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Emissions of N_2O from agricultural soils and mitigation options:

A review with special reference to Norwegian agriculture

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Sammendrag/summary:

Denne rapporten gir en beskrivelse av lystgassutslipp fra jordbruksjord med særlig fokus på norsk landbruk i tillegg til noen forskningsresultater fra andre land. Det gis en introduksjon om prosesser og faktorer som har betydning utslipp av lystgass. Effekten av gjødsel, jordpakking, jordarbeiding, jordfuktighet, drenering, pH og frysing-tining er beskrevet. Det er også foreslått ulike tiltak for reduksjon av N_2O-utslipp som er relevant for norske forhold. Til slutt konkluder rapporten med forslag om framtidig forskning og måling som grunnlag for reduksjon av N_2O-utslipp fra jordbruksjord.

This report gives a comprehensive review of current knowledge on nitrous oxide (N_2O) emissions from mineral soils with special reference to the Norwegian agriculture and some research results from other countries. The report starts by highlighting the importance of N_2O followed by descriptions of the pathways to N_2O production and factors affecting N_2O emissions from agricultural soils and measurement techniques and modelling N_2O emissions. This is followed by reviewing and discussing research results on the effects of soil management practices including fertilizer application, soil compaction, soil tillage, effects of soil moisture and drainage conditions, effects of soil pH, freezing-thawing effects and effects of soil drying and rewetting on N_2O emissions. Furthermore, some relevant mitigation measures to reduce N_2O emissions are presented. The report concludes by suggesting future research needs to measure and mitigate soil N_2O emissions.

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PREFACE

This review report is about nitrous oxide (N_2O) which is one of the potent greenhouse gases emitted predominantly from agricultural soils. The report highlights the importance of N_2O with special reference to Norwegian agriculture by synthesizing information and data from several literatures pertaining to N_2O emissions from Norway and other countries that have similar pedo-climate conditions and agricultural systems.

The information presented in the various chapters of the report will be a valuable document for the scientific community in general, researchers, scholars and college students in particular and also policy makers, planners and development agencies working in agriculture, environment and food security issues.

I would like to thank Dr. Sissel Hansen and Dr. Arne Grønlund for their helpful comments and suggestions on the previous version of the report. Special thanks are due to the AGRILOSS project (no. 8126.3) for financial assistance given to draft the report and Dr. Marianne Bechmann for providing some funds to finalize the report.

Ås, 01.02.16
Mehreteab Tesfai

CONTENTS

1 INTRODUCTION

Today, we are facing a challenge to sustain food security and protect the natural resource bases from environmental problems such as climate change. Climate change induced by increased Green House Gases (GHGs) emissions in the atmosphere results in global warming (IPCC, 2001). One of these GHGs is nitrous oxide (N_2O) which is a potent GHGs emitted predominantly from agricultural soils.

Climate change is expected to increase the incidence of extreme weather events such as drought and floods (IPCC, 2007). In Norway, the intensity of precipitation is supposed to increase during the crop growing seasons. This leads to excessive moisture in the soils and hinders agricultural activities. The natural drainage capacity of most of the agricultural soils in Norway is poor (Semb, 1974; SLF, 2013). Poor drainage and high moisture in soils renders conducive environment for denitrification and subsequent emissions of N_2O and eventually N_2. Poorly and imperfectly drained agricultural soils can potentially have large amounts of applied fertilizer N lost through denitrification which can be one of the major contributors to soil N_2O-N emissions (Nash et al., 2012). To improve the drainage conditions of agricultural soils and increase crop production in Norway, subsurface tile drainages were installed in large parts of the agricultural lands in the early 1900s. During the late 1970s, extensive land levelling of agricultural lands were undertaken by mobilizing the farming community in south-eastern Norway with the support of the Norwegian parliament (Lundekvam et al., 2003) aiming at increasing farmers' income and national food production. Subsidies were given for cultivation of new land, subsurface drainage on poorly drained soils were installed and prices of agricultural products were increased. Despite increased production was realized through cultivating more arable lands, the land levelling operation has led to 65% increase in soil erosion and depletion of soil organic matter (Lundekvam et al., 2003). In Norway, at least 10 percent of the total agricultural lands (i.e. 100,000 ha) have inadequate drainage (SSB, 2010). Although 60 percent of Norwegian agricultural land is equipped with artificial drainage systems, the maintenance of these drainage systems are somewhat overlooked (Njøs, 2005).

A recent report by SLF (2013) estimates that Norwegian agriculture is responsible for 10 percent of Norway's GHG emissions and for about 70 percent of N_2O national emissions. According to the climate prediction report by Hanssen-Bauer et al. (2015) climate change up to the end of 21st century, will result to wetter autumns and warmer summers in the south and eastern part of Norway where most of the agricultural arable lands are located. Under such climate scenarios, the problems of poor drainage in agricultural soils will exacerbate even more and emission of nitrous oxides (from soils with impended drainage) will rise tremendously unless measures are taken to mitigate nitrous oxide emissions. Poor drainage conditions contribute to denitrification and subsequent emissions of N_2O-N from agricultural soils (Granli and Bøckman, 1994).

So far, a comprehensive review on the process, measurements and mitigation measures of N_2O emissions from agricultural mineral soils hardly exists in Norway. Moreover, there are still many uncertainties in our understandings with regard to the measurements, modelling and mitigation measures of N_2O emissions. Hence, I propose to review the state of the art of nitrous oxide emissions from Norwegian agriculture perspectives but also incorporating relevant research experiences of other countries. This review entails the processes involved in N_2O formation and emissions; research efforts made to measure N_2O emissions from agricultural soils and mitigation measures applied and future research needs. The main purpose of this report is to review and bring reader's attention to some recent scientific works on N_2O emissions from agricultural soils. Possible mitigation options are also discussed. I would like to remind readers that this review is not an exhaustive one and was not meant to be, but it helps to improve our understandings on the drivers of N_2O production and emissions at a process level, measurement techniques, modelling, mitigation measures and future research needs on nitrous oxide emissions in Norway, in particular. The report focuses on the direct emission of N_2O from agricultural soils.

During the review process, we firstly systematically searched relevant scientific publications and technical reports, project overviews, and statistical data from relevant official web sites. In total, 260 published studies conducted between 1974-2015 that are composed of papers, reports were reviewed and cited (Table 1.1).

This review report is divided into nine chapters. Chapter 1 (i.e. Introduction) gives an overview of the state of Norwegian agriculture in relation to greenhouse gases emissions particularly nitrous oxide. Chapter 2 describes the significance of N_2O and its pathways and factors controlling the formation and emission of N_2O. Chapter 3 deals with the measurement techniques of N_2O and modelling experience and some of the strengths and limitations of the measurement techniques. Moreover, it gives examples of simple models used to estimate emissions and some process-based modelling approach that can be used to predict N_2O emissions with known level of uncertainty. Chapters 4 and 5 review research results and discuss the effects of soil management practices and soil moisture and drainage systems on nitrous oxide emissions, respectively. The role of soil pH freezing-thawing effects including soil drying rewetting effects on nitrous oxide production and emissions are elucidated in chapters 6 and 7, respectively. Some mitigation measures to reduce N_2O emissions are discussed in chapter 8. Finally, in chapter 9 we concluded by identifying knowledge gaps and by highlighting future research questions regarding measuring and mitigating N_2O emissions from agricultural soils under Norwegian conditions. From chapters 3 to 7, relevant research experiences of other countries on measurements and mitigation of N_2O are also incorporated to learn lessons and to recommend those practices that are relevant to the Norwegian conditions.

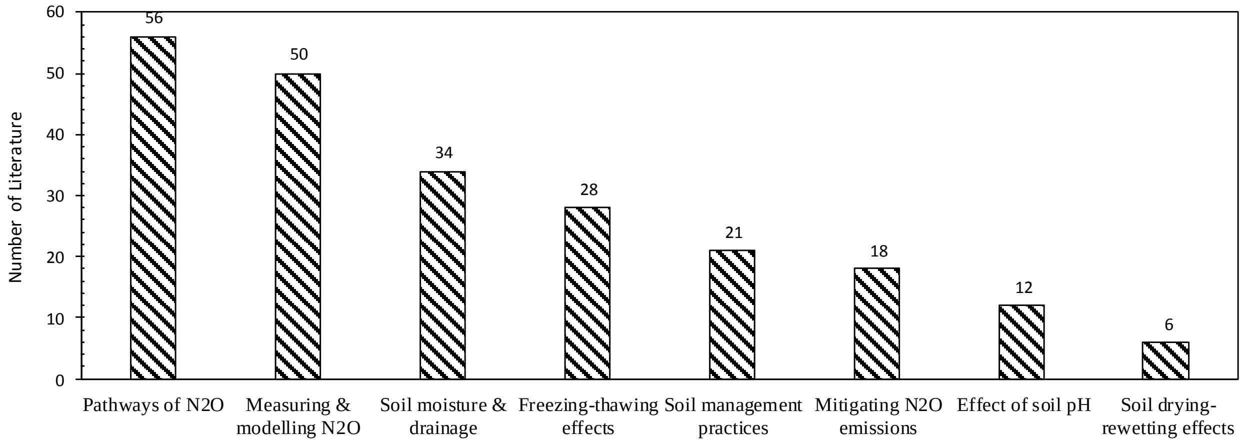

Figure 1.1 Number of literatures reviewed and cited in this report in each topic area (n ~ 225)

2 NITROUS OXIDE

This chapter highlights the significance of nitrous oxide (N_2O) as a greenhouse gas and its effect on climate change and food production. Moreover, it gives a description of the pathways of N_2O to production and emissions processes and the main factors that affect the N_2O emissions from agricultural soils.

2.1 Why N_2O?

Increasing atmospheric content of some trace gases causes serious environmental concern. One of such gas is nitrous oxide- a chemical compound with the formula N_2O (N-O-N). Nitrous oxide is one of the most important GHGs showing a steady increase in the atmosphere due to human activities including agricultural practices notably fertilizer application and fossil fuel combustion. Globally, anthropogenic activities account for more than 50 percent of total N_2O emissions, to which agriculture contributes 81 percent (Isermann, 1994). It is estimated that annual global emissions from agriculture is about 6×10^3 Gg N_2O-N. Of which, 31 percent is contributed by indirect emissions via nitrate leaching, sewage, runoff and atmospheric deposition; 42 percent of the emissions sourced from direct emission of cultivated arable soils and the remaining 27 percent of the emissions caused from animal production (Nevison, 2000; IPCC, 2007). In Norway, the emissions of N_2O from agricultural soils amounted to 1.57 Metric tonnes (calculated in CO_2-equivalents) in 2013. The agricultural soils accounted for about 64 percent of the total Norwegian N_2O emissions in 2013 or about 2.9 percent of the total Norwegian GHG emission of that year (NEA, 2015).

Nitrous oxide is the third largest anthropogenic contributor to greenhouse gases after CO_2 and CH_4. Global anthropogenic emissions of N_2O presently amounts to $4{,}5 \pm 0{,}6$ Tg N yr^{-1} since preindustrial era (Khali and Rasmussen, 1992; IPCC, 2007). The atmospheric N_2O concentration has increased by about 20 percent over the past century and rises steadily at a rate of 0.25 percent per year. Nitrous oxide increased approximately linearly by about 0.8 ppb per year over the past few decades (IPCC, 2007). Despite low concentration of N2O in the atmosphere (~310 ppb) on a molecular weight basis, it has a radiative force of about 300 times that of CO_2 per kg gas in a 100 years perspective (IPCC, 1995; IPCC, 2007) and an average atmospheric lifetime of about 150 years (IAEA, 1992). Furthermore, the breakdown of N2O to NO in the stratosphere results in the depletion of ozone layer (Crutzen and Lelieveld, 2001). Ravishankara et al (2009) coted 'N_2O emissions currently is the single most important ozone-depleting emission and is expected to remain the largest throughout the 21st century'. It has been estimated that doubling the concentration of N_2O in the atmosphere would result in a 10 percent decrease of the ozone layer. This would increase the ultraviolet radiation reaching the earth surface by 20 percent (Crutzen and Ehhalt, 1977) which could result in increased skin cancer and other health-related problems (Lijinsky, 1977). Limiting N_2O emissions would enhance the recovery of the ozone layer from its depleted state and would reduce the anthropogenic forcing of the climate system, representing a 'win-win' for both ozone and climate. Hence, there is a need to broaden the role of N_2O in its wider definition of global change so as to embrace its effect on the ozone layer (Smith, 2010).

Despite N_2O hazardous effect on climate change and human well-being, application of N-fertilizers to agricultural soils will continue in the future. The challenge now and in the future will be how to balance the trade-offs between reduction of N_2O emissions from agricultural soils on one hand and increasing food production on the other hand?

2.2 How is N_2O produced?

To predict how much N_2O is produced from each unit of fixed N (chemically or biologically) that is added to the soil, we must first understand how and where N_2O is produced in the biosphere, what sinks exit for the gas and how the gas moves from where it is produced into the atmosphere. Considerable amounts of N_2O are emitted from natural and cultivated soils through microbial processes, the most important being nitrification and denitrification (Bowden, 1986; Tiedje, 1988). The general requirements for biological denitrification are:

- The presence of bacteria, fungi, other denitrifying eukaryote or archaea possessing metabolic activity;
- Suitable electron donors such as available organic carbon compounds;
- Anaerobic conditions or restricted supply of O_2; and
- Availability of nitrogen oxides: NO_3^-, NO_2^-, NO, or N_2O as terminal electron acceptors.

Nitrous oxide is produced in soils mainly in the course of two contrasting microbial processes, namely nitrification and denitrification. Denitrification is considered as a much more potent source of N2O than nitrification in grassland soils (Saggar *et al.,* 2013) and it accounts globally for about 60 percent of total N_2O emissions to the atmosphere (Mosier *et al.,* 1998a). Both nitrification and denitrification are controlled by the availability of mineral N; soil temperature; mechanisms reducing the redox potential of the soil such as soil wetness (Skiba and Smith, 2000), soil texture and organic carbon; (Sahrawat and Keeney, 1986; Granli and Bøckman, 1994), pH; availability of trace metals; and soil microorganisms. Denitrification enzymes require several metal ions as cofactors including iron, copper and molybdenum (Ferguson, 1998).

(i) Nitrification: Nitrification is an aerobic (O_2 present) process that converts ammonium (NH_4) into nitrate (NO_3) with N_2O as a by-product. This happens when the supply of O_2 is limited by diffusional constraints where by the nitrifying bacteria can use nitrite as an electron acceptor and reduce it to NO and N_2O. Otherwise, the process will proceed as NH_4^+, NO_2^-, NO_3^-. In other words, nitrification occur when soil bacteria convert fertilizer N from the ammonium form (NH_4^+) and/or NH_4-N mineralized from soil organic matter (SOM) to the nitrite (NO_2^-) and nitrate (NO_3^-) forms. In order for nitrification to proceed, fertilizer needs to be added in the form of NH_4^+ or other forms (such as anhydrous ammonia [NH_3] or urea) which are converted to NH_4^+ in the soil. Nitrification is more likely to occur in drier conditions and in coarse-textured soils where rates of N_2O production by nitrification tend to be smaller (Williams *et al.,* 1992).

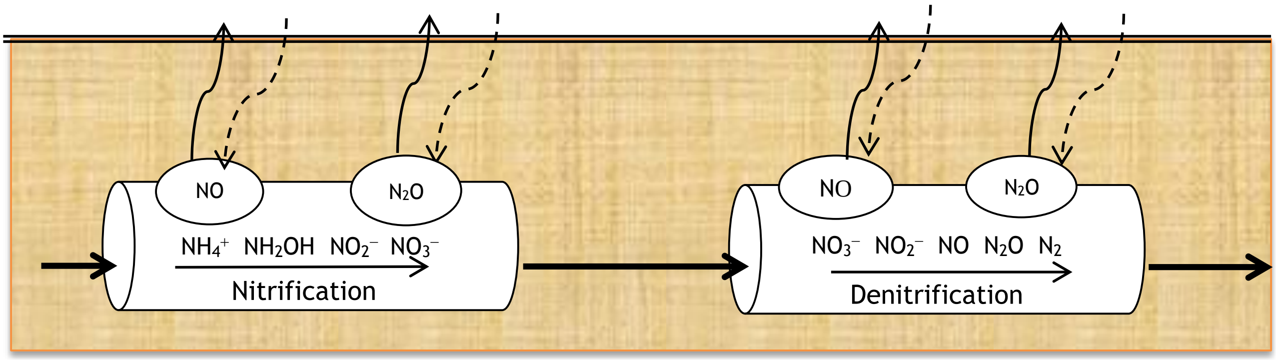

Figure 2.1 Conceptual 'Hole-in-the-pipe' model describing N_2O and NO production/ consumption by the microbial processes nitrification and denitrification. Adapted from Firestone and Davidson (1989) and Davidson et al. (2000).

(ii) Denitrification: Denitrification is an anaerobic (O_2 absent) process that converts nitrate (NO_3^-) into nitrogen gas (N_2) with N_2O being produced as an intermediate product. In other words, denitrification of NO_3^-, N_2O and ultimately to molecular nitrogen (N_2). During denitrification, soil bacteria convert NO_3^- to NO_2^-, then to nitric oxide (NO) gas, and then to N_2O in a sequence of reactions. In order for denitrification to proceed, N must be present in the form of NO_3^-. The NO_3^- can be present either because fertilizer NH_4^+ has been converted to NO_3^- via nitrification, or if NO_3^- is added directly, for example as ammonium nitrate (NH_4NO_3). Denitrification is commonly thought to be the dominant process responsible for the N_2O production in constructed wetlands (Kadlec and Knight, 1996). In general, denitrification is most prevalent in wetter conditions and in fine-textured soils (Velthof *et al.,* 1996).

To elucidate N trace gas production during nitrification and denitrification, Firestone and Davidson (1989) and Davidson et al. (2000) developed a simplified conceptual approach called the "Hole-in-the-pipe" model (see Figure 2.1). In this conceptual model, the amount of N_2O and NO released during microbial N turnover is specified as a function of the flow rate through the "pipe" and the loss rate through the "holes".

The sensitivity of both processes on environmental factors such as pH, temperature, oxygen status, substrate supply, etc. differs. However, both nitrification and denitrification can occur simultaneously in neighbouring micro sites dependent on the surrounding environmental conditions, specifically oxygen and substrate availability. Hence, NO_3^- produced during nitrification can immediately be consumed in denitrification (Nielsen *et al.,* 1996).

Table 2.1 Differences between denitrification and nitrification in terms of their effect on N_2O production and emissions

Denitrification	Nitrification	References
▪ Denitrification is an anaerobic process	▪ Nitrification is aerobic process	Williams *et al.* (1992)
▪ $NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2$	▪ $NH_4^+ \rightarrow NO_2^- \rightarrow N_2O$ or ▪ $NH_4^+ \rightarrow NO_2^- \rightarrow NO_3^- \rightarrow N_2O$	Granli and Bøckman (1994)
▪ Denitrification tends to occur under wet conditions and in slowly draining soils that do not allow for rapid replenishment of O_2	▪ Nitrification tends to occur in moderately well-drained soils and during drier periods.	Davidson *et al.* (2000)
▪ The end product of denitrification could be N_2 under extreme O_2 stress and highly water saturated conditions.	▪ The end product of nitrification is N_2O not N_2.	Davidson *et al.* (2000)
▪ Denitrification requires dissolved organic carbon and is therefore enhanced in soils that have higher carbon levels,	▪ Nitrification does not require organic carbon.	Venterea, *et al.* (2008)
▪ Emissions from the denitrification process are based on anaerobic conditions, NO_3^- concentrations, soil temperature, and the concentration of soluble organic carbon.	▪ N_2O emissions from the nitrification process depend on soil moisture, soil temperature, and soil NH_4^+ concentration.	Li *et al.* (1992)

2.3 Factors affecting N_2O production and emissions

2.3.1 The factors

The most important factors affecting the production of N_2O by nitrification and denitrification are NO_3^- and NH_4^+ concentrations; water content of the soil; soil temperature (both of which affect microbial processes); the amounts of O_2 which controls denitrification; soil pH which influences nitrification and denitrification rates as well as the ratio of N_2/N_2O, and availability of water soluble carbon which is used as energy source for denitrifying bacteria (Bremner, 1997). A brief description is given below:

i) Nitrogen availability: The primary reason for enhanced N_2O emissions from agricultural soils are increased N inputs by mineral fertilizers, symbiotic N_2 fixation, and animal waste application. Generally, the rate of denitrification increases with increasing NO_3^- content in soil under conditions suitable for denitrification (e.g. high moisture) and when factors such as temperature and available organic C are not limiting. Under most circumstances, the presence of NO_3^- inhibits the rate of N_2O reduction to N_2, which results in a higher N_2O/N_2 ratio at similar moisture and oxygen contents. Production of N_2O by nitrification is also enhanced as the soil concentration of the substrate $NH4^+$ increases. Hence, the application of N fertilizers or manure is usually followed by an increase in N_2O emission (Granli and Bøckman, 1994).

ii) Moisture and aeration: Figure 2.2 illustrates schematically the relationships between soil water content (expressed as water filled pore space, WFPS) and N_2O and N_2 emission due to denitrification and nitrification. At low soil water content, N_2O emission is low because microbial activity is low and the O_2 supply is ample so that nitrification goes all the way to NO_3– and denitrification rates are low. With increasing water content, mineralization rate increases and nitrification increasingly produces N_2O. Also denitrification becomes significant with a high N_2O/ N_2 ratio as O_2 diffusion becomes impeded. At high soil water content (above 80-90 percent WFPS), gas diffusion is severely hindered, denitrification proceeds increasingly towards N_2 but N_2O emissions declines. Thus, a soil water content where both denitrification and nitrification can proceed, will generally give the maximum emission of N_2O. The range of this soil water content is normally 45 to 75 percent WFPS, though some studies have indicated a higher level (Klemedtsson *et al.,* 1988; Hansen *et al.,* 1993).

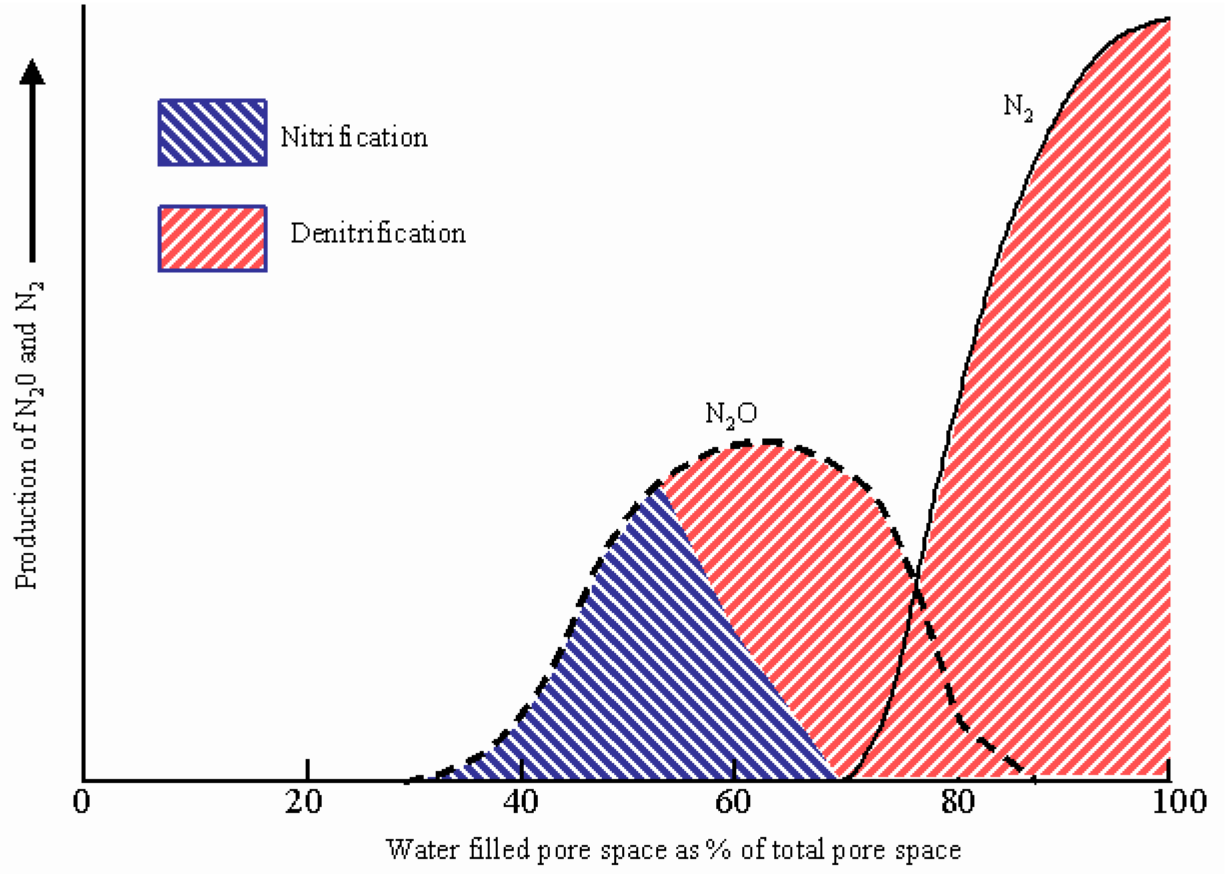

Figure 2.2 The effect of increasing soil moisture content (measured as water filled pore space) on the emission of nitrous oxide and di-nitrogen gasses (Granli and Brockman, 1994).

iii) Soil temperature: Like other biological processes, nitrification and dentrification rates increase with increasing temperature within a certain range. Higher temperature favours a higher ratio of N_2O/NO_3- from nitrification (Goodroad and Keeney, 1984). As soil temperature increases, N_2O emissions also increase, at least up to 37 °C (Castaldi, 2000 as cited by Dalal, *et al.,* 2003).

iv) Organic carbon: Input of degradable plant and animal materials to soil can create conditions favourable for N_2O formation. Microbial activity is enhanced, O_2 is consumed and anaerobic sites can develop. Similarly, soils with high levels of organic carbon content and high water content (≥ 60 % WFPS) have greater propensity for N_2O formation than soils with low levels, notably after application of nitrates N. Under such conditions, very high emission rates have been observed from peat soils manipulated to drainage in Northern Norway (Kløve et al., 2010).

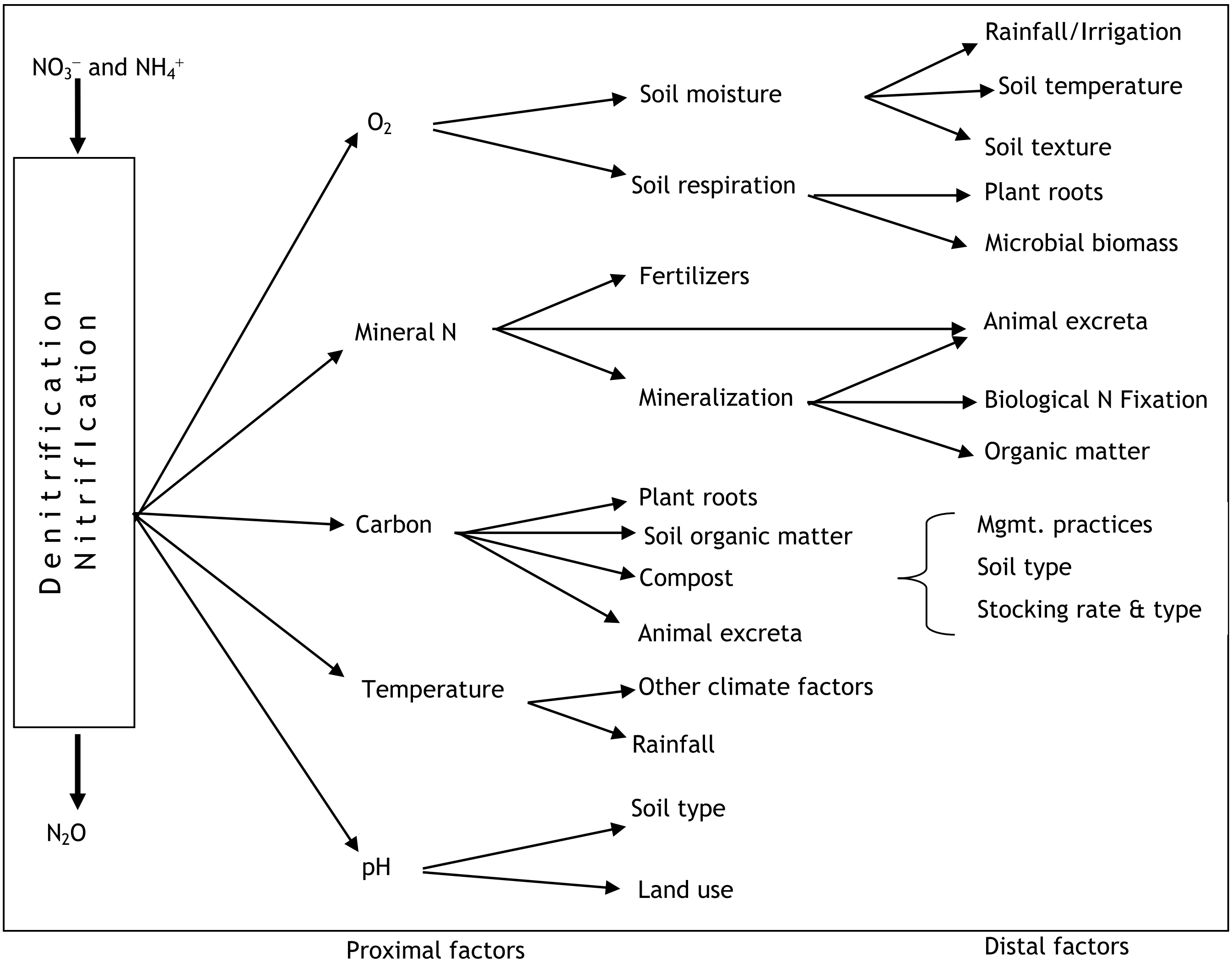

Figure 2.3 Schematic diagram of factors affecting N_2O production in agricultural soils (after Cecile *et al,* 2001).

v) Soil pH: The effect of pH on N_2O emission from soils is complex. Conflicting results are reported. Where denitrification is the main source of N_2O, emissions tend to increase with increasing pH at least in acid soils that have pH below 5 to 6 (Granli and Bøckman, 1994). Where nitrification is the main source of N_2O, emissions tend to increase with increasing pH, at least in the range of pH 6 to 8. However, the rate of N_2O production from autotrophic nitrification decreases with increasing pH in acid soils up to pH 5 (Granli and Bøckman, 1994).

2.3.2 The processes

The N_2O emission at the soil surface is the result of nitrogen production and consumption processes. N_2O emissions originating from agricultural land use include direct and indirect emission of N_2O. Direct emissions are those that occur from N sources within a farming system. While, indirect emissions are those from nitrogen losses through nitrate leaching from agricultural fields into adjacent systems, ammonia volatilization and subsequently re-emissions as N_2O from surface waters or following redeposition of NH_3 to land. Of applied fertilizer or N excreted by animals, 10 to 30 percent may be volatilized as NH_3 (Bouwman *et al,* 2002), which itself is deposited somewhere in the surrounding region, relatively close to its source. N_2O emitted at the soil surface is a mixture of ground water-derived N_2O of mostly agricultural origin, and N_2O produced in soil that originates from industrial, agricultural and natural sources. Industrial sources make up about 20 percent of all anthropogenic sources. Human activity is thought to account for 30 percent and tropical soils and oceanic release account for 50 percent.

Figure 2.3 depicts the interdependency among the environmental factors affecting N_2O emissions. Both nitrification and denitrification are affected by a number of proximal soil factors, such as oxygen content, water content, temperature, mineral N content, C contents and pH (Tiedje 1988; Groffman and Tiedje 1989; Groffman 1999; de Klein et al 2001; Wallenstein et al, 2006). However, these proximal factors are in turn affected by various more distal regulator factors such as management practices, climate, soil type and nutrient supply, which make the regulation of the two processes rather complex (Sirivedhin and Gray, 2006). For example, soil oxygen supply, often regarded as the main factor affecting N_2O emission (Frolking *et al.,* 1998; Luo *et al.*, 1999a), is regulated by soil water content, which in turn depends on rainfall or irrigation and soil texture. As a result, peak N_2O emission or denitrification rates are often found following rainfall or irrigation events (de Klein *et al. 1999*; Luo *et al.,* 1999a).

Due to the heterogeneous nature of both proximal and distal factors, N_2O emissions exhibit a large spatial and temporal variability. Hence, single point measurements at long time intervals are not sufficient to accurately estimate seasonal or annual losses of N_2O on a field scale basis (Mosier and Heinemeyer, 1985). The full characterization of N_2O losses requires, therefore, a large research effort of near-continuous measurements on spatially integrated areas. While considerable experimental work has led to the generalizations about the effects of distal and proximal factors on denitrification and on the N_2O:N_2 ratio (Zaman and Nguyen, 2010). The interactions among these factors are likely to control denitrification and the N_2O:N_2 ratio under field conditions, and these interactions remain poorly understood. We understand the general trends of change in the N_2O:N_2 ratio with changes in some of the individual factors. However, how these factors interact to affect denitrification and N_2O:N_2 ratios under various edaphic and environmental conditions remains poorly understood (Saggar *et al,* 2013).

2.4 Summary

In general, the rate of nitrification increases with increasing N content, oxygen content and decreasing pH, but it decreases with increasing available organic carbon and rising soil temperature. On the other hand, denitrification increases with increasing N content, increasing soil temperature and increasing available organic carbon content but it decreases with increase aeration and at low soil pH. Complete denitrification is promoted by high soil-water content, neutral soil pH, high soil temperature, low rates of O_2 diffusion and the presence of labile C. The balance between factors promoting and factors hindering N_2O emissions determines the outcome of N_2O emissions from a given agricultural system.

3 N_2O MEASUREMENTS & MODELLING

In this section, measurements techniques used to quantify nitrous oxide emissions from agricultural soils are described. And attempts made to develop models to predict N_2O emissions from different land use systems, are reviewed. The estimation of N_2O emissions is still highly uncertain, due to their large variability in time and space. Large variability is caused by the variable rates at which the processes of nitrification and denitrification occur. These processes in turn are controlled by biophysical and chemical conditions in soil microsites, which often show strong non-linear relationships with emissions of N_2O (Bouwman *et al,* 2010). There are broadly two methods for measuring N_2O emissions and theses are flux chamber and micro-meteorological techniques.

3.1 Chamber methods

3.1.1 Introduction

Owing to the dependency of microbial N_2O production and consumption processes on environmental controls such as substrate availability, redox potential and temperature, N_2O fluxes from soils are notoriously variable across various temporal and spatial scales. However, understanding spatial variability of N_2O fluxes is essential to better constrain the magnitude of soil–atmosphere exchange of N_2O and to design statistically valid measurement programs so as to determine flux rates from plot to regional levels. To date, the most widely used measuring technique for quantifying soil N_2O fluxes is the closed chamber technique. Chamber techniques have been used to estimate soil-surface gas emissions for more than eight decades and still remain the most commonly used approach. Chambers can be grouped into two types according to whether the flux is calculated at constant (steady-state) or changing (non-steady-state, NSS) gas concentration. In Norway, the NSS chambers are widely used to measures N_2O emissions from agricultural soils (Photo 3.1).

i) Non-steady state chambers: The NSS chambers are used to measure the soil-surface flux of relatively inert gases such as CO_2, CH_4, and N_2O. For e.g. N_2O emission rates are commonly determined by enclosing the atmosphere above the source (soil, manure, or water body) and measuring the increase in headspace N_2O concentration over time. In Non-Steady State (NSS) chambers, the flux of the gas of interest (F_{N_2O} µg m^{-2} h^{-1}) is calculated using the rate of change of its concentration (dc/dt; mol mol^{-1} s^{-1}) inside the chamber during deployment time (Rochette and Hutchinson, 2005) as presented below:

$$F_{N_2O} = \frac{N_2O}{dt} \times \frac{V_C}{A} \times \frac{Mn}{V_m} \times 60 \qquad [1]$$

where F_{N_2O} is the N_2O flux (µg N_2O-N m^{-2} h^{-1}), d_{N_2O} (ppm N_2O min^{-1}) is the rate of change in gas concentration in the chamber headspace, V_c is the chamber volume (L), A is the surface area covered by the chamber (m^2), M_n is the molecular mass of N in N_2O (28 g N mol^{-1}) and V_m is the molecular volume of the gas at the mean temperature during chamber deployment (L mol^{-1}).

Photo 3.1 Non-steady state closed gas chamber installed in the arable land of south eastern Norway (left photo) and grassland in western Norway (right photo) fields (photo taken by Mehreteab and Sissel).

The V_c is re-calculated for each chamber twice during the growing season i.e. just after fertilisation and before crop harvest by averaging the height from the soil to the top of the frames at four sides ($4 \times 5 = 20$) to account for soil settling.

Cumulative N_2O emissions (kg N_2O-N ha^{-1} $season^{-1}$) are calculated by linear interpolation as mean of the cumulative fluxes of the chambers times the number of days between two adjacent sampling events (Nadeem *et al.,* 2012; Hansen *et al.,* 2014).

One upcoming new method for investigating spatial variability of trace gas fluxes is the use of the fast-box method (Hensen *et al.,* 2006). Here, a chamber is linked to a fast and precisely operating N_2O analyser (e.g. tunable diode laser, TDL). This allows a significant reduction in closure times, so that chamber positions can be changed in minutes, and spatial variability can be explored. By contrast, with standard gas chromatograph (GC) techniques, closure times of 30–60 min are commonly used.

ii) Steady state chambers: In Steady State (SS) chambers, the flux of the gas of interest (F_c) is calculated under constant chamber CO2 concentration $[CO2]_{ch}$ (Rochette and Hutchinson, 2005). An increase in $[CO2]_{ch}$ during chamber deployment has a direct impact on F_c. SS chambers are designed to reduce this impact by measuring F_c at constant $[CO2]_{ch}$. Flow-through steady-state (SS) chambers offer several advantages compared to NSS chambers. Because they offer control on the chamber gas concentration (G), air temperature, and humidity, flux in SS chambers can be measured under conditions that are closer to ambient. Also, their design lends itself more easily to automation and near-continuous flux monitoring. On the other hand, they are more complex to operate than NSS chambers; require on-site gas analyzers; are usually limited to the measurement of one gas at a time; and their performance is sensitive to pressure gradients between the inside and the outside of the chambers.

Gas flux measurements at steady state can also be achieved without air flow through the chamber. Non-flow-through SS chambers have variously been labelled in the past as a static chamber, absorption chamber, or alkali trap chamber. They contain a vessel that is supported above the soil surface and filled with a known amount of a substance that reacts with the gas of interest. Such chambers are typically deployed for long periods, often 12 or 24 h, and the amount of gas trapped by the substance is determined by laboratory analysis. For more information on this type of chamber, the reader can refer to a recent review by Rochette and Hutchinson (2005) and Smith and Conen (2004).

3.1.2 Advantages of chamber method

Closed static gas chambers are simple to use, inexpensive and allow us to study treatment effects as well as to carry out specific process studies. They also permit us measurement of very small fluxes; are relatively inexpensive to build and use; and can be adapted to a wide range of field conditions and experimental objectives; and they allow process-based studies of N_2O emission from soils (Fowler *et al.,* 1997). The closed chamber technique has a relatively good confidence in the N_2O emission measurements due to the fact that the chambers remained in the soil (10 cm depth) throughout the study period (Rochette and Eriksen-Hamel, 2008). Apart from this, the close chambers are robust; locally made stainless steel metal; easy to operate and install in the field. Moreover, they do not also require power supply. The flux chamber techniques can be applied to fragmental landscape and field experiments with multiple small plots and are also suitable to measure fluxes under unstable meteorological conditions (Yao *et al.,* 2009).

3.1.3 Disadvantages of chamber method

The use of closed chamber technique is often known to be associated with severe shortcomings owing to effects on environmental conditions (e.g. temperature effects, soil compaction, plant damage, disturbance of diffusion gradients (Davidson *et al.,* 2002; Butterbach-Bahl and Kiese, 2011). As the method is labour intensive and limited coverage of soil surfaces (usually <1 m^2) is possible, the spatial heterogeneity is often not sufficiently addressed over time. Moreover, collar insertion in the soil lead to cutting of plant roots (Heinemeyer *et al.,* 2011). The closed chamber technique is often associated with high uncertainties due its limited measurement intervals: weekly-to-monthly. In such conditions, the contribution of fluxes during peak emission periods (for e.g. following fertilizer application or during spring–thaw periods) are often not captured. Although the problem of the temporal coverage of flux measurements is increasingly addressed by using automated chamber systems, the problem of the spatial representativeness of chamber-based measurements cannot be easily solved. Spatial variability occurs not only in agricultural but also in natural systems [Ball *et al,* 2000; Butterbach-Bahl *et al.,* 2002) and is often driven by small-scale changes in soil properties (texture, soil organic carbon, gas diffusivity or water availability), plant cover or nutrient availability.

Modification of the microenvironment, pressure distribution, leakage or contamination by lateral diffusion of N_2O and variability of N_2O fluxes due to sampling site and sampling strategy are additional limitations. The main disadvantage of chamber technique is that it measures N_2O emission over a relatively small area and thus a large number of measurements and chambers are required to deal with the large spatial and temporal variability in emissions, which hampers the extrapolation of the results to larger scale. Moreover, precautions should be taken when handling air samples during sampling, storage and analysis; when designing and deploying chambers; and when determining dG/dt to achieve high-quality NSS chamber measurements (Rochette and Bertrand, 2007.

3.2 Micrometeorological method

3.2.1 Advantages of Micrometeorological methods

Micrometeorological techniques involve measurements of N_2O in the atmosphere at two or more points above the soil surface, in combination with meteorological measurements of wind speed, wind direction, and air temperature (Denmead *et al,* 2000). They are derived from the mathematical description of turbulent mass and energy transport above in a relatively large, flat, and homogeneous field. The micrometeorological techniques measure N_2O emissions on a field-scale, and thus spatially integrate N_2O flux measurements. They are non-intrusive and can provide temporally and spatially integrated estimates of the exchange of most gases of interest in agricultural ecosystems. Micrometeorological methods are well suited to measuring real-time fluxes over a large area in response to management interventions. N_2O flux measurements by micrometeorological methods allow small-scale variability of fluxes to be averaged and provide continuous observations of fluxes. The obtained flux estimates for a much larger area and is fundamental for developing and testing up-scaling approaches. Following recent advances in measuring techniques, specifically owing to the commercial availability of laser instruments that allows high precision, accuracy and sensitivity as well as high temporal resolution (less than 1 Hz), the number of studies where micrometeorological methods for e.g. eddy covariance (EC) or gradient techniques in conjunction with TDL or quantum cascade laser spectrometers, used to derive N_2O fluxes for areas more than 0.5–1 ha, is steadily increasing (Eugster *et al.,* 2007; Molodovskaya *et al.,* 2011). The EC technique provides continuous measurements over a large area, without interfering with the processes of gas exchange between the source and the atmosphere (Denmead, 1995; Aubinet *et al.,* 2000).

3.2.2 Disadvantages of micrometeorological methods

The disadvantages of micrometeorological techniques are that they require large homogeneous field sites; are less reliable with low wind speed and atmospheric stability; not suited for comparing different agricultural practices within the same area; and are costly due to expensive N_2O analysis/equipment (Fowler *et al.,* 1997; Rochette and Bertland, 2007). Moreover, the technique is not appropriate in hilly terrain.

3.3 Approaches to quantify denitrification (N_2O and N_2)

The two most commonly used approaches for determining denitrification rate from measurements of N_2 and N_2O production include a technique based on the acetylene (C_2H_2) inhibition (AI) of N_2O reduction (Tiedje, 1988) and an isotopic method using substrates enriched in 15N that allows subsequent 15N gases to be determined by isotope-ratio mass spectrometry (Mosier and Klemedtsson, 1994). More recently, direct quantification of N_2 has been attempted using airtight systems containing Helium (He) or Argon (Ar) either with continuous flow of He+O_2 (Butterbach-Bahl *et al.,* 2002) or a closed system with periodic headspace sampling and replacing the sampled volume with He. The reader can refer to the recent review by Groffman *et al.* (2006) on the methods available to measure and calculate denitrification in terrestrial systems. Some of the advantages and disadvantages of these techniques are summarized in Table 3.1.

Table 3.1 Summary of denitrification measurement methods: their advantages and disadvantages.

Measurement method	Advantages	Disadvantages
▪ Acetylene inhibition (AI) technique Approaches:	▪ Simple to conduct ▪ Can run large number of samples at a time ▪ Removes the spatial and temporal variability of denitrification rate	▪ Can only be used in NO_3^- dominated systems ▪ Inhibits nitrification; and can underestimate denitrification ▪ Slow diffusion of C_2H_2 into soil or sediments limits blockage of N organic
▪ 15N tracer technique	▪ Considered better than Argon technique ▪ Gives reliable estimates of denitrification	▪ Laborious process ▪ Requires costly instruments ▪ Addition of 15N to the N limiting condition results in overestimation of denitrification
▪ Calcium carbide (CaC_2) granules ▪ In situ chambers in field ▪ Static cores	▪ Useful in studying the effect of soil and environmental factors on denitrification and denitrification enzyme activity assay	▪ Rapid decomposition of C_2H_2 by microbes ▪ Contamination of C_2H_2 with other gases can affect denitrification ▪ Scavenging of NO leading to underestimation of denitrification
▪ Direct N_2 quantification	▪ No labelled N or inhibitor is added ▪ Highly sensitive method can even detect low denitrification changes ▪ Can be used to study temperature and moisture effect	▪ Can only be used in enclosed incubation experiments ▪ Complex and difficult system ▪ Not suitable for longer period ▪ Challenge to measure N_2 against high atmospheric N_2 concentration
▪ Molecular approaches Polymerase chain reaction ▪ DNA microarray technique ▪ Immunological techniques	▪ Give reliable and realistic qualitative indication of enzymes and encoding genes involved in denitrification ▪ Can be used both in aquatic and terrestrial	▪ Require very expensive equipment ▪ Require technical expertise to extract, analyse and interpret enzymes and encoding genes ▪ Useful to study microbial diversity

Source: adapted from Saggar *et al.* (2013).

3.4 Modelling N_2O emissions

As a major source of N_2O production from agricultural soils, the denitrification process must be an important part of any process-based soil N_2O model. Nitrous oxide emissions are highly variable both in space and time. And the estimation of N_2O emissions from individual fields based on actual measurements is very costly. Moreover, the high temporal and spatial variability of agricultural nitrous oxide (N_2O) emissions from soil makes their measurement at regional or national scales impractical. Accordingly, robust process-based models are needed. Models have, therefore, become an important means for improving our understandings of the complex interactions between drivers of N_2O emissions, for estimating N_2O emissions from agricultural systems and for evaluating practices that can reduce emissions. Moreover, models provide a valuable complement to measurement, extending limited temporal and spatial measurements to other climatic and edaphic conditions, regions and scales. These models range from relatively simple national inventory or accounting models (like emission factors) to detailed process-based models, and use factors and constants that are derived from measurements under experimental and controlled conditions. The main example of the inventory/accounting model is the IPCC methodology for estimating national greenhouse gas inventories (IPCC, 2006) while the DNDC model (Li *et al.*, 1992a; 1992b) is one of the best known biophysical models for estimating N_2O emissions.

3.4.1 Modelling approaches

Simple empirical models can be used to estimate emissions, but these are limited to the types of soils and management practices used in the model parameterisation. More detailed, process-based models can be used to create emission scenarios and to examine the potential impacts of novel mitigation strategies. The success of a process-based denitrification model depends upon how well the model simulates the other processes affecting the soil environment. A recent review by Heinen (2006) identified three types of denitrification models: i) microbial growth models; ii) soil structural models; and iii) simplified process models that represent the denitrification rate in terms of easily measurable parameters such as soil moisture, temperature and NO_3^- concentration.

i) IPCC emission methodology: N_20 emissions from agriculture are estimated using the IPCC guidelines for national greenhouse gas inventories (IPCC, 1996). National N_20 emissions are estimated on a per year basis, using default emission factors of IPCC (1996). Since, this default factor does not distinguishes between different climates, soils and crops, each countries tries to adjust the emission factor to its own conditions. The guidelines distinguish three sources of N_20 emission from agriculture: for example in Norway, the N_2O emissions are estimated by Statistic Norway (Sandmo et al. 2014) as follows:

(a) Direct emissions of N_20 from agricultural soils: (from application of synthetic fertilizer, animal manure, biological N-fixation, crop residues and cultivation of organic soils). According to report made by Mattilsynet (2015) synthetic fertilizer consumption in Norway is estimated to be about 102.2 Gg N yr^{-1} (Gg = 10^9 g) for 2013-14. Urea and ammonia are the main source of volatilization of fertilizer N. The use of urea and ammonia as fertilizer is negligible in Norway, and therefore the fraction of fertilizer N volatilized is set to be zero which is contrary to the IPCC default value of 10 percent. Direct emission of N_20 from application of synthetic fertilizers in Norway is estimated to be 1.38 Gg N_2O-N yr^{-1} and indirect emission from leached N and runoff N to be 0.83 Gg N_2O-N yr^{-1}.

In Norway, all animal excreta which is not deposited during grazing, is used as manure. The amount of N excretion is estimated around 78 Gg N using parameters specific for Norwegian husbandry. These data are based on number and type of animal. It is assumed that 20 percent of the N in animal excreta is volatilized. Further, the emission from manure management is taken into account. The total N_2O emission from animal manure applied to soil was estimated around 0.6 Gg N (Table 3.2).

Biological N-fixation (F_{BN}) is estimated around 8 Gg N yr^{-1}. Using the IPCC default emission factor (1.25 percent), the N_2O emission from biological N-fixation is estimated around 0.1 Gg N_2O-N yr^{-1} which is added to the crop residues. The N amount in crop residues returned to soil is assumed to be equal to the amount of N in all the yield. N_2O emission from crop residues is estimated to be about 1.1 Gg N_2O-N yr^{-1} (Table 3.2). The area of cultivated organic soil in Norway is approximately 1.8 x 105 ha (Johansen, 1997). Using the IPCC default N_2O emission factor of 5 kg N_2O-N ha^{-1} yr^{-1} (IPCC, 1996), the N_2O emission from organic soils is estimated to be 0.9 Gg N_2O-N yr^{-1} (Table 3.2). The total direct emission of N_2O from agricultural soils is about 3.98 Gg N_2O-N yr^{-1} (Table 3.2).

Table 3.2 Direct N_2O emission (Gg N_2O-N yr^{-1}) from agricultural soils in Norway (N_2O_{DIRECT}).

Emission due to	Gg N_2O-N yr^{-1}	% of the total	Mt $CO_{2\text{-equ.}}$ (2010)*
Synthetic fertilizer use (N_2O_{SN})	1.38	35	0.6
Crop residues returned to soil (N_2O_{CR})	1.10	27	0.1
Cultivated organic soil (N_2O_{OS})	0.90	23	0.3
Animal manure (N_2O_{AW})	0.60	15	0.6
Agricultural soils in country N_2O_{DIRECT})	3.98	100	1.6

Source: Adapted from Statistic Norway (2009) and * Grønlund and Hansen (2010).

(b) Emissions of N_2o from animal production: i.e. emission from droppings on pastures and animal waste storage/treatment previous to application to agricultural soils. The total N_2O emissions from animal excreta on pastures ($N_2O_{ANIMALS}$) was estimated to be ca. 18 Gg N_2O-N yr^{-1}. $Frac_{GASM}$ (fraction of livestock N excreta) that volatilizes as NH_3 and NO (kg NH3-N + NO_x-N/kg N excreted) constituted 0.2 Gg N_2O-N yr^{-1}. EF_3: emission factor for animal N excreta on pastures (kg N_2O-N kg^{-1} N excreted) estimated to be 0.02 Gg N_2O-N yr^{-1}. $N_2O_{ANIMALS}$: N_2o emission from grazing on cultivated and uncultivated land to be 0.29 Gg N_2O-N yr^{-1}.

(c) Indirect emissions of N_2O induced by Agriculture: through N losses by volatilization, leaching and surface runoff, and sewage production can be distinguished as:

- *Atmospheric depositions of N compounds fertilizes:* soils and waters are sources of N_2O emission ($N_2O_{(G)}$). As mentioned above, N_2O volatilization from synthetic fertilizer use in Norway is considered negligible. And, it is assumed that 20 percent of animal manure is volatilized. Using a default N_2O emission factor of 1 percent (IPCC, 1996) ($N_2O_{(G)}$) was estimated to be 0.12 Gg N_2O-N yr^{-1}.
- *The fraction of fertilizer and manure N lost:* to leaching and runoff ranges between 10 and 80 percent. In the IPCC methodology for estimating national N_2O emission, a default factor value of 30 percent is proposed for leached and runoff N. Leached and runoff N is nitrified and denitrified both in groundwater and surface drainage, in rivers and in marine coastal areas. However, to calculate N_2O emission from leached and runoff N, only one emission factor is used for the three environments; i.e., nitrification and denitrification in groundwater, surface drainage and rivers are pooled together. In 1996, N_2O emission from leached and runoff N in Norway was estimated to be about 1.37 Gg N_2O-N yr^{-1}.
- *Human consumption of food results:* in sewage production and food waste. As sewage N is nitrified and denitrified, it is a source of N_2O emission. Estimation of N_2O emission from human sewage is based on protein consumption, fraction of N in protein and number of people in the country. The IPCC methodology suggests to use FAO food statistics as input data for protein intake. These FAO statistics are estimates of whole sale food supply. In Norway, about two thirds of the whole sale food supply are actually consumed. The other third consists of solid waste, which is also a source of N_2O. Aakra and Azzaroli (1997) have treated sewage and solid waste as one post, and estimated the total emission of N_2O ($N_2O_{(s)}$) to be about 0.26 Gg N_2O-N yr^{-1} (Table 3.3). The N_2O emission factor used is the IPCC default factor (i.e. 1 percent). $N_2O_{(s)}$ includes emission both from food produced in Norway and from imported food.

Table 3.3 N_2O emission (Gg N_2O-N yr^{-1}) indirectly induced by Norwegian agriculture (N_2o $_{INDIRECT}$).

	Gg N_2O-N yr^{-1}	% of the total	Mt $CO_{2\text{-equ.}}$ (2009*)
N_2O produced from N leaching and runoff $N_2O_{(L)}$	1.410	77	0.3
N2O produced from human sewage $N_2O_{(s)}$	0.255	14	n.d.
N_2O produced from atmospheric deposition of NO_x and NH_3 $N_2O_{(G)}$	0.156	9	0.1
N_2O produced from N used in agriculture $N_2O_{(INDIRECT)}$	1.821	100	0.4

Source: Adapted from Statistic Norway (2009) and * Grønlund and Hansen (2010).

According to IPCC (1996), the total N_2O emission indirectly induced by agricultural activities N_2O $_{(INDIRECT)}$ = $N_2O_{(G)}$ + $N_2O_{(L)}$ + $N_2O_{(s)}$ in Norway is about 1.8 Gg N_2O-N yr^{-1} (Table 3.3).

Total Emissions of N_2o from Norwegian Agriculture: When estimated according to the IPCC guidelines for national greenhouse gas inventories total emission of N_2O from Norwegian agriculture is 6.1 Gg N_2O-N yr^{-1} (Table 3.4).

Table 3.4 Total emission (Gg N_2O-N yr^{-1}) from agriculture in Norway $N_2O_{(TOTAL)}$.

	Gg N_2O-N yr^{-1}	% of the total
Direct N_2o emission from agricultural soils (N_2O_{DIRECT})	3.98	66
Indirect N_2o emission from N used in agriculture ($N_2O_{INDIRECT}$)	1.82	35
N_2o emission from grazing on cultivated and uncultivated land ($N_2O_{ANIMALS}$)	0.30	5
$N_2O_{(TOTAL)}$	6.10	100

Numerous attempts have been made to model N_2O emission from soils (e.g. Elliott and de Jong, 1993; Kaiser *et al.,* 1996; Li *et al.,* 1996; Velthof *et al.,* 1996c; Muller *et al.,* 1997a; Potter *et al.,* 1997). However, as N_2O emissions are affected by a range of factors, each of which exhibits a large spatial and temporal variability, the development of an N_2O model is not an easy task and hence its general applicability is often limited. Van der Weerden (1999) adapted the model of Muller *et al.* (1997b) to describe N_2O emissions from non-grazed and cultivated arable land. This micro-scale mechanistic model also relies on Michaelis-Menten kinetics to elucidate N_2O fluxes via nitrification and denitrification, which requires the following inputs: Mineral N content, Soil temperature, WFPS, Soil pH, Potential denitrification activity, and Cultivation records.

Several detailed biochemical process-based models of N-gas emissions have been developed in recent years to provide site-specific and regional scale estimates of N_2O emissions. Examples of overseas regional scale models now receiving some attention include the Denitrification-Decomposition (DNDC) model (Li *et al.,* 1992), the Carnegie-Ames-Stanford Approach (CASA) model (Potter *et al.,* 1996), and an approach using boundary-line analysis (Elliott and de Jong, 1993).

ii) The DNDC model: The DNDC (Denitrification–Decomposition) model is a process oriented model simulating temporal changes in the levels of soil organic carbon (SOC) as well as nitrogen cycling. The Field-DNDC model contains four main sub-models: a) the soil climate sub-model calculates hourly and daily soil temperature and moisture fluxes in one dimension, b) the crop growth sub-model simulates crop biomass accumulation and partitioning, c) the decomposition sub-model calculates decomposition, nitrification, NH_3 volatilization and CO_2 production, d) whilst the denitrification sub-model tracks the sequential biochemical reduction from nitrate (NO_3) to NO_2^-, NO, N_2O and N_2 based on soil redox potential and dissolved organic carbon. (Li et al. 1992a &b; Li, 2000).

The denitrification submodel calculates hourly denitrification rates, N_2O and N_2 emissions during periods when the WFPS exceeds 50 percent. During these periods of low oxygen availability, it is assumed that denitrifiers are most active. The DNDC model has been calibrated and validated with a number of field studies including the use of daily N_2O emissions (Li, 1995). The model was validated using seven independent datasets extracted from various field experiments and showed no significant differences between measured and predicted cumulative N_2O emissions. As a process-based model, DNDC is capable of predicting the soil fluxes of all three terrestrial GHGs namely, N_2O, CO_2, and CH_4, as well as other important environmental and economic indicators such as crop production, ammonia (NH_3) volatilisation and nitrate (NO_3^-) leaching. The DNDC model has been widely used internationally, including in the EU nitrogen biogeochemistry projects NOFRETETE and NitroEurope (Zhang *et al.,* 2015).

iii) DAYCENT model: Conceptually, the DAYCENT model is based on the assumption that the total gas emission from soil are proportional to N cycling through the system and the soil gas diffusivity determines the relative amounts of the respective N gas species emitted from soil. The DAYCENT model includes sub-models for soil organic matter decomposition, land surface parameters, plant productivity, and trace gas fluxes. The sub-models for N_2O and N_2 fluxes from denitrification assumes that the N gas flux from denitrification is controlled by the most limiting factor among soil nitrate concentrations, carbon substrate supply, and oxygen supply (Dalal *et al.,* 2003).

Although DAYCENT simulates the seasonal pattern of N_2O fluxes from grasslands reasonably well, it simulates these fluxes very poorly on a daily basis because of the significance of diurnal fluctuations in N_2O emissions due to changes in soil water and temperature. In addition, soil mineral nitrogen appears to be consistently underestimated (Parton et al., 1998). Further improvements in simulation modelling are required, taking into account the soil texture differences, clay mineralogy, and integration of the 'hot spots' over time and field scale (Dalal, *et al.,* 2003).

3.4.2 Model validation

Model validation is indispensable prior to its use and validation should be performed with field observation data to assess simulation accuracy and guide further model development (Dietiker et al., 2010). Testing denitrification models is complicated by the fact that under field conditions, there are many interacting processes that will affect denitrification rates. e.g., heat and water transport through soil and other N transformation processes. Therefore, the success of a process-based denitrification model depends not only on the quality of the denitrification component but also on its ability to simulate soil conditions accurately; requirement to soil data and cost for providing the soil data of sufficient quality. For example, denitrification rates are highly dependent on soil moisture. Improvements to the soil water modelling component have been found to improve predictions of N_2O emission (Saggar *et al.,* 2007; Norman *et al.,* 2008). Therefore, there would be little point in improving the denitrification process in a model, if the poor performance of some other component was the limiting factor.

Models are frequently validated against N_2O measurements. This is not necessarily a good test of the denitrification model, as N_2O is a product of both nitrification and denitrification, while denitrification also produces N_2. So, it is possible for a model to predict the N_2O emissions correctly while incorrectly simulating the overall denitrification rate. Frolking *et al.* (1998) found in a comparison of four different process-based models (DNDC, DayCent, CASA, and Expert-N) that even when the models produced similar N_2O fluxes, they often produced very different estimates of gaseous losses as NO, N_2 and NH_3 indicating differences in the simulated N-transformation processes. Therefore, in order to have confidence that a model is simulating denitrification well, it is necessary to ensure that other processes such as NO_3– leaching, NH_3 volatilisation and plant uptake are also well simulated. Even though, it is not usually feasible to measure all the possible forms of N (e.g. N_2O, N_2, NH_3, soil NO_3– and NH_4^+) in a given experiment, the more these are measured well the more valuable the data set will be for model validation (Saggar et al, 2013).

Table 3.5 Form of f_N used, calibaration of N_2O:N_2, indicator of microbial activity of denitrification models.

Model name	Form of f_N	Variables used in N_2O:N_2 ratio	Indicator of microbial activity	References
ANIMO	Zero-order	n.a.	n.a.	Rijtema and Kroes (1991)
DNDC	n.a.	WFPS	Microbial biomass	Li *et al.* (1992)
FASSET	Michaelis-Menten	Temperature, WFPS, clay, depth	Mineralisation rate	Chatskikh *et al.* (2005)
DAYCENT	n.a.	NO_3^-/CO_2 ratio, WFPS	CO_2 concentration	Stehfest and Müller (2004)
InfoCrop	First-order	n.a.	Microbial biomass	Aggarwal *et al.* (2006)
NEMIS/NOE	Michaelis-Menten	n.a.	Site-specific constant	Hénault and Germon (2000); Hénault *et al.* (2005)
NGAS	n.a.	Soil NO_3^-, respiration and WFPS	Respiration rate	Parton *et al.* (1996)
NOE	Michaelis-Menten	Empirical site-based parameter	n.a.	Hénault *et al.* (2005)
PaSim	n.a.	n.a.	Decomposition rate	Schmid *et al.* (2001)
WNMM	First-order	WFPS	Soil organic carbon	Li *et al.* (2007)

Source: adapted from Saggar et al (2013). f_N: effect of soil nitrate.

Table 3.5 lists the form of f_N used in selecting denitrification models, factors considered in calculating the N_2O:N_2 ratio and the approached used to account for microbial activity in different denitrification models. Models such as DNDC and DAYCENT can be used to simulate N_2O production from soil after parameterization with the local data, and appropriate modification and verification against the measured N_2O emissions under different management practices (Dalal *et al.,* 2003).

3.4.3 Model limitations

Factors affecting the simulation or modelling of N_2O emissions are:

- Most mechanistic models include algorithms describing N_2O production in the soil, but models do not include processes and time steps for the actual N_2O emissions from the soil surface (Smith, 2010).
- All process-based models contain empirical approximations at some level, either from limitations of understanding the underlying processes or from practical difficulties in measuring all the

required parameters. The different approaches to denitrification modeling discussed above differ in empirical approximations.

- Simplified process-models can often give good results for situations for which they have been well-parameterised (e.g. Hénault *et al.,* 2005) and may be more suitable for some applications. However, these simplifications limit the model's ability to only providing insight into the simplified processes. For instance, a model that uses a fixed N_2O:N_2 ratio for denitrification products is not capable of determining situations where the N_2O:N_2 ratio changes. Improved understanding of the underlying processes can lead to improvements in the models. However, as N_2O production is the result of many interacting processes, improving the simulated denitrification process alone will not necessarily improve the model's performance if other processes are poorly simulated.
- Unavailability of appropriate parameter values. Many models are validated at field scale simply by comparing measured N_2O emissions with the modelled results. However, this does not indicate which processes are well modelled and where the model is performing poorly.

3.5 Summary

Flux chamber and micro-meteorological techniques are the two major methods used to measure N_2O emissions. Chambers can be grouped into whether the flux is calculated at constant (steady-state) or changing gas concentration (non-steady-state which are commonly used in Norway). The chamber method is simple to use, relatively inexpensive to install; and can be adapted to a wide range of field conditions. However, chambers are cumbersome, high uncertainties due to limited coverage of soil surfaces and small number of measurements. While, micro-meteorological techniques measure time series fluxes over a large area with higher accuracy despite costly and require large homogeneous field sites. Moreover, they are not suited for comparing different agricultural practices within the same area; and the technique is not appropriate for hilly terrain.

Several models have been developed to predict N_2O fluxes from agricultural fields, worldwide. They include from simple national inventory models (like emission factors) to dynamic process-based models such as DNDC and DAYCENT. National inventories of N_2O emissions have advantages in collating annual inventories but may mask significant variations in emission factors on a regional scale. In Norway, there is lack of information on what type model to use and what are the limitations for getting good data on N_2O flux rate using a model? To address this gap, process–based models such as DNDC (which have been widely used internationally including in many EU projects) should be tested, validated and assessed against N_2O emissions prediction ability across different Norwegian agricultural systems.

4 EFFECTS OF SOIL MANAGEMENT PRACTICES ON N_2O EMISSIONS

In this chapter, the effects of soil management practices notably fertilizer applications and tillage methods (associated with soil compaction) on N_2O emissions from agricultural soils are reviewed and discussed.

4.1 Inorganic and organic fertilization

The availability of N, especially in the form of NO_3^- for denitrifying bacteria is the primary requirement for denitrification and therefore the NO_3^- concentration in the soil solution can be one of the principal factors limiting denitrification. Soil NO_3^- concentrations depend on mineralisation and nitrification rates, plant N uptake, microbial immobilisation and NO_3^- movement by leaching and diffusion (Tiedje, 1988; Zaman *et al.,* 2007).

4.1.1 Norway

Hansen *et al.* (2014) measured N_2O emissions following the application of inorganic and organic fertilizers in Western Norway. The objective of the research was to quantify N_2O emissions from fertile grassland soils dominated by sandy loam, and to estimate the response of seasonal N_2O emissions to added inorganic N, cattle slurry (CS) N and clover N. A field experiment with no tractor traffic was carried out during the summers of 2009 (13 May to 29 October) and 2010 (12 April to 20 August) in intensively managed grassland. The experiment followed a completely randomized block design with seven N application rates replicated four times in 2 × 8 m plots. Fertilizer treatments were given as total kg N and added as NH_4NO_3 in or/and CS per ha and year. Ammonium nitrate (AN) and CS were applied manually at annual rates of 0, 100, 150, 200 and 250 kg AN N ha^{-1}, 80 kg CS-N ha^{-1} or as a combination of 200 kg AN-N ha^{-1} and 80 kg CS-N ha^{-1}.

According to results displayed in Figures 4.1 and 4.2, N_2O emissions throughout the two growing seasons showed that high inter-annual variation with up to 10 times higher emission rates from unfertilized soil during a warm and dry year as compared with a cold and wet year. Fitting N_2O emissions aggregated for growth periods to various N sources (ammonium nitrate: AN, cattle slurry: CS, biologically fixed clover N: Ndfa) also showed that i) the inter annual variation in flux magnitude was mainly due to pronounced differences in background emission activity in a warm and a cold year, and ii) the emissions induced by mineral fertilizers during summer were surprisingly constant between 2009 and 2010 (0.11 percent), irrespective of weather. Hansen, et al (2014) reported that a moderate direct response of N_2O emission to fertilization was revealed in well-drained fertile grassland soils, while showing background emissions can be high and variable.

Hansen *et al.* (2014) concluded that N_2O emissions in fertile Norwegian grasslands are to a great extent controlled by inter-annual variations in background emissions and variable contribution of biologically fixed N and CS-N. Together with the circumstantial evidence for clover-induced emissions, this calls for caution when scaling up N_2O emissions from fertilizer-response alone.

In 2009, emissions of soil N_2O-N is directly proportional with the soil NO_3^--N and soil NH_4^+-N levels in the fertilizer applied plots. The control treatments measured lowest emissions of soil N_2O-N and also lowest soil NO_3^--N and soil NH_4^+-N levels. However, during August to September, an inverse relationships was observed between the soil NO_3^--N and soil NH_4^+-N levels and soil N_2O-N flux rates. In other words, higher emissions of N_2O but lower contents of NO_3^--N and soil NH_4^+-N were measured especially after the second harvest. In general, a declining trend of N_2O-N fluxes, NO_3^--N and NH_4^+-N levels discerned in all treated plots (Figure 4.1).

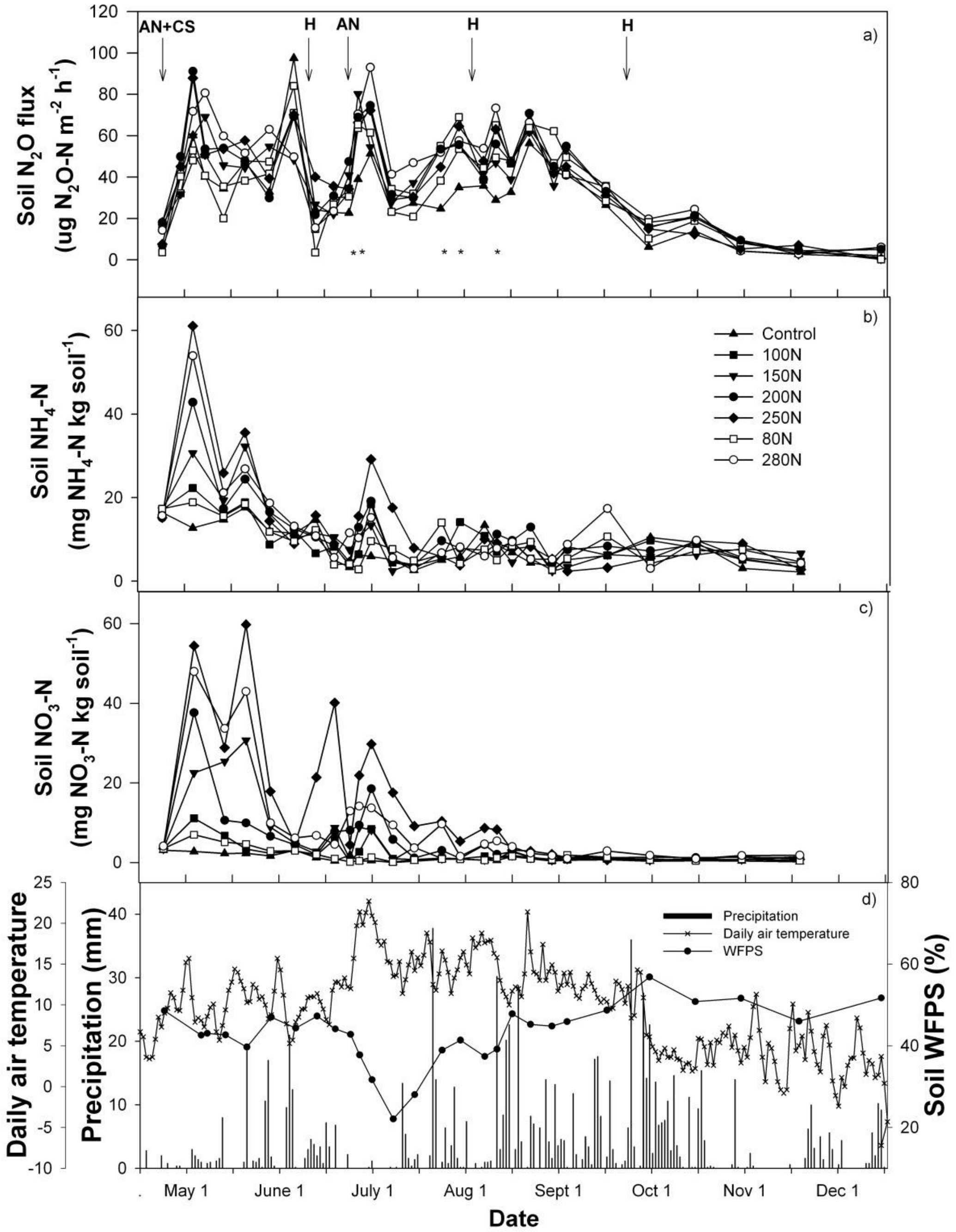

Figure 4.1 Temporal dynamics of a) N_2O fluxes, b) soil NH_4-N concentration, c) soil NO_3-N concentration, and d) ancillary variables measured in intensively managed grassland in Western Norway in 2009. Dates with significant differences in N_2O emission rate are marked with (*).a) n=4; b) and c) Values are from one composite sample per treatment on each date. Fertilizer treatments are total nitrogen added in NH_4NO_3 (AN) or cattle slurry (CS). Black indicators = AN = ammonium nitrate application, White indicators = CS= cattle slurry application, H= harvest. (Adapted from Hansen *et al.,* 2014).

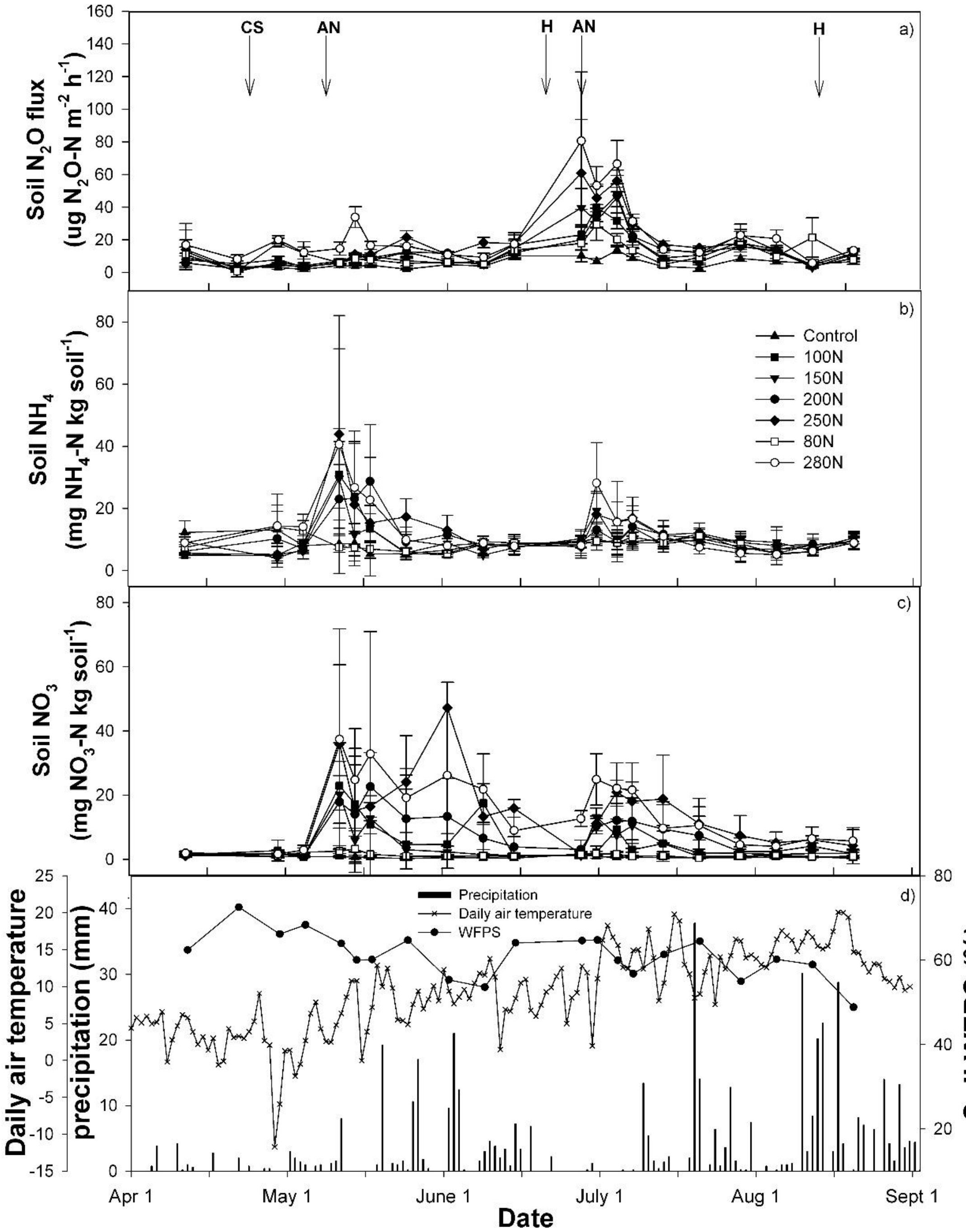

Figure 4.2 Temporal dynamics of a) N_2O fluxes, b) soil NH_4-N concentration, c) soil NO_3-N concentration, and d) ancillary variables measured in intensively managed grassland in Western Norway in 2010. Abbreviations are described in Fig.1. Values in a, b) and c) are averages from 4 replicate plots per treatment. Vertical lines indicate standard deviation. (Adapted from Hansen *et al.,* 2014).

In 2010, an increase in N_2O flux rates and NO_3^--N and NH_4^+-N levels were measured after CS and AN application. Comparatively, the concentrations of NO_3^--N and NH_4^+-N decreased but N_2O-N fluxes rates increased after first harvest and second AN application. Afterwards, a declining trend of N_2O-N fluxes, NO_3^--N and NH_4^+-N levels was observed in all treated plots. The background emissions of N_2O-N fluxes was small (<20 µg N_2O-N/m^2/hr) in 2010 and had low levels of NO_3^--N

(<5 mg/kg) and NH_4^+-N (<20 mg/kg). In both years, the first and last measurements of N_2O flux, NO_3^--N and NH_4^+-N were more or less the same (Figure 4.2).

One of the fertilization treatment in the fertilization trial (referred to as NT) was comparable with the fertilization on the farmers' field surrounding the fertilization trial (3 tons of cattle slurry and 200 kg N (NH_4NO_3 per ha (Hansen et al., 2011). This is referred to as CF. The emissions from NT and CF was compared with emissions from a neighbour field run organic (OF) during the summers of 2009 (13 May to 29 October) and 2010 (12 April to 20 August) (see Table 4.1).

Table 4.1 Cumulative N_2O losses ± standard deviation from grass clover leys in a field trial with no tractor traffic under conventional farm (CF) and organic farm (OF) during the summers of 2009 (13 May to 29 October) and 2010 (12 April to 20 August).

Year	Kg N_2O-N/ha			mg N_2O-N kg/DM herbage		
	NT	CF	OF	NT	CF	OF
2009	1,6 ± 0,2	3,8 ± 3,4	2,0 ± 0,5	132 ± 25	380 ± 437	156 ± 111
2010	0,7 ± 0,3	2,2 ± 1,2	0,2 ± 0,04	71 ± 31	332 ± 203	37 ± 11

Adapted from Hansen *et al.* (2011). DM: Dry matter.

Fertilization had only a minor influence on the N_2O flux in the field trial when there was no tractor traffic in well-drained soil where the N_2O flux rate was ≤100 µg N_2O-N/m²/hr. While, the N_2O flux rate in the field with CF was much higher than the field trial with OF. The variation in N_2O emissions between the trials was large which was strongly affected by fertilization and soil moisture conditions. The maximum N_2O flux rate at each location varied from 90-2000 µg N_2O-N m²/hr in 2009 and 80-1000 µg N_2O-N m²/hr in 2010 (Hansen et al., 2011).

A field experiment was carried out in SE Norway (Frøseth et al 2014; Nadeem *et al.* 2012a) to quantify N_2O emissions from different green manure (GM) management practices such as mulching versus removal of grass-clover herbage during a whole growing season and replacement as biogas residue to a subsequent barley crop. The results showed that grass-clover ley had small but statistically significant higher N_2O emissions as compared with a non-fertilized cereal during the year of green manure production in 2009. Mulching of herbage induced more N_2O emission (+0.37 kg N_2O-N ha^{-1}) throughout the growing season than removing herbage. In spring 2010, all plots were ploughed (with and without GM) and sown with barley, resulting in generally higher N_2O emissions than during the previous year. Application of biogas residue (60 kg NH_4^+-N + 50 kg organic N ha^{-1}) before sowing did not increase emissions neither when applied to previous ley plots nor when applied to previously unfertilized cereal plots. Ley management (mulching versus removing biomass in 2009) had no effect on N_2O emissions during barley production in 2010.

In general, GM ley (mulched or harvested) increased N_2O emissions relative to a cereal reference with low mineral N fertilization (80 kg N ha^{-1}). Based on measurements covering the growing season 2010, organic cereal production emitted 95 g N_2O-N kg^{-1} N yield in barley grain, which was substantially higher than the cereal reference treatment with 80 kg mineral N fertilization (47 g N_2O-N kg^{-1} N yield in barley grain).

4.1.2 Other countries experience

Canada: Field experiments were conducted in 1996–1997 to assess soil N_2O emissions as affected by timing of N fertilizer application and straw removal for crop production under irrigation in southern Alberta. The crops were soft wheat (Triticum aestivum L.) in 1996 and canola (Brassica napus L.) in 1997. N_2O emissions were greater when N fertilizer (100 kg N ha^{-1}) was applied in the fall compared to spring application. Straw removal at harvest in the fall increased N_2O emissions when N fertilizer was applied in the fall, but decreased emissions when no fertilizer was applied. The study showed that N_2O emissions may be minimized by applying N fertilizer in spring, retaining straw, and incorporating it in spring. The estimates of regional N_2O emissions based on a fixed proportion of applied N may be tenuous since N_2O emission varied widely depending on straw and fertilizer management practices (Hao et al. 2001).

United Kingdom: A field experiment was carried out in Edinburgh (UK) from an intensively managed grassland site on imperfectly drained gleysol to compare N_2O emissions from ammonium nitrate (AN), urea (UR), and with urea modified nitrification inhibitors. N_2O fluxes from AN plots were greater than from UR plots probably due to slower hydrolysis of Urea (CH_4N_2O) into NH_4^+ plus no $NO3^-$. There is a scope for reducing N_2O emissions from N-fertilised grassland by applying UR instead of AN particularly in cold conditions on wet soils when grass growth begins in spring. Applying UR with a nitrification inhibitor could cut N_2O emissions further (Dobbie and Smith, 2003).

4.2 Soil compaction

Soil compaction created by tractor traffic is favourable for N_2O production in soil due to its effect on soil aeration. It restricts oxygen diffusion within the soil, thereby increasing the rate of denitrification (Rosswall et al., 1989; Luo et al., 1999a). In grazed pastures, compaction can be caused by animal treading especially with high stocking rates (Drewry et al., 2008) and in arable crop fields, it is caused by heavy machinery traffic during sowing, ploughing, and harvesting particularly when the soils are wet. Soil compaction also affects the nitrification process and thereby influences the supply of NO_3^- for denitrification.

4.2.1 Norway

Soil compaction trials were carried out in western Norway to determine effect of soil compaction on N_2O emissions from grassland soils (for e.g. Sitaula et al. 2000; Sturite et al. 2014).

Sitaula *et al.,* (2000) measured nitrous oxide fluxes in a field experiment in Surnadal (Norway) for four consecutive years (1991-1994) to determine the effect of soil compaction on N_2O emissions. Soil compaction resulted in increased N_2O emission and this compaction effect was more pronounced in NPK fertilized treatments. In unfertilized plots, N_2O emission rate was increased by 44 percent due to compaction, whereas in NPK-fertilized treatment, the average N_2O emission rate was increased by 170 percent and was statistically significant at $p<0.05$. This means that the effect of compaction on increased N_2O emissions was about four times greater than unfertilized treatment (Figure 4.3). Since soil compaction reduces the total soil pore volume (Breland and Hansen, 1996), a higher occurrence of anaerobic sites can be expected in a compacted soil, especially with a high soil moisture status, which lead to increased emissions of N_2O (Figure 4.3).

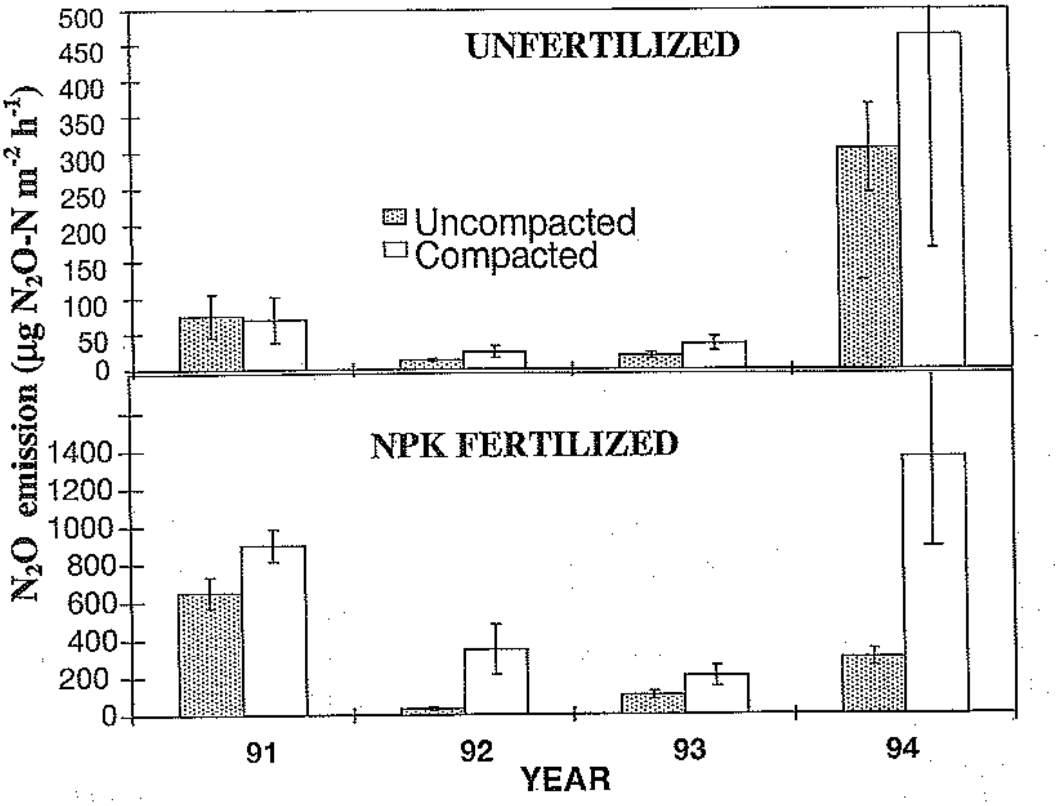

Figure 4.3 N_2O emission rates (mean values for each treatment ±S.E., n =4–16 measurements) in Surnadal field experiments as influenced by soil compaction. Adapted from Sitaula *et al.* (2000).

Similar results were also found at Fureneset in western Norway where soil compaction increased N_2O emissions from mixed grass-clover ley in the first year but emissions were negligible during the second year, indicating that clover can substitute for the input of mineral fertilizer and thus mitigate N_2O emissions during the growing season (Sturite et al. 2014).

4.2.2 Other countries experience

Germany: A field study was conducted in Germany to determine the effect of soil compaction on the fluxes of N_2O in a soil (fine-silty Dystric Eutrochrept) planted with potato (*Solanum tuberosum* L.). The major part (68 percent) of the total N_2O release from the fields during the cropping period was emitted from the compacted tractor tramlines; emissions from the ridges made up only 23 percent. The results indicate that soil compaction was probably the main reason for increased N_2O emission from potato-cropped fields. Soil compaction by tractor traffic strongly increased N_2O emissions (Ruser *et al.,* 1998); whereas soil loosening decreased N_2O fluxes (Flessa *et al.,* 2001).

United Kingdom: A compaction experiment with zero and heavy compaction was carried out in Scotland (UK) on imperfectly drained Cambisols in 1995. The heavy compaction treatment gave a greater N_2O emission than the zero compaction treatment particularly eight days after fertilization despite low rainfall. On the other hand, N_2O emissions were comparatively low (<1000 $\mu g\ N\ m^{-2}\ hr^{-1}$) at high rainfall events (for e.g. just 6 days after fertilization) as shown in Figure 4.4 (Ball *et al.,* 1999).

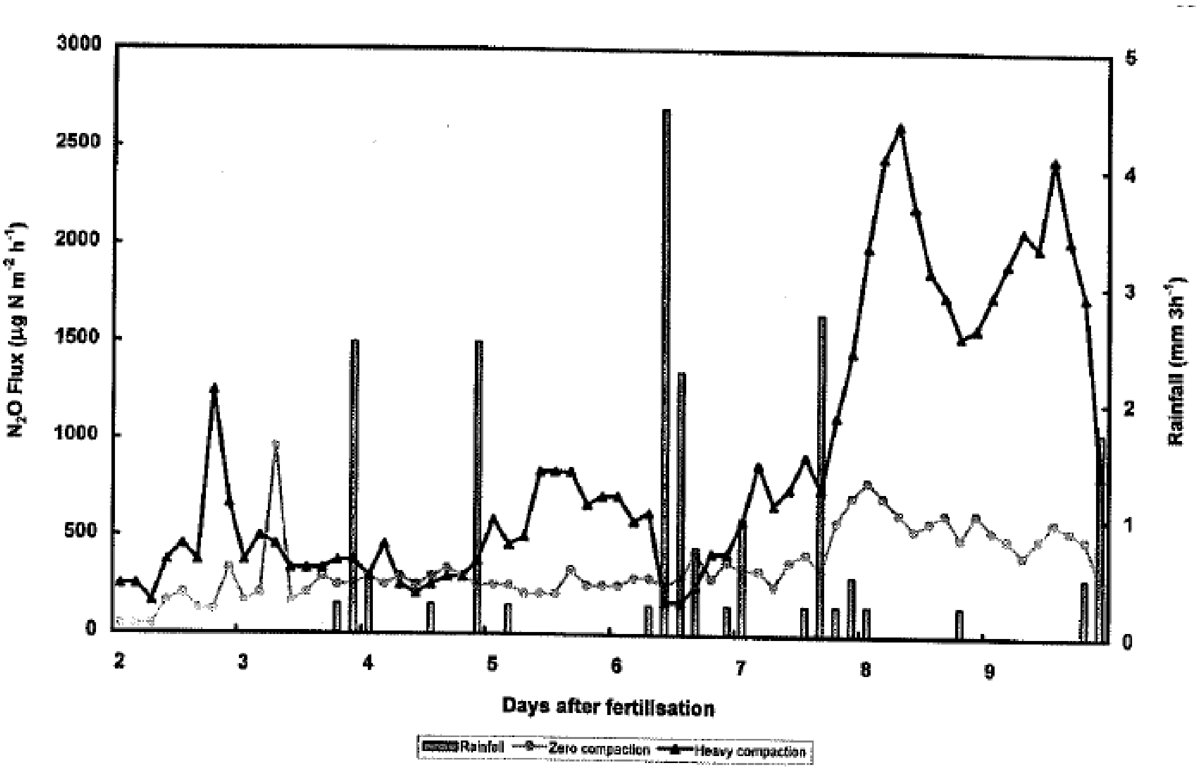

Figure 4.4 Temporal variability of N_2O fluxes and 3-hourly rainfall in the compaction experiment under winter barley, shortly after the main spring fertilization of 110 kg N/ha on 17 April assessed using the automatic chambers. Adapted from Ball *et al.* (1999).

4.3 Soil tillage

The effect of tillage on N_2O emissions is variable and it all depends on climatic conditions and on the time since conversion to no till (NT) or reduced tillage (RT) practice. In the short term, N_2O emissions from NT could be lower than conventional tillage (Wagner-Riddle et al. 2007). Conversely, NT increases N_2O emissions from poorly aerated soils (Rochette, 2008; Ball et al. 1999).

4.3.1 Norway

Yu (2011) has investigated the effect of autumn and spring tillage plus fertilization rates on N_2O emissions from a field site in Ås, Norway. The experiment was carried out in a long-term field trial run by Norwegian University of Life Sciences (NMBU). N_2O emissions were measured periodically throughout the growing season (i.e. April to October 2010) using a NSS chamber method.

Results showed that cumulative N_2O emissions clearly increased with increasing fertilizer rate (0-120 kg N ha^{-1} yr^{-1}). Although not perfectly linear, the increase in cumulative N_2O emissions suggested that about 1 percent of the applied N was lost as N_2O-N which is identical to the Tier 1 emission factor suggested by IPCC (2007) for national GHG inventories. Time of tillage had no significant effect on cumulative N_2O emissions mainly due to counteracting effects of tillage. Emissions were higher in spring ploughed soils as compared with soils that had been ploughed in autumn, presumably because autumn ploughing leads to more N loss throughout winter. Autumn ploughed had lower emissions which could been associated with ploughing- induced changes in soil structure. Both effects cancelled each other out and therefore no effect of ploughing time was seen on an annual basis (i.e. a zero sum game).

Irrespective of the mechanisms involved, it may be concluded that permanent (> 30 years) spring versus autumn ploughing had no fundamental effect on N_2O emission in the studied soil, suggesting that time of ploughing is not a relevant tool for mitigating fertilizer induced N_2O emissions (Yu, 2011). This concluding statement is also shared by Nadeem et al., (2012b; 2014) who conducted fertilization rate and ploughing time effects on nitrous oxide emissions in a cereal field.

4.3.2 Other countries experience

Denmark: A field experiment on a Danish sandy loam soil was carried out by Mutegi *et al.* (2010) to compare the effect of three tillage practices namely, conventional tillage (CT), reduced tillage (RT), and direct drilling (DD) on N_2O emissions. Each of these tillage treatments were further differentiated whether the crop residues were retained (+Res) or removed (−Res). The sampling took place from autumn 2007 to the end of spring 2008.

Results showed that N_2O dynamics were characterized by three peaks: i) during autumn after seed bed preparation (12-43 µg N_2O m^{-2} hr^{-1}), ii) during spring after fertilization (17-52 µg N_2O m^{-2} hr^{-1} and iii) during spring after slurry application (10-32 µg N_2O m^{-2} hr^{-1}). The first two peaks are most likely due to soil disturbance during seedbed preparation and seeding under wet, yet warm autumn conditions. These conditions have increased SOM and crop residues turnover and N mineralization, presumably stimulating N_2O production due to elevated oxygen consumption rates by denitrifying microorganisms. Ploughing in the autumn contributes to increased emissions of N_2O (Ball *et al.,* 1999). In contrast, the other two peaks in spring resulted from transformations of N applied in fertilizers and manure.

Overall, N_2O emissions were 27 and 26 percent lower in DD and RT, respectively, relative to N_2O emissions from CT plots ($P<0.05$). Mutegi *et al.* (2010) observed that in residue removal scenarios N_2O emissions were similar for all tillage treatments, but in residue retention scenarios N_2O emissions were significantly higher in CT than in either DD or RT ($P<0.05$). Irrespective of residue management, N_2O emissions from DD and RT plots never exceeded emissions from CT plots. Retention of residue was estimated to reduce emissions from DD plots by 39 percent and in RT plots by 9 percent, but to increase N_2O emissions from the CT plots by 35 percent. Relative soil gas diffusivity (Rdiff), soil NO_3–N, soil temperature, tillage and residue were important driving forces for N_2O emission ($P<0.05$). A multiple linear regression model using Rdiff to represent the water factor explained N_2O emissions better than a WFPS based model. In this regard, there is need to review the current use of WFPS in N_2O prediction models. Mutegi *et al.,* (2010) concluded that direct drilling has the potential to reduce N_2O emissions when crop residues are returned into the soils, in this case, light textured soils.

United Kingdom: Tillage experiment was conducted on imperfectly drained clay loam soils of Winton series in Scotland by Ball *et al.* (1999). They found that greater emissions of N_2O under no till (NT) than ploughed treatments (Figure 4.5). The greater emissions of N_2O under NT are likely to be associated with reduced gas diffusivity and air filled porosity which presumably is caused by heavy rainfall that has triggered higher water contents near the soil surface than the ploughed treatments. The NT plots showed N_2O peaks (>2000 µg m^{-2} hr^{-1}) after 53, 56, and 61 days of fertilization when rainfall was low. In the same plots, N_2O emissions were small (<1000 µg m^{-2} hr^{-1}) when there was higher rainfall. Although the N_2O flux rates in the ploughed plots (i.e. 20 and 30 cm depth) were considerably lower than NT plots, the changes in N_2O emissions was similar with that of NT plots. The N_2O emissions from the ploughed soil treatment (i.e. 30 cm) were the least (Figure 4.5).

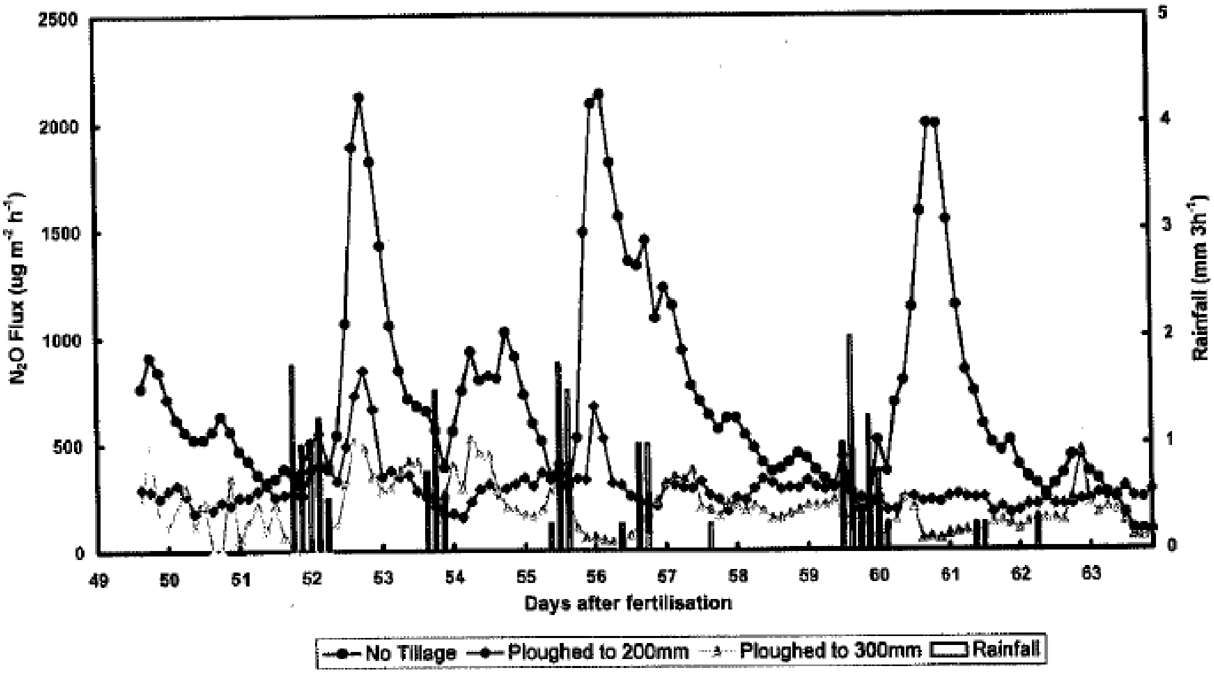

Figure 4.5 Temporal variability of N_2O fluxes and 3-hourly rainfall in the tillage experiment under spring barley 7-10 weeks after sowing and fertilization of 80 kg N/ha in spring 1996, assessed using the automatic chambers. The crop was sown and fertilized on 5 April. Adapted from Ball *et al.* (1999).

Rochette (2008) has written a summarized report on field N_2O emissions under no-till and tilled soils. According to this report, on average soil N_2O emissions under no-till were 0.06 kg N/ha^{-1} (lower) for good aeration, 0.12 kg N ha^{-1} (higher) for medium aeration and 2.0 kg N ha^{-1} (higher) for poor aeration. The ratio of mean cumulative emissions from no-till to tilled soils for good aeration was 0.87, for medium aeration was 1.13 and for poor aeration was 1.50. This means that the mean impact of no-till on N_2O emissions is small in good-aerated soils but often large on soils where aeration is restricted.

4.4 Summary

Under the Norwegian conditions, N_2O emissions from fertilized soils are to a great extent controlled by inter-annual variations of weather conditions and background emissions. In general, N_2O flux rates increase with increasing N fertilization rates and immediately after rain events in arable soils as well as in grasslands soils.

It can be generalized that N_2O emission was higher in compacted than uncompacted soils from the few studies made in Norway, so far. The effects of compaction was significantly higher in the NPK fertilized treatment than unfertilized one. The effect of ploughing time on N_2O emissions is inconsistent which depends mainly on weather conditions that vary from year to year. Overall, time of ploughing (for e.g. spring versus autumn) alone had no measureable effect on growing season N_2O emission. Other environmental factors should also be taken into account when evaluating ploughing time effects on N_2O emission.

5 EFFECTS OF SOIL MOISTURE & DRAINAGE ON N_2O EMISSIONS

Soil moisture is a major driver of N_2O emissions as it regulates the oxygen availability to soil microbes. N_2O emissions have their optimum in the range of 70–80 percent WFPS depending on soil type (Davidson et al., 2000; Skiba and Smith (2000). At higher soil moisture, the major end product of denitrification is N_2 (Butterbach-Bahl *et al,* 2013). The N_2O:N_2 ratio has often been found to decrease with increasing soil water content (Davidson, 1992; Rudaz *et al.,* 1999), particularly when the soil water content exceeds 75 percent WFPS (Weier *et al.,* 1993). Likewise, the measured N_2O:N_2 ratio was highest (≥1) under dry conditions during summer and early autumn when denitrification was relatively inactive (Ruz- Jerez *et al.,* 1994). Changes in N_2O:N_2 ratio with increased WFPS will also depend on the ability of denitrifiers to produce N_2 or N_2O as an end product (Morley *et al.,* 2008).

Another important factor for promoting or hindering N_20 emissions from agricultural soils is the drainage conditions of the soils. Under the context of agriculture, drainage could be defined as the removal of excess water from surface and subsurface agricultural fields in order to enhance crop growth and removal of soluble salts from the soil. Agricultural drainage increases yield, reduces annual yield variability and promotes conservation practices such as conservation tillage. However, poorly drained soils can potentially have large amounts of applied fertilizer N lost through denitrification which can be a major contributor to soil N_2O emissions (Nash *et al.,* 2012; Tesfai et al. 2015). Good drainage has several advantages, among others, i) maintain crop growth and achieve higher yields; ii) promotes effective utilization nitrogen fertilizers and leads to reduce emission of nitrous oxide per kg harvest; iii) enables farming operations in early spring and late autumn; and iv) reduces working hours spent in wet fields (Skaggs and Schilfgaarde, 1999).

5.1 Soil moisture & drainage

5.1.1 Scandinavia countries

Drainage of organic soils and associated effects of N_2O emissions have been well documented in Scandinavian countries (Conen and Neftel, 2010; Kløve et al. 2010). For example, field measurements on cultivated organic soils in Finland for barley fields ranged from 5.4 to 24.1 kg N_2O-N ha^{-1} yr^{-1}, for grass from 1.7 to 11.0 kg N_2O-N ha^{-1} yr^{-1}, for fallow from 3.8 to 37.0 kg N_2O-N ha^{-1} yr^{-1} (Maljanen *et al.,* 2007) and estimated net emissions from a pipe drained peat soils in northern Norway expressed in kg of CO_2 eq. m^{-2} yr^{-1}, was about 0.13 kg for N_2O (Grønlund et al. 2006) and mean annual flux of 1.51 kg N_2O-N ha^{-1} (Kløve et al. 2010) which does not include the winter period.

There is however, little information available on emissions of N_2O from mineral soils with different drainage conditions in agricultural fields of Norway apart from Tesfai and et al. (2015). They carried out a field measurements aimed at quantifying N_2O emissions under three different soil drainage conditions i.e. poorly drained (PD), imperfectly drained (ID) and moderately well drained (MD) plots in a cereal field at Ås in south eastern Norway during the growing seasons of 2011 and 2012. Soil moisture and groundwater levels were monitored and gas samples were collected from closed gas chambers during crop growing seasons (i.e. April to August 2011 and 2012). The gas samples were analysed for N_2O flux by gas chromatograph.

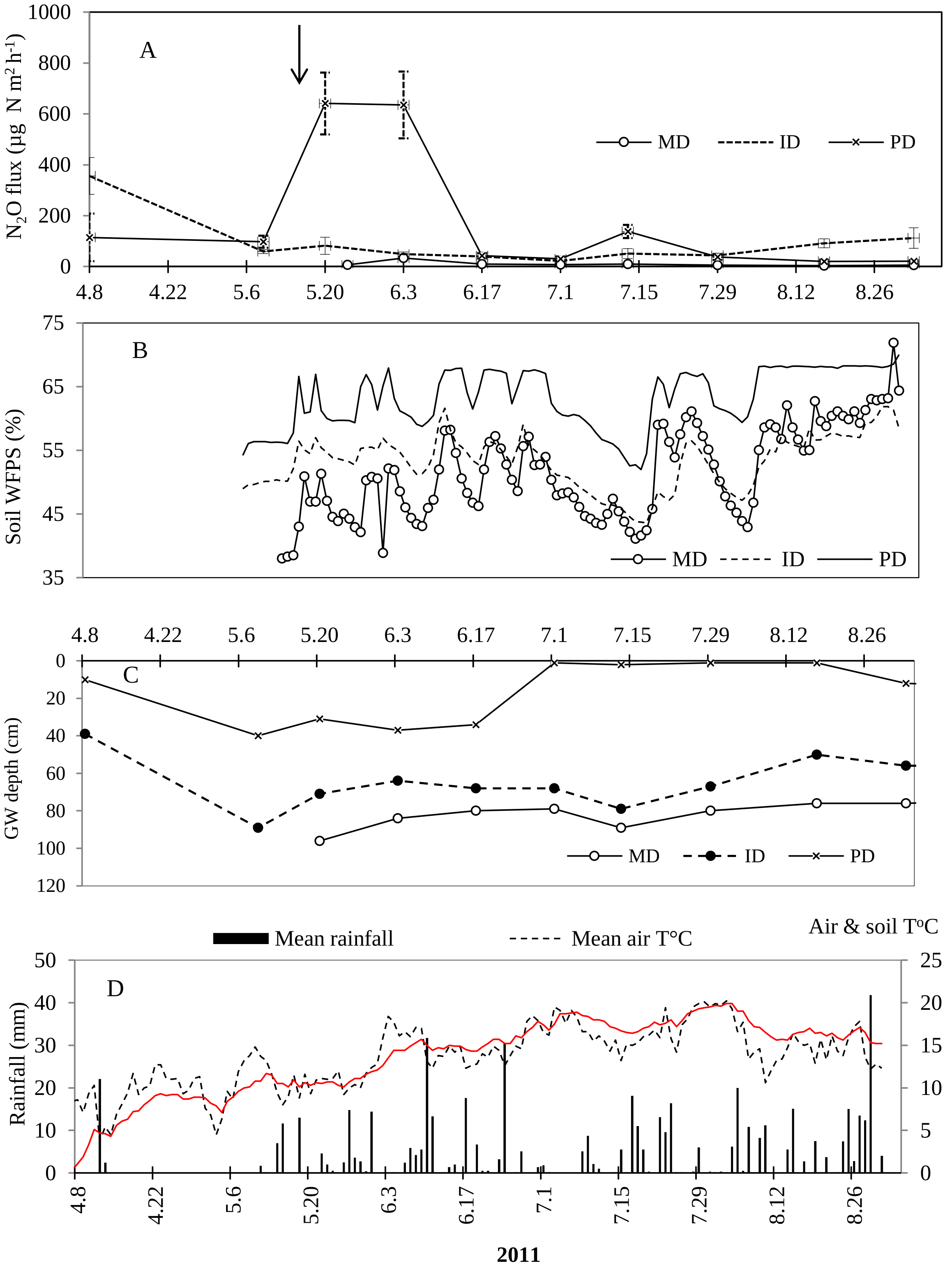

Figure 5.1 Mean N_2O-N flux from MD, ID and PD plots, with standard errors, n = 5, the markers show date of gas sampling), B (mean daily WFPS), C (GW = depth to ground water from soil surface) and D (daily rainfall, air and soil temperature) during April to August 2011. Arrows indicate date of fertilizer application. Source: Tesfai et al. (2015).

In 2011, the highest N_2O flux rates (641 ± 121 µg N m^{-2} h^{-1}, n = 5) was emitted from PD soils one week after fertilisation in which soil NO_3^-contents were high. While, lowest flux (20 ± 6.8 µg N m^{-2} h^{-1}) was measured one week before crop harvest when the soil NO_3^- contents were low (Table 5.1). In ID soils, mean flux rates from the chambers ranged from 22 ± 1.2 to 356 ± 72 and in MD soils varied between 4 ± 0.5 and 33 ± 6.8 µg N m^{-2} h^{-1} in 2011. When comparing the N_2O emissions among MD, ID and PD soils on each sampling day, the emissions were higher from PD soils in 50 percent of the sampling occasions (Figure 5.1).

In 2012, PD soil N_2O fluxes varied in the range of 6 ± 2.8 to 878 ± 229 µg N m^{-2} h^{-1}. N_2O emissions from the PD soils peaked (878 µg N m^{-2} h^{-1}) in mid-June i.e. three weeks after the first fertilizer application and after the first rains as shown in Figure 5.2. A peak flux close to 1607 ± 621 µg N m^{-2} h^{-1} was measured (on 2nd July: second fertilization) from ID soils with lowest flux 11 ± 3,7 µg N m^{-2} h^{-1} on 13 April 2012. In the case of MD soils, flux ranged between 7.6 ± 1.9 to 92 ± 13.6 µg N m^{-2} h^{-1}. All N_2O fluxes from MD soil were below 50 µg m^{-2} h^{-1} except the sample on 27 August 2012 that emitted 92 µg N m^{-2} h^{-1} (Figure 5.2).

The cumulative N_2O-N emissions (kg N ha^{-1}) in MD, ID and PD soils in 2011 and 2012 growing seasons are depicted in Figures 5.3 and 5.4. According to these figures, the cumulative N_2O fluxes from PD soil (6 kg N ha^{-1}) were much higher than ID (4 kg N ha^{-1}) and MD soil N_2O fluxes (0.2 kg N ha^{-1}) in 2011. In 2012, the cumulative N_2O fluxes from PD soil were higher than the ID soil until early July but were lower towards the end of the growing season. On the other hand, cumulative N_2O emissions from MD soil (0.9 kg N ha^{-1}) were much lower than PD and ID soils in both years (10 and 12 kg N ha^{-1}) . Overall, the cumulative N_2O flux rates registered in MD, ID, and PD soils in 2011 were almost twice that of 2012 flux rates (Figure 5.3). In general, poorly drained soils emitted more N_2O-N compared to moderately well drained soils mainly due to higher soil moisture and shallow ground water table.

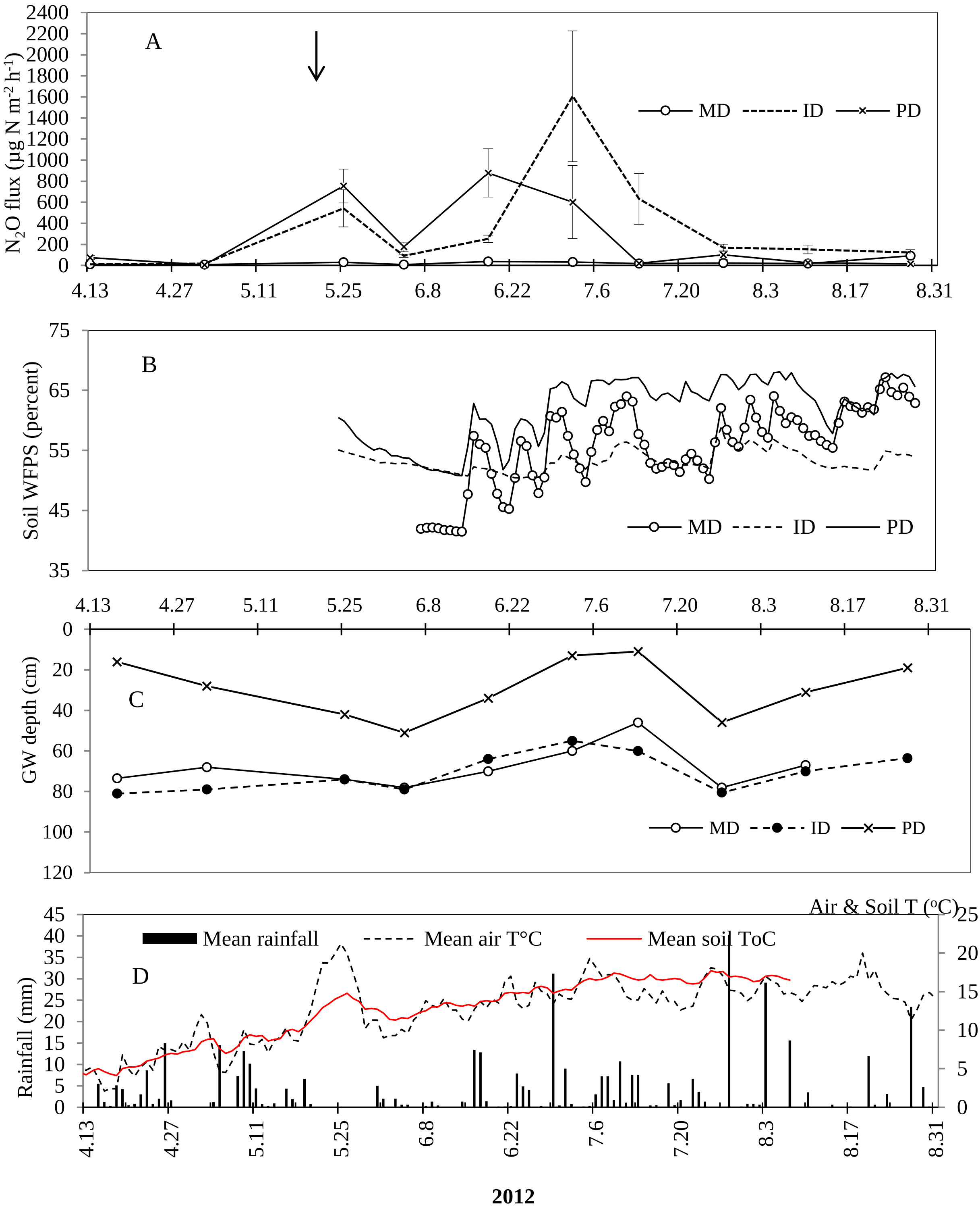

Figure 5.2 Mean N_2O flux from MD, ID and PD plots, with standard errors, n = 5, the markers show date of gas sampling), B (mean daily WFPS), C (depth to ground water from soil surface) and D (daily rainfall, air and soil temperature) during April to August 2012. Note that data on soil temperature is not available after 6 August 2012. Arrows indicate date of fertilizer application. Source: Tesfai et al. (2015).

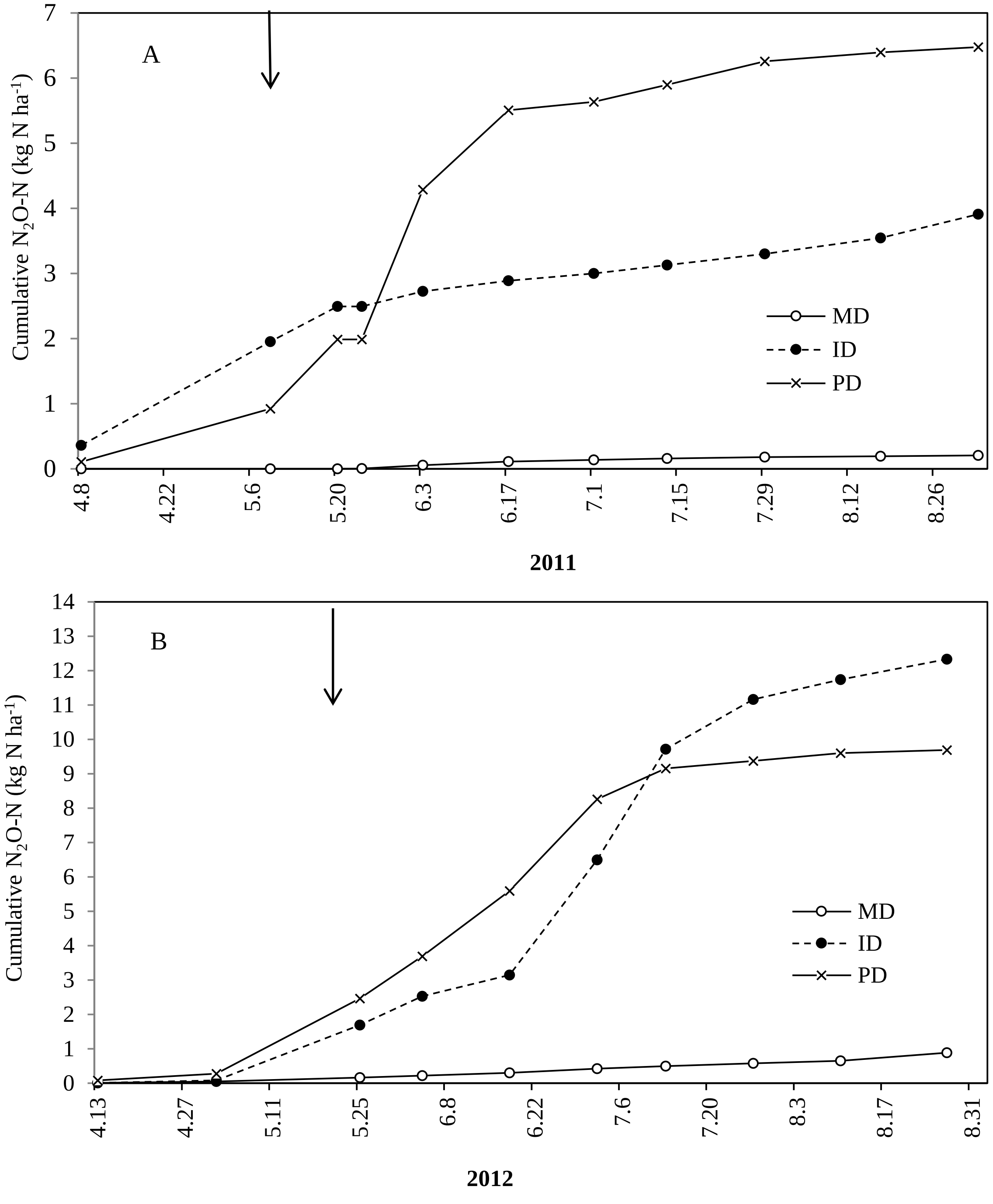

Figure 5.3 Cumulative N_2O-N emissions (kg N ha^{-1}) in MD, ID and PD soils during 2011 (A) and 2012 (B) growing seasons. Arrows indicate dates of fertiliser application. Source: Tesfai et al. (2015).

In western Norway, N_2O emissions were measured in farmers' field at places varying from each other in distance to water table by Hansen *et al.* (2011). Four days after fertilization (70 kg N ha^{-1} as ammonium nitrate), the N_2O emission rate varied from 15 to 1900 µg N_2O-N m^{-2} h^{-1}. During those days, the water table depths ranged from 12 cm to >100 cm. The N_2O fluxes were low after one week of fertilization until the crop harvest (Figure 5.4). Peak N_2O flux (1900 µg N_2O-N m^{-2} h^{-1}) was measured from the plot with shortest distance to water table around the first week of June and then a sharp decline of N_2O flux was observed in all the drainage points.

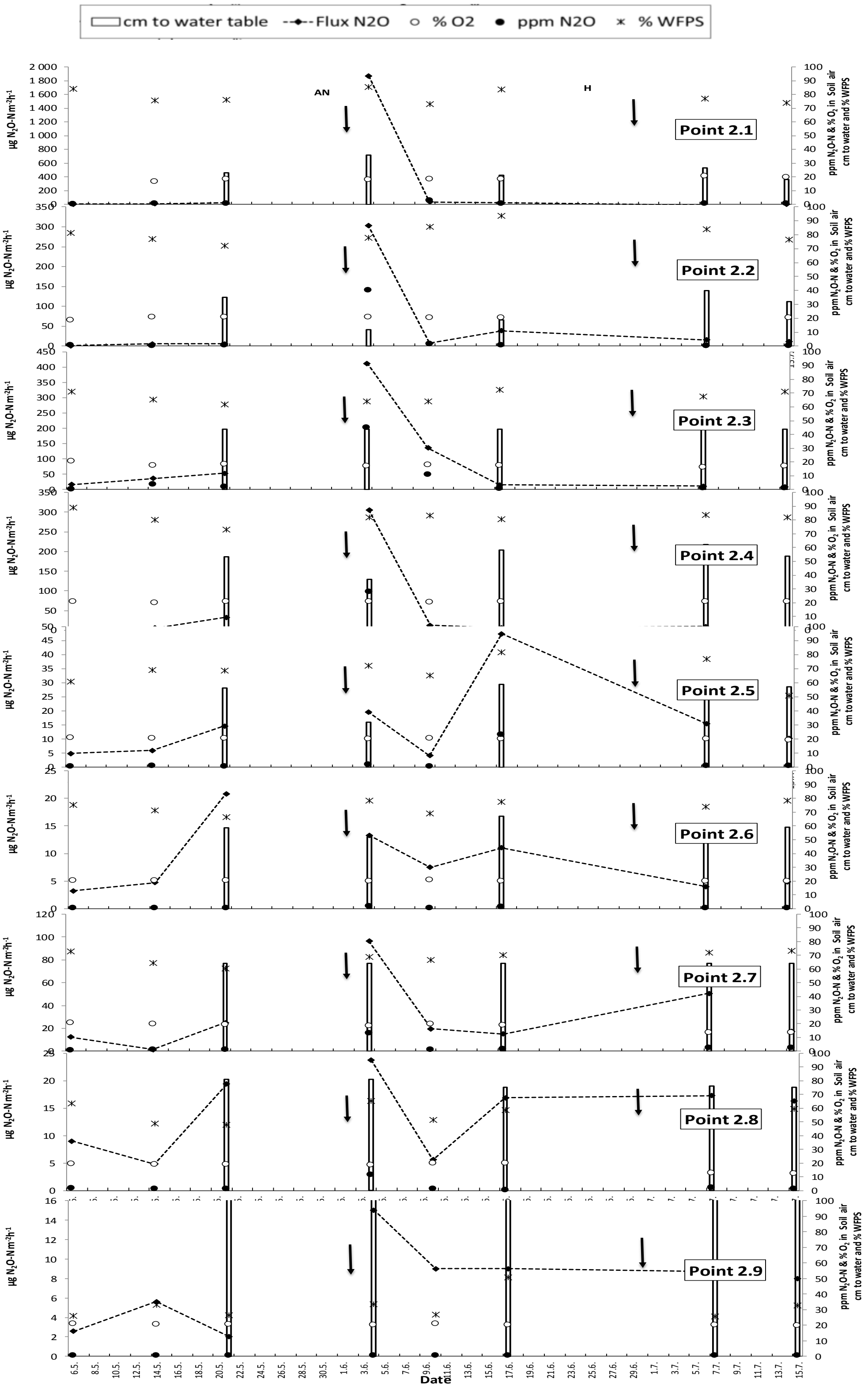

Figure 5.4 N_2O fluxes, N_2O and O_2 concentrations in soil air (7-11 cm depth), distance to water table or dense soil layer and water filled pore space (% WFPS) in grassland on field 1 measurement points 1.1 -1.9 a) from 6th May until 15th July 2010. An arrow with AN indicate when 70 kg ha^{-1} NH_4NO_3-N in combined NPK fertilizer is applied, and an arrow with H when grassland is harvested Source: adapted from Hansen et al. personal communication.

5.1.2 Other countries experience

France: A study was conducted in the Orgeval catchment in France to analyze the influence of landscape position on N_2O emissions. In general, N_2O emissions were highest (4 kg N ha^{-1} yr^{-1}) in the foot slope position (where mean WFPS was 68 percent) when compared to backslope and shoulder positions (Vilain *et al.,* 2010).

Germany: A study was made to determine the effect of soil moisture on emission of N_2O after addition of nitrate fertilizer in Scheyern of southern Germany. Results showed that N_2O emission rates were generally small at soil moisture levels ≤60 percent WFPS with mean flux rates ranging between 1 and 12 µg N_2O-N m^{-2} hr^{-1}. Significant increment of N_2O emission rates were measured at soil moisture contents ≥70 percent WFPS, with the highest N_2O fluxes occurring at the highest soil moisture level (Ruser *et al,* 2006).

United Kingdom: Topsoil samples were taken from an arable field on Imperial College London to determine the contributions of nitrification and denitrification to N_2O emissions from different soil WFPS. Nitrification was the main source of N_2O in soils at 35-60 percent WFPS. The threshold at which soils become too anaerobic for either nitrifier denitrification or aerobic nitrification was above 60 percent WFPS and 70 percent WFPS where nitrification did not contribute to N_2O production. As soil WFPS increases, diffusion of oxygen into soil aggregates will decrease, and a rapidly increasing fraction of the soil volume will become anaerobic, causing increased N_2O production by denitrification to maximum 90 percent WFPS but above which most of the N_2O is reduced to N_2 (Bateman and Baggs, 2005).

USA: Terry *et al.,* (1981) measured field N_2O fluxes and N_2O produced in soil samples incubated under laboratory conditions in Florida (USA) to determine the effect of flooding on N_2O emissions from a drained cultivated Histosols. The N_2O flux decreased from 174 g N ha^{-1} d^{-1} prior to flooding to approximately zero after flooding. A possible explanation could be that the flooded field may have acted as a sink for atmospheric N_2O. However, it should be recognized that a flooded soil has a unique environment that favours reduction of N_2O to N_2 over escape of N_2O to the atmosphere. Denitrification under non-flooded conditions may well lead to substantial emissions of N_2O, because the soil microflora has less ability to further reduce N_2O to N_2. Thus, diffusion of N_2O from the soil is more rapid. Nitrous oxide fluxes from flood-irrigated dairy pasture rose rapidly two to three days after irrigation, when the soil WFPS was >95 per cent. The emissions remained high for further one or two days before gradually subsiding to background levels as the soil moisture decreased. However, the N_2O emission remained low immediately following irrigation, which was most likely the result of complete denitrification, producing mainly N_2 emissions (Rochette and Bertland, 2007).

5.2 Soil drying-rewetting

Soil drying and subsequent rewetting induces N mineralization and denitrification, but the effects of the “extent” or “degree” of drying and rewetting on N_2O emissions remains poorly understood (Guo *et al.,* 2014). Plant water uptake followed by rapid rewetting during precipitation generally creates drying–wetting cycles in the root zone. Increased soil denitrification rates and N_2O emissions following wetting of dry soil by rainfall or irrigation have been reported for various agricultural systems (Nobre *et al.,* 2001) including grazed pastures (Kim *et al.,* 2010). Increased soil N_2O emissions resulting from denitrification following thawing of frozen soils have also been observed in various agricultural systems including grasslands (Virkajärvi *et al.,* 2010). The mechanisms considered responsible for increased denitrification and N_2O emission following rewetting and thawing are the followings:

- Increased availability and accessibility of the substrate (Yergeau and Kowalchuk, 2008);
- Creation of anaerobic conditions (de Bruijn *et al.,* 2009); and
- Release of trapped N_2O and N_2 (Virkajärvi *et al.,* 2010).

5.2.1 Other countries experience

Costa-Rica: In tropical soils of Costa Rica, N_2O flux pulses began within 30 min, peaking no later than 8 h after rewetting and 25 g N_2O–N ha^{-1} was emitted for three simulated rain events over a 22-day period (control emitted 14 g N_2O–N ha^{-1}) and one episodic N_2O production event driven by one moderate rain accounted for less than 15 percent to more than 90 percent of the total weekly production (Nobre *et al.,* 2001).

Canada: Surface soil was collected in December 2011 after fall harvest from the corn phase of a corn-soybean-winter wheat rotation at the Hon. Eugene F. Whelan Experimental farm, Agriculture and Agri-Food Canada to investigate the impacts of different degrees of soil drying (drying to 45, 30, 20, or 10 percent water-filled pore space, WFPS) and subsequent rewetting (rewetting to 75 or 90 percent WFPS) on N_2O emissions, denitrification, and net N mineralization. The highest N_2O emissions (201 µg N_2O-N kg^{-1}) occurred when the soils were dried to 10 percent WFPS followed by rewetting to 90 percent WFPS, whereas the lowest emissions (4.72 µg N_2O-N kg^{-1}) occurred when the soil was dried to 45 percent WFPS followed by rewetting to 75% WFPS. When soil was rewetted from 10 to 90 percent WFPS, cumulative N_2O emissions over 120 h were 7.4 times higher than when the soil was rewetted from 10 to 75 percent WFPS (Figure 5.5). The proportion of N_2O evolved [$N_2O/(N_2O+N_2)$] generally increased as the soil dried. Soil rewetting to 75% WFPS generally produced higher $N_2O/(N_2O+N_2)$ ratios than rewetting to 90 percent WFPS. Net N mineralization rates in soils rewetted to 75 percent WFPS significantly increased from 0.78 mg N kg^{-1} d^{-1} for the soils dried to 45 percent WFPS to 1.69 mg N kg^{-1} d^{-1} for the soils dried to 10 percent WFPS. More extensive soil drying and more extensive rewetting stimulated N_2O emissions and total denitrification losses (Guo *et al.,* 2014).

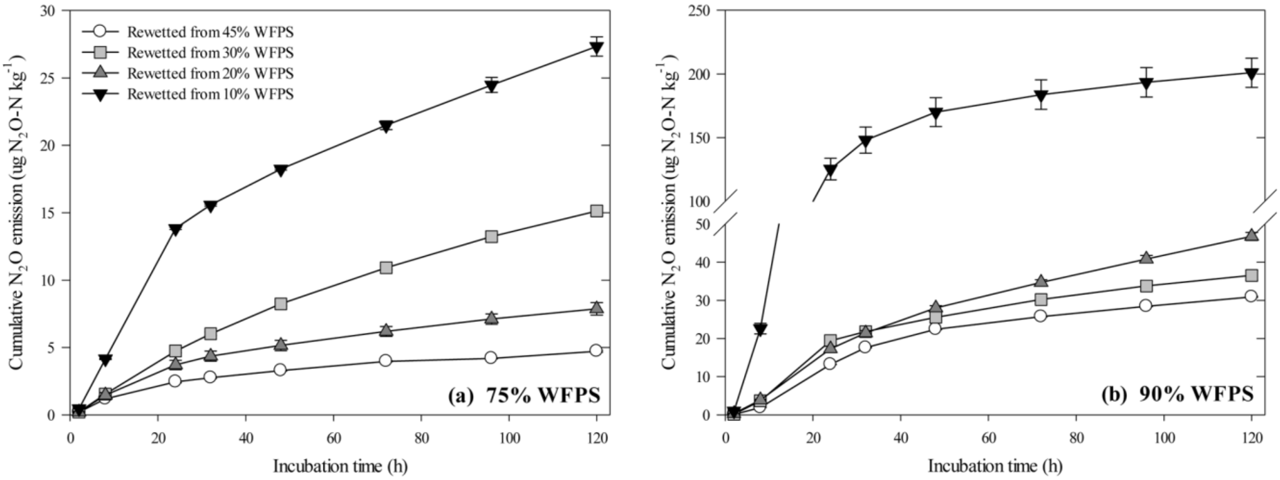

Figure 5.5 Cumulative N2O emission during the 120-h incubation at 75% WFPS (a) and 90% WFPS (b) after rewetting from 45, 30, 20, and 10 percent WFPS. Bars are standard error (n = 4), which are not always visible because the standard error is smaller than the symbol. Source: adapted from Guo *et al.* (2014).

5.3 Summary

It is very difficult to draw a conclusion on the effects of moisture and drainage conditions on N_2O emissions from cultivated organic or mineral soils in Norway. The very few studies carried out so far have limited measurements in time and space. Generally speaking, poorly drained mineral soils emit more N_2O-N compared to moderately drained soils mainly due to higher soil moisture and relatively shallow ground water table. Factors such as the type and amount of fertilizer also influence the N_2O emissions along with the soil moisture. The practice of drainage in peat soils in northern Norway has resulted in release of GHGs to the atmosphere notably CO_2 and N_2O and also leaching of nutrients such as $NO3^-$. As a result, the nitrate concentrations increased in the soil and ground water during the dry period which could be a source for N_2O production.

6 EFFECTS OF SOIL pH ON N_2O EMISSIONS

6.1 Introduction

Soil pH is another key regulator of the microbiological processes that affect N_2O and N_2 production. Soil pH has a potential effect on N_2O production pathways. It is generally agreed that denitrification is slower in acid conditions (Fageria and Baligar, 2008), but denitrification can still occur at pH values as low as 3.5 and can account for significant N losses in naturally acid soils (Weier and Gilliam, 1986). Šimek et al. (2000) found no significant relationship between soil pH and denitrifying enzyme activity, although denitrification potential was significantly correlated with soil pH. This and other work led Šimek and Cooper (2002) to conclude that the expression "optimum pH for denitrification has little or no meaning" unless it is qualified by specifying the particular aspect of denitrification that is being considered. On the other hand, Liu, et al (2010) have reported that the product ratio of $N_2O/\ (N_2O + N_2)$, showed a consistent decline with increasing pH. Similar results were found by Raut et al. (2012) in which $N_2O/(N_2O+N_2)$ ratio increased with decreasing pH. However, the mechanism of pH control on the N_2O: N_2 ratio of denitrification is still not fully understood (Liu et al. 2010; Saggar et al. 2013).

6.1.1 Norway

Mørkved *et al.* (2007) examined long-term effects of liming on the N_2O product ratio of nitrification by aerobic incubation experiments. The objective was to investigate the mechanisms involved in pH control of denitrification by quantifying pools of functional denitrification genes, their transcription and the kinetics of NO, N_2O and N_2 production in soils. Soils were sampled from the long-term liming field experiments located in western Norway and south-eastern Norway at Ås where different amounts of lime had been applied.

Table 6.1: Rates of N_2O production, gross nitrification, gross denitrification, CO_2 production, calculated contribution of nitrification to N_2O production (%) and N_2O product ratio (PR) of nitrification $N_2O/(NO_3{}^- + NO_2{}^-)$ during 67 h incubation in shaken soil slurries and headspace O_2 concentration at the end of incubation. All rates are averages of three replicates (±SD) except denitrification rates which were measured in a single bottle (N_2O production with 10 percent C_2H_2). Nitrifcation rates were calculated by the pool dilution method.

Soil	pH-H_2O	ng N g^{-1} dry wet soil d^{-1}		µg N or Cg^{-1} dw soil d^{-1}		Nitrification contrib$^{\underline{n}}$ to	Nitrification N_2O PR[a] (0/oo)	Headspace O_2[b]
		N_2O prod$^{\underline{n}}$	Denitrification	Nitrifcation	CO_2 prod$^{\underline{n}}$	N_2O (%)	$N_2O/(NO_3^- + NO_2^-)$	%
F2	4.2	200 ± 34	695	9.8 ± 1.2	14.2 ± 0.3	68 ± 1.07	14.1 ± 1.49	20.5 ± 0.01
F3	4.1	1100 ± 37	733	9.6 ± 2.3	10.6 ± 0.2	62.9 ± 0.59	76.2 ± 21.1	20.6 ± 0.01
F4	6.7	31 ± 1.4	8.1	170 ± 8.5	27.7 ± 0.4	95.7 ± 0.08	0.18 ± 0.01	20.0 ± 0.01
F5	6.2	25 ± 0.4	7.6	80 ± 10.4	27.2 ± 1.6	89.8 ± 0.05	0.28 ± 0.04	20.0 ± 0.06
F9	7.8	15 ± 1.1	4.6	61 ± 6.9	26.5 ± 1.0	94.0 ± 1.53	0.24 ± 0.05	19.9 ± 0.03
Å2	5.0	2.3 ± 0.2	0.90	6.3 ±0.5	19.2 ± 0.5	90.8 ± 0.60	0.33a ± 0.02	20.3 ± 0.02
Å5	5.5	2.4 ± 0.4	0.35	5.6 ± 0.4	19.9 ± 1.1	91.4 ±1.95	0.39ab ± 0.04	20.2 ± 0.04
Å7	6.1	2.9 ± 0.3	0.28	5.6 ± 0.6	19.5 ± 2.1	92.5 ± 0.20	0.49b ± 0.07	20.3 ± 0.06

[a]: Treatments not significantly different (ANOVA, P= 0.05) share the same letter . No ANOVA was done for F due to large variation in variance.

[b]: Calculated from measured CO_2 production assuming a 1:1 stoichiometry between CO_2 production and O_2 consumption.

F: peat soils from a meadow in Fureneset in Fjaler in western Norway and Å stagnic albeluvisols from Ås in SE Norway.

Source: adapted from Mørkved *et al* (2007)

For soils with pH ≥5, the N_2O product ratios for nitrification were low (0.2–0.5 percent) and comparable to values found in pure cultures of ammonia oxidizing bacteria. In mineral soils, only a minor increase in the N_2O product ratio with increasing soil pH was found, but the effect was so weak that it justifies a constant N_2O product ratio of nitrification for N_2O emission models. Whereas for the soils with pH 4.1 and 4.2, the apparent N_2O product ratio of nitrification was 76 percent and 14 percent, respectively which is about 1-1.5 orders of magnitude higher than above pH 5 (Table 6.1). This could partly be accounted by the rates of chemodenitrification of NO_2^- (Mørkved, et al., 2007). The study underlines the role of NO_2^-, both for regulating denitrification and for the apparent nitrifier derived N_2O emission.

A laboratory study was carried out by Hovlandsdal (2011) to test the enzyme that reduces N_2O to N_2 during denitrification (N_2O)-reductase, whether it has an effect on in situ N_2O emissions or not. Soil samples were collected from long-term experimental field in Fureneset in outer Sunnfjord, Norway. The study showed that liming has a great potential to reduce fertilizer induced N_2O -emissions from cultivated organic soils.

6.1.2 Other countries experience

Czech Republic: A field experiment with manipulation of the soil pH was carried out in a grassland area at South Bohemia in Czech Republic to investigate how changes in soil pH affect the N_2O and N_2 emissions, denitrification activity, and size of a denitrifier community by Ĉuhel *et al.* (2010). They found that manipulation of soil pH affected the $N_2O/(N_2O + N_2)$ ratio, which increased with decreasing pH due to changes in the total denitrification activity but not in N_2O production. Soil pH is of importance in determining the nature of denitrification end products. Thus, the $N_2O/(N_2O + N_2)$ ratio increased with decreasing pH due to changes in the total denitrification activity (Figure 6.1), while no changes in N_2O production were observed (Ĉuhel *et al.,* 2010).

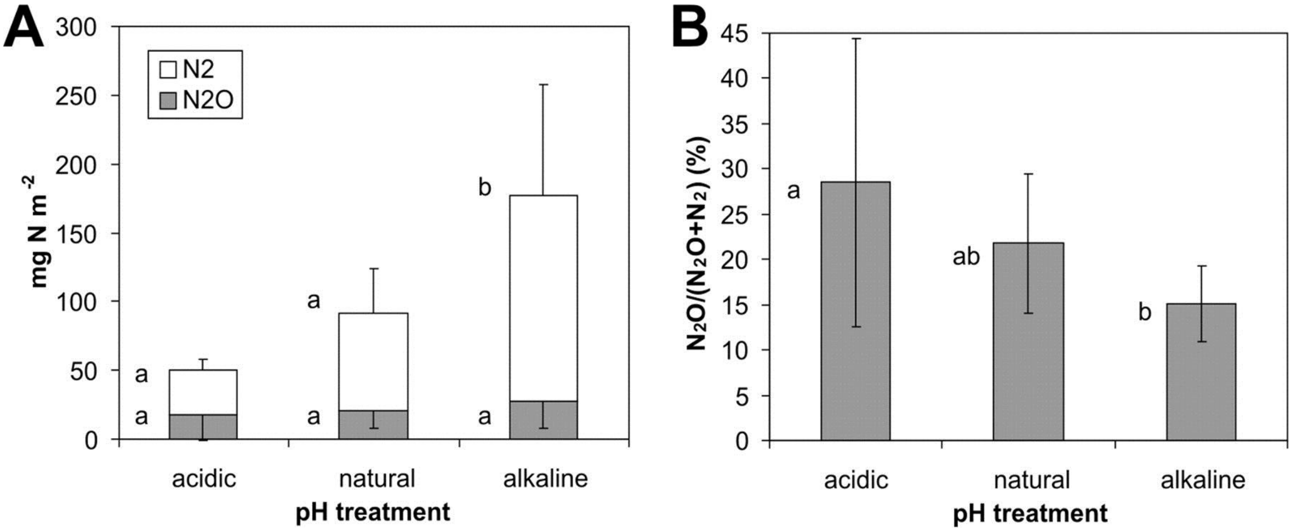

Figure 6.1 In situ cumulative losses of N (separately as N_2O and N_2) (A) and relative N_2O production expressed as the $N_2O/(N_2O + N_2)$ molar ratio (B) with different pH treatments over the 74 h after the addition of 15N-labeled KNO_3. Mean values and ± standard deviations are shown (n = 12). The different letters next to the bars indicate significance differences between the specific pH treatments ($p < 0.05$). Source: adapted from Ĉuhel *et al.* (2010).

Northern Ireland: A laboratory incubations of soil samples were collected from a long term permanent pasture site at County Antrim in Northern Ireland to investigate the effect of soil pH on the processes that are responsible for the production of N_2O and N_2. Stevens *et al.* (1998) reported that the flux of N_2 increased with pH but the effect of pH on the flux of N_2O was inconsistent. The maximum flux of N_2O occurred at pH 6.5 and the minimum fluxes at pH 6 and 8. The N_2O-reductase enzyme was still being inhibited or N_2O was being produced by a process other than denitrification. Partial inhibition of N_2O-reductase at pH 8 may be due to NO_2^- accumulation.

6.2 Summary

It is not possible to generalize the effect of soil pH on N_2O emission from Norwegian agricultural soils. There are very few studies carried out on the direct effects of soil pH on nitrous oxide production from Norwegian agricultural soils. Most of the studies were analysed in soils incubated under laboratory condition to examine the N_2O product ratio of nitrification and denitrifiction by taking soil samples from long term liming field trials. According to these trials result, the product ratio of dentrification $N_2O/(N_2O+N_2)$ showed a consistent decline with increasing pH and vice versa which is in line with other countries result (Šimek and Cooper, 2002; Ĉuhel et al. 2010). While the product ratio of nitrification $N_2O/(NO_2^-+ NO_3^-)$ were low for soil pH between 5.0 to 7.8 but high for soils with pH 4.1 and 4.2 partly due to chemodenitrification rates of NO_2^-. However, the mechanism of pH control on the product ratio of nitrification and denitrification is still not fully understood. Further field studies are required to determine whether soil pH is the best indicator for N_2O production.

7 FREEZING-THAWING EFFECTS ON N_2O EMISSIONS

7.1 Introduction

Temperature is one of the main factors causing temporal fluctuations in denitrification (Ryden, 1983). In general, microbial activity, denitrification and nitrification rates all increase with temperature. The denitrification product ratio (N_2O/N_2) decreases with increasing temperature, while that from nitrification (N_2O/NO_3^-) tends to rise. The combined effect is that N_2O emission rates increase with temperature (Granli and Bøckman, 1994). In temperate regions, a majority of N_2O is emitted during spring soil thawing (Dietzel *et al.,* 2011). Freeze-thaw cycles during the spring are responsible for 50 percent (Rover *et al.,* 1998) to 66 percent (Duxbury *et al.,* 1982; Johnson *et al.,* 2010) of annual N_2O emissions from agricultural soils. However, the mechanisms that control N_2O fluxes following rewetting and thawing events and their impact on annual budgets are not fully understood (Kim *et al.,* 2012).

7.1.1 Norway

A short-term laboratory incubations was conducted by Mørkved *et al.* (2006) under standardized moisture and oxygen (O_2) conditions, using nitrogen (N) tracers (^{15}N) to determine process rates and sources of emitted N_2O after freeze–thaw treatment of soil or after addition of freeze–thaw extract from clover. The N_2O emission response was inversely related to O_2 concentration, indicating denitrification as the quantitatively prevailing process. Denitrification product ratios in the two studied soils (pH 4.5 and 7.0) remained largely unaltered by freeze–thaw or freeze–thaw-released plant material, refuting the hypothesis that high winter emissions are due to frost damage of N_2O reductase activity (Mørkved *et al.,* 2006).

Table 7.1 Average N_2O production rates (±SD; n = 3) with and without acetylene (C_2H_2), and product ratios of denitrification ($N_2O/(N_2+N_2O)$) in response to freeze–thaw and plant extract addition.

		Temp	No C_2H_2	With C_2H_2		
Soil	Treatment	°C	ng N_2O-N g^{-1} d^{-1}	ng N_2O-N g^{-1} d^{-1}	$N_2O/(N_2 + N_2O)$	R^2
Soil 1 (pH 4.5)	Water	5	1502 ± 166	2040 ± 152	0.74 ± 0.10	0.89–0.91
	Water, freeze–thaw	5	4368 ± 226	6379± 526	0.69 ± 0.07	1.00
	Plant extract	5	8301 ± 525	9607 ±172	0.86 ± 0.06	0.99–1.00
	Plant extract, freeze–thaw	5	6770 ± 244	7831 ± 514	0.87 ± 0.07	1.00
	Water	20	7157 ± 136	7212 ± 278	0.99 ± 0.04	0.98–1.00
Soil 2 (pH 7.0)	Water	5	1021 ±160	1938 ±212	0.53 ± 0.10	0.99–1.00
	Water, freeze–thaw	5	2802 ± 55	7365 ± 93	0.38 ± 0.01	0.99–1.00
	Plant extract	5	5621 ± 83	10224 ± 90	0.55 ± 0.01	0.99–1.00
	Plant extract, freeze–thaw	5	4131 ± 6	8274 ± 304	0.50 ± 0.02	0.99–1.00

All ratios are based on linear regression of N_2O accumulation versus time under anaerobic conditions. The minimum and maximum R^2 of the linear regression with time for the six N_2O production rate estimates (± C_2H_2) within each treatment are given. Source: Mørkved *et al.* (2006).

Nitrification rates estimated by nitrate (NO_3^-) pool enrichment were 1.5 to 1.8 mg NO_3^- -N per g dw (dry weight) soil per day in freeze–thaw-treated soil when incubated at O_2 concentrations above 2.3 vol percent and one order of magnitude lower at 0.8 vol percent O_2. Thus, the experiments captured a situation with severely O_2-limited nitrification. As expected, the O_2 stress at 0.8 vol percent resulted in a high nitrification product ratio (0.3 g per g). Despite this high product ratio, only 4.4 percent of the measured N_2O accumulation originated from nitrification, reaffirming that denitrification was the main N_2O source at the various tested O_2 concentrations in freeze–thaw-affected soils (Mørkved *et al.,* 2006).

N_2O emission response to both freeze–thaw and plant extract addition appeared strongly linked to stimulation of carbon (C) respiration, suggesting that freeze–thaw-induced release of decomposable organic C was the major driving force for N_2O emissions in our soils, both by fuelling denitrifiers and by depleting O_2. The soluble C (applied as plant extract) necessary to induce a CO_2 and N_2O production rate comparable with that of freeze–thaw was 20–30 µg C per g soil dry weight. This is in the range of estimates for over-winter soluble C loss from catch crops and green manure plots reported in the literature. Thus, freeze–thaw-released organic C from plants may play a significant role in freeze–thaw-related N_2O emissions (Mørkved *et al.,* 2006).

7.1.2 Other countries experience

The following tables provide a synopsis of research results and conclusions drawn on the effects of freezing and thawing on N_2O emissions from other countries that have similar climatic conditions with Norway. The main objectives of the research along with references are also mentioned. The most recently published literatures that are relevant to the Norwegian conditions are selected and presented in the Tables 7.2 to 7.5.

i) Canada

Table 7.2 presents 2 field and 2 laboratory studies made on freezing and thawing effects on N_2O emissions in Canada.

Table 7.2 Publications that provide research results on the effect of freezing and thawing on N_2O emissions in Canada.

Research objectives	Results and conclusions	Reference
• *To examine the effects of temperatures (near freezing point) on N_2O emissions, denitrification, and on the abundance and structure of soil nitrifiers and denitrifiers communities.*	• Assessing the effects of freeze–thaw cycles of different amplitudes and frequency would help to understand the impacts of climate warming over winter on nitrifier and denitrifier communities with freezing and a surprising stimulation of N_2O emissions at –1 °C when NO_3^- and C are present.	Wertz *et al.* (2012)
• *To identify the source of N_2O and emission during freezing and thawing cycles.*	• Soil atmospheric N_2O concentrations at 7.5 to 45 cm soil depths, increased during the winter of 1995-1996 when air temperatures increased above freezing. • N_2O flux from soil to atmosphere during the thawing events was highly correlated with N_2O concentration in the soil atmosphere near the surface. • Biological processes contribute to significant amount of N_2O emissions during thaw events, even when temperatures are at or below freezing point.	Chang and Hao (2001)
• *To quantify N_2O emissions from agricultural fields during winter and spring thaw*	• N_2O emissions from January to April over 4 years ranged between 0 and 4.8 kg N ha^{-1}. These thaw emissions are substantial and should be considered in the N_2O budgets in regions where thaw periods occur. • Fallowing, manure application and alfalfa incorporation in the autumn lead to high spring emissions, while the presence of plants (as in the case of alfalfa or grass) can result in negligible emissions during thaw. This presents an opportunity for mitigation of N_2O emissions through use of over-wintering cover crops.	Wagner-Riddle and Thurtell (1998)
• *To examine N_2O emission from agricultural soil subjected to different freeze-thaw cycles*	• The mean N_2O fluxes ranged from 0.24 to 0.65 kg N ha^{-1} day^{-1} for non-frozen soil cores. One explanation for increased N_2O emission by freeze-thaw is that freeze-thaw increases the availability of soil organic carbon for microbial utilization. Another explanation is that freeze-thaw enhances nitrification and hence produces more NO_3^- for denitrification.	Chen *et al.* (1995)

ii) Finland

The results of four selected studies (i.e. 3 laboratory and 1 field) on freezing and thawing effects on N_2O emissions in Finland are shown in Table 7.3.

Table 7.3 Research results on the effect of freezing and thawing on N_2O emissions in Finland.

Research objectives	Results and conclusions	Reference
• *To examine the effect of snow cover and no snow cover on N_2O production and emissions.*	• Removal of snow lowered soil temperatures and enhanced N_2O emissions during freezing and thawing events in sands and mull soils. Cumulative emissions with snow removed (without snow cover) over the winter were 0,25 (0,37) in sandy soils, 0,66 (1,3) in mull soils and 3,0 (3,3) N_2O-N m^2 y^{-1} in peat soils. • Laboratory incubations of the soil samples showed high production rates of N_2O at temperatures <0°C, especially in sandy and peat soils.	Maljanen *et al.* (2009)
• *To explore the temperature responses of N_2O emissions using laboratory microcosm during soil freezing and thawing periods*	• At –2.2°C, abandoned soil showed an increase in N_2O emissions (58.7 ± 14.5 µg N_2O-N m^{-2} h^{-1}). While when the temperature was further decreased to –4.9°C, N_2O emissions decreased in abandoned soil as well as in the afforested soils. N_2O emissions increased immediately after the soil thawing started in both sites. The emissions from the abandoned soil were highest (238 ± 47.2 µg N_2O-N m^{-2} h^{-1}) 2 days after the thawing started. The maximum emissions from the afforested soil in mid-winter was 128.7 ± 17.9 µg N_2O-N m^{-2} h^{-1} after 5 days of thawing. • Soil freeze-thaw cycles enhance the N_2O emissions in abandoned soil as well as in the afforested soils in contrast to the NO emission.	Koponen *et al.* (2006)
• *To examine the N_2O production at low temperatures with and without freezing-thawing events*	• When the temperature of the frozen soil cores was increased stepwise from –8°C, the N_2O emissions began to increase at –0.5 °C, and peaked at –0.1 °C in the organic, clay and silty soils, and at +1.6 °C in loam soils. However, a stepwise decrease in soil temperature from +15°C also induced an increase in the N_2O emissions close to the 0°C. These emissions peaked between –0.4 and +2.5°C depending on the soil type and water content. • Soils can have a maximum of N_2O emission near 0°C when soil temperature decrease. These emissions, however, are less than those emitted from thawing soils. The results suggest that N_2O is produced in soils down to a temperature of –6 °C.	Koponen *et al.* (2004)
• *To investigate the effects of freezing-thawing and drying-wetting on the emission of N_2O*	• The total surplus N_2O emissions due to the first wetting event ranged between 3 and 140 mg N-N_2O m^{-2} due to the first thawing event but later declined following two successive freeze-thawing events. • Wetting and thawing produced a greater surplus emission of N_2O from grassland sites compared to arable sites. • Wetting or thawing of soil cores from the Finnish sites did not result in any significant increase in N_2O emission rates perhaps because of a prolonged drought during the time of soil core collection that had changed soil properties considerably.	Priemé and Christensen (2001)

iii) Germany

Table 7.4 shows three selected studies (i.e. 2 laboratory and 1 under field conditions) on freezing and thawing effects on N_2O emissions in Germany.

Table 7.4 Effects of freezing and thawing on N_2O emissions in Germany: main results and conclusions.

Research objectives	Results and conclusions	Reference
• *To examine the processes of N_2O emissions during freezing/thawing periods under laboratory condition.*	• Two periods of higher N_2O emissions were detected, a period of elevated N_2O emissions during continuous soil freezing and a period of brief peak emissions during thawing. • N_2O was produced by microorganisms during continuous soil freezing in an unfrozen water film on the soil matrix which was covered by a layer of frozen water. The frozen water in form of an ice layer represents a diffusion barrier which reduces oxygen supply to the microorganisms and partly prevents the release of N_2O. • Peak emissions during soil thawing were explained by the physical release of trapped N_2O and/or denitrification during thawing.	Teepe *et al.* (2001)
• *To investigate the effects of freezing-thawing and drying-wetting on the emission of N_2O*	• Following thawing, denitrification was responsible for <40 percent of N_2O emission from the German grassland soils.	Priemé and Christensen (2001)
• *To examine the N_2O emissions from soils during thawing and freezing cycles in winter*	• Small N_2O emission rates were measured, when soil freezing started in December. However, N_2O emissions increased with soil warming (up to 0 °C) in January. In mid January the N_2O emissions decreased due to soil freezing (<-2 °C). High N_2O emissions from soil were also found during a second thawing period in February. • N_2O is produced and emitted even in frozen soil (soil temperature -4 °C). Therefore winter emissions are likely important in global N_2O emission budgets.	Röver *et al.* (1998)

iv) Japan

Yanai *et al.* (2011) studied in situ soil gas N_2O and oxygen (O_2) concentrations at two experimental sites in northern Japan over the period of a year from November 2008 to October 2009. The objective was to clarify the factors stimulating N_2O production in soils at low temperatures. The experimental sites were N-fertilized bare arable lands with different soil frost depths and snowmelt rates according to the snow cover management imposed.

Results shows that winter-to-spring net N_2O fluxes, ranging from 0.10 to 1.95 kg N_2O-N ha^{-1} were positively correlated with the annual maximum soil frost depth (ranging from 3 cm to 41 cm; $r = 0.951^{***}$). In the plots with deeper maximum soil frost, winter-to-spring N_2O fluxes represented 58 to 85 percent of the annual values. Soil N_2O production was stimulated when the soil frost depth was greater than 15 cm or the daily mean soil temperature at 5 cm depth was below –2.0 °C. In the soil with the greatest frost depth, soil gas N_2O concentrations at the depth of 10 cm peaked at 46 ppm when soil gas O_2 concentrations fell down to 0.12 m^3 m^{-3} under soil temperature below 0.0 °C. Snowmelt acceleration had no stimulating effect on N_2O production in the soil during the winter-to spring period (Yanai *et al.,* 2011).

Yanai *et al.* (2011) concluded that soil freezing, rather than the melting of snow cover, primarily enhanced soil N_2O production in a one-year plot-scale (snow cover manipulation) experiment conducted in the northern Japan. This is because a decreasing soil gas O_2 concentration in the shallow layer clearly corresponded with an increasing soil gas N_2O concentration, denitrification in the thawing upper layer, which was likely the major process responsible for N_2O production. They asserted that in regions where the maximum soil frost depth exceeds 15 cm or the daily mean soil temperature at a 5 cm depth falls below –2.0 °C, N_2O fluxes might not be negligible, as once thought. To evaluate the annual N_2O fluxes, field monitoring of N_2O emission rates immediately prior to the end of snowmelt might be essential for regions where soils undergo seasonal freezing (Yanai *et al.,* 2011).

v) USA

Two field and two laboratory studies on freezing and thawing effects of N_2O emissions in USA are shown in Table 7.5.

Table 7.5 Research results on the effect of freezing and thawing on N_2O emissions in USA.

Research objectives	Results and conclusions	Reference
• *To test several soil conditions affecting the emission of N_2O during thawing of soil*	• Nitrate addition to surface soil (0-10 cm) enhanced thaw emission. Higher thaw emission for surface soil was related to greater organic matter and microbial biomass C contents and denitrifying enzyme activity than deeper soil. • Increasing the bulk density of soil from 1.1 to 1.25 Mg m^{-3} decreased thaw emission. • Using the acetylene inhibition method, the N_2O:N_2 ratio was higher for frozen (0.17) than cold (0.07) treated soil, respectively, without the addition of NO_3^-. • The addition of NO_3^- increased the N_2O:N_2 ratio being 2.45 and 0.53 for frozen and cold-treated soil.	Tenuta and Sparling (2011)
• *To investigate the effect of the temperature and moisture on the emission of N_2O from arable soils.*	• In the soils with elevated water content (90 percent of the total water capacity) at 25 °C, the loss of fertilizer N in the form of N_2O reached 2.35 percent because of the active denitrification. • The extra N_2O flux initiated by the freeze–thaw processes made up 88 to 98 percent of the total N_2O flux during the entire experiment.	Kurganova and de Gerenyu (2010)
• *To understand the effects of tillage, moisture content and manure application on N_2O emissions from agricultural soils at low temperatures.*	• The maximal emission of 200 µg N m^{-2} h^{-1} was found at soil temperatures >5 °C and at WFPS between 40 to 70 percent. However, emissions dropped dramatically with further increases in soil moisture, to 50 µg N m^{-2} h^{-1} in the most saturated areas (90 percent saturated). • The emission of N_2O from manure-amended soils was not limited to thawing events: emissions began at soil temperatures below 0°C and continued even after complete soil freezing.	Singurindy *et al.* (2009)
• *To quantify nitrous oxide fluxes during soil thaw in early spring.*	• At thawing, high N_2O concentrations (ranging from 1082 to 2066 mg m^{-3}) were found at 10 to 30 cm in the soil profiles of a coniferous forest, and in manure and straw-treated plots. When thawing was complete, soil profile N_2O concentrations and N_2O flux declined. N_2O was released on warming, and cores treated with 2 mL of chloroform ($CHCI_3$), had a slower release rate.	Goodroad and Keeney (1984)

7.2 Summary

There is very limited knowledge on the processes involved in N_2O emissions during and following freezing and thawing of soils in Norwegian agriculture. Hence, it is not possible to draw a conclusion on the state of freezing and thawing effects on N_2O emission. However, the N_2O emission from cultivated soils during the winter and early spring in Norway (when freezing and thawing cycles are common), could be substantial as it has been shown in other cold temperate regions such as in North America and northern Europe. Therefore, N_2O emission induced by freezing and thawing of soils during winter and early spring should be quantified and accounted in the national N_2O budgets and annual GHGs emissions.

8 OPTIONS FOR MITIGATING N_2O EMISSIONS

From an agricultural perspective, denitrification could result in the loss of a valuable plant nutrient notably N and should therefore be minimised. From an environmental perspective, denitrification presents both a threat and a benefit. When denitrification converts NO_3^- to N_2 which otherwise would have been leached from soils and discharges into rivers or lakes, may be an environmental benefit because the evolved N_2 can be trapped later by legumes fixing bacteria and enrich the soil with nitrogen. If, however, denitrification results in emitting N_2O, this poses an environmental threat since it increases global warming and induce climate change. Management of agricultural practices should therefore minimise the accumulation of NO_3^- in soil and thereby restrict both leaching and N_2O emission. However, where denitrification is inevitable, or even desirable to protect water quality, the management should ensure that as much of the denitrified NO_3^- is emitted as N_2 not as N_2O. There are a range of on-farm mitigation options to control denitrification by either emitting less N_2O or shifting the balance between harmful N_2O and the non-greenhouse gas namely N_2 from agricultural soils. Some of the mitigation practices of N_2O emissions from agricultural soils that are relevant to the Norwegian conditions, are described below.

8.1 Match N supply with crop demand

The foundation of good fertiliser stewardships rest on the principles of using the 4R's i.e. right source, at the right rate, at the right time, and with the right placement (IPNI, 2012). The crop requirement for N fertilizer in the form of organic or inorganic N could be determined through undertaking need-based fertilizer application. Need-based fertilizer application is a system where the amount and type of fertilizers applications to the soils and/or plants is based on soil testing (Marx *et al.*, 1999), using leaf colour chart: LCC (Varinderpal-Singh *et al.*, 2012) or chlorophyll meter (Ali *et al.*, 2015) or using generic yield/nutrient uptake and nutrient applied/nutrient uptake ratio (van Duivenbooden *et al.,* 1996). Need-based fertilizer application is getting more and more attention in several countries (for e.g. in India, Ali *et al.,* 2015) as the single blueprint fertilizer doses are no more effective (van Beek *et al.*, 2014). The amount of N-fertilizers required for crops depends on the soil type, weather conditions and the land use. By matching the N supply derived from the soil reserves (through soil testing) with the crop demand for N, an appropriate amount of N-fertiliser could be determined and applied to the soils without mining the soil nutrients and exposing the soils for emissions of N_2O. In other words, by undertaking soil N balance in the given field, the emission of soil N_2O can be controlled at the same time the soil and crop can be nurtured with recommended amount and type of N fertiliser which then contributes to sustain soil fertility and crop production (Varinderpal-Singh *et al.,* 2010).

In line with the above mitigation practice under the Norwegian condition, N_2O emissions could be reduced by 30% when lowering N fertilization from 120 to 90 kg N ha^{-1} yr^{-1} without significantly reducing dry matter grain yield in wheat production areas of SE Norway (Nadeem et al. 2014).

8.2 N fertilizer Best Management Practices

Some of the Best Management Practices (BMP) that can contribute to reduce N_2O emissions from agricultural soils are listed in Table 8.1. In cold temperate climates to prevent or reduce losses of mineral fertilizer nitrogen, it is better to avoid the application of organic and nitrogen containing mineral fertilizers in the autumn and early spring, when soil chills are possible. Another method of limiting such losses can be to leave the stubbles remain on the soil or to mulch the soil with straw or other plant residues for reducing the risk of the arable soil's freezing (Kurganova and de Gerenyu, 2010). The most effective way for farmers to inhibit releases of N_2O is to avoid excess use of N fertilizer and manures and make sure that they are not applied to waterlogged soils (Granli and Bøckman, 1994).

Table 8.1: N fertilizer Best Management Practices (BMP): options to mitigate N_2O emissions from soils.

Best Management Practices	References
• Timely application of N fertilizer to correspond to crop's ability to make the best use of the N	Aguilera *et al.* (2013)
• Uniform application of N-fertilizers and manure and avoiding application on frozen or water logged soils	Granli and Bøckman (1994)
• Split fertilizer application particularly for urea to reduce NH_3 volatilization and subsequent N_2O emissions	de Klein *et al.* (2001)
• Injection of slurry or rapid incorporation of manure into the soils by tillage	Vander Zaag *et al.* (2011)
• Constructing storage facilities (for animal wastes and feed lots) with proper aeration and moisture management	de Klein *et al.* (2001)
• Reducing unintentional N transfer from agriculture to other ecosystems	de Klein *et al.* (2001)
• Keeping the soil NO_3^- concentrations low during off season	de Klein *et al.* (2001)
• Avoiding fallow fields by growing autumn sown or catch crops and ploughing immediately and planting new crops	de Klein *et al.* (2001); Sangeetha *et al.* (2009)
• Minimum tillage in permanent grasslands to cover the field with grass at the end of the growing season	Petersen, *et al.* (2011); D'Haene *et al.* (2008)
• Reducing fertiliser use or grazing intensity in the autumn in grassland soils	IPNI (2012)
• Avoiding soil compaction and erosion in the agricultural fields	Yamulki and Jarvis (2002); Sitaula *et al.* (2000)

The most relevant management options to reduce N_2O emission from Norwegian agricultural soils as suggested by Bleken and Bakken (1997) are reducing the N input to cultivated land through recycling of food wastes, better utilization of final animal and plant products or by changing the food consumption of Norwegian society from less animal food eating to more vegetarian diet.

8.3 Nitrification inhibitors

One of the fertilization techniques used to mitigate N_2O emissions is applying nitrification inhibitors in soils. Nitrification inhibitors are chemical compounds that inhibit the oxidation of NH_4^+ to NO_3^- in soil and thus reduce N_2O emissions from NH_4^+-based fertilizers and from urine (Di and Cameron, 2002). The most widely used inhibitors are nitrapyrin and Dicyandiamide, DCD (de Klein and Eckard, 2008). Fertilizers coated with nitrification inhibitors have shown to be effective in reducing nitrification and N_2O emissions by up to 80 percent, as reported by de Klein *et al.* (2001). Nitrification inhibitors are often used to increase N fertilizer use efficiency and decrease NO_3^- losses through leaching and denitrification. Nitrification inhibitors can effectively reduce N_2O emissions and yield is generally higher when inhibitors are applied with N fertilizer (Misselbrook *et al.,* 2014).

8.4 Improving land drainage

According to the climate prediction report by Hanssen-Bauer *et al.* (2015) climate change (up to the end of 21st century) will lead to wetter autumns and warmer summers in the south and eastern part of Norway where most of the agricultural lands are located. Under such climate scenarios, the problems of poor drainage in agricultural soils will exacerbate even more and emission of nitrous oxides (from soils with impended drainage) will rise tremendously (Hauge and Tesfai, 2013) unless measures are taken to mitigate nitrous oxide emissions. As denitrification is enhanced under conditions of low soil aeration or high moisture, reducing the waterlogging of agricultural land will reduce potential N_2O emissions. Therefore, one key aspect for mitigation is the control of soil moisture content through land drainage.

Preventing too shallow groundwater table through installation of appropriate drainage systems may be one option to reduce N_2O emissions from mineral agricultural soils (Velthof *et al.,* 1998; Brady and Weil, 2002). On the other hand, maintaining the soils for longer periods of saturation may actually lower N_2O emissions with conditions favouring the complete transformation of NO_3^- to N_2 gas (Bateman and Baggs, 2005; Hernandez and Mitsch, 2006; Nash *et al.,* 2012a).

8.4.1 Surface/subsurface drainage channels

A common practice in the management of seasonally wet soils has been to introduce surface or subsurface drains. In Norway, subsurface tile drainage have been installed in most of the agricultural fields in the early 1900s aiming to increase crop production. However the functionality of these drains is questionable and some of these may even be dysfunctional. It is common to observe standing water in the lower agricultural fields after heavy rains in agricultural lands. Most of the subsurface drainage structures need renovation or be replaced by new drainage systems. However prior to this, there is a need to investigate the type of drainage materials to install, the drainage intensity (i.e. spacing and depth of the drains) which depends on the soil type and climatic conditions of the area. In this regard, there is a dearth of information and data for Norwegian conditions.

The impact of subsurface drainage systems on N_2O emissions is not straightforward. There is a potential trade-offs when improving the drainage conditions of the soils. Waterlogged soils will denitrify more efficiently than well-drained soils but improved drainage will increase N losses through the drains (de Klein and Eckard, 2008). The subsurface drainage channels could also serve as a potential transport system for nutrients losses particularly NO_3^- to the downstream where N_2O could also be emitted from the water surface. However, if the improved drainage only reduces the WFPS of the soil system to below saturation (80 percent), and remains above wilting point (40 per cent), this may actually increase N_2O emissions (Granli and Bøckman, 1994) and could lead to increased NO_3^- leaching. In some cases, stimulating denitrification has been recommended as a way of reducing nitrate leaching in nitrate–sensitive areas (Russelle *et al,* 2005).

8.5 Optimizing tillage operations

Optimizing tillage practices including no tillage or minimum tillage, bed planting, autumn/spring tillage, reducing compaction also help in mitigating the nitrous oxide emission from soils through less soil disturbance, and high nitrogen and water use efficiency. Research on the effects of no-till and conventional tillage on N_2O emission has generated mixed results. Some studies have reported higher N_2O emissions from No till (Ball *et al.,* 1999; Baggs *et al.,* 2003), others have observed lower emissions in no till than conventional tillage (Rochette, 2008) and still others have reported no difference among tillage practices and time of tillage (Choudhary *et al.,* 2002; Yu, 2011; Nadeem *et al.,* 2012b). However, there is insufficient information on the impacts of tillage on N_2O emissions in

Norway. Further investigation is required to suggest which tillage practices are desirable under the Norwegian conditions to keep N_2O emissions to a minimum.

8.6 Biochar applications

Biochar is a carbonaceous material produced during the pyrolysis of biomass. It is the term used for the solid product obtained from pyrolysis of lingo-cellulosic materials that concentrate carbon in a stable form, in order for it to be deliberately applied to soils, increasing their carbon storage and providing agronomic benefits. Specific physical and chemical properties of biochar, such as high porosity (Liang *et al.* 2006), contribute to increase water retention (Lehmann *et al.* 2003). Biochar particulate nature (Skjemstad *et al.* 1996), combined with a specific chemical structure (Baldock and Smernik, 2002) provides great resistance to microbial degradation in soils (Cheng *et al.* 2008). Besides its potential positive effects on soil fertility, when applied to agricultural soils, biochar may contribute to reduce N_2O emissions (Spokas and Reikosky, 2009; Cayuela *et al.* 2010; Van Zwieten *et al.* 2010; Bruun *et al.* 2011; Cayuela *et al.* 2014). Biochar for soil application, has recently been studied, because of the growing attention of climate change on GHG emissions from soils.

In Norway, field experiment with biochar application dose (i.e. 25 t C/ha) measured a cumulative flux of 6 kg N_2O-N ha^{-1} which was less than the control (i.e. 9 kg N_2O-N ha^{-1}) during crop growing season from May to September (pers.comm. Adam O'Toole, 2012). Further investigation is required to come up if biochar could be used as an alternative mitigation measures to N_2O emissions. Other researchers for e.g. Wang *et al.* (2012) reported that biochar application decreased N_2O emissions up to 54 percent and 53 percent during rice and wheat seasons, respectively. The use of biochar may be an alternative option to mitigate N_2O soil emissions from agricultural soils (Wang *et al.* 2011; Alho *et al.* 2012). However, considering N_2O emissions are often dependent on the inherent characteristics of biochar, on the addition of exogenous nitrogen, and on soil properties (Clough *et al.* 2010), the mechanisms by which biochar mitigates N_2O emissions still have to be further examined (Alho *et al.* 2012).

Table 8.2 Effects of Biochar applications on cumulative N_2O-N emissions as reported by some authors.

References	Cumulative N_2O-N	
	With biochar	Without biochar
Case *et al.* (2015)	0.05 ± 0.02 mg N_2O-N kg^{-1}	0.61 ± 0.20 mg N_2O-N kg^{-1}
Verhoeven and Six (2014)	4.14 ± 1.14 to 4.24 ± 0.74 kg N_2O-N ha^{-1} yr^{-1}	2.00 ± 0.66 to 1.60 ± 0.28 kg N_2O-N ha^{-1} yr^{-1}
Angst *et al.* (2013)	0.04 to 0.08 mg N_2O-N kg^{-1}	>0.3 mg N_2O-N kg^{-1}
Suddick and Six (2013)	0.10 ± 0.05 kg N_2O-N ha^{-1} day^{-1}	0.09 ± 0.02 kg N_2O-N ha^{-1} day^{-1}
Case *et al.* (2012)	0.01 N_2O-N µg g dry $soil^{-1}$	0.41 N_2O-N µg g dry $soil^{-1}$

While there is clear evidence that, in many cases, emissions of N_2O are reduced after biochar amendment (Table 8.2), there is still a significant lack of understanding in the key mechanisms and processes involved. In other words, the effect of biochar in mitigating N_2O emissions is still a controversial issue among researchers and practitioners. There are a number of studies showing no changes or even an increase in N_2O emissions from soil with biochar application. A recent study made by García *et al.,* (2014) revealed that the addition of biochar increased N_2O emissions by 54 percent in Haplic Calcisols. Biochar systematically stimulated nitrification, which was probably the cause of the increased N_2O emissions. Predicting which N_2O formation pathway pre-dominates in a certain kind of soil will be necessary to guarantee the success of biochar as a N_2O mitigation strategy (García *et al.,* 2014).

8.7 Biogas slurry applications

The use of digested liquid (i.e. biogas slurry) may reduce the emission of greenhouse gases including N_2O because of a decreased need for inorganic fertilizers (Arthurson, 2009). Singla and Inubushi (2014) conducted a pot-based study at Kujukuri, Chiba, Japan to evaluate the effect of application of digested liquid on N_2O flux in paddy field. Analysis revealed that digested liquid treated soils significantly decrease N_2O emission possibly due to affecting the availability of organic C in the soil to microbial activity for methanogenesis. The availability of organic C is often considered as a major factor influencing denitrification under anaerobic conditions (Zou *et al.* 2005). This is one of the probable reasons that the incorporation of straw biochar and other organic matter can decrease seasonal N_2O emissions by providing readily available C in the soil (Zou *et al.* 2005; Zhang *et al.* 2012; Singla and Inubushi, 2014). The presence of C could potentially inhibit the activity of reductase involved in the conversion of NO_2^{-1} and NO_3^{-1} to N_2O (Zwieten *et al.* 2009). Thus, the presence of C may contribute in reducing N_2O emission by digested liquid application.

8.8 Soil liming

Liming may be an option for the mitigation of N_2O from soils (Stevens et al. 1998; van der Weereden *et al.* 1999) by reducing N_2O to N_2. Soil liming enhances nitrification and the cumulative N_2O emissions under field capacity conditions, are reduced with liming (Clough *et al.* 2003). Numerous field-based studies have observed a decline of in-situ N_2O emissions in response to liming (Brumme and Beese, 1992; Hovlandsdal, 2011). By increasing soil pH, the authors decreased the accumulation of NO_2^- and in turn N_2O emissions; all whilst observing an increase in nitrification with increasing soil pH (Feng *et al.* 2003). However, liming may only be effective at decreasing soil N_2O emissions following summer-rainfall events when the soil has a history of N fertilizer inputs (Barton *et al.* 2013). A laboratory study was carried out by Hovlandsdal, (2011) to test the enzyme that reduces N_2O to N_2 during denitrification (N_2O)-reductase, whether it has an effect on in situ N_2O emissions or not. Soil samples were collected from long-term lime experiment field in Fureneset (Norway) in outer Sunnfjord. The study showed that liming has a great potential to reduce fertilizer induced N_2O emissions from cultivated organic soils.

However, the above mentioned studies are contrary to the common view that increasing soil pH will increase soil N_2O emissions due to increases in microbial activity (Simek and Cooper, 2002; Page *et al.* 2009). Field study was carried out by Galbally et al. (2010) in Australia to investigate the effect of liming on N_2O emissions. This study has concluded that liming did not make any significant change to the average N_2O emissions measured for the limed plots (0.88 ± 0.04 mg N m^{-2} d^{-1}) against the non-limed plots (0.96 ± 0.07 mg N m^{-2} d^{-1}). Hence, there is a need for more comprehensive studies on effects of liming regarding its mitigation potential.

8.9 Summary

Overall, there is a lack of comprehensive studies made on the above-mentioned N_2O mitigation practices in Norway. Hence, information are scanty regarding which mitigation practices are best to apply to reduce N_2O emissions across the different Norwegian agricultural systems.

9 CONCLUSIONS

This report has highlighted the importance of nitrous oxide (N_2O), which is one of potent GHGs emitted predominantly from agricultural soils. We have described the main factors that promote the process of dentrification and nitrification of N_2O formation and the techniques and models used to predict N_2O emissions. Around 225 published literatures spanning from 1974 to 2015 were reviewed and selected papers were discussed on their effects of N_2O production and emissions from agricultural soils. The review covers effects of soil management practices including N-fertilization, soil compaction, soil tillage; effects of soil moisture and drainage; effects of soil pH, effects of soil drying-rewetting, freezing and thawing effects on N_2O production and emissions. Moreover, a range of alternative measures that can reduce N_2O emissions from agricultural soils, are presented and future research needs relevant to the Norwegian conditions are suggested.

9.1 Knowledge gaps and research needs

Although, there is a large body of knowledge available on rates of nitrification and denitrification in soils, little is known about N_2O production, transport, and consumption as well as emissions from field to catchment scale over time. Moreover, there is still a significant lack of information on the best mitigation measures to apply to reduce N_2O emissions from soils. The following research gaps are identified and listed in Table 9.1 which are not in order of importance.

Table 9.1 Research needs to measure and mitigate N_2O emissions from Norwegian agricultural soils.

Research needs	*Rationale*
• There is a need to revisit the IPCC emission factor used to estimate national N_2O emissions	• Increased use of N fertilizers will continue to produce sufficient food and sustain food security.
• The agronomic, economic and environmental consequences of reducing N fertilization as a mitigation measure	• Reducing mineral N input can often lead to proportional decrease in crop yields and shortage of food supply in the short term.
• Risk of high N_2O emissions from cultivate organic and/or mineral soils during the winter period	• When weeds are killed by herbicides during early autumn followed by excessive manure application and ploughing in late autumn, is a potential N_2O bomb which should be avoided.
• Manure and urine management to reduce direct N_2O emissions and secondary emissions from NH_3 volatilization	• Efforts must be made to find effective and affordable solutions for spreading animal fertilizers that secure maximum utilization of applied N via 4Rs: Right type, Right dose, Right time, and Right placement.
• Indirect emissions of N_2O and/or enhanced NO_3^- leaching from drainage waters and streams in agricultural catchments	• Very few data are available on indirect emissions of N_2O from nitrate leaching and run-off in agricultural fields.
• Effect of freezing-thawing cycles on emissions of N_2O during winter and early spring	• Lack of knowledge on the mechanisms controlling N_2O emissions during the freezing-thawing periods and their impacts on annual N budgets.
• Effect of soil pH on N_2O emissions via denitrification	• The mechanism of pH control in the N_2O/N_2 ratio of denitrification is not fully understood
• Modelling the different driving factors leading to changes in N_2O/N_2 ratio	• Limited knowledge and data on modelling N_2O emissions
• How does the released N from organic materials affect emissions of N_2O under cereal cropping system?	• Limited knowledge and data on emission of N_2O from organic fertilizers in cereal fields.
• Effects of biochar, biogas slurry and/or liming on N_2O emissions and associated soil improvements	• There are conflicting results on the effects of biochar, biogas slurry and/or liming as an alternative option for mitigation of N_2O soil emissions.

To this end:

- There is a need to increase our knowledge and understandings of the N_2O fluxes at various spatial and temporal scales by taking into account the Nitrogen system as a whole with its various inputs and outputs and interactions during the key microbial N transformation processes via denitrification and nitrification, and
- Improved estimates of N_2O emission from agricultural soils and mitigation options can be achieved through national research program that covers a wider sampling seasons and area, combining different methods (i.e. chamber and micrometeorological) and using high precision analytical instruments and simulation modelling. The research program should include various pedo-climatic conditions and agricultural systems across Norway.

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Norsk institutt for bioøkonomi (NIBIO) ble opprettet 1. juli 2015 som en fusjon av Bioforsk, Norsk institutt for landbruksøkonomisk forskning (NILF) og Norsk institutt for skog og landskap.

Bioøkonomi baserer seg på utnyttelse og forvaltning av biologiske ressurser fra jord og hav, fremfor en fossil økonomi som er basert på kull, olje og gass. NIBIO skal være nasjonalt ledende for utvikling av kunnskap om bioøkonomi.

Gjennom forskning og kunnskapsproduksjon skal instituttet bidra til matsikkerhet, bærekraftig ressursforvaltning, innovasjon og verdiskaping innenfor verdikjedene for mat, skog og andre biobaserte næringer. Instituttet skal levere forskning, forvaltningsstøtte og kunnskap til anvendelse i nasjonal beredskap, forvaltning, næringsliv og samfunnet for øvrig.

NIBIO er eid av Landbruks- og matdepartementet som et forvaltningsorgan med særskilte fullmakter og eget styre. Hovedkontoret er på Ås. Instituttet har flere regionale enheter og et avdelingskontor i Oslo.

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