields that was fertilised (100 kg N annually as urine). Regarding fertilisation of arable peat soils, contradicting results exist, with no difference or higher N_2O emissions from non fertilised soils (Flessa *et al.*, 1998; Rochette *et al.*, 2010) while others have reported small increases in N_2O emission after N-fertilisation (Velthof *et al.*, 1996; Langefeld *et al.*, 1997). Obviously, other factors than nitrogen availability governs the emission. Therefore fertilising effects will not be further discussed.

Table 2. N_2O emissions from peat soil under grass

Country	**pH**	**N_2O g m^{-2} yr^{-1}**	**Study**
Sweden	7.4	0.20	**Paper II**
Finland	5.6	0.62	Regina *et al.*, 2004
Finland	5.8	1.14	Regina *et al.*, 2004
Finland	5.3	1.74*)	Regina *et al.*, 1996
Finland	4.4	0.27-0.60	Maljanen *et al.*, 2004
Finland	5.0	0.58	Maljanen *et al.*, 2009a
Finland	5.0	0.86	Maljanen *et al.*, 2009b
Finland	5.8	0.14	Maljanen et al., 2009a
Finland	6.1	1.7	Maljanen *et al.*, 2003a
Finland	5.3	1.2-1.5	Nykänen *et al.*, 1995
Norway	5.2	0.69	Grølund *et al.*, 2006
The Netherlands	nd	1.4-6.1	Langeveld *et al.*, 1996
The Netherlands	nd	0.66-6.4	Velthof *et al.*, 1997
Germany	5.8	3.1	Flessa *et al.*, 1998
Germany	5.1	0.66	Flessa *et al.*, 1998

nd – not determined

*) not full year, measurement during growing season only

The N_2O emission from the barley field was higher than from the pasture, but in the same range as or lower than other investigations on cereal crops (Table 3).

Table 3. N_2O emissions from peat soils under cereals.

Crop/Country	**pH**	**N_2O g m^{-2} yr^{-1}**	**Study**
Barley/SWE	7.2	$0.99^{*)}$	**Paper II**
Barley/FIN	6.1	1.3	Maljanen *et al.,* 2003a
Barley/FIN	5.0	0.85-1.8	Maljanen *et al.,* 2004
Barley/FIN	5.6	2.0	Regina *et al.,* 2004
Barley/FIN	5.8	2.4	Regina *et al.,* 2004
Rye/GER	4.0	8.9	Flessa *et al.,* 1998
Maize/GER	5.6	2.5	Flessa *et al.,* 1998
Maize,Wheat/USA	nd	4.5-15.2	Elder & Lal, 2008

nd – not determined

$^{*)}$ corrected from **Paper II** which says 1.51 g N_2O m^{-2} y^{-1}.

The difference between pasture and barley at our site could have been due to a tillage effect. Repeated tillage has previously been shown to increase short-term N_2O fluxes (Nykänen *et al.,* 1995; Kessavalou *et al.,*1998; Elder & Lal, 2007). This is in agreement with our investigation (**Paper II**), where large emission events were found after soil cultivation managements at the barley field (see also Figure 2). The elevated emissions lasted for approximately 4 days before returning to the background level. We explained the increased rate of N_2O emission, according to **Paper II**, by soil perturbation, resulting in newly exposed soil particle surfaces. This increases soil mineralisation at a time when there are no roots competing for the released mineral N, which becomes available for nitrification and denitrification and thus enhances N_2O production. At the time of cultivation the soil is moist and the temperature is favourable for sowing the crop, which provides environmental conditions promoting N_2O production.

On basis that the N_2O emission increases after soil perturbation, no-till management has been recommended in order to decrease N_2O emissions. However, most studies comparing tilling and no-tilling systems in respect of N_2O emission has shown that no-till systems tend to enhance N_2O emission due to increased denitrification as a result of soil compaction (Smith *et al.,* 2001; Rochette *et al.,* 2008). Rochette *et al.* (2008) concluded that no-till generally increased N_2O emission at poorly aerated soils while the differences were small for well aerated soils when comparing 25 different mineral soils. Studies comparing N_2O emissions from no-till and tillage systems on peat soils are very scarce. Elder & Lal (2007) studied a peat soil in Ohio and found an almost 3 fold higher N_2O emission from the till treatment compared to the no-till. The lack of till versus no-till studies on peat soils makes it difficult to predict any general effects.

The highest emission of my study was from the carrot field (**Paper III**). These emissions are in the same range as found for potato and vegetable crops in northern climate, but less than the emission in temperate climate (Table 4).

Table 4. N_2O emissions from row sown crops production

Crop/Country	Fertiliser Kg N	pH	N_2O g m^{-2} yr^{-1}	Study
Carrot/SWE	0	5.8	$4.1^{*)}$	**Paper III**
Potato/FIN	100	5.8	1.6	Regina *et al.,* 2004
Onion/USA	nd	nd	13.4	Duxbery *et al.,* 1982
Vegetables/CAN	0	5.0	$1.5\text{-}3.9^{*)}$	Rochette *et al.,* 2010
Vegetables/CAN	50-150	5.0	$0.56\text{-}6.3^{*)}$	Rochette *et al.,* 2010

nd – not determined

$^{*)}$ not full year, measurement during growing season only

When comparing the cropping and post-harvest period for the carrot field (**Paper III**), we found a different pattern. During the cropping period (16 June – 12 Aug) the N_2O emissions were one order of magnitude higher than during the post-harvest period (20 Aug – 01 Oct). This finding is particularly interesting as during the post-harvest period the soil NO_3^- concentrations were much higher compared to the cropping period (Fig. 2b). This finding, as well the negative correlation between N_2O and soil NO_3^- concentration (**Paper III**), points to the fact the lower N_2O emissions during the post-harvest was not due to NO_3^- limitation. A likely explanation is that the denitrifying organisms were instead C limited during the post-harvest period, when no root exudates were available as C source. This interpretation is in line with the fact that denitrifiers are generally stimulated by roots (Woldendorp, 1963; Klemedtsson *et al.*, 1987) and that this is due to increased oxygen consumption by root and heterotrophic respiration. The denitrifiers may further be directly stimulated by the exudates (Klemedtsson *et al.*, 1987), as they are usually limited by C in soil (Granli & Bøckman, 1994). The bare soil plot, within the barley field, showed similar emissions to the barley field in the post-harvest period, but the emissions were one order of magnitude higher during the cropping period. However, the total flux (cropping plus post-harvest period) were similar for barley and bare soils plots. This is in contrast to findings by Duxbury *et al.* (1982), who found, for organic soils, an 8 times higher emission from fallow soil compared to vegetable crop field. However, their study was conducted in a sub-tropical climate, while our study site is in a temperate-boreal region, which might explain the contrasting findings.

Soil nitrogen

A rather high water content at our site may have restricted mineralisation and, hence, N_2O production. In addition, plant growth was vigorous, so it is unlikely that N was limiting. The soil ammonium concentrations show a similar pattern for all treatments at Falköping (Fig. 2) although differing in amounts. For the pasture, barley, forest and carrot sites respectively the average (and minimum) concentrations for NH_4^+ were 11 (2), 8 (0), 9 (2) and 8 (2) kg N ha^{-1}.

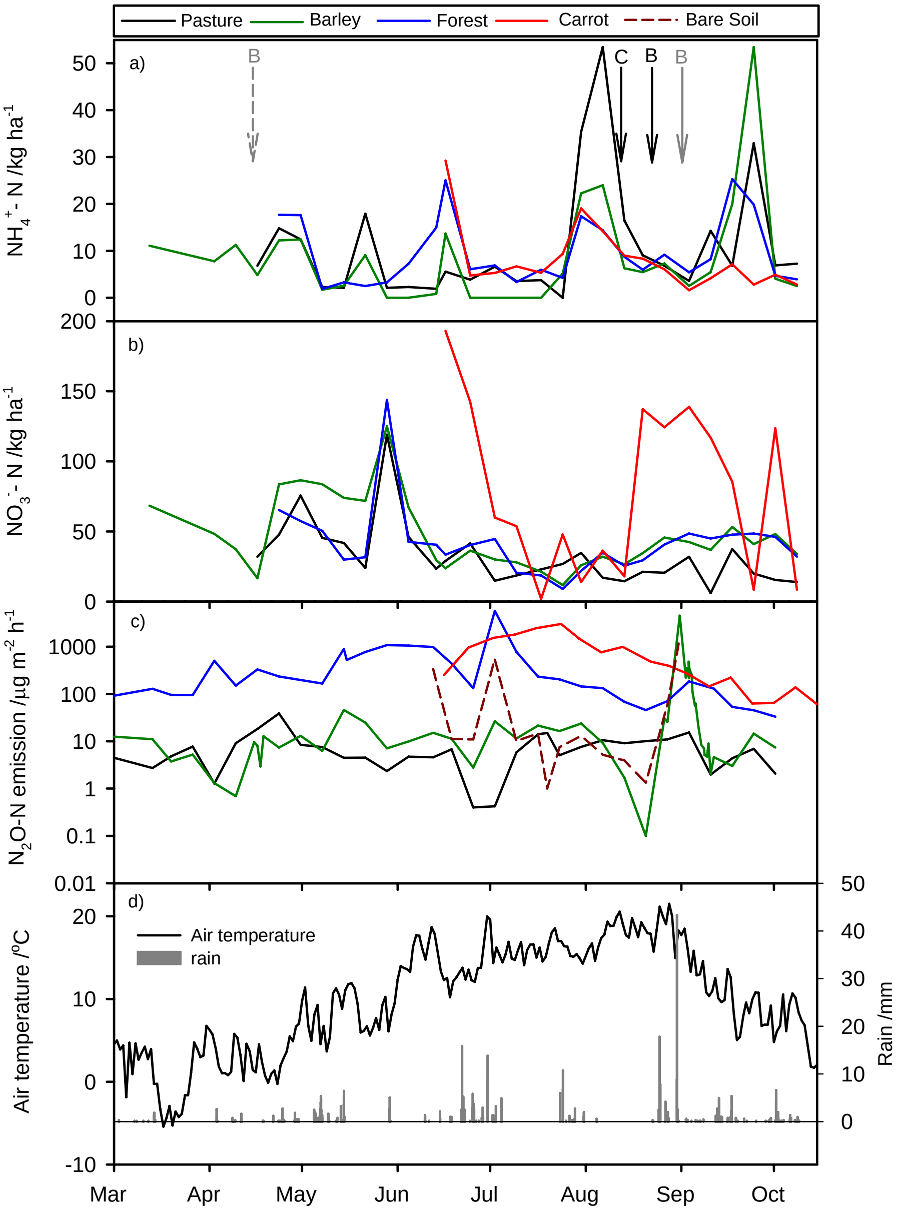

Figure 2. Concentrations of soil NH_4^+ (a) and NO_3^- (b), emissions of N_2O (c) as well as air temperature and precipitation (d) at the four sites in Falköping in 1997. The arrows indicate time of managing events: dashed grey arrow denotes ploughing followed by harrowing and sowing, black arrows harvest, and grey full arrow ploughing (B: barley; C: carrot).

All Falköping sites showed very high mineral nitrate levels, indicating substantial nitrification. The average (and minimum) nitrate concentrations in the top 30 cm were 30 (6), 47 (12), 42 (9) and 77 (2) kg N ha^{-1} for the pasture, barley, forest and carrot sites respectively. The nitrate concentrations were (except for the carrot site) hence never lower than the 5 mg NO_3^--N kg dry soil (equivalent to 4.5 kg ha^{-1} for the carrot soil)limit that Smith *et al.* (1998) and Dobbie *et al.* (1999) suggest as being N-limiting for N_2O production.

The high NO_3^- concentration in the forest soil indicates a high nitrification activity despite the low pH. It should, however, be noted that the soil samples collected from the forest site were a mixed sample from ten sub-samples, hence, high nitrate samples from the high pH chambers could have balanced low nitrate samples from the low pH chambers. Nevertheless, soil nitrogen availability was probably not a limiting factor for N_2O emission at any of the Falköping sites and could not explain the differences in N_2O emission between the sites. The differences in the ammonium and nitrate are most likely affected by the difference in bulk density of the soils (Table 1). The lower the bulk density is the more rapid oxygen diffusion down into the soil, which will affect N mineralisation, nitrification and denitrification. The barely field had the highest bulk density, followed by pasture (with a large range between subplots), forest and the lowest was at the carrot field. The carrot field also had the highest nitrate concentrations, and thus most likely the highest nitrification rates.

During 1997 a bare soil site (fallow) was established (12 June) within the barley field (Fig. 1), but was not presented in **Paper II**. These plots where ploughed at the same time as for the barley. The emissions where in the same range that has been found for boreal climate (Table 5), but lower than for warmer climate in the south of USA.

Table 5. N_2O emissions from bare and fallow peat soils

Country	**pH**	**N_2O-N** **g m^{-2} yr^{-1}**	**Study**
Sweden	7.4	$0.94^{a)}$	**Bare soil at Falköping**
Finland	5.8	0.69	Regina *et al.*, 2004
Finland	5.6	3.9	Regina *et al.*, 2004
Finland	6.1	1.1	Maljanen *et al.*, 2003a
Finland	5.0	0.63-5.8	Maljanen *et al.*, 2004
Finland	5.3	0.58	Nykänen *et al.*,1995
USA	nd	26	Duxbery *et al.*, 1982

a) not full year, scaled to annual emissions using data from barely field outside growing period.

The bare soil treatment at the barley site emitted 0.33±0.09 g m^{-2} while the accumulated emission from the barley site during the same period was 0.37±0.11 g m^{-2}. The emission pattern of the bare soil followed in general the barley, with the exceptions after a heavy rain in July, which resulted in higher emissions from the bare plots than for the barley (Fig. 2). This was probably an effect of differences in WFPS in the soil. During the summer period the evapotranspiration from the barely most likely reduced the WFPS in the barely field compared to the bare plots. After harvest the emission levels from both sites increased to the same level (Figure 2).

Soil water status

Groundwater level could explain the differences in N_2O emissions between the sub-sites within the treatment but not the difference between the treatments (Barley and Pasture, **Paper II**). Events with very high emissions were observed after intense rainfalls at both the barley and forest site (Fig. 2). The very high peak at the barley site in August 1997 coincided with harvesting and ploughing, which makes it hard to distinguish whether the emission peak was an effect of soil perturbation or rain or both.

Anoxic conditions also occurred in our drained arable soil (**Paper II**), where more than 60% WFPS results in a net CH_4 emission, which is in line with reports of rapid increases in the denitrification rate when WFPS exceeds 60% (Davidson, 1991; McTaggart *et al.*, 1997; Maljanen, 2003). Water-filled pore space is one of the most important factors controlling N_2O emissions (Davidson, 1991), which have been found to increase when the soil is close to saturation, because of the increasingly anoxic conditions (Davidson, 1991; Granli & Bøckman, 1994; Rudaz *et al.*, 1999; Ruser *et al.*, 2001; Dobbie & Smith, 2006), but to decrease when the soil becomes saturated because complete anoxic conditions favour N_2 production. In order to model N_2O emissions, the so called ‘anaerobic balloon’ was suggested as governing N_2O production in the soil (Li, 2000). Smith *et al.* (1998) proposed a threshold of 70% WFPS, below which very little N_2O is emitted. A similar threshold effect of soil water has also been found for denitrification (Klemedtsson *et al.*, 1991). On a peat soil in Finland the largest N_2O emissions were found during wet conditions, 60% WFPS (Maljanen *et al.*, 2003a), but their peak emission were 10 times larger than we found in **Paper II**.

Forest sites

The emissions from the birch forest were in the upper range from what has been found earlier (**Paper I**). This site had not been used, to my knowledge, for any intensive agricultural use, only extensive grazing. Peat harvest had been conducted at the site, but not in the area where the flux measurements were conducted. Thus, I do not classify it as an afforested former agricultural land. The site had both high ammonium as well as nitrate concentrations, thus nitrification and denitrification ought not to be limited by available N (Fig. 2). There was a remarkably large spatial variability with respect to pH and C content within the site, which was found at the end of the measurement period (September 1997). Until that date, soil was collected as combined samples (according to the sampling protocol for the adjacent arable sites), and the variability was not revealed. The soil pH for the adjacent arable fields was alkaline (6.8–7.4) because of the large calcium carbonate content originating from mollusc shells (**Paper II**). The forest soil contained no calcium carbonate, but it may have had some impact on the origin of the soil pH variability over the site. Berglund, (1996) examined the range of physical and chemical properties of drained organic soils in Sweden and found that the pH was normally between 4 and 5. The mean pH of the site is in the same order as found for Swedish and Finnish forest on drained organic soils (Table 6). The emissions were high compared to other drained forest soils, which could be explained by the low C/N ratio (further discussed below). At the site the emissions of N_2O were negatively correlated with ground water level and positively correlated with WFPS. These expected correlations could be interpreted as the emissions are positively correlated to the likelihood of increased anoxia in the soil, especially as the nitrate concentration was high in the soil, well drained to promote nitrification, which result in high N_2O emissions. Data from soil gas probes (not presented) showed the highest N_2O concentrations at the largest profile depth (30-40 cm), suggesting that N_2O production occurred just above the fluctuating water table. However, during winter periods highest soil N_2O concentrations were found at 5-10 cm depth. Similar, pattern where found at the Skogaryd research station, where today more intensive studies are conducted to link chamber flux to soil gas profiles.

Table 6. N_2O emissions (g $m^{-2}year^{-1}$) from boreal forests on drained peatsoils with data on C/N and pH.

Forest	**C/N**	**pH**	**N_2O g m^{-2} yr^{-1}**	**Study**
Birch, SWE	13	4.6	3.05	**Paper I**
Birch, Pine, FIN	21	3.7	4.1	Maljanen *et al.*, 2009b
Birch, FIN	20	4.5	0.66	Maljanen *et al.*, 2003a
Spruce, SWE	21	4.5	0.27	Ernfors, 2009
Spruce, SWE	34	4.9	<0.01	Ernfors, 2009
Pine, SWE	44	2.7	0.04	von Arnold *et al.*, 2005b
Spruce “young” , SWE	26	3.3	0.08	von Arnold *et al.*, 2005b
Spruce “old” , SWE	29	3.2	0.05	von Arnold *et al.*, 2005b
Birch, SWE	25	3.4	0.2	von Arnold *et al.*, 2005b
Afforested				
Alder, SWE	16	4.5	0.9	von Arnold *et al.*, 2005a
Spruce, SWE	23	4.5	0.38	**Paper V**
Birch, FIN	19	5.3	0.43	Maljanen *et al.*, 2001b
Birch, FIN	19	4.7	0.97	Maljanen *et al.*, 2001b
Birch, FIN	19	5.4	0.11	Mäkiranta *et al.*, 2007
Birch, FIN	19	4.1	0.17	Mäkiranta *et al.*, 2007
Birch, FIN	19	4.2	0.73	Mäkiranta *et al.*, 2007
Pine, FIN	18	4.7	1.8	Mäkiranta *et al.*, 2007
Pine, FIN	17	4.1	3.1	Mäkiranta *et al.*, 2007
Birch, FIN	18	3.4	3.4	Mäkiranta *et al.*, 2007
Birch, FIN	13	4.7	3.5	Mäkiranta *et al.*, 2007

The emissions found from afforested sites where in the same range as fluxes found from the agricultural soils (Table 2 and 3). This suggests that afforestation does not decrease N_2O emissions and, hence, is not a suitable mitigation option to control N_2O emissions. The emissions from the spruce forest at Skogaryd (**Paper V)** were about one magnitude lower than from the birch forest, but showed the same temporal pattern, with both winter and summer peak emissions. The winter emissions from the Skogaryd site (**Paper V**) were 0.26 kg g N_2O m^{-2} compared to the 0.46 g N_2O m^{-2} at the birch forest (**Paper I**). The largest difference in emissions was found to be the summer emission period, with much lower emissions at the spruce site compared to the birch site. This could have been due to a higher N uptake by the forest at Skogaryd, as the spruce had a higher estimated annual biomass increment than the birch; 8440 kg dry weight ha^{-1} compared with 6100 kg dry weight ha^{-1} (unpublished data). Thus, when estimating the emissions from forested sites the tree growth ought to be added as a variable affecting the annual emissions. However, the C/N ratio was lower at the birch forest than at the spruce, which might be the main driving factor (see below).

Soil pH

The soil pH has been found to be one of the most important factors controlling denitrification in soils (Nömmik, 1956; Simek & Cooper, 2002). Although contradicting results exist, there are consisting results showing that lower pH increase the $N_2O:N_2$ ratio. Decreased pH increases the amount of N_2O produced from autotrophic nitrification in Finnish forest soils (Martikainen, 1985). Similar results have been found for Dutch forest soils: acidic conditions favour N_2O production from both autotrophic and heterotrophic nitrifiers (Martikainen & de Boer, 1993). Thus, it was expected that the pH gradient, found at the Falköping forest site (**Paper I**), would affect the N_2O emissions. The pH effect on N_2O emissions, from the single chambers annual emission, could be described by the equation:

$$N_2O\text{-}N = 636.6\ e^{-0.8028\ pH}$$

The pH respond explained 89% of the spatial variation in the emissions between the chambers in **Paper I**. This is in line with the findings of Bandibas *et al.,* (1994) in which pH was the greatest predictive value for overall N_2O emissions in a study of 18 Belgian soils. Using the equation on the wood ash addition treatments (**Paper V**), after calibrating the intercept to the control winter emissions (pH 4.5 and emission 0.26 g N_2O m^{-2}), results in a fairly good estimation of the winter emissions from the ash amended treatments; 0.20 and 0.16 g N_2O m^{-2} compared to the measured values of 0.15 and 0.14 g N_2O m^{-2} for the low and high ash addition respectively. The winter emissions where used to test the equation as the emissions during this time of the year are most likely from the top soil, and thus affected by the ash. During the summer the highest production was deeper down in the soil profile as indicated by soil gas probe measurement (*Ernfors, pers. comm.*). Thus, wood ash could be a mitigation option, but it will only function for upper part of the top soil for well drained soils. The long term effect is not known, but the investigations at Skogaryd are continued.

In the compilation of N_2O fluxes from northern drained peat soils by Maljanen *et al.* (2009), low emissions below pH 4 and above pH 6 were found, but not the significant correlation as in **Paper I** when comparing pH with emissions from all kinds of land use. However, this was expected, as the emission pattern of a site is a sum effect of a large number of driving variables.

The production and consumption of N_2O is regulated in a complex manner by a large number of soil variables, including water, N, C, pH and the bacterial community (Firestone & Davidson, 1989; Granli & Bøckman, 1994). If one of these factors is not at an appropriate level, this can suppress emissions. Removal of one or the other constraining factor can result in the promotion of processes controlling N_2O flow. Thus, to explain the size of the N_2O emissions is difficult, unless one of the driving variables changes dramatically in one of the treatments. This was the case for the investigation of the birch forest (**Paper I)** and thus gave the clear pH connection. However, not only the differences in soil physical/chemical properties, plant community, management and climate have to be evaluated. It is also important to know if these variables affect the microorganism active in the nitrogen cycle.

Holtan-Hartwig *et al.* (2000) compared the Falköping barley soil with two other soils from Finland and Germany. They found that the lower N_2O production rates from the Falköping soil could be explained by different denitrifier communities. The denitrifier community from the Falköping soil was more effective in reducing N_2O to N_2, which was in line with the lower measured N_2O emissions. They also suggested that the effective N_2O reduction for the Falköping soil could have been an effect of the high pH (**Paper I**).

In further studies (Holtan-Hartwig *et al.*, 2002) on the same soils they found significant differences in temperature response of denitrifier N_2O production and that the Falköping denitrifier community had a lower potential to act as a net N_2O emitter after sudden anaerobisis, which may lead to lower “event based” N_2O emission after for example a heavy rainfall.

It is possible that occasions with large gas emissions, resulting from denitrification, are dominated by N_2 and not N_2O emissions, and are, thus, not detected by our investigation. The quality of the denitrifier community, along with the alkaline pH, could thus be factors to take into account when explaining the small emissions reported here compared with other drained farmland. Applying the equation from the forest site on the agricultural sites at Falköping failed. Clearly pH alone cannot be used to estimate fluxes from soils with different properties and management, although it might be useful for peat soils with the same origin. Although the response to the pH change might be more mechanism specific, the mechanisms involved varies between microbial communities and are affected by abiotic variables.

Soil fertility and CN ratio

The relationship we found in **Paper IV** showed that the C/N ratio of the soil could be used for scaling N_2O fluxes from organic forest soils. Applying the identified relationship for the Skogaryd site (C/N = 23) gives an estimate of 0.26 g N_2O m^{-2} $year^{-1}$. The measured annual N_2O flux from the Skogaryd control site was 0.38 g N_2O m^{-2} $year^{-1}$, which is fairly close to the C/N-ratio derived emission. Ernfors *et al.* (2007) slightly modified the equation in order to estimate uncertainties and applied it to estimate the N_2O emissions from forested drained organic soils, using national inventory data. The estimated total emission for drained organic forest soils in Sweden was 4,700 tones N_2O

annually, which would increase the current estimate of the Swedish anthropogenic N_2O source strength by 18%. The emissions from drained organic soils are currently not included in the Swedish national reporting.

However, the C/N ratio is just a proxy of the potential gross mineralisation rate. The actual gross mineralisation rate will also be affected by the oxygen status in the peat, determined by the ground water level. Mineralised N can be temporally stored in the soil, taken up by trees and understorey vegetation, or can be nitrified. Nitrification results in the production of nitrate, which is the substrate for denitrification. Both these processes may produce N_2O when the circumstances are fulfilled (Firestone & Davidson, 1989). The previous land use may be of importance, as high N_2O emissions were found from afforested peatlands previously used for agriculture (Maljanen *et al.*, 2007). However, former land use is also linked to the site fertility, as the most fertile soils have historically been used for agriculture. Furthermore, N_2O emissions depend on the current forest growth, whereby the amount of mineral N uptake by plants is determining the N availability for N_2O production pathways. In the review by Maljanen *et al.* (2009) they compiled emission data from drained peat soil in northern countries. They found that the correlation fitted well for forest soils, but not for agriculture soils. This suggests that for intensively managed soils the influence of the C/N ratio on the N_2O emission looses its importance. Ojanen *et al.* (in press) found that forested agricultural peat soils neither fitted the C/N – N_2O emission, suggesting that the land use history can greatly affect the N_2O emission. However, the Skogaryd site (**Paper V**) has a long history of agricultural use, but fitted fairly well with C/N response. The forest at the site is planted, is well growing and is in the same age as most of the forests on the afforested peatlands former used for agriculture in Sweden. The two above mentioned Finnish compilations included natural afforested sites as well, with higher emission than predicted, which could have been due to a lower N-uptake by the vegetation.

Modelling

Several interacting factors are important for the magnitude of N_2O emission, such as site fertility, ground water status, pH and competition from vegetation. Thus, it is difficult to predict N_2O emission from environmental factors because of the complexity of the driving variables. The best approach to estimate emissions is probably the use of mechanistic models (Bakken & Dörsch, 2007), as for example the CoupModel (Norman *et al.*, 2008), which are driven by long-term climate records. This is because the emission response to changes in soil conditions is not linear with respect to changing climatic conditions. The driving factors for both nitrification and denitrification and N_2O emissions are complex, with a large number of interactions and feedbacks (Brumme *et al.*, 1999). The large inter-annual variation found in this thesis highlights the need for long term measurements in order to generate reliable data for upscaling and GHG budgets. The CoupModel have been calibrated on the mean pH emissions from the birch site at Falköping and was capable of predicting the mean annual emissions for the lower and higher pH at the site. However, the model emissions had a temporal shift in the dynamics (Nylinder, 2010).

Can one mitigate the emissions?

To afforest the farmed organic soils does not seem to be a mitigation option. The forest emissions are in the same order as for the farmed sites, especially for the nutrient rich soils. In order to reduce the emissions from the forest soil, the N-supply has to be reduced. This can only be done by raising the groundwater table, and thus decrease the N-mineralisation, but also the forest production will be decreased if the mean water table is above 0.5 m. The emissions can partly be reduced by a pH increase from wood ash application, but this alone will have limited effect. A combination of moderately raised ground water and wood ash/lime addition might be a good mitigation option, but it has to be tested.

Table 7: Cumulative N_2O emissions from five different land use options on organic soil from Falköping for cropping and post-harvest period (1997). Modified from **Paper III**

	Pre harvest	Post harvest	Sum	
	kg N_2O ha^{-1} $period^{-1}$			
Pasture	**0.2**	**0.1**	**0.3**	**Paper II**
Barley	**0.3**	**5.6**	**5.9**	**Paper II**
Forest	**21.8**	**1.5**	**23.3**	**Paper I**
Carrot	**36.8**	**3.6**	**40.4**	**Paper III**
Bare soil	**2.0**	**3.5**	**5.5**	

The emissions from farmed land can be mitigated by a change in land use and management. Certainly management intensity in form of soil perturbation, i.e. harrowing, ploughing and mechanical weed removal, is one important factor. Soil perturbation enhances soil N mineralisation and as a consequence the N_2O emission from the arable sites clearly increases with increased soil perturbation. Although the higher emission from the carrot crop could have been an effect of the difference in pH and the direct effect of the crop, it could also have been an effect of more soil perturbation. Reducing emission by no-till practices might be an option, but are yet to be tested as very few studies have been conducted on organic soils. The very low emissions from the pasture (Table 7) suggest that avoiding cereals and row sown crops totally in favour of grass production seems to be a powerful mitigation option, possibly in combination with liming.

My thesis has focused on N_2O emissions from drained organic soils, but these soils are also large sources for CO_2. Mitigation options proposed for N_2O discussed herein would in most cases also reduce CO_2 emissions. When trading CO_2 and N_2O emissions with changing land use or management, one should, however, always bear in mind that “CO_2 sequestration is reversible, whereas N_2O emission is not” (Smith & Conen, 2004).

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