Effect of using a catch crop to reduce nitrate leaching losses from winter grazed fodder beet soil

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Abstract of a Dissertation submitted in partial fulfilment of the requirements for the Degree of Bachelor of Agricultural Science with Honours.

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Grazing of winter forage crops such as fodder beet (*Beta vulgaris* L.) and kale (*Brassica oleracea* L.) is a common management option for increasing winter feed supply on New Zealand farms. This practice is most often implemented on South Island farms for feeding nonlactating dairy cows prior to calving. These crops produce high yields of 10-16 t DM/ha by the time of winter grazing. However, there is growing concern that the high stock density during grazing leads to large amounts of urinary nitrogen (N) deposited onto bare soil. The soil often remains fallow until early spring and thus could led to large nitrate (NO_3^-) leaching losses. Catch crops have been proposed as a mitigation strategy for reducing the N leaching losses from these winter forage grazing systems. The objective of this trial was to determine the effectiveness of the use of a catch crop of oats (*Avena sativa* L. cv. Milton) to reduce N leaching following winter grazing of fodder beet (*Beta vulgaris* L. cv. Rivage). This trial used the recently developed suction cup and lysimeter array (SCALAR) system to collect field scale measurements NO_3^- and ammonium (NH_4^+) leaching. The results show a catch crop of oats had no significant effect (P>0.05) on N leaching from winter grazed fodder beet. This result is thought to be due to the rapid leaching of N from the root zone prior to the establishment of the oats. This occurred because of the above average rainfall which occurred over the trial period. Total N leaching losses from winter grazed fodder beet soil were nevertheless low at 19 and 23 kg N/ha for the fallow and catch crop treatments respectively. This was attributed to the potentially large quantities of gaseous N loss resulting from denitrification due to the wet anaerobic conditions and soil compaction due to trampling damage.

Keywords: Nitrate, NO_3^-, ammonium, NH_4^+, nitrogen, N, Suction Cup and Lysimeter Array (SCALAR), winter forage, fodder beet, catch crop, oat, mitigation, leaching.

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Chapter 1

Introduction

In New Zealand's dairy industry, pregnant, nonlactating dairy cows are often grazed on winter forage crops such as fodder beet (*Beta vulgaris* L.) in order to meet the body condition score targets which are required for calving in early spring (Judson *et al*., 2010). These winter forage crops are usually sown in the previous spring and generally reach yields of between 10-16 t DM/ha by the time of winter grazing (Brown *et al*., 2007). The areas where these crops are grown often remains fallow until the following crop is grown in spring approximately 2-3 months later. These winter forage crops provide a large feed supply at a time when pasture production is low thus filling a vital feed gap that exists in the majority of New Zealand's traditional dairy farm systems. Unfortunately grazing of these winter forage crops has the potential to cause large amounts of N leaching to occur as a result of the high stocking rates used during grazing, winter drainage rates and lack of plant uptake of urine-N from the grazing cattle. The loss of this N represents both economic loss from farming systems as well as an adverse environmental issue. N leaching causes a number of environmental issues including eutrophication of water ways and algae growth which can lead to animal, as well as human health issues such as methemoglobinemia (blue baby syndrome). The NO_3^- can also be denitrified to form N gases such as nitrous oxide (N_2O) which can be lost to the atmosphere. These gases can then cause a variety of issues for example, N_2O is a strong Green House Gas (GHG) with 300 times the warming potential of carbon dioxide (CO_2) (Keller and Hartley, 2003). N_2O also depletes the ozone layer increasing the risk of health problems caused by UV rays such as skin cancer (Scottish Environmental Protection Agency, n.d.). Gaseous N losses of ammonia (NH_3) can also led to acidification of many ecosystems therefore negatively affecting a variety of species. Because of the issues caused by N leaching, methods to minimise it are extremely valuable to society. One mitigation which has shown potential to substantially reduce N leaching from winter forage crops is the use of fast growing winter active catch crops (McLenaghen *et al*., 1996; Francis *et al*., 1998; Shepherd, 1999; Di & Cameron, 2002). However, there is a lack of information regarding the effect of catch crops following fodder beet and little field scale information regarding N leaching from these winter forage systems. Due to this, the objective of this trial was to determine the effectiveness of the use of a catch crop of oats (*Avena sativa* L. cv. Milton) to reduce N leaching following winter grazing of fodder beet. This trial will use the recently developed suction cup and lysimeter array (SCALAR) system to collect field scale measurements NO_3^- and NH_4^+ leaching.

Chapter 2

Literature Review

2.1. Introduction

Nitrogen (N) is a plant nutrient which is needed in large quantities due to it being a vital macronutrient (Cameron, 1992). However, N can be leached from the soil in the form of nitrate (NO_3^-). The loss of this NO_3^- represents an economic loss for farming systems as increased N fertilizer application is required to replace the lost N and plant yields are reduced. As well as causing economic and production losses NO_3^- leaching causes adverse environmental issues. These issues include eutrophication of water ways, algae growth which can lead to animal and human health issues such as methemoglobinemia (blue baby syndrome). The NO_3^- can also be denitrified to form N gases such as nitrous oxide (N_2O) which can be lost to the atmosphere. N_2O is a strong Green House Gas (GHG) with 310 times the warming potential of carbon dioxide (CO_2) (Keller and Hartley, 2003). N_2O also weakens the ozone layer increasing the risk of health problems caused by UV rays (Scottish Environmental Protection Agency, n.d.). Gaseous N losses of ammonia (NH_3) can also lead to acidification of many ecosystems therefore negatively affecting many species. Because of the issues caused by NO_3^-, knowledge of NO_3^- leaching and methods to minimise it are extremely valuable to society. The objective of this literature review is to summarise the current state of knowledge relating to N leaching from winter forage crops and the potential to use a catch crop to reduce those losses

2.2. The Nitrogen Cycle

Replace the content of this page with your own content. The N cycle in agriculture is shown in Figure 2.1. The key soil processes that affect nitrate leaching from annual urine deposits include nitrification, denitrification, immobilisation and solute transport.

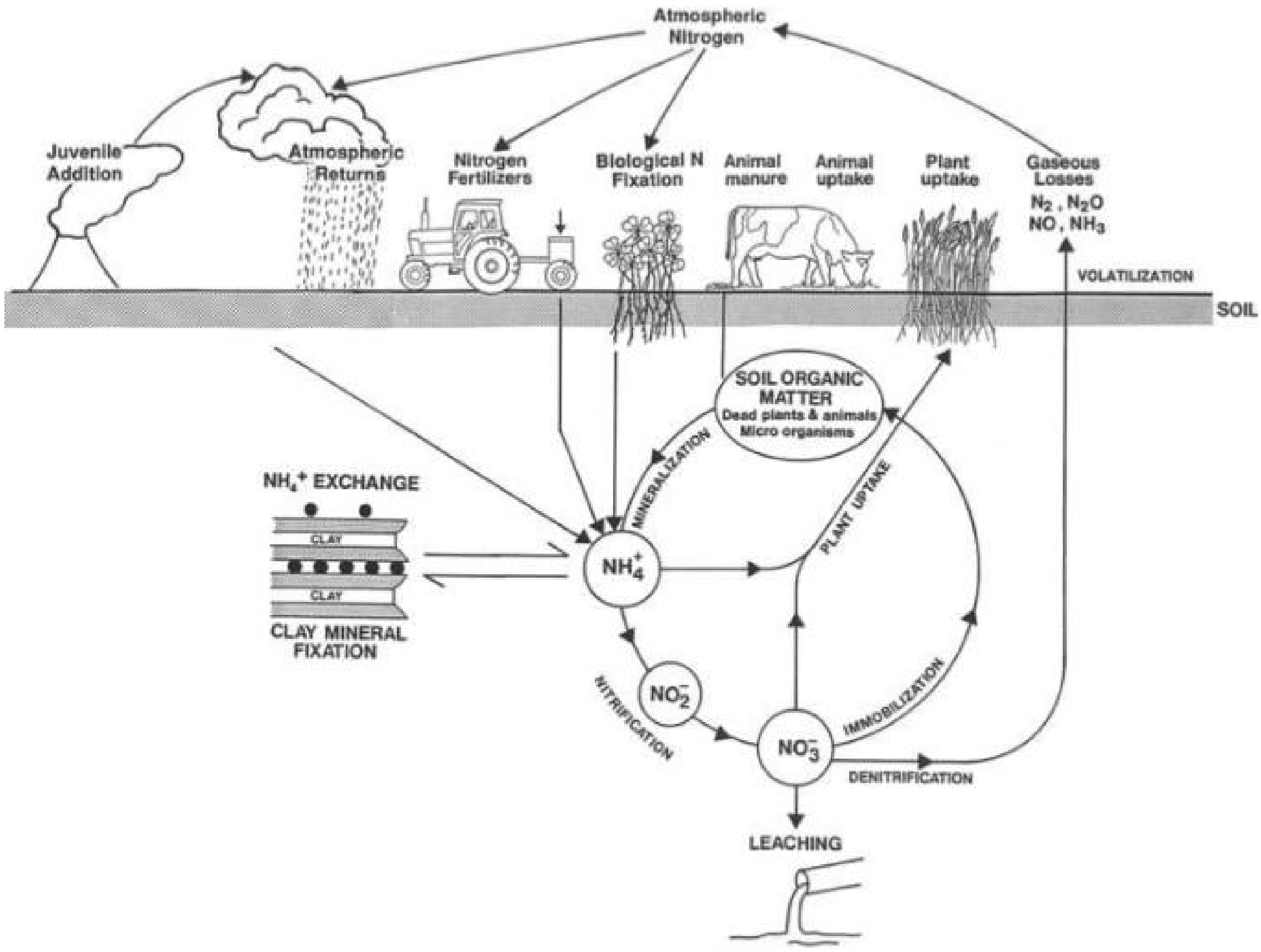

Figure 2.1: Nitrogen cycle (Adapted from Cameron *et al.*, 2013).

2.2.1. Nitrification

Nitrification is the process of oxidation of ammonium (NH_4^+) or NH_3 to nitrite (NO_2^-) and then to NO_3^- (Di *et al.*, 2010). This is an aerobic process which was previously believed to be carried out by ammonia oxidising bacteria (AOB) but the recent discovery of amoA gene in archaea has created the prospect of ammonia oxidising archaea (AOA) being involved too. However, it has been found that the majority of nitrification is carried out by AOB in agricultural soil (Di *et al.*, 2010).

The first step of bacterial nitrification is carried out by a group of obligate autotrophic bacteria known as *Nitrosospira* by the following reaction which produces NO_2^-:

$2NH_4^+ + 3O_2 = 2NO_2^- + H_2O + 4H^+ + \text{energy}$

The conversion from NO_2^- to NO_3^- is affected largely by a second group of obligate autotrophic bacteria known as *Nitrobacter*. This reaction is shown by the equation:

$2NO_2^- + O_2 = 2NO_3^- + \text{energy}$

The overall nitrification process can be represented by the equation:

$NH_4^+ + 2O_2 = NO_3^- + H_2O + 2H^+$

The *Nitrobacter*, *Nitrosomonas* and *Nitrosopira* bacteria are collectively known as *Nitrobacteria* (Sahrawat, 2008). Oxidation of NO_2^- to NO_3^- is a fast process thus NO_2^- does not build up in soil (Cameron, 1992).

The NO_3^- which is produced is highly mobile in soil due to it being a negatively charged ion causing it to be repelled from cation exchange sites.

Some of the most important factors which cause nitrification rate to vary are the population of nitrifying organisms, pH, temperature, oxygen, moisture, and substrate concentration and availability (Sahrawat, 2008). Table 2.1 demonstrates the effect of some of the factors listed above on nitrification rate in soil. For example, soils with acidic pH such as Louisiana clay have lower NO_3^- content than those of higher pH such as Pila clay.

Table 2.1: Nitrification of soil nitrogen in 10 soils with a range in texture, pH, organic C, and total N, incubated aerobically at 30°C for 4 weeks (Sahrawat, 2008).

Soil	pH	Organic C (g/kg)	Total N (mg/kg)	NO_3^- N (mg/kg)
Calalahan sandy loam	3.4	15.7	1100	0
Malinao loamy sand	3.7	12.2	900	0
Luisiana clay	4.4	15.2	1750	0
Morong peat	5.6	128.0	5600	5
Lam Aw peat	6.1	227.0	12000	116
Maahas clay	6.5	15.0	1200	106
Quingua silty loam	6.5	12.8	1150	115
Pila clay	7.5	22.7	1850	123
Lipa loam	7.5	25.0	1900	98
Maahas Clay, alkalized**	8.6	15.0	1200	118

*Maahas clay plus 1.3% Na_2CO_3.

2.2.2. Denitrification

Denitrification is the process of NO_3^- reduction. This process occurs via biological denitrification and chemical denitrification (chemo-denitrification).

Biological denitrification occurs in soils which have anaerobic conditions such as poorly drained waterlogged soil. In these conditions facultative anaerobic bacteria biochemically reduce NO_3^- in sequence to nitric oxide (NO), N_2O and eventually to dinitrogen gas (N_2) (McLaren and Cameron, 1996). These gasses can be lost to the atmosphere during the process as illistrated by Figure 2.2 and each step requires a specific reductase enzyme.

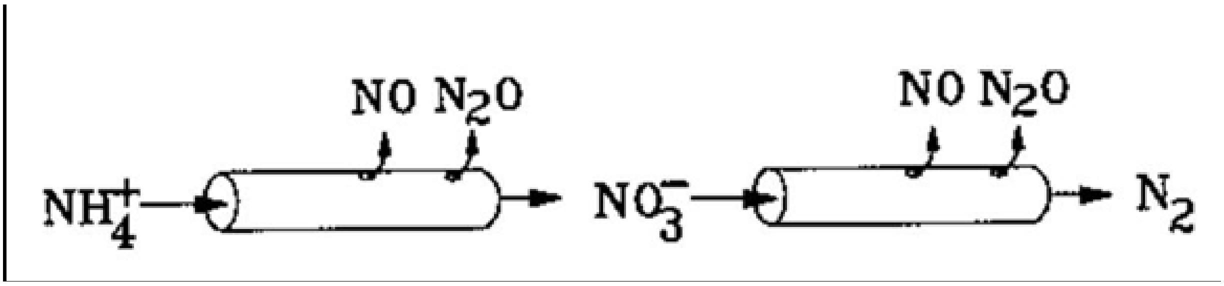

Figure 2.2: "Hole in the pipe" model. A conceptual model of the two levels of regulation of N trace gas production via nitrification and denitrification: flux of N through the process "pipes" and holes in the pipes through which trace N-gases "leak". Taken from (Firestone & Davidson 1989).

The complete process of denitrification is shown by the equation bellow:

$C_6H_{12}O_6 + 4NO_3^- = 6CO_2 + 6H_2O + 2N_2$

The sequence of reduction steps in biological denitrification processes are as follows:

$NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2$

The rate of biological denitrification is affected by many factors such as NO_3^- concentration, oxygen diffusion rate, availability of carbon substrates, temperature and pH. The rate of denitrification increases as NO_3^- concentrations and carbon substrate availability increase but decreases as oxygen diffusion rate increases (Cameron, 1992). Soil pH can have a large effect on denitrification causing the rate to decrease substantially when pH is less than 5 as demonstrated by Figure 2.3 Temperature also has a large effect on the rate of denitrification with the rate being greatly decreased when below 10°C and stopping completely when temperatures drop below 2°C (McLaren and Cameron, 1996). Although this process occurs in anaerobic soils the entire soil does not need to be anaerobic because denitrification may occur in specific parts of the soil, for example, above a poorly drained layer such as an iron pan, or within an aggregate. Studies have shown that if soil redox potential is below 320 mV in any location denitrification is likely to occur (McLaren and Cameron, 1996). Bremner and Shaw (1958) showed the denitrification is minimal when soil moisture content is less than 60% of soil water holding which illustrates the importance of anaerobic conditions to this mechanism.

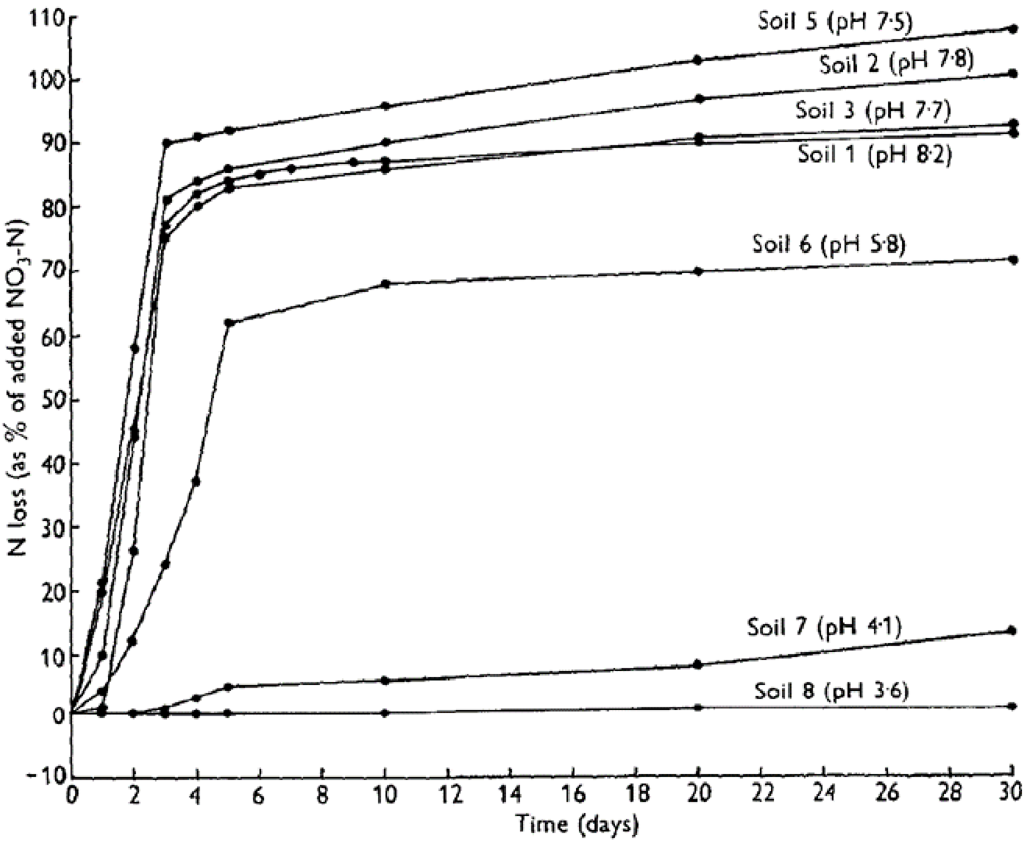

Figure 2.3: Rate of loss of N from different soils on incubation with NO_3^- and glucose. 5 g samples were incubated at 25°C with 11 ml water containing 5 mg NO_3^- N (as KNO_3) and the amount of glucose found to induce maximal loss of N in 20 days (25 mg with soils 2 and 5; 37.5 mg with soils 1, 3, 7 and 8; 62.5 mg with soil 6). (Adapted from Bremner and Shaw, 1958).

Chemo-denitrification is a term used to describe various chemical reactions of NO_2^- within the soil which cause losses of N gases such as N_2 and NO_2 (Cameron, 1992). These chemo-denitrification reactions can occur in soils which are not anaerobic, for instance when NH_4^+ fertilizer has been applied. In this case the chemo-denitrification occurs as the high levels of NH_4^+ restricts the activity of *Nitrobacter* resulting in a build-up of NO_2^- which can cause a volatile loss of N_2 independent of microbial activity. The losses from these chemo-denitrification reactions are highest in soils with acid to neutral pH and mainly involve reactions of NO_2^- or N_2O with organic matter, NH_3 or urea.

The chemical reactions are likely to be similar to those below (McLaren and Cameron, 1996):

Reaction with organic compounds ($R\text{-}NH_2$ = amino acid, HNO_2 = nitrous acid)

$R\text{-}NH_2 + HNO_2 = R\text{-}OH + 2H_2O + N_2$

Reaction with NH_3

$NH_3 + HNO_2 = 2H_2O + N_2$

Reaction with urea

$(NH_2)2CO + 2HNO_2 = CO2 + 3H_2) + 2N_2$

Denitrification may account for the total loss of N from some systems such as rice paddies but in temperate agricultural systems such as in New Zealand, losses are variable but usually less than 40% of the applied N and account for less N loss than that NO_3^- leaching (McLaren and Cameron, 1996). Although limited, results in New Zealand have indicated that average denitrification losses are less than 30 kg N/ha/yr. The climate in New Zealand results in low temperatures (less the 10°C) during the winter which reduces the amount of denitrification that occurs during this often waterlogged period of the year. Thus much of the denitrification occurs in the milder spring and autumn months which have adequate temperature and waterlogging for denitrification to occur.

2.2.3. Mineralization and Immobilisation

Mineralization is the conversion of organic N into mineral N which is performed by soil microbes. This process involves the breakdown of complex proteins to form amino acids and then NH_3. The NH_3 is then rapidly hydrolysed to form NH_4^+ (Cabrera *et al*., 2005). The stage at which NH_3 is released is called ammonification. This process provides both energy and N to organisms. The chemical equation below demonstrates the mineralization/ammonification process. Note that R represents amino-N (Cameron, 1992).

$R - NH_2 + H_2O = NH_3 + R - OH + \text{energy}$

Immobilisation is the reverse of the mineralization process as it is the incorporation of N into microbial tissue. Mineralization and immobilisation occur simultaneously in soil. However, the carbon to nitrogen ratio (C:N ratio) of the material being decomposed will determine whether net mineralization or net immobilisation occurs. When the material being decomposed has a high N content (e.g. Alfalfa 3% N) the C:N ratio is low (<25-1). This causes more mineral N to be produced than is required by the microbes thus the excess mineral N which is produced enters the soil solution where is quickly hydrolysed to NH_4^+ and possibly converted to NO_3^-. If the organic material has a low N content the C:N ratio is greater (>25:1) for instance Table 2.2 shows cereal straw has an organic carbon content of 40% which an C:N ratio of 80:1. This will result in mineral N being taken from the soil solution and a reduction in the amount of N available to plants (net immobilisation) as insufficient N is available to microbes for assimilation.

Table 2.2: Approximate content of organic carbon and nitrogen and C:N ratio of various organic materials (adapted from McLaren and Cameron, 1996).

Organic material	Organic carbon (% dry weight basis)	Total nitrogen (% dry weight basis)	C:N Ratio
Soil bacteria	50	10	5:1
Soil actinomycetes	50	8.5	6:1
Soil humus	2	0.2	10:1
Soil fungi	50	5	10:1
Alfalfa	40	3	13:1
Cereal straw	40	0.5	80:1
Clover	40	2	20:1

There are a variety of factors which can influence the rates of mineralization and immobilisation such as aeration and temperature. These factors have an influence due to their effect on the microbes which carry out the mineralization process. For example Sierra (1997) showed that mineralization rate increased continuously with an increase of temperature and an increase in water filled porosity as shown by Figure 2.4.

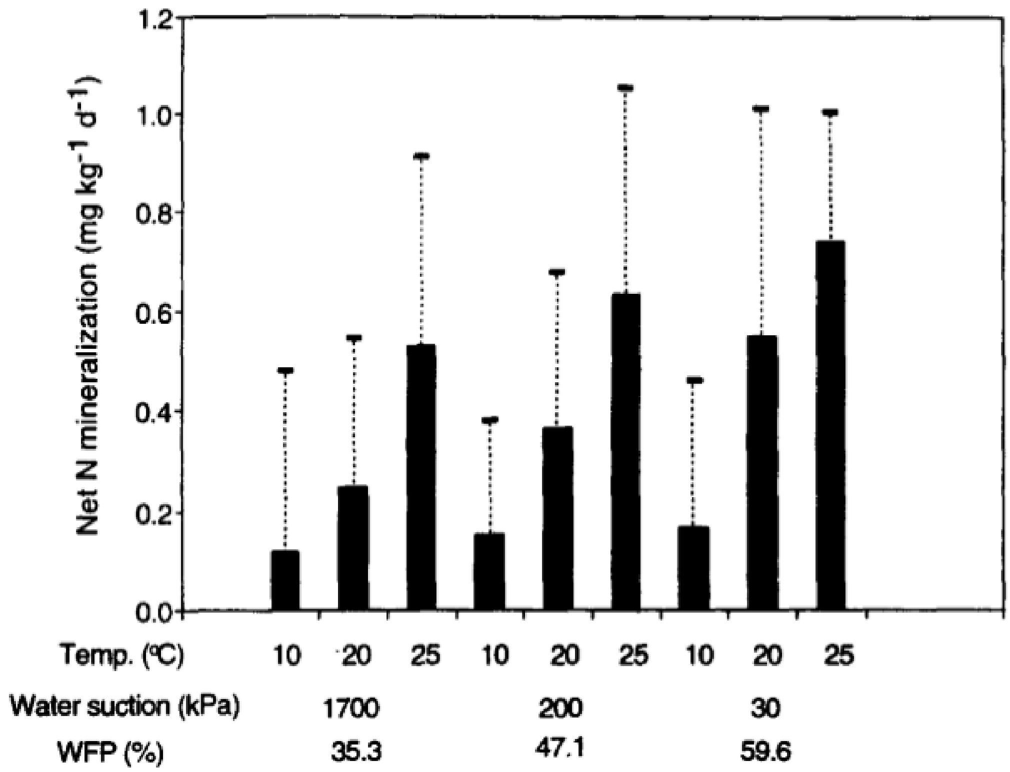

Figure 2.4: Rates of net N mineralization for the nine temperature-soil moisture treatments. Vertical bars indicate standard error (n = 25). WFP is the water filled porosity.

A variety of farm practices can cause variation in these factors and subsequently cause changes in mineralization and immobilisation (Figure 2.4). One of these practices is cultivation, as it aerates and warms the soil, improving the conditions for the microbes, thus resulting in increased mineralization rate.

2.2.4. Solute Transport Mechanisms

Understanding of solute transport is crucial when trying to understanding NO_3^- leaching and the mitigation strategies available. The follow sections discuss the mechanisms of solute movement throughout the soil profile.

Convection

Convection can be described by a modified version of Darcy's law:

$j_c = qc = -c(K*dH/dx).$

where j_c equals the convective solute flux, q equals water flux, c is the solute concentration, K equals the hydraulic conductivity and dH/dx is the hydraulic gradient (McLaren and Cameron, 1996). This flow is predominantly vertical but can occur horizontally.

Convection is the process by which solutes are transported through the soil due to the mass flow of water (Hillel, 2013). Darcy's equation shows that solute flow by convection is dependent upon water flux, thus, the more connected the soil and greater the hydraulic gradient present, the greater the water flux will be. Soil texture has a major effect on this as larger pores result in greater water flux as demonstrated by Poiseuille's law which calculates flow rate of a pore (Figure 2.5).

Poiseuille's law:

$Q=(\pi\ p\ g/\ 8\eta)\ r^4$

where Q is the flow rate, p equals density, g is acceleration due to gravity, η viscosity and r equals the radius of the soil pore.

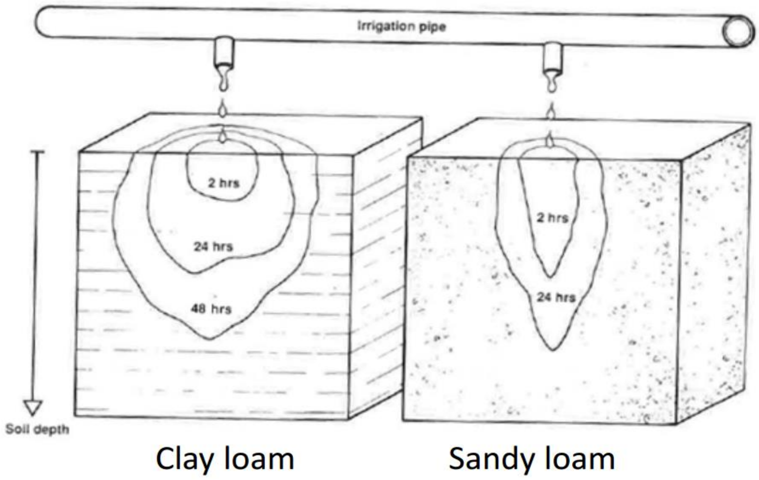

Figure 2.5: Illustration of convection (Adapted from McLaren and Cameron, 1996).

The distance a solute travels per unit time by convection is therefore determined by the average water velocity (U) as shown by the calculation $U = q/\theta_v$ (θ_v = volumetric water content). Hence, if a band of solute was placed onto of a soil it would move down the profile uniformly like a brick moving down the profile if convection was the only mechanism of solute transport. In reality this does not occur due to diffusion and hydrodynamic dispersion.

Diffusion

Diffusion is the mechanism by which solutes move through the soil from areas of high solute concentration to areas of lower solute concentration (Figure 2.6). This occurs as a result of the molecules colliding with one another causing them to be propelled toward areas with less resistance (lower solute concentration). This movement throughout the profile causes equalisation of solute concentrations (Hillel, 2013). The extent of diffusion depends on the solute's concentration and the water content (hydraulic conductivity). For example, soils which are dry and have low solute concentration will have less diffusion occur than wet soils which are high in solutes.

Diffusion can be described by Fick's law:

$J_d = -D_s (\theta) \, dc/dx$

where J_d is the rate of diffusion, $-D_s$ equals the diffusive coefficient of solute in the soil which is dependent upon the moisture content of the soil (θ) and dc/dx which represents the solute concentration gradient (McLaren and Cameron, 1996). Thus greater moisture content and solute concentration gradient results in high levels of diffusion.

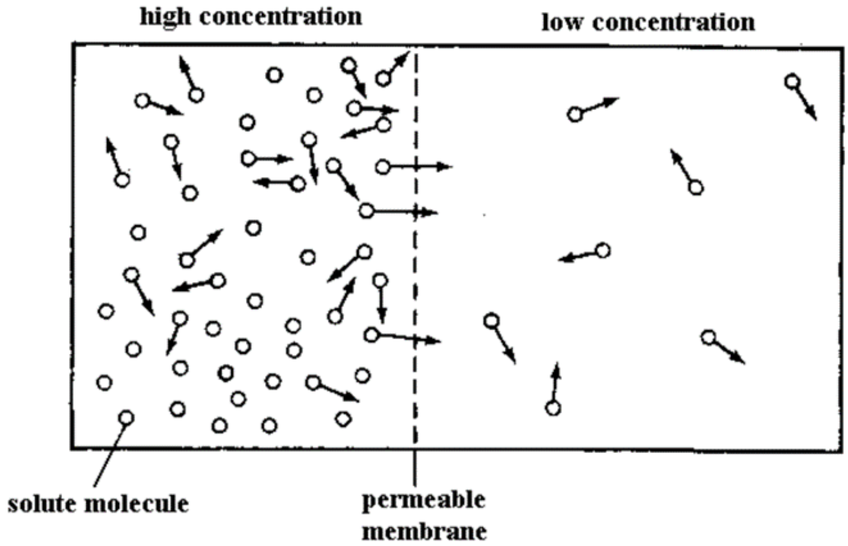

Figure 2.6: Illustration of diffusion (Adapted from Cameron, 2017b).

Hydrodynamic Dispersion

Hydrodynamic dispersion is a mechanism which causes solutes to be transported at different rates resulting in mixing and spreading of solutes throughout the soil profile. This often resembles diffusion and can mask diffusions effects as both cause equalisation of solute distribution through the soil profile. But this process differs from diffusion because hydrodynamic dispersion is not dependent upon solute gradient as it is driven by water flow through the soil. Hydrodynamic dispersion is caused by a variety of factors including the tortuosity of pores which causes flow path length to differ and inconsistent water velocity both within individual pores and between pores (Figure 2.7). Because of these factors, solutes which enter the soil at the same time can be at very different locations after a similar duration of movement. For example, a NO_3^- ion that moves down a vertical macropore will get to much greater depth than a NO_3^- ion which travelled down a tortuous micropore over the same period of time.

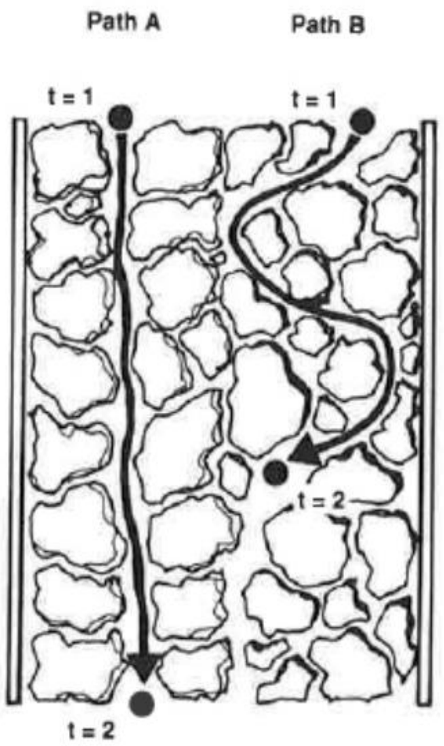

Figure 2.7: Illustration of effect of hydrodynamic dispersion (McLaren and Cameron, 1996).

Convection-Diffusive-Dispersive Transport

The combination of the effects of convection, diffusion and dispersion equal the total flux of dissolved solute. This can be shown by the convective dispersive equation (CDE) which accounts for the transit state conditions, in which fluxes and concentrations can vary in time and space (McLaren and Cameron, 1996).

Convection-dispersive equation: $\delta c/\delta t = D_a\ \delta 2c/\delta x2 - U\ \delta c/\delta x$ (where Da = Ds + mU

Where D_a is the apparent diffusion coefficient as a sum of diffusion and dispersion, c is the concentration of solute, t is time, x equals distance (in the x dimension), m is the dispersivity and U represents the average pore water velocity.

2.3. Factors Affecting Nitrate Leaching

2.3.1. Season and Climate

Season and climate can affect NO_3^- leaching dramatically. One of the main reasons that season and climate affect NO_3^- leaching is due to their effects on drainage, as drainage is a critical factor in determining NO_3^- leaching. For example, seasons with more rainfall and lower evapotranspiration such as autumn and winter have greater drainage resulting in increased NO_3^- leaching. The results of Di *et al*. (1999) supported this showing that within one year 15.1-18.8% of N applied during autumn was leached whereas only 8.5-11.4% of N was leached when applied in spring. This is likely due to the longer period before drainage occurred when N was applied in spring enabling greater plant uptake and gas loss to the atmosphere.

Spring and summer conditions have also shown to affect NO_3^- leaching. This is because when temperatures are high and rainfall is low during these times a greater amount of NO_3^- is leached the following winter. This is because these conditions cause the level of NO_3^- in the soil to build up more than usual, resulting in a greater amount of NO_3^- in the soil come winter. The NO_3^- accumulates as a result of the reduced plant uptake, reduced drainage and reduced denitrification as well as the increased mineralization that occurs during early autumn rainfall as a result of dry hot spring/summer conditions. Because of these seasonal changes, in New Zealand the level of NO_3^- in soil generally builds up over spring and summer and then is leached in drainage during autumn and winter. However, this is not the case in some other climates for example, where spring and summer monsoons occur, NO_3^- leaching can be at its greatest over spring and summer due to the increased drainage.

Higher temperatures can also affect NO_3^- leaching due to their affect on nitrification. As temperature increases so too will the rate of nitrification to a point. Malhi and McGill (1982) supported this with results which showed the optimum temperature for nitrification in soils from central Alberta was 20°C and at 30° activity had almost ceased. This shows that climate and seasonality changes can have a large effect on NO_3^- leaching in a variety of ways.

2.3.2. Soil Properties

There are a variety soil properties that alter NO_3^- leaching. One main soil property that affects this is texture, because fine textured soils cause slower water movement which decreases drainage. Reduced drainage also results greater potential for denitrification to occur. For example, coarse sandy soils of poor structure have shown to be extremely susceptible to NO_3^- leaching when compared to finer textured soil such as clays. The depth of the soil profile above gravels or ground water is also important as NO_3^- reaches the groundwater faster in shallow soils (Di and Cameron, 2002a).

Drainage status also affects NO_3^- leaching (Figure 2.8). This is probably because better drainage causes increased N mineralization and faster movement of NO_3^- into waterbodies (Scholefield *et al.*, 1993). Drainage also affects denitrification as the increased aeration provided by drainage reduces denitrification. The combination of these factors causes more leaching from well drained soils as the factors cause the total amount of NO_3^- in the soil to increase and it to be leached regularly due to the more frequent drainage events.

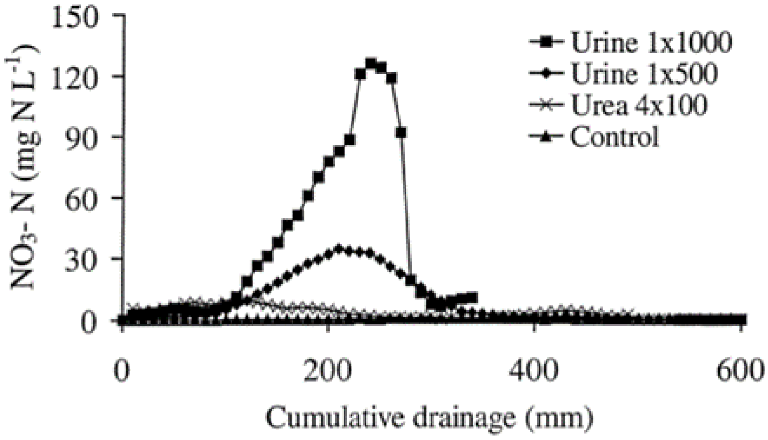

Figure 2.8: Nitrate leaching losses under cow (1000 kg N/ha) and sheep (500 kg N/ha) urine patches (single application) compared to urea applied over 4 applications (Silva *et al.*, 1999).

The abundance of continuous macropores created by earthworms, plant roots or wetting and drying events also causes variation in NO_3^- leaching. This is because theses macropores allow preferential flow which enables a large amount of soil solution to travel through the macropores at high speed carrying N from the soil surface and within the macropores causing the rate of NO_3^- leaching to be increased (Silva *et al*., 2000). But these macropores can also cause bypass flow. This is when water moves down the macropores without distribution through the soil matrix, therefore a large amount of the NO_3^- contained within aggregates remains undisturbed. Thus the variation of NO_3^- leaching caused by macropores is highly dependent upon the location of the NO_3^- throughout the soil profile.

Another soil property which affects NO_3^- leaching is the soil organic matter content and the rate of mineralization of this organic matter. As the rate of mineralization increases more NO_3^- is released into soil solution which will cause increased NO_3^- leaching if not utilised by plants before drainage occurs.

2.3.3. Land Use

The data presented in Table 2.3 and Figure 2.9 demonstrate the effect of land use on N leaching. For example, Table 2.3 shows that beef cattle on clay soil with N application of 200 kg N/ha/yr was estimated to leach 39 kg N/ha/yr, whereas a dairy cow farm on silt loam soil receiving 200 kg N/ha/yr was estimated to leach 59 kg N/ha/yr.

Table 2.3: Example of nitrate leaching losses from production systems (Adapted from Cameron *et al.*, 2013)

N Applied (kg N/ha/year)	Soil texture	Grazing animal/ crop type	Leaching loss (kg N/ha/year)	Location
0	Silty clay loam	Cereal rotation	8	UK
0	Clay loam	Cattle	30	NZ
240	Silty clay loam	Cereal rotation	24	UK
200	Clay loam	Cattle	46	NZ
0	Silt loam	Mixed cropping: Autumn	14-102	NZ
0	Silt loam	Dairy cows	25	NZ
200	Silt loam	No-till corn	8-77	USA
200	Silt loam	Dairy cows	59	NZ
200	Loamy sand	Cereal rotation: Spring wheat	17-87	UK
200	Sandy loam	Dairy cows	54	NZ
169	Clay	Cotton	35	USA
200	Clay	Beef cattle	39	NZ
396	Sandy loam	Corn, carrots	155	USA
400	Sandy loam	Sheep	11-41	NZ

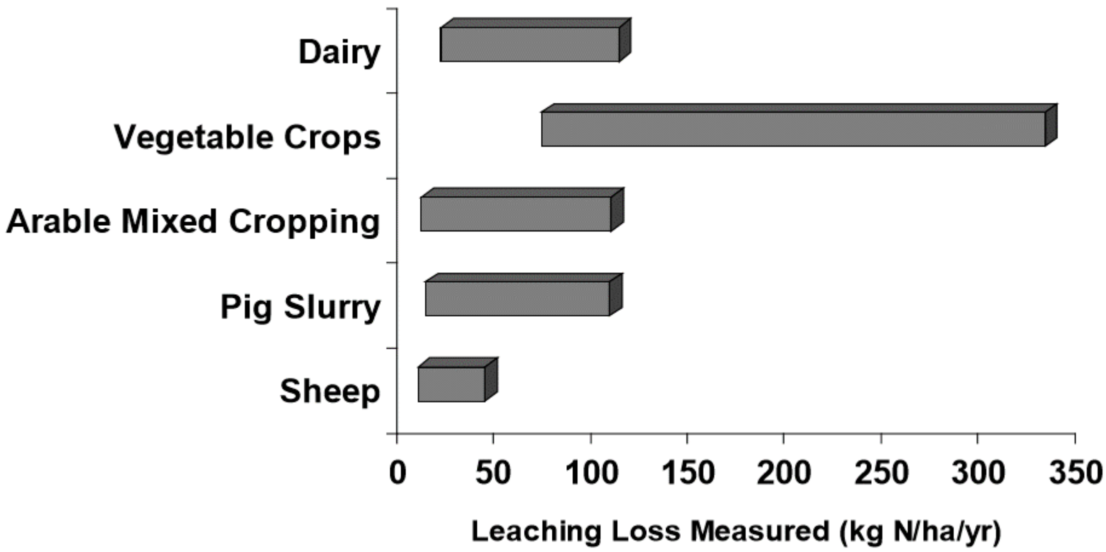

Figure 2.9: N leaching ranges from systems (Cameron, 2017a).

Forest

The amount of NO_3^- leached from forest soil is lower than other agricultural systems such as horticulture and livestock based systems. This is due to a variety of factors such as soil acidity, low N fertilizer application rates and that N is cycled efficiently in forest systems. Soils in forests are generally acidic which is unfavourable to the nitrifying bacteria, thus the production of NO_3^- in these systems is deceased. This along with the low input of N fertilizer (normally zero) into the forest systems decreases the risk of NO_3^- leaching as there are low levels of NO_3^- in the soils. However, by harvesting or burning the forests large amounts of N can be released which can then be leached into surrounding waterbodies. For example, the results of Hornbeck *et al*. (1975) showed an N loss of more than 340 kg

N/ha during a three year period after the clear cutting of a North America forest. But the C:N ratio will ultimately determine the amount of mineralization and thus the potential NO_3^- leaching possible from forest systems.

Grasslands

Leaching from grassland systems is primarily determined by two factors, the amount of excess NO_3^- above plant requirement in the soil and the amount of drainage that occurs (Figure 2.8) (Jarvis, 2000). The NO_3^- in the soil profile comes from N in fertilizer, animal excreta, legume fixation as well as mineralization of organic N. Because of this, grassland systems can vary in NO_3^- leaching dramatically, depending upon farm intensity (N inputs) and the management practices carried out on farm (Figure 2.10). For example Di and Cameron (2002a) found that when N was applied at between 200 – 400 kg N/ha /yr in four applications, NO_3^- leaching ranged from 6 – 17 kg N/ha/yr but when the same amount of fertilizer was applied in two applications, the NO_3^- leaching increased to between 13-49 kg N/ha/yr. This increase in leaching is due to an increased NO_3^- excess in soils as a result of NO_3^- application being less synchronised with plant demand. This demonstrates that management of N supply can have a large effect on NO_3^- leaching in grassland systems.

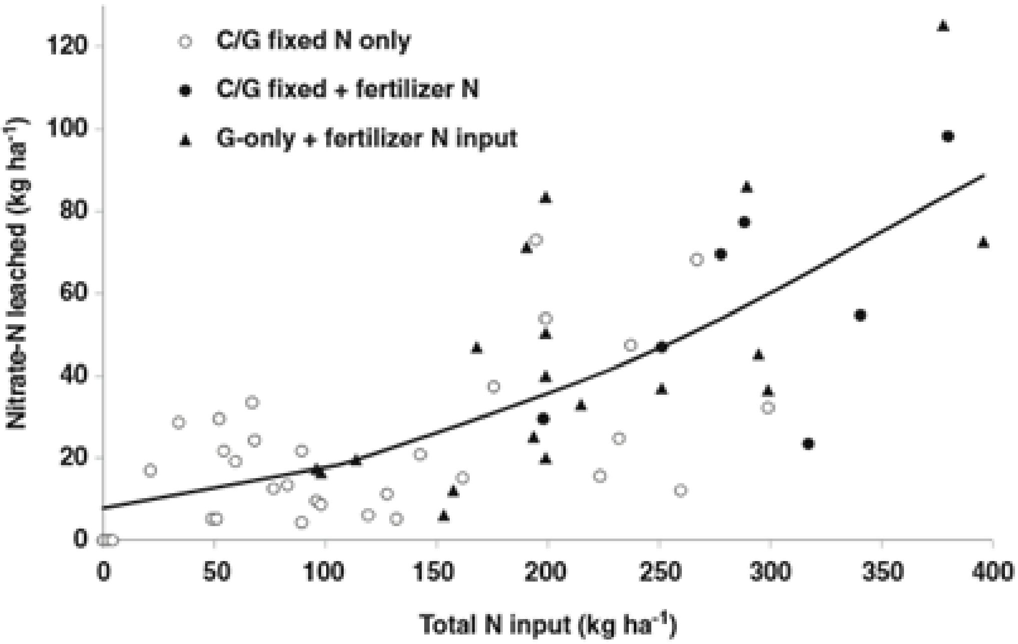

Figure 2.10: Relationship between nitrogen input and nitrate leaching on grassland pastures. (Di and Cameron, 2002a)

Managing N is much easier in grassland systems where the pasture is removed via cutting. Thus they have less NO_3^- leaching than systems where pasture is grazed in situ (Ryden *et al.*, 1984). This is because N is applied in N fertilizers and waste effluent or fixed by legumes in cutting systems thus the amount of N in the system can be better controlled and distributed to mitigate the risk of NO_3^- leaching. This control and distribution of N is not as simple in grazed pastures. This is because of the N which is returned to soil in stock excreta in particular urine as it delivers highly concentrated N which is mobile in soil. Of the N in excreta over 70% is contained in the urine of which 70-90% is in the form of urea (Di and Cameron, 2002a). This urine is deposited to the soil in very small patches (0.5-0.7 m^2) thus the high concentration of N is far too much for plants to utilise. Therefore these urine patches result in a large quantity of excess NO_3^- in the soil which is then leached in drainage. For example the amount of N in a cow's urine patch is equivalent to an application of approximately 700-1000 kg N/ha which is much more than that required by pasture thus leaching occurs as shown by Figure 2.10. This excess of N in soil from urine is often worsened by application of N fertilizers or waste effluent, as the blanket application of these adds to the excess N in the urine patch areas which occupy approximately 25% of the field (Silva *et al*.1999).

2.3.4. Irrigation

Irrigation can have a large effect on NO_3^- leaching both positively and negatively. If irrigation is scheduled correctly increasing plant production without causing additional drainage to occur throughout the year NO_3^- leaching can be reduced. This is because the increased plant production requires more nutrients from the soil thus reducing the NO_3^- content of the soil which is one of the main determinates of NO_3^- leaching. Hahne *et al*. (1977) supported this showing that optimum fertilizer and irrigation rates decreased leaching losses from 48% to 5% of N applied. However, if irrigation is not properly managed, additional drainage caused by over watering can occur, resulting in an increase in leaching. This was demonstrated by the results of Gheysari *et al*. (2009) who showed a significant change when NO_3^- leaching increased from 25 kg N/ha/yr to 47 kg N/ha/yr as a result of an irrigation regime that applied 1.13 times the required irrigation. This shows that excessive irrigation can have a large effect on NO_3^- leaching. Unfortunately the water requirement of soil varies over small distances thus, throughout a field many areas will be over irrigated and other areas under irrigated by irrigation systems which are not capable of performing variable rate application.

Application Rate of Fertiliser and Effluent

Application rates of N fertilizers and effluent can have a notable effect on NO_3^- leaching from production systems. This is because if application rates apply excess N above plant requirement NO_3^- accumulates in the soil. This accumulation of NO_3^- can then be leached from the soil when drainage occurs. Often when annual applications of N are greater than 400 kg N/ha N leaching begins to increase substantially.

Horticulture/Arable

Horticultural and arable systems are some of the most intensively fertilised and cultivated production systems. These systems are often characterised by high N inputs, intensive cultivation, and short periods of plant growth and low nutrient use efficient crops. Also, after the crops are harvested crop residues containing up to 100 kg N/ha are mixed into the soil which release mineral N into the soil when decomposed (Di and Cameron, 2002a). These factors cause horticultural and arable systems to be very susceptible to high levels of NO_3^- leaching thus as a production system they are the highest contributors to NO_3^- leaching on a per hectare basis (Figure 2.9).

The regular cultivation of these systems causes a large amount of NO_3^- leaching as a result of the consequential increase in mineralization resulting in increased NO_3^- concentrations in the soil. For example Francis *et al*. (1992) reported a 60 kg N/ha/year loss after cultivation of a 3 year grass ley. These arable and horticulture systems also often leave fields bare for periods of time which causes NO_3^- leaching to increase as a result of the absence of plant uptake of NO_3^-. The results of Miller (1906) supported this showing lysimeters with bare ground and no fertilizer still leached 42 kg N/ha.

Arable and horticulture systems are similar in terms of leaching. But horticulture systems use higher rates of N fertilizer and the plants have shallow root systems which cause greater NO_3^- losses than from most arable systems. The shallower plant root systems of horticulture crops reduce the plant's ability to uptake nutrients from the soil resulting in a larger amount of unutilised NO_3^- being leached. For example, horticulture application rates of nitrogen can be up to 600 – 900 kg N/ha of which often only 50% is utilised by the crop but this utilisation can be as low as 20% (Pionke *et al*., 1990). This results in a large amount of excess NO_3^- in the soil which is frequently leached in drainage as shown by recorded NO_3^- losses of up to 300 kg N/ha/yr from horticultural farms in New Zealand (Spiers *et al*., 1996).

Stock Type

Stock type can have a large effect on the quantity of NO_3^- leached from pastoral systems. The main reasons for this is that the concentration of N deposited from stock urination varies between stock types. For example, sheep urine generally deposits N onto soil at a rate of 300-500 kg N/ha which is half that of the approximately 700-1000 kg N/ha deposited in cow urine (Cameron *et al*., 2013). Because of this variation in urine-N deposits, pasture grazed by cows has greater NO_3^- excess under urine patches than that of pastures grazed by sheep resulting in higher levels of N leaching from these areas.

Winter forage grazing

Winter forage crops such as kale or fodder beet are sown in the previous spring for use as winter feed. These crops reach yield of between 10-16t DM/ ha by the time of grazing at the start of winter (Brown *et al*., 2007). This provides a large amount of stock feed at a time were growth of pastures is low due to cold temperatures. Unfortunately winter forage grazing can cause a large amount of NO_3^- leaching to occur. There are a variety of factors which contribute to the large N leaching loss from these winter forages such as high stocking rates used during grazing, high winter drainage rates and lack of plant uptake of urine-N from the grazing cattle as the paddocks generally remain fallow until spring. The combination of these factors can lead to extreme N leaching from areas where these crops are grown. For example New Zealand studies have resulted in nitrate leaching losses ranging from 52 to 173 kg N/ha from winter forage crop grazing (Shepherd *et al*., 2008; Smith *et al*., 2012; Monaghan *et al*., 2013; Malcolm *et al*., 2015b). Because of this, mitigation options which focus on the management of winter forage crops have the potential reduce farming systems N leaching substantially.

2.4. Strategies for Decreasing Leaching Losses from Winter Grazed Fodder Beet soils

Because NO_3^- leaching has many sources and is affected by a variety of factors there are a lot of strategies available that will enable NO_3^- leaching to be decreased. A combination of many of these strategies will be necessary to develop best management practice (BMP) to optimize the N use efficiency of plants for optimum production, while significantly reducing NO_3^- leaching. The general aim of BMP is to reduce excess NO_3^- accumulation above plant requirement especially during periods of frequent drainage. However, knowledge of the use of catch crops is the main purpose of this literature review, hence catch crop use will be the only mitigation strategy discussed in detail. Other strategies for decreasing NO_3^- leaching from production systems include reducing N surplus through the use of standoff pads, N budgets, improved reproductive performance (to reduce quantities of replacement stock), genetic improvement, exploitation of plant traits, use of buffer zones, nitrification inhibitors, N application management as well as reduced stocking rate and increased production per stock unit.

2.4.1. Catch Crops

Catch crops are an effective method for reducing NO_3^- leaching as they decrease the soil NO_3^- pool due to plant uptake. Grazing of winter forage crops presents a new challenge for reducing N losses because large volumes of urine are deposited onto bare soil with no opportunity for plant uptake of N at a time of year when drainage rates are typically large. Because of this, the use of catch crops to soak up excess N from soil are a potential mitigation method to reduce N leaching. For example, Wyland *et al*., (1996) used two catch crop treatments, Phacelia and Merced rye (*Phacelia tanacetifolia* cv. 'Phaci', and *Secale cereale* cv. 'Merced'), with a fallow control and showed a 65-70% reduction in NO_3^- leaching due to catch crops illustrating the potential benefits which are possible. The results of Carey *et al*. (2016) reinforced this showing a catch crop of oats (*Avena sativa* L.) sown between 1 and 63 days after the urine deposition onto Kale (*Brassica oleracea* L.) in early winter reduced N leaching losses from urine patches by ~34% on average (range: 19–49%) over the winter–spring period compared with no catch crop. However, both of these studies were lysimeter based trials, thus lacking the realism of a field-scale trial. Also neither study used fodder beet as the winter feed crop. Malcolm *et al*. (2016) has done a trial on N leaching from fodder beet and found total NO_3^- leached from the urine treated lysimeter represented an equivalent of 21% (64 kg NO_3^- N /ha) and 32% (84 kg NO_3^- N /ha) of the total urine-N applied in 2012 and 2013 respectively. However, this trial was also lysimeter based and did not incorporate the use of a catch crop.

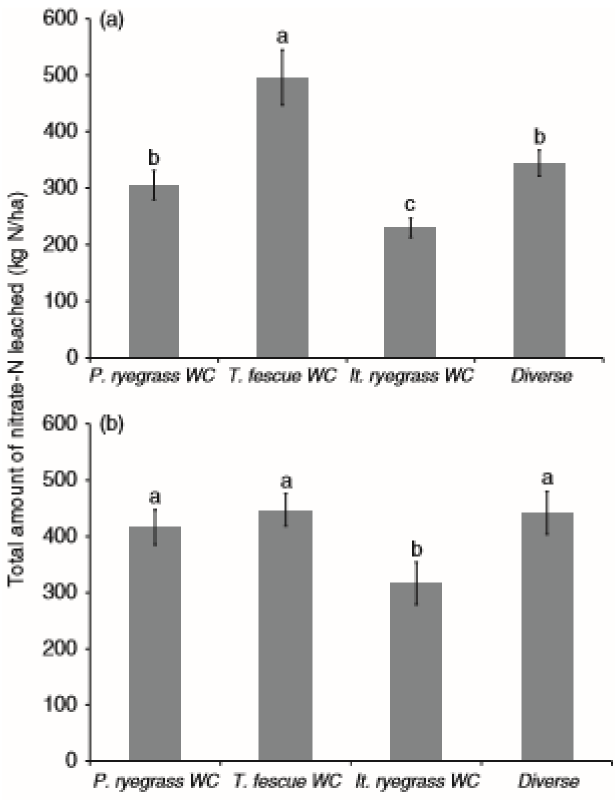

Figure 2.11: Total amount of NO_3^- leached from (a) 2010/11, and (b) 2011/12 lysimeters. Vertical bars indicate +/- 1 standard error of the mean (SEM). Means with a letter in common are not significantly different at the 5% level. (Adapted from Malcolm *et al*., 2014).

Plants such as Italian ryegrass and oats are suitable for the purpose of catch crops as they remove large amounts of N from the soil and are winter active (Figure 2.11). It is important for the catch crops to be winter active as it is vital that the crop can be planted and quickly begin to utilise the NO_3^- from the soil before leaching removes the NO_3^- from the root zone. For example Malcolm *et al*., (2014) found that a pasture mix of Italian ryegrass and white clover resulted in 24-54% less leaching under cow urine patches than other less winter active pasture mixes tested (Figure 2.11). The results of Woods *et al*. (2016) reinforced this in part as shown by the winter uptake portion of Table 2.4 which shows winter N uptake (June-August) from the urine treated ryegrass white clover mix was 37.3% less than the urine treated Italian ryegrass. However, herbage yields for urine-treated forage during the experimental period of the 17 month trial were 16% lower for Italian ryegrass than for the ryegrass white clover mix. This indicates that Italian ryegrass grew more during the cool winter period than ryegrass white clover mix and was able to take up more N in this period. This was supported by both Malcolm *et al*. (2014)

and (2015a). This in turn supports the hypothesis that high winter activity is a key requirement of a successful winter catch crop for reducing NO_3^- leaching.

Table 2.4: Total herbage dry matter yield harvested (t DM/ha), nitrogen uptake harvested (kg N/ha) from lysimeters for the experimental period: 7 May 2014 to 1 October 2015, and nitrogen uptake harvested (kg N/ha/d) for winter herbage. Perennial ryegrass and white clover (RGWC) and Italian ryegrass (Italian RG) treated in May 2014 with either: control (no urine, water) or urine (700 kg N/ha). Means with the same letter (a-d) are not significantly different at the 5% level (Adapted from Woods *et al*., 2016)

Forage Type	Treatment	Total herbage yield harvested (t DM/ha)	Herbage N uptake harvested (kg N/ha)	Harvest 23/06/2014 (kg N/ha/d)	Harvest 07/08/2014 (kg N/ha/d)	Mean winter N uptake (kg N/ha/d)
RGWC	Control	20.9c	692b	0.72	0.5	0.62c
RGWC	Urine	24.3a	811a	1.6	1.62	1.61b
IRG	Control	12.2d	246d	0.22	0.1	0.16d
IRG	Urine	20c	629b	1.91	2.3	2.10a
P Value	Forage type	<0.001	<0.001			<0.001
LSD		2.050	83.50			0.1594

Use of catch crops can also have economic benefits, for example, the grown crop can be utilised by harvesting it for use as supplementary feed. This method would result in the nutrients being recycled back to the pasture eventually to further increase production while also achieving the overall goal of reducing N leaching losses.

2.5. Conclusions

- Nitrate leaching is affected by a variety of factors including climate, season, soil properties and land use.
- Increased drainage causes greater NO_3^- leaching.
- Improved plant production can decrease soil NO_3^- levels.
- Winter grazing of forage crops such as kale can lead to large amounts of NO_3^- leaching form agricultural systems.
- The use of catch crops to reduce NO_3^- leaching from winter grazed forages has shown to be effective in studies thus far.
- Winter activity is a key trait needed for a successful catch crop.
- This review has lead me to conclude that there is a lack of information regarding the effect of a catch crop following fodder beet and that there is a lack of paddock scale information regarding N leaching.

Chapter 3
Materials and Methods

3.1. Site Description

The study was conducted using the suction cup and lysimeter array (SCALAR) system at the Lincoln University Ashley Dean Research and Development Station near Springston on the Canterbury Plains of New Zealand (43°38’43.4”S 172°20’50.8”E; 17 m above sea level)(Plate 3.1). The site is 50 m x 50 m and was selected due to its flat topography and because it fitted between a lateral span of an irrigator (Figure 3.1). The soil type at this site was Balmoral stony silt loam (Mottled Argillic Pallic Malcolm *et al.*, 2015b). This soil type is stony with silt loam texture and is free draining which causes it to be prone to N leaching.

Plate 3.1: Drone view of SCALAR site showing oat and fallow plots (27/8/2017; Lincoln University Ashley Dene Research and Development Station SCALAR site).

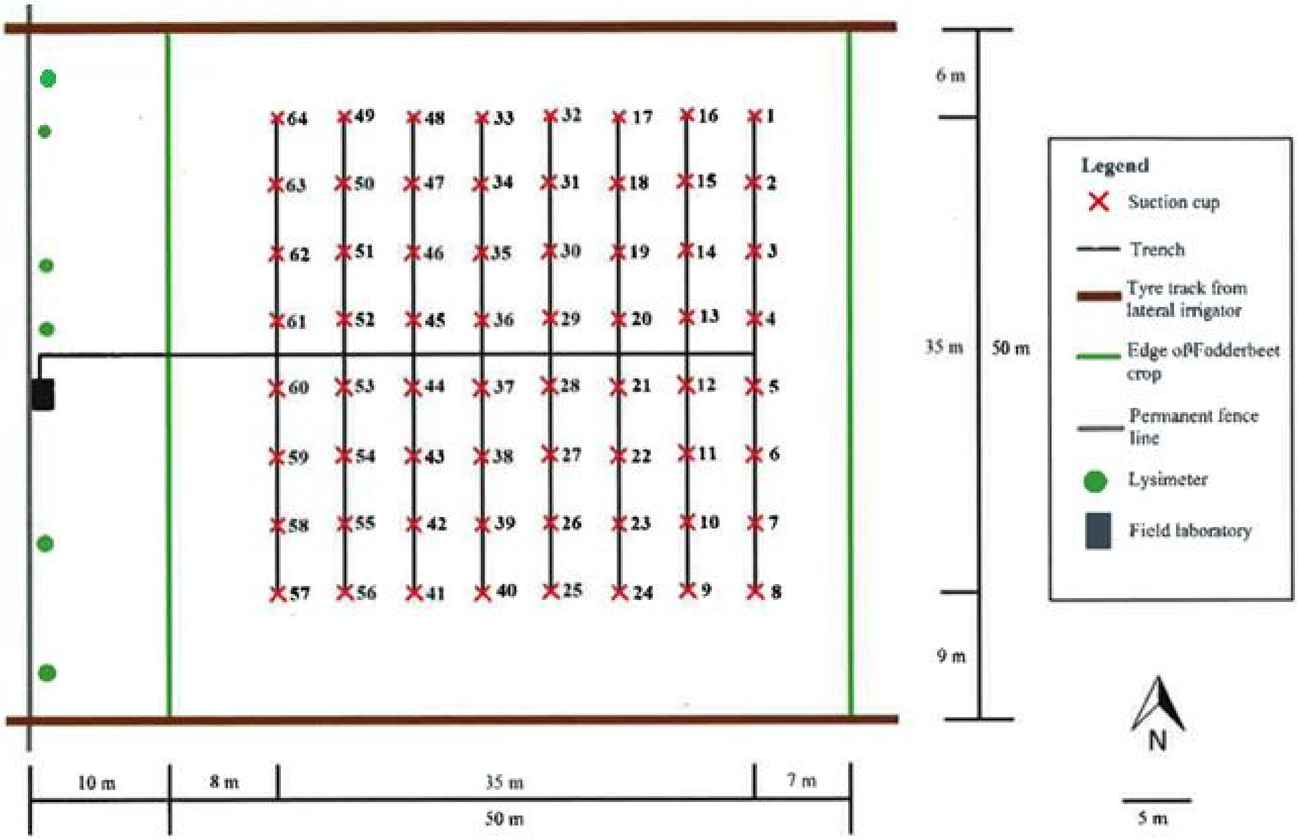

Figure 3.1: Site layout for suction cup and lysimeter array (SCALAR) at Ashley Dene Research and Development Station.

The SCALAR system is comprised of 64 suction cups and 6 lysimeters. The suction cups are installed at a 45° angle bellow the cultivation layer (35cm deep) at 5 m intervals in an 8 x 8 grid pattern allowing for independent drainage measurements over a 40 m x 40 m (0.16 ha) area as show by Figure 3.1. This design allows for measurements within +/-20% of the true value, based on data from Lilburne *et al.* (2012). Each suction cup has two tubes attached to it which are also under the cultivation layer: one is for collection of the samples and one for adding vacuum and pressure (Figure 3.2). The suction cups are under -10 to -16 k Pa when drainage is occurring so that samples of drainage solution are taken rather than samples of total soil solution. The 6 lysimeters are installed on the west side of the suction cup array and are for the purpose of measuring drainage rate and volume. When drainage occurs from the lysimeters causing the tipping bucket beneath to operate a signal is sent to start the vacuum pump causing suction to be placed on each suction cup. The suction cups will then collect samples of drainage which they store until collection takes place at which time positive pressure is added to the suction cups causing the drainage samples to be pushed up the individual collection tubes into vials in the field laboratory (Plate 3.2). These samples are then manually taken to the Lincoln University where they are stored at -4°C until analysis.

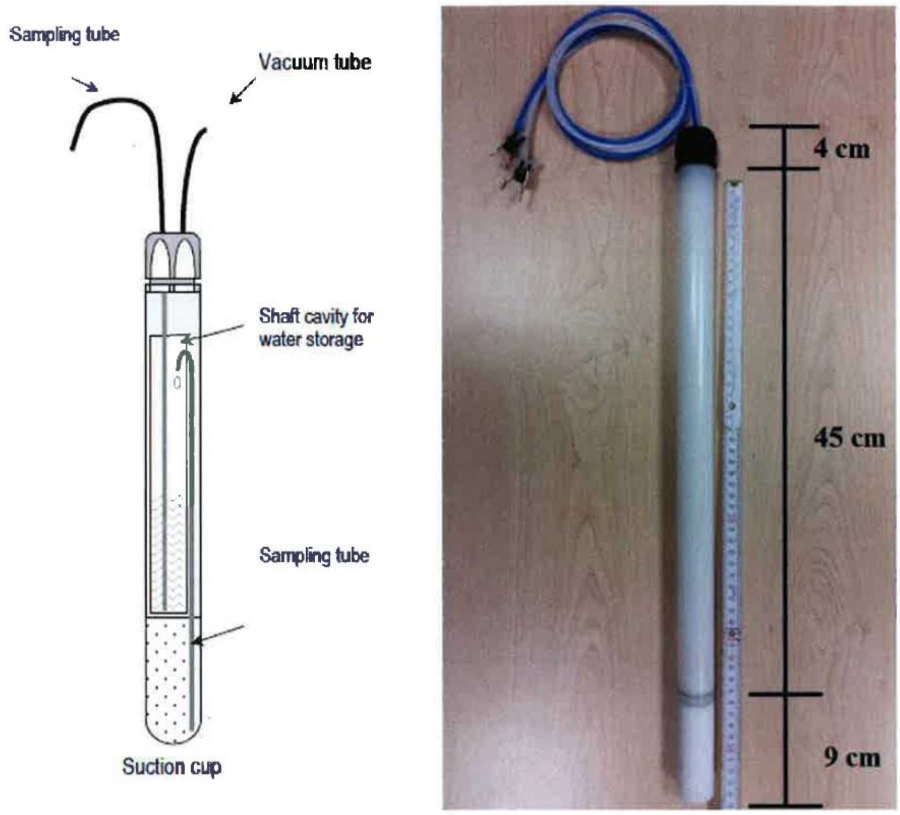

Figure 3.2: Design of suction cups used in SCALAR system.

Plate 3.2: Manifold and collection board inside the field laboratory (6/6/2017; Lincoln University Ashley Dene Research and Development Station SCALAR site).

3.2. Site Preparation

The area was under fodder beet in the previous year, before which it had been under pasture. The paddock was cultivated and sown with fodder beet (cv 'Rivage') at a rate of 100,000 seeds/ha in

October of 2016. This crop received the following management in terms of cultivation, herbicide, fertilizer and irrigation treatments as shown in Table 3.1.

Table 3.1: Management of Fodder beet (cv 'Rivage').

Management	Description
Cultivation date	15th October 2016
Sowing date	29th October 2016
Sowing rate	100,000 seeds/ha
Fertilizer at sowing	250 kg/ha Cropmaster 20, 350 kg/ha NaCl, 200 kg/ha KCl, 15 kg/ha Boronate (10%B)
N fertilizer during growth	100 kg N/ha as urea (28th November 2016) 70 kg N/ha as urea (25th January 2017)
Herbicide application	2 L/ha Norton (29th October 2016) 0.2 L/ha Norton , 0.5 L/ha Bentanal Forte and 1 kg/ha Goltix (29th November 2016)
Irrigation	15 mm (23rd December 2016) 150 mm (January – March 2017)

3.3. Grazing

The fodder beet above the SCALAR was grazed by 85 in-calf mixed-age cows from the 18th to 29th of June 2017. Over the grazing period of the trial the stock spent 24 hours per day on the site. However, the site was not back fenced meaning that stock were able to move off the site into areas of the paddock where suction cups were not present. After the site had been grazed the stock had access until the 10th of July. This was to account for movement of stock to the new break and back grazing to make the trial more realistic. When grazing the site the break fence was moved forward approximately 3 m /day at roughly 10 am. The cows were adjusted to the fodder beet diet prior to grazing the site. This was done by grazing an earlier section of the paddock for approximately 13 days before beginning grazing of the SCALAR site as shown in Plate 3.3 . Table 3.2 illustrates the grazing which took place.

Table 3.2: Timing of SCALAR row grazing events.

Suction cup row (South to North)	Date
Row 1	18th June 2017
Row 2	20th June 2017
Row 3	22nd June 2017
Row 4	23rd June 2017
Row 5	25th June 2017
Row 6	26th June 2017
Row 7	27th June 2017
Row 8	29th June 2017
Fenced off from stock	10th July 2017

Plate 3.3: Cows adjusting to fodder beet diet prior to grazing SCALAR site (6/6/2017; Lincoln University Ashley Dene Research and Development Station SCALAR site).

3.4. Treatment establishment

The oats (*Avena sativa* (cv 'Milton')) and the fallow plots were established in a randomised block design with each block unit as a 5 m x 20 m area above four suction cups running west to east across a grazing row as shown by Figure A 1. The oat crop received the management as shown in Table 3.3. The surface of the lysimeters were also altered at this time with three being cultivated and sown with cv 'Milton' oats and the other three being pugged using a manually operated trampling device, designed to provide c. 200 kPa, similar to that of the mechanical hoof described by Di *et al*. (2001). This meant that the lysimeters better represented the treatments so that drainage rate could then be recorded separately for the two treatments from this point onwards.

Table 3.3: Oats (cv 'Milton') treatment management.

Management	Date
Grubbing	1st August 2017
Power harrow and Cambridge roll	7th August 2017
Sowing date	8th August 2017
Bird covers laid	9th – 23rd August 2017

3.5. Measurements and Data Collection

A total of 64 suction cups were used for the collection of drainage samples from the SCALAR site. Leachate samples from these suction cups were first collected on 6th of April to analyse for carry over N from the previous year. Leachate was then collected between the 6th of June and the 16th October when sufficient drainage occurred. These samples were analysed for NO_3^- and NH_4^+ concentrations using sequential flow injection analysis (FIA).

Drainage measurements were collected via tipping bucket systems located under each of the lysimeters. These measurements were automatically sent to a database accessible at Lincoln University. This automated drainage measurement system ran continuously from 1/1/2017 onwards. An average of five tipping buckets were used to calculate drainage up to the point of sowing of the oats, at which time a sixth lysimeter and tipping bucket was installed. This enabled the lysimeters to be altered as described in part 2.5 so that measurements from three of the tipping buckets would represent drainage of the fallow treatment, and the remaining tipping buckets would represent drainage from oat the treatment.

The SCALAR site also has its own weather station capable of measurements such as rainfall, temperature, soil moisture, relative humidity and vapour pressure. These measurements were measured hourly and automatically updated to the database accessible at Lincoln University. Additional rainfall measurements were also collected via five manual rain gauges located on the west side of the SCALAR site. Figure 3.3 illustrates the measurement devices present at the SCALAR site.

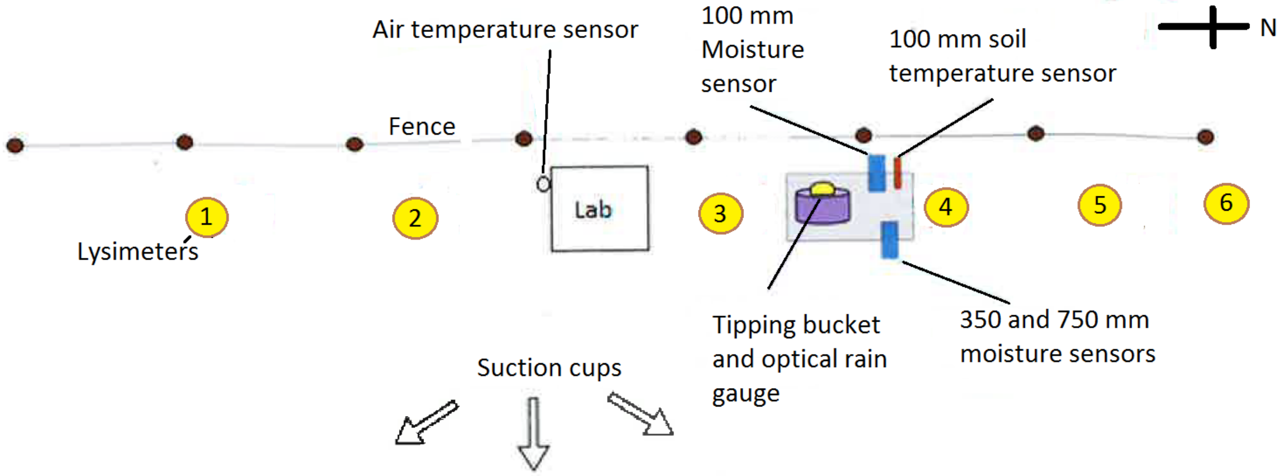

Figure 3.3: Sensor systems located on site Lincoln University Ashley Dene Research and Development Station SCALAR site).

3.6. Statistical Analysis

Statistical analysis involved a 't'-test comparing the oat and fallow treatments at the end of the pre-and-post sowing periods to test for differences in mean NO_3^-, NH_4^+ and total N leaching. ANOVA was also performed to analyse for differences in total N leachate between grazing rows over the pre sowing period.

Chapter 4

Results

4.1. Pre Sowing Results.

4.1.1. Climate

The results presented in Figure 4.1 show 248.5 mm of rainfall occurred prior to sowing of the oats. This resulted in drainage over this period reaching a total of 192.5 mm.

Mean daily soil temperatures at 50 mm, 100 mm and 150 mm deep were 5.8°C, 6.5°C and 6.8°C respectively. Daily air temperature over the pre sowing period averaged 5.9°C. Air temperature fluctuation over the pre sowing period ranged from 1.8°C to 13.9°C whereas the soil temperatures at 50 mm and 150 mm ranged from 1.8°C to 9.1°C and 4°C to 8.8°C respectively (Figure 4.2).

Mean daily soil moisture at 150 mm, and 50 mm deep were 0.29 and 0.34 Vw/Vt and ranged from 0.253 to 0.348 and 0.28 to 0.407 respectively as illustrated in Figure 4.3.

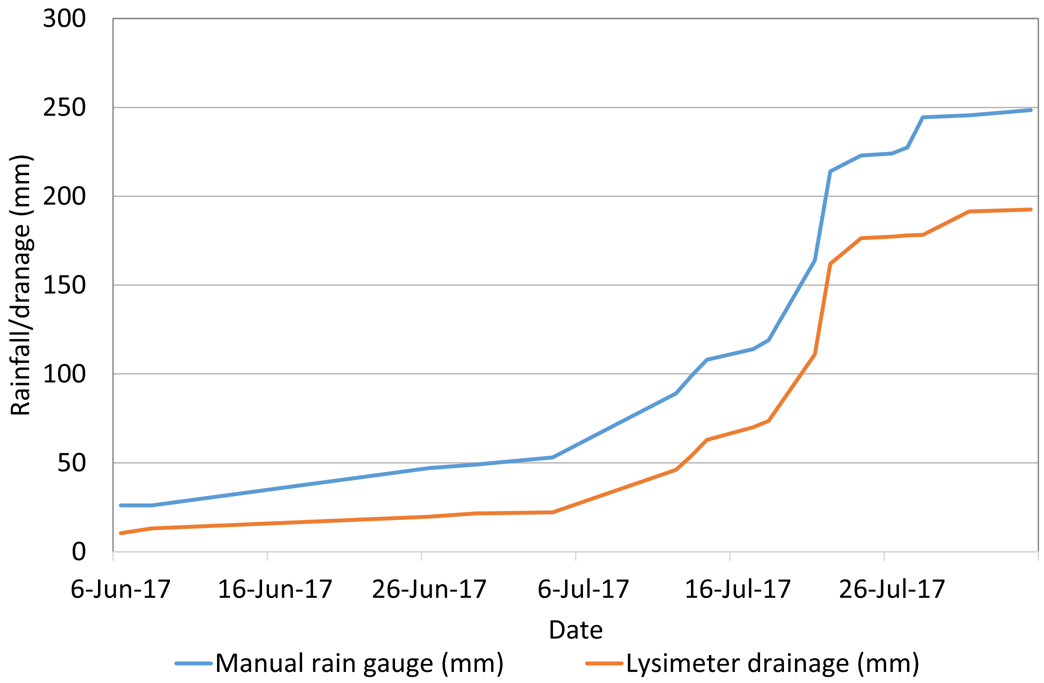

Figure 4.1: Cumulative average lysimeter drainage and rainfall inputs pre sowing.

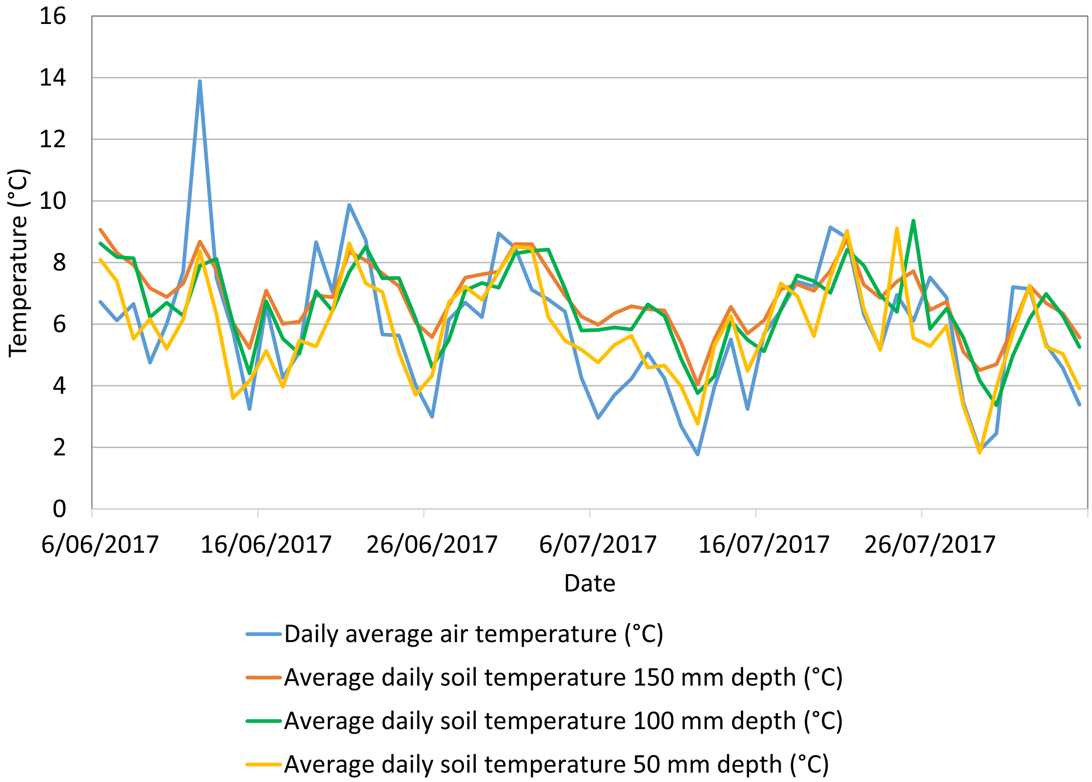

Figure 4.2: Daily mean temperatures at SCALAR site pre sowing period.

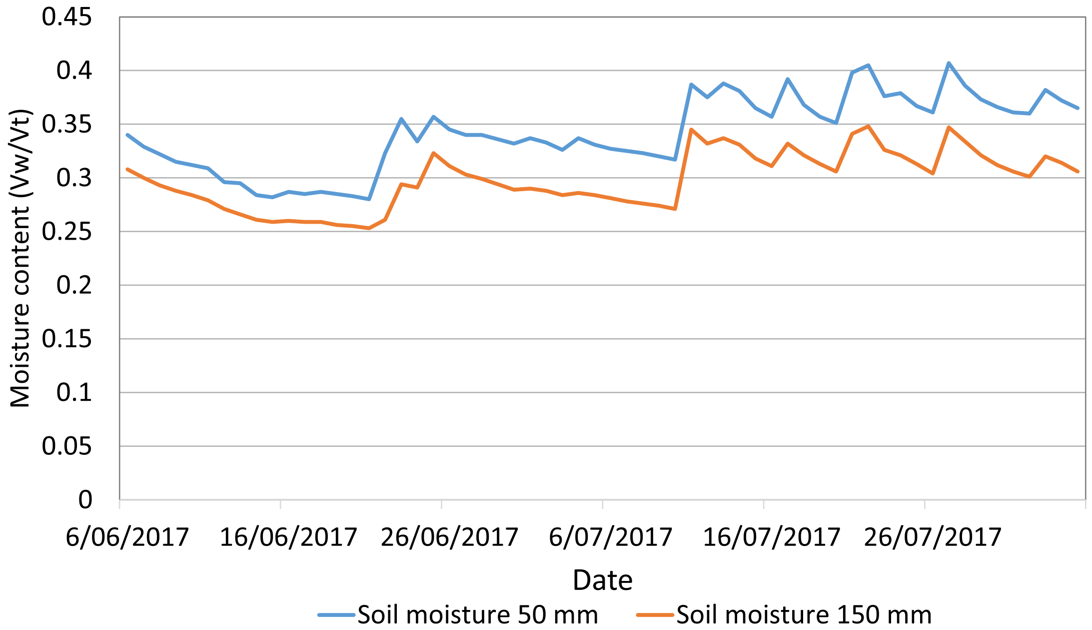

Figure 4.3: Soil moisture content during pre-sowing period.

4.1.2. Nitrogen Leaching

The concentration of NO_3^- increased as a result of grazing of the fodder beet (i.e. after 18-29th June) (Figure 4.4). NO_3^- concentrations in leachate continued to rise until the time of sowing where the peak average concentration pre sowing was reached at 6.8 mg N/L as illustrated in Figure 4.4. The mean concentration of NO_3^- in leachate over the pre sowing period was 4.5mg /L.

Figure 4.5 shows that the NH_4^+ concentration also increased as a result of fodder beet grazing. However the concentrations were much smaller and more variable than that of NO_3^-. Average NH_4^+ concentration leached per cup pre sowing reached a peak of 0.74 mg N/L on the 18/7/2017. The mean concentration of NH_4^+ over the pre sowing period of the trial was 0.5 mg /L.

Using the drainage rate and the concentration data from the leachate such as illustrated in Figure 4.1, Figure 4.4 and Figure 4.5 it was calculated that the average amount of N leached during the pre-sowing period was 9.6 kg N/ha. Of this, 9.43 kg N/ha, 8.6 kg N/ha was leached in the form of NO_3^- and 0.83 Kg N/ha was leached as NH_4^+ as shown in Figure 4.6. This meant that on average 91.2% of the N leached was in the form of NO_3^-.

The rows of suction cups in Figure 4.6 are presented in the order in which they were grazed during the trial. Row 1 was grazed first on the 18th of June and row 8 last on the 29th of June as shown by Table 3.2.

Figure 4.7 shows the cumulative amount of nitrogen leached per cup over the pre sowing period of the trial. This illistrates the range of nitrogen leached over the area of the scalar site. Additionally, statistical analysis showed that there was no significant difference (P=0.49) in N leaching between suction cups used for the fallow and catch crop areas pre sowing.

Table 4.1: Summary of Mean Cumulative Nitrogen leached over pre-sowing period. All were insignificant (P>0.05).

	Fallow plots (kg N/ha)	Catch crop plots (kg N/ha)	P value
Total nitrogen	8.83	10.03	P=0.493
Nitrate	8.26	8.93	P=0.642
Ammonium	0.57	1.1	P=0.464

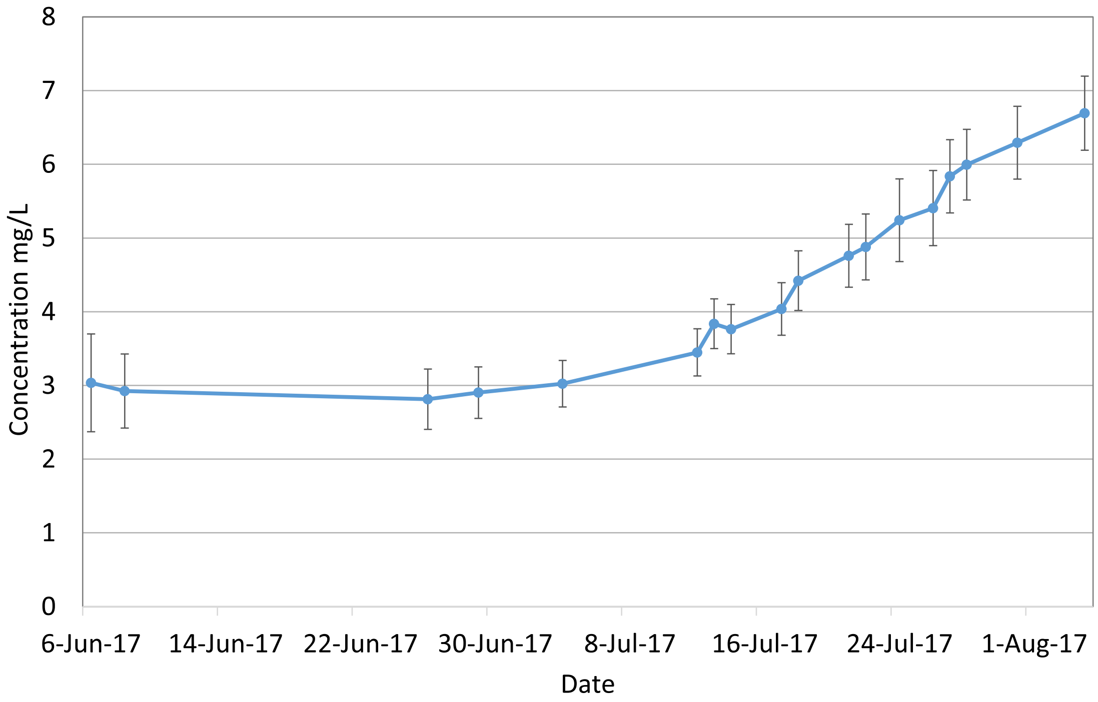

Figure 4.4: Concentration of nitrate in leachate pre sowing. Vertical bars represent standard error of the mean.

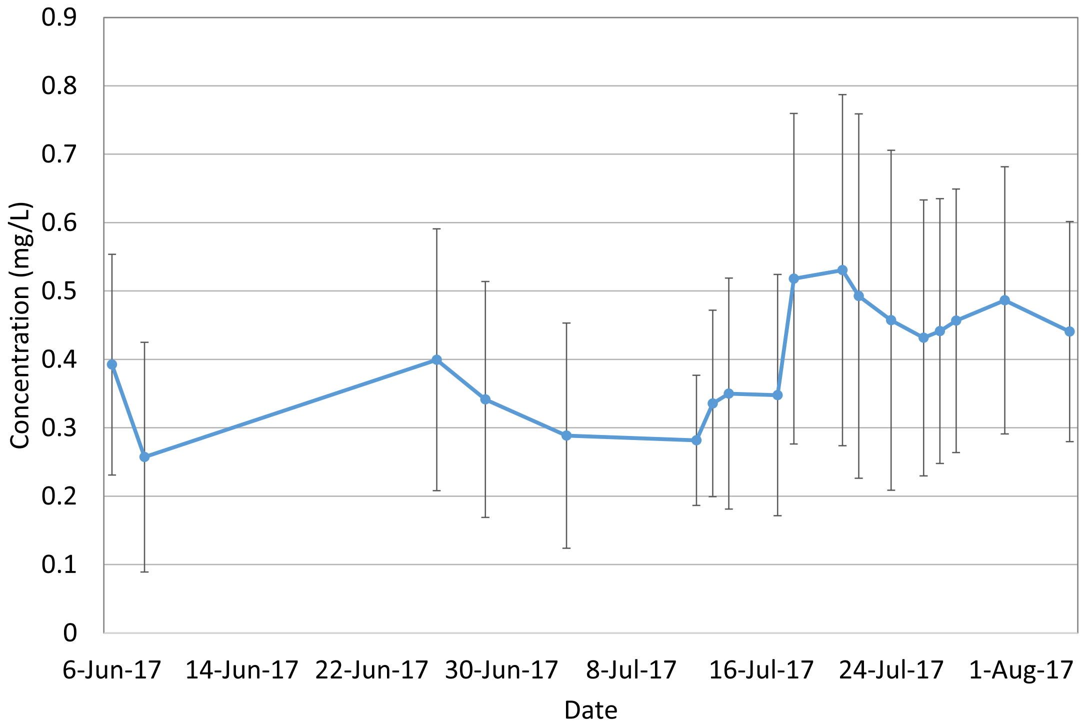

Figure 4.5: Concentration of ammonium in leachate pre sowing. Vertical bars represent standard error of the mean.

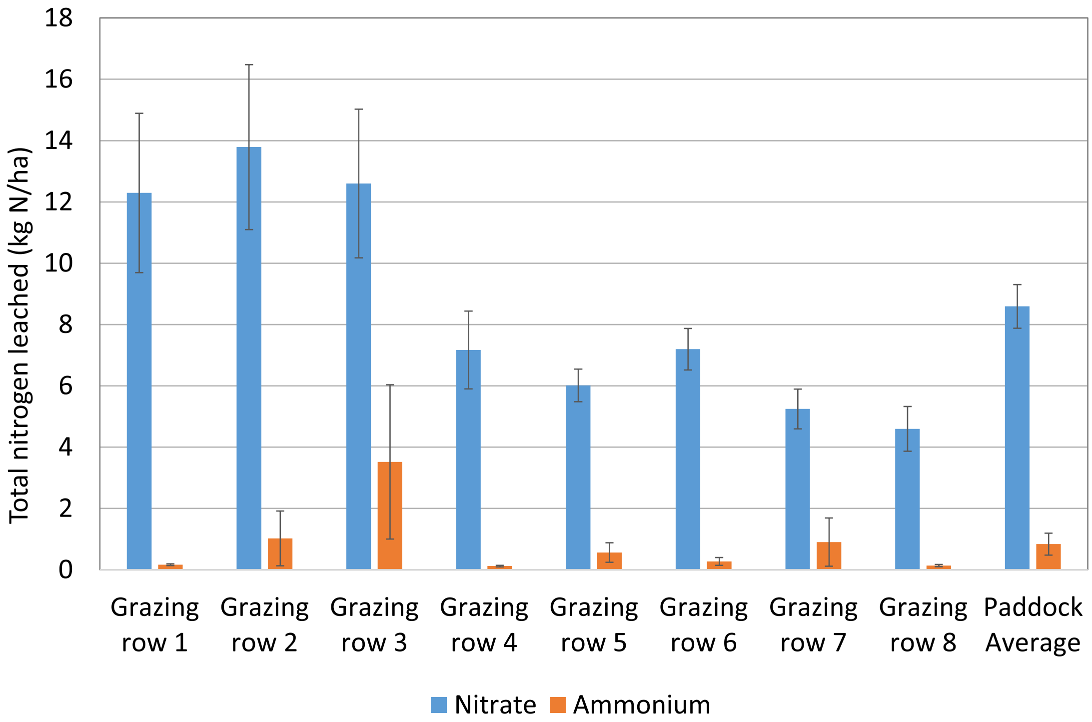

Figure 4.6: Mean cumulative nitrate and ammonium leached per row pre sowing. Vertical bars indicate standard error of the mean. There was found to be a significant difference between rows of nitrate (P<0.001) but not that of ammonium (P=0.256).

Table 4.2: Mean Cumulative amount of Nitrogen leached pre-sowing.

Grazing row	Mean Nitrate leached (kg N/ha)
Row 1	12.46 ab
Row 2	14.78 b
Row 3	16.15 b
Row 4	7.29 ac
Row 5	6.58 ac
Row 6	7.47 ac
Row 7	6.24 c
Row 8	4.73 c

Means within the same columns with differing superscripts are significantly different (P<0.05)

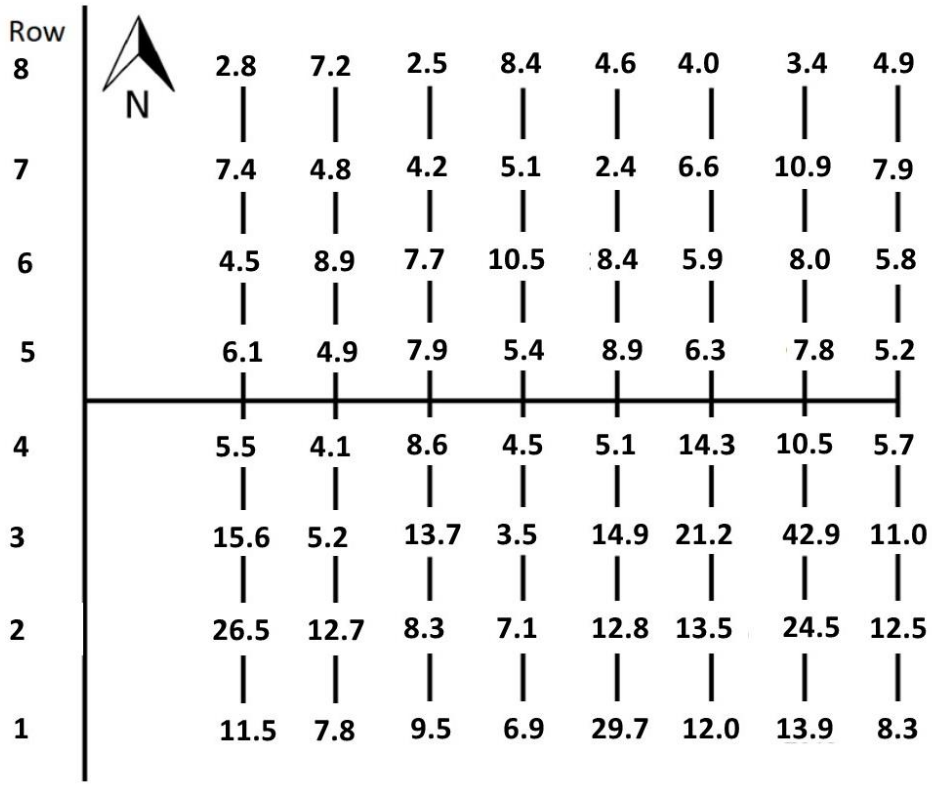

Figure 4.7: Cumulative nitrogen leached kg N/ha per suction cup pre sowing.

4.2. Total Trial Period

4.2.1. Climate

The results presented in Figure 4.8 show that the cumulative rainfall reached 434.7 mm while total cumulative drainage reached 263.2 mm over the trial period. The July period received the greatest rainfall reaching 196.6 mm within the single month (Figure 4.9).

Mean daily soil temperatures at 50 mm, 100 mm and 150 mm deep were 8°C, 8.7°C and 8.9°C respectively. Daily air temperature over the trial averaged 8°C (Figure 4.10). Mean daily air temperature fluctuation over the trial ranged from 1.8°C to 18.5°C whereas the mean soil temperatures at 50 mm and 150 mm ranged from 1.8°C to 14.5°C and 4°C to 14.8°C respectively as shown by Figure 4.10.

Mean daily soil moisture at 150mm, and 50mm deep were 0.29 and 0.34 Vw/Vt and ranged from 0.197 to 0.356 and 0.204 to 0.427 respectively as illustrated by Figure 4.11.

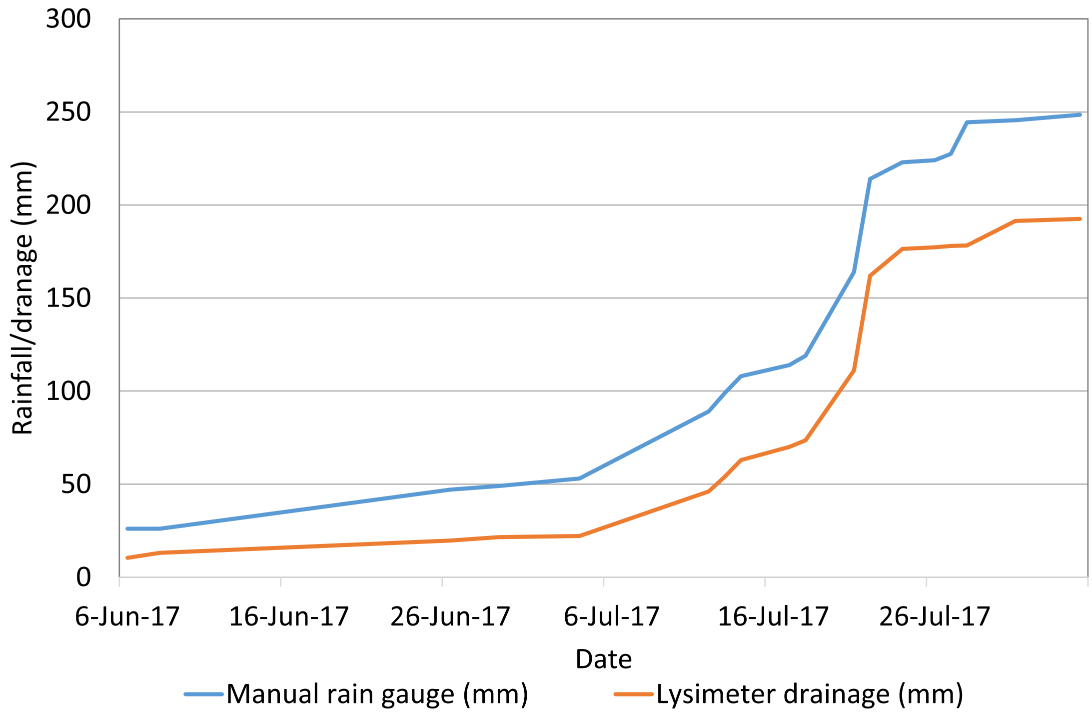

Figure 4.8: Cumulative average lysimeter drainage and rainfall inputs over trial period.

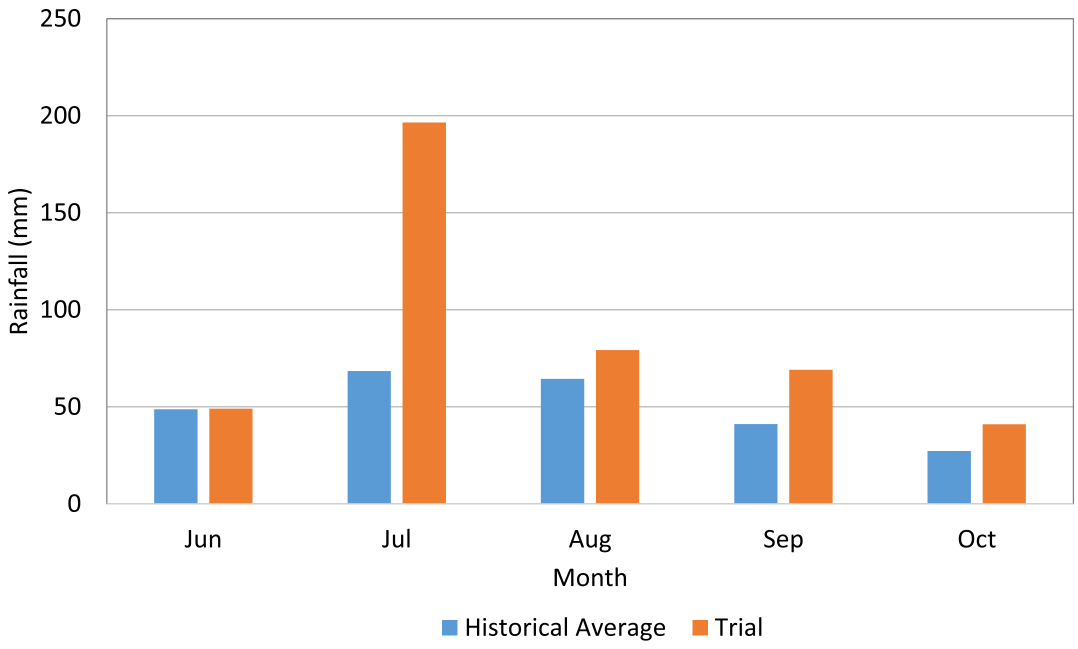

Figure 4.9: Rainfall inputs over trial period actual vs Christchurch historical average 1981-2010 (Adapted from NIWA, 2010).

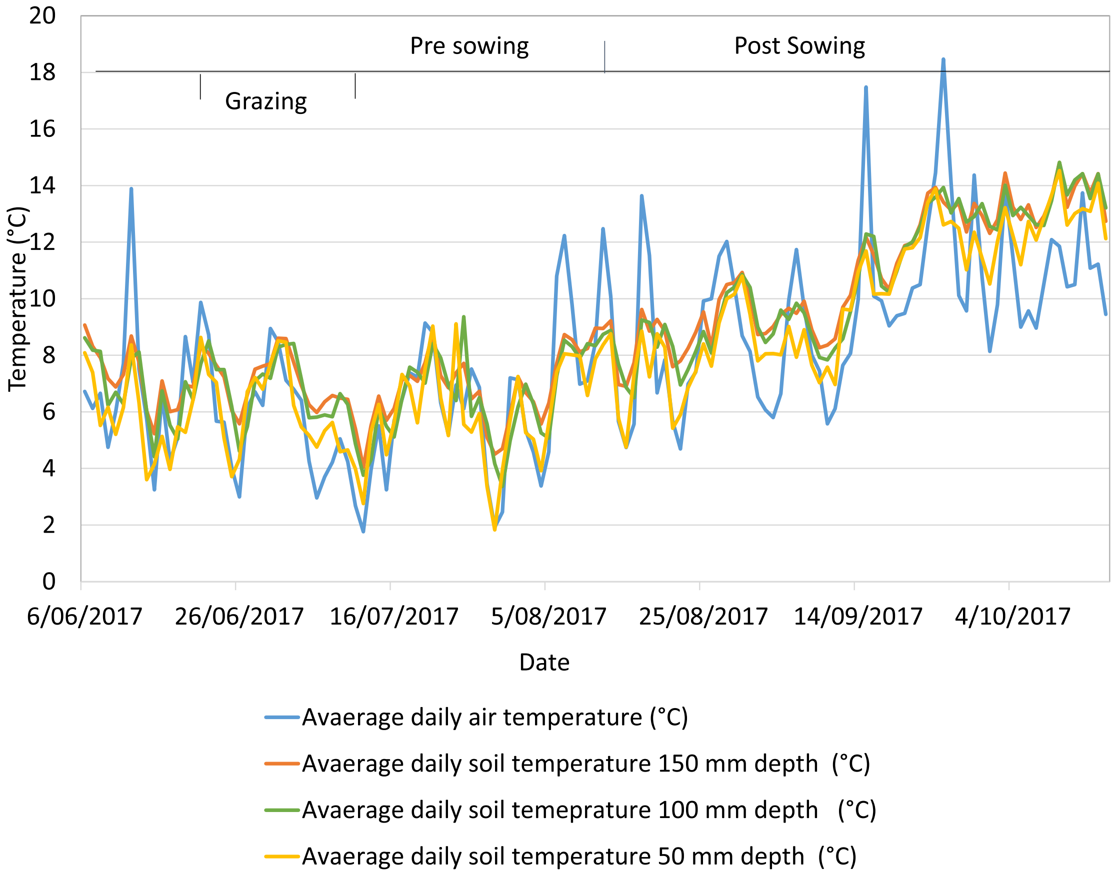

Figure 4.10: Daily mean temperatures at SCALAR site total trial period.

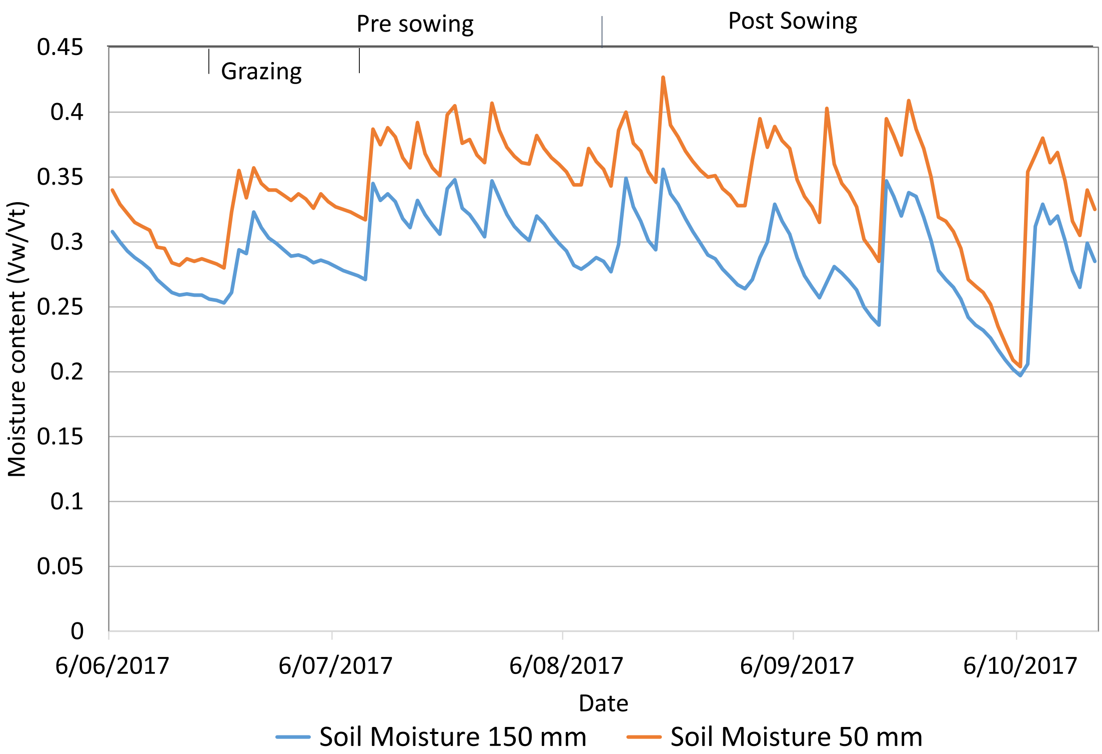

Figure 4.11: Mean daily soil moisture content total over trial period.

4.2.2. Nitrogen Leaching

The results showed that by the end of the trial period there was no significant difference ($P>0.05$) in the amount of total amount of mineral N leached between the catch crop and fallow treatments, nor was there a significant difference ($P>0.05$) in leaching of either NH_4^+ or NO_3^- individually over the treatments (Table 4.3). Total N leached reached a mean of 19 and 23 kg N/ha for the fallow treatment and catch crop treatments respectively (Figure 4.12). However there was large variation between suction cups as illistrated by Figure 4.13. Cumulative NO_3^- leached reached a mean of 22.17 kg N/ha for the catch crop treatment and 17.63 kg N/ha for the fallow treatment whereas cumulative NH_4^+ leached reached a mean of 0.83 and 1.43 kg N/ha for the catch crop and fallow treatments respectively over the trial period (Figure 4.14, and Figure 4.16). This meant that on average 96.4% and 92.6% of the N leached in the form of NO_3^- for the catch crop and fallow treatments respectively.

After sowing NO_3^- concentrations increased rapidly to a peak of 42 mg NO_3^--N/L for the catch crop treatment and 33.4 mg NO_3^--N/L for the fallow treatment as illustrated in Figure 4.18. Mean NH_4^+ concentrations increased to a peak of 0.74 and 0.52 mg NH_4^+-N/L for the fallow and catch crop treatments respectively. This peak occurred during the pre-sowing period of the trial however, over this period variation in NH_4^+ concentration between suction cups was large as shown by Figure 4.20. The concentrations of NH_4^+ began to decrease shortly after sowing of the catch crop, decreasing to a mean of 0.38 and 0.24 mg NH_4^+-N/L for the fallow and catch crop treatment respectively in the final samples (Figure 4.20).

Nitrate and NH_4^+ were also analysed ralatvive to drainage up to 260 mm. This showed that NH_4^+ concentration begain to decrease after 220 mm of drainage had occurred as illistrated by Figure 4.21, whereas NO_3^- concentration continued to increased to a peak of 19.3 and 32.89 mg NO_3^- N/L for the fallow and catch crop treatments respectively at 260mm of drainage (Figure 4.19). Cumulative NH_4^+ and NO_3^- leaching losses reached 1.47 and 16.95 kg N/ha for the fallow treatment and 0.83 and 20.86 kg N/ha for the catch crop treatment respectively by 260 mm drainage (Figure 4.15 and Figure 4.17).

Table 4.3: Summary of Mean Cumulative Nitrogen leached over trial period. All were insignificant (P>0.05).

	Fallow plots (kg N/ha)	Catch crop plots (kg N/ha)	P value
Total nitrogen	19	23	P=225
Nitrate	17.6	22.17	P=0.102
Ammonium	1.4	0.83	P=0.49

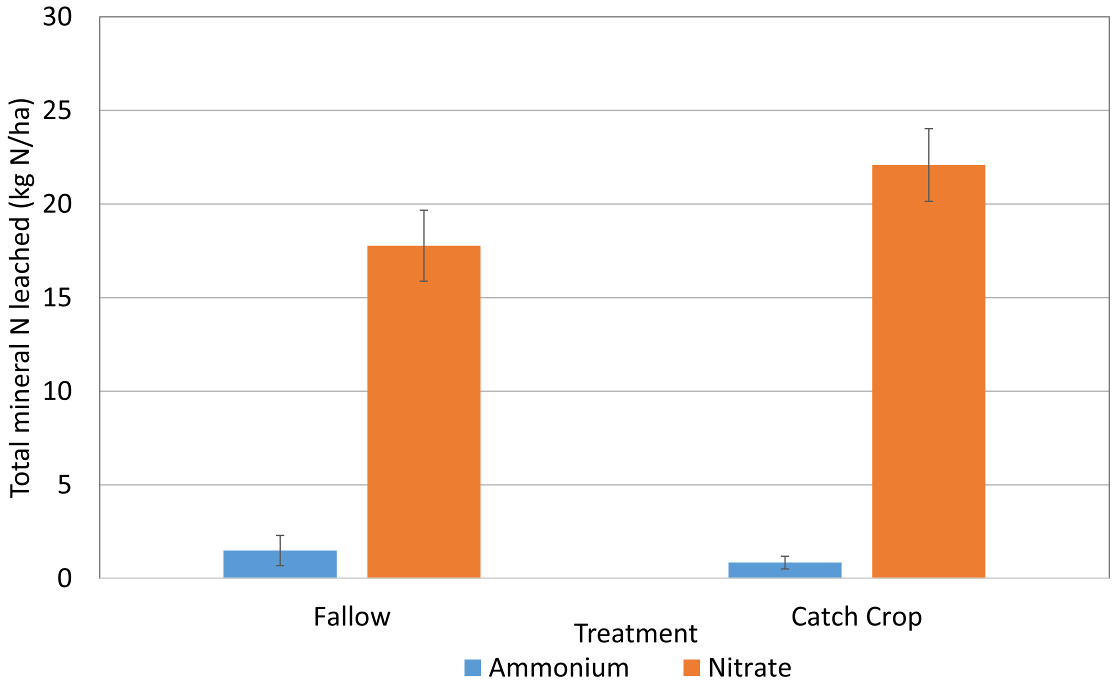

Figure 4.12: Total N leached fallow vs catch crop. Vertical bars represent standard error of the mean.

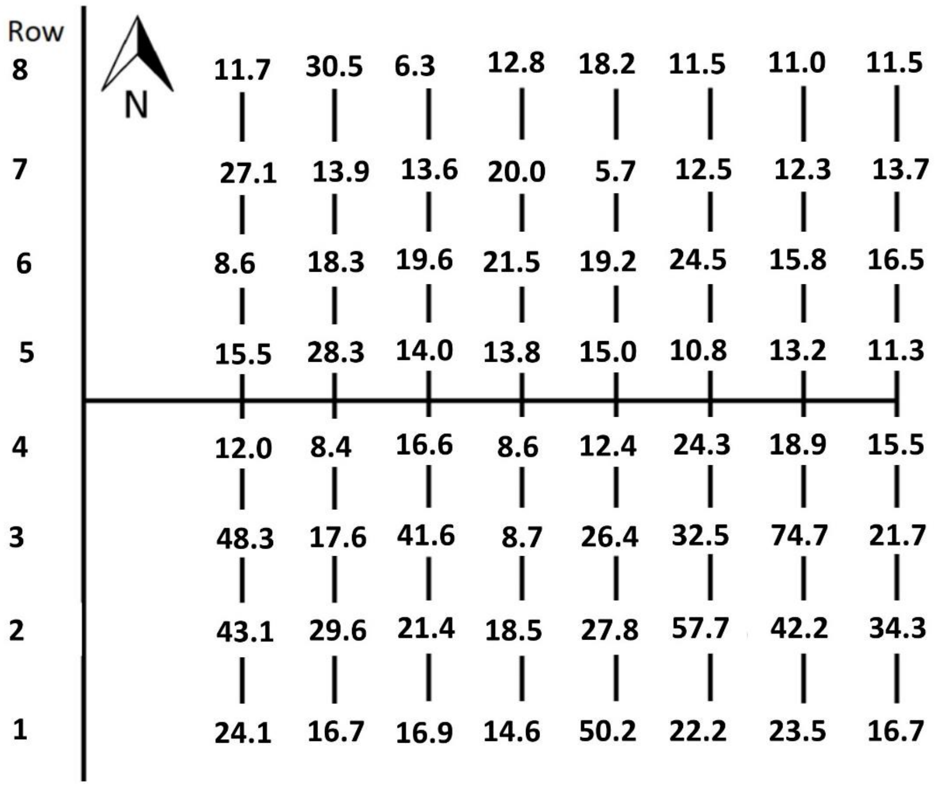

Figure 4.13: Cumulative nitrogen leached kg N/ha per suction cup.

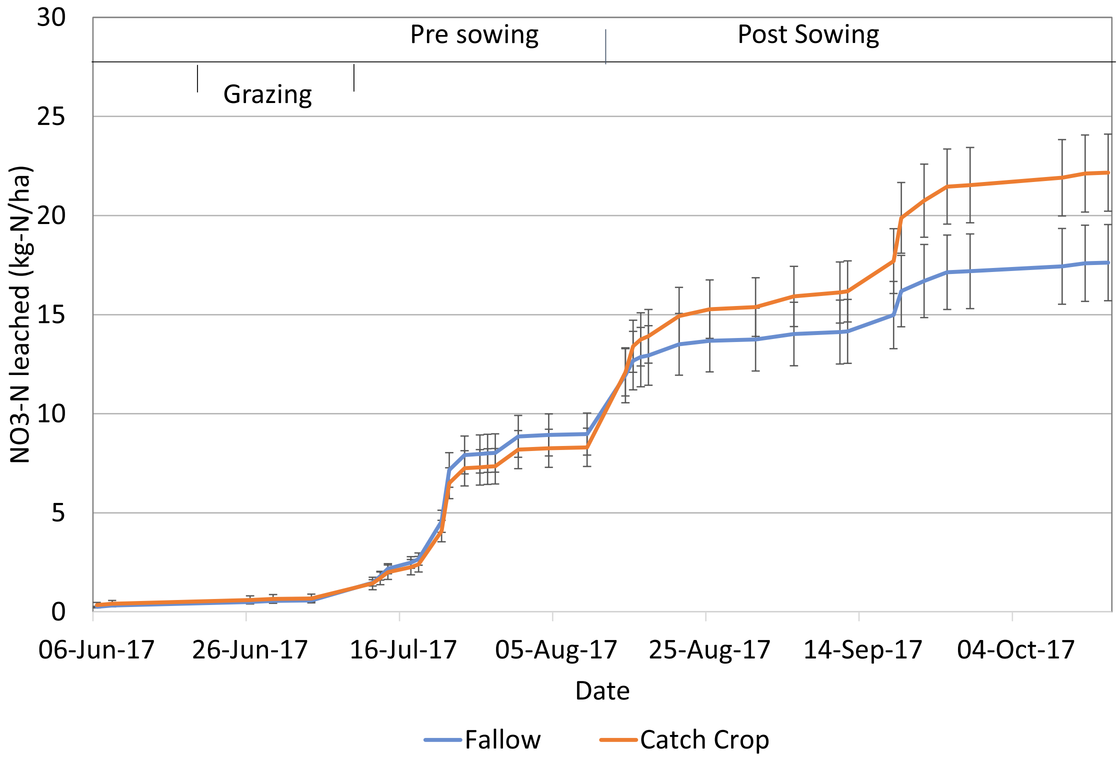

Figure 4.14: Cumulative nitrate leached fallow vs catch crop over trial period. Vertical bars represent standard error of the mean.

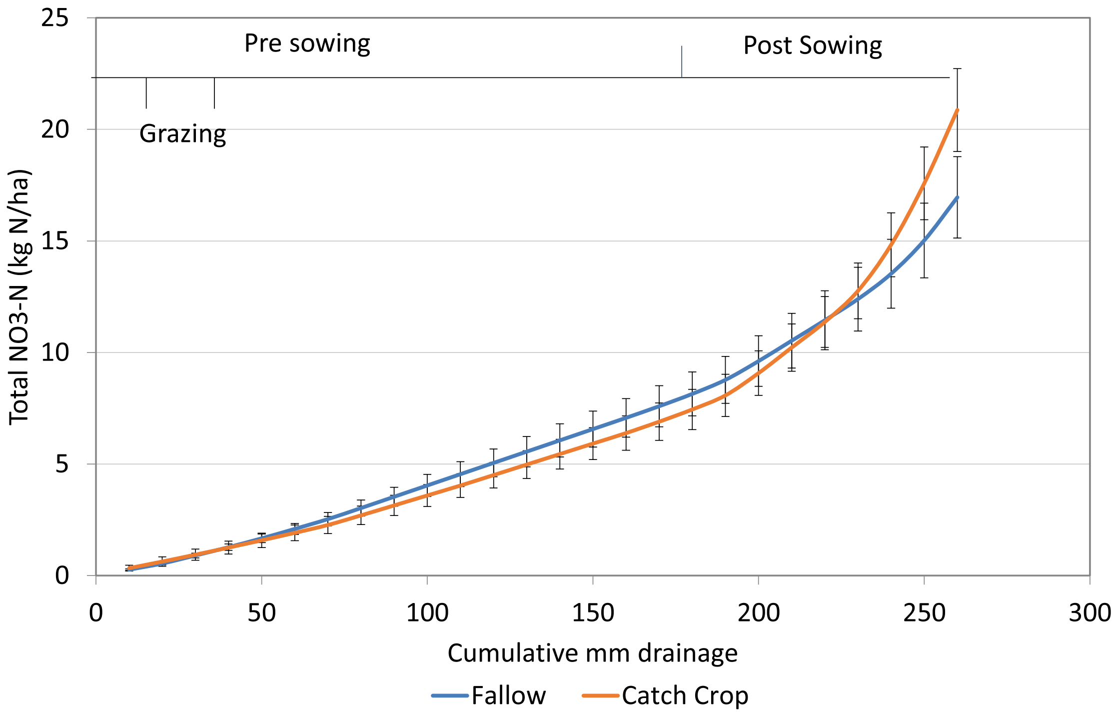

Figure 4.15: Cumulative nitrate leached fallow vs catch crop over trial drainage scale. Vertical bars represent standard error of the mean.

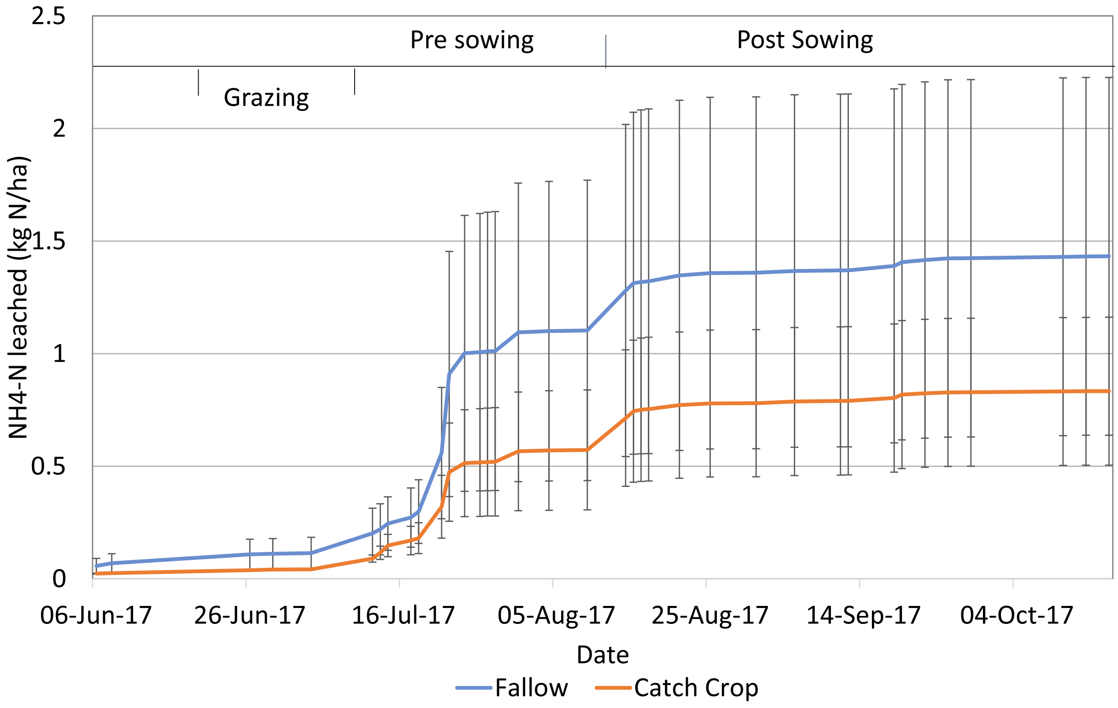

Figure 4.16: Cumulative Ammonium leached fallow vs catch crop over trial period. Vertical bars represent standard error of the mean.

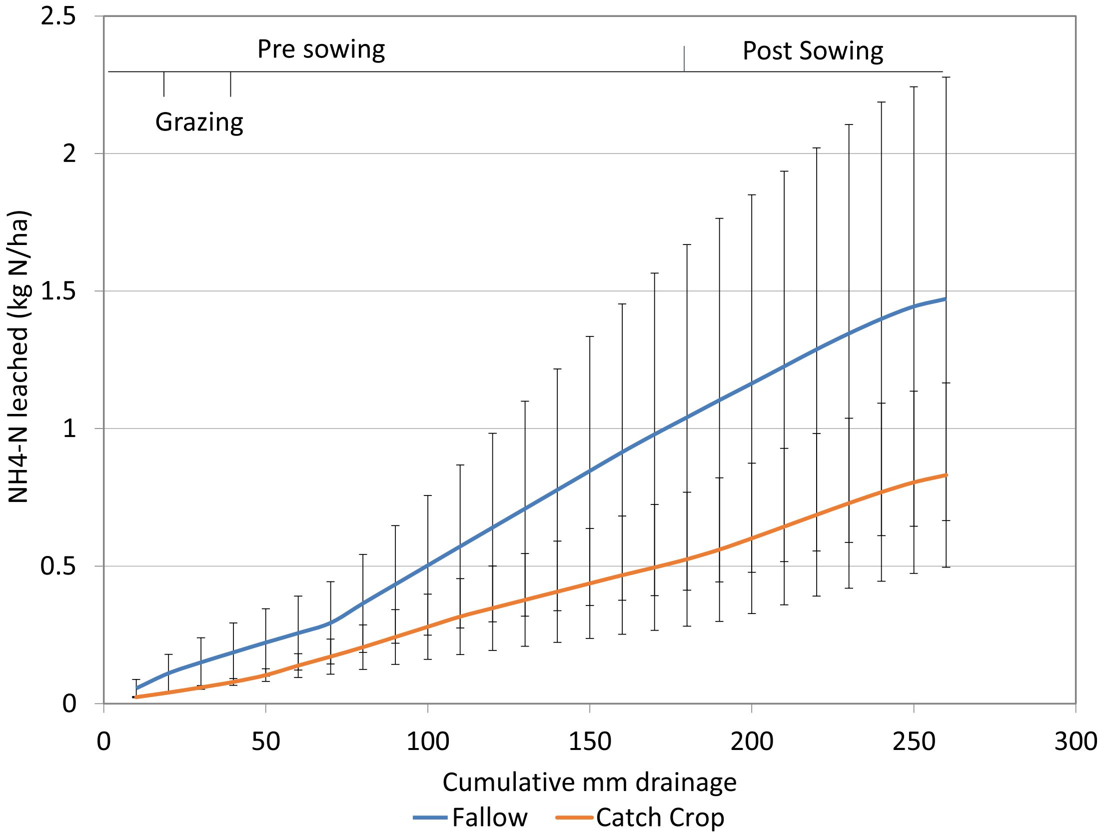

Figure 4.17: Cumulative Ammonium leached fallow vs catch crop over trial drainage scale. Vertical bars represent standard error of the mean.

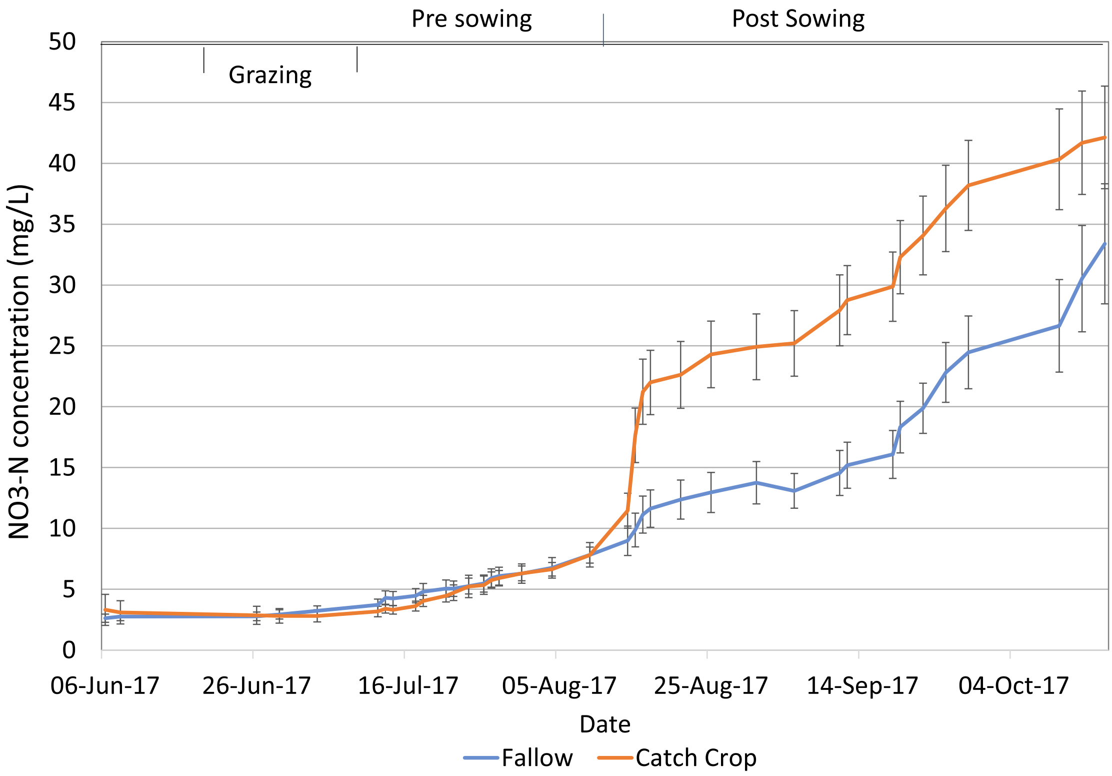

Figure 4.18: Concentration of nitrate in leachate fallow vs catch crop over trial period. Vertical bars represent standard error of the mean.

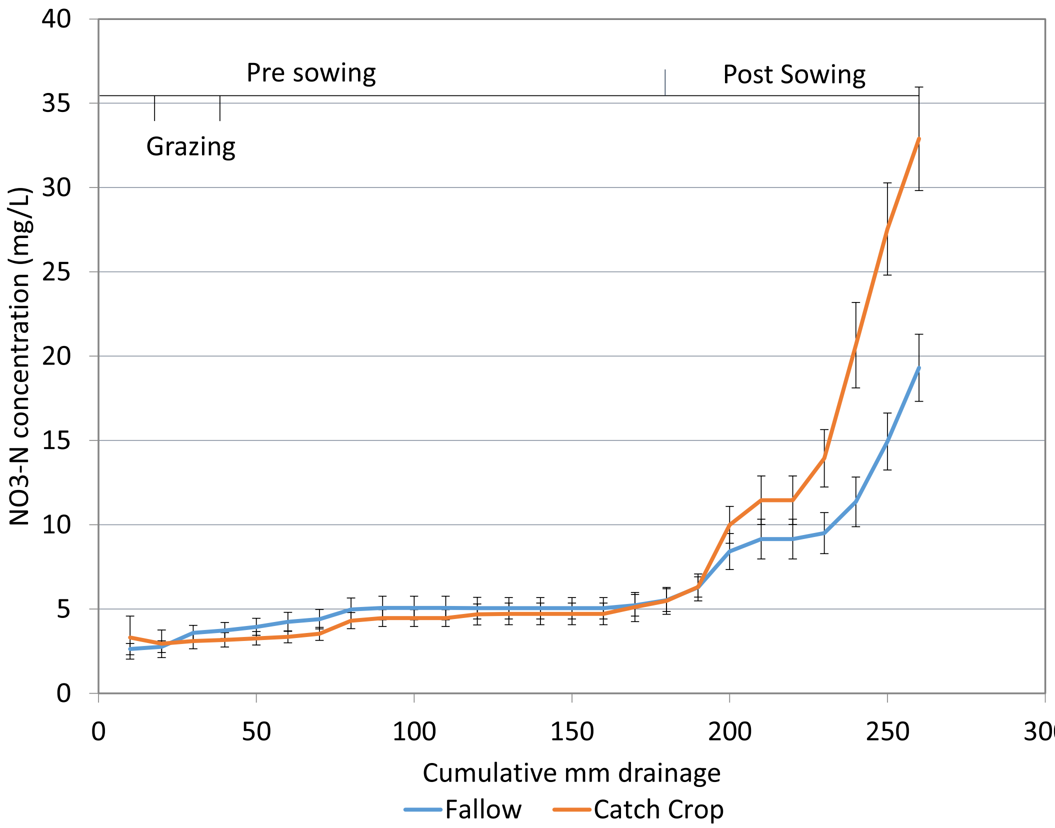

Figure 4.19: Concentration of nitrate in leachate fallow vs catch crop over trial drainage scale. Vertical bars represent standard error of the mean.

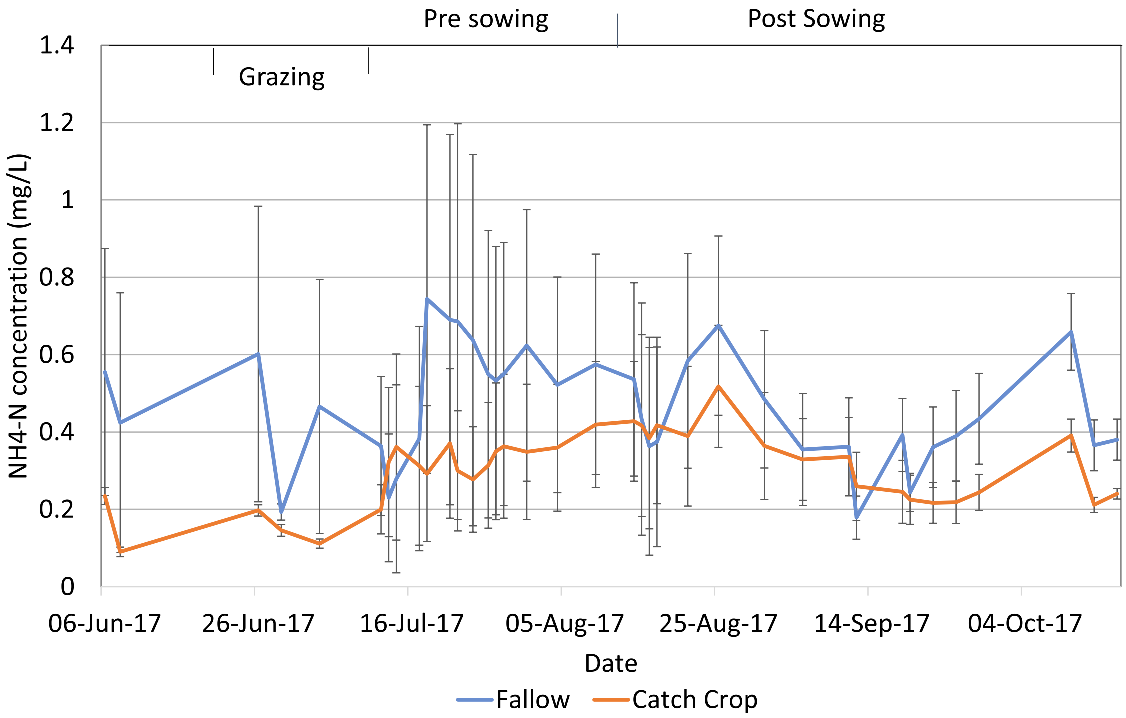

Figure 4.20: Concentration of ammonium in leachate fallow vs catch crop over trial period. Vertical bars represent standard error of the mean.

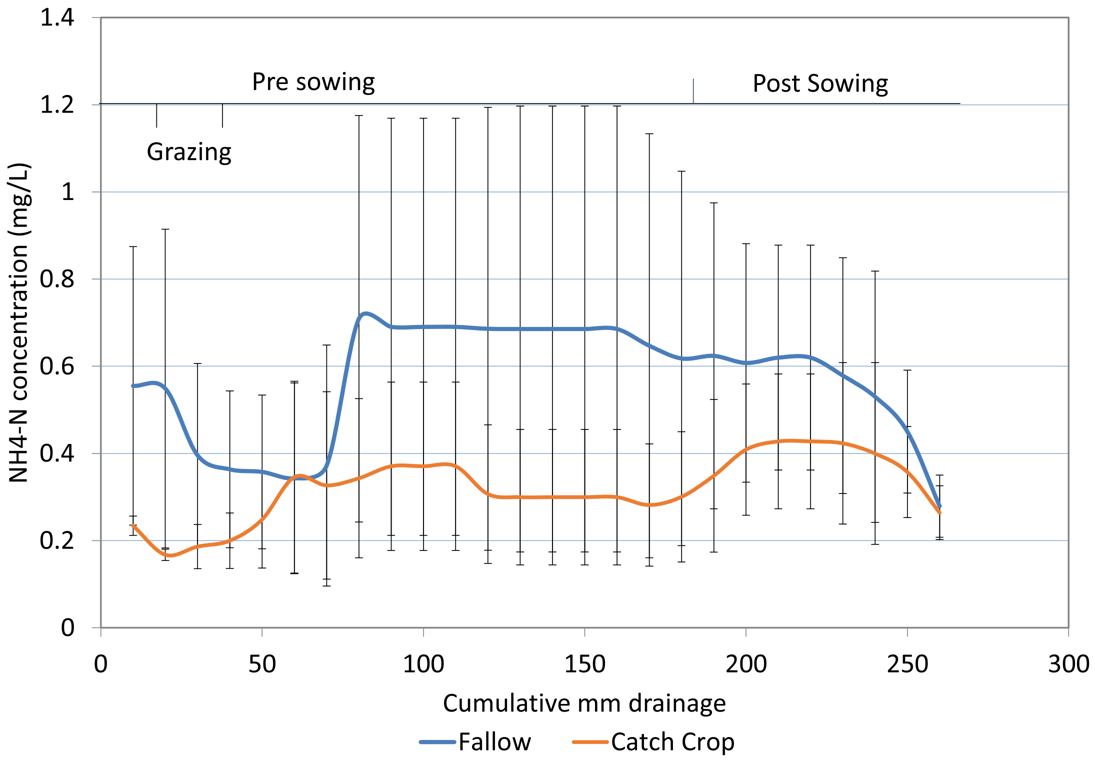

Figure 4.21: Concentration of ammonium in leachate fallow vs catch crop over trial drainage scale. Vertical bars represent standard error of the mean.

Chapter 5

Discussion

5.1. Pre-Sowing Nitrogen Leaching Loss

Results showed that during the pre-sowing period there was a significant difference ($p<0.001$) in the amount of N leaching loss between the grazing rows (Figure 4.6 and Table 4.2). This is likely due to the differences in timing of grazing causing N to be at differing points in the profile. However, Figure 4.7 shows that some suction cups leached much greater amounts of total N than the average. This is likely due to distribution of urine patches throughout the trial site. Additionally, statistical analysis confirmed that there was no significant difference between the oat and fallow treatment areas over the pre-sowing period ($P=0.49$) (Table 4.1).

An average of 9.43 kg N/ha was leached during the pre-sowing period of the trial (6/6/17 to 4/8/17). Of this N, 8.6 kg N/ha was leached as NO_3^- and 0.83 kg N/ha as NH_4^+ which meant that on average 91.2% of the N leached pre-sowing was in the form of NO_3^- as illustrated by Figure 4.6. The mean concentration of the leachate was 4.4 mg /L and 0.4 mg /L for NO_3^- and NH_4^+ respectively in the 192.5 mm of drainage that occurred during the pre-sowing phase (Figure 4.1: Figure 4.4: Figure 4.5). The results of Carey *et al*., (2016) had a much greater concentration of NO_3^- with a peak of 230 mg N/L for the early sown fallow treatment (1st June). Overall this meant that N leaching totalled 272 kg N/ha based on the annual drainage estimate. Of this N, NO_3^- comprised the majority (NH_4^+ was <3% of mineral N).

Rainfall was much greater than the regional average over the pre-sowing period, for example data from NIWA (2010) showed that 1981-2010 average rainfall in Christchurch for this period was 176.5 mm. Whilst 249 mm of rainfall was received over the pre-sowing period (Figure 4.1).

Denitrification over this period would have been substantial due to the anaerobic conditions caused by winter waterlogging which resulted from the high rainfall and low plant uptake. Pugging damage caused by the grazing of the fodder beet resulted in surface capping and subsequent surface water ponding thus further decreasing air movement into the soil profile and increasing the potential for denitrification. The volumetric water content averaged 0.34 and 0.3 at 50 mm and 150 mm depth respectively over the pre-sowing period (Figure 4.3). Due to these conditions the soil would have been prone to excessive denitrification, removing N from the soil system. For example, Carey *et al*., (2017) reported total denitrification loss under winter forage grazing, and as proportion of urinary-N applied at 350–700 kg N /ha, can probably be estimated at 30– 35%. This was supported by Fraser *et al*., (1994) who showed denitrification losses of 28%. Additionally, Selbie *et al*. (2015) found in an Irish grassland study N_2 losses as high as 56% from urine application. In the study by Selbie *et al*. (2015) soil conditions in terms of temperature (>9°C), water filled poor space, labile Carbon, and nitrate supply were probably near optimal for denitrification by late August/ early September thus enhancing gaseous N losses (Haynes and Williams 1993; Di *et al*. 2014). Additionally Ball *et al*., (2012) found that livestock trampling increased average cumulative N_2O emissions from 1.74 to 4.66% of urine-N applied. Given the similarity in conditions between Selbie *et al*. (2015) and this study as well as the tramping damage caused via intensive fodder beet grazing it is likely that denitrification would have been substantial over the pre-sowing period. The reduced N content due to denitrification is thought to be one of the key factors responsible for the low N leaching which occurred over the pre-sowing period of this trial. In contrast the Carey *et al*., (2016) received rainfall below that of the regional average therefore a lower proportion of the N would have denitrified resulting in a greater amount of N in the soil available to be leached. Additionally, due to Carey *et al*., (2016) being a lysimeter based study the urine N was applied directly thus removing the effect of inter-urinary areas.

Another factor which is believed to have contributed to this pre-sowing N leaching was the lack of plant uptake and evapotranspiration. The presence of plants is one of the crucial factors determining N leaching losses from agricultural systems. This is because without the presence of plants there is no opportunity for N to be utilised. Additionally, increased drainage occurs due to reduced water uptake from the soil as a result of decreased evapotranspiration. Therefore fallow areas such as the pre-sowing site are prone to N leaching. Overall lack of plant N uptake and reduced evapotranspiration should lead to increases in both soil N concentration and total drainage over the pre-sowing period. For example, Wyland *et al*., (1996) used two catch crop treatments, Phacelia and Merced rye (*Phacelia tanacetifolia* cv. 'Phaci', and *Secale cereale* cv. 'Merced'), with a fallow control and found a 65-70% reduction in NO3- leaching due to catch crops illustrating that leaching from fallow areas can be substantial. The results of Carey *et al*. (2016) reinforced this, showing fallow lysimeters leached an average of 34% more than areas of oats (*Avena sativa* L.) sown between 1 and 63 days after the urine deposition onto Kale (*Brassica oleracea* L.) in early winter. These studies illustrate the potential N leaching which can occur during fallow periods.

Temperature may have also had an effect on N leaching during the pre-sowing period of the trial. This is because as temperature increases so too will the rate of nitrification. This increased rate of nitrification is driven by the effect of temperature on the microbes which carry out nitrification. Malhi and McGill (1982) showed that the optimum temperature for nitrification in soils from central Alberta was 20°C. The SCALAR site mean daily soil temperatures at 50 mm, 100 mm and 150 mm deep were 5.8°C, 6.5°C and 6.8°C respectively with an average over all depths of 6.4°C (Figure 4.2). Daily air temperature over the pre-sowing period averaged 5.9°C as illustrated by Figure 4.2. Air temperature fluctuation over the pre-sowing period ranged from 1.8°C to 13.9°C whereas the soil temperatures at

50 mm and 150 mm ranged from 1.8°C to 9.1°C and 4°C to 8.8°C respectively. The soil temperatures recorded over the pre-sowing period were similar to that of the average winter 2013-2014 soil temperature recorded by Carey *et al*. (2017) of 5.9°C but greater than the long term winter average of 4.8 recorded between 2000 and 2016 (Carey et a., 2017). This may have resulted in a greater amount of nitrification than the long term average thus increasing the amount of nitrate available to be leached or denitrified from the soil over the pre-sowing period.

Macropore flow may also have contributed to the N leaching during the trial. Macropores allow preferential flow which enables a large amount of soil solution to travel through the macropores at high speed carrying N from the soil surface and within the macropores causing the rate of N leaching to be increased (Silva *et al*., 2000). Due to much of the N being deposited over the relatively short grazing period a large proportion would have been in areas of the profile which deemed it vulnerable to being transferred through the soil profile via macropore flow. The Balmoral stony silt loam has a large proportion of continuous macropores as a result of the high stone content and moderate structure. These factors suggest that N would have leached via macropores during the pre-sowing period of the trial. However, it appears that denitrification may have minimised this effect, as N was probably denitrified before having the opportunity to leach.

Overall the N leaching loss pre-sowing was much less than expected from previous reports in the literature. This has been attributed to the potential for large amounts of gaseous N being lost due to denitrification which reduced the N available in the soil profile. This result suggests that N leaching losses following grazing of winter catch can be minimal in wet years when favourable conditions for denitrification are present. The results of Carey *et al*., (2016) support this hypothesis because the drier conditions which minimise denitrification meant that more N was available in the soil for leaching thus N leaching from the early established fallow treatments reached an average of 272 kg N/ha based on the annual drainage estimate.

5.2. Total Trial Period

The results of this trial found no significant (P=0.23) difference between in total N leaching loss treatments of a catch crop of oats compared to fallow after winter grazing of fodder beet. Additionally, there was no significant difference (P>0.05) in leaching of either NH_4^+ or NO_3^- individually between the catch crop and fallow treatments (Table 4.3). An average of 19 and 23 kg N/ha was leached during the trial period (6/06/2017 to 16/10/2017) for the fallow and catch crop treatments respectively. Of this N, 17.6 kg N/ha was leached as NO_3^- and 1.4 kg N/ha as NH_4^+ for the fallow treatment and 22.17 kg NO_3^--N/ha and 0.83 kg NH_4^+-N/ha for the catch crop treatment (Figure 4.12). Thus on average 92.6% and 96.4% of the N leached in the form of NO_3^- for the fallow and catch crop treatments respectively. Leachate concentrations of NO_3^- and NH_4^+ are illustrated in Figure 4.18 and Figure 4.20. The results of this study contradicted the findings of Carey *et al*. (2016) which concluded that use of a catch crop sown between 1 and 63 days after urine deposition in early winter was able to reduce NO_3- leaching losses by approximately 34% on average (range: 19 – 49%) compared to fallow over the winter-spring period. Differences in rainfall over the trial periods are thought to be the main factor causing the differing results in these studies. Rainfall during this trial affected both N movement and transformation in the soil profile, as well as affecting the management of the site.

Rainfall over the trial period reached 434.7 mm and this led to total drainage of 263 mm as illustrated in Figure 4.8. This rainfall was greater than the regional average for this period of 244.5 mm as calculated by NIWA (2010) and far greater than the water input of Carey *et al*., (2016) as illustrated by Figure 4.9 and Figure 5.1. This above average rainfall and subsequent drainage is believed to have caused the N from the grazing of the winter fodder beet to be leached bellow the root zone before the catch crop of oats was able to utilise it. Due to this the catch crop did not have access to the N for uptake from the soil system. It is thought that the lower rainfall during the trial of Carey *et al*. (2016) meant that much of the N remained in the root zone available to the catch crop for uptake thus enabling N leaching to be reduced, hence the differing results of the studies.

The above average rainfall which occurred over the trial period also led to management issues which affected the ability of the catch crop to remove N from the soil. The main issue was that after grazing, the site was unsuitable for sowing the oat crop. This meant that the period between the termination of grazing and sowing of the catch crop was prolonged to enable the soil to dry to an adequate moisture content for sowing. The soil took 29 days to dry adequately over which time 195.5 mm of rainfall occurred washing the soil N further down the profile subsequently reducing the quantity available for plant uptake. Because the study of Carey *et al*. (2016) used lysimeters and had lower precipitation this did not affect the sowing of the catch crop and enabled the crop to be sown as desired between 1 and 63 days after urine deposition. This management issue affects the practicality of the use of winter catch crops at farm scale as it highlights a major limitation.

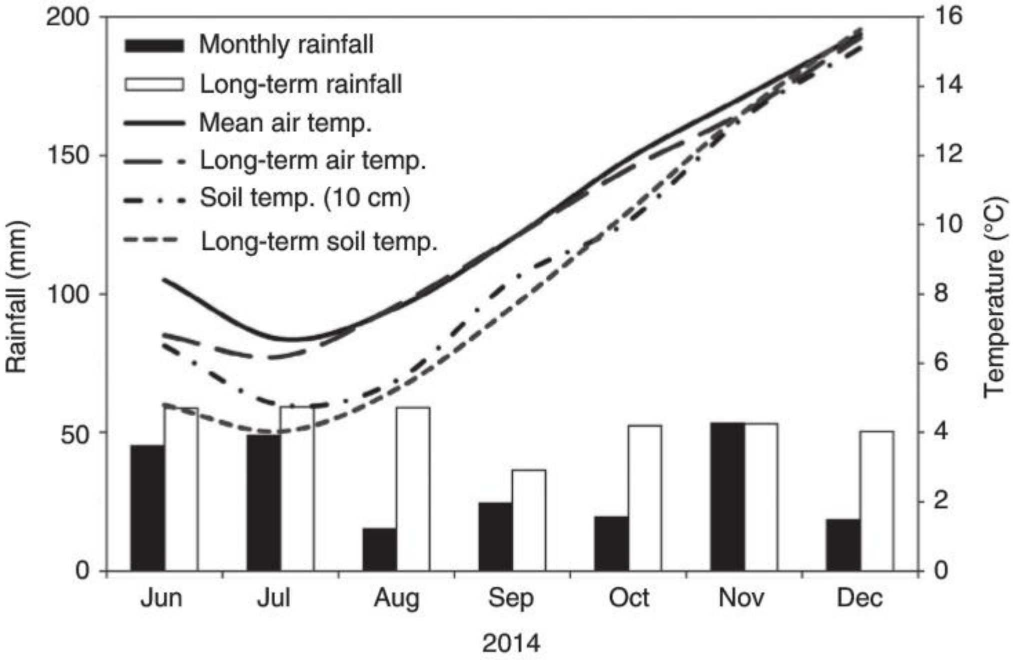

Figure 5.1: Mean Lincoln monthly rainfall, air and soil (10 cm) temperature for the 2014 winter-spring period compared with long-term means (1980-2014) (Adapted from Carey *et al.*, 2016).

Total N leaching losses of 19 and 23 kg N/ha for fallow and oats treatments respectively was much lower than expected compared to recent New Zealand studies of winter forage grazing that have reported nitrate leaching losses ranging from 52 to 173 kg N/ha (Shepherd *et al.*, 2012; Smith *et al.*, 2012; Monaghan *et al.*, 2013; Malcolm *et al.*, 2015). The low N leached during this study is believed to have been caused by large amounts of denitrification and subsequent gaseous N losses (Selbie *et al.*, 2015). One reason for this was the anaerobic soil conditions. These conditions resulted from the above average rainfall and lack of plant uptake over much of the trial period which in turn led to large periods of the trial having saturated soil conditions which facilitated denitrification, as illustrated by Figure 4.11.

Another aspect which is believed to have contributed to the denitrification over the trial period is trampling damage caused by the intensive grazing of the fodder beet as illustrated by Plate 5.1. Trampling has shown to decrease the air permeability of the soil, suppress plant growth and alter some of the denitrifying microbial communities thus making the soil more anaerobic and overall increasing denitrification (Ball *et al.*, 2012; Treweek *et al.*, 2016). For example, Ball *et al.* (2012) found trampling increased average cumulative N_2O emissions from 1.74 to 4.66% of urine-N applied. Treweek *et al.* (2016) supported this with results which showed mean N_2O emissions from treatments which had been trampled and received 400 kg N/ha was 6.24 kg N_2O-N/ha whereas those treated with 400 kg N/ha which were not trampled emitted only a mean of 0.8 kg N_2O-N/ha (Figure 5.2).

Plate 5.1: Trampling damage over SCALAR site (24/6/2017; Lincoln University Ashley Dean Research and Development Station SCALAR site).

The lower N leaching loss measured in this study suggests that during wet years the N leaching loss from winter forage grazing may be low as a result of increased losses of gaseous N resulting from denitrification. The study of Carey *et al*. (2016) resulted in far greater total N leaching over the trial period reporting N leaching losses for the annual drainage estimate 272, 224 and 172 kg N/ha for early, mid and late fallow treatments, respectively. This greater N leaching is likely because the lower rainfall over the trial period of Carey *et al*. (2016) did not cause anaerobic conditions and saturation of the soil profile, thus did not create suitable conditions which favoured denitrification (Figure 5.1). Therefore this study and Carey *et al*. (2016) show the contrasting effects of high and low rainfall conditions on total N leaching from winter forage grazing.

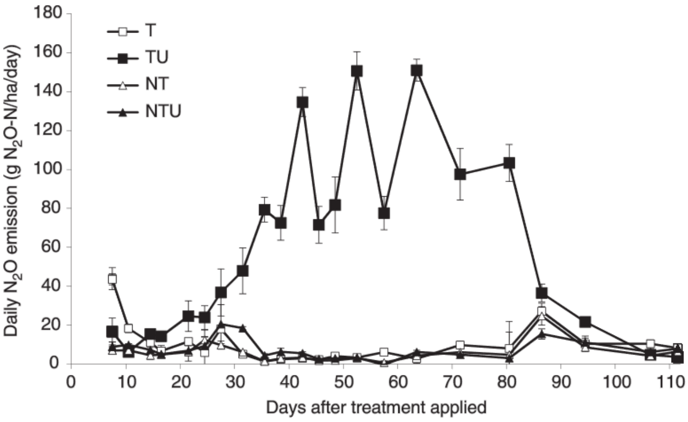

Figure 5.2: Daily N_2O emissions flux from all treatment, T = trampling no urine, TU = trampling with 400 kg N/ha, NT = no trampling, NTU = no tramping with 400 kg N/ha (N applied as urine) (Adapted from Treweek *et al*., 2016).

The concentrations of NO_3^- and NH_4^+ in leachate fluctuated throughout the trial period. Although variable between suction cups the average NH_4^+ concentration increased slightly over the pre-sowing period while NO_3^- leachate concentration remained low (Figure 4.19 and Figure 4.21). NH_4^+ concentrations increased as a result of N deposition from fodder beet grazing and reduced transformation of NH_4^+ into NO_3^- by nitrification due to anaerobic conditions present during this period (Figure 4.11). These conditions also caused the potential for a large proportion of the NO_3^- that was produced to denitrify, removing NO_3^- from the soil system. Post-sowing, the mean concentration of NO_3^- in leachate increased while mean NH_4^+ concentration decreased as illustrated by Figure 4.18 and Figure 4.20. This is because of the aerated condition and increased soil temperature which occurred in the top soil as a result of cultivation is thought to have increased nitrification and decreased denitrification in the upper profile of the soil therefore reducing NH_4^+ content and increasing the NO_3^- content of the soil (McLaren and Cameron, 1996; Di *et al*., 2010).

The high rainfall over this winter period, as show in Figure 4.9, is thought to have rapidly leached the NO_3^- downwards through the soil before plant uptake could occur. Although some of the NO_3^- was leached from the root zone, much of the NO_3^- is thought to have been denitrified due to the anaerobic conditions of the soil. Thus concentrations of NO_3^- in leachate of the catch crop treatment only reached a peak of 42 mg NO_3^--N/L by the end of the trial period as illustrated by Figure 4.18 which is much lower than that of previous studies. For example, a lysimeter study by McDowell & Houlbrooke (2009) resulted in peak leachate NO_3^- concentration of 250 mg NO_3^--N/L when cow urine was applied at a rate of 580 kg N/ha on a Timaru silt loam soil. The study of Malcolm *et al*. (2015b) also reported a greater peak NO_3^- concentration of 236 mg NO_3^--N/L from lysimeters containing Balmoral stony silt loam treated with 700 kg N/ha applied as cow urine. The low rainfall study by Carey *et al*. (2016) also resulted in much greater NO_3^- concentration in leachate as illustrated by Figure 5.3. The contrasting NO_3^- leachate concentrations of these studies supports the hypothesis that denitrification in the soil profile reduced the NO_3^- concentration of leachate over the post sowing period of this trial.

Plate 5.2: Comparison of treatments early spring (7/11/2017; Lincoln University Ashley Dene Research and Development Station SCALAR site).

Despite having no significant effect on N leaching from the winter grazed fodder beet, the catch crop would provide an extra source of feed to the farm system which would have otherwise not existed (Plate 5.2).

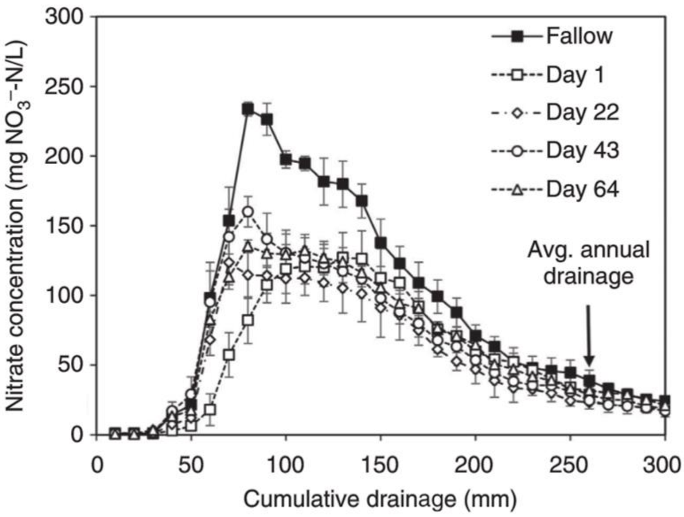

Figure 5.3: Nitrate leaching breakthrough curves (and standard errors) for fallow and catch crop treatments for early urine application (equivalent to 350 kg N/ha). Crops were sown approximately 1, 22, 43 or 64 days after the urine application. Standard error bars are shown. Annual average drainage shown for an irrigated Balmoral soil in the district (Adapted from Carey *et al.*, 2016).

Chapter 6

Conclusions and Suggestions for Future Research

The main conclusions and suggestions for future research are:

- Use of a catch crop of oats had no significant effect on N leaching losses from winter grazed fodder beet in this trial. This result is thought to be due to removal of N from the root zone prior to establishment of the oats because of the above average rainfall which occurred over the trial period.

- Total N leaching losses from soil under winter grazed fodder beet can be low in wet seasons. This has been attributed to large quantities of gaseous N loss resulting from denitrification due to the wet anaerobic conditions and soil compaction due to trampling damage.

- Despite not affecting N leaching losses, the oat crop provided extra feed for the farming system.

- Further research is still required in this area to better quantify the effectiveness of catch crops for reducing N leaching from winter grazed forage systems as studies thus far have encountered climatic extremes.

Appendix A

Randomised Block Design

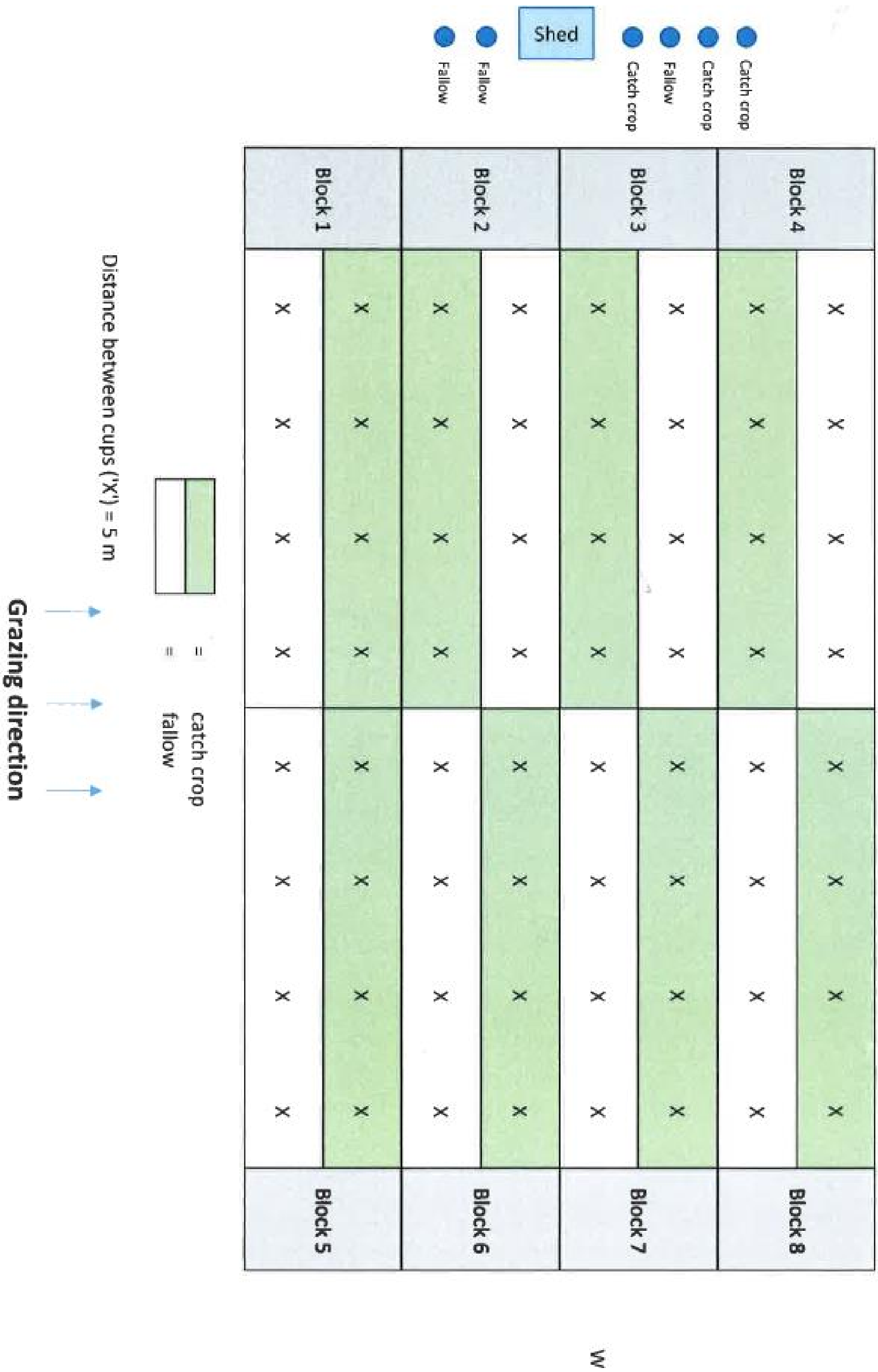

Figure A 1: Randomised block design of the field trial showing placement of ceramic cups, treatment replicates and grazing direction.

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