

The impact artificial sub-surface drainage on greenhouse gas emissions, change in soil carbon storage and nutrient losses from grassland

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By: N ria Valbuena-Parralejo

Supervised by:

Dr. Michael Williams (Trinity College Dublin)

Dr. James Humphreys (Teagasc)

Dr. Gary Lanigan (Teagasc)

Dr. Owen Fenton (Teagasc)

*Department of Botany, School of Natural Sciences, Trinity College Dublin, College Green,
Dublin 2, Ireland*

DECLARATION

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Signed:

Núria Valbuena-Parralejo

August 2018

PREFIX

All field measurements were carried out by the author from October 2013 to the end of the data collection phase of this study in August 2015. Data collected before October 2010 was supplied by Teagasc and the relevant studies are referenced where appropriate.

One of the mole and gravel mole drainage system used in this study was already designed and established prior to the commencement of this study and was managed by Teagasc personnel throughout the duration of the study. This included the management of the pastures and livestock on the system.

ABSTRACT

Irish dairy production sector has started an expansion process while achieving a 20% reduction in the National greenhouse gas (GHG) emissions and fulfilling the Water Framework Directive. Moreover, the 30% of Irish dairy farms are located on poorly permeable soils. Mole and gravel mole drainage are widely implemented as a management technique in order to improve soil permeability, maximize grass utilization and ultimately increase milk production on dairy farms. Hence, assessing the environmental impact of such systems becomes a requirement to reach the above targets. This study carried out detailed soil GHG flux and soil phosphorus (P) and nitrogen (N) losses to water measurements on a pasture-based livestock production system. This study also allowed for a much more holistic view of soil N and carbon cycle after subsurface artificial drainage implementation compared to most previous studies which have primarily focused on soil nutrients losses to water. Encompassed in this holistic study was a study of the impact of weather variability on the soil GHG emissions. The experiments were carried out on a perennial ryegrass (*Lolium perenne* L.)/white clover (*Trifolium repens* L.; 18.2%) based at Solohead Research Farm (52° 51' N, 08° 21' W), on a gently sloping (1.4%) site (2.5 ha) bounded by the river Pope. Four treatments (plot size: 100 x 15 m) were arranged in a randomised complete block design replicated four times were: (i) un-drained, (ii) mole drainage 1, (iii) mole drainage 2 and (iv) gravel mole drainage. Representative grab water samples associated with flow measurements were taken, filtered and analysed for P and N (from 31 March 2014 until 31 March 2015). Soil nitrous oxide (N₂O), soil root respiration (CO₂) and soil methane (CH₄) fluxes were measured in two experiments conducted over a 36 month period. These measurements were combined with measurements of soil watertable depth (WTD), soil water filled pore space (WFPS), soil temperature, soil mineral nitrogen (N), herbage N uptake, soil organic carbon (SOC) and soil total N (TN), that were measured on a weekly and seasonal basis. The data collected over the sampling period was used to calibrate and validate the DNDC ecosystem process model in order to assess the effect of each drainage treatment on soil GHG. Historical weather (1986-2013) was used to generate 27 weather scenarios. The effects of weather variability on the soil GHG emissions were analysed and also key drivers of soil GHG fluxes were identified.

Results showed drainage treatments ($P < 0.05$) deepened the WTD and decreased WFPS. However no ($P > 0.05$) impact of drainage treatment was detected on the soil GHG flux, soil mineral N, herbage N uptake, SOC and TN. Drainage increased the extent of dissolved inorganic N losses in the subsurface pathways. However, total P losses were lowered reduced by on avenged (114 g ha⁻¹ year⁻¹) by improving soil permeability and enhancing soil P sorption.

Modelling results showed that gravel mole drainage can significantly lower the annual cumulative soil N₂O emissions and that increased annual rainfall and mean air temperature can significantly increase soil cumulative N₂O emissions under drained and un-drained soils. The clay-loamy soils with high soil organic carbon and moist maritime climate at this site contained conditions that enhanced the sorption of the P dissolved in the water percolating through the soil drainage system. Moreover, it also contained conditions for denitrification, which may explain the lack of differences between treatments in the cumulative soil N₂O flux between drained and undrained treatment and relatively low N losses to ground water in the present study. Although modelling results showed that drainage treatment can significantly lower the overall soil N₂O emissions, the accounted difference was $0.3 \pm 0.07 \text{ kg N}_2\text{O-N ha}^{-1} \text{ year}^{-1}$. Further analysis also showed that years with greater rainfall and mean air temperature increased the soil N₂O-N emissions, with no significant difference between soil cumulative N₂O emissions from drained and undrained treatments. This study highlights that mole and gravel mole drainage techniques for improving trafficability and grass utilization did not have a major impact the eutrophication of water bodies and on the extent of soil GHG emissions under the management and conditions described in the present study. It also shows the complexity underlying the soil GHG emissions and particularly soil N₂O flux. And the strong influence of weather variables on the soil N and carbon cycles. As weather cannot be controlled the emphasis should be made in developing management techniques for the control of the key parameters that drive soil GHG emissions and avoid favourable conditions for soil nutrient losses and soil GHG emissions.

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CHAPTER 1: GENERAL INTRODUCTION

1.1 Introduction

The aim of this project was to assess the impact on the environment of land drainage in terms of greenhouse gas emissions, soil carbon change and nutrient losses to water.

Since the phasing out of the European Union milk quota there has been a 45% increase in milk production in Ireland in 2017 compared with the 10-year average of 2001 to 2010. In Ireland, many dairy farms are located in areas with high rainfall and low evapotranspiration and consequently to enable the above expansion, land that is wet and marginal is being brought into more intensive production.

Agricultural productivity can benefit from drainage, achieving a reduction of soil compaction, improving air exchange between soil and atmosphere, improving rooting and enhancing water uptake and use (Franklin et al., 2007b). Nevertheless, drainage is increasingly perceived as a major contributor to detrimental environmental impacts such as nutrient losses from the soil to water that can lead to eutrophication (Skaggs et al., 1994). Agricultural land covers about 37% Earth land surface and produces great amounts (12%) of the total greenhouse gas (GHG) emissions to the atmosphere (Linguist et al. 2012), which include carbon dioxide (CO₂), nitrous oxide (N₂O) and methane (CH₄).

Previous studies have shown that N₂O emissions from the soil can be affected by soil aeration, water filled pore space (WFPS), soil temperature (Ruser et al. 2006), soil compaction (Bhandral et al. 2007) soil carbon and soil nitrification rates (Hefting et al. 2003) and soil electrical conductivity and pH (Baggs et al., 2010). Other studies have shown that soil respiration rates (CO₂ fluxes) can be affected by modification of soil water status (Nangia et al. 2013) and soil temperature (Jabro et al. 2008). Often models of soil respiration rely mainly on two variables; soil moisture and soil temperature (Schlentner and Cleve 1985, Lloyd and Taylor 1994). Necpálová et al. (2013) found that changing of soil physical characteristics of permanent grassland by grassland renovation reduced the quantity of carbon stored in soil due to oxidation of the soil organic carbon. While Ibrahim et al. (2013) showed that an improvement of soil permeability can change pathway of nutrient (nitrogen and phosphorus) losses to water. However, there have been few studies investigating the environmental impact of artificial subsurface drainage of intensively managed permanent grassland. Therefore there are three hypotheses underlying this study: that the implementation of artificial subsurface drainage systems will change the extent of (i) GHG emissions, (ii) soil carbon stocks and (iii) nutrient losses to water.

This work involves the measurement of soil GHG emissions, soil total carbon and nitrogen and phosphorus losses to water, over a two year period in an effort to answer the following questions: ‘Will subsurface drainage (mole and gravel mole drainage) have an impact on:

- (a) the extent of N₂O, CO₂ and methane CH₄ emissions?
- (b) the total carbon and total nitrogen stocks in soil?
- (c) the nutrient losses to water?’

if so:

- (d) For how long will this impact last?
- (e) Can this impact be quantified?
- (f) Which are the main drivers of this impact?

The novel aspect of this work will be to investigate the environmental impact of artificial land drainage of intensively managed permanent grassland.

This thesis is divided into five linked chapters; a literature review, two chapters describing field experiments concerned with GHG emissions and P and N losses from grassland under artificial drainage management, a chapter concerned with the modelling of GHG fluxes from grassland under artificial drainage management, and a final discussion chapter.

Grassland productivity

Managed grasslands occupy more than 33 million square kilometres or 25% of the global land surface area, making these the single most extensive form of land use on the planet (Asner et al., 2004). Lowland areas in North West of Europe are more than suitable for the production and utilisation of forage i.e. grasslands and maize, and are thus the main European dairy farming regions (Taube et al., 2014). There is however, a diversity of dairy production systems in northwest Europe largely determined by differences in climate, soils and socio-economic circumstances (Humphreys et al. 2009).

Commercial dairy farms vary significantly in their forage production system. Dairy production systems in the Netherlands and northern Germany aim for the maximisation of the milk yield per cow, and rely on energy-rich feed crops (grain), whole-plant silage from maize with external protein-rich feed such as soybean. In contrast, there are regions with a large proportion of grassland, such as Ireland and western France, where ruminant livestock is predominantly fed by forage obtained from the grassland, through either grazing or cutting. These systems are characterized by a minimization of the costs although the milk yield per cow is constrained by the feed efficiency (Taube et al. 2014). In grassland systems is often challenging to achieve the

maximum feed efficiency, highly determined by the cow breed, however with a well-managed grassland good quality feed can achieve an annual average milk yield of approximately 6,000 litres with a minim feed-stuff input ($<130 \text{ kg DM cow}^{-1}$) (Dillon et al., 2005).

Dairy production systems based on grazed grass provide the opportunity to produce milk in the cheapest way, converting grass into milk during the grazing season (April to October). The typical system of dairy production in Ireland is unique in Europe; it has evolved to a seasonal production system that makes maximum use of grazed grass in the diet of dairy cows (Humphreys et al. 2009). Soil and climatic conditions in Ireland favour grassland farming because soil temperatures are generally sufficient to allow grass growth throughout the year, albeit at low rates during the winter, and soil moisture deficit generally (with exceptions i.e. summer 2018) does not impede grass growth during the summer (Humphreys et al. 2009). Therefore the design of the pasture systems includes fencing to control livestock and divide the grassland paddocks, provision of drinking water and put in place several management tools, such us cut and weigh or raising plate meters, to be able to stablish a rotation system in order within the grassland swards. Pasture based systems are highly dependent on climatic conditions. Grassland systems are gear assembly between cow fertility and grass production. The latter is highly dependent on climate conditions hence efforts are made to match the peak of grassland production with the pick of milk production (ten weeks after calving). However, it does not always fits perfectly and often there is a surplus or deficient on grass production. The key to success of the grassland systems is to be able to manage those extras and deficits in order to ensure the grass quality to feed the cows and achieve high milk production rates. Hence, due to the seasonality and exposure to the unpredictable weather conditions, there is a need of an intensive observation of climatic and edaphic variables. Furthermore, those measurement will eventually become decision making tools that will greatly help and improve the management of grass production and overcome challenges such as changes in soil moisture over the seasons.

1.1.2 Length of the grazing season

Climatic conditions, such as those experienced in Ireland, UK and North-West Europe, allow a long growing season (between 18 and 21 days) for pasture with the potential to produce between 12 and 16 t DM ha⁻¹ on average from February to November (Dillon et al., 2005). This is competitive advantage, being able to produce good quality grass over a long period of time, over other countries. Dillon et al., 2005 showed a strong relationship between total costs of production and proportion of grass in the cow's diet in a number of countries (Figure 1.1).

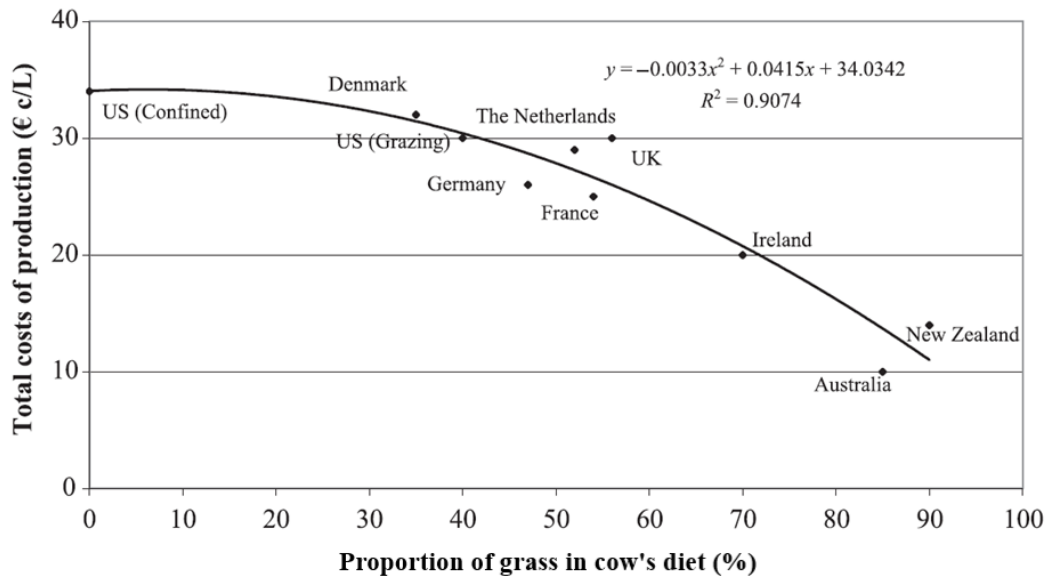


Figure 1.1 Relationship between total costs of production and proportion of grass in cow's diet (Dillon et al., 2005).

However regions with greater proportion of grass in cow's diet, are also characterized by high annual precipitation and low evapotranspiration rates, which lead to poor soil infiltration, high soil moisture content and poaching damage to pasture by cattle (Beukes et al., 2013). Conditions as such impede the grass utilization over certain seasons (winter) or over longer periods due to soil hydraulic conductivity characteristics (clay soils), common soil type formation on the mentioned regions.

However improving soil drainage by developing infrastructure like land drainage practices, can facilitate management and allow for higher stocking rate (Dobbie and Smith, 2006, Fitzgerald et al., 2008). Previous studies have shown that lengthening the grazing season by 27 days reduced the cost of milk production annually by 1 cent L⁻¹ (Dillon et al 2005). Authors of the same study emphasized the potential of grazing systems to further reduce the cost of milk production by increasing the proportion of grass in the diet.

1.1.3 Drainage as a management technique to improve grass utilization under grazing

Agricultural land drainage has been proven a successful soil management technique for soils with poor hydraulic conductivity. Drainage, and particular mole drainage, is a locally and direct modification of the soil structure that directly impacts on the physical characteristics of the soil.

The major aim of drainage is to facilitate the water flow through the soil profile, therefore improving the permeability and the control of the soil water table. This will be no longer controlled by the soil properties alone, but also by the drain depth and drain spacing (Bernd Lennartz et al., 2011).

Previous studies have shown how agricultural drainage improved the management in grassland production systems. Tyson et al. (1992) conducted an experiment over 5 years with beef-cattle and concluded that benefits from drainage were modest and for beef production unlikely to pay for its costs on the short term. However, authors observed that during the spring drained plots had an 11% higher dry matter production. Fitzgerald et al., (2008) simulated the influence of poor drained soil on grass based dairy production systems in Ireland and results showed that by draining the soil a higher stocking rate was achieved, hence the productivity per hectare increased. Furthermore authors of the latter study also observed that under poorly drained conditions the housing days over winter increased on average 48 days compared with well drained sites. The improvement on grass utilization through drainage can be achieved by a combination of higher herbage yield and less poaching damage due to a lower soil moisture in the top soil (Tyson et al., 1992).

1.1.4 Drainage to improve trafficability

Most of routine agricultural management such as fertilizer spreading, for example, is done by heavy machinery. Soils that suffer from poor infiltration and water logging impede the passage of machinery, which delays the management. Traffic causes soil compaction, which can have a negative effect on the grass growth rates because compacted soil layers can impede water infiltration and negatively impact root development over the long term (Vero et al., 2014)).

Lowering soil moisture content following artificial land drainage has been shown to decrease soil compaction, increase grass production and improve trafficability (Vero et al., 2014; Armstrong and Garwood, 1991). Several studies have reported a reduction on crop growth by up to 30% due to shallow water table depth that increase soil moisture content (Brereton and Hope-Cawdery, 1988; Bruke et al., 2004; Laidlaw, 2009).

1.2 Impact of drainage on soil conditions

1.2.1 Soil water movement and energy balance

Soil water movement

Water movement through soil is regulated by soil structure. Water that enters the soil matrix is stored in micropores approximately $< 30 \mu\text{m}$ in diameter, whereas large macropores ($> 30 \mu\text{m}$ in diameter) remain full of air allowing drainage once there is no impediment to hydraulic conductivity. If an impediment exists these pores temporally fill and the soil can be saturated with water (Figure 1.2). The balance of water stored in the soil is generally explained by the following equation:

Equation 1: $\Delta W = Pe + I - R - D - RO - T - ET$

Where:

ΔW = the change in water storage

Pe = rainfall

I = irrigation

R = deep percolation

D = discharge from drains

RO = surface runoff

T = interflow

ET = evapotranspiration

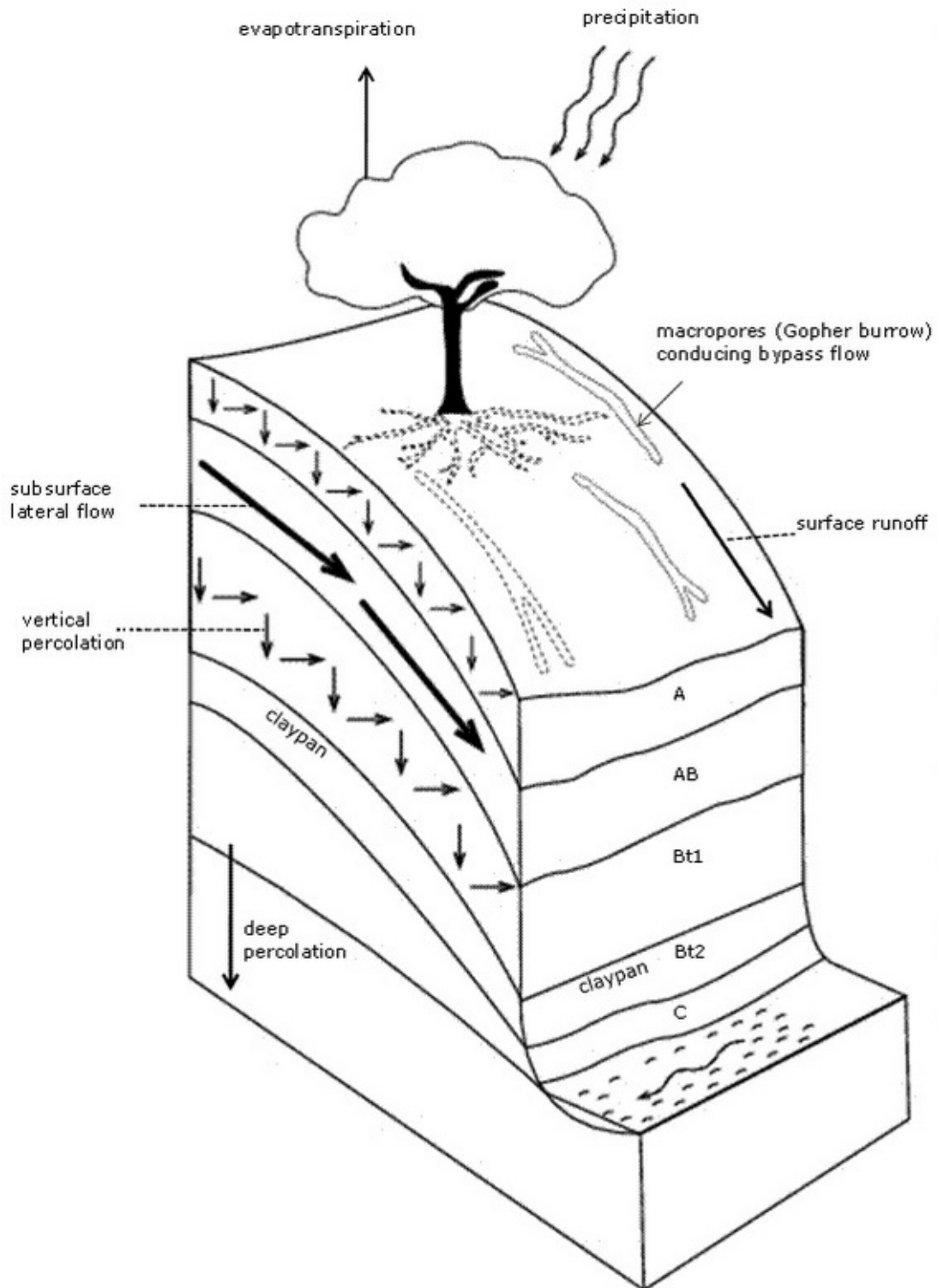


Figure 1.2. Conceptual diagram of a soil profile illustrating the multiple flow paths through which water moves through soil. © 2013 Nature Education Modified from O'Geen *et al.* 2010.

A soil water balance (SWB; e.g. equation 1) generates an estimate of the surplus water available for loss as drainage or surface runoff.

Approximately 40 to 60% of the soil volume is made of solid material (minerals and organic matter) that are surrounded by spaces filled with water, with gas or with roots or other living microorganisms. Small pores hold water more strongly by capillary forces than larger pores. According to the classification of Greenland (1977) interconnected pores with diameter range larger than 50 μm (transmission pores) drain under gravity. This will allow air to enter and therefore aerobic conditions. Pores ranking between 50 – 0.5 μm (storage pores) can hold water against the force of gravity but available for root extraction, forces up to -1500 kPa. While with pores smaller than 0.2 μm , water is available neither for gravity drainage nor for root extraction. The total volume of these spaces (pores), their interconnectivity, stability and relative proportion of each sizes class will have a major impact on the amount and speed of water movement through the soil profile (hydraulic conductivity) and the soil water retention capacity (Baisden, 2001).

Mole and gravel mole drainage has a direct impact on the soil structure, creating fissures in the soil through which water can move readily (Armstrong and Garwood, 1991). More information in section 1.5 Mole and gravel mole drainage installation on site. This can directly impact on the soil hydraulic conductivity, improve the soil vertical drainage rates and prevent the flooding of the soil surface. Numerous studies have shown that agriculture land drainage can remove the excess water on the soil matrix and therefore increase the soil strength leaving it less vulnerable to damage (Mulqueen and Roche 1975; Oosterbaan and Nijland 1994).

Water and energy balance

In order to understand and explain the water movements within the ecosystem an integrative understanding of living systems within the soil–plant–atmosphere continuum (SPAC) has been deeply studied for many years. Water is the most obvious component of the SPAC, and the relation of this water with energy is the key of our understanding of how life interacts with its environment. The heat (energy) of the sun is the driver of the extraction of energy from the surface as a result of evaporation and the release of energy to the atmosphere by condensation. This ‘latent heat,’ associated with the change of phase of water between liquid and vapor phase, provides the key feedback between water and energy cycles that stabilizes the soil–plant–atmosphere system, permitting it to sustain life. (J.M. Norm and M.C. Anderson, 2005).

Water movement through the SPAC is driven by a generated energy gradient. The energy gradient is a result of between two different water potential. Water potential quantifies the tendency of water to move from one point to another due to osmosis, gravity, mechanical pressure, or matrix effects such as capillary action (Cornelis et al., 2009). When looking at a vertical section of a plant and soil the highest water potential lies in the soil while gets gradually

lower as it goes up through the plant. Hence, it is essential to allow the water movement and substituent nutrient transportation to have a gradient on the water potential (with very negative water potential in the roots and less negative at the highest appendix of the plant, assuming a continuous water potential).

Soil water potential is highly dependent on the soil moisture. As mentioned in the previous section, pore size can play an important role retain water. Moreover, the amount of water in the soil will be determining the soil water potential. For example, at saturation point all soil pores are filled with water and water potential is 0 kPa, typically water will drain from macro-pores by gravity and water potential will be reduce. While at field capacity water potential can be approximately -33 kPa (depending on soil type), in general field capacity is viewed as the optimal condition for plant growth. However, when water potential reaches -1500 kPa, it is considered the permanent wilting point where water is held by soil particles and becomes unavailable for uptake by plants. In contrast, atmospheric water potentials are more negative than soil water potential (approximately -100 kPa) in order to allow the driving factor of this process (Cornelis et al., 2009). Hence it could be expected that by lowering soil water content through agricultural drainage soil water potential might be decreased. An impact on soil water movement can have a direct impact on nutrients taken up by plants. The latter is highly influenced by the soil-to-root contact zone (Cornelis et al., 2009). For the roots to be able to uptake nutrients those need a more negative water potential than the soil water potential. This water potential deepens on several factors such as soil hydraulic conductivity, rooting depth and density, depth of the water table and the atmospheric evaporative demand. Overall it is a complex process; however, in order to observe an impact in the soil water movement through the plant, changes between dry and wet soil should be persistent and enough differentiated i.e. dry soil reaches wilting point (Cornelis et al., 2009)

Not only the water movement but also the overall water balance can be affected due to a lower water content in the soil matrix. There is a very tight relationship between water and energy balances. In order to understand the relationship it is important to examine the energy balance equation (Equation 2).

$$\text{Equation 2: } R_n + G + LE + H = 0$$

where,

R_n is net radiation of sun

G is soil heat flux

LE is the latent heat flux (evaporation to the atmosphere) and is the product of the evaporative flux (E) and the latent heat of the evapotranspiration (λ).

This relationship between both balances is illustrated in Figure 1.3.

Figure 1.3. Water and energy balance components. Water balance component are in black, energy balance component are in white (Huang et al., 2011)

It is well established in the literature that the components of the energy balance, in particular LE, are highly influenced by season, type of crop, type of soil (bare soil) and also irrigation or rainfall regimes (Huang et al., 2011). However the impact of agricultural land drainage on the energy balance is generally understudied. When looking at a wide range of climates, for

example within Europe. Teuling et al., (2010) studied the impact of heat waves across grassland and forest land cover within Europe, they concluded that in grasslands this process accelerates soil moisture depletion and induces a critical shift in the regional climate system that leads to increased heating. In general, most studies including the one mentioned above that investigated the impact of changing water balance on energy balance have focussed mainly on changes driven by extreme climatic conditions and also the impact of land cover type (Wolf et al., 2013; Wang et al., 2019; Xu et al., 2019). However, all the above mentioned studies agree that by lowering soil water content, on grassland sites, there was a reduction in gross primary production persistent for several weeks to months, depending on the site. Therefore, by applying subsurface agriculture drainage and changing the water balance, the overall energy balance will change and this can have further implications regarding the nutrient uptake from plants.

Soil temperature

Soil temperature has an impact on the types and rates of chemical reactions in the soil. Temperature is one of the main variables that controls the enzyme activity and hence the microbial activity (Six et al., 2002). The latter control and influence the biochemical processes that undergo in the soil. Those processes allow that nutrients present in the soil become available by plants and other microorganisms. Therefore, soil temperature directly influences biological processes, such as crop growth and root development.

The heat transport within the soil depends on the heat conduction and convection capacities; both are strongly influenced by soil colour, texture and water content (Jury et al. 1991).

Hence lower soil water content can have an impact on soil temperature. Most of the literature agrees on the fact that agricultural land drainage increases soil temperatures during the spring in regions with humid climates due to lower energy required to heat the soil profile when there is less water held in the soil. For example, it has been found that drainage decreased the heat capacity of the wet soils and raised the soil temperature during the spring (Baver et al. 1972, Taylor and Jackson 1965a,b, Jin et al., 2008). However, more recent studies have concluded that when high water content is present in the soil much more heat is needed to achieve a temperature change because the heat capacity is three to five times more than for mineral soils (Foth 1990, Miller and Donahue 1990 and Jin et al., 2008). Hence the changes in soil moisture content in mineral soils are likely to require greater heat in order to influence in the same degree the soil temperature as the non-mineral soils.

1.2.2 Soil pH

Soil pH is a measure of the activity of ionized hydrogen (H^+) in the soil solution (McClean., 1982). Soil pH is determined by soil composition and the nature of the particles that form the soil. It measures the alkalinity and acidity on a scale from 0 to 14. Alkaline solutions have a pH that is greater than 7 while acidic solution have a pH less than 7. As the concentration of hydrogen ions increases, the pH of soil decreases. By implementing agricultural drainage a decrease in soil moisture and increase of oxygen present in the soil profile is achieved, and both variables greatly influence the biochemical process within the soil (Smith, 1990). Furthermore both are strongly linked, oxygen partial pressures being an important factor for O_2 diffusion, soil WFPS is the most important regulating factor as water presents a barrier to impede diffusion through the soil profile (Smith, 1990). Hence, by draining the soil and increasing O_2 diffusion the latter reduces the activity of denitrification enzymes by regulating the flow of electrons, suppressing the expression of structural denitrifying genes (Berks et al., 1995) and inhibiting the NO_3^- up take systems (Hernandez and Rowe, 1987).

All those changes will have an impact on concentrations of ions available in the soil matrix that will determine soil pH. There have been few studies that have investigated the impact of agricultural drainage on soil pH, however it is well established that changes in soil WFPS have an impact on soil pH. For example, Zhang and Wienhold, (2002) observed that pH increased when WFPS reached levels of 80% due to established anaerobic conditions that increased the presence of $NH_4^- N$ in the soil matrix. More recently, Giles et al., 2012 established that and increased soil WFPS, as mentioned above strongly linked to availability of O_2 in the soil, will enhance the denitrification and therefore also influence the concentration of ions in the soil matrix.

Overall a shift in soil pH promoted by agricultural drainage can strongly determine the predominant biochemical process, such as nitrification, complete or incomplete nitrification, in the soil and ultimately influence the output of those process. For example resulting on release of N_2O from the soil to the atmosphere due to inhibition of N_2O reductase (Baggs et al., 2010). Further details of the impact of pH on soil N_2O emissions is addressed in this Chapter, section 1.3.3.4.

1.2.3 Soil organic matter

Soil organic matter (SOM) can be physically, chemically and biochemically protected (Figure 1.3)

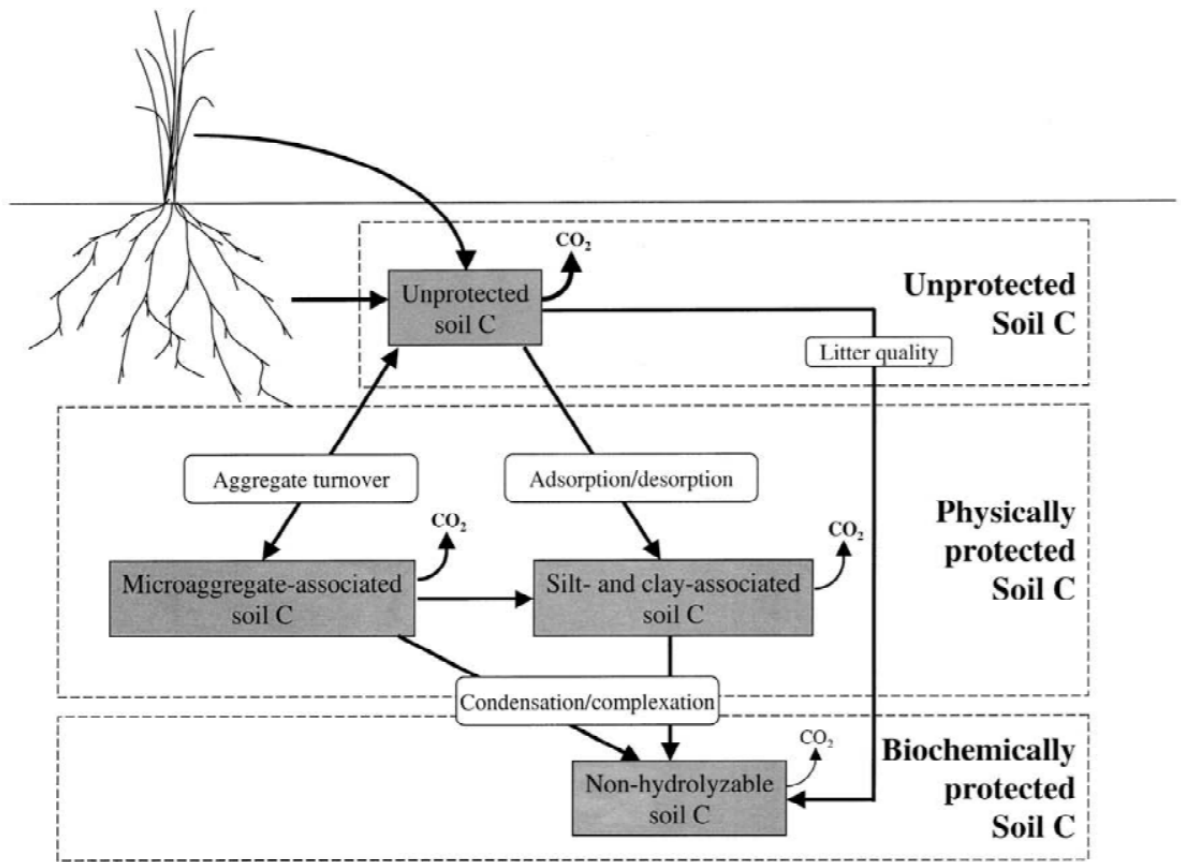


Figure 1.4. Conceptual model of soil organic matter (SOM) dynamics with measurable pools. The soil processes of aggregate formation/degradation, SOM adsorption/desorption and SOM condensation/complexation and the litter quality of the SOM determine the SOM pool dynamics. Source Six et al. (2002)

- Physical protection is performed by soil aggregation, which reduces contact between the chemical compounds of the SOM and microorganisms, enzymes and oxygen.
- Chemical protection takes place when organic materials are associated with soil minerals either directly or indirectly through the cation-binding.
- Biochemical protection results from condensation and polymerisation reactions forming organic macromolecules.

The breakdown of organic matter involves a complex process, including chemical alterations of organic matter, physical fragmentation and release of mineral nutrients. Several soil microorganisms are involved in this process, such as microorganisms, earthworms, microarthropods, ants and beetles. Organic matter breakdown is regulated by many factors, including soil moisture, thermal regimes, soil texture, bedrock type, nutrient status (cation

exchange capacity), water capacity, illuviation and bioturbation rates, root penetration and the availability of oxygen to support aerobic microbial respiration. Disturbances such as deforestation, logging, agricultural and grazing and biomass burning lead to a reduction of the soil organic carbon (SOC) either due to an increase of carbon release or lower carbon inputs. For example, ploughing damages soil structure, aerates the soil and accelerates the decomposition of SOM (Six et al., 2002).

Quantities of SOM present within a given soil are generally constant; changes tend to result in an equilibrium concentration shift. Necpálová et al. (2013) found that after ploughing grassland SOC decreased 32.2 t ha^{-1} and the difference between the permanent grassland occurred within the first four month period after ploughing. An earlier study by Jensen et al. (2001) measured soil respiration directly using a dynamic-chamber method during the first three months following ploughing of an old grass-clover pasture in grass-arable rotation. Their results showed that cultivation caused losses of $1.2 \text{ t carbon (C) ha}^{-1}$ over a three-month period, compared with undisturbed grassland. It has also been shown that the time required to reach the new equilibrium can vary from a few years or decades to several centuries (Six et al., 2002).

Hence, the literature has provided a lot of evidences of how ploughing had significantly decreased SOC. The impact of mole and gravel mole drainage on SOC has not yet been investigated, however due to the similarities of the above mentioned drainage techniques with soil ploughing, those might have a similar impact on SOC.

1.3 Impact of drainage on GHG emissions

Globally, agricultural soil contributes from 10 to 12 % of anthropogenic sources and accounts for 60% of global N_2O and 50% of global CH_4 emissions (Smith et al. 2007). The concentration of CO_2 in the atmosphere has increased by 35% percent since the beginning of the industrial revolution. Soil and land management practices affect the SOC content of the soils hence affecting the emission of for example soil CO_2 (Kumar et al. 2014).

A recent study by Tubiello et al., (2015) found out that more than 25 million hectares of organic soils were drained worldwide for agricultural use of which about 35% were in tropical areas, 60% in boreal and temperate cool areas, and 5% in warm temperate areas. According to this study, the emission from the drainage was significant, amounting to approximately one billion tonnes CO_2 equivalents annually, of which about 780 million tonnes were CO_2 (which was more than one-fourth of total amount of CO_2 emissions) from agriculture, land use, and forestry. A large amount

of the emission came from a few tropical countries in Southeast Asia linked mostly to drainage and land clearing for crop cultivation. Suggesting that land use change such as drainage, that implies changes in the soil chemical and physical characteristics, is a major contributor to the soil GHG emissions and emissions require to be accounted.

1.3.1 Soil carbon dioxide (CO₂)

1.3.1.1 System balance (photosynthesis/respiration)

The carbon cycle starts with the fixation of atmospheric CO₂ by plants that are then converted to organic carbon compounds through photosynthesis. Some of those compounds will be used for the growth of plant cells and tissues and some will be break down to supply energy to the plant. However as the carbon goes through the cycle, CO₂ is produced and emitted back to the atmosphere. The main sources of CO₂ are plant respiration that can be divided into the above ground respiration and below ground respiration. Below ground respiration is equivalent to a combination of autotrophic and plant respiration. Autotrophic respiration accounts from approximately half of the total soil respiration but it can vary between 10 and 90% among different studies (Hanson et al 2000).

The CO₂ flux measured at the soil surface can be considered as soil respiration when CO₂ production and transport are at steady state. Curtis et al., (2005) found that the relative contribution of soil respiration to ecosystems respiration varies considerably in a year. Soil respiration contributes nearly 100% of the ecosystem respiration for most of the winter while the contribution drops to about 60% during the period of fast leaf expansion. It then gradually increases during the growing season as the soil warms, reaching about 75% at the time of leaf abscission in the autumn (Curtis et al., 2005). Therefore soil respiration is closely related to the ecosystem production such as gross primary production (GPP) and net primary production (NPP). GPP is annual carbon assimilation by photosynthesis ignoring photorespiration and NPP can be estimated by measuring yearly increments in plant biomass. A combination of different measuring strategies is required in order to assess the carbon balance on crop land (Figure 1.5).

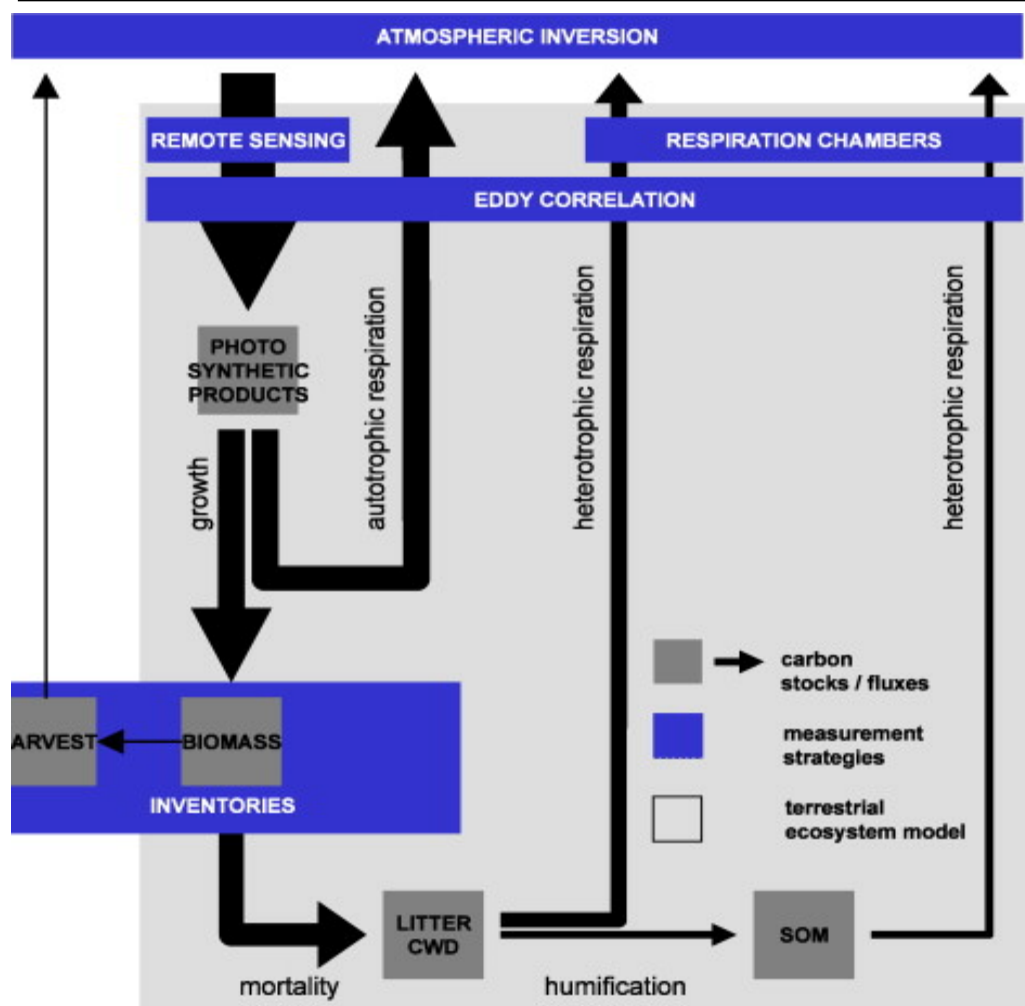


Figure 1.5 Schematic representation of the components of the carbon balance of croplands (Smith *et al.*, 2010)

All the process that take part in the system balance are influenced by chemical and physical characteristics of the soil.

1.3.1.2 Soil WFPS

Soil CO₂ flux is greatly influenced by water filled pore space (WFPS). The common conceptual relationship states that soil respiration is low under dry conditions, reaching the maximal rate at intermediate WFPS level (i.e. 60%) and rapidly decreases at a high WFPS level when anaerobic conditions depress aerobic microbial activity (Figure 1.4). The optimum water content is usually when the soil macropores are mostly air-filled therefore allowing O₂ diffusion, while the soil micropores are water-filled, hence facilitating diffusion of soluble substrates.

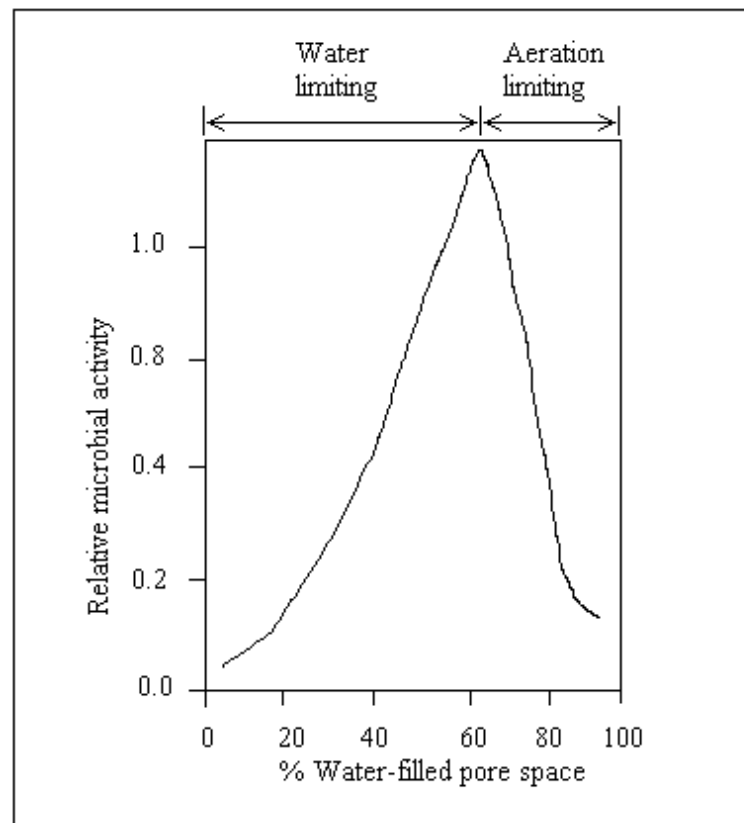


Figure 1.6. Idealized relationship between WFPS and relative amount of microbial respiration. It assumes that there is one optimal soil moisture content based primarily on Papendick and Campbell (1981).

Therefore changes in soil WFPS will have a direct impact on the overall soil CO₂ flux. For example wet-dry cycles in the soil driven by rainfall events can induce peak fluxes of CO₂ flux, as well as other gasses such as N₂O. At first, during the rainfall, water infiltrates and replaces CO₂ resulting in a degassing, this happens usually minutes after the rewetting and last for a few hours, strictly this is considered as release of stored CO₂ in soil from past microbial and root respiration. After rainfall, from several hours to a few days, microbial activities are activated (Glinski and Stepniewski 1985) resulting in an increase of soil CO₂ efflux.

1.3.1.3 Soil temperature

Another important variable that regulates soil respiration is soil temperature (Lloyd and Taylor, 1994). It is well established that maximum soil respiration occurs during warmer weather than during cooler weather (Deng et al. 2010). Moreover, it is widely accepted that rising temperatures would accelerate the release of CO₂ leading to further increase global warming (Luo, 2007; Allan and Soden, 2008; Smith et al., 2003).

The release of CO₂ from the soils is mostly through microbial decomposition of organic matter and root respiration (Davidson and Janssens, 2006). Microbial decomposition and root activity are affected by environmental factors and substrate changes associated with phenological processes (Gaumont-Guay et al., 2006; Curiel Yuste et al. 2007). Hence, the soil respiration will be influenced not only by the soil moisture but also by the soil temperature, many studies have shown that soil respiration is mostly affected an interaction from both variables. For examples, a study in Dinghushan Nature Reserve in the center of Guangdong Province in southern China have shown that soil respiration increased exponentially with soil temperature and has highly positively and linearly correlated to soil moisture (Deng et al., 2010, 2012; Tang et al. 2006). Results of these Chinese studies agree with recent findings by Jiang et al. (2013). In the latter study, authors found that an increase in soil temperature led to an increase in soil respiration. Moreover, it was also concluded that soil respiration would decrease with lower soil moisture.

1.3.1.4 pH

Soil respiration is primarily affected by soil moisture and soil temperature. However soil parameters such as soil pH, salinity and nutrient supply can also influence the CO₂ flux.

Very few studies have established a correlation between soil pH and soil respiration. Kowalenko and Ivarson 1978 found that CO₂ flux usually increases with pH when pH is less than 7 and decreases when pH is above 7. Similar results were found in more recent studies (Rao and Pathak 1996, Xu and Qi 2001a). Other studies have also shown that an increased soil acidity can lead to a greater accumulation of soil organic matter due to a reduced rate of microbial mineralization. Similarly, the effect of increased pH can impact on plant litter quality and increase in the C-to-N ratio of soil organic matter (Motavalli et al., 1995). However more recent studies have shown that an alteration of soil pH had no significant effect on the soil's C-to-N ratio (Kemmitt et al., 2006). Guggenberger et al., (1994, 1995), Van Bergen et al., (1997, 1998) and Bull et al., (2000) suggest the soil pH has direct impact on the metabolic modifications during the decomposition of specific components of the organic matter (e.g. lignin-cellulose, lipids), aside from the varying litter input at different pH. The latter points out the need of further studies, as the relative importance of biotic and abiotic processes in this process is unknown (van Bergen et al., 1997).

1.3.1.5 Soil structure

Unlike soil pH, soil structure has a great influence on soil respiration. The influence is mainly through its effects on soil porosity and soil moisture. The soil porosity determines the water

holding capacity, water movement and gas diffusion in the soil. Hence a change in soil structure can potentially impact on soil moisture and subsequently soil respiration. Moreover structure can also influence on rooting systems, such as rooting depth, thus indirectly impact soil respiration (Yiqi et al., 2010).

Furthermore, soil structure (porosity) changes across the soil profile and soil macro and microbiota play an important role in ensuring this. There are different decomposers acting on organic matter in the soil in different spatial scales (Filser et al., 2016). Decomposers break down plants and animal residues, carry out bioturbation, distribute organic matters, and produce large and smaller organic matter such as termite mounds and faecal pellet (Wilkinson et al., 2009; Sanders et al., 2014). The structures affect plants and microorganisms and determine the stability of the soil. Earthworms feed on mineral and organic parts of the soil and mix them affecting the retention of SOM (Bottinelli et al., 2015). Decomposers have different impacts on soil. For example, some species of earthworms change physically the soil by creating a network of conduits while others are associated with altering the SOM content (Jimenez et al., 2012).

Finally, as pointed out in the earlier section other studies have also observed that when changes in soil structure are promoted by disturbances, for example ploughing, the soil respiration increased (Jensen et al., 2001). In this case, the authors attributed the increase of soil respiration to an improved soil aeration and moisture conditions that enhanced the decomposition of the SOM.

1.3.2 Soil methane (CH₄)

Anaerobic microbial processes are widely considered the source of atmospheric methane in the biosphere. A large amount of this CH₄ produced is used by the soil through methanotrophic consumption. However, soil can also release (emission of CH₄ to the atmosphere) methane due to changes in soil water content and soil temperature.

1.3.2.1 Soil temperature

As mentioned in earlier sections (1.2.3 and 1.3.1.3) soil temperature is a key driver of microbial soil processes. Likewise for CO₂, production of CH₄ increases with an increase in temperature and anoxic conditions. Previous studies have reported a negative correlation between SOC and soil temperature (Balogh et al., 2011). Moreover Hamdi et al. (2013) observed that this relationship differed depending on the region, whereby SOC was more sensitive to temperature changes in colder than in temperate environments.

Hence, depletion of SOC through higher temperature can lead not only to higher soil CO₂ emissions but also to higher CH₄ flux (Jugold et al., 2012). Depending on the climatic region this change in soil temperature might have different effects on the soil CH₄ flux, for example a shift from CH₄ uptake to CH₄ emissions. This effect was observed in a previous study by Borken et al. (2006) where a significant and negative correlation between temperature and soil methane emission was established.

1.3.2.2 Soil moisture

Soil moisture is one of the most influential variables on the soil CH₄ flux. On an overall global scale, the greatest soil CH₄ emissions come from wetlands. However, this may change due to the increasing atmospheric temperatures in those regions. Higher temperatures and increasing drought events will lead to lower water table depths, which in mineral soils can decrease the emission of CH₄ because of less anaerobic conditions. Several studies have been carried out to demonstrate the impact of reduction of soil water on soil CH₄ flux. In general the literature agrees that there is a positive linear relationship between soil CH₄ flux and soil water content. For example, Merbold et al. (2009) experimentally drained a 400 meters diameter region of wetland tundra in the Siberian Arctic leading to reduction of the average CH₄ flux to below 4% of their control plot. Moreover, Sturtevant et al., (2012) also reported that an increased inundation increased the soil moisture and led to a greater release, by more than 50% compared to the control, of CH₄. In agreement with the latter study, Zona et al. (2009) observed a strong relationship between soil CH₄ flux and the soil moisture. But in their study, the authors found that an increase in soil water content led to the reduction in the soil methane flux. Differences amongst studies can be attributed to the differences in the diffusive resistance, authors of Zona et al. (2009) increased soil water content by inundation (water level above the soil surface), while other studies rarely reached the saturation point.

1.3.2.3 pH

Soil pH is one of the key environmental parameters affecting soil CH₄ emissions and uptake, through methanotrophy and methanogenesis. However the optimal pH range for methanogens and methanotrophy overlaps, while the first ranges from 6 to 8 the latter goes beyond the more acidic conditions (Semrau et al., 2010). Moreover both communities reach to threshold pH levels below 5.75 and above 8.75, where their activity is inhibited (Jeffery et al., 2016). Because of the overlapping it is expected that the two process coexist and are similarly affected by a change in of the soil pH and also soil moisture and soil oxygen. Hence it is often difficult to predict if the change of pH will eventually increase or decrease the overall soil CH₄

emissions. However, it is widely accepted that methanogenic bacteria are very sensitive to acid conditions and that the optimum soil pH is 7.0 (Jeffery et al., 2016).

Previous studies have examined the relationship between pH and CH₄. For example, Dunfield et al. (2007) found that production of CH₄ in the soil tends to decrease by increasing the acidity. This is due to the increased activity of methanotrophic community and decreased of the methanogeni. In contrast, Jeffery et al. 2016 reported an increase in the CH₄ with acidity of the soil. Authors of the latter study attributed this effect to the smaller size/structures of methanotrophic communities in their site.

1.3.3 Nitrous oxide (N₂O)

Nitrous oxide is a GHG that is released from denitrification and nitrification. N₂O in the atmosphere has increased by 19% since the pre-industrial times increasing by an average of 0.77 parts per billion (ppbv) yr⁻¹ from 2000 to 2009. N₂O has a 100-year global warming ability 298 times more than that of CO₂. Moreover, it can deplete the stratospheric ozone. Most of soil N₂O is sourced from microbial production processes in sediments, water bodies, and soils. The N₂O emissions are influenced by several factors such as temperature, moisture, and soil pH (Chen, Whalen and Guo 2013).

1.3.3.1 Soil N system balance

Nitrogen is important for plants and living organisms. Living organisms use N to form different organic compounds such as amino and nucleic acids, and proteins. Nitrogen cycling in terrestrial ecosystems and landscapes is mainly driven by microbiological and plant processes, with physicochemical processes such as diffusion, emission, volatilization, leaching or erosion leading to displacement of N on site, regional and global scales (Sutton et al., 2011).

Biological nitrogen fixation (BNF)

Biological nitrogen fixation (BNF) is the process where the atmospheric nitrogen (N₂) is converted to ammonia (NH₃) (Harrison 2003). It is carried out by a specialized group of prokaryotes that avail of the nitrogenase enzyme to catalyze the conversion of N₂. The dinitrogen molecule is joined by a triple covalent bound; nitrogenase catalyzes the breaking of this bound and the addition of three hydrogen atoms to each nitrogen atom. The reduction of atmospheric nitrogen is a complex process with a high energy requirement (Wagner 2011). Microorganisms that fix nitrogen require 16 moles of adenosine triphosphate (ATP) to reduce each mole of nitrogen (Smith et al., 2004). This energy is obtained by oxidizing organic

molecules e.g. sugars. Depending on their metabolic pathway those molecules are obtained from other organisms (non-photosynthetic) or produced by photosynthesis. In agriculture it is possible to find many microorganisms that fix nitrogen symbiotically by partnering with a host plant. The plant provides the sugars from photosynthesis that are utilized by the nitrogen-fixing microorganism as a source of energy to achieve the nitrogen fixation. In exchange, the microbe provides the plant with an additional source of plant-available nitrogen for its growth. By far the most important nitrogen-fixing symbiotic associations are the legumes and *Rhizobium* and *Bradyrhizobium* bacteria. Important legumes used in arable and grassland systems include alfalfa, beans, clovers, cowpeas, lupines, peanut, soybean, and vetches (Vance 2001).

Hence BNF contributes to generate plant material and plant growth. When looking at the N cycle it is observed that N₂O emissions can come from decomposition of plant material, hence a question can be arise of how much BNF would impact to the N₂O emissions. There have been few studies that showed how several isolated *Rhizobium* species can denitrify nitrate and release N₂O (Rochette and Janzen, 2005). Li et al., (2011) conducted a study were N₂O emissions from grassland with (white clover and ryegrass) and without BNF (ryegrass) were examined. Authors concluded that the fixed N in the mixed crop was higher in the ryegrass because fertilization was found to suppress BNF. Despite the residue from BNF authors did not observed difference between crops. They attributed it to a greater N use efficiency of the white clover due to the grass-legume interactions and efficient transformation of N into biomass. This is in agreement with other studies that suggest direct N₂O emission from BNF are negligible (Carter and Ambus, 2006; Zhong et al., 2009). Authors agree that N₂O emissions induced by the growth of legume crops/forages are estimated solely as a function of the above-ground and below-ground N inputs from crop residues (Rochette and Janzen, 2005; Carter and Ambus, 2006; Li et al., 2011; IPCC 2014). Hence, N₂O emissions from legume-based grasslands are lower than fertilized grasslands. For example, Ruzjerez et al., (1994) reported up to 5-fold of more N₂O emission from heavily N fertilized grasslands than from their legume-based counterparts. Furthermore, a data synthesis conducted by Jensen et al., (2012) indicated that the average soil N₂O emissions from field-grown legumes, N fertilized grass pastures and crops, and unfertilized soils are 1.29, 3.22 and 1.20 kg N ha⁻¹year⁻¹, respectively. Hence, all studies conclude that N₂O emissions from grass legume mixes are only slightly greater than background emissions (Rochette and Janzen, 2005; Jensen et al., 2012).

Industrial N fixation

Fritz Haber (Germany 9 December 1868 – 29 January 1934) developed in 1913 the Haber – Bosh process achieving the synthesis of ammonia. This process consists of combining atmospheric N to form NH_4^+ , under high pressure (150–250 bars) and a temperature of 600 °C, with the use of an iron catalyst and hydrogen (usually derived from natural gas or petroleum) (Erisman et al. 2008). This discovery led to the production of mineral fertilizers and formed the foundation of modern agriculture. Some authors attribute the population explosion during the second half of twentieth century to this discovery (Smil 1999).

Nitrogen uptake (NH_4^+ to Organic N)

Plants avail of NH_4^+ and NO_3^- to cover their N requirements, those are quickly incorporated into protein and other organic N compounds, by a host plant, bacteria or another soil microorganism (Harrison et al., 2007, Jackson et al., 2008). The uptake of N by the soil solution it is regulated by the transport systems in the plasma membrane of root cells. If the plant has a symbiotic relationship with *Rhizobium*, some of the N assimilated by the plant in form of NH_4^+ ion comes from the *Rhizobium*. External factors can also regulate N uptake such as soil pH, soil NH_4^+ and NO_3^- , light, temperature and organic compounds (Jackson et al., 2008). Ultimately the N uptake of each plant over a certain period will depend on the N supply and demand and on the plant species. Perennial ryegrass belongs to the C3 metabolism group of plants and can be considered a relatively high-N (>30 g N kg⁻¹ DM) photosynthetic foliage rather than low-N (14 to 20 g N kg⁻¹ DM) than can be found C4 plants such as maize, in storage organs or seeds as is the case for most arable crops. It is generally accepted that a sward producing close to maximum yields, in Ireland, of between 14 and 16 t dry matter (DM) ha⁻¹ annually needs to take up around 450 kg N ha⁻¹ yr⁻¹ (Humphreys et al., 2003)

Nitrogen mineralisation – ammonification (Organic N to NH_4^+)

Nitrogen enters in the SOM pool after internal N cycle via plants, livestock and other living microorganisms. When the depolymerisation of the SOM takes place, carried out by extracellular enzymes of fungi and bacteria (Jackson et al., 2008), the N becomes bioavailable for plants and microorganisms. Microorganisms can further convert bioavailable N into ammonium (NH_4^+) through mineralisation or ammonification process. Depolymerizing and ammonifying heterotrophic microbes are sensitive the C-limited conditions and those can be negatively affected by low C:N ratios. However, some studies agreed that this can be compensated by root exudation, root turn over and mycorrhizal turnover adding labile N and carbon sources to the soil (Bais et al., 2006; Hogberg and Read, 2006). This process shows the

strong dependence of depolymerizing and ammonifying heterotrophic microorganisms and the tight linkage between the soil N and soil carbon cycles.

Nitrification (NH_4^+ to NO_3^-)

Nitrification is the biological oxidation of NH_4^+ to nitrate (NO_3^-). Nitrification occurs through both the heterotrophic and autotrophic pathways. Autotrophic nitrification is performed in two different steps, in the first step NH_4^+ is reduced to hydroxylamine (NH_2OH), which acts as an intermediate between NH_4^+ and NO_2^- , and hydroxylamine is reduced to NO_2^- in the subsequent step. This first step is carried out by ammonium oxidizing bacteria and in some instances by ammonium oxidizing archaea (Costa et al., 2006). The second step (NO_2^- to NO_3^-) is carried out by NO_2^- oxidizers and nitrous oxide can be produced as a by-product during autotrophic nitrification (Stevens et al., 1998). While heterotrophic nitrification results in the same products, intermediates and has the same substrates as autotrophic nitrification it has been found to oxidise organic forms of N such as urea, as well as NH_3 . In general, heterotrophic nitrification is regarded to be only a minor source of N_2O compared to autotrophic nitrification. However, under certain conditions such as low pH, high oxygen contents and high availability of organic material it may produce significant amounts of N_2O (Stevens et al., 1998).

During nitrification certain enzymes are required, those are outlined on Figure 1.7. Enzymes regulate the nitrification process, and hence can trigger or stop the process going through the next step. This has been a subject of study in order to develop inhibitors to reduce N_2O emissions, for example dicyandiamide (DCD) has been found to reduce the rate of conversion of NH_4^+ to NO_2^- . This inhibitor acts deactivating the ammonia mono-oxygenase enzyme of soil ammonia oxidizers, making the enzyme ineffective for the first step of the nitrification process (Amberger and Amberger, 1989; Skiba et al., 2011). The use of such nitrification inhibitors is an effective mitigation strategy for N_2O emissions (Di and Cameron, 2012; Dai et al., 2013; Wakelin et al., 2013).

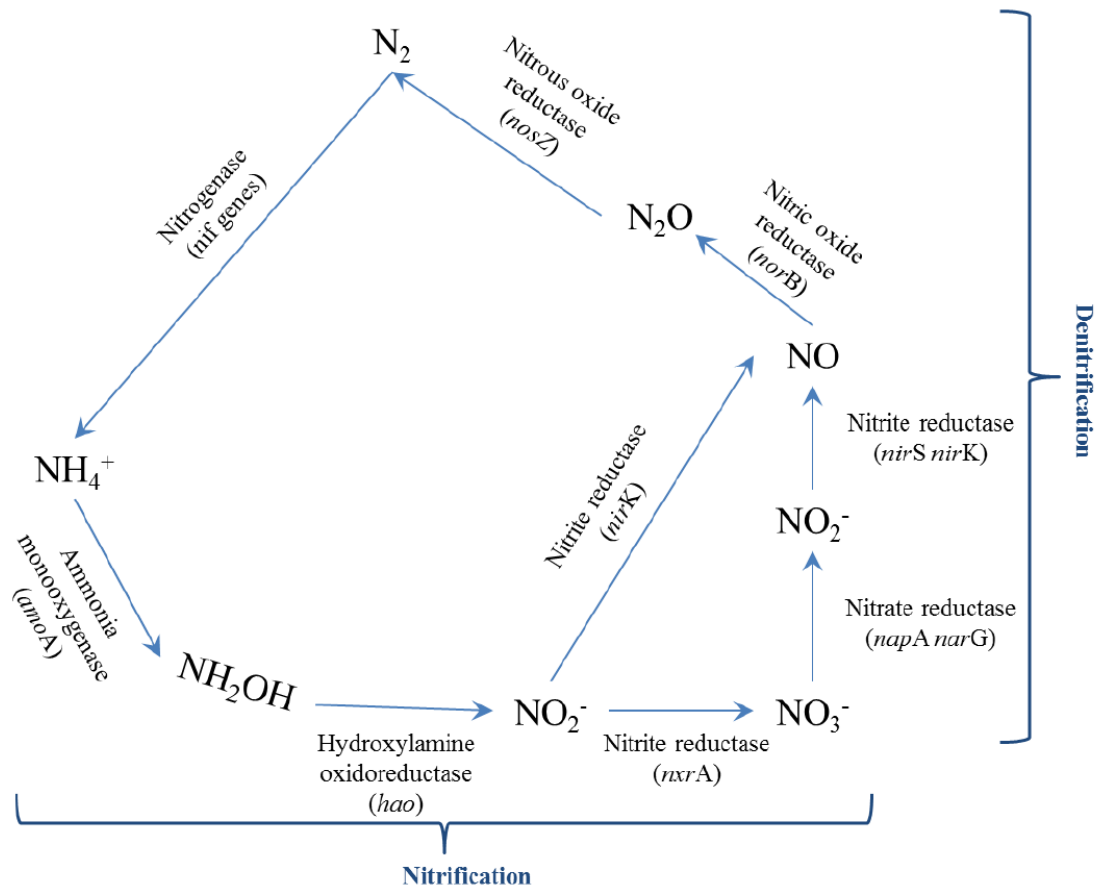


Figure 1.7. Enzymes and its functional genes (in parenthesis) which, encode these enzymes, involved in the nitrification and denitrification process. Source; (Baggs, 2011; Müller and Clough, 2012)

Furthermore, nitrification of ammonium increases the probability of reactive N (such as NO_3^-) forms being lost from the ecosystem, since these forms are susceptible to being lost by leaching through hydrological pathways or further reductions to gaseous forms such as NO , N_2O and N_2 via denitrification. Hence if nitrate is not being leached or does not undergo complete or incomplete denitrification, it can potentially be reduced to N_2O . Moreover, there is also the likelihood that nitrate can be immobilized by microbes or taken up by plants, since N can only be incorporated into the cell walls in reduced form (Glass et al., 2002). Moreover, the anion exchange capacity in the soil matrix is considerably lower than the cation exchange capacity. Then ammonium retention is promoted over nitrate, which is more susceptible to leaching.

Finally nitrification is, in general, an aerobic process, which is highly dependent on the oxygen availability in the soil profile (Conrad 2002). The optimum soil water-filled pore space (WFPS), depends on the soil texture and ranges between 30% and 60% (Bouwman 1998).

Denitrification ($\text{NO}_3^- \rightarrow \text{N}_2 + \text{N}_2\text{O}$) is the process whereby the NO_3^- is reduced to N_2O or N_2 (Figure 1.7) by an array of denitrifying micro-organisms (bacteria, archaea, fungi and other eukaryotes) (Baggs, 2011). Bacteria are considered to be the most important denitrifiers, however fungi have also been found to have important contribution to N_2O , particularly in soils under permanent grassland (Firestone 1982). During denitrification bacteria produce energy by oxidizing N and organic carbon (Firestone, 1982). Denitrifying bacteria can use as an electron acceptor either oxygen or NO_3^- , hence when oxygen is unavailable those use NO_3^- (i.e. anaerobic conditions).

As for nitrification enzymes are required in each step of the denitrification process (Figure 1.7) and therefore play a major role regulating N_2O and N_2 emissions. The genes which encode the enzymes involved in denitrification have been widely studied in order to reduce N_2O emissions. Research has been focused on developing molecular markers, which have been linked to denitrification process and rates in order to mitigate NO_2 emissions (Müller and Clough, 2012).

The main factors that control denitrification are the soil carbon content, organic carbon acts as an electron donor in the respiration process, and soil aeration (Firestone and Davidson 1989). However other factors such as WFPS, soil temperature and soil pH have also an influence in the process (Coyne 2008). Finally, other soil managements such as tillage can indirectly affect denitrification through influencing the variables previously stated. Similarly, the application of any organic amendments to the soil will influence denitrification through the provision of a new carbon source.

1.3.3.2 Factors affecting N_2O emissions

The main factors that can impact soil N_2O emissions are soil NO_3^- and NH_4^+ concentrations, soil moisture (particularly WFPS), SOC, soil pH, and soil temperature (Luo et al., 1999; van Spanning et al., 2007; Cuhel et al., 2010). Therefore grassland management and climatic factors, such as rainfall and temperature, have a large influence on the overall N_2O emissions from agricultural soils.

Soil WFPS

Many studies have found WFPS being a factor with a major impact on soil N_2O emissions (Figure 1.8), with emissions generally increasing under lower aeration or elevated WFPS (Bouwman 1998; Luo et al., 1999; Rudaz et al., 1999; Dobbie and Smith, 2003a; Bateman and Baggs, 2005; Liu et al., 2007; Luo et al., 2007). This can be partly caused by increased

precipitation or irrigation and soil compaction (i.e. grazing), which often lead to reduced aeration of the soil profile, creating anaerobic conditions that can induce denitrification and therefore increased N_2O emissions (Bhandral et al., 2007, Rafique et al., 2011, Rafique et al., 2012).

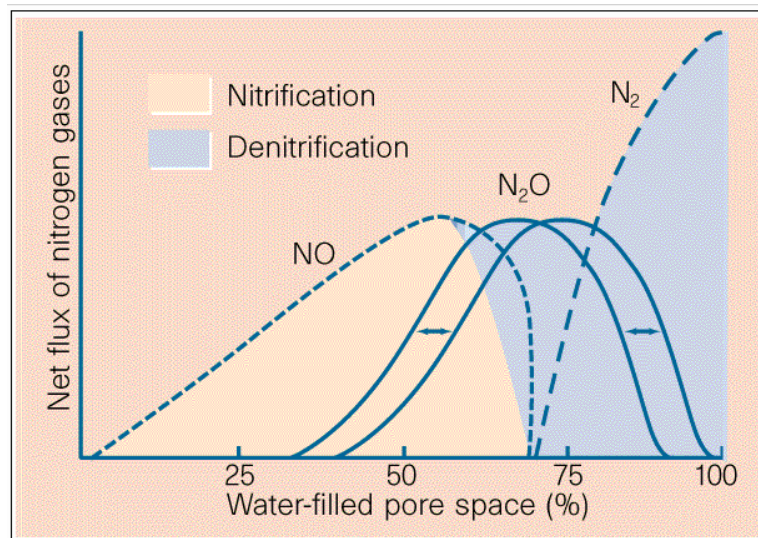


Figure 1.8 Effect of WFPS on the net flux of nitrogen gases (Bouwman 1998).

However the relationship is complex and previous studies have reported that N_2O emissions increase when soil moisture content range between certain threshold values of WFPS (Ruser et al., 2006; Turner et al., 2008). Moreover, those values differ according the soil type, hence influenced by the field capacity. For example reported threshold values for sandy soil range from 74% to 83%, sandy loamy soils from 62% to 83% and in loamy and clay soils from 50% to 74% (Barton et al., 1999).

Therefore it is expected that a change in the soil WFPS will induce an impact on the outputs of the nitrogen biochemical process within the soil. For example, rapid rewetting of dry soils can cause production of large volumes of N_2O attributed to release of osmolytes during the dry phase and the breakdown of abundant N substrates as the soil gets wetter (Kool et al., 2011). Similar results were reported in earlier dry-rewetting studies, authors concluded that emission of N_2O can be affected by factors such as soil compaction, frequency of the drying and rewetting cycles, fertilizer inputs (Ruser et al., 2006), and the type of crop the residue came from (Zhong et al., 2011).

Soil temperature

Similarly to WFPS, soil temperature directly impacts the activity of the nitrifying and denitrifying bacteria and therefore the N₂O emissions. An increase in soil temperature can lead to an increase in the activity of other microbial communities in the soil micro biota. Many biochemical and physiological studies usually show a general temperature-response curve that respiration increases exponentially with temperature. Low temperatures can limit the activity of both membrane-bounded and soluble enzymes (Atkin and Tjoelker 2003). Whereas in the high temperature range the main regulation factors are the adenylates (adenosine monophosphate [AMP], adenosine diphosphate [ADP], and adenosine triphosphate [ATP]) and the substrate supply (Svensson et al., 2002, Douce and Neuburger 1989, Atkin et al., 2002, Atkin and Tjoelker 2003).

The temperature sensitivity of denitrification, N₂O emissions and soil respiration is often described by Q₁₀ (a quotient of change in respiration caused by change in temperature by 10°C). Values vary widely across different studies from 1 (low sensitive) to 10 (high sensitive). These values depend on the soil type and geographic location (Peterjohn et al., 1993, 1994; Lloyd and Taylor 1994; Kirschbaum 1995; Simmons et al., 1996; Chen et al 2000). Abdalla et al. (2009a) reported Q₁₀ values ranging from 4 to 6 for temperatures between 10 to 25°C, whilst Dobbie and Smith, (2001) reported values of 3.7 and 2.3 for temperatures between 5 to 12°C and 12 to 18°C respectively for N₂O emissions. Based on data compiled nearly 20 years ago, the global median value of Q₁₀ for soil respiration rates is 2.4, with a range of 1.3 to 3.3 (Raich and Schlesinger 1992). Moreover a number of previous studies have demonstrated that increased rates of soil N₂O results from increases of soil temperature (Winkler et al., 1996, Christensen et al., 1997, Jarbo et al., 2008, M. Abdalla et al., 2011).

Soil pH

As mentioned previously in this Chapter, soil pH can greatly influence enzymatic activity, microbial activity and thus the nitrification and denitrification process. Denitrification rates are often thought to decrease with pH (Šimek and Hopkins 1999) and it is generally agreed that the optimum pH for denitrification is between 7 and 8 (Šimek et al., 2000). Several studies have proven that acid soil pH (range 4 to 6) increases soil N₂ and N₂O emissions. For example Liu et al., (2010) concluded that N₂ and N₂O emissions were higher when pH was lower (below 6). Furthermore, results of the latter study agree with the findings of Saleh-Lakha et al., (2009), whose authors concluded that nitrifiers have more favourable conditions to produce N₂O under acidic conditions (pH range 4 to 6). Both studies also suggest that the optimum pH range for soil N₂O emissions is between 4 and 6, and any changes outside this range might have an effect

on the soil N₂O and N₂ emissions. However, there are evidences were denitrification can be the dominant process below pH 4.9 (Ellis et al., 1998). Hence, the impact of soil pH on denitrification is more complex.

Most authors agree on the fact that changes in soil pH have a direct impact not only in the extent of denitrifier community composition (Enwall et al., 2005) that can influence the adaptation of denitrifiers to low pH (Parkin et al. 1985), but also in bacterial biomass (Blagodatskiy et al., 2008). In agreement with previous authors Baggs et al., (2010) found denitrification being the predominant process (accounted 80% of N₂O-N emissions) at soil pH 4.5. Authors of the latter study attributed that to the fact that N₂O reductase (reduction of N₂O to N₂) in denitrification is more sensitive to pH than other denitrification reductases (Knowles, 1982). Similarly, on an earlier study Thomsen et al., (1994) showed that *P. denitrificans*, an N₂O reductase, was inhibited at low soil pH.

Another variable, the time taken for a change in pH, should also be considered in order to assess the impact of soil pH change on biochemical processes. Baggs et al., (2010) conducted a study where they examined the long-term effect (from 1 to 2 months) of soil pH change, found that long term changes in soil pH had significantly lower N₂O emissions than short term changes. Authors attributed the effect of the capacity of bacteria to become more tolerant to the new soil pH conditions. Hence this will be highly influenced by the denitrifying phylogenetically diversity of the community (Enwall et al., 2005) and adaptation to different pH (Parkin et al., 1985). Some strains of denitrifying and ammoniaoxidising bacteria may be more acid tolerant than others and prevailing low pH conditions may select for these more tolerant types (Parkin et al. 1985).

Overall, the short term changes in soil pH are likely generate greater N₂O emissions than long term changes and mostly due to the adaptation of bacteria (Baggs et al., 2010). Thus, the surrounding environment where the bacterial communities are might act as natural selection for bacteria. There are some evidences in the literature where authors observed selection of different archaeal and bacterial ammonia oxidizer phylotypes, for example Nicol et al., (2008) in the observed a similar effect on a maintained in the pH range 4.5–7.5 since 1961. Authors of the latter study also investigate the gene expression, they found that bacterial *amoA* gene expression was much lower than that of archaea at low pH but increased with increasing pH. The authors suggested that this was a reflection of different preferences for available ammonia or fundamental differences in their physiology and metabolism (Nicol et al., 2008).

Soil structure and texture

As much as the soil temperature, WFPS and pH can influence the soil N₂O flux, soil structure plays a key role influencing denitrification and nitrification. For example, in clay soils there tends to be increased denitrification activity because the soil aggregates prevent the flow of oxygen leading to anaerobic microsites promoting the process (Stolk et al., 2011). Hence, the structure of the soil influences denitrification by influencing the availability of oxygen in soil.

Moreover, in the recent years, the change in soil structure by macrofauna and its impact on the soil N₂O emissions has been widely studied. Lubbers et al., 2013 showed that there are interactions of soil macrofana and soil microorganisms that lead to an increase in soil N₂O emissions of up to 42%. Earlier studies concluded that earthworms were able to increase soil N₂O emissions through the physical modification of the soil structure (Drake and Horn, 2007) and also due to the release of 0 to 11 nmol N₂O g⁻¹ h⁻¹ from their bodies (Horn et al., 2006).

1.4 Nutrient losses to water

1.4.1 Impact of drainage on soil hydrology

The hydrologic cycle is made up of two phases the first being the atmospheric phase that involves movement of gas (water vapor) and solid/liquid (snow/rain). The second phase is the terrestrial phase, which is about movement principally of water in, over, and through the Earth. The terrestrial phase can be broken into the groundwater phases involving aquifer recharge, percolation, and infiltration and surface water phase that consist of streamflow and runoff.

After precipitation, the amount of water that infiltrates the soil depends on various factors that include soil texture, moisture content, bulk density, porosity, permeability, and presence of restrictive layers. Landscape scale, climatic, and land management factors can also affect infiltration. For example, frozen and compacted soils have lower infiltration. The infiltrated water can end up being taken up by plants or can flow further due to the pull of gravity to be stored in an aquifer or can flow further to a surface water body such as a lake, river, or a stream where it can be turn into water vapor to start the process again (Easton, 2015).

Soil hydrology, how the water moves through the soil profile, can be affected by soil drainage. The effects of drainage on hydrology depend on soil properties, vegetative cover, topography, growing season, climate, and soil drainage intensity and system (spacing between drain pipes and type of artificial drainage system). A number of studies have been done on the impact of drainage on soil hydrology. For example, Rahma et al. (2013) studied the applicability of the

Soil and Water Assessment Tool (SWAT) in examining subsurface drainage in a cold environment. Authors s

tudied the potential impact of extensive subsurface drainage development in the Red River Valley (RRV) on streamflows. In the study, Rahma et al. (2013) found out that extensive subsurface drainage in the RRV would likely increase the amount of small peak flows and likely reduce the amount of large peak flows. They attributed this to the reduction of the flow volumes and not the increase in the time-to-peak hydrograph. Rahma et al. (2013) also noted that extensive subsurface drainage could cause movement of more water in the surface (soil surface runoff) to the river in the autumn leading to an increase in the soil storage capacity for water during the winter. Later on, Muma, Rousseau, and Gumiere (2016) did a study on the impact of subsurface agricultural drainage on soil water storage and flows of a small watershed and found that subsurface drainage raised base and total flow and reduce peak flows, in terms of volume and number of flow events.

1.4.2 Overland flow and drainage flow

Overland flow is part of the hydrology cycle and occurs after precipitation. It is mainly influenced by a number of factors such as soil type. In general, overland flow can occur when rainfall exceeds the infiltration capacity of the soil, leading to excess water running off the soil surface. Infiltration excess runoff is common in compacted soils or soils with low porosity such as clay soils. The overland flow can also occur when the soil surfaces reaches the saturation point; the soil has no further capacity to hold water. In this case, the predominant variables regulating the process are depth and topography (Easton, 2015). Depending on the type of overland flow, there will be a different impact on the soil nutrient losses.

The water that has not been lost through the overland flow percolates through the soil, recharges the water table. If the soil has been artificially drained part of this water will reach the drain conduit and be discharged from the soil profile as drain water. Hence, after sub-surface drainage installation the water movements across the soil profile such as the recharge of water table will be no longer controlled by the soil properties only, but also by the drain depth and drain spacing (Kamara et al., 1999, Bernd Lennartz 2011).

1.4.3 Soil nitrogen losses

The nitrogen balance in agroecosystems is important for environmental and soil quality and sustainability of agriculture. The N can be added to the soil from fertilizer and manure, biological N fixation, irrigation water, and atmospheric depositions. However, it can be removed from the agroecosystem in products and through leaching, volatilization of ammonia, surface runoff, denitrification, erosion, and trace gas emissions (Sainju, 2017). A recent study of a pasture-based dairy production system by Burchill et al., (2016) showed that only between 15 and 35% of the N entering the grassland based production livestock systems was converted to tradable agriculture products. From the N surplus, almost 50% was lost as various gasses (N_2 , N_2O , NH_4) and the remainder was lost to water. Improving hydraulic conductivity by drainage will impact on the overall N losses to water.

1.4.3.1 Nitrate losses to water

Nitrate is soluble in water and mobile, so it can be lost from the agroecosystem through drainage water. Nitrate is typically taken up by plants when it occurs within the root-zone. However, when there is a surplus it can be carried by water to below the root-zone and reach the ground water table or drainage systems. Nitrate leaching from agroecosystems has been a matter of growing concern around the world. A study by Piccini et al. (2016) stated that most of the European agricultural sectors contribute more than 50% of total nitrate leached to groundwater within their regions.

Soil texture and porosity has also been reported to have an impact on nitrate leaching. That is partly because the degree of aggregation of the soil will have an impact on the downward movement of water through the soil profile. Previous work by Su et al. (2014) studied the nitrate leaching in different managed sandy soils with different silt+clay contents. They established correlations between different soil textures and N leaching, finding significant correlations between soil nitrate concentration and silt and clay content.

Thus there is a potential to increase soil nitrates losses when the water flow changes. Hence it is important to monitor and control the nitrate losses to water. Nitrate becomes an environmental pollutant when it reaches high concentrations on fresh water bodies such as rivers and lakes. The EC Nitrates Directive 91/676 is specifically aimed at reducing pollution of water by nitrates originating from agricultural sources and to prevent further such pollution. It follows a previous Directive 75/440, which fixed the maximum allowed levels of nitrate in drinking water at 11.3 mg L^{-1} in the EU. Furthermore, safe levels for human consumption should not exceed 22.6 mg L^{-1} according to the World Health Organisation (WHO) (European Council, 1991).

1.4.3.2 Ammonium losses to water

Ammonium in the soil can result from the hydrolysis of urea, application of synthetic fertilizer, and mineralisation of the SOM (Whitehead, 1995). Usually it is present in the topsoil and is leached down through the profile. The solubility and mobility of ammonium will highly depend on the soil temperature and the concentration. Under low temperatures and high concentration rate where ammonization and ammonification are inhibited ammonium has been proved to be highly soluble. Moreover NH_4^+ ion is positively charged and therefore can easily bind with the negatively charged clay particles and the negative exchange site of SOM (Feigenbaum et al., 1994; Whitehead, 1995; Stark and Richards, 2008b). This will provide some degree of protection towards the susceptibility of rapid nitrification and subsequent leaching.

In clay soils particles such as mica and vermiculite are capable of fixing NH_4^+ . This is done by replacement of cations in the interchangeable zones. However this process is strongly influenced by the soil moisture content and the soil temperature, therefore the stability of the NH_4^+ binding will be controlled by wetting-drying and freeze-thawing cycles. Although the agricultural fixation of NH_4^+ is not very important, for certain soils, it can play a key role and give insights of the underlying biochemical process (Walsh et al., 1960; Havlin et al., 2005).

1.4.3.3 Dissolved organic nitrogen losses to water

Dissolved organic nitrogen (DON) is defined as the organic N present in the soil solution, collected without using extractants (Murphy et al., 2000).

In the soil, there are two pools of DON (Yu et al., 2002; Jones et al., 2004). The first one is the labile DON pool, comprised of low molecular weight DON (LMW-DON); formed by free amino acids, amino sugars and proteins (Stevenson, 1982; Antia et al., 1991), it is turned over rapidly by microorganisms, therefore it doesn't accumulate in the soil. Therefore, a low concentration of amino acids in the soil solution indicates that labile DON in the soil is rapidly removed from solution by either roots or microorganisms preventing its accumulation (Jones et al., 2004). The second pool is a high molecular weight DON (HMG-DON), rich in complex humic substances that are resistant to microbial attack (Peierls and Paerl, 1997; Stepanauskas et al., 1999, 2002; Marschner and Kalbitz, 2003; Jones et al., 2004). It has a slow turnover and represents the major DON loss to water.

In agriculture soil the concentrations of DON vary between 25 $\mu\text{g N L}^{-1}$ and 10 mg N L^{-1} (Lawes and Gilbert, 1881; Watson et al., 2000a; Perakis and Hedin, 2002; Siemens and

Kaupenjohann, 2002; Jones et al., 2004; Vinther et al., 2006) accounting for the 0.1-3.0% of the soil total N (Haynes, 2005).

1.4.4 Soil phosphorus losses

Phosphorus (P) is a naturally occurring element in the minerals, organic matter, water and living organisms in the soil and is required by plants for growth and development.

Surplus P in agroecosystems can be an environmental hazard. For crop production, there can be economic benefits from increasing phosphorus fertilization. However, excessive soil P fertilization can be carried to water bodies leading to increase in the growth of aquatic plants such as algae resulting in depletion of oxygen and consequently the destruction of aquatic ecosystems. The reduction in oxygen and the toxicity of certain algal blooms can cause death of fish and harm to wildlife and livestock (Liu et al., 2009).

Unlike nitrogen, P is the least mobile of most plant nutrients. Soil P, in general, is strongly held in the soil matrix and would not readily leach out in the water drains (Li et al., 2017). However, more recent studies showed that soil P can also be lost through runoff and leaching. The latter studies have drawn the attention to the fact that a large proportion of the area sorbing P are the smallest particles, such as clay, that can easily become detached from the soil matrix (McGechan, 2002). Some authors from recent studies agree that most of the P losses from soils are mainly through overland flow (Hart et al., 2004; Sharpley et al., 1999). A recent study by Parvage et al., (2010) found that grazing significantly increased soil P losses in the runoff water. Particularly, grassland systems that are rich in P at the soil surface due to the fact that it is a perennial crop and the soil surface is not regularly inverted. Furthermore, Li et al. (2017) found that a large amount of P is lost through runoff especially when there are higher volumes of runoff. However, it is now becoming clear that an important mechanism leading to P leaching to drains is the sorption onto mobile suspended colloidal soil particles. The leaching is promoted by the larger pores of the soil 'macropores' that impose very little restriction and allow rapid flow on the above mentioned mobile particles (Figure 1.9). Moreover, it has been observed that in clay soils the rapid flow takes place under very wet soil conditions (McGechan, 2002).

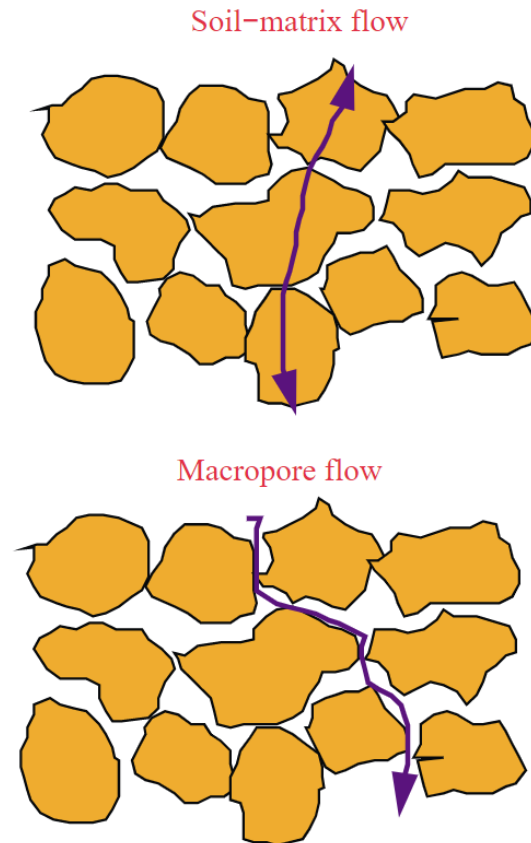


Figure 1.9 Water and particles movement in soil-matrix pores and macropores of aggregated soil; soil-matrix flow leads to straining of filtration of particles (colloid-sorbed) P particles; macropores flow occur when inter-aggregate pore spaces are water filled, P pass rapidly through soil without restrictions (McGechan 2002).

This is in agreement with the study conducted on a grassland site in the UK by Hooda et al., (1999) where high P loads were found in the field drains during the first two weeks after slurry applications during the winter, this can only be accounted by macropore flow. Similar results were found by Zhang et al., (2015) where authors concluded that long-term grasslands may become a significant P source in tile-drained systems when they receive regular P addition (Zhang et al., 2015).

Overall, depending on the P losses pathway the predominant soil P particle lost will differ among: dissolved reactive P (DRP), particulate (PP), dissolved unreactive P (DUP) and organic P.

1.4.4.1 Dissolved reactive phosphorus (DRP)

Dissolved reactive phosphorus is readily available P for plants. Most P enters water bodies attached to sediments but end up dissolving to become available for growth of aquatic plants and algae. The concentration of DRP indicates that a water body can support algae and plant growth.

1.4.4.2 Dissolved unreactive phosphorus (DUP)

Dissolved unreactive P does not react with phosphorus reagents. They are usually measured as the difference between DRP and soluble phosphorus. Particulate phosphorus (PP)

Particulate phosphorus (PP) is P fixed to soil particles. The amount of P that is transported through surface runoff is dependent on the characteristics of the soil. Soils that are highly susceptible to erosion cause transport of large amounts of PP. In a recent study on P losses at a catchment level by Ockenden et al., (2017) authors reported that extreme weather events led to an increase of 30% on losses of P and particularly of PP.

1.4.4.3 Impact of drainage on P losses

The amount of P losses is determined by factors such as the amount and intensity of rain, antecedent conditions, tillage and cropping, soil texture, and drainage systems design (Moore, 2016, p.5). In general, sub-surface artificial drainage increases the discharge of water from the soil profile and reduces runoff (McDowell et al., 2001). As mentioned above, in agriculture most of the P is being lost through runoff. Hence, sub-surface drainage will reduce the overall P exported by lowering the runoff. However, the DRP will be susceptible to be lost through the drain channels. For this reason, drained land tends to have high concentration of DRP than any other form of P (McDowell, Gongol, and Woodward, 2012).

1.5 Mole and gravel mole drainage installation on site

Prior to introduce the experimental Chapters in this section it is summarized the treatment installation on site. Accompanied by pictures of the site that will illustrate how the measurement from the coming Chapters were carried out.

1.5.1 Location of experimental site A and B

Sites were located at north-east site of the Solohead Research farm (further details 2.3.2). Both site were separated by a drain ditch, the brown line shoed in Figure X.1.

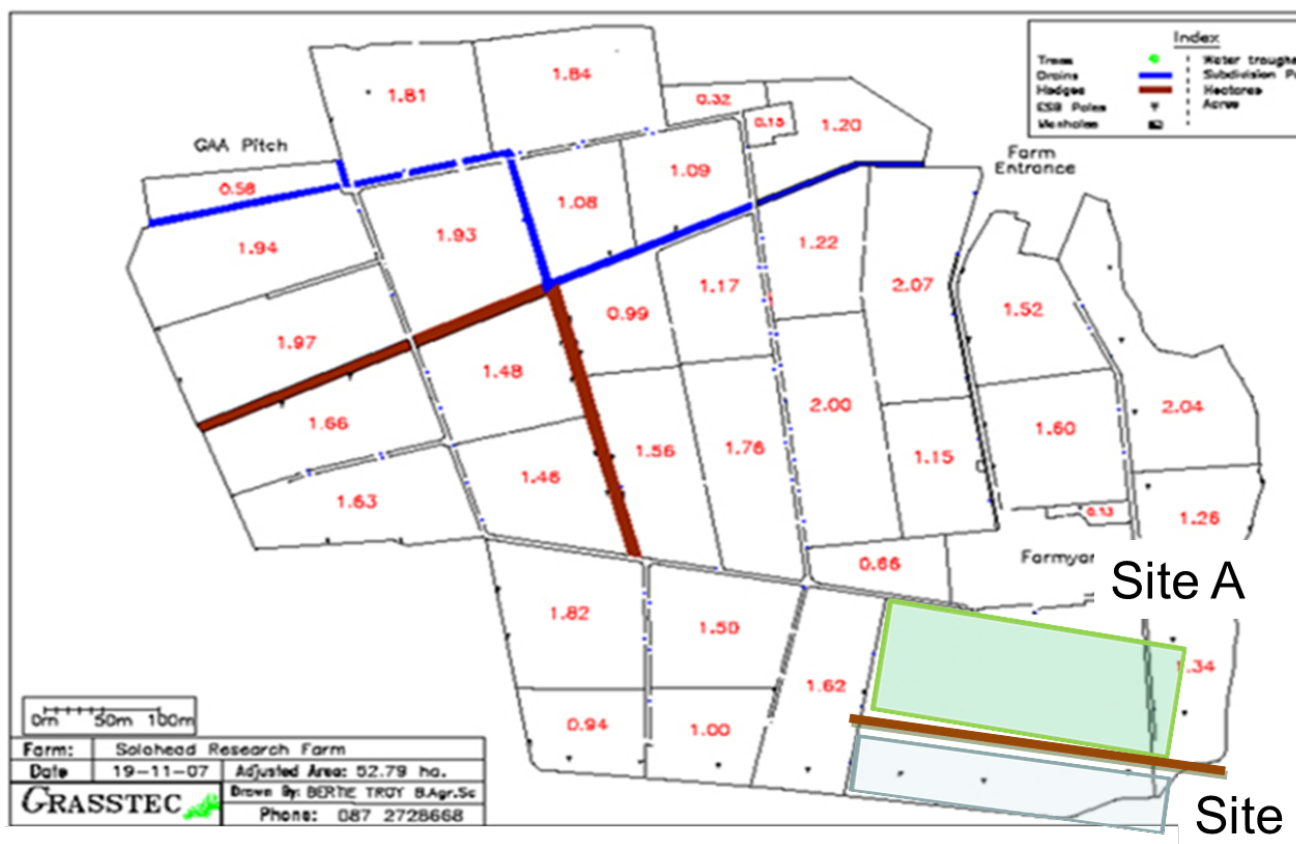


Figure X.1.
Solohead
farm.
Experiments
carried out on
and site B,
by drain ditch
The drainage
excavated in



Map of
Research

were
site A
separated
(brown).

ditch was
order to

collect the water flow from the drainage water and subsurface water, and this discharged both into the river (Figure X.2)

Figure X.2. Drain ditch

Both, Site A and Site B were hydrologically isolated by trenches of > 2 m deep backed filled with stones (Figure X.3) on 3 sides. This was done in order to avoid that water coming from other parts of the farm into the experimental site.



Figure X.3. Hydrological isolation trench

1.5.2 Experimental Site A

1.5.2.1 Treatment and site implements

On site A there were 4 treatments and plots were isolated from each other with isolation berms (0.4 m high), that were also used to collect the water flowing from the surface to get discharged Figure X.4, further details are described in section 2.3.2.



Figure X.4. Isolation berms

Mole drainage treatment consisted on creating a mole channel 40 cm from the soil surface with a torpedo pugged on a tractor. As the channel was created in the soil surface fissures from the channel to the top soil were also generated. Those will allow the water to percolate and be discharged by the mole channel. Figure X.5 shows how the mole channel is created and how the fissures look like in a soil horizontal section.

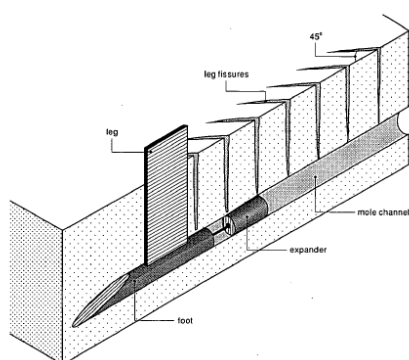


Figure X.5. Mole drainage treatment

Similarly, gravel mole drainage is undertaken using the same techniques as described above. However in this instance the fissures created are backed filled with gravel in order increase the operational lifespan of the drainage systems because those can become clogged with soil and stop discharging water (Figure X.6).



Figure X.6. Gravel mole drainage

1.5.2.2 Water flow and sampling measurements

As mentioned above, the isolation berms between plots also allowed to collect the subsurface water. The water was channel to a stone V (Figure X.7) and was then piped into a collection tank (Figure X.8)



Figure X.7. Stone V that channels the surface flow

For each plot there were two collection tanks from which water samples were collected and analysed for P and N losses to water (more details are provided in Chapter 2 section 2.4.7), except for the undrained that only had surface flow. As shown in Figure X.8, the tanks were fitted with flowmeters that allowed the water flowing from the site to be measured every 15 minutes.



Figure X.8. Measurement tanks and flowmeters

1.5.3 Experimental Site B

On Site B there was only one drainage treatment applied. In this case was more extensively drained than on Site A (further details are provided in Chapter 3 section 3.3.2). Gravel mole drainage was installed across collector drains filled with stone (Figure X.9).

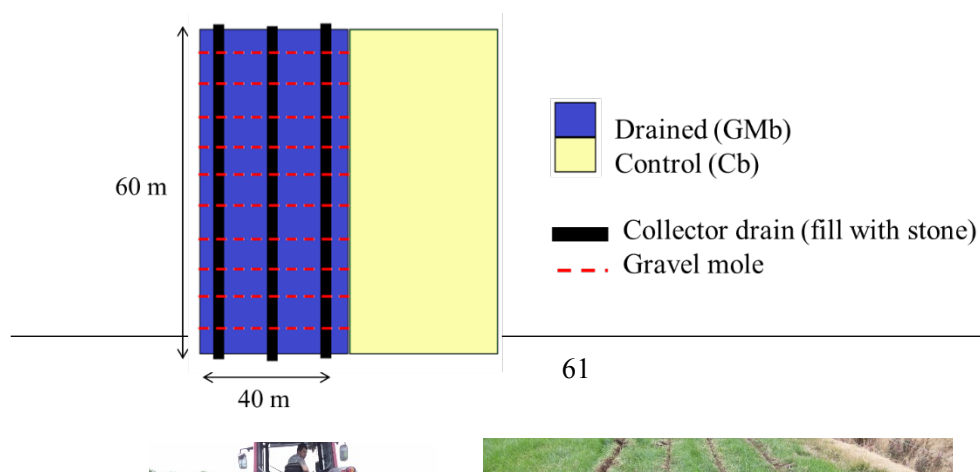


Figure X.9. Site B, layout and drainage installation

CHAPTER 2: PHOSPHOR AND NITROGEN LOSSES FROM TEMPERATE PERMANENT GRASSLAND ON CLAY-LOAMY SOIL AFTER INSTALLATION OF ARTIFICIAL DRAINAGE

Based on: Valbuena-Parralejo, N., Fenton, O., Tuohy, P., Williams, M., Lanigan, G. J., & Humphreys, J. (2019). Phosphorus and nitrogen losses from temperate permanent grassland on clay-loam soil after the installation of artificial mole and gravel mole drainage. *Science of The Total Environment*, 659, 1428-1436.

2.1 Abstract

Mole (M) and gravel-mole (GM) drainage systems improve the permeability of soils with high clay contents. They collect and carry away infiltrating water during episodic rainfall events. Characterisation of nutrient fluxes (concentration and flows) in overland flow (OF) and in mole drain flow (MF) across sequential rainfall events is important for environmental assessment of such drainage systems. The objective of this study is to assess the impact of drainage systems on soil nutrient losses. Three treatments were imposed on grazed permanent grassland on a clay loam soil in Ireland (52°30'N, 08°12'W) slope 1.4%: undrained control (C), mole drainage (M) and gravel mole drainage (GM). Plots (100 m x 15 m) were arranged in a randomised complete block design with four replicated blocks. Nitrogen (N) and phosphorus (P) concentrations in OF, MF and groundwater (GW) were measured from each plot over 15 consecutive rainfall events. The results showed that M and GM ($P < 0.05$) deepened the watertable depth and decreased OF. M and GM increased losses of nitrate-N (22%) and ammonium-N (14%) in GW. Nitrate-N concentrations from all the flow pathways (mean and standard error (s.e.): 0.99 s.e. 0.10 mg L⁻¹) were well below the 11.3 mg L⁻¹ threshold for drinking water. Ammonium-N concentrations from all the flow pathways (mean: 0.64 s.e. 0.14 mg L⁻¹) exceeded drinking water quality standards. On the other hand M and GM lowered total P losses (mean annual losses from C, M and GM: 918, 755 and 853 s.e. 14.1 g ha⁻¹ year⁻¹) by enhancing soil P sorption. Hence M and GM can be implemented on farms under similar management to that described in the present study with a minor impact on N (increased concentration on averaged 18% to GW) and P (reduced by on averaged 114 g ha⁻¹ year⁻¹).

2.2 Introduction

Mole drainage (M) is widely used internationally as a shallow drainage technique in soils exhibiting low permeability in upper horizons (Burke et al., 1974; Haygarth et al., 1998; Ibrahim et al., 2013; Sharpley and Syers, 1979; Tuohy et al., 2015). Mole drainage can improve the permeability of upper soil horizons and can thereby increase the rate of water discharged through the soil profile (Tuohy et al., 2015). In Ireland, many dairy farms are located in high rainfall areas, dominated by poorly-permeable soils and low evapotranspiration rates, which results in poor trafficability by livestock and machinery after rainfall events, especially in spring and autumn (Fitzgerald et al., 2008; Keane, 1992; Tuohy et al., 2016). Achieving a long grazing season on permanent grassland is key to profitable ruminant livestock production in many temperate grassland regions (Dillon et al., 2005; Shalloo, 2009). Therefore, mole and gravel

mole drainage (GM) are land management techniques that improve trafficability and, ultimately, profitability on farms.

Although phosphorus (P) and nitrogen (N) are present in the soil, nutrient losses in overland flow and flow emanating from mole drains highly depend on soil permeability (McDowell et al., 2001, Ibrahim et al., 2013). Moreover, areas with high soil nutrient contents along with high volumes of runoff and drainage are recognized as being of high risk of nutrient loss to water (Haygarth et al., 1998; Peukert et al., 2014; White et al., 2009). Previous research has shown that installation of M or GM drains can change the volume of flows between overland flow and mole flow (Tuohy et al., 2016; Tuohy et al., 2015). Hence to fully assess the impact of installation of mole or gravel mole drains there is a need to make an integrated assessment of N and P losses encompassing the various loss pathways including overland flow, mole flow and loss to groundwater.

The objective of this experiment was to quantify the effects of mole and gravel mole drainage systems on losses of P and N in (i) overland flow, (ii) mole flow and (iii) ground water flow under a grass-clover based dairy production system over 15 episodic rainfall events during one year. Phosphorus and N species were investigated in terms of mean concentration (mg L⁻¹) and mean fluxes (g ha⁻¹h⁻¹) from the following treatments: un-drained control (C); mole drainage (M) and gravel mole drainage (GM).

2.3 Materials and methods

2.3.1 Study site

This experiment was conducted at Solohead Research Farm (52,30'N, 08,12'W), on a gently sloping (1.4%) site (2.5 ha) bounded by the river Pope. The average annual rainfall from the preceding 10 years (2004 to 2013) was 1017 mm with annual potential evapotranspiration (calculated using the model by Schulte et al., 2005) of approximately 510 mm. During the study period, 31 March 2014 until 30 March 2015, total annual effective drainage was 565 mm, occurring on 260 days (due to the lag time there is not effective drainage every rainfall day) and the 73% between November and March. The soils were poorly drained Gleys with a clay loam texture; Subsoil was quaternary till with a shallow watertable (0 to 2.2 m below ground level) (Tuohy et al., 2016). The soil type was grouped under the Elton soil association (Creamer et al., 2014). Soil physical and chemical properties are presented in Table 2.1.

Table 2.1 Soil properties at Solohead Research Farm at four different depths. All values are means of at least three determinations. Bulk density (BD), hydraulic conductivity (HD), total nitrogen (TN), total carbon (TC), soil organic carbon (SOC).

Depth (cm)	Sand (%)	Silt (%)	Clay (%)	BD (g cm ⁻³)	HD (mm h ⁻¹)	Total porosity (%)	pH	TN (t ha ⁻¹)	TC (t ha ⁻¹)	Total inorganic C (t ha ⁻¹)	SOC (t ha ⁻¹)
0 to 10	40	29	29	1.01	0.0012	61.71	6.5	12.7	119.7	2.4	117.3
10 to 30	41	20	39	1.05	0.0020	60.32	6.8	8.8	83.1	2.5	80.6
30 to 60	34	20	46	1.21	0.0016	54.31	7.5	3.3	7.2	0.7	6.5
60 to 80	11	31	58	1.33	0.0028	49.81	7.5	2.4	6.8	0.8	6.0

Prior to 1996 there were an excess to agronomic response nutrient inputs to this site applied as dairy manure. Although these applications ceased in 1996 and there were no subsequent applications of P and K fertilizers there remained a legacy of high soil P and potassium (K) concentrations at this site (0 to 10 cm depth). Following extraction using Morgan's solution (Na acetate + acetic acid, pH 4.8), mean soil Morgans P concentrations (SMP) was 27.4 (s.e. 1.12) mg L⁻¹ and mean soil Morgans K concentrations (SMK) was 184 (s.e. 24) mg L⁻¹. To put these concentrations in context, no agronomic response is expected to SMP >8.0 mg L⁻¹ and SMK >150 mg L⁻¹ (Coulter et al., 2008).

2.3.2 Experimental layout and design

The experimental site, layout and design has been described previously by Tuohy et al. (2016) and Valbuena-Parralejo et al., 2019. Three of the four treatments described on previous studies were used in the present study. They were an un-drained control (C), mole drainage (M) and gravel mole drainage (GM). In August 2010 a large open V-shaped drain (2 m depth and 4 m wide at ground level) was excavated along a natural depression in the site running along a west-east axis with water exiting into the river Pope from the eastern end of the open drain (Figure 2.1). All the runoff and drainage flow discharged to the open drain and then to the river Pope. The experimental site was hydrologically isolated from external lateral water movement by an isolation ditch, comprising a 1 m deep trench filled to the soil surface with stone aggregate (Figure 2.1) that ensured to capture the water coming from other parts of the farm. In January 2011 the site was divided into four blocks (each 60 m-wide, 100 m-long). Each block was sub-

divided into four 15 m wide plots. To avoid lateral surface flow between adjacent plots, each plot was flanked by berms consisting of a 0.2 m deep trench and 0.30 m high embankment of compacted soil. Treatments were assigned to plots in a randomized complete block design with four replications.

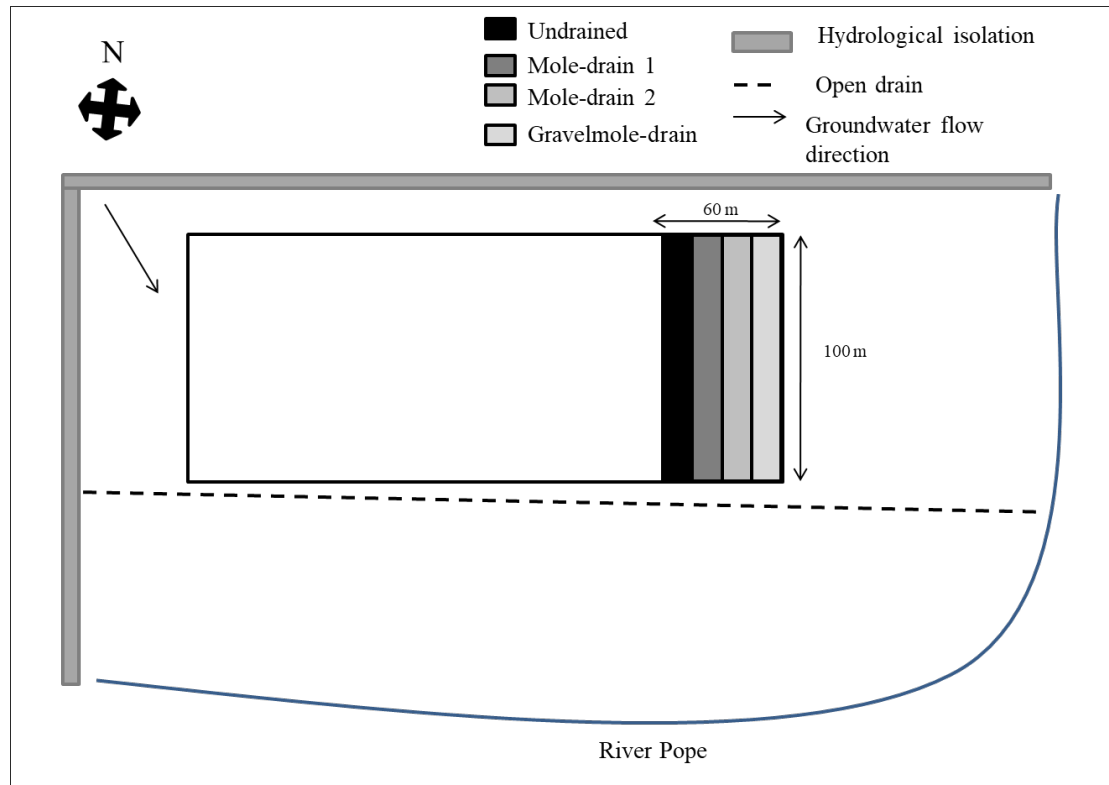


Figure 2.1 Location of the study area, experimental sites, groundwater flow direction and open drain. North is upslope position and south is down slope. Hydrological isolation consisted on a 1 m deep by 0.5 m wide ditch filled with gravel (20-30 mm grade). Data presented in the currents study was from treatments, uncrained (C), mole drain 2 (M) and gravel mole drain (GM).

2.3.3 Treatment installation

Treatments were imposed in July 2011. Treatment M was installed using a simple mole plough at a depth of 0.55 m. This installation depth ensured that the mole channels were located in a heavy, plastic, poorly permeable and stone-free layer of the soil. All the latter soil properties were required in order to succeed on the mole channel formation and stabilisation. The distance between each mole channel, laid parallel to each other, was 1.2 m.

Treatment GM was installed using a gravel mole plough. It consisted on the same principle as the simple mole plough but back-filled with gravel. The GM was installed at a depth of 0.40 m, which was the maximum attainable by the implement used. The distance between each channel

was 1.2 m. The gravel mole plough was set to install a 0.20 m high column of clean gravel aggregate (10 to 20 mm diameter) along the formed channel (Figure 2.2).

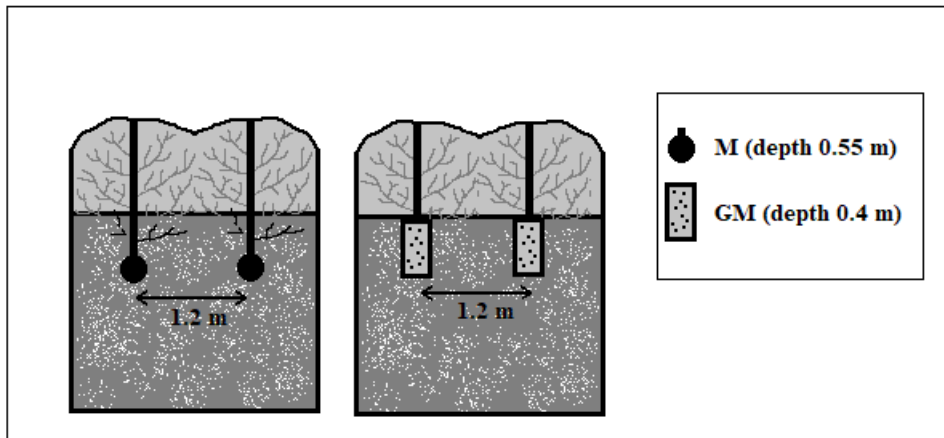


Figure 2.2. Soil vertical section of Mole (M) and Gravel mole (GM) drainage treatment drainage systems.

Treatment M consisted of unlined channels formed in subsoil by pulling a cylindrical torpedo-shaped foot through the subsoil. The torpedo-shaped foot was attached to a blade-shaped leg that was connected to the three-point linkage of the tractor. A plug (or expander) was attached to the end of the foot, which helped to compact the channel wall. Treatment GM was based on the same principal except that the channel created in the soil was immediately back-filled with gravel flowing from a hopper on top of the mole-plough down into and out through the leg and foot of the mole-plough.

Both M and GM treatments were installed to the full length of the plots (100 m). Further details are available in Tuohy et al. (2016). There were 11 mole or gravel mole channels installed in each of the drained plots. In order to avoid border effects, measurements were conducted on flows from the outlets of the five central mole channels in each drained plot (see Figure X.8).

Two sub-plots (2×11 m) for watertable depth (WTD) measurements were established along the long axis of each plot. One sub-plot was established in the upper half (up slope) and the second in the lower half (down slope) of the plots. Two piezometer, for WTD measurements, were installed in each sub-plot. Details of the piezometers are provided below.

2.4 Experimental measurements

2.4.1 Meteorological data

An automated weather-station (Campbell Scientific Ltd. Loughborough, UK) recorded rainfall, air temperature, soil temperature, wind speed, wind direction and solar radiation (resolution 30 min). The hybrid grassland model of Schulte et al. (2005), assuming poorly drained soil criterion, was used for estimation of the soil moisture deficit and effective drainage. There were calculated as the difference between precipitation and estimated evapotranspiration. The soil moisture deficit of a certain day was calculated in the model as soil moisture deficit of the previous day (both in mm) minus the actual rain plus the evapotranspiration (mm/day) and plus the amount of water drained (mm/day). The evapotranspiration was expressed as a function of the potential or reference crop evapotranspiration and the current soil moisture deficit. Equations are described by Schulte et al. (2005) and all values are estimated for the root zone, hence the first 15 cm of soil depth.

2.4.2 Water table depth

Two fully screened piezometers (HDPE pipes, internal diameter 19.6 cm - Eijkelkamp, Agrisearch Equipment, Giesbeek, The Netherlands) were installed in each sub-plot. The piezometers were installed 3 m below ground level using a soil pneumatic drill (Giddings Machine Co, Windsor, USA). Near the soil surface, the area between the soil and the pipe (0.03 m) was filled with bentonite to prevent water channelling down the outside the piezometer. The WTD readings was measured from all the piezometers on weekly basis using an electronic dipper (Marton Geotechnical Services Ltd, Suffolk, U.K.).

2.4.3 Plant biomass production and N uptake

The grass-clover swards were rotationally grazed by lactating dairy cows at approximately monthly intervals during the main grazing season (March to November). The grazing rotation length varied from 21 days in late spring/early summer to 42 days in autumn. Cows were allowed to graze the swards when the sward height, measured using a Filips rising plate meter (www.grasstec.ie), reached 10 cm above ground level. Cows were removed from the plots when the sward height was 4 cm above the ground level. The swards received no inorganic fertiliser input; white clover (*Trifolium repens*) in the pasture was the main source of plant-available N via biologically fixed N.

Herbage production was measured from an area of 10×1.2 m within each plot. The herbage was harvested (Etesia UK Ltd, Warwick, UK) and weighed before each grazing. A sub-sample of 100 g was dried for 16 h at 100 °C to determine the herbage matter (DM) content. A second 100 g sub-sample was stored at -20 °C before being freeze dried, milled to pass a 0.2 mm sieve and analysed for total N using a LECO 528 auto-analyser (LECO Corporation, St. Joseph, MI, USA). Uptake of N in herbage was calculated by multiplying herbage DM yield by the N concentration in harvested herbage for each sampling date; annual N uptake was calculated by summing values over one year.

2.4.4 Soil fertility

Plant-available P and K concentrations were determined for each plot before the experiment started. Fifteen soil samples were taken to 10 cm depth for each plot and extracted in Morgan's solution (1:5 (v/v) soil:solution ratio with a 10% sodium acetate solution buffered at pH 4.8).

2.4.5 Flow event delineation and antecedent conditions

A period of 12 hours without rainfall was used to separate one rainfall event from another, similar to other studies (Ibrahim et al., 2013; Kurz et al., 2005; Tuohy et al., 2015). The mole and gravel mole drains were only transmitting water during and shortly after rainfall events. Therefore the start and the end of each flow event was clearly defined as the periods when water was not flowing from the mole drains outside the timeframe of rainfall events.

A total of 15 rainfall events were captured and analysed in this study (31 March 2014 until 31 March 2015) (Table 2.2).

Table 2.2 Pre-event and event rainfall characteristics of the 15 selected rainfall events.

Event (dates)	Event duration (h)	Rainfall during event (mm)	7-day antecedent precipitatio n (mm)	30- day antecedent precipitatio n (mm)	Max intensity (mm/h)	Mean intensity (mm/h)	ED (mm)
1 (08/05/2014 to 09/05/2014)	28	11.2	4.2	68.8	5.2	0.42	10.4
2 (06/06/2014 to 07/06/2014)	32	4.2	7.7	91.0	1.6	0.18	3.1
3 (09/06/2014 to 10/06/2014)	37.5	11.6	11.8	113.2	5.6	0.52	12.0
4 (26/06/2014)	19.5	10.2	3.2	40.8	4.4	0.52	8.6
5 (07/07/2014 to 08/07/2014)	21	6.0	3.8	69.8	11.2	0.34	4.0
6 (19/07/2014)	17	18.6	12.0	71.7	13.6	1.1	15.3
7 (02/08/2014 to 03/08/2014)	22	8.0	4.0	76.8	3.6	0.36	0.7
8 (05/08/2014)	18	7.4	17.2	38.0	2.0	0.42	5.8
9 (06/11/2014)	13.5	15.8	22.4	104.4	12.4	1.17	5.7
10 (13/11/2014 to 14/11/2014)	38	45.8	43.0	121.0	7.2	1.22	45.6
11 (07/12/2014 to 08/12/2014)	29	16.4	3.2	105.6	2.4	0.56	4.7
12 (10/12/2014 to 12/12/2014)	28.5	29.4	14.4	120.5	3.6	0.98	27.9
13 (14/01/2015)	11	18.0	19.0	89.7	5.2	1.62	15.3
14 (15/02/2015)	10	5.4	23.6	59.0	2.4	0.38	4.2
15 (12/03/2015)	14	15.0	12.6	102.8	2.4	1.06	14.0

2.4.6 Overland and mole drain flows

The overland flow from each plot was collected through the lateral berms and channelled towards the lower end of each plot. The berms were piped towards a specific overland flow measurement tank for each plot. Likewise mole flow from the five central channels was piped to a specific mole flow measurement tank for each plot with Mole and Gravel Mole drainage channels. Each measurement tank (1.0 x 0.6 x 0.6 m, Carbery Plastics, Clonakilty, Co. Cork, Ireland) contained a v-notch weir through which overland flow or mole flow flowed towards the

monitoring tank. To monitor flow rates each tank was fitted with a Sigma area/velocity probe connected to a Sigma 920 flowmeter (HACH Company, Maryland, USA). Flow rate from each tank was measured continuously with a resolution of 15 min. Mole flow results were adjusted to total plot area taking into account for the mole flow from five of the eleven Mole and Gravel Mole channels.

The ground water recharge was calculated based on the estimated effective drainage (Schulte et al. 2005) and the measured overland flow and mole flow. Ground water flow (mm) was calculated as: effective drainage – [overland flow + mole flow] = ground water flow.

2.4.7 Water sampling and analysis

Volumes of the overland flow and mole flow were measured in this study. Water samples associated with the measured volume were taken within a timeframe, approximately 180 min once rainfall started. There were a total of four overland flow measurement tanks for treatment control and four overland flow and four mole flow for treatments Mole and Gravel Mole, respectively. At every sampling occasion, throughout the rainfall event, a water sample of each flow pathway (overland flow and mole flow) was taken from each measurement tank. Hence a total of 60 samples were taken at each rainfall event (36 from overland flow and 24 from mole flow). Approximately 100 ml of water was taken from each flowing pipe using a sampling tube (Polylab Centrifuge Tube Conical Bottom, 50 ml. Sigma Aldrich, Co Wicklow, Ireland). Sampling was conducted 15 times between 8 May 2014 and 13 March 2015 (Table 2.2), which represented the total overland flow during this year.

In each plot one piezometer located in the up-slope was used to determine nutrient concentrations in ground water flow. Water samples were taken weekly, after purging the piezometers, during the rainfall event. At each sampling occasion each of the piezometers were emptied in the morning; all of the water standing in each well was pumped out using a manual hand pump and discarded. During the following 2 h the piezometers were allowed to recharge; water seeped back into the piezometer from the surrounding soil and through the piezometer screen. This 'recharged' water in each well was sampled. Approximately 100 ml of water was taken from each well using a large syringe.

From all the flow pathways (overland flow, mole flow and ground water flow) two 50 ml samples (A and B) were taken on each sampling occasion. Sample A was filtered using Whatman milipore filter paper, d=132 mm, 0.45µm, and sample B remained unfiltered. All samples were analysed for total phosphorus (TP), dissolved reactive P (DRP), ammonium-N and nitrate-N using a Thermo Konelab 20 analyser (Technical Lab Services, Ontario, Canada).

By combining the results of the analyses of the filtered (A) and unfiltered (B) samples the results for the following P species were determined using the methodology described by Ibrahim et al. (2013); i.e. TP is the sum of the dissolved (TDP) and particulate P (PP), and TDP is the sum of DRP and dissolved unreactive P (DUP). Hence, the DUP was calculated from by $TDP - DRP$. The key results presented in this study are DRP, DUP and PP.

Nitrogen and P losses (kg ha^{-1}) for each treatment were estimated by multiplying the mean N and P concentrations (mg L^{-1}) recorded on the sampling date with the volume of overland flow or mole flow (mm) between two sampling occasions. For the ground water flow pathway DRP (kg ha^{-1}) were estimated by multiplying the mean DRP concentrations (mg L^{-1}) recorded on the sampling date with the volume of ground water flow (mm) between two sampling occasions. Likewise losses of all N species (kg ha^{-1}) were estimated by multiplying the mean N concentrations (mg L^{-1}) recorded on the sampling date with the volume of ground water flow (mm) between two sampling occasions.

2.4.8 Annual nutrient losses to water

The 15 rainfall events encompassed in this study accounted for 0.56 of the annual rainfall during the study period. The extent of annual losses was extrapolated from the 15 events to allow us to compare the results of the present study with other previous studies, where losses were quantified on an annual basis.

2.4.9 Statistical Analysis

The water table depth, concentrations and fluxes (concentration $\times$ flow) (TP, PP, TDP, DRP, DUP, nitrate-N and ammonium-N) data were subjected to ANOVA using SAS 9.3 (SAS 2011, Institute Inc., Cary, NC, USA). All data were checked for normality by using residual normality and variance analysis. All data required a natural logarithmic transformation [$y = \log(x)$]. Some of the P and N species data required a different logarithmic transformation [$y = \log(x+0.01)$]. The value of 0.01 was added before log transforming the data, which was sufficient to prevent the generation of zero transformed values. The variables included in the model were treatment, sampling date and treatment by sampling date interaction. Sampling date was included as a repeated measure in the ANOVA. Extrapolated annual losses ($\text{g ha}^{-1} \text{ year}^{-1}$) of TP, PP, TDP, DRP, DUP, nitrate-N and ammonium-N, N uptake in herbage, soil P and K concentrations were also subjected to ANOVA. All data were checked for normality by using residual normality and variance analysis. The variables included in the model were treatment and replicate. For both ANOVA models a Tukey-Kramer method test (Jaccard, J. et al., 1984) was carried out to

determine differences between treatments. A statistical probability of $P < 0.05$ was considered significant for all statistical tests.

2.5 Results

2.5.1 Meteorological data

Annual rainfall during the study period was 1013 mm. Daily precipitation, ED and SMD data for the year beginning 31 March 2014 are presented in Figure 2.3; the timing of rainfall events is also indicated.

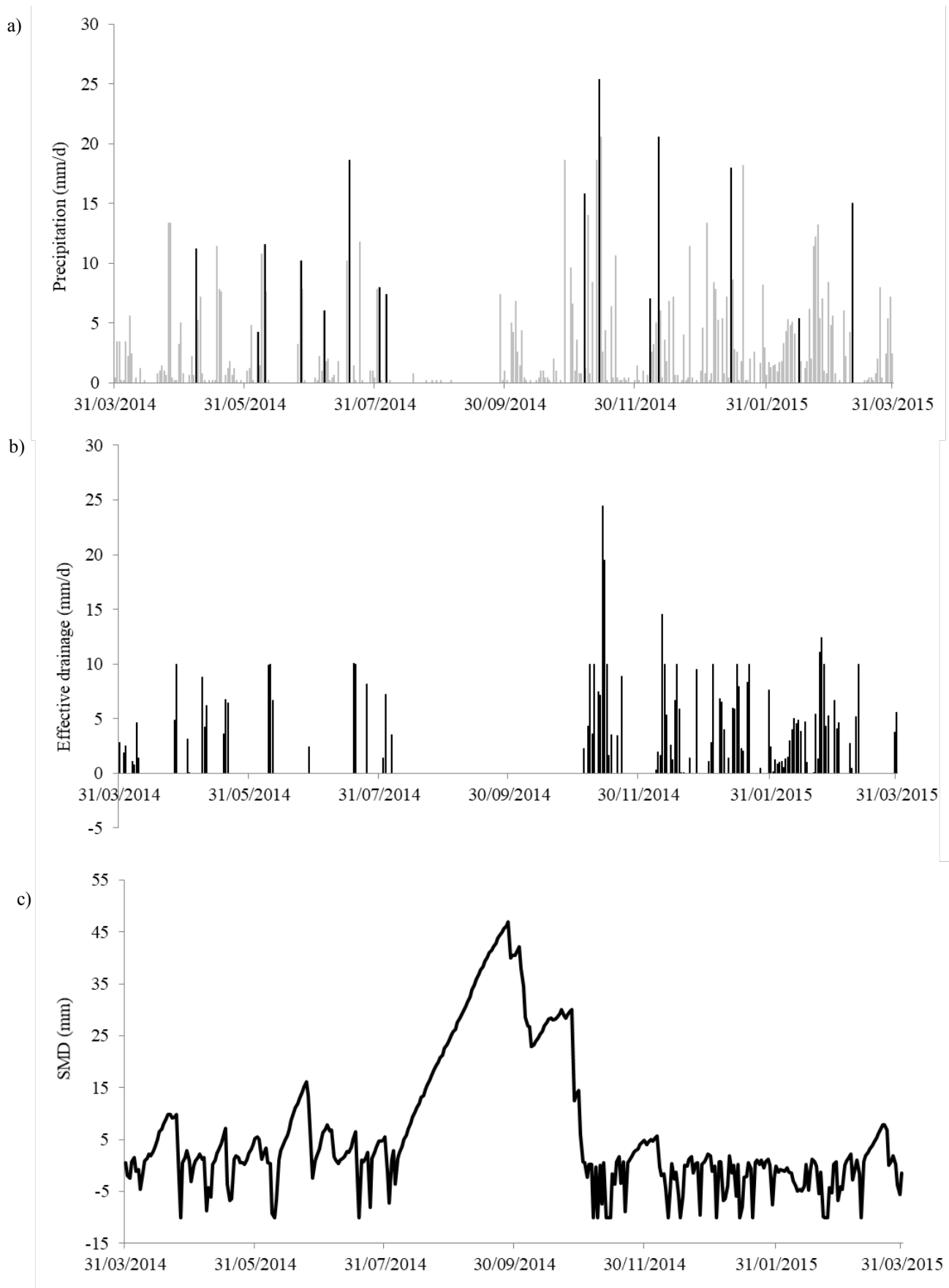


Figure 2.2 .(a) Daily precipitation, (b) Effective drainage (ED) and (c) Soil moisture deficit (SMD) values for the period 01/03/14 to 01/04/15 during which the rainfall events were selected. Lines in black in (a) indicate the start of the monitored events.

A total of 15 rainfall events were captured (Table 2.2). Of those, one event had less than 5 mm of rainfall, four had between 5 and 10 mm and rainfall during each of the other ten events was greater than 10 mm. Effective drainage during each of the 15 events ranged from 0.7 to 45.6 mm. Total event precipitation was the greatest for Event 10 (45.8 mm) and the least for Event 2 (4.2 mm).

Treatment	C	M	GM	P-	SEM [†]
				value	

2.5.2 Water table depth

The water table depth was consistently shallower ($P < 0.05$) in C than in the drainage treatments. Mean pre-event water table depth was 1.2 m below ground in C compared with 1.3 and 1.5 m below ground for Mole and Gravel Mole, respectively, (standard error of the mean (SEM) 0.021); Post-event below ground means of 0.67, 0.90 and 1.06 m were recorded for Control, Mole and Gravel Mole, respectively, ($P < 0.05$, SEM 0.033).

2.5.3 Soil fertility

There was no difference in soil Morgan's potassium concentrations between treatments. Mean soil Morgan's potassium was 184 (s.e. 24) mg L^{-1} , which was in excess of an agronomic response to applications of fertilizer potassium (100 to 150 mg L^{-1}). Similarly, there was no difference in soil Morgan's P concentrations between treatments. Mean soil Morgan's P concentrations was 27.4 (s.e. 1.12) mg L^{-1} , which was very high, being well in excess of an agronomic response to applications of fertilizer P (5 to 8 mg L^{-1}) and classified as being at risk of environmental losses in overland flow (Coulter et al., 2008). Furthermore, there was no difference in annual N uptake in herbage between treatments. Mean annual N uptake in herbage across treatments was 459 (s.e. 6) kg ha^{-1} .

2.5.4 Volumes of Flow from the flow pathways

There was a greater overland flow from Control than from the Mole and Gravel Mole, however Mole and Gravel Mole were not different from each other (Table 2.3). Mole drain flow was greater from Gravel Mole than Mole. There was no difference in flow being discharged to ground water flow between treatments. Overland flow accounted for 0.6 of the total flow from Control, which was a greater proportion than for the drainage treatments i.e. 0.36 and 0.37 from Gravel Mole and Mole, respectively (Table 2.3).

	<u>Pathway</u>		(mm h ⁻¹)				
Table 2.3 overland flow and the 15 events.	Overland flow	0.16 ^a	0.11 ^b	0.12 ^b	< 0.05	0.003	Mean flows rates in flow, mole drain discharge to groundwater over captured rainfall
	Mole Drain flow		0.09	0.13	< 0.001	0.002	
	Groundwater	0.11	0.10	0.08	ns [‡]	0.01	

2.5.5 Concentrations of P

There were no differences in TP, DRP and DUP concentrations in overland flow between treatments (Table 4). The mean concentration of PP in overland flow was lower in Mole than Control and Gravel Mole, which were not different from each other.

Table 2.4 Flow weighed mean concentration of nutrients over the captured 15 rainfall events in overland flow drain flow and groundwater.

Treatment	C	M	GM	P-value	SEM [†]
Overland flow		(mg L ⁻¹)			
TP	0.93	0.92	0.97	ns [‡]	0.02
DRP	0.62	0.66	0.63	ns	0.03
DUP	0.11	0.17	0.17	ns	0.06
PP	0.19	0.09	0.18	< 0.01	0.05
Ammonium-N	0.20	0.14	0.33	ns	0.07
Nitrate-N	0.75	0.65	0.95	ns	0.15
Drain flow		(mg L ⁻¹)			
TP		0.22	0.20	ns	0.02
DRP		0.04	0.10	< 0.001	0.01
DUP		0.08	0.05	ns	0.02
PP		0.10	0.05	< 0.001	0.02
Ammonium-N		0.65	0.58	ns	0.09
Nitrate-N		1.21	1.55	< 0.01	0.06
Groundwater		(mg L ⁻¹)			
DRP	0.01	0.00	0.01	ns	0.01
Ammonium-N	1.00	1.16	1.12	ns	0.27
Nitrate-N	0.81	1.01	1.06	ns	0.59

C = drained control, M = mole drainage and GM = gravel mole drainage. P = Phosphorus, TP = total P, PP = particulate P, DRP = dissolved reactive P and DUP = dissolved unreactive P. TP = PP+ (DRP+DUP). [†]Standard error of the mean; [‡]not significant

In mole flow there were no differences in TP and DUP concentrations between Mole and Gravel Mole (Table 2.4). The mean concentration of DRP in mole flow was higher ($P<0.001$) from Gravel Mole than Mole. In contrast, PP concentration in mole flow was higher ($P<0.001$) in Mole than Gravel Mole. There was no difference in DRP concentrations in ground water flow between treatments (Table 2.4).

2.5.6 Concentrations of N

There were no ($P<0.05$) differences in nitrate-N and ammonium-N concentrations in overland flow and in ground water flow between treatments (Table 2.4). Likewise there was no difference in ammonium-N concentrations in mole flow between treatments. In contrast, nitrate-N concentrations in mole flow were higher in Gravel Mole than in Mole.

2.5.7 Fluxes of P

The fluxes of TP ($P<0.05$) and DRP ($P<0.01$) in overland flow were higher from Control than from the drainage treatments, which were not different from each other (Table 2.5). There was no difference in the flux of DUP in OF between treatments. The flux of PP in overland flow from Control was greater ($P<0.05$) than that from Gravel Mole, which in turn was greater ($P<0.05$) than that from Mole (Table 2.5).

There was no difference ($P>0.05$) in the flux of TP and DUP in mole flow between the drainage treatments (Table 2.5). DRP losses in mole flow from Gravel Mole were greater ($P<0.001$) than from Mole. PP losses in mole flow were greater ($P<0.05$) from Mole than Gravel Mole.

Table 2.5 Mean flux.Product of concentration and flow of nutrients over the captured 15 rainfall events in overland flow, drain flow and groundwater.

Treatment	C	M	GM	P-value	SEM [†]
Overland flow		(g ha ⁻¹ h ⁻¹)			
TP	1.52 ^a	1.06 ^b	1.16 ^b	< 0.05	0.10
DRP	1.02 ^a	0.76 ^b	0.75 ^b	< 0.01	0.05
DUP	0.19	0.19	0.20	ns [‡]	0.05
PP	0.31 ^a	0.11 ^b	0.21 ^c	< 0.05	0.04
Ammonium-N	0.33 ^a	0.16 ^b	0.40 ^c	< 0.05	0.06
Nitrate-N	1.20 ^a	0.74 ^b	1.13 ^c	< 0.05	0.13
Drain flow		(g ha ⁻¹ h ⁻¹)			
TP		0.19	0.25	ns	0.04
DRP		0.03	0.13	< 0.001	0.03
DUP		0.07	0.06	ns	0.04
PP		0.09	0.06	< 0.05	0.01
Ammonium-N		0.56	0.73	ns	0.09
Nitrate-N		1.04	1.96	< 0.0001	0.06
Groundwater		(g ha ⁻¹ h ⁻¹)			
DRP	-	-	-		
Ammonium-N	1.13 ^a	1.30 ^b	0.86 ^c	ns	0.30
Nitrate-N	0.91 ^a	1.14 ^b	0.81 ^c	ns	0.28

C = un-drained control, M = mole drainage and GM = gravel mole drainage. P = Phosphorus, TP = total P, PP = particulate P, DRP = dissolved reactive P and DUP = dissolved unreactive P. TP = PP+ (DRP+DUP).[†]Standard error of the mean; [‡]not significant, - measured but very low/not detected. Different letters from each row indicate significantly different.

2.5.8 Fluxes of N

The losses of ammonium-N and nitrate-N in overland flow were lower ($P<0.05$) from Mole than from the other two treatments, which were not different from each other (Table 2.5).

In the mole flow there was no difference in ammonium N between the drainage treatments (Table 2.5). Nitrate-N losses in mole flow were greater ($P<0.001$) from Gravel Mole than Mole.

There were no differences in losses of ammonium-N and nitrate-N to ground water flow between treatments (Table 2.5).

2.5.9 Annual losses

Statistical analysis showed that the captured rainfall events were representative of annual rainfall events during this study. The mean volume of rainfall for the annual rainfall events was 10 (s.e. 1.1) mm and the mean for the captured events was 15 (s.e. 2.8) mm. The captured

rainfall events accounted for the 0.56 of annual rainfall. Hence annual losses were estimated based on the above.

The annual losses of TP ($P < 0.001$) and DRP ($P < 0.01$) across all the pathways (overland flow, mole flow and ground water flow) were the highest from Control, followed by Gravel Mole and Mole (Table 2.6). There was no difference in the annual losses of DUP and PP between treatments.

Table 2.6 Total estimated annual losses of nutrients to water, which is the sum of losses in runoff, drain flow and groundwater.

Treatment	C	M	GM	P-value	SEM [†]
	(g ha ⁻¹ year ⁻¹)				
TP	918 ^a	755 ^c	853 ^b	< 0.001	14.1
DRP	615 ^a	479 ^c	526 ^b	< 0.01	14.7
DUP	113	159	160	ns [‡]	3.6
PP	190	117	167	ns	8.5
Ammonium-N	885 ^c	1200 ^b	1225 ^a	< 0.001	8.9
Nitrate-N	1293 ^c	1767 ^b	2364 ^a	< 0.001	33.1

C = un-drained control, M = mole drainage and GM = gravel mole drainage. P = Phosphorus, TP = total P, PP = particulate P, DRP = dissolved reactive P and DUP = dissolved unreactive P. TP = PP+ (DRP+DUP). [†]Standard error of the mean; [‡]not significant. Different letters from each row indicate significantly different.

The annual losses of TP ($P < 0.001$) and DRP ($P < 0.01$) across all the pathways (overland flow, mole flow and ground water flow) were the highest from Control, followed by Gravel Mole and Mole (Table 2.6). There was no difference in the annual losses of DUP and PP between treatments.

The annual losses of nitrate-N across all the pathways were lower ($P < 0.001$) from Control than from the other two drainage treatments, which were not different from each other (Table 6). The annual ammonium-N losses across all the pathways were greater ($P < 0.001$) from Gravel Mole than Mole, which was greater ($P < 0.001$) than Control (Table 2.6).

2.6 Discussion

During this study losses of P and N were measured during 15 rainfall events and annual losses were also calculated in order to compare the results of the current study with previous studies that measured annual losses from the agri-ecosystem.

2.6.1 Phosphorus loss to water

This site had a legacy of high nutrient inputs as described above and evidenced by the high soil Morgan's potassium and soil Morgan's P concentrations. Very high soil Morgan's P concentrations combined with the high volumes of overland flow, due to the low permeability of the clay loam soil in the present study, indicates that this was a site with high risk of TP loss to water (Hart et al., 2004). Overall, annual TP losses estimated from the current study (775 to 918 g ha⁻¹ y⁻¹ Table 2.6) would be classified as low to moderate according to Jordan et al. (2012). Furthermore, the TP values in the present study were very much lower than 2000 to 3000 g ha⁻¹ year⁻¹ from mole drains under grazed grassland reported by Haygarth et al. (1998).

The transport of TP from the soil to water occurred in dissolved and particulate forms. In the present study the greatest losses were in the form of DRP accounting between 0.62 and 0.67 of TP losses to water via the three loss pathways (overland flow, mole flow and ground water flow). Overland flow was the main loss pathway for DRP, accounting for >0.85 of DRP losses across the three treatments, most likely due to the soil P content from the soil upper layers.

Installation of Mole and Gravel Mole improved water infiltration into the soil and decreased the volume of overland flow from these treatments (Table 2.3) and hence lowered the losses of DRP and TP in overland flow from these treatments. It is well established that most of the P exported from soils is mainly through overland flow (Hart et al., 2004; Sharpley et al., 1999). Furthermore, it is likely that improved infiltration associated with Mole and Gravel Mole facilitated the adsorption of DRP as it percolated through the soil profile (Chardon et al., 1997; McDowell et al., 2001), hence, lowering the losses of DRP (and TP) in mole flow and ground water flow (Table 2.6). The pH of the soils was from 6.5 to 7.5 (Table 2.1); in this pH range, the main DRP species are the H₂PO₄⁻ and H₂PO₄⁻. Under acidic conditions, phosphate sorption is pH driven (Scarseth 1935). The most likely sorption mechanism that could have happened in the current study would be the presence of metal oxides, iron and aluminium hydroxides, poorly crystalline aluminosilicates, which are found within a wide range of soil orders, as well as organo-mineral complexes (Violante et al., 2002). However we did not measured the above components. Finally, the very low concentrations of DRP in ground water flow also indicated that there might be P sorption capacity of the overlying soil layers in this study.

The DRP losses from Gravel Mole were greater than Mole. This can be explained by the improved infiltration capacity by drainage. However, a Langmuir isotherm was conducted to determine the P sorption capacity of the gravel and the surrounding soil following the methodology described by Fenton et al., (2009). Results showed that gravel had a maximum sorption capacity of approximately 0.07 mg PO₄P kg⁻¹, which was much lower than the P

sorption capacity of the soil (Djodjic et al., 2004). Hence the greater DRP losses from Gravel Mole can be partly attributed to the presence of gravel in the Gravel Mole channels that lowered the interaction between drainage water and the soil. Moreover, the channels for the Mole treatment were 0.1 m deeper than the Gravel Mole. This increased the travel distance of the water through the soil and hence allowed greater potential for P-sorption.

Particulate P accounted for between 0.16 and 0.20 of P losses to water via the three pathways. McDowell et al., (2001) also concluded that in grassland overland flow carries relatively little sediment due to minimal erosion. The majority of PP across all the treatments (0.69) was lost through the overland flow pathway. However PP flux in the overland flow was lower in mole and gravel Mole than Control because of lower overland flow and higher mole flow associated with Mole and Gravel Mole.

There was a greater flux of PP in mole flow from Mole than Gravel Mole (Table 2.5), which can be attributed to the absence of gravel in Mole leaving the walls of the Mole drain channels exposed to a greater degree of erosion than Gravel Mole.

2.6.2 Nitrogen loss to water

Despite no inorganic fertilizers being applied during this study, the background N in the soil combined with biologically fixed N was able to support a high level of N uptake by the herbage (459 kg ha⁻¹). It can be assumed that much of this N was subsequently recycled in urine and dung by the grazing dairy cows (although this aspect was not investigated in this study).

In contrast to P, improving infiltration by installing the drainage treatments lowered the proportion of nitrate-N and ammonium-N lost in overland flow and increased the proportion lost in mole flow and to ground water flow (Table 2.5), with the overall result that more nitrate-N and ammonium-N were lost from Mole and Gravel Mole than from Control. This difference can be attributed to greater attenuation and denitrification of nitrate-N in Control, whereas drainage and Gravel Mole facilitated the rapid loss of nitrate-N from the soil in mole flow. Both nitrate-N concentrations and fluxes were higher from Gravel Mole than from Mole. This can be partly attributed to the Gravel Mole channels being shallower than Mole channels. Moreover, nitrate-N losses were higher than ammonium-N in mole flow from Gravel Mole. Again, this can be explained by Gravel Mole facilitating a more rapid discharge of water via the mole channels; in this case lowering the travel time and decreasing the extent of denitrification taking place. Similar phenomena was reported by Gold et al., (2001) that found that artificially lowering the watertable resulted in nitrate-N bypassing areas of high natural attenuation, thus increasing nitrate-N losses.

Most (>0.54) of the ammonium-N was lost to ground water flow. This can be attributed to the impact of the microbial activity on the chemical transformation of N in the soil. Previous studies at this site (Necpalova et al., 2012) reported that biochemical transformation i.e. nitrate reduction to ammonium, is facilitated by the thickness of the saturated zone in the soil profile. This process is called dissimilatory nitrate reduction to ammonium. Soil dissimilatory nitrate reduction to ammonium capacity was likely to have been unaffected by drainage treatment (below drains depth) in the present study, this was shown on the lack of difference between treatments on the ammonium-N losses to the ground water (Table 2.4 and 2.5).

Annual nitrate-N and ammonium-N losses in the present study were lower than that found by Monaghan et al., (2002) and are within the range of a previous study on the present site, where losses in the shallow ground water under circumstances similar to the undrained control system in the present study were examined (Necpalova et al., 2012). A recent study at this site, also under undrained conditions, by Burchill et al., (2016) reported annual N losses to ground water (mean $\pm$ standard deviation) of 21 ± 8 and 5 ± 2 kg ha⁻¹ during 2011 and 2012, respectively. Authors reported a large inter-annual variability most likely affected by the climatic conditions. In the current study cumulative rainfall was similar to the year 2012 (1147 mm), despite this losses in the latter study were greater than annual N losses to water found in the present study. Burchill et al., (2016) concluded that the majority of N was lost from the soil as dinitrogen gas (0.44) followed by ammonia (0.42) whereas N losses (encompassing nitrate, ammonium etc.) to ground water flow accounted for 0.06 of total N losses.

2.6.3 Phosphorus and nitrogen concentrations in water

In this study the mean concentration of DRP lost via the overland flow and mole flow pathways was 0.41 mg L^{-1} , which is above the threshold of 0.03 mg L^{-1} necessary to avoid eutrophication in water bodies (European Council, 2003). These high concentrations can be explained by the extremely high soil P concentrations at this site.

Mean nitrate-N concentrations reported in this study were very low relative to that reported in other studies; i.e. 15 (range 5 to 21) mg nitrate-N L⁻¹ (Sharpley and Syers, 1979) and 6.9 (range 1 to 28) mg nitrate-N L⁻¹ (Monaghan et al., 2002). In the present study the mean nitrate-N concentration lost via the various pathways were 0.96 mg L^{-1} , well below the maximum allowed concentration (MAC) of 11.3 mg L^{-1} (European Council, 2000). However, the mean ammonium-N concentration reported in the present study was 1.09 mg L^{-1} , higher than the MAC for ground water, $0.065 \text{ mg N L}^{-1}$ for surface water and $0.175 \text{ mg N L}^{-1}$ for drinking water (European Council, 2009).

2.7 Conclusions

Over a full year of observations, we concluded that Mole and Gravel Mole drainage treatments deepened the watertable depth and decreased the volume of overland flow by improving soil permeability. This had direct implications for the interconnectivity of the hydrological pathways and associated nutrient losses in mole flow and to ground water flow. Mole and Gravel Mole drainage treatments increased the extent of nitrate-N and ammonium-N losses, although nitrate-N concentrations remained well below the threshold for drinking water. Ammonium-N concentrations were above the drinking water quality standards. Conversely, Mole and Gravel Mole lowered DRP and TP losses, by enhancing soil water infiltration. Therefore, Mole and Gravel Mole management techniques for improving trafficability and grass utilization are not likely to increase P losses to water, in particular DRP, on farms under similar management to that described in the present study. However nitrate-N and ammonium-N losses to water are likely to be higher, albeit in this study overall annual losses were very low.

CHAPTER 3: GREENHOUSE GAS EMISSIONS FROM TEMPERATE PERMANENT GRASSLAND ON CLAY- LOAMY SOIL FOLLOWING THE INSTALATION OF ARTIFICIAL DRAINAGE

Based on: Valbuena-Parralejo, N., Tuohy, P., Fenton, O., Burchill, W., Williams, M., Lanigan, G. J., & Humphreys, J. (2019). Greenhouse gas emissions from temperate permanent grassland on clay-loam soil following the installation of artificial drainage. *Agriculture, ecosystems & environment*, 269, 39-50.

3.1 Abstract

Shallow drainage (e.g. mole and gravel mole drainage installed at 0.55 m below soil surface) is used on pasture-based farms to reduce soil moisture, deepen water table depth and thus increase grass utilization on soils with impeded drainage. There is, however, a lack of research on the effect that these drainage techniques have on soil greenhouse gas (GHG) emissions. Hence the objective of this study was to investigate the effects of mole and gravel mole drainage on soil GHG fluxes. Soil nitrous oxide (N₂O), soil root respiration (CO₂) and soil methane (CH₄) fluxes were measured in two experiments conducted over a 36 month period. The experiments were conducted on permanent grassland with impeded drainage on a eutric gleysol (humic) in Ireland (52°30'N, 08°12'W). Both experiments were arranged in randomized complete block design. There were three treatments: (i) undrained control, (ii) mole drainage and (iii) gravel mole drainage. Soil GHG fluxes, soil water filled pore space (WFPS) and soil temperature were measured on a weekly basis and soil mineral nitrogen (N), herbage N uptake, soil organic carbon (SOC) and soil total N (TN) were measured seasonally and annually over the sampling period. Drainage treatments ($P < 0.05$) deepened (mean $\pm$ SE) the water table by 0.17 ± 0.03 m and decreased WFPS by $7 \pm 0.2\%$. However no ($P > 0.05$) impact of drainage treatment was detected on soil average daily GHG fluxes (9.84 ± 6.34 g N₂O-N ha⁻¹ day⁻¹, 34.05 ± 12.90 kg CO₂-C ha⁻¹ day⁻¹ and -0.91 ± 3.70 g CH₄-C ha⁻¹ day⁻¹), soil N mineralisation, soil N nitrification, SOC and TN in both experiments. The results indicate that mole and gravel mole drainage can be installed on farms with similar management and soil type to that described in the present study without impact on the soil GHG emissions.

3.2 Introduction

Permanent grassland accounts for 68% of agricultural land globally (FAO, 2014). Pasture-based systems are, in general, concentrated in temperate grassland regions. In temperate zones, pasture-based systems predominate in areas with high rainfall, low evapotranspiration and with soils with poor-drainage characteristics- often unsuitable for other forms of agricultural production (Humphreys et al., 2009). Such conditions coincide with poor soil trafficability by livestock and machinery, and poor grass utilization by grazing livestock (Keane, 1992; MacEwan et al., 1992; Drewry, 2006; Fitzgerald et al., 2008; Tuohy et al., 2016). Grass utilization under grazing is one of the main components underpinning the profitability in these systems (Dillon et al., 2005). Mole and gravel mole drainage have been used widely as shallow drainage techniques in these areas and have been found to reduce soil moisture content by

lowering the watertable depth (WTD) and improving surface water infiltration (Burke et al., 1974; Sharpley and Syers, 1979; Haygarth et al., 1998; Ibrahim et al., 2013; Tuohy et al., 2015)

Soil greenhouse gas (GHG) fluxes are a major contributor to global GHG emissions (Robertson et al., 2000). Grasslands have been reported to account for between 16% and 33% global agricultural N₂O emissions and are also an important carbon sink (De Klein et al., 2008). Soil GHG emissions are highly dependent on soil microbial activity, which is regulated by soil moisture content, soil temperature, soil pH and available soil organic carbon (SOC). Previous research has shown that soil disturbance, for example by ploughing, decreases soil moisture content and increases soil temperature (Davidson et al., 2000; Lal, 2004; Angers et al., 2010), and significantly increases soil N₂O and soil carbon dioxide (CO₂) fluxes (Schlesinger and Andrews, 2000; Ruan and Robertson, 2013; Garcia-Marco et al., 2016). At present, there is very little research on how mole drainage techniques impact on soil GHG fluxes, despite the wide application of mole drainage techniques on grasslands and their impact on soil microbial environment.

Hence the objective of this study was to investigate the effects of mole and gravel mole drainage on soil GHG emissions. We examined the soil cumulative (kg ha⁻¹ year⁻¹) and daily (g ha⁻¹) GHG fluxes from mole and gravel mole drainage systems throughout their life-span i) immediately after drainage installation, ii) during first drainage season and iii) three years after drainage installation.

3.3 Materials and methods

3.3.1 Site description

This study was conducted on a grassland site (4 ha) at Solohead Research Farm in Ireland (52°30'N, 08°12'W), with a gentle slope (1.4%) and bounded on the south and east by the river Pope (Figure X.1). The average annual rainfall in the preceding 10 years was 1089 mm with annual potential evapotranspiration of approximately 510 mm. Mean daily soil temperature at 10 cm depth ranged from 3.3 to 13.9 °C between November and January and from 10.1 to 20.0 °C between May and July. The WTD fluctuates seasonally from <1.0 m to > 2.5 m below ground level (Tuohy et al., 2015). The soils are classified as eutric gleysol (humic) under the 2014 World Reference Base (WRB, 2014). Soil physical and chemical properties are presented in Table 1.

The grassland site was reseeded with perennial ryegrass (*Lolium perenne*) and white clover (*Trifolium repens*) in August 2010. During the study the swards were predominantly perennial ryegrass with 20% white clover on a dry matter (DM) basis. In August 2010 a large open V-shaped drain (2 m depth and 4 m wide at ground level) was excavated along a natural depression in the site running along a west-east axis with water exiting into the river Pope from the eastern end of the open drain (Figure 2.1). The large open drain separated the study area into two sites: site A and site B (Figure 2.1) where experiments A (Expt A) and B (Expt B) were conducted. All the runoff and drainage flow originating in both sites was discharged to the open drain and then to the river Pope. The experimental site was hydrologically isolated from external lateral water movement by an isolation ditch, comprising a 1 m deep trench filled to the soil surface with stone aggregate (Figure 2.1).

3.3.2 Experimental layout and design

3.3.2.1 Experiment A

In January 2011, the area was divided into four blocks; each 60 m wide and 100 m long, with the long axis of each block running from north to south. Each block was subdivided into four plots (100 m long and 15 m wide), with the long axis of each plot running perpendicular to the open drain. One of four treatments was imposed in each plot in a randomised complete block design with four replicates. The four treatments were (i) undrained control (C), (ii) mole drainage installed in January 2011(M1), (iii) mole drainage installed in July 2011(M2) and (iv) gravel mole drainage installed in July 2011 (GM). Treatment installation dates were chosen in order to evaluate the drainage treatments performance with different soil conditions at installation. Soil moisture conditions at the time of installation are presented in Table 3.1. Further details of the treatments are available in Tuohy et al., (2015) and Tuohy et al., (2016).

Table 3.1 Soil gravimetric moisture content (g g^{-1}) on the treatment installation dates. M1: 31 January 2011, M2: 12 July 2011, GM: 14 July 2011 and GMb: 25 June 2014.

Depth (cm)	Experiment A			Experiment B
	M1	M2	GM	GMb
0 to 10	0.49	0.39	0.39	0.32
10 to 30	0.49	0.39	0.39	0.25
30 to 60	0.35	0.21	0.21	0.25
60 to 80	0.35	0.21	0.21	0.22

Adjacent plots were separated by soil berms (0.30 m high). The mole plough (M. Tighe Engineering, Navan, Co. Meath, Ireland) had a 0.075-m-diameter foot, a 0.08-m-diameter

expander plug and an adjustable leg capable of reaching a depth of 0.55 m. Mole drains were installed at a depth of 0.55 m (Table 1.2) with a spacing of 1.2 m between the parallel mole drains. The depth was chosen taking into account the soil clay content and the maximum attainable depth by the mole plough in order to induce maximum soil disruption.

The gravel mole plough (O' Keeffe Engineering, Listowel, Co. Kerry, Ireland) had a 0.08-m-wide foot and could reach depths up to 0.40 m. It was set to install a 0.20 m diameter row of gravel (10–20 mm grade) within the channel. Gravel mole drainage was installed at a depth of 0.40 m and spacing of 1.2 m. There were a total of 11 mole or gravel mole channels in each drained plot.

Two sub-plots (2×11 m) for herbage, WTD and N_2O measurements were installed along the long axis of each plot. One sub-plot was established in the upper half (up slope) and the second in the lower half (down slope) of the plots. Two piezometers, for WTD measurements, were installed in each sub-plot. Details of the piezometers are provided below.

3.3.2.2 Experiment B

In September 2011, the area of site B was divided into four blocks (80 m wide, 60 m long). Each block was subdivided into two 40×60 m plots; the long axis of each plot running perpendicular to the open drain. There were two treatments per replicated block in a randomised complete block design: (i) gravel mole drained (GMb) and (ii) undrained control (Cb). The Cb treatment received no drainage. In the GMb treatment three collector drains were installed per plot in September 2011. There was 10 m between each collector drain and 5 m between the collector drain and the border of the main plots. Each collector drain was 1 m deep and 60 m long and aligned along the long axis of each GMb plot. Drainage water exited each collector drain into the open drain. Each collector drain was backfilled with stone (50-100 mm grade) to within 200 mm of the soil surface and then filled with topsoil up to the soil surface.

In June 2014, gravel mole drains were installed in each GMb plot perpendicular to the collector drains. The gravel mole plough settings and installation depth and spacing were the same as described for Expt A above. There were a total of 46 gravel mole channels per GMb plot. Soil moisture conditions at the time of installation are presented in Table 3.1.

Two sub-plots (2×11 m) were laid down at randomly assigned locations in the centre (well away from the borders) of each plot. Similar to that described above, two piezometers were installed at each sub-plot.

3.3.2.3 Grassland management

The plots were rotationally grazed by lactating dairy cows at approximately monthly intervals during the main grazing season (March to November). The grazing rotation length varied from 21 days in late spring/early summer to 42 days in autumn. Postgrazing sward height, maintained at 4 cm, was measured using a rising plate meter (Grasstec, Charleville, Ireland) and used to determine when cows were moved to the next plot. The sub-plots were fenced to prevent grazing. Herbage from the sub-plots was harvested and measured at intervals coinciding with the grazing events.

No fertilizer was applied to the plots during the study period. However the sward was composed predominantly of perennial ryegrass ($>600 \text{ g kg}^{-1} \text{ DM}$) and white clover ($230 \text{ g kg}^{-1} \text{ DM}$) and the clover sward provided nitrogen (N) input through the biological nitrogen fixation. The soil Morgan's potassium (K) concentrations (mean $\pm$ standard error (SE)) was $184 \pm 24 \text{ mg L}^{-1}$, which was in excess of an agronomic response to applications of fertilizer K (100 to 150 mg L^{-1}) (Wall and Plunkett, 2016). Likewise soil Morgan's phosphorus (P) concentrations was well over of an agronomic response to applications of fertilizer P (5 to 8 mg L^{-1}), averaging $27.4 \pm 1.1 \text{ mg L}^{-1}$.

3.4 Experimental measurements

3.4.1 Sampling periods and sampling regime

All experimental measurements took place in the sub-plots. There were three distinct sampling periods (SP) on Expt A, hereafter referred to as SP1, SP2 and SP3 (Table 3.2).

Table 3.2 Sampling periods of the study and variables measured in each sampling period. The ✓ indicates that the corresponding variable was measured during that time frame.

	Experiment A			Experiment B
	SP 1	SP 2	SP 3	SP4
Dates	27 May to 26 October 2011	1 January 2012 to 31 December 2013	10 January to 10 July 2014	26 June 2014 to 25 July 2015
N ₂ O flux	✓		✓	✓
CH ₄ flux	✓		✓	✓
CO ₂ flux			✓	✓
WTD		✓	✓	✓
WFPS	✓	✓	✓	✓
N _{min} and N _{nit}			✓	✓
SOC	✓	✓	✓	✓
Herbage yield	✓	✓	✓	✓
N herbage uptake	✓	✓	✓	✓

SP = sampling period

N₂O flux = soil nitrous oxide flux.

CH₄ flux = soil methane flux.

CO₂ flux = soil heterotrophic respiration.

WTD = water table depth.

WFPS = water filled pore space.

N_{min} = soil nitrogen mineralisation rate

N_{nit} = soil nitrogen nitrification rate.

SOC = soil organic carbon.

The first (SP1) was from the 27 May 2011 until 26 October 2011. SP1 captured any initial effects of mole drainage implemented in January 2011(M1) on soil N₂O and CH₄ flux. During the second sampling period (SP2), from the 1 January 2012 until the 31 December 2013, the performance of the drainage systems was examined in terms of water filled pore space (WFPS) at the soil surface, WTD and herbage yields. The third period (SP3) was from the 10 January 2014 until the 9 July 2014, and examined effects of the mole and gravel mole drainage on soil GHG flux three years after drainage installation. Mole drainage (M1 and M2) has always been considered to have a shorter effective life-time span than GM and must be regularly repeated to maintain effectiveness (Mulqueen, 1985). Hence SP3 represented a timeframe when the effectiveness of both M1 and M2 had deteriorated to a greater extent than GM due to their shorter life-time span (Tuohy et al., 2015).

There was one measurement period on Expt B, hereafter referred to as SP4. This period commenced immediately after GMb treatment installation on the 26 June 2014 and continued until the 25 June 2015. This period was chosen to capture any effects of gravel mole drainage for the first full year after installation.

3.4.2 Meteorological measurements

An automated weather-station (Campbell Scientific Ltd. Loughborough, UK) recorded rainfall, air temperature, soil temperature, wind speed, wind direction and solar radiation with a resolution of 30 min. The hybrid grassland model of Schulte *et al.* (2005), assuming poorly drained soil classification within the model, was used for estimation of the soil moisture deficit (SMD) and effective drainage (ED), which was calculated as the difference between precipitation and estimated evapotranspiration during this study.

3.4.3 Soil N₂O and CH₄ flux

A closed static chamber technique (Hutchinson and Mosier, 1981) was used to measure N₂O and CH₄ fluxes. The chamber consisted of two parts: a permanent collar and a chamber, both of which were constructed from polyvinylchloride (PVC) pipe. The headspace height of the chamber was 40 cm with an internal diameter of 29.5 cm, resulting in a cover area of 0.0683 m² and a headspace volume of 0.0273 m³. A rubber turnover stopper 16 mm diameter (Rubber Suba Seal, Sigma-Aldrich Ireland) was permanently inserted in the top of each chamber and this was changed every 4 months. The collars were permanently inserted into the soil to a depth of 12 cm and extended 4 cm above the soil surface following installation. In Expt A two chambers and collars were deployed within each sub-plot, this resulted in a total of 16 chambers for each treatment, which was greater than the minimum sample size calculated using a power

analysis, taking into account the standard deviation and the difference of the means of a pre-experimental data set. In experiment B a total of four chambers and collars were deployed in each sub-plot, which was a total of 32 chambers for each treatment. The N₂O and CH₄ sampling was conducted (between 9:30 am and 1:00 pm) once per week on a weekly basis.

At each sampling date the chambers were placed on the collars siled. Gas samples were taken from the chamber with a gas-tight syringe (BD, New Jersey, USA) immediately after chamber deployment (T0), and again after 15 (T15) and 30 (T30) minutes. All gas samples were transferred to pre-evacuated 7 ml screw-cap exetainer with Teflon-coated butyl septa (Perbio Science, Cambridge, UK). Air temperature inside the chamber and soil temperature at 5 cm depth beside the chamber were recorded using a thermo-sensor (Sensor-Tech Ltd., Castlebellingham, Ireland). Gas samples were analysed for N₂O and CH₄ using a gas chromatograph (GC) (Varian GC 450; Netherlands) as described in full by Burchill *et al.* (2014). The CH₄ and N₂O were analysed simultaneously with the same procedure, the calibration curves consisted on N₂O standards of 0.2, 0.5, 1, 5, 10, 20, 50 and 100 ppm and CH₄ standards of 0.1, 0.2, 0.5, 1, 3 and 5 ppm.

3.4.4 Measurements of soil respiration

Measurements of soil respiration (CO₂) were made using a CIRAS-1 gas exchange system (PP systems, Norfolk UK) fitted with the SRC-1 soil respiration chamber. The soil chamber was cylindrical (height = 150 mm; diameter = 100 mm). The method for measuring soil respiration was that described by Parkinson (1981). Measurements were made once per week each week during SP3 and SP4 (Table 3) between 10:00 am and 01:00 pm which represent day time soil respiration average.

The above ground biomass was cut to expose bare soil before placing the chamber on the soil surface. The chamber was pushed into the ground to a depth of 1.5 cm to create an air tight seal between the chamber and the soil surface. The chamber was placed onto the ground for approximately 1 to 2 min periods, during which the air in the chamber was re-circulated between the chamber and CIRAS-1. Concentration of CO₂ in the air was determined using infra-red gas analysis. The flux of CO₂ was calculated by fitting equation to the relationship between the increasing CO₂ concentration and elapsed time (PP-Systems, 2003).

3.4.5 Calculations of GHG flux

Pre-experimental testing showed a linear increase in N₂O concentration and a linear decrease in CH₄ concentration in the chambers over a 40 minute test period. Linearity of all the chambers at every sampling occasion was checked as a quality control to accept or reject the flux. The

criterion used to accept a reading was $R^2 \geq 0.8$. Hence, the change in the N_2O and CH_4 concentration in the headspace over the 30 min cover period was used to calculate N_2O and CH_4 fluxes. Daily N_2O and CH_4 fluxes from each treatment were calculated using the arithmetic means of (i) sixteen chambers for experiment A and (ii) thirty-two chambers for experiment B. Cumulative N_2O and CH_4 ($kg\ ha^{-1}$) emissions for the entire observation periods were calculated by linear interpolation between sampling occasions.

Daily CO_2 fluxes ($g\ C\ ha^{-1}\ day^{-1}$) for each treatment were calculated using the arithmetic means of twelve measurements. Cumulative CO_2 emissions ($kg\ C\ ha^{-1}$) emissions for the entire observation periods were calculated by linear interpolation between sampling occasions.

Greenhouse gases were converted in CO_2 -C equivalents (CO_2 -C eq). This was calculated using the 100 year horizon global warming potential of 34 for CH_4 and 298 for N_2O (IPCC, 2013). In the GHG budget we included the soil cumulative production and consumption processes measured in the study: soil N_2O , CH_4 emissions and soil root respiration (CO_2).

3.4.6 Soil water table depth and soil moisture content

Two fully screened piezometers (HDPE pipes, internal diameter 19.6 cm - Eijkelkamp, Agrisearch Equipment, Giesbeek, The Netherlands) were installed in each sub-plot. The piezometers were installed 3 m below ground level using a soil pneumatic drill (Giddings Machine Co, Windsor, Colorado, USA). Near the soil surface, the area between the soil and the pipe (0.03 m) was filled with bentonite to prevent water channelling down the outside the piezometer. All piezometers provided WTD readings on weekly basis using an electronic dipper (Marton Geotechnical Services Ltd, Suffolk, U.K.). On Expt A soil WTD was measured continuously between 1 April 2012 and 1 August 2014. On Expt B WTD was measured between the 10 November 2014 and 11 December 2015.

At each gas sampling date, volumetric soil water content was recorded at 5 cm depth using a HH2 moisture meter in conjunction with a Theta Probe sensor (Delta-T Devices, Cambridge, UK). The WFPS at 5 cm depth was calculated according to Ri et al., (2003). On Expt A WFPS was measured once per week between 25 May 2011 until 10 July 2014. On Expt B measurements took place between 1 June 2014 and 15 July 2015.

3.4.7 Soil net mineralisation and net nitrification

The in situ bag technique of Eno (1960) was used to quantify seasonal changes in N mineralisation and nitrification. This procedure was carried out during four periods: autumn (26 September to 26 November), winter (19 December to 20 January), spring (14 April to 12 May) and summer (19 June to 20 July) during 2013/2014 and 2014/2015 on both experiments. On

each sampling occasion two pairs of intact soil cores from each sub-plot (1.5 cm diameter) were taken from the soil, near the N₂O chamber, to a depth of 10 cm. One of the cores (N initial) was used to determine the initial amount of soil inorganic-N. The other one (N final) was sealed tightly in a 20 µm thickness polyethylene bag, and it was reburied in its original hole incubate and in the field for approximately one month. The polyethylene bag was permeable to water and gas, allowing the soil core sample to be in similar conditions as the adjacent soil. After one month the soil core was removed from the bag and prepared for potassium chloride (KCl) extraction to determine the soil inorganic N (NH₄⁺ and NO₃⁻) contents. Each soil core was subjected to extraction within an hour of sampling. Extracts were obtained by shaking sieved (< 2 mm) samples of field moist soil with 2 M KCl continuously for 3 h at a solution ratio of 2:1 (100 ml:50 g, ratio v/w; Bremner (1965)). Following shaking, the soil suspension was left to stand for 5 min before filtering (Whatman no. 2 filtered paper, d = 12.5 cm, 0.45 µm).

The rate of the net mineralisation and net N nitrification was expressed on a dry soil basis (kg N ha⁻¹ day⁻¹) and calculated using the following equations 1 and 2 respectively (Nelson *et al.*, 2006).

$$[1] \text{ Net N mineralisation} = \frac{(NH_4N + NO_3N)_{final} - (NH_4N + NO_3N)_{initial}}{time}$$

$$[2] \text{ Net N nitrification} = \frac{NO_3N_{final} - NO_3N_{initial}}{time}$$

The conversion of net N mineralisation and nitrification rates from soil per unit area was calculated multiplying the above results by the volume sampled (soil core area) using the soil bulk density (g/cm³) and soil dry matter (%). Results were expressed in kg N ha⁻¹.

3.4.8 Soil organic carbon and soil total nitrogen

Soil samples from all treatments on Expt A were taken in January 2011, April 2014 and January 2015. Likewise, on Expt B samples were taken in April 2014, January 2015 and January 2016. At each sampling occasion, fifteen cores per main plot were taken randomly using a Geonor MCL3 hydraulic auger (Geonor A/S, Oslo, Norway) to a depth of 30 cm. Samples were bulked to provide a composite sample per plot. The soil samples were processed and analysed for total carbon (TC), SOC and total nitrogen (TN) using a LECO CN-2000 elemental analyser (Leco, St. Joseph, MI, USA) as described in full by Necpálová *et al.* (2013a).

3.4.9 Herbage yield and N uptake in herbage

Herbage yield and N uptake was measured in sub-plots, coinciding with the grazing rotation on the main plot, which was at approximately monthly intervals during the grazing season (March to November) each year encompassing 2011 to 2015. A 1.2×10 m strip from the sub-plot was harvested to 4 cm above ground level using a rotary mower (Etesia UK Ltd, Warwick, UK) and weighed. A sub-sample of 100 g was dried for 16 h at 100 °C to determine the herbage DM content. A second 100 g sub-sample was stored at -20 °C before being freeze dried, milled to pass a 0.2 mm sieve and analysed for total N using a LECO 528 auto-analyser (LECO Corporation, St. Joseph, MI, USA). Uptake of N in herbage was calculated by multiplying herbage DM yield by the N concentration in harvested herbage for each sampling date; annual N uptake was calculated by summing values over one year.

3.4.10 Statistical analysis

All daily data WTD, soil WFPS, N₂O, CO₂ and CH₄ flux were subjected to ANOVA using SAS 9.3 (SAS 2011, Institute Inc., Cary, NC, USA). The variables included in the model were treatment, sampling date and treatment by sampling date interaction. Fixed effects were treatment and block. Sampling date was included as a repeated measure in the ANOVA. The N₂O data required a natural logarithmic transformation [$y = \log(x)$]. The annual cumulative GHG flux, cumulative GHG flux transformed in CO₂-C eq, seasonal soil net mineralisation and nitrification rates, N herbage uptake, SOC and TN were subjected to ANOVA and the variables included in the model were treatment and block. Post-hoc Tukey-Kramer multiple comparison tests were carried out following all ANOVAs on cumulative data to determine differences between treatment means.

Finally, multiple linear regression (stepwise) analysis was carried out to create simple explanatory models with weather and soil variables to account of variation in daily N₂O emissions, using data from each plot in each year emissions (experiment A: n = 185 and experiment B: n= 342). A statistical probability of $P < 0.05$ was considered significant for all statistical tests.

3.5 Results

3.5.1 Meteorological data

Total rainfall during SP1 was 519 mm. Annual rainfall was 1147 mm in 2012, 975 mm in 2013, 1201 mm in 2014 and 1100 mm in 2015. Long term average (10 preceding years) cumulative rainfall was 1017 mm. Daily precipitation, ED and SMD data for SP3 and SP4 are presented in Figure 3.1. Average daily soil temperature (10 cm depth) ranged from 2.2 to 19.8 °C during the timeframe of this study.

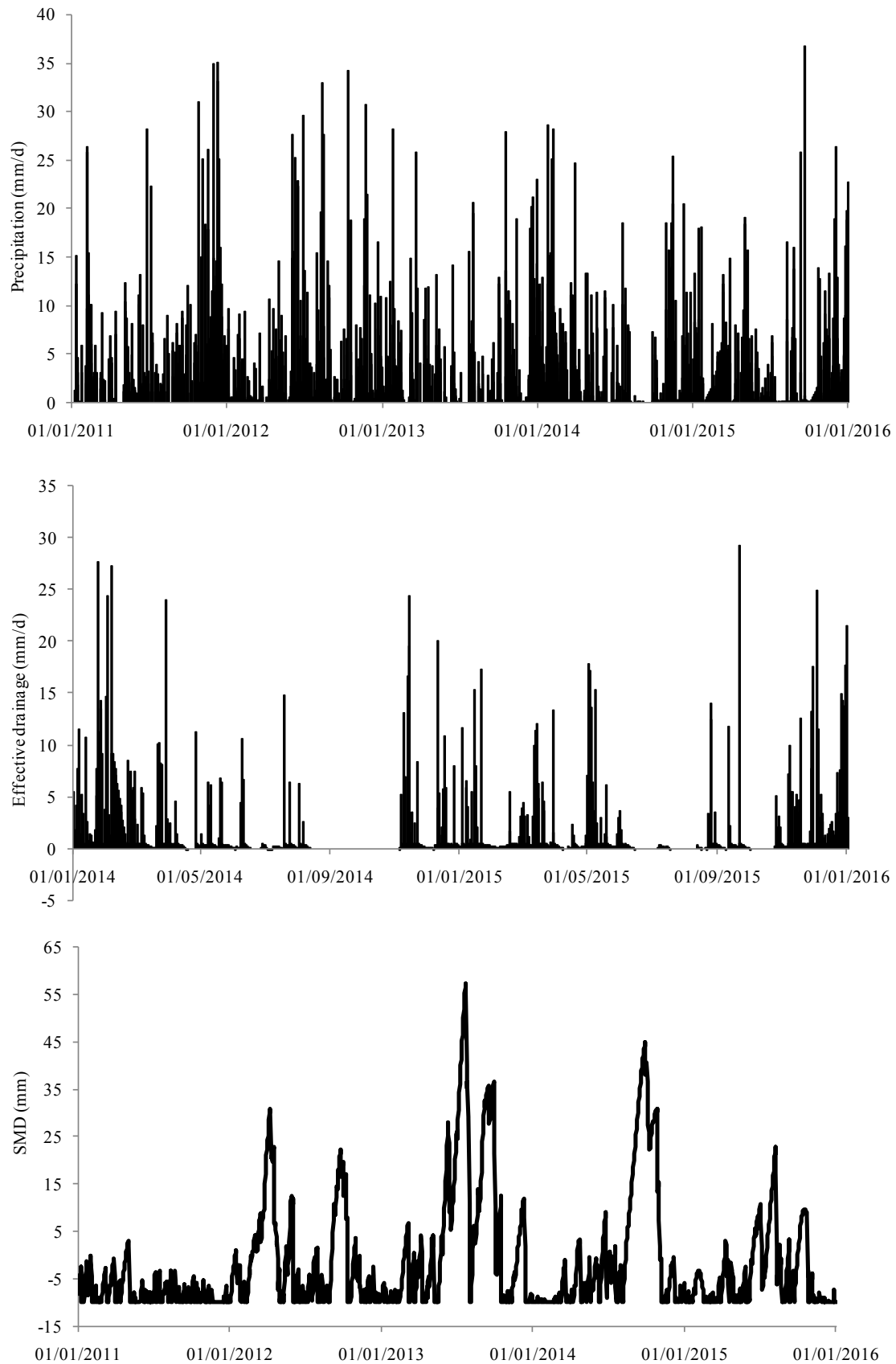


Figure 3.1 Daily precipitation, Effective drainage (ED) and Soil moisture deficit (SMD) during the study.

3.5.2 Soil water table depth and soil water filled pore space

The WTD was ($P<0.05$) lower in the gravel mole treatments (GM and GMb) than in the control in both Expt A and B (Table 3.3). In Expt A the mean WTD in M2 was also ($P<0.05$) lower than C, but only during SP2. In Expt A, during SP2 and SP3 WTD up-slope (mean $\pm$ standard error) was 1.36 ± 0.07 m, deeper ($P<0.05$) than down-slope (1.07 ± 0.07 m).

Table 3.3 Impact of drainage treatments on watertable depth (WTD) and soil water filled pore space (WFPS) during sampling periods (SP) 1, 2 and 3 in Experiment A and SP4 in experiment B Treatments control (C and Cb), mole 1 (M1), mole 2 (M2), gravel mole (GM), and gravel mole into collectors drains (GMb). Standard error of the mean (SEM). Values with the same letter are not significantly different.

		M 1	M 2	GM	C	SEM	P-value
WTD (m)							
Expt A	SP 1						
	SP 2	1.12 ^{a,b}	1.13 ^b	1.13 ^b	0.80 ^a	0.126	< 0.0001
	SP 3	1.21 ^{a,b}	1.19 ^a	1.38 ^b	1.09 ^a	0.851	< 0.001
		WFPS (%)					
	SP 1	84 ^b	-	-	90 ^a	1.7	< 0.05
	SP 2	83 ^b	83 ^b	84 ^b	93 ^a	0.6	< 0.05
GMb Cb SEM P-value							
Expt B		WTD (m)					
	SP 4	-	-	0.84	0.76	0.275	< 0.05
		WFPS (%)					
	SP 4	-	-	50	55	0.2	< 0.05

Soil WFPS was ($P<0.05$) lower in the gravel mole treatments (GM and GMb) than in the controls throughout all four sampling periods (SPs) (Table 3.4). During SP1 and SP2 the WFPS in M1 and M2 were ($P<0.05$) lower than C. During SP3 mean WFPS in M2 was also ($P<0.05$) lower than C.

The temporal trend of WFPS and WTD during SP3 and SP4 is presented in Figure 3.2. The WTD fluctuated seasonally during these sampling periods. Mean daily WTD was the deepest during the summer. The WFPS ranged between 15% and 100% and remained above 50% for the majority of these two SPs in both experiments. Likewise for WTD there was a seasonal trend in all the treatments. There was a ($P < 0.01$) linear correlation between WTD and WFPS ($y = -10.831x + 79.699$, $R^2 = 0.17$).

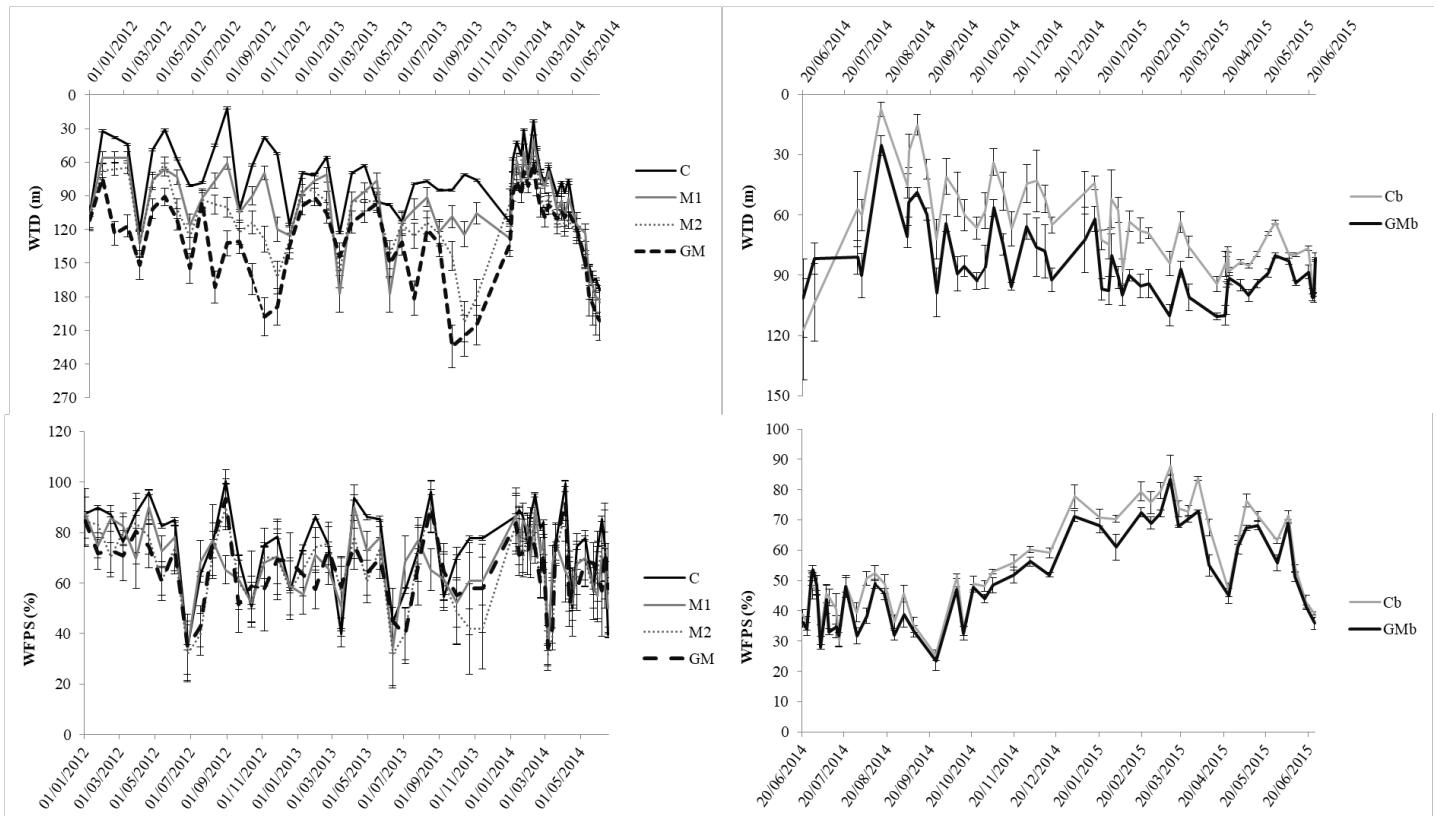


Figure 3.2 Water table depth (m; mean $\pm$ SE) and water filled pore space (%; mean $\pm$ SE) at 0–5 cm depth from control (C/Cb), mole (M1 and M2) and gravel mole drained (GM and GMb) during sampling period (SP) 2, SP3 and SP4.

3.5.3 Temporal trends in daily N_2O , CO_2 and CH_4

In both experiments drainage treatment did not impact ($P > 0.05$) on soil daily N_2O , CO_2 and CH_4 fluxes. Daily N_2O , CO_2 and CH_4 fluxes during SP3 and SP4 are presented in Figure 3.3.

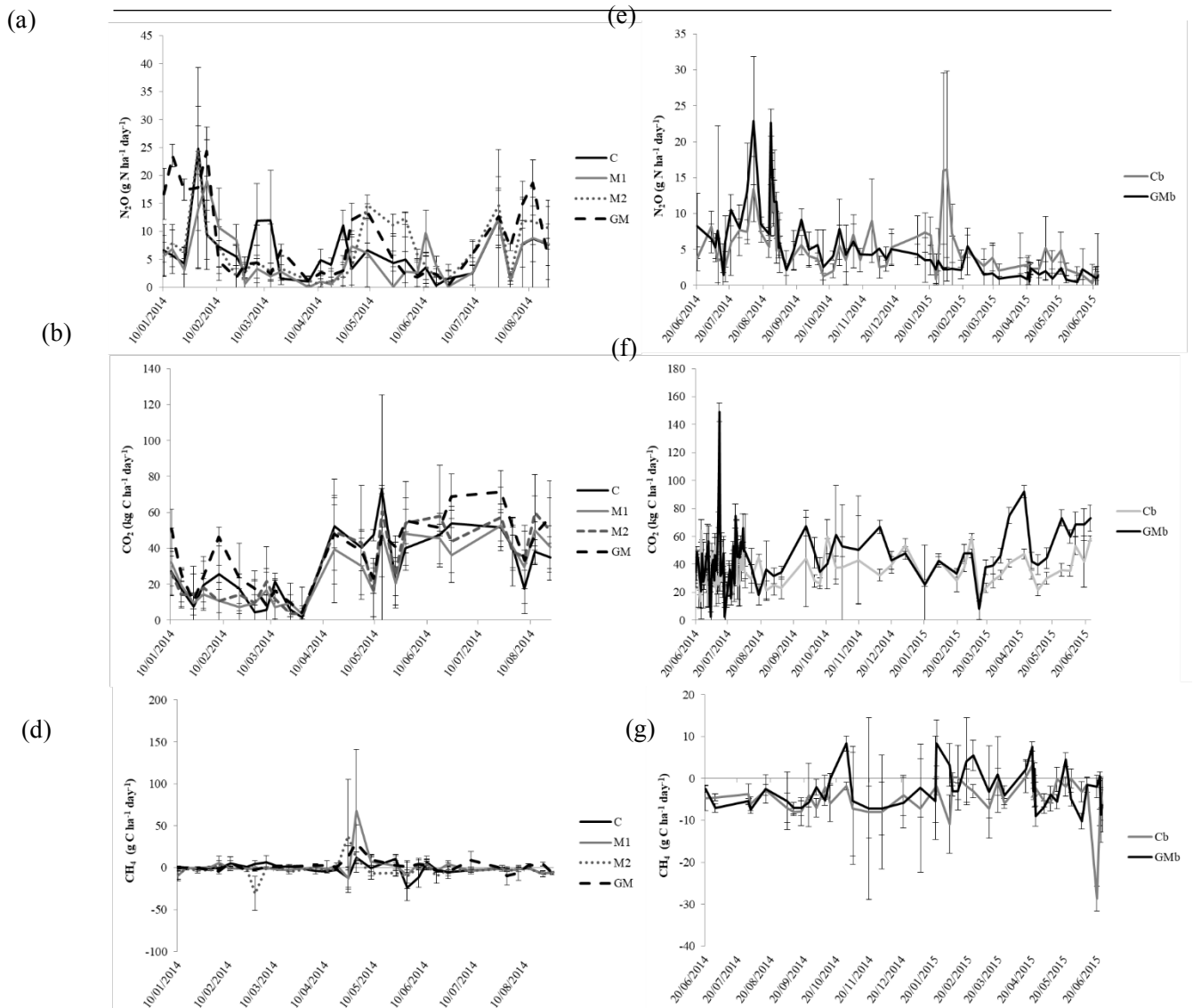
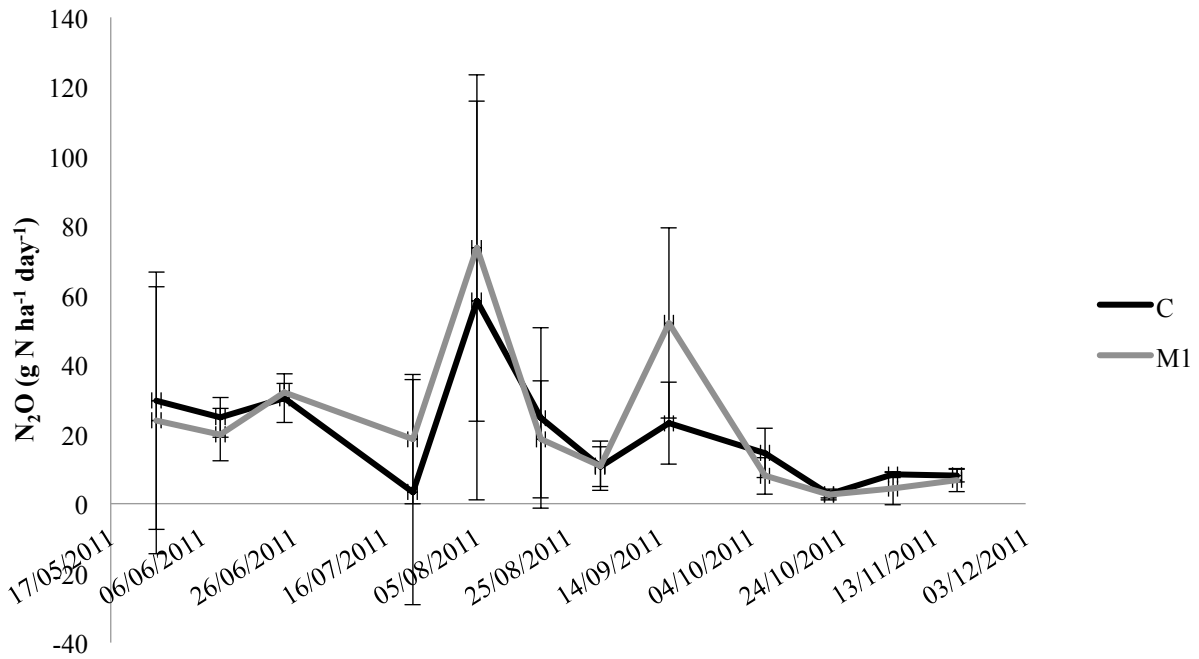


Figure 3.3 .(a) Daily N_2O fluxes (mean $\pm$ SE), (b) daily CO_2 fluxes (mean $\pm$ SE) and (d) daily CH_4 fluxes (mean $\pm$ SE) from control (C), mole drained (M1 and M2) and gravel mole drained (GM) during SP3 in Experiment A. (e) Daily N_2O fluxes (mean $\pm$ SE), (f) daily CO_2 fluxes (mean $\pm$ SE) and (g) daily CH_4 fluxes (mean $\pm$ SE) from control (Cb) and gravel mole drained (GMb) during SP4 in Experiment B. There was no ($P>0.05$) impact of drainage on daily N_2O , CO_2 and CH_4 fluxes. SP = sample period.

Nitrous oxide fluxes from all treatments were characterized by episodic peaks with daily mean values displaying large variation throughout the study. During SP1, daily N_2O flux ranged between -3 and $73.5 \text{ g N ha}^{-1} \text{ day}^{-1}$ (Figure 3.4a).

(a)



(b)

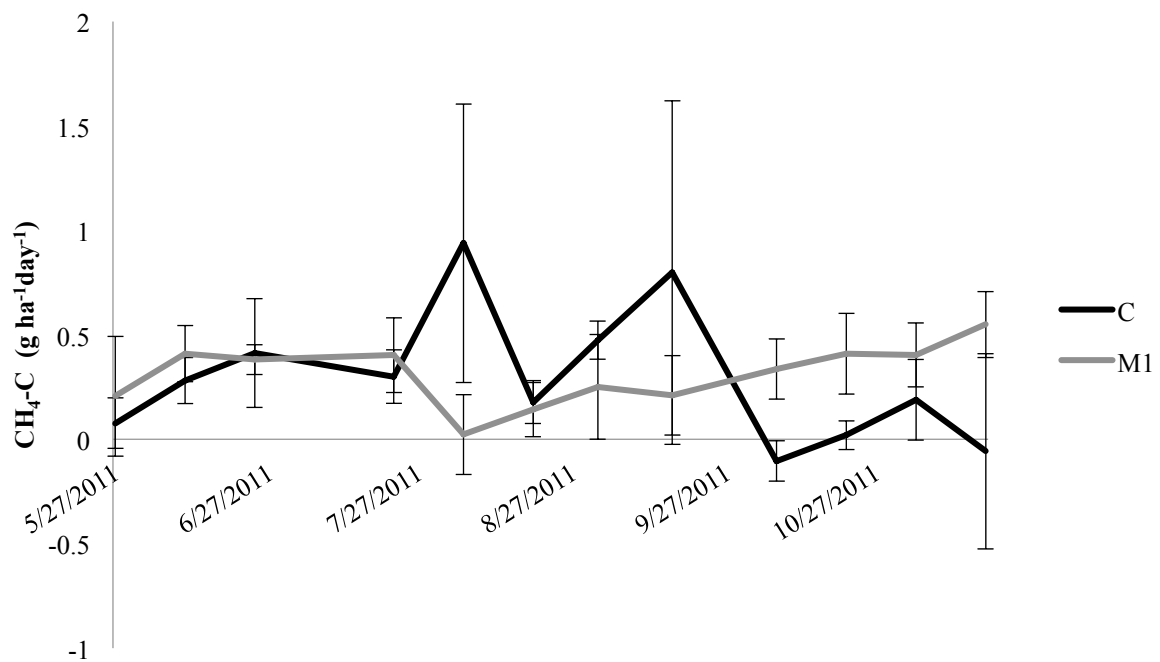


Figure 3.4 (a) Daily N_2O fluxes (mean $\pm$ SE) and (b) daily CH_4 fluxes (mean $\pm$ SE) from control (C), mole drained (M1 during SP1 in Experiment A. There was no ($P>0.05$) impact of drainage on daily N_2O and CH_4 fluxes. SP = sample period.

Across the entire study the highest daily N_2O fluxes in both drained and control treatments occurred on 05/08/2011, 04/02/2014, 11/08/2014 and 27/08/2014. The daily N_2O fluxes from all treatments were generally highest during the summer particularly during August 2011.

The daily average ($\pm$ SE) soil CO₂ fluxes during SP3 and SP4 were 34.05 ± 12.898 kg CO₂-C ha⁻¹ day⁻¹ (Figure 3.3b and 3.3f). Soil respiration followed a seasonal pattern and was ($P < 0.05$) correlated with soil temperature ($y = 0.0375x + 0.0105$, $R^2 = 0.41$)

Mean daily CH₄ fluxes during SP1, SP3 and SP4 were very small, averaging -0.91 ± 3.701 g CH₄-C ha⁻¹ day⁻¹. Mean daily CH₄ fluxes during SP1 are presented in Figure 3.3b and fluxes during SP3 and SP4 are presented in Figure 3.3d and Figure 3.3g.

3.5.4 Soil cumulative N₂O, CO₂, CH₄ flux and GHG budget

During SP1 there was no ($P > 0.05$) difference in cumulative GHG fluxes between M1 and C. Soil cumulative (i) N₂O fluxes were 3.4 and 3.7 (standard error of the mean (SEM) 1.29) kg N ha⁻¹ and (ii) soil cumulative CH₄ fluxes were 0.05 and 0.06 (SEM 0.021) kg C ha⁻¹ for C and M1, respectively.

Cumulative N₂O, CH₄ and CO₂ flux were not ($P > 0.05$) affected by treatment during SP3 and during SP4. During SP3, soil N₂O, CO₂, CH₄ cumulative fluxes were 0.8 ± 0.2 kg N₂O-N ha⁻¹, 686.6 ± 39.4 kg CO₂-C ha⁻¹ and -0.1 ± 0.03 kg CH₄-C ha⁻¹. Cumulative fluxes during SP4 were 1.6 ± 0.2 kg N₂O-N ha⁻¹ year⁻¹, 14274.5 ± 258.5 kg CO₂-C ha⁻¹ year⁻¹ and -0.6 ± 0.2 kg CH₄-C ha⁻¹ year⁻¹.

Drainage treatment did not ($P > 0.05$) impact on soil cumulative GHG fluxes (Table 3.4).

Table 3.4 Greenhouse gas budgets presented as CO₂-C equivalents (CO₂-Ceq) for control (C/Cb), mole 1 (M1), mole 2 (M2), gravel mole drained (GM) and gravel mole drained into collector drains (GMb) grassland. CO₂-C eq were calculated based on the GWP of 100 years from the annual cumulative N₂O, CO₂ and CH₄ emissions from three sampling periods. The CH₄ flux negative values mean C uptake by the soil.

		C/Cb	M 1	M 2	GM	GMb	SEM	P-value
N₂O-N (kg CO₂-C eq ha⁻¹)								
Expt A	SP 1	472.5	434.2	-	-	-	166.0	> 0.05
	SP 3	97.5	178.8	113.8	146.3	-	32.5	> 0.05
Expt B	SP 4*	204.0	-	-	-	191.6	28.1	> 0.05
CH₄-C (kg CO₂-C eq ha⁻¹)								
Expt A	SP 1	0.6	0.7	-	-	-	0.2	> 0.05
	SP 3	-0.02	0.01	-0.02	0.02	-	0.77	> 0.05
Expt B	SP 4*	-9.9	-	-	-	-4.9	3.0	> 0.05
CO₂-C (kg CO₂-C ha⁻¹)								
Expt A	SP 1	-	-	-	-	-		
	SP 3	588	680	698	780	-	63	> 0.05
Expt B	SP 4*	14799	-	-	-	13750	258	> 0.05

*SP: sampling period. For SP 4 C treatment is Cb (undrained).

A summary of the average daily fluxes from all the SP and the cumulative fluxes from SP4 is presented in Table 3.5.

Table 3.5 Summary table of average daily and cumulative annual greenhouse gas (GHG) emissions. The GHG consisted on nitrous oxide (N₂O), soil respiration (CO₂) and methane (CH₄). Treatments are control (C/Cb), mole 1 (M1), mole 2 (M2), gravel mole drained (GM) and gravel mole drained into collector drains (GMb) grassland. All the GHG is expressed in CO₂-C equivalents (CO₂-Ceq), the CO₂-C eq were calculated based on the GWP of 100 years from the annual cumulative N₂O, CO₂ and CH₄ emissions. The CH₄ flux negative values

GHG	Units	C/Cb	M1	M2	GM/GMb
N ₂ O	g N ₂ O-N ha ⁻¹ day ⁻¹	12.7 (9.39)	13.9 (8.96)	6.6 (4.42)	6.1 (2.58)
	*kg N ₂ O-N ha ⁻¹ year ⁻¹	1.6 (0.32)	-	-	1.5 (0.12)
CO ₂	kg CO ₂ -C ha ⁻¹ day ⁻¹	33.2 (7.41)	28.7 (15.75)	32.6 (19.21)	41.7 (9.22)
	*kg CO ₂ -C ha ⁻¹ year ⁻¹	14799 (258.51)	-	-	13750 (237.81)
CH ₄	g CH ₄ -C ha ⁻¹ day ⁻¹	-1.5 (1.94)	0.7 (3.16)	-1.9 (5.93)	-0.9 (3.76)
	*kg CH ₄ -C ha ⁻¹ year ⁻¹	-0.8 (0.16)	-	-	-0.4 (0.21)
All GHG	*kg CO ₂ -C eq ha ⁻¹ year ⁻¹	15040 (4903.9)	-	-	14209 (4514.8)

mean C uptake by the soil.

*annual values are from the fourth sampling period (SP4) only, because it accounted for the entire 12 months period. For SP 4 treatment is Cb (undrained) and GMb (drained).

3.5.5 Soil TN, SOC, soil net mineralisation and nitrification

In both experiments, drainage treatment did not impact ($P > 0.05$) on the TN and SOC (Table 3.6).

Likewise, soil net mineralisation and nitrification was not affected ($P > 0.05$) by drainage treatment. During SP3, mean ($\pm$ SEM) soil net mineralisation and nitrification from all the treatments was 0.431 ± 0.39 mg N kg soil⁻¹ day⁻¹ and 0.432 ± 0.28 mg N kg soil⁻¹ day⁻¹. During SP4 soil net mineralisation and nitrification rates averaged 0.101 ± 0.051 mg N kg soil⁻¹ day⁻¹ and 0.127 ± 0.056 mg N kg soil⁻¹ day⁻¹ respectively. There was a ($P < 0.05$) effect of season. During SP3 and SP4 mean ($\pm$ SE) soil net mineralisation and nitrification increased during summer (19 June to 20 July) to 0.26 ± 0.1 mg N kg soil⁻¹ day⁻¹ and 0.24 ± 0.1 mg N kg soil⁻¹ day⁻¹. The minimum soil net mineralisation and nitrification (-0.35 ± 0.1 mg N kg soil⁻¹ day⁻¹ and -0.13 ± 0.1 mg N kg soil⁻¹ day⁻¹) was reached during autumn (26 September to 26 November). Negative values mean that soil mineral nitrogen was uptaken by other processes.

Table 3.6 Impact of drainage treatments from experiment A and B on soil organic carbon (SOC) and soil total nitrogen (TN). Treatments control (C and Cb), mole 1 (M1), mole 2 (M2), gravel mole (GM) and gravel mole into collectors drains (GMb). Standard error of the mean (SEM). Sampling period (SP).

	SP	Year	C	M1	M2	GM	SEM	p-value	
Expt A	SOC (t C ha ⁻¹)								
	SP2	2013	151	138	-	-	5.8	> 0.05	
	SP3	2014	149	149	147	143	4.0	> 0.05	
	SP3	2015	131	119	116	131	4.0	> 0.05	
	TN (t N ha ⁻¹)								
	SP2	2013	16	15	-	-	0.6	> 0.05	
	SP3	2014	16	16	16	15	0.4	> 0.05	
	SP3	2015	15	13	12	14	0.4	> 0.05	
				Cb	GMb				
Expt B	SOC (t C ha ⁻¹)								
	SP4	2014	145				140	4.6	> 0.05
	SP4	2015	90				84	7.3	> 0.05
	SP4	2016	157				138	6.0	> 0.05
	TN (t N ha ⁻¹)								
	SP4	2014	16				15	0.5	> 0.05
	SP4	2015	10				9	0.8	> 0.05
	SP4	2016	17				15	0.6	> 0.05

3.5.6 Herbage yield and N uptake in herbage

Herbage yield and N herbage uptake during SP1, SP2 and SP3 was not ($P>0.05$) affected by drainage treatments, except during SP4 when herbage yields and N uptake of GMb were ($P<0.05$) higher than Cb (Tables 3.6 and 3.7).

Table 3.7 Impact of drainage treatments from experiment A and B on annual herbage yields and herbage N uptake. Treatments control/ undrained (C and Cb), mole 1 (M1), mole 2 (M2), gravel mole (GM) and gravel mole into collectors drains (GMb). Standard error of the mean (SEM). SP = sample period.

Herbage (t DM ha⁻¹ year⁻¹)								
		Years	C	M 1	M 2	GM	SEM	P-value
Experiment A	SP1	2011	13.3	12.5	-	-	0.30	> 0.05
	SP2	2012	11.9	12.7	11.7	11.0	0.52	> 0.05
		2013	12.8	12.9	13.4	13.3	0.48	> 0.05
	SP3	2014 [‡]	15.0	15.4	15.9	15.9	0.62	> 0.05

			Cb			Gmb		
Experiment B	SP4	2015	11.0	-	-	12.3	0.65	<0.01
Herbage N uptake (kg N ha⁻¹ year⁻¹)								
			C	M 1	M 2	GM		
Experiment A	SP1	2011	374.8	352.3	-	-	8.46	> 0.05
	SP2	2012	351.1	374.7	345.2	324.6	15.34	> 0.05
		2013	365.0	367.8	382.1	379.2	13.69	> 0.05
	SP3	2014‡	445.9	453.6	467.2	471.1	17.47	> 0.05
			Cb			Gmb		
Experiment B	SP4	2015	310.0	-	-	467.9	18.32	< 0.01

‡Measurements only between March and July 2014.

3.5.7 Factors affecting daily N₂O emissions

A multiple regression analysis was performed to investigate the effect of WTD and WFPS on soil N₂O flux. Data included in the multiple regression models was WTD $\geq$ 0.1 m deep. Variables included in the model were soil temperature, air temperature, WTD and WFPS. There was an experiment effect hence data from each experiment were analysed separately (Table 3.8).

Table 3.8 Multiple linear regression accounting for variation in mean daily N₂O emissions (experiment A: n = 185 and experiment B: n= 342) using mean daily explanatory variables.

	Variable	Estimate	SE	Semi- partial R ²	P-value	Model R ²
Experiment A N ₂ O§	WTD (cm)	0.002	0.0001	-	<0.0001	0.55
Experiment B N ₂ O§	WFPS (cm)	-0.014	0.005	0.52	<0.001	0.61
	Soil temperature (°C)	0.004	0.001	0.09	<0.001	

§N₂O data were transformed using the logarithmic transformation: log (N₂O).

WTD = daily mean water table depth.

WFPS = daily mean water filled pore space (0 – 5 cm).

Soil temperature = daily mean soil temperature (0 – 5 cm)

The best regression model for annual N₂O emissions for Expt A only included average daily WTD. On the contrary, for Expt B the best fitted model included the average daily soil temperature and WFPS. The latter variable accounted for the largest proportion of variation (highest semi-partial R²) hence variation in N₂O emissions from Expt B. Results of the fitted model showed that the soil N₂O emissions increased with increasing WTD and lower WFPS (Figures 3.5 and 3.6).

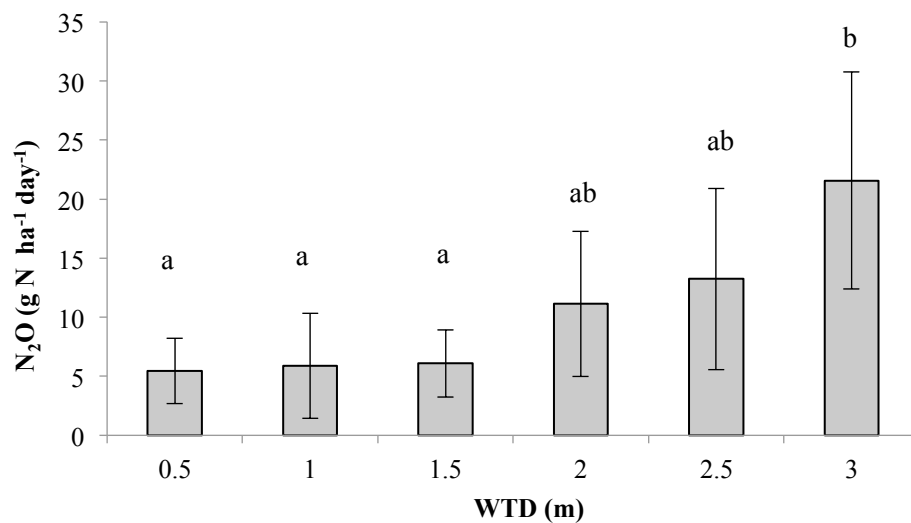


Figure 3.5 Experiment A. Mean daily estimates of N₂O emission at each water table depth (WTD) level ($\pm$ SE).
Bars with the same letter are not significantly different ($P < 0.05$).

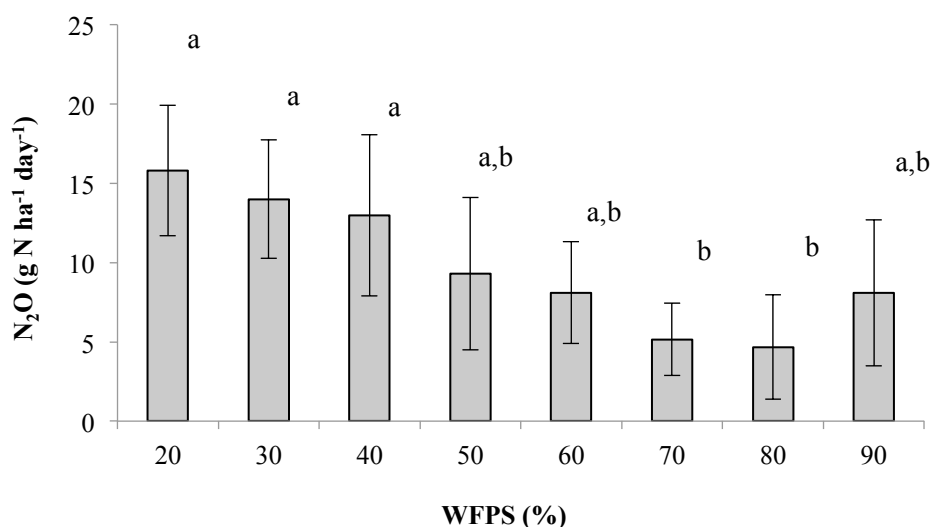


Figure 3.6 Experiment B. Mean daily estimates of N₂O emission in each water filled pore space (WFPS) level ($\pm$ SE). Bars with the same letter are not significantly different ($P < 0.05$).

3.6 Discussion

The aim of this study was to assess the soil GHG fluxes in response to artificial subsurface drainage. In the context of this study on the impact of land drainage on GHG, only soil N₂O, CO₂ and CH₄ fluxes are considered, as these represent the main GHG from agricultural ecosystems (Cole et al., 1997; IPCC, 2013; Paustian et al., 2016).

In Expt A, the first GHG sampling period (SP1) did not start immediately after the drainage treatment was imposed due to time taken to the set-up the experimental measurements. Nevertheless, this study examined the impact of mole and gravel mole drainage on GHG emissions under a range of conditions: (i) moderate to intensive artificial drainage in Expt A; (ii) intensive artificial drainage in Expt B (iii), immediately after drainage installation and during the first twelve months of life-span (SP4); (iv) shortly after drainage installation (SP1); and (v) three years after drainage installation (SP3), when all drainage treatments were established and M1 and M2 were beginning to deteriorate and needed to be re-installed.

3.6.1 The impact of artificial subsurface drainage on WTD, WFPS and herbage production

Annual rainfall during the present study was between 10 and 25% higher than long term average, with the exception of 2013. Mole drainage (M2), during SP2 and SP3, and gravel mole

drainage (GM and GMb), significantly ($P<0.05$) lowered the WTD and WFPS in the present study (Table 3.3). Drainage treatment allowed the discharge of water from the soil profile, which is consistent with previous findings on the same site (Tuohy et al., 2015; Tuohy et al., 2016). Despite this, there were no detectable differences on herbage yield between the drained treatments and the control in Expt A. During Expt B the GMb treatment yielded on average 1.3 ± 0.7 t DM ha⁻¹ year⁻¹ more than Cb. Hence, results of the present study suggest that lowering the WTD can enhance grass growth. This was most evident in Expt B, where more intensive artificial drainage was imposed than in Expt A. This is in agreement with previous studies that reported a reduction on crop production by up to 30% due to shallow WTD (Brereton and Hope-Cawdery, 1988; Burke et al., 2004; Laidlaw, 2009).

Drainage offers other advantages in terms of trafficability (Armstrong, 1986) and length of the grazing season. Gupta and Gupta (1987) reported that mole drainage on grasslands under high rainfall reduced poaching damage, increased yield and extended the grazing season by 6 weeks. In the present study there was clear evidence to show that the artificial drainage, particularly gravel mole drainage, lowered soil moisture at the soil surface.

3.6.2 The impact of artificial subsurface drainage on soil N and soil N₂O flux

3.6.2.1 Herbage N uptake, soil TN and N net mineralisation and nitrification

The high levels of herbage production in this study are reflected in N uptake in herbage with annual means ranging between 310 kg ha⁻¹ and 471 kg ha⁻¹ (Table 3.7). Although no artificial fertilizer N was applied to treatments during this study it is clear that the soil, enhanced by biological N fixation in association with white clover, had substantial capacity to supply N to fuel herbage production. The implication is that there was also substantial capacity to lose N from the soil through the processes of nitrification and denitrification.

In the present study there were no ($P>0.05$) impact of treatments on TN, soil net N mineralisation and nitrification, which, for the most part, is consistent with herbage production and N uptake in herbage. In a previous study by Nangia et al., (2013) soil net N mineralisation and nitrification rates were also not affected by drainage treatment. In the present study this can be attributed to the relatively small differences in WFPS between drained and undrained treatments (Table 3.3) and the absence of differences in soil temperature between treatments.

3.6.2.2 Influence of subsurface artificial drainage on soil cumulative N₂O emissions

The above results are consistent with the absence of differences in cumulative N₂O emissions between treatments during this study. Annual cumulative N₂O flux of 1.6 ± 0.9 kg N₂O-N ha⁻¹ recorded during SP4 is similar to emissions previously recorded for grassland at Solohead

Research Farm (Necpálová et al., 2013b; Burchill et al., 2014) and elsewhere in Europe. For example, Burchill et al. (2014) reported annual cumulative emission from grassland of 1.7 ± 0.8 kg N₂O-N ha⁻¹. The present results are similar to the range, -0.5 to 1.2 kg N₂O-N ha⁻¹, reported by Flechard et al., (2007) for multiple grassland sites across Europe. Cumulative N₂O emissions (0.8 ± 0.4 kg N₂O-N ha⁻¹) during SP3 were slightly higher than SP4 and the previous cited studies, for the equivalent time period, which can be attributed to the inter-annual variation similar to that reported by Burchill et al., (2014).

3.6.2.3 Influence of subsurface artificial drainage on daily N₂O emissions

Although there were no significant differences in cumulative N₂O fluxes between treatments, mean daily N₂O emissions were episodic and peaked when WTD was the deepest (summer). In the present study according to the regression results, WTD and WFPS statistically accounted for the 55% and 52% of the variability of N₂O flux from Expt A and B respectively. This agrees with previous studies (Kliewer and Gilliam, 1995; Smith et al., 2003). Moreover, results of the present study indicate that when WTD was more than 2.5 m and up to 3 m deep, the N₂O emissions were significantly higher ($P < 0.01$) than when WTD was 2.5 m or less from the soil surface (Figure 3.5). This was not detected in Expt B, partly because the deepest WTD recorded at this experiment was 1.3 m. Nevertheless, in Expt B a similar relationship was found between N₂O emissions and WFPS (Figure 3.6). It was shown that when WFPS was between 30 and 40% (dry conditions) the N₂O flux was significantly higher ($P < 0.05$) than when it was between 70 and 80%. Similar results were reported in the Netherlands by Van Beek et al., (2010), where emissions from the dry fields were about three times higher than the emissions from the wet fields. Goldberg et al., (2010), in Germany, also observed peaks on N₂O emissions, however concluded that artificially drained plots did not have any impact on N₂O flux. In the latter study artificial drainage caused a moderate reduction in WTD (i.e. 0.1 to 0.3 m deeper than undrained plots). In contrast Grossel et al., (2016) conducted an experiment in a cereal field in France and concluded that shallower WTD resulted in significantly higher N₂O emissions, however conditions on the latter study were very dry (i.e. annual precipitation 598 and evapotranspiration 740 mm) compared with the study of Goldberg et al., (2010) and the present study.

Conditions such as annual precipitation, evapotranspiration, and soil type have a direct impact on soil aerobic and anaerobic status. Authors from previous studies claimed that by artificially lowering the WTD the soil shifted from anaerobic conditions (and incomplete denitrification generating N₂O) towards aerobic conditions resulting in lower N₂O flux from the artificially drained treatment (Dobbie and Smith, 2006; Tesfai et al., 2015; Grossel et al., 2016). However, in the present study results suggest that when WTD was closer to the soil surface the soil shifted

from wet to saturated conditions creating more favourable conditions for complete denitrification and hence the generation of dinitrogen gas (N_2) and concurrently lower N_2O flux. When WTD was deeper (below 2.5 m from the soil surface) drying conditions in the soil promoted incomplete denitrification to become the dominant process increasing the N_2O flux (Stevens et al., 1997; Ball, 2013). However another possible explanation might be the effect of soil temperature and N availability for N_2O flux production. Soil temperature explained the 9% of N_2O flux variability but only in Expt B. However the seasonal changes on soil mineralisation and nitrification rates showed that despite there was not a treatment effect there was greater available N in the soil when WTD and WFPS was the lowest (during summer, section 3.5), hence this could have enhanced the soil N_2O emissions.

Overall in our study the effect of WTD on the N_2O flux was only observed when WTD was below 2.5 m from the soil surface. At this depth the artificial subsurface drainage systems did not control the WTD (installation depth 0.55 m from the soil surface), hence this explains why there were no significant difference between treatments.

3.6.3 Influence of subsurface artificial drainage on soil carbon pools

Artificial land drainage techniques are associated with microbial oxidation of the SOC and subsequent soil GHG realised from the soil to the atmosphere (Six et al., 2004; Buss et al., 2005; Velthof et al., 2010). In the present study there was no ($P>0.05$) impact of drainage on SOC, which can be attributed to the relatively small differences in WFPS (Table 3.1) and the unchanged soil temperature between treatments. In contrast O'sullivan et al., (2015) reported that land drainage techniques in Ireland contributed to SOC losses. The dissimilar results between the latter modelling study and the present study might be due to the timeframe under examination. While in the current study SOC losses were evaluated over a three year period following installation of drainage treatments, the study of O'sullivan et al., (2015) examined the losses with a model that simulated SOC losses over 30 years.

3.6.3.1 Soil CH_4 flux The lack of impact of drainage treatment on the soil carbon pools was reflected on the soil cumulative CH_4 flux. Previous studies have found that by lowering the WTD with artificial drainage the soil CH_4 emissions decreased (Rizzo et al., 2015). In the present study the WFPS and WTD were significantly lowered by drainage treatments, however it was not sufficient to influence the CH_4 oxidation rate. In the current study mean daily fluxes from all the sampling periods averaged (mean $\pm$ SE) -0.91 ± 3.7 g CH_4 -C ha⁻¹ day⁻¹, results were similar to results reported by Schaufler et al., 2010. Authors from the latter study conducted

measurements on incubated soil cores from across Europe. There were three grassland sites: UK, Hungary and Switzerland. The UK soil had a similar and negative mean daily flux (mean $\pm$ SE; -0.312 ± 0.216 g CH₄-C ha⁻¹ day⁻¹) to the current study, on the contrary the Swiss and Hungarian soil had positive fluxes averaging 0.6 ± 0.1 and 0.072 ± 0.168 g CH₄-C ha⁻¹ day⁻¹ respectively. Differences between daily emissions from the sampling locations could be attributed to the differences in the climatological zones. Overall, annual cumulative fluxes from the present study were < 1 kg CH₄-C ha⁻¹ year⁻¹, which is also in agreement with the results presented by Merino et al., 2004.

3.6.3.2 Soil CO₂ flux These present results are consistent with the absence of impact of drainage treatment on soil CO₂ flux. Soil CO₂ flux is primarily influenced by soil temperature (Schlentner and Cleve, 1985; Angers et al., 2010; Koncz et al., 2015) which was unchanged by the drainage treatments. However, previous studies have reported that soil CO₂ flux increased when dry matter production increased (Ludlow, 1985; Schaufler et al., 2010). In contrast, Brito et al. (2015) detected that CO₂ flux was decreased when pasture height was higher. In the present study despite higher grass growth on Gmb was observed during SP4 this did not result in higher CO₂ flux than Cb. During SP4 the shallow WTD recorded over the winter most likely had reduced the photosynthesis and growth rates, driven by anaerobic stress (Burke et al., 2004; Laidlaw, 2009). The latter accounts for the positive correlation between WTD and herbage production. However the majority of soil CO₂ emissions were observed during the summer months, when soil temperatures were highest (Figure 3.3f). During the summer WTD was at its deepest and not controlled by drainage treatment. This explains why no differences in CO₂ flux were detected between treatments.

Finally annual soil CO₂ flux from the present study (14274.5 ± 258.51 kg CO₂-C ha⁻¹ year⁻¹) were within the range of previously reported values from European grassland sites that ranged from 580 to 19880 kg CO₂-C ha⁻¹ year⁻¹ (Bahn et al., 2008) and very similar to other studies from grasslands that ranged between 15000 and 19000 kg CO₂-C ha⁻¹ year⁻¹ (Davidson et al., 2000; Owensby et al., 2006).

3.6.4 Impact of drainage treatment on the greenhouse gas budget

Results of the present study show that drainage treatment successfully lowered the WFPS, deepened the WTD and significantly increased the herbage yield during SP4. However the cumulative GHG fluxes were unaffected due to (i) the lowered WFPS was not sufficient to influence soil N₂O and CH₄ flux and (ii) the reduced herbage yield on the undrained treatments took place during shallow WTD and did not affect the CO₂ cumulative flux. Those are

consistent with the calculated cumulative CO₂-C_{eq} presented in Table 3.4. Drainage treatment did not have a significant impact on the GHG budget.

We calculated that on average soil GHG emissions amounted to 1.5 t CO₂-C_{eq} ha⁻¹ year⁻¹ across all the treatments. The major source of GHG cumulative emissions was soil respiration (CO₂), this is in agreement with Merbold et al., (2014) who conducted a full GHG budget on intensively managed grassland in Switzerland and measured annual emissions of 1.3 t CO₂-C_{eq} ha⁻¹ year⁻¹.

3.7 Conclusions

Mole and gravel mole drainage treatments lowered the WTD, decreased the WFPS and had no influence on SOC or TN, but increased herbage yield. However cumulative GHG emissions were unaffected. Furthermore, the impact of artificial drainage on WFPS was not enough to cause a detectable difference in cumulative N₂O or CH₄ flux. Increased herbage yield did not impact on soil CO₂ flux, which was mainly influenced by soil temperature and mostly occurred during the summer, when drainage treatment did not impact on the WTD. Furthermore, results suggest that N₂O flux increased when WTD was deeper under the particular conditions of the present study. This indicates that the relationship between WTD and N₂O emissions is complex and there are many processes involved in N₂O production throughout the soil profile.

Overall, mole and gravel mole drainage techniques for improving trafficability and grass utilization did not increase the extent of soil GHG emissions under the management and conditions described in the present study. Further studies are required in order to identify and understand the interactions of the underlying processes involving the soil N₂O emissions.

CHAPTER 4: SIMULATING SOIL GREENHOUSE GASES FROM IRISH GRASSLAND UNDER DRAINAGE MANAGEMENT

Based on: N. Valbuena-Parralejo, M. Necpálóva, M. Williams and J. Humphreys. Agriculture Ecosystems and the Environment (submitted September 2019)

4.1 Abstract

Shallow drainage (e.g. gravel mole drainage installed at 0.40 m below soil surface) is used on pasture-based farms to reduce soil moisture, deepen water table depth and thus increase grass utilization on soils with impeded drainage. There is a lack of research on the effect that this drainage technique has on soil greenhouse gas (GHG) emissions, particularly under changing climate. Hence, the objective of this study was to investigate the effects of gravel mole drainage and weather variability on soil GHG fluxes. Data was collected from two experiments with undrained control (C) and a gravel mole drainage (GM) arranged in randomized complete block design on a permanent grassland with impeded drainage on eutricgleysol (humic) in Ireland. Soil biogeochemical properties and GHG fluxes were measured between 2014 and 2015. We calibrated and validated DNDC ecosystem process model with the latter dataset. We used historical weather data (1986-2013) to generate 28 weather scenarios. These were used to analyse effects of weather variability on the soil GHG emissions and to identify key drivers of soil GHG fluxes in C and GM. The GM treatment ($P < 0.05$) decreased (mean $\pm$ SE) the water filled pore space (WFPS) by $7.0 \pm 0.2\%$. However no ($P > 0.05$) impact of GM treatment was detected on cumulative soil CO_2 and CH_4 fluxes (mean $\pm$ SE) over the 27 scenarios (1554 ± 20.9 kg $\text{CO}_2\text{-C ha}^{-1} \text{ year}^{-1}$ and -0.48 ± 0.12 kg $\text{CH}_4\text{-C ha}^{-1} \text{ year}^{-1}$). Despite GM treatment and weather variability did not impact cumulative CO_2 and CH_4 emissions, for N_2O emissions weather variability increased those on warmer and wetter years and the identified key drivers of emissions were different for GM and C. Cumulative soil N_2O emissions were ($P < 0.05$) lower for GM than for C (1.4 and 1.7 ± 0.07 kg $\text{N}_2\text{O-N ha}^{-1} \text{ year}^{-1}$, respectively) and were primarily influenced by dissolved organic carbon and ammonium under C, and soil organic carbon and WFPS under GM. These results indicate a significant decrease in soil temperature and WFPS in response to drainage significantly decreases the annual cumulative N_2O emissions, which are much more potent and persistent GHG than CO_2 and CH_4 .

4.2 Introduction

Permanent grassland accounts for 68% of agricultural land globally (FAO, 2017). Pasture-based systems are, in general, concentrated in temperate grassland regions, where they predominate in areas with high rainfall, low evapotranspiration and soils with poor-drainage characteristics, often unsuitable for other forms of agricultural production (Humphreys et al., 2009). Such conditions coincide with poor soil trafficability by livestock and machinery and poor grass utilization by grazing (Fitzgerald et al., 2008; Tuohy et al., 2016). Grass utilization under grazing is one of the main components underpinning the profitability in these systems (Dillon et al., 2005). To improve grazing conditions in these areas, gravel mole drainage have been widely used as a shallow drainage techniques to reduce the soil moisture content by lowering the watertable depth (WTD) and improving surface water infiltration (Ibrahim et al., 2013; Tuohy et al., 2015).

Recently, there is a global concern related to the increase in greenhouse gas (GHG) emissions from agricultural activities, primarily CO₂, N₂O, and CH₄ and their impact on climate change. Soil GHG fluxes, that contribute to the global GHG emissions (Robertson et al., 2000), are highly dependent on the soil microbial activity, which is regulated by soil moisture content, soil temperature, soil pH and available soil organic carbon (SOC) (Chadwick et al., 2011; Curtin et al., 2000; Smith et al., 2003). Global warming potential of CO₂, CH₄ and N₂O is 1, 28 and 298, in addition atmospheric lifetime can be on average 100 years. The direction and magnitude of changes in SOC, N₂O and CH₄ emissions associated with the management implementation compared to conventional management depends on pedo-climatic conditions and historical land use (Smith, 2012). Previous research has shown that soil disturbance, for example by ploughing, decreased soil moisture content and increased soil temperature (Davidson et al., 2000; Lal, 2004; Angers et al., 2010) and thereby significantly increased soil N₂O and soil CO₂ fluxes (Schlesinger and Andrews, 2000; Garcia-Marco et al., 2016). Despite a wide application of gravel mole drainage techniques (North west Europe, New Zealand and Australia) there is a lack of research on how this management affects key drivers and biogeochemical processes leading to soil GHG emissions compared to undrained grasslands.

Process-based biogeochemical ecosystem models, that are robustly calibrated and validated with empirical data, provide an opportunity to understand and quantify how changes in management or climate impact the ecosystem responses of the agriculture systems, particularly compare alternative management scenarios intended to reduce emissions (Abdalla et al., 2010;

Iqbal et al., 2018). This capability being more relevant under a changing climate, where measurements of emissions cannot be easily obtained (Gillespy et al., 2014). Moreover, models with adequate representation of the modelled processes allow to explain the combined effects of processes contributing to the observed results. Furthermore, previous research has shown that dynamic ecosystem process-based models can be used to evaluate interactions between pedo-climate properties and agricultural management, and are thus considered as powerful tools for optimizing management practices or production systems under complex environmental conditions (Fitzgerald et al., 2008).

The international panel of climate change (IPCC) has expressed a growing certainty in global warming and climate change and that human activity is a principal cause (IPCC, 2018). In order to reduce the level of global warming well below 2 °C by the end of century, all countries have to establish a feasible climate change mitigation strategies in the agriculture sector that does not compromise capacity for sustainable food production (Pachauri et al., 2014).

Previous studies across north-west Europe have reported that climate change has a major influence on grassland productivity due to the changing weather patterns (Fitzgerald et al., 2008). It is expected that in Atlantic central regions like Ireland extreme weather episodes, such as droughts and flooding, will occur more often (Reheul et al., 2017). Moreover, the agriculture GHG emissions in Ireland, mainly from enteric fermentation, soil management and manure management are projected to increase by 6-7% during the period of 2014-2020 (EPA, 2016) in response to milk quota abolition and intensification of agriculture. Despite the great concern about the increasing trend in soil GHG emissions, there are very few studies examining how weather variability affects the processes and overall soil GHG emissions from grassland production systems (Bell et al., 2012; Del Prado et al., 2013; Chang et al., 2017). Bell et al., (2012) assessed the effect of future climate scenarios on the balance between productivity and GHG emissions from sheep grazing systems, authors concluded that within grazing systems, the warmer temperatures will likely increase soil N₂O emissions by denitrification, although the productivity and emissions changes under climate change will heavily dependent on the pedo-climatic conditions and pasture species.

The objectives of this study were to investigate the effects of gravel mole drainage and the weather variability on soil GHG emissions through DNDC modelling. We i) calibrated and validated the DNDC model to represent the modelled agroecosystem and drainage management using empirical data collected at the single grassland site in South-East Ireland over 2014 and 2015; ii) studied the effects of undrained and drained managements and weather variability on

soil GHG emissions at this experimental site (this was achieved by running the model under 27 weather scenarios, representing 27 years of historical weather data recorded over a period 1986 to 2013); and iii) we identified key drivers of soil GHG emissions under undrained and drained soil conditions. We considered impact only on soil N₂O, CO₂ and CH₄ emissions.

4.3 Materials and Methods

4.3.1 Field observations

The study site is a permanent grassland with impeded drainage in Solohead, Ireland (52°30'N, 08°12'W). It has a mixed maritime/humic oceanic climate with high, annual rainfall (mean of the 10 preceding years 1089 mm). Mean annual air temperature was 11.5°C. The soil is classified as humic eutricgleysol according to the World Reference Base (WRB, 2014). Soil characteristics are describe in Table 1. The study site (4 ha) has a slope of 1.4% and mean perched water table at 1 m below ground level (BGL) (Figure1). A collector drain divided the site in two site areas, hereafter referred as Experiment A (Expt A) and Experiment B (Expt B). It was excavated 2 m BGL intercepting perched groundwater along the lower axis of Expt A. The remaining sides were hydrologically isolated using a ditch 1 m deep and filled with aggregate (100 to 50 mm). The area under Expt A was divided in four blocks (60 x 100 m). Each block was sub-divided into four main plots (15 m x 100 m). There were four treatments (with four replicates) allocated to the main plots in a randomized complete block design. The four treatments consisted of (i) undrained control (C), (ii) and (iii) mole drainage installed at a depth of 0.55 m in January and July 2011, respectively and (iv) gravel mole drainage installed at a depth of 0.40 m in July 2011 (GM). In order to have a continuous two-year data set across the two experimental areas, for modelling we selected C and GM treatments, which represent the most contrasting managements in the experimental design. Each main plot contained two sub-plots (2 x 10 m) near the highest point of the plot (deep water table (WT)) and in the lower point of the plot (shallow WT). All experimental measurements were taken from sub-plots, those were un-grazed and unfertilized in order to assess the impact of drainage treatments without other confounding factors.. Area under Expt B was divided into four blocks (80 x 60 m). Each block was subdivided into two 40 × 60 m plots the way that the long axis of each plot run perpendicularly to the open drain. There were two treatments arranged within blocks in a randomised complete block design: (i) gravel mole drained (GM) and (ii) undrained control (C). There were two sub-plots (2 × 11 m) laid down at randomly assigned locations in the centre (well away from the edges) of each plot. Similar to those described above, two piezometers to

monitor WT depth were installed in each sub-plot. Sub-plots were managed the same as in Exp A.

Soil GHG fluxes, soil WT depth, soil water filled pore space (WFPS) was measured on a weekly basis . The aboveground biomass was also recorded over the whole experimental period. Soil N₂O and CH₄ fluxes were measured on a weekly basis using the static chamber technique in each treatment sub-plot in Expt A between January and June 2014 and in Expt B between July 2014 and July 2015 and subsequently analysed by gas chromatography. Soil respiration (CO₂) flux was measured weekly, between 10:00 am 1:00 pm, using CIRAS-1 with SRC chamber (PPS Systems, USA). This measurement interval was selected to represent daytime soil respiration average.

4.3.2 DNDC model overview

The DNDC (DeNitrification-DeComposition) is a process-based model of soil C and N biogeochemistry in agroecosystems (Li et al., 1992a,b, 2000) used to examine potential impacts of management and climate change in agriculture. DNDC simulates crop growth, soil temperature and moisture, soil carbon dynamics, nitrogen leaching, and emissions of greenhouse and trace gases including N₂O, nitric oxide, dinitrogen, ammonia (NH₃), CH₄ and CO₂ at a daily time step. The primary ecological drivers (i.e. model inputs) such as climate, soil, vegetation and management determine soil temperature, moisture, pH, redox potential and substrate concentration gradients (NO₃⁻, NH₄⁺ and dissolved organic C). These then control microbially mediated biochemical processes of nitrification, denitrification and fermentation. The model consist of six interacting sub-models: soil-climate/thermal-hydraulic, aerobic decomposition, denitrification, nitrification, fermentation, and plant growth as affected by land management (Li et al. 1992; Li et al. 1994, Li, 2000). Nitrification rate is calculated from soil NH₄⁺ and NH₃ under aerobic conditions and N₂O from nitrification represents a fraction of the nitrification rate. Denitrification is initiated once soil is saturated. Following soil saturation, soil reduction potential is determined with the Nernst thermodynamic equation and used to determine anaerobic microbial functional group activity under a given soil conditions. When microbial capacity is established, the oxidation/reduction reaction rates are primarily driven by the relevant substrate concentrations using the Michaelis-Menten Equation.

4.3.4 Model set-up, calibration and validation

In this study the DNDC model version 9.3 (2010) was used to simulate WFPS, grass yields and soil GHG emissions in response to the drainage management and weather variability.

The drainage treatments were imposed with time series of plot-specific WT depth input data and input on WFPS at field capacity (FC). The WFPS at FC for each modelled plot was estimated from measured time series of WFPS. The simulations were executed at the plot level as soil properties, WT, WFPS, soil GHG and yield measurements were available for each treatment plot.

The model calibration and validation simulations were conducted for eight plots (four plots under C and GM management, respectively). Model calibration was based on data collect in 2014, while independent validation on data from 2015. Each simulation was driven by plot specific soil inputs, i.e. pH, bulk density, porosity, soil NO_3^- and NH_4^+ and SOC at 0–30 cm. Saturated hydraulic conductivity and WFPS at wilting point for each plot were estimated from soil texture and SOC using pedotransfer functions (Saxton and Rawls, 2006) embodied in the Soil Water Characteristics Calculator software (SWCC), version 6.02.74 (USDA Agricultural Research Service, Washington). These simulations were driven by 2014 and 2015 weather data (i.e., daily precipitation, maximum and minimum temperature) recorded on a local weather station (Campbell Scientific Ltd. Loughborough, UK). Model performance was evaluated against cumulative soil N_2O , CO_2 , CH_4 , WFPS, and grass yields over the same period. The model was firstly run with default DNDC parameters (DEF simulations). Then we improved the model capacity to simulate WFPS, aboveground biomass, soil N_2O , CO_2 and CH_4 emissions. The aboveground biomass was calibrated by adjusting the maximum biomass productivity, soil biological N fixation and plant biomass C:N ratio based on previous measurements on the site (Burchill et al., 2016). How were the other outputs calibrated?

The model performance against calibration and validation data set was determined using various statistical criteria e.g. bias error (bias), absolute root mean square error (RMSE) and relative root mean square error (rRMSE). The independent model validation determined the level of confidence for the predictions under these site-specific conditions.

4.3.5 Model application

Through the DNDC modelling, we examined the impact of drainage management and weather variability on soil GHG emissions at this experimental site. This was addressed by running the model under 28 scenarios, representing 28 years of historical weather data collected over period 1986 to 2013. As no site-specific weather data was available for a longer period, data from the

nearest weather station provided by the Irish meteorological services (Met Eireann) within a distance of 30 km of the site, was used.

Initially, we analysed the 28 years of weather data and identified the year 2000 as a base line year (BLY), as a year that represented the average precipitation and air temperature of the dataset, with an annual precipitation (1036 mm) and air temperature (10.3 °C) close to the long-term means (Figure 4.1). One year long weather scenarios were representative of weather conditions (daily precipitation, maximum and minimum air temperature) of individual years from 1986 up to 2013. Annual precipitation, air temperature and intra-annual variation in these variables was highly variable between the years, providing a robust dataset for studying the sensitivity of soil GHG emissions to local climate. Model simulations for these scenarios were driven by soil properties of C and GM plots used for independent model validation. In order to achieve identical initial conditions for each weather scenario, and thus to disentangle residual effects of previous years from the climate effects imposed in weather scenarios, we ran the model with the baseline weather data for 5 years prior to each weather scenario. This was also important to ensure a sufficient crop growth sub-model stabilization. Moreover, we also investigated sensitivity of soil GHG emissions to the variables such as soil WFPS, soil temperature, mineral N, SOC content and grass yield, and thereby identified key drivers under C and GM conditions. This was studied across all the weather scenarios and independently for each treatment.

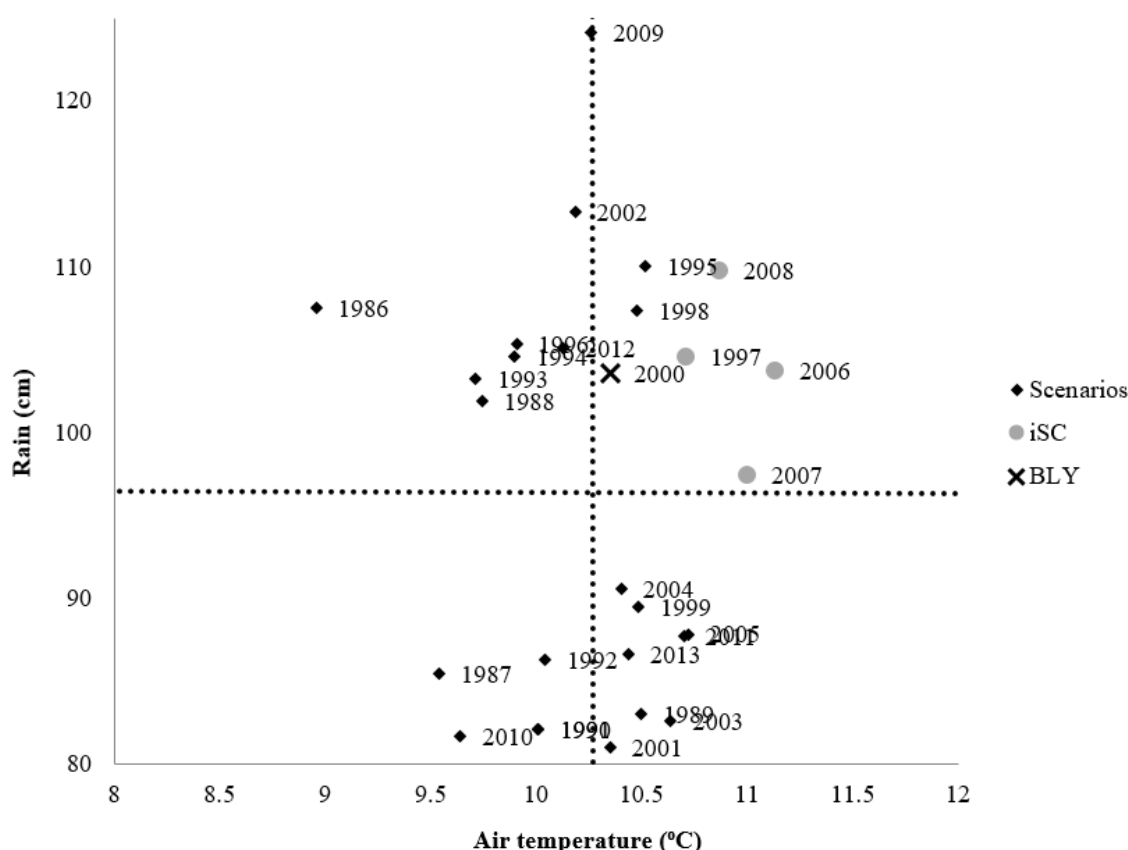


Figure 4.1 Annual cumulative precipitation and annual average air temperature from 1986 to 2013 recorded at the weather station 30 km from the study location, used to study the effect of changing climate on soil greenhouse gases. Years (1986 to 2013) are classified as follows: year 2000 = base line year (BLY), years 1997, 2006, 2007 and 2008 = identified scenarios (iSC) with significantly greater soil N₂O emissions from C and GM treatments than the BLY, and the other years = other climate scenarios. Vertical and horizontal dashed lines represent the annual average air temperature and annual average cumulative precipitation from 1986 to 2013

4.3.6 Calculations and statistical analysis

Cumulative soil N₂O, CO₂ and CH₄ (kg ha⁻¹) emissions for the entire observation periods were calculated by linear interpolation between sampling occasions as described in Valbuena-Parralejo et al. (2019). Simulated and measured values were compared taking into account the same start and finishing date of measurement, by linear interpolation of the daily output.

The GHG emissions were converted to CO₂ equivalents (CO₂ eq) using the 100 year horizon global warming potentials of 28 for CH₄ and 265 for N₂O (IPCC, 2014). The net GHG budget (kg CO₂ eq ha⁻¹) accounted for soil production and consumption processes for soil N₂O, CH₄ emissions and soil respiration (CO₂) on an annual basis.

Simulated soil cumulative N₂O, CO₂, CH₄, net soil GHG emissions and mean soil WFPS were subjected to ANOVA using SPSS (Armonk, NY: IBM Corp, USA). The model included

following fixed factors: drainage treatment, climate scenario, treatment by climate scenario interaction and experimental block. Climate scenario was included as a repeated measurement. All data were checked for normality; N₂O data required a natural logarithmic transformation [$y = \log(x)$]. The weather scenarios with significant effect on GHG emissions were identified with a Tuckey's test. In order to identify the key drivers and processes in GHG emission production, the whole dataset was subjected to several multiple linear regression (stepwise) analyses. Two types of multiple linear regression analyses were performed, one to identify the key drivers and the other one to identify predominant processes. The explanatory variables considered were the following: a) soil temperature, soil WFPS (0 to 50 cm), dissolved organic C (dSOC) (0 to 50 cm), soil mineral N (0 to 50 cm) and SOC (0 to 50 cm) for soil N₂O emissions; b) dSOC (0 to 50 cm), aboveground and belowground plant biomass, soil temperature (0 to 50 cm), soil WFPS (0 to 50 cm), root C litter input and SOC (0 to 50 cm) for soil CO₂ emissions; and c) soil mineral N (0 to 50 cm), dSOC (0 to 50 cm), leached C, soil temperature (from 0 to 50 cm depth), aboveground and belowground plant biomass, WFPS (0 to 50 cm) and SOC (0 to 50 cm) for soil CH₄ flux (Hashimoto et al., 2011). The selection of the drives was based on the per-reviewed literature (Hashimoto et al., 2011; Horrocks et al., 2015; Louro et al., 2013). The variables were averaged across several soil horizons to a depth of 50 cm as the drainage management is assumed to affect soil profile at least down to 45 cm depth. Highly correlated variables with $r \geq 0.7$ were excluded from the analyses in order to avoid multicollinearity assumption. As the main aim was to determine the key driving variables of soil GHG emissions, the interaction effects were omitted form the model statements. In order to identify predominant N₂O production processes, i.e., nitrification vs. denitrification, the N₂O emissions were also regressed against the rates of these processes. In all the linear regression models, the explanatory variables with higher beta standardised regression coefficients as well as partial correlation coefficients were considered as the most influential in the production of the particular GHG. Explanatory variables with a negligible influence on the response variables were eliminated by a stepwise regression model selection technique. Statistical probability of $P < 0.05$ was considered significant for all statistical tests.

4.4 Results

4.4.1 Model calibration and evaluation

The mean measured and simulated values are presented in Table 4.2, results on the model performance for the default model run, model calibration and evaluation are presented in Table 4.3. In general, all indicators showed that the calibration of the model improved the

performance of the model simulating the variables of interest. Under evaluation, model performed very well simulating the soil WFPS. The mean absolute error (MAE), relative mean absolute error (RMAE) and *Bias* were very small.

Table 4.1. Mean measured and modelled WFPS (0 to 30 cm depth), grass yield and soil GHG cumulative emissions simulated from undrained (Control) and Gravel Mole drained treatments by the default (DEF), calibrated (CAL) and evaluated (VAL) model. Measured (M) and simulated (s) values. Presented values are mean and between brackets standard deviation.

C						
		WFPS (%)	Grass yield (kg C ha ⁻¹ ya ⁻¹)	N ₂ O (kg N ha ⁻¹ ya ⁻¹)	CO ₂ (kg C ha ⁻¹ ya ⁻¹)	CH ₄ (kg C ha ⁻¹ ya ⁻¹)
DEF	Measured	0.74 (0.01)	7099 (72)	0.92 (0.55)	422 (94)	0.45 (0.88)
	Simulated	0.54 (0.00)	5000 (109)	0.03 (0.01)	244 (34)	-0.51 (0.08)
CAL	Measured	0.74 (0.01)	7099 (72)	0.92 (0.55)	422 (94)	0.45 (0.88)
	Simulated	0.85 (0.02)	6914 (0)	0.86 (0.15)	442 (94)	-0.12 (0.02)
VAL	Measured	0.79 (0.09)	7194 (257)	1.65 (0.74)	1375 (69)	-0.81 (0.33)
	Simulated	0.85 (0.02)	7114 (0)	0.95 (0.14)	1242 (88)	-0.23 (0.07)
GM						
DEF	Measured	0.65 (0.01)	6730 (579)	0.66 (0.08)	451 (32)	-0.14 (0.38)
	Simulated	0.54 (0.00)	4938 (102)	0.02 (0.01)	228 (27)	-0.48 (0.09)
CAL	Measured	0.65 (0.01)	6730 (579)	0.66 (0.08)	451 (32)	-0.14 (0.38)
	Simulated	0.79 (0.01)	6981 (0)	0.57 (0.04)	424 (49)	-0.16 (0.03)
VAL	Measured	0.77 (0.08)	7284 (104)	1.49 (0.21)	1479 (21)	-0.41 (0.42)
	Simulated	0.77 (0.02)	7114 (0)	0.67 (0.15)	1286 (90)	-0.36 (0.06)

Table 4.2. Model performance indicators presented for each modelled variable simulated with a default model (DEF), during model calibration (CAL) and validation (VAL) for undrained control (C) and gravel mole drained (GM) treatments: bias error (bias), root mean square error (RMAE), and Willmott's index and mean absolute error (MAE).

			WFPS (%)	Grass yield (kg C ha ⁻¹ ya ⁻¹)	N ₂ O (kg N ha ⁻¹ ya ⁻¹)	CO ₂ (kg C ha ⁻¹ ya ⁻¹)	CH ₄ (kg C ha ⁻¹ ya ⁻¹)
Bias	DEF	C	0.20	2098.9	0.89	177.7	0.97
		GM	0.11	1791.3	0.64	223.1	0.34
	CAL	C	-0.11	118.60	0.06	-20.05	0.58
		GM	-0.14	-0.80	0.09	21.51	0.02
	VAL	C	-0.06	80.20	0.70	132.81	-0.59
		GM	0.00	169.67	0.83	193.16	-0.05
RMAE	DEF	C	0.27	0.30	0.96	0.41	0.42
		GM	0.17	0.26	0.97	0.49	-58.96
	CAL	C	0.15	0.02	0.53	0.13	0.25
		GM	0.21	0.02	0.17	0.12	-20.02
	VAL	C	0.14	0.03	0.33	0.09	-0.67
		GM	0.10	0.02	0.54	0.13	-1.12
Willmott's index	DEF	C	0.04	0.05	0.46	0.429	0.48
		GM	0.14	0.32	0.16	0.186	0.44
	CAL	C	0.07	0.41	0.54	0.799	0.49
		GM	0.12	0.01	0.31	0.436	0.11
	VAL	C	0.21	0.37	0.53	0.36	0.37
		GM	0.17	0.41	0.25	0.07	0.24
MAE	DEF	C	0.20	2098.87	0.89	177.74	0.97
		GM	0.11	1791.27	0.64	223.14	0.45
	CAL	C	0.11	118.60	0.40	49.97	0.77
		GM	0.14	106.15	0.11	55.60	0.29
	VAL	C	0.11	201.28	0.70	132.81	0.59
		GM	0.07	169.67	0.83	193.16	0.30

Default model = DEF, model calibration = CAL and model validation = VAL. C = undrained control; GM = gravel mole drained

Water filled pore space (WFPS) (0 to 30 cm depth).

The model succeeded to simulate aboveground biomass even if it was slightly underestimated (*Bias*), however the simulated differed on averaged only 3% from the mean measured aboveground biomass for both treatments. Moreover the model simulated it with a precision of 236.84 and 192.29 kg C ha⁻¹ y⁻¹ that represented 3 and 2% fraction of the mean measured aboveground biomass for C and GM.

Despite the model performed very well simulated the above variables, it slightly underestimated the overall soil GHG emissions. However, soil N₂O and CH₄ flux were simulated relatively well. The model simulated N₂O flux with a precision of 0.90 and 0.88 kg N ha⁻¹ y⁻¹ that represented the 56 and 59% fraction of the mean measured N₂O flux from C and GM. The CH₄ flux was simulated with a precision of 0.68 and 0.38 kg C ha⁻¹ y⁻¹ which represented the 82 and 92% fraction of the mean measured soil CH₄ flux from C and GM. Furthermore, Willmott's index indicated that both simulated soil fluxes followed the variability of measured fluxes fairly well. On the contrary, the model measured slightly poorly the soil CO₂ flux. Flux was under predicted and the overall model error for this simulation was 162.99 kg C ha⁻¹ y⁻¹. However, the model was able to measure the CO₂ flux with a precision of 132.81 and 193.16 kg C ha⁻¹ y⁻¹ that represented the 12 and 14% fraction of the mean measured CO₂ flux from C and GM treatments. Moreover, the Willmott's index indicated that soil CO₂ flux followed the variability of the measured flux fairly well for both treatments.

4.4.2 Weather scenario characteristics

Average annual rainfall across the scenarios was 1036.3 mm. The lowest rainfall recorded was 810.3, 817.4 and 821.2 from years 2001, 2010 and 1990 respectively. Whereas years 2009, 2002 and 1986 recorded 1241.9, 1133.0 and 1075.5 mm. Mean air temperature from all the scenarios was 9.5°C, minimum air temperature recorded was -9.7°C in December 2010 and maximum was 28.8°C, recorded in July 2006. Finally, the averaged soil WFPS was 78%, the lowest WFPS was 75% and the highest was 81%, observed in years 1988 and 1989.

4.4.3 Soil N₂O emissions

4.4.3.1 Drainage management effects and effects of the weather variability

Results of the analysis of variance showed that GM treatment significantly lowered the annual cumulative soil N₂O flux (Table 4.3). Soil WFPS was significantly ($P=0.001$) lower for GM

than C. Soil temperature (from 0 to 50 cm) was unaffected by GM. Soil nitrate, soil ammonium and leached nitrate (from 0 to 50 cm deep) were significantly greater for GM than for C.

Table 4.3. Drainage management effect on soil N₂O, CH₄ and CO₂ emissions, net soil GHG emissions, soil microclimate parameters and soil mineral nitrogen. Values are means across 27 climate scenarios (standard error of the mean).

Treatment	N ₂ O	CO ₂	CH ₄	Net GHG	WFPS	Soil temp.	Soil NO ₃ ⁻	Soil NH ₄ ⁺	Leached NO ₃ ⁻
Units	kg N ha ⁻¹ ya ⁻¹	kg C ha ⁻¹ ya ⁻¹	kg C ha ⁻¹ ya ⁻¹	kg CO ₂ -eq ha ⁻¹ ya ⁻¹	%	°C	kg N ha ⁻¹	kg N ha ⁻¹	kg N ha ⁻¹
C	1.7 (0.07) ^{**}	1521(2 0.9) ^{ns}	-0.4 (0.1) ^{ns}	2016 (85.9) ^{ns}	79 (0.2) ^{**}	10.3 (0.04) ^{ns}	25 (0.9) [*]	14 (0.2) [*]	13 (0.8) [*]
GM	1.4 (0.07) ^{**}	1588(2 0.9) ^{ns}	-0.6 (0.1) ^{ns}	1877 (85.9) ^{ns}	72 (0.2) ^{**}	10.2 (0.04) ^{ns}	37 (0.9) [*]	21 (0.2) [*]	21 (0.8) [*]

Level of significance ^{*}, ^{**}, ^{***} are p ≤ 0.05, p ≤ 0.01 and p ≤ 0.001. ^{ns} = not significant based on analysis of variance.

There was a significant effect of weather scenario on soil N₂O emissions and there was not a significant interaction between weather scenario and treatment. Post-hoc analysis (Tukey's test) identified that the iSC weather scenarios 1997, 2006, 2007 and 2008 significantly increased the soil N₂O emissions by (mean±standard deviation) 0.85±0.322 kg N ha⁻¹ year⁻¹ compared to BLY. Results show that the iSC are characterised by on average significantly higher air temperature and greater WFPS. Mean annual rainfall WFPS and air temperature from iSC were 3mm, 5.6% and 0.5°C greater than the BLY.

4.4.3.2 Key drivers and dominant process

Variables included in the regression analysis were: soil temperature, soil WFPS (from 0 to 50 cm depth), dissolved organic carbon (dSOC) (from 0 to 50 cm depth) and SOC (from 0 to 50 cm depth). Highly correlated variables such as mineral nitrogen were excluded from the analysis because calculation of N₂O flux was contained those parameters in the equations.

In the current and following linear regression models, the explanatory variables with high beta standardised regression coefficients and high partial correlation coefficients were considered as the most influential in the production of the particular GHG. Factors with negligible influence on the response variables were eliminated by stepwise regression model selection technique.

For C the best fitted linear regression model included dSOC, soil WFPS and SOC (from 0 to 50 cm depth). Model explained the 76% of the variability in annual N₂O emissions under C treatment. For GM the linear regression model included soil temperature, WFPS and SOC. Model R² = 0.533 (standard error = 0.3434). Standardised regression coefficient Beta and partial correlation coefficients of all explanatory variables in both models are presented in Table 4.4. The most influential variables were dSOC and SOC for C and GM respectively. The dSOC was highly and negatively correlated to N₂O emissions under C, while the SOC was highly and positively correlated to N₂O flux under GM.

Table 4.4. Key drivers and predominant processes of soil N₂O emissions as identified through multiple linear regression analysis for undrained control (C) and gravel mole drainage (GM) treatments independently. B as the intercept for the regression equation, the standard error associated with the coefficients (SE), the beta standardised regression coefficient (Beta), partial correlation coefficient (corr. partial) and model R². Number of observations (n) in each model n = 112 (28 weather scenarios x 4 plot replicates).

	C					GM				
	B	SE	Beta	Corr. (partial)	Model R ²	B	SE	Beta	Corr. (partial)	Model R ²
Intercept	-13.17 (p<0.001)					-8.55 (p<0.001)				
SOC		0.01	-	-0.45	0.761	0.00	0.00	0.765	0.571	0.533
			1.97							
WFPS		2.28	0.19	0.32		3.66	0.54	0.442	0.546	
dSOC		0.00	-	-0.59		-	-	-	-	
			1.80							
NH₄		0.13	1.07	0.46		-	-	-	-	
L. NO₃⁻		-	-	-		0.02	0.01	0.441	0.512	
Soil temp.		-	-	-		0.41	0.06	0.402	0.374	
Intercept	0.43 (p<0.001)					0.307 (p<0.001)				
Denitrification		0.320	1.30	0.865	0.947	3.58	0.24	0.816	0.816	0.663
Nitrification		0.011	-	-0.422		-	-	-	-	
			0.35							

SOC= soil organic carbon, dSOC= dissolved soil organic carbon, NH₄⁺ = ammonium, L. NO₃⁻ = leached nitrate and soil temp. = soil temperature.

4.4.4 Soil respiration

4.4.4.1 Drainage management effects and effects of weather variability

Soil respiration was not significantly affected by drainage management nor by weather scenarios. Mean CO₂ emissions were 15455±257 kg C ha⁻¹ year⁻¹ and 15641±279 kg C ha⁻¹ year⁻¹ for C and GM, respectively.

4.4.4.2 Key drivers

Variables included in the final regression analysis were: dissolved organic carbon (from 0 to 50 cm depth), aboveground plant biomass, belowground plant biomass, soil temperature (from 0 to 50 cm depth), soil water filled pore space (from 0 to 50 cm depth), root carbon litter input and soil organic carbon (from 0 to 50 cm depth). For C, the best-fitted linear regression model included dSOC, soil temperature, root carbon, root litter carbon, leached carbon, and aboveground plant biomass. Model R² = 0.994 (standard error = 20.7136). For GM the model included dSOC, soil temperature, root carbon, SOC and aboveground biomass. Model R² = 0.926 (standard error = 21.7446). Standardized Beta regression coefficient and partial correlation coefficients for all explanatory variables are presented in Table 4.6. The most influential variables were dSOC and SOC for C and GM respectively. The dSOC was highly and negatively correlated to CO₂ emissions under C, while the SOC was highly and positively correlated to CO₂ flux under GM.

Table 4.5. Key drivers of soil CO₂ emissions as identified through multiple linear regression analysis for gravel mole drainage (GM) and undrained control (C) treatments independently. Beta refers to beta standardised regression coefficient and corr. partial refers to partial correlation coefficient. Each model was developed based on 112 observations (28 weather scenarios x 4 replicates).

	C			GM		
	Beta	Corr. (partial)	Model R ²	Beta	Corr. (partial)	Model R ²
Intercept	1258.65 (p<0.001)			1398.12 (p<0.001)		
dSOC		-0.847	0.994		-0.956	0.926
SOC		0.489			0.703	
Soil temp.		0.497			0.685	
Root C		0.578			0.608	
Leached C		0.270			-	
Abo-gr. biom.		0.232			0.261	

SOC= soil organic carbon, dSOC= dissolved soil organic carbon, soil temp. = soil temperature, root C= root carbon content, leached C = leached carbon and abo-gr. biom = above ground biomass.

4.4.5 Soil CH₄ emissions

4.4.5.1 Treatment and weather variability effect

The CH₄ emissions were unaffected ($p>0.05$) by GM and weather scenarios. Annual mean (mean±standard deviation) emissions were -0.4 ± 0.09 kg C ha⁻¹ year⁻¹ and -0.5 ± 0.1 kg C ha⁻¹ year⁻¹ for C and GM, respectively.

4.4.5.2 Key drivers

Variables included in the linear regression analysis were: soil mineral N (from 0 to 50 cm depth), dSOC (from 0 to 50 cm depth), leached carbon, soil temperature (from 0 to 50 cm depth), aboveground and belowground plant biomass, WFPS (from 0 to 50 cm depth) and mean SOC (from 0 to 50 cm depth). For C, the best fitted linear regression model included dSOC and WFPS. Model $R^2 = 0.872$ (standard error = 0.0355). For GM the model included SOC, WFPS, soil temperature, leached C and dSOC. Model $R^2 = 0.962$ (standard error = 0.03102). Standardised regression coefficient Beta and partial correlation coefficients for all explanatory variables are presented in Table 4.7. The most influential variables were dSOC and WFPS for C and GM respectively. Both variables were highly and positively correlated to CH₄ emissions under C and GM.

Table 4.6. Key drivers of soil CH₄ identified through multiple linear regression analysis for undrained control (C) and gravel mole drainage (GM) treatments independently. B as the intercept for the regression equation, the standard error associated with the coefficients (SE), the beta standardised regression coefficient (Beta), partial correlation coefficient (corr. partial) and model R². Number of observations (n) in each model n = 112 (28 weather scenarios x 4 plot replicates).

	C					GM				
	B	SE	Beta	Corr. (partial)	R ²	B	SE	Beta	Corr. (partial)	R ²
Intercept	-1.05 (p<0.001)					-0.88 (p<0.001)				
dSOC	0.00	0.00	0.911	0.942	0.872	0.00	0.00	-	-0.420	0.962
WFPS	1.03	0.13	0.244	0.602		2.23	0.05	0.851	0.975	
SOC	-	-	-	-		0.0	0.0	-	-0.853	
Soil temp.	-	-	-	-		-0.06	0.01	-	-0.635	
Leached C	-	-	-	-		-0.02	0.00	-	-0.578	
NH ₄	-	-	-	-		-	-	-	-	
L. NO ₃ ⁻	-	-	-	-		-	-	-	-	
Abo-gr. biom.	-	-	-	-		-	-	-	-	
Root C	-	-	-	-		-	-	-	-	

SOC= soil organic carbon, WFPS= water filled pore space, dSOC= dissolved soil organic carbon, soil temp. = soil temperature and leached C = leached carbon, NH₄⁺ = ammonium, L. NO₃⁻ = leached nitrated, root C= root carbon content and abo-gr. biom = above ground biomass.

4.4.6 Net soil GHG emissions

Net soil GHG emissions were unaffected by weather scenarios and the drainage management treatments. Annual mean (mean±standard error) net soil GHG emissions were 2016±85.9 kg CO₂eq ha⁻¹ year⁻¹ and 1877±85.9 kg CO₂eq ha⁻¹ year⁻¹ for C and GM, respectively.

4.5 Discussion

In our study calibrating DNDC with measured soil variables at a plot level to obtain the best fit of modelled fluxes considerably improved the model performance (Table 4.1). Model indicators such as MAE, RMAE, RMSE and *Bias* indicated that model performed very well simulating the

WFPS for both treatments. This was very important in order to establish the differences between the soil WFPS of the two treatments. Furthermore the model simulated very well the soil above biomass, which is considered a key variable to influence the soil biochemical cycles and it is widely accepted as one of the main influence of process based models on N and C pools (Hashimoto et al., 2011; Smith et al., 2003). The model simulated the annual above biomass production with a very accurate precision of 236.84 and 192.29 kg C ha⁻¹ y⁻¹ for C and GM respectively.

Hence, the very good simulation of the soil WFPS and above biomass had a direct impact on the soil GHG fluxes simulation. Overall, the modelled fluxes, relative to the measured fluxes, were very well represented for both treatments. The RMAE, RMSE and *Bias* of soil GHG flux were comparable with other modelling studies (Abdalla et al., 2010, 2009; Iqbal et al., 2018). However, in the current study the CO₂ emissions were slightly poorly estimated (based on indicator *Bias*) compared to previous studies that had a relatively well estimation of the CO₂ emissions such as Hashimoto et al., (2011), difference among studies are likely to be explained by using different models that represent slightly differences the biochemical process. Overall, despite the slightly underestimation of the annual GHG emissions in the current study the model performed very well simulating the spatial variability of the soil GHG emissions.

4.5.1 Drainage management effects on soil GHG emissions

The GM treatment significantly lowered WFPS across all simulation scenarios (Table 4.3). Drainage treatment allowed the discharge of water from the soil profile, which is consistent with previous findings on the same site (Tuohy et al., 2015; Tuohy et al., 2016, Valbuena-Parralejo et al., 2018). Despite this there was no detectable difference in annual cumulative soil CO₂ and CH₄ emissions. Soil CO₂ flux is primarily influenced by soil temperature (Schlentner and Cleve, 1985; Angers et al., 2010; Koncz et al., 2015) which was unchanged by the drainage treatments. Other studies found that by lowering soil water content with artificial drainage the soil CH₄ emissions decreased (Rizzo et al., 2015). Present results indicate that lowering the WFPS was not sufficient to influence the CH₄ oxidation rate.

However, simulated data indicated that GM treatment had significantly lower annual cumulative soil N₂O emissions than C. This agrees with the findings of Dobbie and Smith (2006) who reported that by lowering the WTD to 45 cm would reduce soil N₂O emissions by over 80%. Other studies also reported lower emissions from drained plots (Tesfai et al., 2015; Grossel et al., 2016). On the contrary, Goldberg et al. (2010) studied the impact of altering the WTD and observed peaks in N₂O emissions from drained fields, however authors concluded that artificially drained plots did not have any impact on N₂O flux. Disagreement between

studies reflects the complexity of the process generating soil N₂O emissions. Saggar et al., (2013) acknowledged the complex interactions between soil properties, soil micro-organisms, climatic factors and management practices, hence all parameters have to be taken into account when comparing results between studies.

In the present study, results indicate that higher N₂O emissions from C treatment were driven primarily by greater WFPS and this enhanced the denitrification process in the soil profile (Table 4.5).

4.5.2 Effects of inter-annual weather variability on soil GHG emissions

The above results are consistent with the impact of inter-annual weather variability on soil GHG emissions. Inter-annual weather variability did not have a detectable impact on soil CO₂ and CH₄ emissions. However the iSC had significantly higher N₂O emissions by 0.85 ± 0.322 kg N ha⁻¹ year⁻¹ than the BLY. The iSC were characterised by a significantly greater WFPS than the BLY. Previous modelling studies that looked at three soil GHG have also identified higher WFPS to enhance N₂O emissions, whereas CH₄ flux decreased and CO₂ emission were mainly influenced by soil temperature (Hashimoto et al., 2011). Decock and Six, (2013) identified higher rainfall to promote greater N₂O emissions, increased soil moisture stimulated carbon and nitrogen turnover (Davidson, 1992). Authors attributed that to the dominance processes generating N₂O emissions as well as the effect of wet and dry cycles. In the present study precipitation from iSC was not significantly different than the rainfall from the BLY. However, air temperature was identified as significantly greater for iSC than BLY. Previous research have established a positive and strong correlations between air temperature and soil N₂O emissions, concluding that an increase by 1.5 °C in the daily average air temperature resulted in a 62% increase in N₂O flux (Abdalla et al., 2009). Authors attributed the increase to an increase in soil denitrification.

Overall, results of the current study show that soil GHG emissions can be influenced by ‘environmental’ or by ‘input’ factors. The environmental factors are those that have an impact on regulation of the biochemical processes, for example the soil temperature and soil moisture (Decock and Six, 2013). While the ‘input’ factors are soil nutrient concentrations that have an impact on these processes, for example soil ammonium or SOC content. Effects of environmental and input factors are often occurring simultaneously and difficult to underpin due to its complex relationship.

4.5.3 Key drivers of GHG drainage treatments

For better understanding of the combination effects of environmental and input factors on individual soil GHG emissions we performed several multiple linear regression analyses.

Soil N₂O emissions

Across the weather scenarios, the main drivers of soil N₂O emissions differed depending upon the drainage conditions. Under C management treatment, results indicate that dSOC was the main driver of N₂O emissions, followed by soil ammonium content and SOC. This was also found on previous grassland studies (Krauss et al., 2017). However, while the latter authors reported a positive correlation between dSOC and SOC and negative correlation with ammonium content, in the current study it was the other way around. Difference between studies might be due to different management, Krauss et al., (2017) applied organic fertilizer and ploughed the soil. Under unfertilized conditions, like in the current study, the N₂O emissions are highly dependent on SOC content, hence when SOC is depleted the N₂O emissions can also decrease (Oenema et al., 2001). In the current study, soil ammonium had a positive linear association with N₂O emissions. This can be explained because after the soil organic matter is mineralised, soil ammonium is free in the soil matrix. Soil ammonium is one of the primaries precursors of soil nitrification and denitrification (Saggar et al., 2013). Hence the concentration of soil ammonium will have a proportionally positive impact on soil N₂O emissions.

However, by applying GM results indicated that SOC had the strongest linear association with N₂O emissions, followed by WFPS and leached nitrate (50 cm depth), all positively correlated with soil N₂O emissions. Results of the present study suggest that across the weather scenarios GM changed the conditions for soil N₂O emissions, environmental parameters such as WFPS and soil temperature happened to have a greater influence on emissions than under C. Moreover soil nitrate leached was positively correlated to soil N₂O emissions, suggesting that nitrogen losses due to greater soil organic matter mineralisation were highly related to soil N₂O emissions. Results of the GM treatment agree with the positive linear correlations established by Krauss et al., (2017). On the contrary Horrocks et al., (2015) conducted a study on a multiple grassland sites study between South East Scotland and South West England and reported inconsistent relationships of soil properties, such as WFPS and soil temperature, with soil N₂O emissions. Authors attributed this to unusual weather patterns throughout their sampling period (i.e. the cool temperatures, particularly in 2010, and the little rainfall, except for two unusually wet months in April and June 2012) likely to have affected key soil environmental factors and plant and microbial communities.

Soil environmental factors are closely related to soil N₂O generation process. In the present study we also observed a drainage management effect on the identified dominant process, denitrification and nitrification. The analysis indicated that denitrification was the dominant process under both GM and C management treatments. The N₂O emissions were positively correlated with the denitrification rates, which showed a very strong positive association with soil N₂O emissions. However, under C management treatment nitrification process was negatively correlated to soil N₂O emissions and despite being significant, had a much weaker linear relationship with soil N₂O emissions than denitrification. Overall, results indicated that on our site for both management treatments denitrification is the dominant process that generated N₂O emissions. Results of the present study agree with Bateman and Baggs, (2005) that conducted a study to identify the contributions of nitrification and denitrification to N₂O emissions from soils at different WFPS. Authors identified denitrification to be predominant at the same ranges of WFPS as the ones reported on our study (i.e. > 70%).

Soil CO₂ emissions

Unlike the N₂O emissions, weather inter-annual variability and drainage management did not have an impact on soil respiration. However, regression analysis indicated that variables influencing soil CO₂ emissions from C and GM management treatments were different.

Under C treatment, soil respiration was primarily influenced by dSOC and negatively correlated. Under C soil CO₂ emissions were positively correlated to SOC, soil temperature, root carbon, leached carbon and aboveground plant biomass. The dSOC as the labile fraction of SOC is able to facilitate biogeochemical redox reactions affecting soil respiration. The strong negative correlation reported in the current study between dSOC and soil CO₂ emissions suggested that the dSOC content inhibits the soil respiration. This was also reported in a previous study by Bi et al., (2013). Authors concluded that the electron transfer capacity of dSOC can greatly improve SOC stability, which was of great significance to soil carbon sequestration and GHG mitigation.

Similarly, under GM treatment dSOC was strongly and negatively correlated to soil CO₂ emissions. However, under this treatment the leached carbon was not significantly correlated to soil CO₂ emissions. Other factors such as soil temperature, SOC and root C had a stronger linear correlation for GM than under C treatment. The differences could be attributed to the different soil characteristics between C and GM treatment, for example soil hydraulic conductivity (Tuohy et al., 2016). Results of the current study show how WFPS was

significantly lower under GM, hence allowing a greater discharge of water from the soil profile that can be attributed to an improved hydraulic conductivity.

Soil CH₄ emissions

In agreement with soil respiration, methane emissions were unaffected by inter-annual weather variability and drainage management. However results of the multiple linear regression analysis indicated that key drivers were different between C and GM. Despite the different drivers, emissions were negative for both treatments, indicating that there was methane uptake, not production. The latter is in agreement with previous European studies under temperate grassland (Merino et al., 2004; Schaufler et al., 2010)

The analysis showed that under C there was a positive and strong linear relationship between soil CH₄ and dSOC content, followed by the soil WFPS. While under GM treatment the WFPS was identified as the main parameter influencing CH₄ emissions, followed by SOC, soil temperature, leached carbon and dSOC. Under GM the dSOC was also identify as a significant driver of CH₄ emission, however the linear relationship was negatively correlated and less strong than under C treatment. Hence results suggest that GM treatment changed the relation between methane uptake and parameters regulating the process.

4.6 Conclusion

Gravel mole drainage treatment decreased the WFPS and had no influence on soil CO₂ and CH₄ emissions, but decreased annual soil cumulative N₂O emissions. The impact of artificial drainage on WFPS was not enough to cause a detectable difference in cumulative CO₂ and CH₄ flux. However, decreased WFPS lowered the cumulative N₂O by 0.3 kg N ha⁻¹ year⁻¹ under the GM treatment. Despite this, denitrification was identified as the main process originating N₂O emissions under both management treatments. Furthermore, results suggest that soil N₂O flux from wet grassland systems are sensitive to weather characteristics. Years with significantly higher precipitation and thus greater WFPS and also air temperature may significantly increase soil N₂O emissions up to 0.85±0.322 kg N ha⁻¹ year⁻¹.

Despite the lack of impact of drainage management on soil cumulative CO₂ and CH₄ fluxes, results indicated that the GM treatment had a significant impact on drivers affecting the soil GHG emissions such as nitrification and denitrification. Hence results provided some insights into soil GHG controls under undrained and drained conditions which can guide the design of future laboratory and field experiments concerning soil GHG mitigation.

Overall, gravel mole drainage management for improving water infiltration through the soil profile did not increase the extent of soil net GHG emissions under site-specific conditions described in the present study.

CHAPTER 5. GENERAL DISCUSSION AND CONCLUSIONS

5.1. Findings of the Study

5.1.1 Impact of subsurface artificial soil drainage on water-filled pore space, soil water table and biomass production

Overall this thesis has concluded that, as expected, artificial soil drainage significantly lowered soil WPFS and the soil WTD. Furthermore, in Chapter 2 it has been shown that by implementing mole and gravel mole drains, the overland flow was significantly reduced and there was a greater proportion of water discharged through the drain flows, hence lowering the soil water content across the soil profile. Previous studies reported a reduction in crop production by up to 30% due to shallow WTD (Brereton and Hope-Cawdery, 1988; Burke et al., 2004; Laidlaw, 2009). However, in the current study the impact of the lower WPFS and WTD in crop yield was not consistent throughout experiments, where only in Expt B was the drained treatment yield on average $1.3 \pm 0.65 \text{ t DM ha}^{-1} \text{ year}^{-1}$ more than that of the undrained treatment. This highlights the fact that intensively drained soil systems have a greater capacity to discharge water from the soil, due to the implemented infrastructure on site, and that this has a direct impact upon crop yield. This is in agreement with previous studies that have extensively studied the impact of mole and gravel mole drainage systems on the subsurface hydrology of soils (Tuohy et al., 2015; Tuohy et al., 2016).

However, despite the impact on biomass yield being inconsistent, the fact that WPFS and WTD were lowered throughout all experiments, contributes to and reinforces the current literature in pointing out that drainage implementation is far more complex than the widely used rule of ‘one design fits all’. For example, Tuohy et al., (2018) examined a wide variety of subsurface drainage installations and concluded that in order to successfully achieve a shallow WTD while minimising the economic cost, an individual assessment must be carried out and a mixed combination of drainage systems should be applied for every site.

5.1.2. Impact of subsurface artificial drainage on water infiltration and N and P losses to water

Most literature agrees that by enhancing water discharge from agricultural soils, the risk of nutrient loss to water is increased (Haygarth et al., 1998; Peukert et al., 2014; White et al., 2009). However, results of the current study elucidate that when water flow is shifted (from runoff to subsurface flow) by subsurface artificial drainage soil N and P do not necessarily increase the concentration loads in drainage water.

As mentioned in Chapter 1, soil N and P particles behave differently due to their properties, solubility being one of the most important. Soil nitrate-N and ammonium-N are highly soluble in water, hence the results of the current study which showed an increase of those losses in agreement with what has been previously reported in the literature (Gold et al., 2001; McGechan; 2002). However, the current study points out that the implications of these losses will vary depending on the soil type and field management techniques. On the one hand, under grassland in loamy clay soils (the soils most likely to have mole and gravel mole drainage applied) and with optimum levels of soil N content that do not require fertilizer applications, both soil nitrate-N and ammonium-N will be well above the 11.3 mg L⁻¹ concentration threshold for drinking water. However, care should be taken when fertilizer applications are required to sustain crop growth. The current study concludes that by applying agricultural subsurface drainage, the water flow is enhanced, and the N dissolved in the drainage water is therefore more likely to be discharged from the soil and reach water catchments.

On the other hand, P is less soluble and tends to be absorbed by small particles of the soil (such as clay). Many studies have reported a high P sorption capacity in clay soils (Chardon et al., 1997; McDowell et al., 2001). Although not measuring soil sorption capacity, results of the current study showed an overall reduction of P losses to water, driven by reduced DRP losses in drainage flow. On the contrary, more recent studies state that is becoming clearer that sorption into mobile suspended or colloidal soil particles is an important mechanism leading to the leaching of P to drains. This leaching is driven by the soil macropores that impose very little restriction on movements of particles and also allow a faster flow of water (McGechan; 2002). In this study, there was no evidence of an increase overall soil P losses to either drainage water or to ground water. However, it was observed that DRP losses from gravel mole were greater than that of mole drainage. This is in agreement with McGechan; (2002), mole drainage has greater ‘macropores’ allowing water with the dissolved particles to move freely and faster. Hence, by applying agricultural drainage, the runoff flow is reduced and hence the overall losses of P are also reduced. However, as highlighted previously regarding N, care should be taken when applying fertiliser because the artificially created macrospores will allow the water to percolate and discharge freely and quickly through the drain.

Finally, the current study established that soil nitrate-N and ammonium-N losses to water increased, leading to the question of how this will impact the N soil cycle and potentially impact the N₂O flux from the soil. Furthermore, changes in physical and chemical soil conditions induced by mole and gravel mole drainage can also further influence the overall soil N and C cycles and the CO₂ and CH₄ related emissions. This established the rationale behind Chapter 3.

5.1.3. Nitrogen loss as N₂O

Despite greater soil nitrate-N and ammonium-N losses to water being observed, there was not an impact on the annual cumulative emission of N₂O from mole and gravel mole drained grassland. These results were not significantly different to undrained grassland averaging 1.6 ± 0.32 and 1.5 ± 0.12 kg N₂O-N ha⁻¹ for drained and undrained, respectively. In this study it was explained as being due to the lack of impact of drainage treatment on soil TN, soil net N mineralisation and nitrification, which, for the most part, is consistent with herbage production and N uptake in herbage.

Although there were no significant differences in cumulative N₂O fluxes between treatments, there was some evidence that mean daily N₂O emissions were episodic and peaked when WTD was at its deepest (summer). This indicated relationships between the emissions and fluctuation in soil water content. This agrees with most of the factors affecting N₂O emission (Chapter 1) and motivated the need to further study the interactions underlying the process involving the soil N₂O emissions and establishing the rationale behind Chapter 4.

Chapter 4 of this thesis allowed us not only to underpin the key drivers that affected the overall GHG emissions coming from the soil, but it also presented the opportunity to investigate the impact of weather variability on soil GHG fluxes. The results of this study showed that in contrast with Chapter 3, drainage treatment significantly lowered the annual cumulative soil N₂O flux. This agrees with the findings of Dobbie and Smith (2006) who reported that by lowering the WTD to 45cm, soil N₂O emissions would be reduced by over 80%. Other studies also reported lower emissions from drained plots (Tesfai et al., 2015; Grossel et al., 2016). Furthermore, the key drivers differed depending on if the soil was drained or undrained. The latter proved that the drainage treatment had a greater influence on change in environmental parameters such as WFPS and soil temperature than under undrained conditions. Results and established linear relationships were expected and in agreement with the literature (Chapter 1; Krauss et al., 2017).

The relevance of these results are of importance as most of the studies published to date provide relationships between key drivers and N₂O emissions (Oenema et al., 2001; Horrocks et al., 2015; Krauss et al., 2017). However, current results show that after applying agricultural drainage, the key drivers and processes for N₂O emissions vary and that drained sites will be more sensitive to changing weather conditions than non-drained sites.

5.1.4. Soil C losses as CO₂ and CH₄

Results of this study demonstrate that agriculture subsurface drainage did not impact either on the overall soil C or on soil CO₂ and CH₄ emissions. The results of this thesis provide valuable data as annual estimates of GHG emissions from drainage treatment are scarce in literature.

There is evidence in literature that physical soil disturbance such as ploughing can significantly enhance the soil CO₂ flux by oxidation of the SOM (Davidson et al., 2000; Lal, 2004; Angers et al., 2010). Moreover, the lower WFPS from the drainage treatment can significantly lower the soil CH₄ flux (Rizzo et al., 2015). However, in the current study, results showed that soil CO₂ and CH₄ emissions were not affected by drainage treatment (Chapter 3 and 4). Furthermore, in Chapter 4 the modelling results showed that weather variability did not have a significant impact CO₂ and CH₄ flux. Results of the current study can be explained by the lack of effect of weather variability and drainage treatment on SOM content.

However, the modelling work pointed out that that the variables influencing soil CO₂ emissions from undrained and drained management treatments were different. For example, factors such as soil temperature, SOC and root C had a stronger linear correlation with CO₂ emissions under artificial drainage than under undrained conditions. Indicating, likewise for N₂O, that agricultural drainage treatment can be more sensitive to changing weather conditions.

5.1.5. Environmental Impact of subsurface artificial drainage

The environmental impact of subsurface artificial drainage was assessed in the SOGGYLAND Project by comparing (i) nitrogen and phosphorous losses to water, (ii) total soil nitrogen and total carbon (iii) soil GHG emissions and (iv) weather variability between undrained and mole and gravel mole drained loamy clay soils in Ireland. The overall nitrate-N concentrations were well below the threshold for drinking water, however ammonium-N concentrations exceeded drinking water quality standards. On the contrary, drainage treatment lowered total P (reduced by on average 114 g ha⁻¹ year⁻¹) losses by enhancing soil P sorption. Furthermore, no detectable effect of artificial drainage was observed in soil TN, TC, SOM, soil net N mineralisation and nitrification, which, for the most part, is consistent with the lack of impact on annual cumulative GHG emissions. Despite differences between treatments on the cumulative soil, N₂O emissions were undetected, the soil daily N₂O emission were episodic and peaked over the summer with the deepest WTD (Chapter 2). The modelling work analysed in further detail the key drivers that impact the soil GHG flux. Results showed clearly that drainage treatment significantly lowered the annual cumulative N₂O emissions by 0.16 kg N₂O-N ha⁻¹ year⁻¹ and did not impact the soil CO₂ and CH₄ flux.

Hence mole and gravel mole drainage can be implemented on farms under similar management to that described in the present study with no impact on soil TN, TC, SOM, soil net N mineralisation and nitrification and annual cumulative soil CO₂ and CH₄ flux. And with a minor impact N and reduced P losses to water. Finally, annual cumulative soil N₂O flux can be lowered by applying mole and gravel mole drainage, however these would be highly dependent upon weather variability. Results showed that higher rainfall and increased air temperature will increase the overall soil N₂O flux from drained and undrained soils.

5.2. Implications

The N and P losses to water has major implications on the eutrophication of water bodies. To ensure that the ecosystems of waterbodies are preserved, the N and P concentrations are monitored under the Water Framework Directive (European Council, 2003). At the same time, milk quota abolition has led to an expansion of the Irish national dairy herd and more previously wet or unproductive land is being brought into intensive production. Hence, land management techniques such as artificial land drainage techniques are being implemented more often. Results reported in the current study show that mole and gravel mole drainage systems successfully lowered the soil water content, lowered the TP losses to water and had only a minor impact on the N losses to water. The substantial datasets (i.e. 24 months N₂O, CO₂ and CH₄ flux) of GHG fluxes gathered during this study are quite rare. These detailed datasets have been very valuable for the modelling validation purposes in this thesis. The results obtained with the validated model can be used to upscale emission estimates from the field and system scale to the catchment/regional or even national scale.

In addition, the results of this study highlight the dominance of weather-driven variables in regulating soil GHG emissions. The results also have major implications in the context of future projected climate change where increased weather volatility, including temperature extremes and patterns of rainfall are predicted (Solomon et al., 2007; Collins et al., 2013). These changes in weather conditions had a direct impact on soil moisture conditions. The N₂O emissions were highest under anoxic soil conditions, under undrained treatment, in this study. While the influence of weather variability on soil moisture conditions cannot be controlled, implementation of management decisions and support tools should be used in order to avoid, for example, cattle grazing under wet soil conditions that will lead to soil compaction and anoxic soil conditions.

Moreover, the results of these studies point out that by a change of land use, those factors influencing the process across the soil N and C cycle change. These findings are very valuable, Regulation (EU) No 525/2013 and Decision No 529/2013/EU, both of which drive policies of land use change in Ireland and in Europe. Recently, 30 May 2018, regulations on the latter included an update on bioenergy plans, climate change adaptation and mitigation strategies. Furthermore, in Europe policymakers and the research community are demanding the need for more accurate information on land use changes. For example, in Ireland, land use maps are produced every six years with the latest published in 2018. Information included in the map is too basic i.e. agricultural area, wetland, forest. However, the more site-specific data that is produced and simulations such as the one performed in the current thesis will provide more accurate data that will be of enormous help to decision makers. For example, to identify which parts are best suited for grazing or forestry, soils that act as carbon sinks and wetlands and areas where habitats are at risk. Furthermore, under a Climate Change Scenario, predictions such as those performed in this study will be very valuable for the national GHG inventory. Irelands' agricultural GHG emissions are currently estimated using the IPCC Tier 1 approach for national GHG inventory reporting required under the EC Climate and Energy Package (See section 1.6.2). Ireland is currently in the process of generating country specific agricultural GHG emissions for the Tier 2 approach. Results of the soil GHG emissions under agriculture drainage from current study will provide valuable data for a more refined estimation of the national GHG inventory. Overall, results will not only improve the estimation of GHG but also by identifying the factors driving the GHG, the research community will be able to further investigate mitigation strategies and ultimately, fulfil the Paris Agreement whereby emissions should be cut by at least 40% in comparison to those in 1990 (Falkner, 2016).

Finally, and no less important, this work will establish the basis to transpose the environmental impact assessment (Directive 2011/92/EU, S.I. No. 456 of 2011) in which all agriculture drainage works above 15 hectares will be required to fulfil before installing the system.

1. 5.3. Conclusions

Mole and gravel mole drainage are unlikely to increase phosphorus losses to water, in particular dissolved reactive phosphorus, on farms under similar management to that described in the present study. However, nitrate-N and ammonium-N losses to water are likely to be higher, albeit that in this study, overall annual losses were seen to be very low (Chapter 2).

The extent of soil GHG emissions were not increased. However, further studies might be required in order to understand the interactions of the underlying process involving the soil N₂O emissions (Chapter 3).

The modelling results showed that agricultural drainage did not affect the annual cumulative soil CO₂ and CH₄ flux and significantly lowered the N₂O flux. Moreover, drainage treatment had a significant impact on drivers affecting the soil GHG emissions. Hence, results provided some insights into soil GHG controls under both drained and undrained conditions, which could be a guide in designing future laboratory and field experiments concerning soil GHG mitigation. Finally, results also showed that weather variability such as increased mean air temperature and rainfall can significantly increase soil N₂O emissions in drained and undrained treatments (Chapter 4).

5.4. Recommendations for future work

- The losses of nitrate-N and ammonium-N were significantly greater for the drainage treatment. It would be interesting to investigate the impact of different N fertilizer inputs on this relationship.
- In order to remediate nitrate-N and ammonium-N losses of the drainage system upstream it would be interesting to further investigate on bioreactor structure installed before the drainage outlet.
- It would be useful to have an estimate of the N₂O emission factor of drainage treatment. It would also be interesting to assess the effects of different types N fertiliser inputs.
- Soil organic N mineralisation/immobilisation were measured in this study on a seasonal basis. It would be interesting to have measured it more often and would have helped to explain the episodic N₂O fluxes.
- This study did not measure the ammonia emissions and the dinitrogen emissions. It would be interesting to gather this data for further N balance studies under agricultural drainage systems.
- The results of this study highlights the complexity of the parameters affecting the soil GHG emissions and particularly the soil N₂O flux. It would be interesting to do a sensitivity analysis of the identified key parameters in order to establish range of effects and interactions.

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APPENDIX

Scientific publications and disseminations

Valbuena-Parralejo, N., Tuohy, P., Fenton, O., Burchill, W., Williams, M., Lanigan, G. J., & Humphreys, J. (2019). Greenhouse gas emissions from temperate permanent grassland on clay-loam soil following the installation of artificial drainage. *Agriculture, ecosystems & environment*, 269, 39-50.

Valbuena-Parralejo, N., Fenton, O., Tuohy, P., Williams, M., Lanigan, G. J., & Humphreys, J. (2019). Phosphorus and nitrogen losses from temperate permanent grassland on clay-loam soil after the installation of artificial mole and gravel mole drainage. *Science of The Total Environment*, 659, 1428-1436.

Scientific papers (published and submitted for publication)

N. Valbuena-Parralejo, M. Necpalova and J. Humphreys. Simulating soil greenhouse gases from Irish grassland under drainage management. Submitted to *Agriculture Ecosystems and Environment* (submitted September 2019).

Conference presentations

N. Valbuena-Parralejo, O. Fenton, P. Tuohy, M. Williams, G.J. Lanigan, W. Burchill and J. Humphreys. 'The effect of mole drainage on N₂O emissions from a clay-loam soil under grassland'. Wageningen International Soils Conference 2015, in Wageningen, The Nederland on the 27th August 2015.

N. Valbuena-Parralejo, P. Tuohy, O. Fenton, M. Williams, G.J. Lanigan, W. Burchill and J. Humphreys. 'The effect of mole drainage on background N₂O emissions from an unfertilised clay-loam soil'. The Agriculture Research Forum 2015, in Tullamore, Co. Offaly, Ireland. The 9th March 2015.

N. Valbuena-Parralejo, P. Tuohy, O. Fenton, M. Williams, G.J. Lanigan, W. Burchill and J. Humphreys. 'Environmental impact of agriculture subsurface artificial drainage – materials and methods'. Postgraduate symposium of the Botany Department at Trinity Collage Dublin, on the 20th February 2014.

Project meetings and seminars

N. Valbuena-Parralejo, P. Tuohy, O. Fenton, M. Williams, G.J. Lanigan, W. Burchill and J. Humphreys. 'Environmental impact of land drainage'. Seminar was given to the Dairy Advisors for the Sustainable Dairy Course. Teagasc, AGRIC, Moorepark, Fermoy, Co. Cork on the 14th June 2016.

N. Valbuena-Parralejo, P. Tuohy, O. Fenton, M. Williams, G.J. Lanigan, W. Burchill and J. Humphreys. 'Environmental impact of agriculture subsurface artificial drainage' presentation was given at the Moorepark Walsh Fellow student's seminar. With and outline of the PhD thesis. Teagasc, AGRIC, Moorepark, Fermoy, Co. Cork on the 20th February 2016.

N. Valbuena-Parralejo, P. Tuohy, O. Fenton, M. Williams, G.J. Lanigan, W. Burchill and J. Humphreys. 'Environmental impact of agriculture subsurface artificial drainage'. annual project meeting of 'SOGGYLAND' at Teagasc, AGRIC, Moorepark, Fermoy, Co. Cork on the 30th April 2014.

Awards and funding

February 2017: Selected for hosting a placement student during 7 months within the FARO program. Consisting in a grant that will provide the student subsistence expenses and travel expenses. Funded by the *Ministerio de Cultura y Deporte*, Spain. (5,000€)

January 2016: Selected for hosting a placement student during 6 months within the ARGO program. Consisting in a grant that will provide the student subsistence expenses and travel expenses. Funded by the *Ministerio de Cultura y Deporte*, Spain. (4,500€).

May 2015: Teagasc Overseas Training Award. Consisting in one month training on DNDC and Century biochemical models. Working in ETH Zurich, Department of Sustainable Agroecosystems. Hosted by Prof. Dr. Johan Six and Dr. Magdalena Necpalova. (5,000€)

September 2013: Teagasc Walsh Fellowship. Consisting in 3 years funded PhD within the Project financed by the Department of Agriculture, Food and the Marine (RSF 6441), Ireland. (70,000€)