Emissions of N$_2$O from agricultural soils and mitigation options:
A review with special reference to Norwegian agriculture

Mehreteab Tesfai
Avdeling for Jordressurser og Areal bruk
**Tittel/Title:**
Emissions of N₂O from Agricultural Soils and Mitigation Options: A review with special reference to Norwegian agriculture

**Forfatter/author:**
Mehreteab Tesfai

<table>
<thead>
<tr>
<th>Dato/date:</th>
<th>Rapport nr./report no.:</th>
<th>Tilgjengelighet/availability:</th>
<th>Prosjekt nr./project no.:</th>
<th>Saksnr./archive no.:</th>
</tr>
</thead>
<tbody>
<tr>
<td>01.02.2016</td>
<td>2(25) 2016</td>
<td>Åpen</td>
<td>8126.3 &amp; 8877.4</td>
<td>2016/211</td>
</tr>
</tbody>
</table>

**ISBN-NR./ISBN-NO:**
978-82-17-01585-7

**Oppdragsgiver/employer:**
Jordressurser og Arealbruk

**Stikkord/keywords:**
Agricultural soils, emission, nitrous oxide, mitigation

**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 for lystgassutslipp fra jordbruksjord. Effekten av gjødsel, jordpakking, jordarbeiding, jordfuktighet, drenering, pH og froste-tving er beskrevet. Det er også foreslått ulike tiltak for reduksjon av N₂O-utslipp som er relevant for norske forhold. Til slutt konkluder rapporten med forslag om framtidig forskning og måling som grunnlag for reduksjon av N₂O-utslipp fra jordbruksjord.

This report gives a comprehensive review of current knowledge on nitrous oxide (N₂O) 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 N₂O followed by descriptions of the pathways to N₂O production and factors affecting N₂O emissions from agricultural soils and measurement techniques and modelling N₂O 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 N₂O emissions. Furthermore, some relevant mitigation measures to reduce N₂O emissions are presented. The report concludes by suggesting future research needs to measure and mitigate soil N₂O emissions.

**GODKJENT /APPROVED**

[Signature]

Jannes Stolte

**PROSJEKTELEDER /PROJECT LEADER**

[Signature]

Marianne Bechmann
PREFACE

This review report is about nitrous oxide (N₂O) which is one of the potent greenhouse gases emitted predominantly from agricultural soils. The report highlights the importance of N₂O with special reference to Norwegian agriculture by synthesizing information and data from several literatures pertaining to N₂O 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 AGRLOSS 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 .......................................................................................................................... 1

2 NITROUS OXIDE .......................................................................................................................... 3
   2.1 Why N\textsubscript{2}O? ........................................................................................................... 3
   2.2 How is N\textsubscript{2}O produced? ......................................................................................... 4
   2.3 Factors affecting N\textsubscript{2}O production and emissions ................................................ 6
      2.3.1 The factors ................................................................................................................... 6
      2.3.2 The processes ............................................................................................................... 8
   2.4 Summary .............................................................................................................................. 9

3 N\textsubscript{2}O MEASUREMENTS & MODELLING ......................................................................... 10
   3.1 Chamber methods ............................................................................................................... 10
      3.1.1 Introduction ................................................................................................................... 10
      3.1.2 Advantages of chamber method ................................................................................... 12
      3.1.3 Disadvantages of chamber method .............................................................................. 12
   3.2 Micrometeorological method .............................................................................................. 13
      3.2.1 Advantages of Micrometeorological methods ............................................................... 13
      3.2.2 Disadvantages of micrometeorological methods .......................................................... 13
   3.3 Approaches to quantify denitrification (N\textsubscript{2}O and N\textsubscript{2}) .................................... 13
   3.4 Modelling N\textsubscript{2}O emissions ...................................................................................... 14
      3.4.1 Modelling approaches ................................................................................................. 15
      3.4.2 Model validation ........................................................................................................... 18
      3.4.3 Model limitations .......................................................................................................... 19
   3.5 Summary .............................................................................................................................. 20

4 EFFECTS OF SOIL MANAGEMENT PRACTICES ON N\textsubscript{2}O EMISSIONS ........................ 21
   4.1 Inorganic and organic fertilization ...................................................................................... 21
      4.1.1 Norway .......................................................................................................................... 21
      4.1.2 Other countries experience .......................................................................................... 25
   4.2 Soil compaction ................................................................................................................... 25
      4.2.1 Norway .......................................................................................................................... 25
      4.2.2 Other countries experience .......................................................................................... 26
   4.3 Soil tillage ............................................................................................................................. 27
      4.3.1 Norway .......................................................................................................................... 27
      4.3.2 Other countries experience .......................................................................................... 28
   4.4 Summary .............................................................................................................................. 29

5 EFFECTS OF SOIL MOISTURE & DRAINAGE ON N\textsubscript{2}O EMISSIONS .................................. 30
   5.1 Soil moisture & drainage .................................................................................................... 30
      5.1.1 Scandinavia countries ................................................................................................... 30
      5.1.2 Other countries experience ........................................................................................ 36
   5.2 Soil drying-rewetting ......................................................................................................... 36
      5.2.1 Other countries experience ........................................................................................ 37
   5.3 Summary .............................................................................................................................. 38

6 EFFECTS OF SOIL PH ON N\textsubscript{2}O EMISSIONS .................................................................. 39
   6.1 Introduction ......................................................................................................................... 39
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\textsubscript{2}O) 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\textsubscript{2}O and eventually N\textsubscript{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\textsubscript{2}O-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\textsubscript{2}O national emissions. According to the climate prediction report by Hanssen-Bauer et al. (2015) climate change up to the end of 21\textsuperscript{st} 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\textsubscript{2}O-N from agricultural soils (Granli and Bøckman, 1994).

So far, a comprehensive review on the process, measurements and mitigation measures of N\textsubscript{2}O 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\textsubscript{2}O 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\textsubscript{2}O formation and emissions; research efforts made to measure N\textsubscript{2}O 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\textsubscript{2}O 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\textsubscript{2}O 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\textsubscript{2}O 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\textsubscript{2}O and its pathways and factors controlling the formation and emission of N\textsubscript{2}O. Chapter 3 deals with the measurement techniques of N\textsubscript{2}O 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\textsubscript{2}O 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\textsubscript{2}O 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\textsubscript{2}O emissions from agricultural soils under Norwegian conditions. From chapters 3 to 7, relevant research experiences of other countries on measurements and mitigation of N\textsubscript{2}O 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)](image-url)
2 NITROUS OXIDE

This chapter highlights the significance of nitrous oxide (N₂O) as a greenhouse gas and its effect on climate change and food production. Moreover, it gives a description of the pathways of N₂O to production and emissions processes and the main factors that affect the N₂O emissions from agricultural soils.

2.1 Why N₂O?

Increasing atmospheric content of some trace gases causes serious environmental concern. One of such gas is nitrous oxide - a chemical compound with the formula N₂O (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 N₂O emissions, to which agriculture contributes 81 percent (Isermann, 1994). It is estimated that annual global emissions from agriculture is about 6 ×10³ Gg N₂O-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 N₂O from agricultural soils amounted to 1.57 Metric tonnes (calculated in CO₂-equivalents) in 2013. The agricultural soils accounted for about 64 percent of the total Norwegian N₂O 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 CO₂ and CH₄. Global anthropogenic emissions of N₂O presently amounts to 4,5 ± 0,6 Tg N yr⁻¹ since preindustrial era (Khali and Rasmussen, 1992; IPCC, 2007). The atmospheric N₂O 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 CO₂ 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 ‘N₂O 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 N₂O 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 N₂O 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 N₂O in its wider definition of global change so as to embrace its effect on the ozone layer (Smith, 2010).

Despite N₂O 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 N₂O emissions from agricultural soils on one hand and increasing food production on the other hand.
2.2 How is N₂O produced?

To predict how much N₂O is produced from each unit of fixed N (chemically or biologically) that is added to the soil, we must first understand how and where N₂O 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 N₂O 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 O₂; and
- Availability of nitrogen oxides: NO₃⁻, NO₂⁻, NO, or N₂O 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 N₂O 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 (O₂ present) process that converts ammonium (NH₄⁺) into nitrate (NO₃⁻) with N₂O as a by-product. This happens when the supply of O₂ is limited by diffusional constraints where by the nitrifying bacteria can use nitrite as an electron acceptor and reduce it to NO and N₂O. Otherwise, the process will proceed as NH₄⁺, NO₂⁻, NO₃⁻. In other words, nitrification occur when soil bacteria convert fertilizer N from the ammonium form (NH₄⁺) and/or NH₄⁻N mineralized from soil organic matter (SOM) to the nitrite (NO₂⁻) and nitrate (NO₃⁻) forms. In order for nitrification to proceed, fertilizer needs to be added in the form of NH₄⁺ or other forms (such as anhydrous ammonia [NH₃] or urea) which are converted to NH₄⁺ in the soil. Nitrification is more likely to occur in drier conditions and in coarse-textured soils where rates of N₂O production by nitrification tend to be smaller (Williams et al., 1992).
(ii) Denitrification: Denitrification is an anaerobic (O₂ absent) process that converts nitrate (NO₃⁻) into nitrogen gas (N₂) with N₂O being produced as an intermediate product. In other words, denitrification of NO₃⁻, N₂O and ultimately to molecular nitrogen (N₂). During denitrification, soil bacteria convert NO₃⁻ to NO₂⁻, then to nitric oxide (NO) gas, and then to N₂O in a sequence of reactions. In order for denitrification to proceed, N must be present in the form of NO₃⁻. The NO₃⁻ can be present either because fertilizer NH₄⁺ has been converted to NO₃⁻ via nitrification, or if NO₃⁻ is added directly, for example as ammonium nitrate (NH₄NO₃). Denitrification is commonly thought to be the dominant process responsible for the N₂O 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 N₂O 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, NO₃⁻ 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

<table>
<thead>
<tr>
<th>Denitrification</th>
<th>Nitrification</th>
<th>References</th>
</tr>
</thead>
<tbody>
<tr>
<td>• Denitrification is an anaerobic process</td>
<td>• Nitrification is aerobic process</td>
<td>Williams et al. (1992)</td>
</tr>
<tr>
<td>• (\text{NO}_3^- \rightarrow \text{NO}_2^- \rightarrow \text{NO} \rightarrow N_2O \rightarrow N_2)</td>
<td>• (\text{NH}_4^+ \rightarrow \text{NO}_3^- \rightarrow N_2O) or (\text{NH}_4^+ \rightarrow \text{NO}_2^- \rightarrow \text{NO}_3^- \rightarrow N_2O)</td>
<td>Granli and Bøckman (1994)</td>
</tr>
<tr>
<td>• Denitrification tends to occur under wet conditions and in slowly draining soils that do not allow for rapid replenishment of (O_2)</td>
<td>• Nitrification tends to occur in moderately well-drained soils and during drier periods.</td>
<td>Davidson et al. (2000)</td>
</tr>
</tbody>
</table>
| • 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, \(\text{NO}_3^-\) 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 \(\text{NH}_4^+\) 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 \(\text{NO}_3^-\) and \(\text{NH}_4^+\) 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 \(\text{NO}_3^-\) 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 \(\text{NO}_3^-\) 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 \(\text{NH}_4^+\) 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 \(\text{NO}_3^-\) 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 denitrification 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 ºC (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 ($\geq 60$ % 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).
v) Soil pH: The effect of pH on N\(_2\)O emission from soils is complex. Conflicting results are reported. Where denitrification is the main source of N\(_2\)O, 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\(_2\)O, emissions tend to increase with increasing pH, at least in the range of pH 6 to 8. However, the rate of N\(_2\)O 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\(_2\)O emission at the soil surface is the result of nitrogen production and consumption processes. N\(_2\)O emissions originating from agricultural land use include direct and indirect emission of N\(_2\)O. 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\(_2\)O 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\(_2\)O emitted at the soil surface is a mixture of
ground water-derived N₂O of mostly agricultural origin, and N₂O 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 N₂O 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 N₂O 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 N₂O 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, N₂O 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 N₂O on a field scale basis (Mosier and Heinemeyer, 1985). The full characterization of N₂O 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 N₂O:N₂ ratio (Zaman and Nguyen, 2010). The interactions among these factors are likely to control denitrification and the N₂O:N₂ ratio under field conditions, and these interactions remain poorly understood. We understand the general trends of change in the N₂O:N₂ ratio with changes in some of the individual factors. However, how these factors interact to affect denitrification and N₂O:N₂ 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 O₂ diffusion and the presence of labile C. The balance between factors promoting and factors hindering N₂O emissions determines the outcome of N₂O emissions from a given agricultural system.
3  \textbf{N}_{2}\text{O} \text{ 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\textsubscript{2}O emissions from different land use systems, are reviewed. The estimation of N\textsubscript{2}O 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\textsubscript{2}O (Bouwman et al, 2010). There are broadly two methods for measuring N\textsubscript{2}O 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\textsubscript{2}O production and consumption processes on environmental controls such as substrate availability, redox potential and temperature, N\textsubscript{2}O fluxes from soils are notoriously variable across various temporal and spatial scales. However, understanding spatial variability of N\textsubscript{2}O fluxes is essential to better constrain the magnitude of soil–atmosphere exchange of N\textsubscript{2}O 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\textsubscript{2}O 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 measure soil \textsubscript{2}N\text{O} emissions from agricultural soils (Photo 3.1).

\textit{i) Non-steady state chambers:} The NSS chambers are used to measure the soil-surface flux of relatively inert gases such as CO\textsubscript{2}, CH\textsubscript{4}, and N\textsubscript{2}O. For e.g. N\textsubscript{2}O emission rates are commonly determined by enclosing the atmosphere above the source (soil, manure, or water body) and measuring the increase in headspace N\textsubscript{2}O concentration over time. In Non-Steady State (NSS) chambers, the flux of the gas of interest (F\textsubscript{N\textsubscript{2}O} \ \mu g \ \text{m}^{-2} \ \text{h}^{-1}) is calculated using the rate of change of its concentration \ ((\text{dc/dt; mol mol}^{-1} \ \text{s}^{-1})) inside the chamber during deployment time (Rochette and Hutchinson, 2005) as presented below:

\[ F_{N_2O} = \frac{N_2O}{dt} \times \frac{V_C}{A} \times \frac{M_n}{V_m} \times 60 \]  \[ \text{[1]} \]

where \( F_{N_2O} \) is the N\textsubscript{2}O flux (\( \mu g \ \text{N}_2\text{O-N m}^{-2} \ \text{h}^{-1} \)), \( d_{N_2O} \) (\( \text{ppm N}_2\text{O min}^{-1} \)) 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\textsuperscript{2}), \( M_n \) is the molecular mass of N in N\textsubscript{2}O (28 g N mol\textsuperscript{-1}) and \( V_m \) is the molecular volume of the gas at the mean temperature during chamber deployment (L mol\textsuperscript{-1}).
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 $\text{N}_2\text{O}$ emissions (kg $\text{N}_2\text{O}$-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 $\text{N}_2\text{O}$ 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 $[\text{CO2}]_{ch}$ (Rochette and Hutchinson, 2005). An increase in $[\text{CO2}]_{ch}$ during chamber deployment has a direct impact on $F_c$. SS chambers are designed to reduce this impact by measuring $F_c$ at constant $[\text{CO2}]_{ch}$. 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 N₂O emission from soils (Fowler et al., 1997). The closed chamber technique has a relatively good confidence in the N₂O 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 m²) 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 N₂O and variability of N₂O fluxes due to sampling site and sampling strategy are additional limitations. The main disadvantage of chamber technique is that it measures N₂O 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$_2$O 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$_2$O emissions on a field-scale, and thus spatially integrate N$_2$O 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$_2$O 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$_2$O 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$_2$O 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$_2$O and N$_2$)

The two most commonly used approaches for determining denitrification rate from measurements of N$_2$ and N$_2$O production include a technique based on the acetylene (C$_2$H$_2$) inhibition (AI) of N$_2$O 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>
<thead>
<tr>
<th>Measurement method</th>
<th>Advantages</th>
<th>Disadvantages</th>
</tr>
</thead>
<tbody>
<tr>
<td>• Acetylene inhibition (AI) technique Approaches:</td>
<td>• Simple to conduct</td>
<td>• Can only be used in NO₃⁻ dominated systems</td>
</tr>
<tr>
<td></td>
<td>• Can run large number of samples at a time</td>
<td>• Inhibits nitrification; and can underestimate denitrification</td>
</tr>
<tr>
<td></td>
<td>• Removes the spatial and temporal variability of denitrification rate</td>
<td>• Slow diffusion of C₂H₂ into soil or sediments limits blockage of N organic</td>
</tr>
<tr>
<td>• ¹⁵N tracer technique</td>
<td>• Considered better than Argon technique</td>
<td>• Laborious process</td>
</tr>
<tr>
<td></td>
<td>• Gives reliable estimates of denitrification</td>
<td>• Requires costly instruments</td>
</tr>
<tr>
<td>• Calcium carbide (CaC₂) granules In situ chambers in field Static cores</td>
<td>• Useful in studying the effect of soil and environmental factors on denitrification and denitrification enzyme activity assay</td>
<td>• Addition of ¹⁵N to the N limiting condition results in overestimation of denitrification</td>
</tr>
<tr>
<td>• Direct N₂ quantification</td>
<td>• No labelled N or inhibitor is added</td>
<td>• Rapid decomposition of C₂H₂ by microbes</td>
</tr>
<tr>
<td></td>
<td>• Highly sensitive method can even detect low denitrification changes</td>
<td>• Contamination of C₂H₂ with other gases can affect denitrification</td>
</tr>
<tr>
<td></td>
<td>• Can be used to study temperature and moisture effect</td>
<td>• Scavenging of NO leading to underestimation of denitrification</td>
</tr>
<tr>
<td>• Molecular approaches</td>
<td>• Give reliable and realistic qualitative indication of enzymes and encoding genes involved in denitrification</td>
<td>• Can only be used in enclosed incubation experiments</td>
</tr>
<tr>
<td>Polymerase chain reaction DNA microarray technique Immunological techniques</td>
<td>• Can be used both in aquatic and terrestrial</td>
<td>• Complex and difficult system</td>
</tr>
<tr>
<td></td>
<td></td>
<td>• Not suitable for longer period</td>
</tr>
<tr>
<td></td>
<td></td>
<td>• Challenge to measure N₂ against high atmospheric N₂ concentration</td>
</tr>
<tr>
<td></td>
<td></td>
<td>• Require very expensive equipment</td>
</tr>
<tr>
<td></td>
<td></td>
<td>• Require technical expertise to extract, analyse and interpret enzymes and encoding genes</td>
</tr>
<tr>
<td></td>
<td></td>
<td>• Useful to study microbial diversity</td>
</tr>
</tbody>
</table>

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

### 3.4 Modelling N₂O emissions

As a major source of N₂O production from agricultural soils, the denitrification process must be an important part of any process-based soil N₂O model. Nitrous oxide emissions are highly variable both in space and time. And the estimation of N₂O emissions from individual fields based on actual measurements is very costly. Moreover, the high temporal and spatial variability of agricultural nitrous oxide (N₂O) 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 N₂O emissions, for estimating N₂O 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 N₂O 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 NO₃⁻ concentration.

i) IPCC emission methodology: N₂O emissions from agriculture are estimated using the IPCC guidelines for national greenhouse gas inventories (IPCC, 1996). National N₂O 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 N₂O emission from agriculture: for example in Norway, the N₂O emissions are estimated by Statistic Norway (Sandmo et al. 2014) as follows:

(a) Direct emissions of N₂O 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 = 10⁹ 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 N₂O from application of synthetic fertilizers in Norway is estimated to be 1.38 Gg N₂O-N yr⁻¹ and indirect emission from leached N and runoff N to be 0.83 Gg N₂O-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 N₂O emission from animal manure applied to soil was estimated around 0.6 Gg N (Table 3.2).

Biological N-fixation (FBN) is estimated around 8 Gg N yr⁻¹. Using the IPCC default emission factor (1.25 percent), the N₂O emission from biological N-fixation is estimated around 0.1 Gg N₂O-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. N₂O emission from crop residues is estimated to be about 1.1 Gg N₂O-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 N₂O emission factor of 5 kg N₂O-N ha⁻¹ yr⁻¹ (IPCC,
1996), the N\textsubscript{2}O emission from organic soils is estimated to be 0.9 Gg N\textsubscript{2}O-N yr\textsuperscript{-1} (Table 3.2). The total direct emission of N\textsubscript{2}O from agricultural soils is about 3.98 Gg N\textsubscript{2}O-N yr\textsuperscript{-1} (Table 3.2).

Table 3.2: Direct N\textsubscript{2}O emission (Gg N\textsubscript{2}O-N yr\textsuperscript{-1}) from agricultural soils in Norway (N\textsubscript{2}O\textsubscript{DIRECT}).

<table>
<thead>
<tr>
<th>Emission due to</th>
<th>Gg N\textsubscript{2}O-N yr\textsuperscript{-1}</th>
<th>% of the total</th>
<th>Mt CO\textsubscript{2}-eq. (2010)*</th>
</tr>
</thead>
<tbody>
<tr>
<td>Synthetic fertilizer use (N\textsubscript{2}O\textsubscript{SN})</td>
<td>1.38</td>
<td>35</td>
<td>0.6</td>
</tr>
<tr>
<td>Crop residues returned to soil (N\textsubscript{2}O\textsubscript{CR})</td>
<td>1.10</td>
<td>27</td>
<td>0.1</td>
</tr>
<tr>
<td>Cultivated organic soil (N\textsubscript{2}O\textsubscript{OS})</td>
<td>0.90</td>
<td>23</td>
<td>0.3</td>
</tr>
<tr>
<td>Animal manure (N\textsubscript{2}O\textsubscript{ANIMALS})</td>
<td>0.60</td>
<td>15</td>
<td>0.6</td>
</tr>
<tr>
<td>Agricultural soils in country N\textsubscript{2}OdIRECT)</td>
<td>3.98</td>
<td>100</td>
<td>1.6</td>
</tr>
</tbody>
</table>

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

(b) Emissions of N\textsubscript{2}O from animal production: i.e. emission from droppings on pastures and animal waste storage/treatment previous to application to agricultural soils. The total N\textsubscript{2}O emissions from animal excreta on pastures (N\textsubscript{2}O\textsubscript{ANIMALS}) was estimated to be ca. 18 Gg N\textsubscript{2}O-N yr\textsuperscript{-1}. F\textsubscript{E}\textsubscript{GASM} (fraction of livestock N excreta) that volatilizes as NH\textsubscript{3} and NO (kg NH\textsubscript{3}-N + NO\textsubscript{x}-N/kg N excreted) constituted 0.2 Gg N\textsubscript{2}O-N yr\textsuperscript{-1}. F\textsubscript{EF}\textsubscript{3}; emission factor for animal N excreta on pastures (kg N\textsubscript{2}O-N/kg N excreted) estimated to be 0.02 Gg N\textsubscript{2}O-N yr\textsuperscript{-1}. N\textsubscript{2}O\textsubscript{ANIMALS}: N\textsubscript{2}O emission from grazing on cultivated and uncultivated land to be 0.29 Gg N\textsubscript{2}O-N yr\textsuperscript{-1}.

(c) Indirect emissions of N\textsubscript{2}O 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\textsubscript{2}O emission (N\textsubscript{2}O\textsubscript{G}). As mentioned above, N\textsubscript{2}O 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\textsubscript{2}O emission factor of 1 percent (IPCC, 1996) (N\textsubscript{2}O\textsubscript{G}) was estimated to be 0.12 Gg N\textsubscript{2}O-N yr\textsuperscript{-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\textsubscript{2}O 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\textsubscript{2}O 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\textsubscript{2}O emission from leached and runoff N in Norway was estimated to be about 1.37 Gg N\textsubscript{2}O-N yr\textsuperscript{-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\textsubscript{2}O emission. Estimation of N\textsubscript{2}O 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\textsubscript{2}O. Aakre and Azzaroli (1997) have treated sewage and solid waste as one post, and estimated the total emission of N\textsubscript{2}O (N\textsubscript{2}O\textsubscript{g}) to be about 0.26 Gg N\textsubscript{2}O-N yr\textsuperscript{-1} (Table 3.3). The N\textsubscript{2}O emission factor used is the IPCC default factor (i.e. 1 percent). N\textsubscript{2}O\textsubscript{g} 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 ($N_2O$\text{\textsc{indirect}}).

| Source: Adapted from Statistic Norway (2009) and * Grønlund and Hansen (2010). |
| --- | --- | --- | --- |
| **$N_2O$ produced from N leaching and runoff $N_2O(L)$** | 1.410 | 77 | 0.3 |
| **$N_2O$ produced from human sewage $N_2O(G)$** | 0.255 | 14 | n.d. |
| **$N_2O$ produced from atmospheric deposition of NOx and NH$_3$ $N_2O(G)$** | 0.156 | 9 | 0.1 |
| **$N_2O$ produced from N used in agriculture $N_2O$\text{\textsc{indirect}}** | 1.821 | 100 | 0.4 |

According to IPCC (1996), the total $N_2O$ emission indirectly induced by agricultural activities $N_2O$\text{\textsc{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 $N_2O$\text{\textsc{total}}.

| Source: Adapted from Statistic Norway (2009) and * Grønlund and Hansen (2010). |
| --- | --- | --- |
| **Direct $N_2O$ emission from agricultural soils ($N_2O$\text{\textsc{direct}})** | 3.98 | 66 |
| **Indirect $N_2O$ emission from N used in agriculture ($N_2O$\text{\textsc{indirect}})** | 1.82 | 35 |
| **$N_2O$ emission from grazing on cultivated and uncultivated land ($N_2O$\text{\textsc{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, N₂O and N₂ based on soil redox potential and dissolved organic carbon. (Li et al. 1992a &b; Li, 2000).

The denitrification submodel calculates hourly denitrification rates, N₂O and N₂ 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 N₂O 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 N₂O emissions. As a process-based model, DNDC is capable of predicting the soil fluxes of all three terrestrial GHGs namely, N₂O, CO₂, and CH₄, as well as other important environmental and economic indicators such as crop production, ammonia (NH₃) volatilisation and nitrate (NO₃⁻) 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 N₂O and N₂ 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 N₂O fluxes from grasslands reasonably well, it simulates these fluxes very poorly on a daily basis because of the significance of diurnal fluctuations in N₂O 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 N₂O 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 N₂O measurements. This is not necessarily a good test of the denitrification model, as N₂O is a product of both nitrification and denitrification, while denitrification also produces N₂. So, it is possible for a model to predict the N₂O 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 N₂O fluxes, they often produced very different estimates of gaseous losses as NO, N₂ and NH₃ 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 NO₃⁻ leaching, NH₃ 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. N₂O, N₂, NH₃, soil NO₃⁻ and NH₄⁺) 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 fₜ used, calibration of N₂O:N₂, indicator of microbial activity of denitrification models.

<table>
<thead>
<tr>
<th>Model name</th>
<th>Form of fₜ</th>
<th>Variables used in N₂O:N₂ ratio</th>
<th>Indicator of microbial activity</th>
<th>References</th>
</tr>
</thead>
<tbody>
<tr>
<td>DNDC</td>
<td>n.a.</td>
<td>WFPS</td>
<td>Microbial biomass</td>
<td>Li et al. (1992)</td>
</tr>
<tr>
<td>FASSET</td>
<td>Michaelis-Menten</td>
<td>Temperature, WFPS, clay, depth</td>
<td>Mineralisation rate</td>
<td>Chatskikh et al. (2005)</td>
</tr>
<tr>
<td>DAYCENT</td>
<td>n.a.</td>
<td>NO₃⁻/CO₂ ratio, WFPS</td>
<td>CO₂ concentration</td>
<td>Stehfest and Müller (2004)</td>
</tr>
<tr>
<td>InfoCrop</td>
<td>First-order</td>
<td>n.a.</td>
<td>Microbial biomass</td>
<td>Aggarwal et al. (2006)</td>
</tr>
<tr>
<td>NEMIS/NOE</td>
<td>Michaelis-Menten</td>
<td>n.a.</td>
<td>Site-specific constant</td>
<td>Hénault and Germon (2000); Hénault et al. (2005)</td>
</tr>
<tr>
<td>NGAS</td>
<td>n.a.</td>
<td>Soil respiration and NO₃⁻, WFPS</td>
<td>Respiration rate</td>
<td>Parton et al. (1996)</td>
</tr>
<tr>
<td>NOE</td>
<td>Michaelis-Menten</td>
<td>Empirical site-based parameter</td>
<td>n.a.</td>
<td>Hénault et al. (2005)</td>
</tr>
<tr>
<td>PaSim</td>
<td>n.a.</td>
<td>n.a.</td>
<td>Decomposition rate</td>
<td>Schmid et al. (2001)</td>
</tr>
<tr>
<td>WNMM</td>
<td>First-order</td>
<td>WFPS</td>
<td>Soil organic carbon</td>
<td>Li et al. (2007)</td>
</tr>
</tbody>
</table>


Table 3.5 lists the form of fₜ used in selecting denitrification models, factors considered in calculating the N₂O:N₂ 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 N₂O production from soil after parameterization with the local data, and appropriate modification and verification against the measured N₂O emissions under different management practices (Dalal et al., 2003).

3.4.3 Model limitations

Factors affecting the simulation or modelling of N₂O emissions are:

- Most mechanistic models include algorithms describing N₂O production in the soil, but models do not include processes and time steps for the actual N₂O 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 N₂O:N₂ ratio for denitrification products is not capable of determining situations where the N₂O:N₂ ratio changes. Improved understanding of the underlying processes can lead to improvements in the models. However, as N₂O 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 N₂O 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 N₂O 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 N₂O 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 N₂O 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 N₂O 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 N₂O emissions prediction ability across different Norwegian agricultural systems.
4 EFFECTS OF SOIL MANAGEMENT PRACTICES ON N$_2$O EMISSIONS

In this chapter, the effects of soil management practices notably fertilizer applications and tillage methods (associated with soil compaction) on N$_2$O 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$_2$O emissions following the application of inorganic and organic fertilizers in Western Norway. The objective of the research was to quantify N$_2$O emissions from fertile grassland soils dominated by sandy loam, and to estimate the response of seasonal N$_2$O 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$_4$NO$_3$ 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$_2$O 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$_2$O 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$_2$O 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$_2$O 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$_2$O emissions from fertilizer-response alone.

In 2009, emissions of soil N$_2$O-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$_2$O-N and
also lowest soil NO$_3^-$-N and soil NH$_4^+$-N levels. However, during August to September, an inverse relationship was observed between the soil NO$_3^-$-N and soil NH$_4^+$-N levels and soil N$_2$O-N flux rates. In other words, higher emissions of N$_2$O 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$_2$O-N fluxes, NO$_3^-$-N and NH$_4^+$-N levels discerned in all treated plots (Figure 4.1).

![Temporal dynamics of N$_2$O fluxes, soil NH$_4^+$-N concentration, soil NO$_3^-$-N concentration, and ancillary variables measured in intensively managed grassland in Western Norway in 2009.](image)

Figure 4.1 Temporal dynamics of a) N$_2$O 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$_2$O 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$_4$NO$_3$ (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 ($<20 \mu g \ N_2O-N/m^2/hr$) 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$_2$O 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$_4$NO$_3$ 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$_2$O 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).

<table>
<thead>
<tr>
<th>Year</th>
<th>Kg N$_2$O-N/ha</th>
<th>mg N$_2$O-N kg/DM herbage</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>NT CF OF NT CF OF</td>
<td></td>
</tr>
<tr>
<td>2009</td>
<td>1,6 ± 0,2 3,8 ± 3,4 2,0 ± 0,5</td>
<td>132 ± 25 380 ± 437 156 ± 111</td>
</tr>
<tr>
<td>2010</td>
<td>0,7 ± 0,3 2,2 ± 1,2 0,2 ± 0,04</td>
<td>71 ± 31 332 ± 203 37 ± 11</td>
</tr>
</tbody>
</table>

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

Fertilization had only a minor influence on the N$_2$O flux in the field trial when there was no tractor traffic in well-drained soil where the N$_2$O flux rate was ≤100 μg N$_2$O-N m$^{-2}$/hr. While, the N$_2$O flux rate in the field with CF was much higher than the field trial with OF. The variation in N$_2$O emissions between the trials was large which was strongly affected by fertilization and soil moisture conditions. The maximum N$_2$O flux rate at each location varied from 90-2000 μg N$_2$O-N m$^{-2}$/hr in 2009 and 80-1000 μg N$_2$O-N m$^{-2}$/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$_2$O 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$_2$O emissions as compared with a non-fertilized cereal during the year of green manure production in 2009. Mulching of herbage induced more N$_2$O emission (+0.37 kg N$_2$O-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$_2$O 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$_2$O emissions during barley production in 2010.

In general, GM ley (mulched or harvested) increased N$_2$O 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$_2$O-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$_2$O-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\textsubscript{2}O 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\textsubscript{2}O emissions were greater when N fertilizer (100 kg N ha\textsuperscript{-1}) was applied in the fall compared to spring application. Straw removal at harvest in the fall increased N\textsubscript{2}O emissions when N fertilizer was applied in the fall, but decreased emissions when no fertilizer was applied. The study showed that N\textsubscript{2}O emissions may be minimized by applying N fertilizer in spring, retaining straw, and incorporating it in spring. The estimates of regional N\textsubscript{2}O emissions based on a fixed proportion of applied N may be tenuous since N\textsubscript{2}O 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\textsubscript{2}O emissions from ammonium nitrate (AN), urea (UR), and with urea modified nitrification inhibitors. N\textsubscript{2}O fluxes from AN plots were greater than from UR plots probably due to slower hydrolysis of Urea (CH\textsubscript{4}N\textsubscript{2}O) into NH\textsubscript{4}\textsuperscript{+} plus no NO\textsubscript{3}\textsuperscript{-}. There is a scope for reducing N\textsubscript{2}O 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\textsubscript{2}O emissions further (Dobbie and Smith, 2003).

4.2 Soil compaction

Soil compaction created by tractor traffic is favourable for N\textsubscript{2}O 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\textsubscript{3}\textsuperscript{-} for denitrification.

4.2.1 Norway

Soil compaction trials were carried out in western Norway to determine effect of soil compaction on N\textsubscript{2}O 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\textsubscript{2}O emissions. Soil compaction resulted in increased N\textsubscript{2}O emission and this compaction effect was more pronounced in NPK fertilized treatments. In unfertilized plots, N\textsubscript{2}O emission rate was increased by 44 percent due to compaction, whereas in NPK-fertilized treatment, the average N\textsubscript{2}O 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\textsubscript{2}O 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\textsubscript{2}O (Figure 4.3).
Figure 4.3 $\text{N}_2\text{O}$ 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 $\text{N}_2\text{O}$ 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 $\text{N}_2\text{O}$ 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 $\text{N}_2\text{O}$ in a soil (fine-silty Dystric Eutrochrept) planted with potato ($\text{Solanum tuberosum}$ L.). The major part (68 percent) of the total $\text{N}_2\text{O}$ 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 $\text{N}_2\text{O}$ emission from potato-cropped fields. Soil compaction by tractor traffic strongly increased $\text{N}_2\text{O}$ emissions (Ruser et al., 1998); whereas soil loosening decreased $\text{N}_2\text{O}$ 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 $\text{N}_2\text{O}$ emission than the zero compaction treatment particularly eight days after fertilization despite low rainfall. On the other hand, $\text{N}_2\text{O}$ 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$_2$O 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$_2$O 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$_2$O emissions from NT could be lower than conventional tillage (Wagner-Riddle et al. 2007). Conversely, NT increases N$_2$O 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$_2$O 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$_2$O emissions were measured periodically throughout the growing season (i.e. April to October 2010) using a NSS chamber method.

Results showed that cumulative N$_2$O emissions clearly increased with increasing fertilizer rate (0-120 kg N ha$^{-1}$ yr$^{-1}$). Although not perfectly linear, the increase in cumulative N$_2$O emissions suggested that about 1 percent of the applied N was lost as N$_2$O-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$_2$O 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 be 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 \( \text{N}_2\text{O} \) emission in the studied soil, suggesting that time of ploughing is not a relevant tool for mitigating fertilizer induced \( \text{N}_2\text{O} \) 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 \( \text{N}_2\text{O} \) 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 \( \text{N}_2\text{O} \) dynamics were characterized by three peaks: i) during autumn after seed bed preparation (12-43 µg \( \text{N}_2\text{O} \text{ m}^2\text{hr}^{-1} \)), ii) during spring after fertilization (17-52 µg \( \text{N}_2\text{O} \text{ m}^2\text{hr}^{-1} \) and iii) during spring after slurry application (10-32 µg \( \text{N}_2\text{O} \text{ m}^2\text{hr}^{-1} \)). 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 \( \text{N}_2\text{O} \) production due to elevated oxygen consumption rates by denitrifying microorganisms. Ploughing in the autumn contributes to increased emissions of \( \text{N}_2\text{O} \) (Ball et al., 1999). In contrast, the other two peaks in spring resulted from transformations of N applied in fertilizers and manure.

Overall, \( \text{N}_2\text{O} \) emissions were 27 and 26 percent lower in DD and RT, respectively, relative to \( \text{N}_2\text{O} \) emissions from CT plots (\( P<0.05 \)). Mutegi et al. (2010) observed that in residue removal scenarios \( \text{N}_2\text{O} \) emissions were similar for all tillage treatments, but in residue retention scenarios \( \text{N}_2\text{O} \) emissions were significantly higher in CT than in either DD or RT (\( P<0.05 \)). Irrespective of residue management, \( \text{N}_2\text{O} \) 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 \( \text{N}_2\text{O} \) emissions from the CT plots by 35 percent. Relative soil gas diffusivity (Rdiff), soil NO\textsubscript{3}–N, soil temperature, tillage and residue were important driving forces for \( \text{N}_2\text{O} \) emission (\( P<0.05 \)). A multiple linear regression model using Rdiff to represent the water factor explained \( \text{N}_2\text{O} \) emissions better than a WFPS based model. In this regard, there is need to review the current use of WFPS in \( \text{N}_2\text{O} \) prediction models. Mutegi et al., (2010) concluded that direct drilling has the potential to reduce \( \text{N}_2\text{O} \) 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 \( \text{N}_2\text{O} \) under no till (NT) than ploughed treatments (Figure 4.5). The greater emissions of \( \text{N}_2\text{O} \) 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 \( \text{N}_2\text{O} \) peaks (>2000 µg m\textsuperscript{-2} hr\textsuperscript{-1}) after 53, 56, and 61 days of fertilization when rainfall was low. In the same plots, \( \text{N}_2\text{O} \) emissions were small (<1000 µg m\textsuperscript{-2} hr\textsuperscript{-1}) when there was higher rainfall. Although the \( \text{N}_2\text{O} \) flux rates in the ploughed plots (i.e. 20 and 30 cm depth) were
considerably lower than NT plots, the changes in N\textsubscript{2}O emissions was similar with that of NT plots. The N\textsubscript{2}O emissions from the ploughed soil treatment (i.e. 30 cm) were the least (Figure 4.5).  

Figure 4.5 Temporal variability of N\textsubscript{2}O 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\textsubscript{2}O emissions under no-till and tilled soils. According to this report, on average soil N\textsubscript{2}O emissions under no-till were 0.06 kg N/ha\textsuperscript{-1} (lower) for good aeration, 0.12 kg N ha\textsuperscript{-1} (higher) for medium aeration and 2.0 kg N ha\textsuperscript{-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\textsubscript{2}O emissions is small in good-aerated soils but often large on soils where aeration is restricted.

4.4 Summary
Under the Norwegian conditions, N\textsubscript{2}O emissions from fertilized soils are to a great extent controlled by inter-annual variations of weather conditions and background emissions. In general, N\textsubscript{2}O 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\textsubscript{2}O 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\textsubscript{2}O 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\textsubscript{2}O emission. Other environmental factors should also be taken into account when evaluating ploughing time effects on N\textsubscript{2}O emission.
5 EFFECTS OF SOIL MOISTURE & DRAINAGE ON N\textsubscript{2}O EMISSIONS

Soil moisture is a major driver of N\textsubscript{2}O emissions as it regulates the oxygen availability to soil microbes. N\textsubscript{2}O 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\textsubscript{2} (Butterbach-Bahl et al, 2013). The N\textsubscript{2}O:N\textsubscript{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\textsubscript{2}O:N\textsubscript{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\textsubscript{2}O:N\textsubscript{2} ratio with increased WFPS will also depend on the ability of denitrifiers to produce N\textsubscript{2} or N\textsubscript{2}O as an end product (Morley et al., 2008).

Another important factor for promoting or hindering N\textsubscript{2}O 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\textsubscript{2}O 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\textsubscript{2}O emissions have been well documented in Scandinavian countries (Conen and Neftel, 2010; Klove 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\textsubscript{2}O-N ha\textsuperscript{-1} yr\textsuperscript{-1}, for grass from 1.7 to 11.0 kg N\textsubscript{2}O-N ha\textsuperscript{-1} yr\textsuperscript{-1}, for fallow from 3.8 to 37.0 kg N\textsubscript{2}O-N ha\textsuperscript{-1} yr\textsuperscript{-1} (Maljanen et al., 2007) and estimated net emissions from a pipe drained peat soils in northern Norway expressed in kg of CO\textsubscript{2} eq. m\textsuperscript{-2} yr\textsuperscript{-1}, was about 0.13 kg for N\textsubscript{2}O (Gronlund et al. 2006) and mean annual flux of 1.51 kg N\textsubscript{2}O-N ha\textsuperscript{-1} (Klove et al. 2010) which does not include the winter period.

There is however, little information available on emissions of N\textsubscript{2}O 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\textsubscript{2}O 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\textsubscript{2}O flux by gas chromatograph.
Figure 5.1 Mean N$_2$O-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$_2$O 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$_2$O 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$_2$O fluxes varied in the range of 6 ± 2.8 to 878 ± 229 µg N m$^{-2}$ h$^{-1}$. N$_2$O 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$_2$O 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$_2$O-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$_2$O fluxes from PD soil (6 kg N ha$^{-1}$) were much higher than ID (4 kg N ha$^{-1}$) and MD soil N$_2$O fluxes (0.2 kg N ha$^{-1}$) in 2011. In 2012, the cumulative N$_2$O 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$_2$O 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$_2$O 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$_2$O-N compared to moderately well drained soils mainly due to higher soil moisture and shallow ground water table.
Figure 5.2 Mean N$_2$O 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).
In western Norway, N₂O 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 N₂O emission rate varied from 15 to 1900 μg N₂O-N m⁻² h⁻¹. During those days, the water table depths ranged from 12 cm to >100 cm. The N₂O fluxes were low after one week of fertilization until the crop harvest (Figure 5.4). Peak N₂O flux (1900 μg N₂O-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 N₂O flux was observed in all the drainage points.

Figure 5.3 Cumulative N₂O-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).
Figure 5.4 N₂O fluxes, N₂O and O₃ 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 NH₄NO₃-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 N₂O emissions. In general, N₂O 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 N₂O after addition of nitrate fertilizer in Scheyern of southern Germany. Results showed that N₂O emission rates were generally small at soil moisture levels ≤60 percent WFPS with mean flux rates ranging between 1 and 12 µg N₂O-N m⁻² hr⁻¹. Significant increment of N₂O emission rates were measured at soil moisture contents ≥70 percent WFPS, with the highest N₂O 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 N₂O emissions from different soil WFPS. Nitrification was the main source of N₂O 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 N₂O 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 N₂O production by denitrification to maximum 90 percent WFPS but above which most of the N₂O is reduced to N₂ (Bateman and Baggs, 2005).

USA: Terry et al., (1981) measured field N₂O fluxes and N₂O produced in soil samples incubated under laboratory conditions in Florida (USA) to determine the effect of flooding on N₂O emissions from a drained cultivated Histosols. The N₂O 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 N₂O. However, it should be recognized that a flooded soil has a unique environment that favours reduction of N₂O to N₂ over escape of N₂O to the atmosphere. Denitrification under non-flooded conditions may well lead to substantial emissions of N₂O, because the soil microflora has less ability to further reduce N₂O to N₂. Thus, diffusion of N₂O 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 N₂O emission remained low immediately following irrigation, which was most likely the result of complete denitrification, producing mainly N₂ 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 N₂O 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 N₂O 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 N₂O 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 N₂O 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 N₂O and N₂ (Virkajärvi et al., 2010).

### 5.2.1 Other countries experience

**Costa Rica:** In tropical soils of Costa Rica, N₂O flux pulses began within 30 min, peaking no later than 8 h after rewetting and 25 g N₂O–N ha⁻¹ was emitted for three simulated rain events over a 22-day period (control emitted 14 g N₂O–N ha⁻¹) and one episodic N₂O 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 N₂O emissions, denitrification, and net N mineralization. The highest N₂O emissions (201 µg N₂O-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 N₂O-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 N₂O 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 N₂O evolved [N₂O/(N₂O+N₂)] generally increased as the soil dried. Soil rewetting to 75% WFPS generally produced higher N₂O/(N₂O+N₂) 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 N₂O 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).](image-url)
5.3 Summary

It is very difficult to draw a conclusion on the effects of moisture and drainage conditions on N\textsubscript{2}O 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\textsubscript{2}O-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\textsubscript{2}O 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\textsubscript{2} and N\textsubscript{2}O and also leaching of nutrients such as NO\textsubscript{3}\textsuperscript{-}. As a result, the nitrate concentrations increased in the soil and ground water during the dry period which could be a source for N\textsubscript{2}O production.
6  EFFECTS OF SOIL pH ON N₂O EMISSIONS

6.1  Introduction

Soil pH is another key regulator of the microbiological processes that affect N₂O and N₂ production. Soil pH has a potential effect on N₂O 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 N₂O/ (N₂O + N₂) showed a consistent decline with increasing pH. Similar results were found by Raut et al. (2012) in which N₂O/(N₂O+N₂) ratio increased with decreasing pH. However, the mechanism of pH control on the N₂O: N₂ 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 N₂O 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, N₂O and N₂ 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 N₂O production, gross nitrification, gross denitrification, CO₂ production, calculated contribution of nitrification to N₂O production (%) and N₂O product ratio (PR) of nitrification N₂O/(NO₃⁻ + NO₂⁻) during 67 h incubation in shaken soil slurries and headspace O₂ concentration at the end of incubation. All rates are averages of three replicates (±SD) except denitrification rates which were measured in a single bottle (N₂O production with 10 percent C₂H₆). Nitrification rates were calculated by the pool dilution method.

<table>
<thead>
<tr>
<th>Soil</th>
<th>pH-H₂O</th>
<th>N₂O prod a</th>
<th>Denitrification</th>
<th>Nitrification</th>
<th>CO₂ prod b</th>
<th>N₂O (%)</th>
<th>N₂O/(NO₃⁻ + NO₂⁻) PR c</th>
<th>Headspace O₂ d (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>F2</td>
<td>4.2</td>
<td>200 ± 34</td>
<td>695</td>
<td>9.8 ± 1.2</td>
<td>14.2 ± 0.3</td>
<td>68.1 ± 0.7</td>
<td>14.1 ± 1.49</td>
<td>20.5 ± 0.01</td>
</tr>
<tr>
<td>F3</td>
<td>4.1</td>
<td>1100 ± 37</td>
<td>733</td>
<td>9.6 ± 2.3</td>
<td>10.6 ± 0.2</td>
<td>62.9 ± 0.59</td>
<td>76.2 ± 21.1</td>
<td>20.6 ± 0.01</td>
</tr>
<tr>
<td>F4</td>
<td>6.7</td>
<td>31 ± 1.4</td>
<td>8.1</td>
<td>170 ± 8.5</td>
<td>27.7 ± 0.4</td>
<td>95.7 ± 0.08</td>
<td>0.18 ± 0.01</td>
<td>20.0 ± 0.01</td>
</tr>
<tr>
<td>F5</td>
<td>6.2</td>
<td>25 ± 0.4</td>
<td>7.6</td>
<td>80 ± 10.4</td>
<td>27.2 ± 1.6</td>
<td>89.8 ± 0.05</td>
<td>0.28 ± 0.04</td>
<td>20.0 ± 0.06</td>
</tr>
<tr>
<td>F9</td>
<td>7.8</td>
<td>15 ± 1.1</td>
<td>4.6</td>
<td>61 ± 6.9</td>
<td>26.5 ± 1.0</td>
<td>94.0 ± 1.53</td>
<td>0.24 ± 0.05</td>
<td>19.9 ± 0.03</td>
</tr>
<tr>
<td>Å2</td>
<td>5.0</td>
<td>2.3 ± 0.2</td>
<td>0.90</td>
<td>6.3 ± 0.5</td>
<td>19.2 ± 0.5</td>
<td>90.8 ± 0.60</td>
<td>0.33 ± 0.02</td>
<td>20.3 ± 0.02</td>
</tr>
<tr>
<td>Å5</td>
<td>5.5</td>
<td>2.4 ± 0.4</td>
<td>0.35</td>
<td>5.6 ± 0.4</td>
<td>19.9 ± 1.1</td>
<td>91.4 ± 1.95</td>
<td>0.39ab ± 0.04</td>
<td>20.2 ± 0.04</td>
</tr>
<tr>
<td>Å7</td>
<td>6.1</td>
<td>2.9 ± 0.3</td>
<td>0.28</td>
<td>5.6 ± 0.6</td>
<td>19.5 ± 2.1</td>
<td>92.5 ± 0.20</td>
<td>0.49b ± 0.07</td>
<td>20.3 ± 0.06</td>
</tr>
</tbody>
</table>

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 CO₂ production assuming a 1:1 stoichiometry between CO₂ production and O₂ 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 N₂O 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 N₂O product ratio with increasing soil pH was found, but the effect was so weak that it justifies a constant N₂O product ratio of nitrification for N₂O emission models. Whereas for the soils with pH 4.1 and 4.2, the apparent N₂O 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 NO₂⁻ (Mørkved, et al., 2007). The study underlines the role of NO₂⁻, both for regulating denitrification and for the apparent nitrifier derived N₂O emission.

A laboratory study was carried out by Hovlandsdal (2011) to test the enzyme that reduces N₂O to N₂ during denitrification (N₂O)-reductase, whether it has an effect on in situ N₂O 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 N₂O 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 N₂O and N₂ emissions, denitrification activity, and size of a denitrifier community by Čuhel et al. (2010). They found that manipulation of soil pH affected the N₂O/(N₂O + N₂) ratio, which increased with decreasing pH due to changes in the total denitrification activity but not in N₂O production. Soil pH is of importance in determining the nature of denitrification end products. Thus, the N₂O/(N₂O + N₂) ratio increased with decreasing pH due to changes in the total denitrification activity (Figure 6.1), while no changes in N₂O production were observed (Čuhel et al., 2010).

![Figure 6.1](image-url) In situ cumulative losses of N (separately as N₂O and N₂) (A) and relative N₂O production expressed as the N₂O/(N₂O + N₂) molar ratio (B) with different pH treatments over the 74 h after the addition of 15N-labeled KNO₃. 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 Čuhel 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 N₂O and N₂. Stevens et al. (1998) reported that the flux of N₂ increased with pH but the effect of pH on the flux of N₂O was inconsistent. The maximum flux of N₂O occurred at pH 6.5 and the minimum fluxes at pH 6 and 8. The N₂O-reductase enzyme was still being inhibited or N₂O was being produced by a process other than denitrification. Partial inhibition of N₂O-reductase at pH 8 may be due to NO₃⁻ accumulation.

6.2 Summary

It is not possible to generalize the effect of soil pH on N₂O 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 N₂O product ratio of nitrification and denitrification by taking soil samples from long term liming field trials. According to these trials result, the product ratio of denitrification N₂O/(N₂O+N₂) showed a consistent decline with increasing pH and vice versa which is in line with other countries result (Šimek and Cooper, 2002; Čuhel et al. 2010). While the product ratio of nitrification N₂O/(NO₂⁻+ NO₃⁻) 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 NO₂⁻. 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 N₂O production.
7 FREEZING-THAWING EFFECTS ON N₂O 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 (N₂O/N₂) decreases with increasing temperature, while that from nitrification (N₂O/NO₃⁻) tends to rise. The combined effect is that N₂O emission rates increase with temperature (Granli and Bøckman, 1994). In temperate regions, a majority of N₂O 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 N₂O emissions from agricultural soils. However, the mechanisms that control N₂O 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 (O₂) conditions, using nitrogen (N) tracers (¹⁵N) to determine process rates and sources of emitted N₂O after freeze–thaw treatment of soil or after addition of freeze–thaw extract from clover. The N₂O emission response was inversely related to O₂ 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 N₂O reductase activity (Mørkved et al., 2006).

Table 7.1 Average N₂O production rates (±SD; n = 3) with and without acetylene (C₂H₂), and product ratios of denitrification (N₂O/(N₂+ N₂O)) in response to freeze–thaw and plant extract addition.

<table>
<thead>
<tr>
<th>Soil</th>
<th>Treatment</th>
<th>Temp °C</th>
<th>No C₂H₂ ng N₂O-N g⁻¹ d⁻¹</th>
<th>With C₂H₂ ng N₂O-N g⁻¹ d⁻¹</th>
<th>N₂O/ (N₂ + N₂O)</th>
<th>R²</th>
</tr>
</thead>
<tbody>
<tr>
<td>Soil 1  (pH 4.5)</td>
<td>Water</td>
<td>5</td>
<td>1502 ± 166</td>
<td>2040 ± 152</td>
<td>0.74 ± 0.10</td>
<td>0.89–0.91</td>
</tr>
<tr>
<td></td>
<td>Water, freeze–thaw</td>
<td>5</td>
<td>4368 ± 226</td>
<td>9379 ± 526</td>
<td>0.69 ± 0.07</td>
<td>1.00</td>
</tr>
<tr>
<td></td>
<td>Plant extract</td>
<td>5</td>
<td>8301 ± 525</td>
<td>9607 ± 172</td>
<td>0.86 ± 0.06</td>
<td>0.99–1.00</td>
</tr>
<tr>
<td></td>
<td>Plant extract, freeze–thaw</td>
<td>5</td>
<td>7770 ± 244</td>
<td>7831 ± 514</td>
<td>0.87 ± 0.07</td>
<td>1.00</td>
</tr>
<tr>
<td></td>
<td>Water</td>
<td>20</td>
<td>7157 ± 136</td>
<td>7212 ± 278</td>
<td>0.99 ± 0.04</td>
<td>0.98–1.00</td>
</tr>
<tr>
<td>Soil 2  (pH 7.0)</td>
<td>Water</td>
<td>5</td>
<td>1021 ± 160</td>
<td>1938 ± 212</td>
<td>0.53 ± 0.10</td>
<td>0.99–1.00</td>
</tr>
<tr>
<td></td>
<td>Water, freeze–thaw</td>
<td>5</td>
<td>2802 ± 55</td>
<td>7365 ± 93</td>
<td>0.38 ± 0.01</td>
<td>0.99–1.00</td>
</tr>
<tr>
<td></td>
<td>Plant extract</td>
<td>5</td>
<td>5621 ± 83</td>
<td>10224 ± 90</td>
<td>0.55 ± 0.01</td>
<td>0.99–1.00</td>
</tr>
<tr>
<td></td>
<td>Plant extract, freeze–thaw</td>
<td>5</td>
<td>4131 ± 6</td>
<td>8274 ± 304</td>
<td>0.50 ± 0.02</td>
<td>0.99–1.00</td>
</tr>
</tbody>
</table>

All ratios are based on linear regression of N₂O accumulation versus time under anaerobic conditions. The minimum and maximum R² of the linear regression with time for the six N₂O production rate estimates (± C₂H₂) within each treatment are given. Source: Mørkved et al. (2006).

Nitrification rates estimated by nitrate (NO₃⁻) pool enrichment were 1.5 to 1.8 mg NO₃⁻-N per g dw (dry weight) soil per day in freeze–thaw-treated soil when incubated at O₂ concentrations above 2.3 vol percent and one order of magnitude lower at 0.8 vol percent O₂. Thus, the experiments captured a situation with severely O₂-limited nitrification. As expected, the O₂ 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 N₂O accumulation originated from nitrification, reaffirming that denitrification was the main N₂O source at the various tested O₂ concentrations in freeze–thaw-affected soils (Mørkved et al., 2006).

N₂O 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 N₂O emissions in our soils, both by fuelling denitrifiers and by depleting O₂. The soluble C (applied as plant extract) necessary to induce a CO₂ and N₂O 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 N₂O 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 N₂O 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 N₂O emissions in Canada.
<table>
<thead>
<tr>
<th>Research objectives</th>
<th>Results and conclusions</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>• To examine the effects of temperatures (near freezing point) on $\text{N}_2\text{O}$ emissions, denitrification, and on the abundance and structure of soil nitrifiers and denitrifiers communities.</td>
<td>• 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 $\text{N}_2\text{O}$ emissions at $-1 , ^\circ\text{C}$ when NO$_3^-$ and C are present.</td>
<td>Wertz et al. (2012)</td>
</tr>
<tr>
<td>• To identify the source of $\text{N}_2\text{O}$ and emission during freezing and thawing cycles.</td>
<td>• Soil atmospheric $\text{N}_2\text{O}$ concentrations at 7.5 to 45 cm soil depths, increased during the winter of 1995-1996 when air temperatures increased above freezing. • $\text{N}_2\text{O}$ flux from soil to atmosphere during the thawing events was highly correlated with $\text{N}_2\text{O}$ concentration in the soil atmosphere near the surface. • Biological processes contribute to significant amount of $\text{N}_2\text{O}$ emissions during thaw events, even when temperatures are at or below freezing point.</td>
<td>Chang and Hao (2001)</td>
</tr>
<tr>
<td>• To quantify $\text{N}_2\text{O}$ emissions from agricultural fields during winter and spring thaw</td>
<td>• $\text{N}_2\text{O}$ 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 $\text{N}_2\text{O}$ 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 $\text{N}_2\text{O}$ emissions through use of over-wintering cover crops.</td>
<td>Wagner-Riddle and Thurtell (1998)</td>
</tr>
<tr>
<td>• To examine $\text{N}_2\text{O}$ emission from agricultural soil subjected to different freeze-thaw cycles</td>
<td>• The mean $\text{N}_2\text{O}$ fluxes ranged from 0.24 to 0.65 kg N ha$^{-1}$ day$^{-1}$ for non-frozen soil cores. One explanation for increased $\text{N}_2\text{O}$ 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.</td>
<td>Chen et al. (1995)</td>
</tr>
</tbody>
</table>

ii) Finland
The results of four selected studies (i.e. 3 laboratory and 1 field) on freezing and thawing effects on $\text{N}_2\text{O}$ emissions in Finland are shown in Table 7.3.
Table 7.3 Research results on the effect of freezing and thawing on N₂O emissions in Finland.

<table>
<thead>
<tr>
<th>Research objectives</th>
<th>Results and conclusions</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>• To examine the effect of snow cover and no snow cover on N₂O production and emissions.</td>
<td>• Removal of snow lowered soil temperatures and enhanced N₂O 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) N₂O-N m⁻² y⁻¹ in peat soils. • Laboratory incubations of the soil samples showed high production rates of N₂O at temperatures &lt;0°C, especially in sandy and peat soils.</td>
<td>Maljänen et al. (2009)</td>
</tr>
<tr>
<td>• To explore the temperature responses of N₂O emissions using laboratory microcosm during soil freezing and thawing periods</td>
<td>• At −2.2°C, abandoned soil showed an increase in N₂O emissions (58.7 ± 14.5 µg N₂O-N m⁻² h⁻¹). While when the temperature was further decreased to −4.9°C, N₂O emissions decreased in abandoned soil as well as in the afforested soils. N₂O emissions increased immediately after the soil thawing started in both sites. The emissions from the abandoned soil were highest (238 ± 47.2 µg N₂O-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 N₂O-N m⁻² h⁻¹ after 5 days of thawing. • Soil freeze-thaw cycles enhance the N₂O emissions in abandoned soil as well as in the afforested soils in contrast to the NO emission.</td>
<td>Koponen et al. (2006)</td>
</tr>
<tr>
<td>• To examine the N₂O production at low temperatures with and without freezing-thawing events</td>
<td>• When the temperature of the frozen soil cores was increased stepwise from −8°C, the N₂O 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 N₂O 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 N₂O emission near 0°C when soil temperature decreases. These emissions, however, are less than those emitted from thawing soils. The results suggest that N₂O is produced in soils down to a temperature of −6 °C.</td>
<td>Koponen et al. (2004)</td>
</tr>
<tr>
<td>• To investigate the effects of freezing-thawing and drying-wetting on the emission of N₂O</td>
<td>• The total surplus N₂O emissions due to the first wetting event ranged between 3 and 140 mg N-N₂O 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 N₂O 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 N₂O emission rates perhaps because of a prolonged drought during the time of soil core collection that had changed soil properties considerably.</td>
<td>Priemé and Christensen (2001)</td>
</tr>
</tbody>
</table>
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\textsubscript{2}O emissions in Germany.

<table>
<thead>
<tr>
<th>Research objectives</th>
<th>Results and conclusions</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>• To examine the processes of N\textsubscript{2}O emissions during freezing/thawing periods under laboratory condition.</td>
<td>• Two periods of higher N\textsubscript{2}O emissions were detected, a period of elevated N\textsubscript{2}O emissions during continuous soil freezing and a period of brief peak emissions during thawing. • N\textsubscript{2}O 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\textsubscript{2}O. • Peak emissions during soil thawing were explained by the physical release of trapped N\textsubscript{2}O and/or denitrification during thawing.</td>
<td>Teepe et al. (2001)</td>
</tr>
<tr>
<td>• To investigate the effects of freezing-thawing and drying-wetting on the emission of N\textsubscript{2}O</td>
<td>• Following thawing, denitrification was responsible for &lt;40 percent of N\textsubscript{2}O emission from the German grassland soils.</td>
<td>Priemé and Christensen (2001)</td>
</tr>
<tr>
<td>• To examine the N\textsubscript{2}O emissions from soils during thawing and freezing cycles in winter</td>
<td>• Small N\textsubscript{2}O emission rates were measured, when soil freezing started in December. However, N\textsubscript{2}O emissions increased with soil warming (up to 0 °C) in January. In mid January the N\textsubscript{2}O emissions decreased due to soil freezing (&lt;-2 °C). High N\textsubscript{2}O emissions from soil were also found during a second thawing period in February. • N\textsubscript{2}O is produced and emitted even in frozen soil (soil temperature -4 °C). Therefore winter emissions are likely important in global N\textsubscript{2}O emission budgets.</td>
<td>Röver et al. (1998)</td>
</tr>
</tbody>
</table>

iv) Japan

Yanai et al. (2011) studied in situ soil gas N\textsubscript{2}O and oxygen (O\textsubscript{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\textsubscript{2}O 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\textsubscript{2}O fluxes, ranging from 0.10 to 1.95 kg N\textsubscript{2}O-N ha\textsuperscript{-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\textsubscript{2}O fluxes represented 58 to 85 percent of the annual values. Soil N\textsubscript{2}O 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\textsubscript{2}O concentrations at the depth of 10 cm peaked at 46 ppm when soil gas O\textsubscript{2} concentrations fell down to 0.12 m\textsuperscript{3} m\textsuperscript{-3} under soil temperature below 0.0 °C. Snowmelt acceleration had no stimulating effect on N\textsubscript{2}O 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 N₂O production in a one-year plot-scale (snow cover manipulation) experiment conducted in the northern Japan. This is because a decreasing soil gas O₂ concentration in the shallow layer clearly corresponded with an increasing soil gas N₂O concentration, denitrification in the thawing upper layer, which was likely the major process responsible for N₂O 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, N₂O fluxes might not be negligible, as once thought. To evaluate the annual N₂O fluxes, field monitoring of N₂O 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 N₂O emissions in USA are shown in Table 7.5.

### Table 7.5 Research results on the effect of freezing and thawing on N₂O emissions in USA.

<table>
<thead>
<tr>
<th>Research objectives</th>
<th>Results and conclusions</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>• To test several soil conditions affecting the emission of N₂O during thawing of soil</td>
<td>• 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 N₂O:N₂ ratio was higher for frozen (0.17) than cold (0.07) treated soil, respectively, without the addition of NO₃⁻. • The addition of NO₃⁻ increased the N₂O:N₂ ratio being 2.45 and 0.53 for frozen and cold-treated soil.</td>
<td>Tenuta and Sparling (2011)</td>
</tr>
<tr>
<td>• To investigate the effect of the temperature and moisture on the emission of N₂O from arable soils.</td>
<td>• 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 N₂O reached 2.35 percent because of the active denitrification. • The extra N₂O flux initiated by the freeze–thaw processes made up 88 to 98 percent of the total N₂O flux during the entire experiment.</td>
<td>Kurganova and de Gerenyu (2010)</td>
</tr>
<tr>
<td>• To understand the effects of tillage, moisture content and manure application on N₂O emissions from agricultural soils at low temperatures.</td>
<td>• The maximal emission of 200 µg N m⁻² h⁻¹ was found at soil temperatures &gt;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 N₂O 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.</td>
<td>Singurindy et al. (2009)</td>
</tr>
<tr>
<td>• To quantify nitrous oxide fluxes during soil thaw in early spring.</td>
<td>• At thawing, high N₂O 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 N₂O concentrations and N₂O flux declined. N₂O was released on warming, and cores treated with 2 mL of chloroform (CHCl₃), had a slower release rate.</td>
<td>Goodroad and Keeney (1984)</td>
</tr>
</tbody>
</table>
7.2 Summary
There is very limited knowledge on the processes involved in \( \text{N}_2\text{O} \) 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 \( \text{N}_2\text{O} \) emission. However, the \( \text{N}_2\text{O} \) 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, \( \text{N}_2\text{O} \) emission induced by freezing and thawing of soils during winter and early spring should be quantified and accounted in the national \( \text{N}_2\text{O} \) budgets and annual GHGs emissions.
8 OPTIONS FOR MITIGATING N₂O 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 NO₃⁻ to N₂ which otherwise would have been leached from soils and discharges into rivers or lakes, may be an environmental benefit because the evolved N₂ can be trapped later by legumes fixing bacteria and enrich the soil with nitrogen. If, however, denitrification results in emitting N₂O, this poses an environmental threat since it increases global warming and induce climate change. Management of agricultural practices should therefore minimise the accumulation of NO₃⁻ in soil and thereby restrict both leaching and N₂O emission. However, where denitrification is inevitable, or even desirable to protect water quality, the management should ensure that as much of the denitrified NO₃⁻ is emitted as N₂ not as N₂O. There are a range of on-farm mitigation options to control denitrification by either emitting less N₂O or shifting the balance between harmful N₂O and the non-greenhouse gas namely N₂ from agricultural soils. Some of the mitigation practices of N₂O 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 N₂O. In other words, by undertaking soil N balance in the given field, the emission of soil N₂O 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, N₂O 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 N₂O 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$_2$O 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$_2$O emissions from soils.

<table>
<thead>
<tr>
<th>Best Management Practices</th>
<th>References</th>
</tr>
</thead>
<tbody>
<tr>
<td>• Timely application of N fertilizer to correspond to crop’s ability to make the best use of the N</td>
<td>Aguilera et al. (2013)</td>
</tr>
<tr>
<td>• Uniform application of N-fertilizers and manure and avoiding application on frozen or water logged soils</td>
<td>Granli and Bøckman (1994)</td>
</tr>
<tr>
<td>• Split fertilizer application particularly for urea to reduce NH$_3$ volatilization and subsequent N$_2$O emissions</td>
<td>de Klein et al. (2001)</td>
</tr>
<tr>
<td>• Injection of slurry or rapid incorporation of manure into the soils by tillage</td>
<td>Vander Zaag et al. (2011)</td>
</tr>
<tr>
<td>• Constructing storage facilities (for animal wastes and feed lots) with proper aeration and moisture management</td>
<td>de Klein et al. (2001)</td>
</tr>
<tr>
<td>• Reducing unintentional N transfer from agriculture to other ecosystems</td>
<td>de Klein et al. (2001)</td>
</tr>
<tr>
<td>• Keeping the soil NO$_3^-$ concentrations low during off season</td>
<td>de Klein et al. (2001)</td>
</tr>
<tr>
<td>• Avoiding fallow fields by growing autumn sown or catch crops and ploughing immediately and planting new crops</td>
<td>de Klein et al. (2001); Sangeetha et al. (2009)</td>
</tr>
<tr>
<td>• Minimum tillage in permanent grasslands to cover the field with grass at the end of the growing season</td>
<td>Petersen, et al. (2011); D’Haene et al. (2008)</td>
</tr>
<tr>
<td>• Reducing fertiliser use or grazing intensity in the autumn in grassland soils</td>
<td>IPNI (2012)</td>
</tr>
<tr>
<td>• Avoiding soil compaction and erosion in the agricultural fields</td>
<td>Yamulki and Jarvis (2002); Sitaula et al. (2000)</td>
</tr>
</tbody>
</table>

The most relevant management options to reduce N$_2$O 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$_2$O 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$_2$O 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$_2$O 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$_2$O emissions and yield is generally higher when inhibitors are applied with N fertilizer (Missonelbrook 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\textsubscript{2}O 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\textsubscript{2}O 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\textsubscript{2}O emissions with conditions favouring the complete transformation of NO\textsubscript{3}\textsuperscript{-} to N\textsubscript{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\textsubscript{2}O 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\textsubscript{3}\textsuperscript{-} to the downstream where N\textsubscript{2}O 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 percent), this may actually increase N\textsubscript{2}O emissions (Granli and Bøckman, 1994) and could lead to increased NO\textsubscript{3}\textsuperscript{-} 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\textsubscript{2}O emission has generated mixed results. Some studies have reported higher N\textsubscript{2}O 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\textsubscript{2}O emissions in
Norway. Further investigation is required to suggest which tillage practices are desirable under the Norwegian conditions to keep N\textsubscript{2}O 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\textsubscript{2}O 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\textsubscript{2}O-N ha\textsuperscript{-1} which was less than the control (i.e. 9 kg N\textsubscript{2}O-N ha\textsuperscript{-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\textsubscript{2}O emissions. Other researchers for e.g. Wang et al. (2012) reported that biochar application decreased N\textsubscript{2}O 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\textsubscript{2}O soil emissions from agricultural soils (Wang et al. 2011; Alho et al. 2012). However, considering N\textsubscript{2}O 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\textsubscript{2}O emissions still have to be further examined (Alho et al. 2012).

Table 8.2 Effects of Biochar applications on cumulative N\textsubscript{2}O-N emissions as reported by some authors.

<table>
<thead>
<tr>
<th>References</th>
<th>Cumulative N\textsubscript{2}O-N</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>With biochar</td>
</tr>
<tr>
<td>Case et al. (2015)</td>
<td>0.05 ± 0.02 mg N\textsubscript{2}O-N kg\textsuperscript{-1}</td>
</tr>
<tr>
<td>Verhoeven and Six (2014)</td>
<td>4.14 ± 1.14 to 4.24 ± 0.74 kg N\textsubscript{2}O-N ha\textsuperscript{-1} yr\textsuperscript{-1}</td>
</tr>
<tr>
<td>Angst et al. (2013)</td>
<td>0.04 to 0.08 mg N\textsubscript{2}O-N kg\textsuperscript{-1}</td>
</tr>
<tr>
<td>Suddick and Six (2013)</td>
<td>0.10 ± 0.05 kg N\textsubscript{2}O-N ha\textsuperscript{-1} day\textsuperscript{-1}</td>
</tr>
<tr>
<td>Case et al. (2012)</td>
<td>0.01 N\textsubscript{2}O-N µg g dry soil\textsuperscript{-1}</td>
</tr>
</tbody>
</table>

While there is clear evidence that, in many cases, emissions of N\textsubscript{2}O 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\textsubscript{2}O 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\textsubscript{2}O emissions from soil with biochar application. A recent study made by García et al., (2014) revealed that the addition of biochar increased N\textsubscript{2}O emissions by 54 percent in Haplic Calcisols. Biochar systematically stimulated nitrification, which was probably the cause of the increased N\textsubscript{2}O emissions. Predicting which N\textsubscript{2}O formation pathway pre-dominates in a
certain kind of soil will be necessary to guarantee the success of biochar as a N$_2$O 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$_2$O 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$_2$O flux in paddy field. Analysis revealed that digested liquid treated soils significantly decrease N$_2$O 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$_2$O 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$_2^-$ and NO$_3^-$ to N$_2$O (Zwieten et al. 2009). Thus, the presence of C may contribute in reducing N$_2$O emission by digested liquid application.

8.8 Soil liming

Liming may be an option for the mitigation of N$_2$O from soils (Stevens et al. 1998; van der Weereden et al. 1999) by reducing N$_2$O to N$_2$. Soil liming enhances nitrification and the cumulative N$_2$O 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$_2$O 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$_2$O 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$_2$O 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$_2$O to N$_2$ during denitrification (N$_2$O)-reductase, whether it has an effect on in situ N$_2$O 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$_2$O emissions from cultivated organic soils.

However, the above mentioned studies are contrary to the common view that increasing soil pH will increase soil N$_2$O 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$_2$O emissions. This study has concluded that liming did not make any significant change to the average N$_2$O 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$_2$O mitigation practices in Norway. Hence, information are scanty regarding which mitigation practices are best to apply to reduce N$_2$O emissions across the different Norwegian agricultural systems.
9 CONCLUSIONS

This report has highlighted the importance of nitrous oxide (N\textsubscript{2}O), which is one of potent GHGs emitted predominantly from agricultural soils. We have described the main factors that promote the process of denitrification and nitrification of N\textsubscript{2}O formation and the techniques and models used to predict N\textsubscript{2}O emissions. Around 225 published literatures spanning from 1974 to 2015 were reviewed and selected papers were discussed on their effects of N\textsubscript{2}O 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\textsubscript{2}O production and emissions. Moreover, a range of alternative measures that can reduce N\textsubscript{2}O 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\textsubscript{2}O 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\textsubscript{2}O 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 N₂O emissions from Norwegian agricultural soils.

<table>
<thead>
<tr>
<th>Research needs</th>
<th>Rationale</th>
</tr>
</thead>
<tbody>
<tr>
<td>• There is a need to revisit the IPCC emission factor used to estimate national N₂O emissions</td>
<td>• Increased use of N fertilizers will continue to produce sufficient food and sustain food security.</td>
</tr>
<tr>
<td>• The agronomic, economic and environmental consequences of reducing N fertilization as a mitigation measure</td>
<td>• Reducing mineral N input can often lead to proportional decrease in crop yields and shortage of food supply in the short term.</td>
</tr>
<tr>
<td>• Risk of high N₂O emissions from cultivate organic and/or mineral soils during the winter period</td>
<td>• When weeds are killed by herbicides during early autumn followed by excessive manure application and ploughing in late autumn, is a potential N₂O bomb which should be avoided.</td>
</tr>
<tr>
<td>• Manure and urine management to reduce direct N₂O emissions and secondary emissions from NH₃ volatilization</td>
<td>• 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.</td>
</tr>
<tr>
<td>• Indirect emissions of N₂O and/or enhanced NO₃⁻ leaching from drainage waters and streams in agricultural catchments</td>
<td>• Very few data are available on indirect emissions of N₂O from nitrate leaching and run-off in agricultural fields.</td>
</tr>
<tr>
<td>• Effect of freezing-thawing cycles on emissions of N₂O during winter and early spring</td>
<td>• Lack of knowledge on the mechanisms controlling N₂O emissions during the freezing-thawing periods and their impacts on annual N budgets.</td>
</tr>
<tr>
<td>• Effect of soil pH on N₂O emissions via denitrification</td>
<td>• The mechanism of pH control in the N₂O/N₂ ratio of denitrification is not fully understood.</td>
</tr>
<tr>
<td>• Modelling the different driving factors leading to changes in N₂O/N₂ ratio</td>
<td>• Limited knowledge and data on modelling N₂O emissions.</td>
</tr>
<tr>
<td>• How does the released N from organic materials affect emissions of N₂O under cereal cropping system?</td>
<td>• Limited knowledge and data on emission of N₂O from organic fertilizers in cereal fields.</td>
</tr>
<tr>
<td>• Effects of biochar, biogas slurry and/or liming on N₂O emissions and associated soil improvements</td>
<td>• There are conflicting results on the effects of biochar, biogas slurry and/or liming as an alternative option for mitigation of N₂O soil emissions.</td>
</tr>
</tbody>
</table>

To this end:

- There is a need to increase our knowledge and understandings of the N₂O 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 N₂O 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.
10 REFERENCES

Aakra, Å. and Marina Azzaroli, B. (1997). N₂O Emission from Norwegian Agriculture as Estimated by the IPCC Methodology, Agricultural University of Norway.


Brunme and Beece (1992)


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.

Forsidefoto: [Sett inn fotografens navn/eventuell fjernes denne teksten]

nibio.no