DURATION-CONTROLLED GRAZING OF DAIRY COWS: IMPACTS ON PASTURE PRODUCTION AND LOSSES OF NUTRIENTS AND FAECAL MICROBES TO WATER

A thesis presented in partial fulfilment

of the requirements for the degree of

Doctor of Philosophy

in

Soil Science

at Massey University, Palmerston North, New Zealand

MASSEY UNIVERSITY
TE KUNENGA KI PŪREHUROA
UNIVERSITY OF NEW ZEALAND

Christine Lynne Christensen

2013

Abstract

Mitigation strategies for improved environmental sustainability of the New Zealand dairy industry need to focus on reducing the transport of nitrogen (N) from urine patches and phosphorus (P) and faecal microbes from dung patches to waterways. One strategy is Duration-controlled grazing (*DC* grazing), a system based upon shorter grazing periods on pasture (4 hours) and removing cows to a stand-off facility for rumination and excretion. The stored effluent is applied to pasture as a slurry at an appropriate time when nutrients are required and soil conditions are suitable.

A three year field study was established in the Manawatu to compare key features of *DC* grazing with a standard grazed (*SG*) system. This thesis explores the impact of a *DC* grazing system on the losses of N, P, potassium (K) and faecal microbes to water through drainage and surface runoff. It also investigates the effects of such a system on pasture production and intakes of pasture by cows.

Pasture accumulation was the same for both treatments in the first year, but there was a 20% and 9% decline on the *DC* treatment in the subsequent two years. This was due to the way that slurry applications were managed. A large amount of slurry (212 kg N/ha) was applied in the first year, and no slurry was applied in the second year. In the third year slurry was applied four times at a total rate of 115 kg N/ha. The study indicates more frequent application of all nutrients captured in the effluent from standing cows off is required to maintain pasture production.

Compared to the *SG* plots, the reductions in N losses from *DC* grazed plots were large, with an average 52% reduction in NO_3^- and 42% reduction in total N leached. Reducing urine deposition during autumn grazings appeared to have the largest impact on reducing NO_3^- leaching. Runoff losses of N were small and similar between treatments. The losses of P were small through both surface runoff and drainage. There was a large variation in runoff volume, which resulted in highly variable P runoff loads across plots and between treatments. The average 32% reduction in total P load from *DC* grazed plots was not significantly different from *SG* plots. Useful predictors of P load lost from all plots were runoff depth and the time cows spent grazing. Faecal microbe losses were also similar between treatments, with the useful predictors of faecal microbe concentration across all plots being the number of days since grazing and the climate after grazing.

The amount of K applied in slurry and urine had a large influence on both soil and herbage K. It was determined that in a *DC* grazing situation, the K-rich liquid component must be included in the applied slurry to maintain soil K levels.

The OVERSEER® nutrient budgeting software was able to simulate nutrient cycling in the *DC* grazing system reasonably well. The total N loss from the system was predicted accurately, although the relative proportion of N in drainage and runoff was not.

Several opportunities for further work arise from this research. While *DC* grazing is a tool that could be implemented to significantly reduce N leaching losses, the management of collected excreta needs to be further developed to ensure pasture production gains are realised, or at least maintained. The combined effects of reducing treading damage and *DC* grazing should be investigated. Finally, a comprehensive economic analysis of standing cows off should be undertaken.

Acknowledgements

Completing a PhD thesis is the destination reached after a journey filled with discovery, emotional turmoil, the brightest of highs, and some very dark lows. Throughout the journey, I was extremely grateful to have several people by my side.

Firstly, to my supervisors. Professor Mike Hedley, my chief supervisor and the man who ignited my interest in nutrient cycling in soils, back when I was an undergraduate Ag student. I thank you for sharing your bottomless pit of knowledge, your critical analysis of my writing, and your passion for the wider FLRC group. Most of all, I thank you for your truly continuous inspiration – I am pursuing a career in Agricultural Science because of people like you.

Dr James Hanly, you have been a constant help. Your practical knowledge and assistance with field work, lab work, and writing has been much appreciated. I thank you for putting up with me constantly knocking at your door with ideas and ‘interesting facts’ about my research. I hope we will continue to enjoy a friendly working alliance as FLRC colleagues.

Associate Professor David Horne, your willingness to help me is valued. Your expertise and patience in anything to do with soil water has certainly helped me over the past few years, and your impeccable knowledge and use of the English language is enviable.

If it wasn’t for the FLRC (Fertilizer & Lime Research Centre) technical team, then my large scale study could not have been completed. Bob Toes, your endless hours in the field have not gone unnoticed, thank you so much. Glenys Wallace, the time you have spent in the lab to help get thousands of samples analysed has been remarkable – thank you for your dedication, and also for your help with editing. Also Ross Wallace, Anja Moebis, Ian Furkert and Mike Bretherton, your assistance in the field and lab has been most appreciated, and Liza Haarhoff and Lance Currie, your help at an administrative and business management level is valued. I also appreciate the help I received from Zhao He and Mark Bebbington when carrying out statistical analysis. Thanks to the No 4 dairy farm staff for your cooperation in carrying out the research trial at the farm. Thank you to all the FLRC team and wider IAE (Institute of Agriculture & Environment) group for your continued friendship and camaraderie – long may it continue.

I am grateful to FLRC and IAE for funding my PhD studies, and also for allowing me the time to complete my studies. The trial was funded through the Pastoral 21 (Environment) programme (C10X0603), an important programme in providing real and usable answers for our agricultural industry into the future.

To Mum and Dad, Blair and Donna, Graham and Sue, and my extended family, thanks to you for instilling in me the values that were required to begin and complete this journey. You have always been very supportive of my decisions, which I appreciate. Also a special thanks to my loyal friends and colleagues who have provided support and advice over the duration of the journey - you know who you are.

Finally, but importantly, I would like to thank my husband James. You have been my rock of encouragement while completing this thesis and our partnership grows stronger every day. I thank you for your listening ear, your constructive comments, and your support in helping me to achieve my personal goals, while we continue to strive to reach our joint aspirations.

This destination has now been reached, but just around the corner are more journeys, to provide me with more challenge, fulfilment, and friendship, just like this one has.

Table of Contents

List of Figures

List of Tables

List of Plates

Chapter 1 Introduction

The New Zealand dairy industry produces 15 billion litres of milk annually (Fonterra, 2011). The majority of this produce is exported, providing over 20% of this country's export earnings (Ministry of Agriculture and Forestry, 2011). New Zealand is perceived by our overseas markets as being 'clean and green'. However, the freshwater river, stream and lake systems throughout the country are declining in quality (Parliamentary Commissioner for the Environment, 2012). A large proportion of this decline has been attributed to losses of nitrogen (N) and phosphorus (P) in drainage and surface runoff from intensively grazed dairy systems. The losses of N and P are predominantly from urine and dung patches, respectively (Haynes and Williams, 1993). Therefore, these losses tend to increase with higher stocking rates which have increased over the last 20 years (Livestock Improvement Corporation and DairyNZ, 2011), and are often sustained through increased fertiliser applications, and/or brought-in supplements.

Nitrogen is mostly lost from a grazed system through subsurface drainage, leading to contamination of freshwater systems. Furthermore, mole and pipe drainage of fine-textured soils to support higher cow numbers per hectare can accelerate the rate of contamination, as the mole and pipe system creates pathways of direct loss (Sharpley and Syers, 1979). Much of the N loss is derived from the urine patches deposited by cows when grazing (Sharpley and Syers, 1979; Ledgard and Menneer, 2005). The high concentrations of N deposited in urine patches are unlikely to be recovered by pasture uptake of N from the urine-affected soil, and therefore the excess N is lost from the soil profile when soil drainage occurs. Most of the N reported to be lost in drainage is in the form of nitrate (NO_3^-), a loosely bound anion that is largely found in soil solution (Haynes and Williams, 1993).

Conversely, P is typically lost through surface runoff, when rainfall intensity exceeds that of soil infiltration rates and overland flow occurs. The high concentration of P in dung pats (up to 25 g P/ 2 kg fresh dung pat (Haynes and Williams, 1993)) increases the concentration of P in the surface runoff (McDowell *et al.*, 2006), ultimately leading to P losses to freshwater rivers and streams. In addition, the loss of faecal microbial organisms is also accelerated with surface runoff transporting dung material (Muirhead *et al.*, 2005).

There are several farm management strategies that have been proposed to reduce N, P and faecal microbe loss to water from New Zealand's intensively grazed dairy pastures. They include lowering the stocking rate, decreasing N fertiliser applications and hence lowering the amount of N cycling in the farm system, applying chemicals such as nitrification inhibitors (Di and Cameron, 2005) and improving storage and application of farm dairy effluent (Houlbrooke, 2005).

Another strategy is duration-controlled grazing, which involves allowing cows to be at pasture for grazing only (not resting or ruminating), then removing them to a stand-off area. Here, the excreta is collected and stored for uniform application to pastures at a later date. This strategy reduces the number of nutrient-rich dung and urine patches and therefore the major sources of N and P loss. If the cows deposit fewer nutrient-rich urine and dung patches in the paddock, the amount of N and P lost in drainage and surface runoff water should decrease. Researchers in Ireland (Kennedy *et al.*, 2009) and in Denmark (Oudshoorn *et al.*, 2008), have demonstrated the potential reduction in N leaching using this technique termed 'restricted access time' or 'time-limited grazing', respectively.

In New Zealand, autumn studies (de Klein *et al.*, 2006) and modelling (de Klein, 2001; de Klein and Ledgard, 2001; de Klein *et al.*, 2006; Monaghan *et al.*, 2008) have also suggested that duration-controlled grazing can markedly reduce NO_3^- leaching from dairy pastures. The use of restricted grazing in the autumn was predicted to decrease NO_3^- -N leaching by 35-50% (de Klein and Ledgard, 2001) and by 41% when implemented in autumn and winter (de Klein *et al.*, 2006) in southern New Zealand on fine-textured, mole and pipe drained soils. The effects of year-round duration-controlled grazing, however, are not well known and thus there is an opportunity to study this in an intensive grazing system.

The successful application of duration-controlled grazing to reduce N and P losses will require the uniform return of effluent from the standoff facilities at optimum times and conditions.

The research objectives of this study are described fully in Figure 1.1. To meet these research goals, a large scale grazing trial was established in the Manawatu region of New Zealand, on a mole and pipe drained pallic soil (Tokomaru silt loam). Fourteen grazing plots, each with isolated drainage and surface runoff systems, were installed. Duration-controlled grazing was operated as one treatment (on seven plots) and compared to a standard grazing system as the other treatment (on seven plots). The standard grazing treatment was managed as a best-management system, with grazings managed to minimise treading damage. Therefore, the only treatment difference throughout was grazing duration. The trial facilitated the investigation and measurement of pasture production, pasture intake per cow, and losses of N, P, K and faecal microbes through drainage and surface runoff.

1.1 Thesis Structure

This thesis is made up of nine chapters including this ‘Introduction’, and a review of the literature (Chapter 2). Chapters 3-8 cover the field trial research conducted over three years. Each of these chapters has been written in the form of a journal paper, comprising introduction, methods, results, discussion, and conclusion sections. There is a main theme for each chapter. Chapter 3 focusses on pasture production and cow intakes, Chapters 4-7 detail nitrogen, phosphorus, *E. coli*, and potassium losses respectively, and Chapter 8 evaluates the capability of OVERSEER® to model the *DC* grazing system. The main conclusions from the study are summarised in Chapter 9.

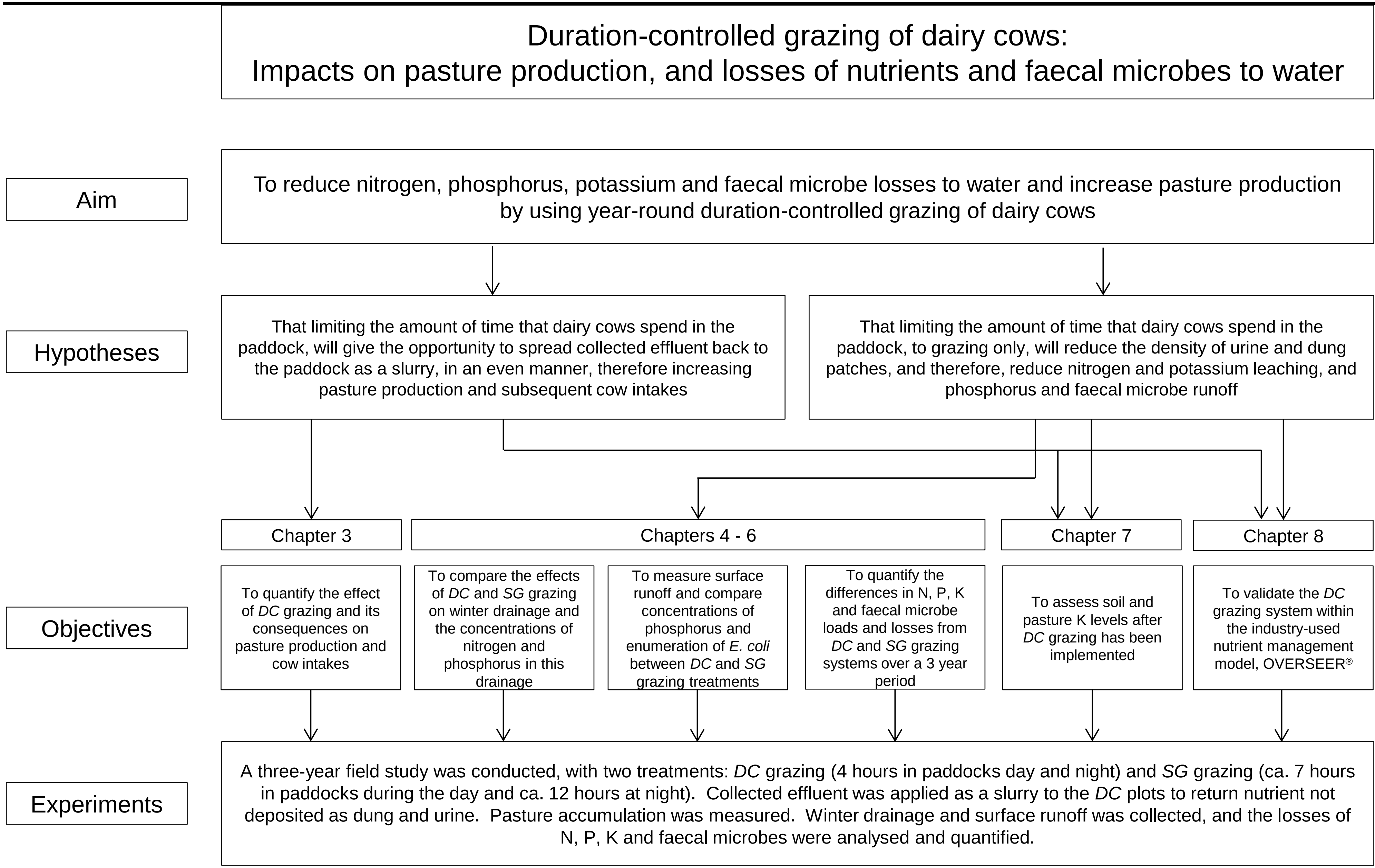

Figure 1.1: Overall aim of experiment and specific hypotheses and objectives for each component of the study.

1.2 References

de Klein, C.A.M., 2001. An analysis of environmental and economic implications of nil and restricted grazing systems designed to reduce nitrate leaching from New Zealand dairy farms. II. Pasture production and cost/benefit analysis. New Zealand Journal of Agricultural Research 44, 217-235.

de Klein, C.A.M., Ledgard, S.F., 2001. An analysis of environmental and economic implications of nil and restricted grazing systems designed to reduce nitrate leaching from New Zealand dairy farms. I. Nitrogen losses. New Zealand Journal of Agricultural Research 44, 201-215.

de Klein, C.A.M., Smith, L.C., Monaghan, R.M., 2006. Restricted autumn grazing to reduce nitrous oxide emissions from dairy pastures in Southland, New Zealand. Agriculture, Ecosystems and Environment 112, 192-199.

Di, H.J., Cameron, K.C., 2005. Reducing environmental impacts of agriculture by using a fine particle suspension nitrification inhibitor to decrease nitrate leaching from grazed pastures. Agriculture, Ecosystems and Environment 109, 202-212.

Fonterra, 2011. http://www.fonterra.com/wps/wcm/connect/fonterracom/fonterra.com/our+business/fonterra+at+a+glance/about+us/key+facts.

Haynes, R.J., Williams, P.H., 1993. Nutrient cycling and soil fertility in the grazed pasture ecosystem. Advances in Agronomy 49, 119-199.

Houlbrooke, D.J., 2005. A study of the quality of articificial drainage under intensive dairy farming and the improved management of farm dairy effluent using 'deferred irrigation'. Insititute of Natural Resources. Massey University, Palmerston North.

Kennedy, E., McEvoy, M., Murphy, J.P., O'Donovan, M., 2009. Effect of restricted access time to pasture on dairy cow milk production, grazing behavior, and dry matter intake. Journal of Dairy Science 92, 168-176.

Laven, R.A., Holmes, C.W., 2008. A review of the potential impact of increased use of housing on the health and welfare of dairy cattle in New Zealand. New Zealand Veterinary Journal 56, 151-157.

Ledgard, S.F., Menneer, J.C., 2005. Nitrate leaching in grazing systems and management strategies to reduce losses. In: Currie, L.D., Hanly, J.A. (Eds.), Developments in fertiliser application technologies and nutrient management. Fertiliser and Lime Research Centre, Massey University, Palmerston North, pp. 79-92.

Livestock Improvement Corporation, DairyNZ, 2011. New Zealand Dairy Statistics 2010-11. DairyNZ.

McDowell, R.W., Muirhead, R.W., Monaghan, R.M., 2006. Nutrient, Sediment, and Bacterial Losses in Overland Flow from Pasture and Cropping Soils Following Cattle Dung Deposition. Communications in Soil Science and Plant Analysis 37, 93-108.

Ministry of Agriculture and Forestry, 2011. http://www.maf.govt.nz/agriculture/pastoral/dairy.

Monaghan, R.M., de Klein, C.A.M., Muirhead, R.W., 2008. Prioritisation of farm scale remediation efforts for reducing losses of nutrients and faecal indicator organisms to waterways: A case study of New Zealand dairy farming. Journal of Environmental Management 87, 609-622.

Muirhead, R.W., Collins, R.P., Bremer, P.J., 2005. Erosion and Subsequent Transport State of *Escherichia coli* from Cowpats. Applied and Environmental Microbiology 71, 2875-2879.

Oudshoorn, F.W., Kristensen, T., Nadimi, E.S., 2008. Dairy cow defecation and urination frequency and spatial distribution in relation to time-limited grazing. Livestock Science 113, 62-73.

Parliamentary Commissioner for the Environment, 2012. Water Quality in New Zealand: Understanding the Science. Parliamentary Commissioner for the Environment, Wellington.

Sharpley, A.N., Syers, J.K., 1979. Loss of nitrogen and phosphorus in tile drainage as influenced by urea application and grazing animals. New Zealand Journal of Agricultural Research 22, 127-131.

Chapter 2 A review of the literature

2.1 Introduction

Dairy farming in New Zealand is based on a predominantly grazed pasture system (Monaghan *et al.*, 2008), giving the country an advantage of producing milk at a low cost per kilogram of milk solids (Holmes *et al.*, 2007). However, there are increasing concerns for the levels of environmental sustainability of New Zealand's dairy systems, and their impact on surface and groundwater quality. The 'clean green' image of the industry's export products is under threat (Parliamentary Commissioner for the Environment, 2012). Nitrogen (N) leaching and phosphorus (P) runoff are the major causes of dairying's adverse impact on water quality (Silva *et al.*, 1999; Di and Cameron, 2002a; Houlbrooke *et al.*, 2004). Faecal coliform-contaminated runoff can also reduce the diversity of water use (ASM, 1999; Muirhead and Littlejohn, 2009). The source of these contaminants is mostly concentrated urine and dung patches that cows deposit when grazing (Ledgard and Menneer, 2005; McDowell *et al.*, 2008).

This review of literature is intended to provide an overview of the information presently in the research literature concerning N, P, faecal coliform, and potassium (K) losses from dairy grazed pastures. It also investigates previous work conducted on mitigation strategies to reduce these losses. In the introduction to subsequent chapters in this thesis (Chapters 3-8) there is a more thorough review for each aspect.

2.2 Nitrogen losses

The amount of N cycling in a dairy system is dependent on several factors, with pasture ingestion by the cow being a significant component of the N cycle. Approximately 20% of ingested N is partitioned to meat and milk production with the remaining 60% of N being excreted as urine and 20% as dung (During, 1972; Ledgard and Steele, 1992; Haynes and Williams, 1993). Approximately 60% of the N ingested as pasture is excreted in urine, with 20% in dung.

Urine is the main source of the nitrate leached from dairy farms (Sharpley and Syers, 1979; Ledgard and Menneer, 2005). The concentrations of nutrients in dung and urine patches are very high and may be equivalent to 1000 kg N/ha in the urine patch (Haynes and Williams, 1993; Di and Cameron, 2002b). Annual pasture production of up to 20 t/ha in urine patches (Di and Cameron, 2002a) at a concentration of 5.5% N straight after deposition but decreasing to 2.5% N within 3 months (Menneer *et al.*, 2003), fails to recover all N prior to the winter drainage season (Silva *et al.*, 1999; de Klein, 2001; Di and Cameron, 2002b; Ledgard and Menneer, 2005; McDowell *et al.*, 2008).

Take, for example, 100 dairy cows grazing one hectare for a full day (20 hours grazing with 4 hours milking), ingesting 1200 kg dry matter (DM) of pasture and 350 kg DM of supplemented maize silage (Figure 2.1). The pasture will consist of, on average, 2.5% N, and the maize silage 1% N (Care and Hedley, 2008). Therefore, the cows will ingest 33.5 kg N/ha/day. Of this, 5.7 kg will go toward milk production and 20 kg will be excreted as urine (Haynes and Williams, 1993). A small amount of N (1.5 kg) will be volatilised as ammonia (NH_3) from the urine patches (Bussink, 1994). These figures are based on percentage values from Haynes and Williams (1993) and are slightly different to those stated by Care and Hedley (2008) in their workshop for dairy farmers on housing cows. However, they follow similar principles for the N cycle.

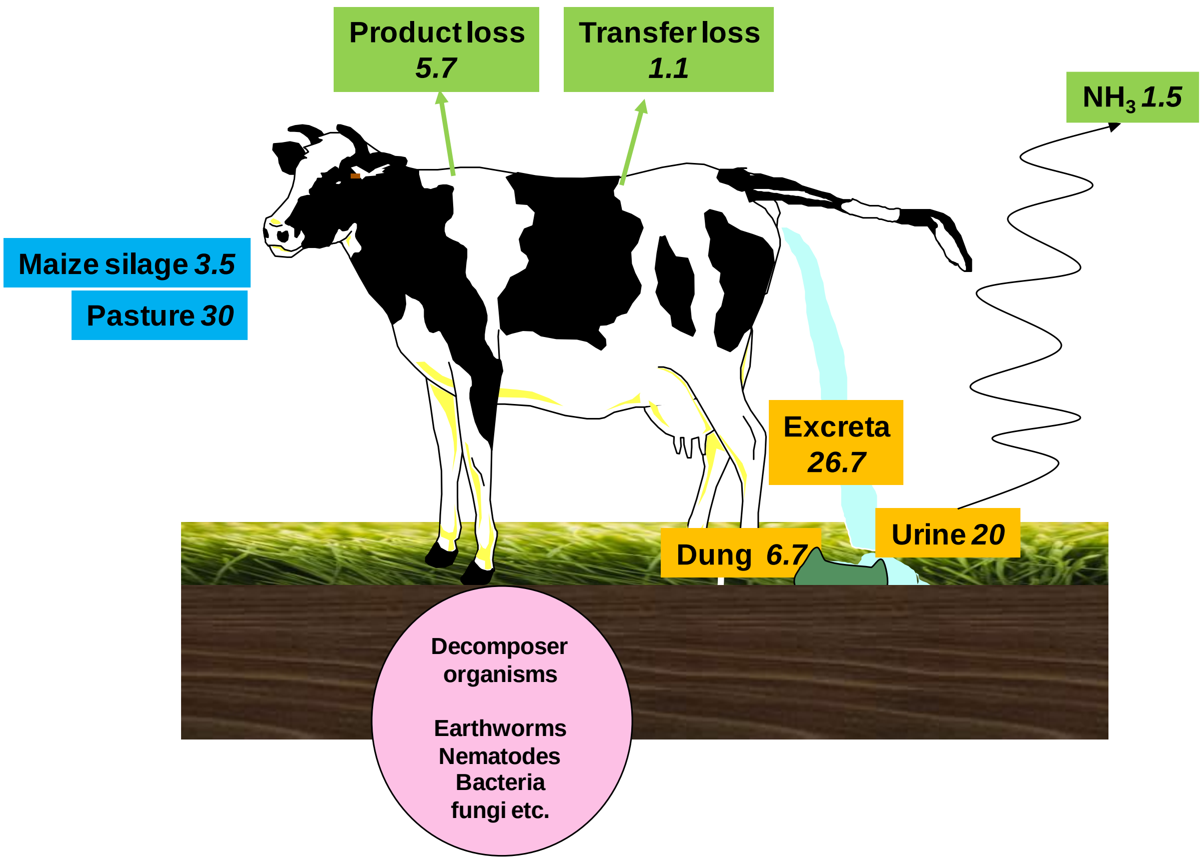

Figure 2.1: Nitrogen (N) transfers in a dairy cow grazing system, for one day's grazing, assuming 100 cows/ha ingesting 1200 kg DM pasture @ 2.5% N and 350 kg DM maize silage @ 1% N. Numbers in *italics* denote kg of N (Adapted from Care and Hedley (2008)).

Dairy cows excrete 11 urinations on average, per cow, per day (Haynes and Williams, 1993). In the New Zealand grazed pasture production system, 9 of these urinations may be deposited on to pasture, covering approximately 0.3 m^2 each (Care and Hedley, 2008). This is based on the proportion of time that cows spend walking to and from the milking shed, where 2 urinations will be deposited on races and tracks, and in the milking shed (contributing to the 1.1 kg N as transfer loss). If the same scenario is used as above, with 100 cows/ha grazing on pasture, and assuming a urine patch area of 0.3 m^2, then the area of pasture covered in urine spots will be 0.027 ha after one day (100 cows × 9 urinations × 0.3 m^2 × 0.0001 ha/m^2). Assuming there is no overlap of urine patches over the year, then after 9 grazings, 24% of the pasture will have had urine deposition (0.027 ha × 9 grazings).

If we assume that 20 kg N is excreted as urine, then 9/11th of this (16.4 kg N) is deposited on the paddock in 0.027 ha of urine patches, applying the equivalent of 607 kg N/ha to the urine patch areas. Menneer *et al.* (2003) measured grass growth of 8340 kg DM/ha under urine spots from the time of deposition to the start of winter drainage. At the first harvest after urination, the N concentration in the grass was 5.5% N, falling to 2.5% a year later. During this period, a total of 342 kg N/ha equivalent was taken up by grass growing in urine spots (Menneer *et al.*, 2003). This equates to 9.2 kg N taken up by grass in the 0.027 ha urine patch area. Thus, 7.2 kg N (16.4– 9.2 kg N) is left in the soil under one-year old urine patches, which, if an insignificant amount is volatilised or immobilised into soil organic N, remains susceptible to winter leaching.

If this occurs on 9 grazings over a year, then a pool of 65 kg N is liable to be leached in winter drainage. This is similar to the amount of N left in the soil estimated by Care and Hedley (2008), where they assumed two thirds of the N deposited as urine was utilised for pasture production. This does not include the 6.7 kg N deposited within dung at each grazing, as it has been well documented that due to the organic nature of N in dung, it is not a major contributor to N leaching (Hack-ten Broeke *et al.*, 1996; Wachendorf *et al.*, 2005).

In a practical situation however, there is likely to be an overlap of urine patches (Auerswald *et al.*, 2010), hence less area of the pasture will have had urine deposited on it, and some areas will be more concentrated from urine patches overlapping. Nevertheless, the model described above, with no urine overlap, identifies a major contributor of N leaching to be the grazing cow concentrating N in urine patches.

The result of a small area having urine deposition is that it uncouples the N cycle, with N being harvested (eaten) and deposited in small areas, with the majority of the area having no urine deposited (Vinther, 1998; Schnyder *et al.*, 2010). Pasture growth may be N limited until such time as excreta is deposited on these areas. The effect is magnified with time as cows preferentially eat areas that have not been contaminated by urine or particularly dung deposition at the previous grazing (Auerswald *et al.*, 2010). Conversely, it has been shown that in some cases, urine patches are preferred at grazing (Day and Detling, 1990), probably because of a higher N content in that area of pasture, particularly in the grazing directly following deposition. It is universally observed however that dung patches tend to be avoided at the next grazing if pasture allocation exceeds the cows' energy requirements.

The spatial uncoupling that occurs reduces the amount of N left in non-urine patch areas, and consequently more N as fertiliser is applied to maintain productivity. In turn, this fertiliser N increases pasture N concentrations causing further N to be excreted as urine (Castillo *et al.*, 2000) and more lost from urine deposition as leaching, and the cycle continues. The quantitative detail of such a cycle is examined in more detail in Chapter 4.

Most of the N lost through the soil profile as leaching is in the form of nitrate (NO_3^-) (Monaghan *et al.*, 2002; Decau *et al.*, 2003; Ledgard and Menneer, 2005; van Es *et al.*, 2006). This is largely due to the fact that NO_3^- ions, unlike phosphate ions, do not undergo specific adsorption by positively charged hydrous oxide surfaces in soils (Wild, 1950).

In New Zealand, the losses of NO_3^- from agricultural practices have been well documented, particularly those losses from dairy pastures. The urine patches deposited by cows are the main source of loss (Haynes and Williams, 1993). Considerable research has been conducted investigating N leaching from dairy farms on fine-textured, mole and pipe drained soils in Southland (Monaghan *et al.*, 2002; Monaghan *et al.*, 2005) and Manawatu (Sharpley, 1977; Houlbrooke, 2005), as well as on free-draining soils in areas such as the Waikato (Shepherd *et al.*, 2011).

In Southland, NO_3^- leaching losses were highest with high amounts of N fertiliser applied (up to 400 kg N/ha/yr), and consequently, higher stocking rates (to match pasture growth) (Monaghan *et al.*, 2005). The annual average loss of NO_3^- in drainage over 3 years ranged from 30 kg N/ha/yr with no N fertiliser, to 56 kg N/ha/yr with 400 kg N/ha fertiliser. This increase in N leaching corresponded to the increase in stocking rate, which increased from the equivalent of 2.4 cows/ha on the treatment that had no N fertiliser, to 3.3 cows/ha on the 400 kg N/ha treatment. Therefore, adding the N fertiliser was not the only cause of increased N leaching, as the increase in stocking rate and consequent urine deposition was likely to have had a major influence.

The timing of N losses in drainage has also been investigated (Monaghan *et al.*, 2002; Houlbrooke, 2005). In Southland studies, the concentration of NO_3^- was highest at the beginning of the drainage season, decreasing to levels of ca. 2 ppm after 300 mm of drainage. This was also the case in the Manawatu, with Houlbrooke (2005) reporting decreasing levels of NO_3^- in drainage from a pallic soil. However, the levels decreased to ca. 2 ppm after only 200 mm drainage in this case. During this time, grazing of dairy cows increased NO_3^- concentrations in drainage by ca. 5 ppm in the subsequent drainage event, and then decreased back to previous levels. The increases in NO_3^- concentrations in drainage were higher after winter grazings than in spring, which Houlbrooke (2005) concluded was most likely due to greater plant uptake of N in the spring than during the winter months.

In his early work in the Manawatu, Sharpley (1977) reported that grazing in the winter months (July) at a stocking rate of 300 cows/ha for 12 hours, increased NO_3^- and total N (TN) concentrations in drainage by 12 and 16 ppm, respectively (Figure 2.2). These elevated levels lasted for three weeks before decreasing back to base levels of ca. 2 ppm by the start of spring (September). Also, when urea was added to plots, the concentration of NO_3^- and TN in drainage approximately doubled; however the peak concentrations did not last as long, and levels returned to base values within 10 weeks of the urea application. Contrary to the later work of Monaghan *et al.* (2002) and Houlbrooke (2005), the concentrations of NO_3^- in drainage at the start of the season were not as high, but this may have been simply due to lower stocking rates and/or N fertiliser use in the preceding months.

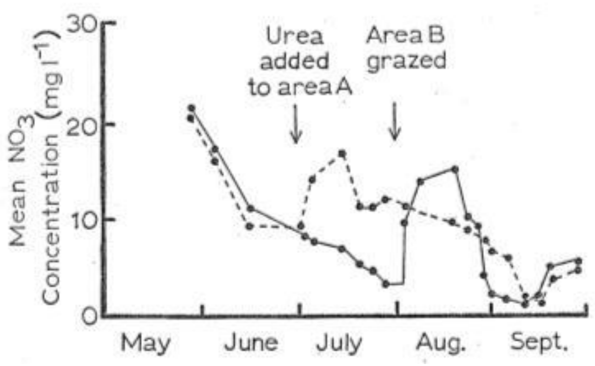

Figure 2.2: Mean NO_3^- concentration in drainage, with arrows indicating urea application and grazing (dashed line = Area A, solid line = Area B) (Sharpley, 1977).

There has been work conducted overseas, to determine the influence of N uptake by pasture prior to the winter drainage season, on N leaching losses (Cuttle and Bourne, 1993; Decau *et al.*, 2003, 2004). In the UK, Cuttle and Bourne (1993) reported that there was a greater uptake by pasture of N from urine deposited in summer (i.e. July) than from urine deposited in late autumn (November). Uptake gradually decreased from 40% of the N applied in the first deposition, to 1% of the N applied in the final deposition in late autumn. Therefore much of the urine that was deposited closer to winter remained in the soil ready for leaching when drainage occurred.

Further to this, a French study also showed that the amount of urine N recovered by pasture decreased with the deposition date (Decau *et al.*, 2003, 2004), regardless of differing soil types. In this case, N leaching of the deposited urine increased with later urine deposition. Between 43% and 77% of the autumn-deposited urine N was leached, compared with between 4% and 34% of the spring-deposited urine N. The comparisons with recent New Zealand work (Shepherd *et al.*, 2011) will be made in more detail in Chapter 4.

In summary, the dairy cow both increases the movement of N and uncouples the N cycle in grazed pasture. Nitrogen is concentrated and spatially uncoupled from the greater area of pasture in urine return. Separate deposition of urine and dung partially uncouples the cycling of nutrients. Uncoupled surplus N in urine patches is liable for leaching in seasonal drainage water. Grazing dairy cows have a large part to play in N losses through leaching, and therefore the management of cows is the main method of reducing N losses. The variation in reports from N leaching studies indicates that more work should be done on determining the most important timing for grazing and N fertiliser application, and how these relate to drainage N losses.

2.3 Phosphorus losses

Phosphorus is an important nutrient for all organisms within pastoral systems and has a fundamental role in cell division and meristem tissue development in plants (Russell, 1973), and cell membrane and skeletal structure of animals. It is also important for buffering pH changes in the rumen of grazing animals (Satter *et al.*, 2005). For these reasons, New Zealand dairy pastures receive regular P fertiliser application to maintain optimum levels of plant-available P in soils (Roberts and Morton, 1999). The form taken up by plants is the inorganic phosphate ion, which is also the form in which it is lost to waterways. Phosphate ions are adsorbed and absorbed strongly by hydrous oxides of iron and aluminium on the surfaces of soil particles (Ryden and Syers, 1975), and therefore the major pathway of phosphate loss to waterways is through surface runoff of particulate soil material (Sharpley, 1985; McDowell *et al.*, 2008), whereas very little phosphate reaches waterways through soil drainage.

When cows graze, the deposition of dung is a primary contributor to phosphate return to the soil. The concentration of phosphate is higher in dung than in urine, and dung is thus an important source of phosphate when surface runoff occurs. If surface runoff occurs soon after grazing by dairy cattle, the amount of P that is lost in this runoff is higher than if the pasture had not been grazed in recent times (Smith *et al.*, 2001; Smith and Monaghan, 2003; McDowell *et al.*, 2006).

If we take the same grazing example as was used in Section 2.2, i.e. 100 cows per hectare grazing on pasture, ingesting 1200 kg DM of pasture and 350 kg DM maize silage per day (2 grazings), we can estimate the amount of P being utilised at each part of the P cycle (Figure 2.3). The pasture ingested will consist of 0.33% P (Betteridge, 1986), while the silage will consist of approximately 0.15% P (Jagusch and Hollard, 1974).

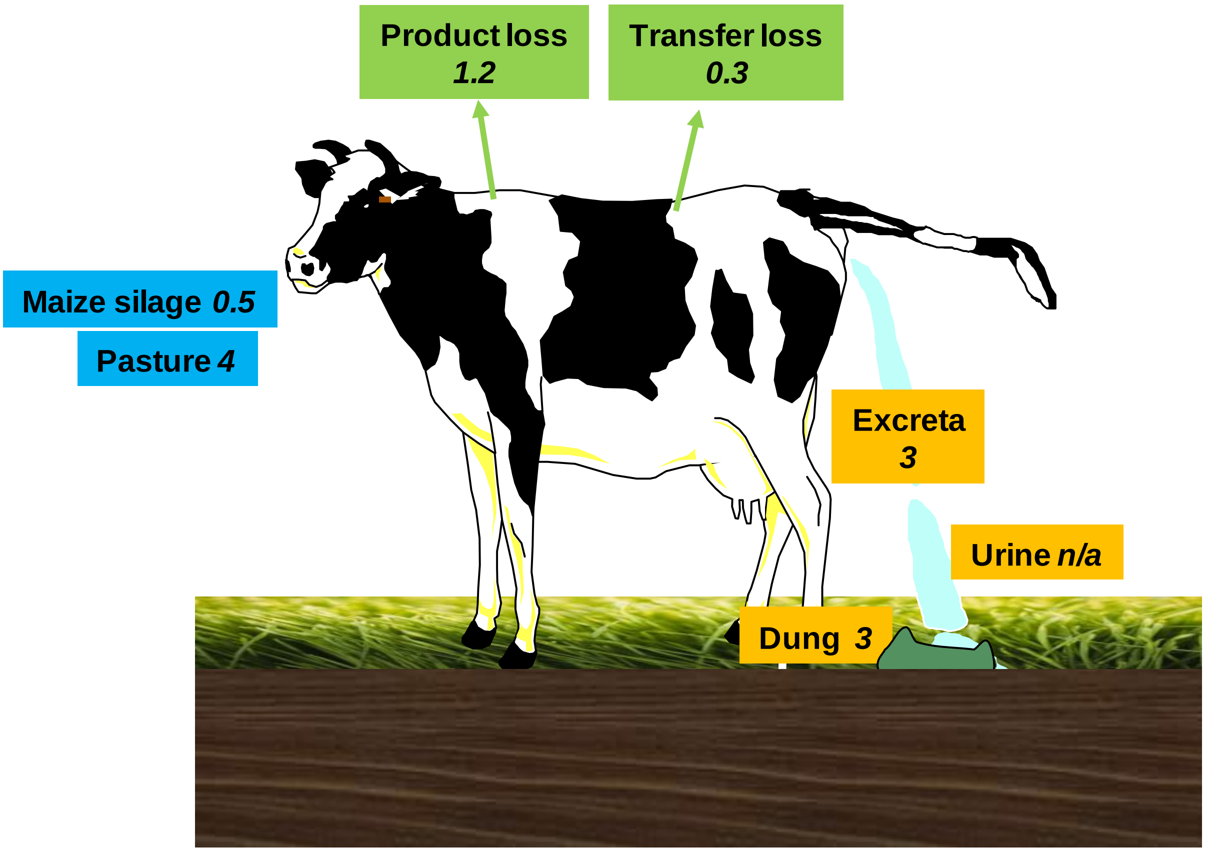

Figure 2.3: Phosphorus (P) cycle in a dairy cow grazing system, for one day's grazing, assuming 100 cows/ha ingesting 1200 kg DM pasture @ 0.33% P and 350 kg DM maize silage @ 0.15% P. Numbers in *italics* denote kg of P.

The amount of P excreted in faeces as a percentage of that ingested is around 65% (Haynes and Williams, 1993). Milk production accounts for 26% of the P, while the remaining P is retained in the cow (Haynes and Williams, 1993) and subsequently transferred to other areas of the farm as dung.

In New Zealand pasture based dairying, the interval between evening and morning milkings is the greatest. Therefore, paddocks that are routinely grazed at night will have greater return of P in dung, and hence these paddocks will have a greater amount of P cycling in them over time (Bolan *et al.*, 2004). Paddocks that are generally kept as 'day' paddocks will, over time, have less P in the soil and the pasture, and the cows will transfer the P from these paddocks to night paddocks through grazing. Furthermore, with 26% of the P ingested going into milk production (Haynes and Williams, 1993), there are significant losses of P from the farm as product.

Grazing of cows, especially when soil moisture conditions are high, can cause treading damage to the soil (Willatt and Pullar, 1983; Horne and Singleton, 1997). Soils suffering from treading damage and wet soils have greater potential for P loss via surface runoff (McDowell *et al.*, 2008), particularly where fresh dung has been deposited. High stocking rates of cattle contribute more to this P loss through greater treading damage, such as compaction, and hence decreased water infiltration rates (Drewry and Paton, 2000; Smith and Monaghan, 2003).

As with N losses, the grazing management of cows has a large part to play in reducing P losses to waterways. This is mostly associated with the amount of dung deposited and treading damage. More detailed discussion will be given in Chapter 5.

2.4 Faecal coliform losses

Cows are asymptomatic carriers of several zoonotic enteric pathogens. Microbial organisms, for example Campylobacter, Giardia, Cryptosporidium and Escherichia coli (*E. coli*) are all sourced from faecal matter and can enter waterways through drainage or runoff (Collins, 2004).

Concentrations are generally higher in surface runoff from grazed pastures (Wilcock *et al.*, 1999) due to dung being freely transported in runoff (NIWA, 2006). The rate of dispersion in runoff of the dung pat is related to how quickly a hard skin forms over the pat (Haynes and Williams, 1993), and therefore the weather following grazing has an impact on the probability of faecal microbes being lost via surface runoff (Muirhead *et al.*, 2005; McDowell *et al.*, 2008). *E. coli* is often used as a faecal indicator organism (FIO) – i.e. if the concentrations of *E. coli* are high in surface runoff, it indicates that other faecal organism concentrations could also be high (Toranzos and McFeters, 1997). The entry of such organisms to waterways can pose serious health risks, with levels of 550 *E. coli* / 100 mL of water from recreational waterways being deemed as the upper limit for safe levels for the public (Ministry for the Environment, 2010).

The large flows of N, P and enteric pathogens via dairy cow excreta return indicates that control measures to reduce the losses of these contaminants to waterways should focus on the spatial and temporal management of dairy cows. More detail on the pathways of loss for *E. coli* will be given in Chapter 6.

2.5 Potassium losses

Potassium is an important element in the pastoral system for plant growth (Williams, 1988), in regulating processes such as photosynthesis and protein synthesis in the plant. It is taken up by the plant as a cation (K^+) and is in abundance on soil surfaces and in soil solution.

In pasture herbage, K is found in similar concentrations to that of N, with a wide range being reported (0.98 – 5.21%; Smith and Middleton, 1978). Less than 4% of the K ingested by the cow is retained in animal tissue, with up to 92% being excreted. Furthermore, the excreted K is partitioned mostly into urine (90%), with very little K excreted as dung (10%; Williams, 1988).

Therefore, the cycling of K is similar to N, being ingested from large areas and deposited to small, concentrated areas. The annual uptake of K in pasture is also similar to N, leaving relatively large amounts of K in the soil solution under urine patches, that is available for soil surface exchange or leaching (Williams, 1988).

There are large differences in the consequences of K and N leaching, however. As described in Section 2.2, N leaching can cause eutrophication in waterways with detrimental effects on New Zealand’s freshwater quality. Conversely, K leaching from the grazed pastoral system does not have any direct environmental consequences, but does affect the uncoupling of the K cycle and thus requires reapplication of K fertiliser to the pasture (Williams, 1988).

Due to the research emphasis on N and P losses from pasture, there have been few practical field studies in New Zealand reporting on the amount of K that is leached from dairy farms. Most reports have been based on simple modelling exercises (Williams, 1988). While the present study is being undertaken, the opportunity arises for K leaching to be monitored. This will be reported in more detail in Chapter 7.

2.6 Management options to reduce N, P, K and faecal microbe losses to waterways

There have been many options investigated for the management of dairy cows to reduce nutrient losses while maintaining or increasing on-farm pasture and milk production (Monaghan *et al.*, 2008). Riparian strips, particularly for the capture of P and faecal microbes, have been used on farm to remove nutrients from surface runoff before it enters waterways (Wilcock *et al.*, 1999; Collins *et al.*, 2004; Muirhead *et al.*, 2005).

2.6.1 Changes by farmers

Simple changes, such as grazing cattle away from waterways, have been implemented by most farmers, which have been encouraged by Regional Council regulations.

The Clean Streams Accord was signed in 2003 by representatives of Fonterra, Ministry of Agriculture and Forestry, Ministry for the Environment and Local Government New Zealand. This Accord was a statement of intent to promote sustainable dairy farming in New Zealand. It included priorities to fence off streams and exclude cows from waterways, and for all farms to have systems in place to manage nutrients (Fonterra *et al.*, 2003).

Furthermore cows are also being housed in situations where surface runoff is likely to occur following grazing, or where treading damage is likely to be a serious issue in wet conditions (Care and Hedley, 2008).

2.6.2 Nitrification Inhibitors

In recent times, nitrification inhibitors (e.g. dicyandiamide (DCD)) have been used to reduce the prevalence of NO_3^- leaching, particularly from urine patches (Di and Cameron, 2005). They are recommended to be applied twice yearly, in the autumn and late winter. Larger decreases in NO_3^- leaching from using nitrification inhibitors have been seen in cooler climates (e.g. Southland) by up to 55% (Monaghan, 2009), but there has been little and inconclusive evidence of their efficacy in warmer climates (e.g. North Island of New Zealand). These parameters around temperature and timing of application have been identified as important for future work to test the effectiveness of nitrification inhibitors nationwide (Monaghan *et al.*, 2007). As this thesis goes to print, DCD has been withdrawn from use on NZ dairy farms because of traces becoming apparent in milk products (Ministry for Primary Industries, 2013). Thus it may be some time before nitrification inhibitors can be used again to reduce NO_3^- leaching from dairy farms.

2.6.3 Housing cows

One way of reducing the N, P and faecal microbe losses from grazing is by removing cows from paddocks and adopting a ‘European-like’ system, where feed is cut and carried to the herd which are housed indoors. The excreta that is collected from the indoor facility can be stored and spread on to crops to enhance growth for further cutting and carrying. This method has been used successfully in European countries, particularly those with adverse climates. However, it is an expensive option due to the level of infrastructure that is needed. To maintain its current competitive advantage, the New Zealand dairy industry must continue to produce high-quality milk products at a low cost – which means it must take advantage of the pastoral grazing system that provides this milk for relatively lower cost compared with our European competitors. Therefore, the industry needs to continue to utilise the pasture, but reduce the environmental impacts from grazing it *in situ*.

2.6.4 Duration-controlled grazing

An option to mitigate the nutrient losses while retaining pastoral grazing is to adopt duration-controlled (*DC*) grazing practices, which involves limiting the time cows are grazed in paddocks. In these systems the cows spend approximately 8 to 12 hours grazing per day and the rest of the time in an animal shelter or standoff facility where they receive supplementary feed (de Klein *et al.*, 2000; de Klein and Ledgard, 2001; Ledgard and Menneer, 2005; Oudshoorn *et al.*, 2008).

Through modelling, de Klein *et al.* (2000) and de Klein and Ledgard (2001) investigated the effects of restricted grazing where cows were kept off paddocks from April to August, and conventionally grazed through spring and summer (September to March). The pasture was cut and carried during the restricted grazing period, and 85% of the effluent N was returned to the pasture via surface spreading. It was estimated that 4% of this N was leached through the soil profile and into waterways. These studies found that compared to a conventionally grazed system, restricted grazing (i.e. nil grazing through autumn and winter) had the potential to decrease N leaching by 35-50%, depending on the N inputs to the farm via fertiliser and supplementary feed.

This practice of removing cows after a grazing period decreases the number of urine and dung patches deposited in the paddock, and hence the amount of N leaching and P runoff. It has been shown that the distribution of urine shows a similar pattern to that of dung (Hilder, 1966), and thus dung can be used as an indicator of the distribution of both forms of excreta. The additional effluent collected during the time spent in the animal shelter can be stored and returned to paddocks when conditions are favourable, i.e. soil moisture is not too deficient for pasture growth, and not too wet for machinery application (de Klein, 2001). This excreta is able to be spread evenly and at a nutrient rate suitable for plant uptake (Chadwick *et al.*, 2002), and can be spread several times throughout the year and also over a greater area of a farm than a conventional effluent system (farm dairy effluent collected from the milking shed and yards). Furthermore, depending upon the quantity of imported supplements fed to housed cows, the nutrient return through spreading effluent from animal shelters may be sufficient to retain or increase soil nutrient levels, ceasing or at least reducing the need for the use of manufactured fertilisers. This can have economic benefits over an extended period of time.

Another advantage of spreading the effluent collected in the animal shelter is the ability of the pasture to utilize the nutrients soon after it has been spread (de Klein, 2001). This can result in an increase in pasture production over time for *DC* grazed areas, particularly on areas that have had low manufactured N fertiliser applications and hence have lower soil N levels than highly fertilised areas (de Klein, 2001). Subsequently, with an increase in pasture production, the farm practising *DC* grazing requires fewer purchased supplements to maintain milk production, which, again, has economic benefits, as well as decreasing the risk of not being able to source the desired supplements.

Animal shelters allow a measured amount of feed to be offered to dairy cows, with less wastage than other means of feeding, for example, feeding supplements in paddocks (Dexcel, 2005). In turn, this requires less supplementary feed to be brought in to the system, with high utilization rates. A more balanced and possibly specialized diet is also able to be offered.

A disadvantage of storing excreta before it is spread evenly to paddocks however, is the losses of N as ammonia (Longhurst *et al.*, 2006). It was found in a study of various Herd Homes, that depending on the type of material incorporated into the bunker of the standoff facility, losses of N were between 10 and 70%. An average of the losses of N over 9 months was approximately 50%, which are the losses that will be used in the following example (Figure 2.4).

Take the example that was used to explain the N cycle for grazing cows in Section 2.2; i.e. 100 cows grazing per hectare for a day (2 grazings), ingesting 1200 kg DM pasture and 350 kg DM maize silage, equivalent to ingesting 33.5 kg N (Figures 2.1 and 2.4). If these cows graze for, say, 8 hours per day (4 hours per grazing), and they are housed for 12 hours of the day, the number of urinations to the paddock will decrease to 3.7 urinations/cow (8 / 24 hours x 11 urinations / day). Therefore, only 0.011 ha of the paddock will be covered in urine patches (as compared with fulltime grazing of 0.027 ha, Figure 2.1). Furthermore, only 8.9 kg N will be excreted on to the paddock (8 / 24 x 26.7 kg N), with 6.7 kg N being excreted as urine (8 / 24 x 20 kg N). Using the same proportions and assumptions as were used in Section 2.2 from Menneer *et al.* (2003), then the pasture will take up 4.5 kg N. Thus, 2.2 kg will be left in the soil, and from 9 grazings over a year, this will leave only 20 kg N liable for leaching in winter drainage (or immobilisation). This equates to 31% of the N that was liable for leaching using an all-grazing system (Figure 2.1).

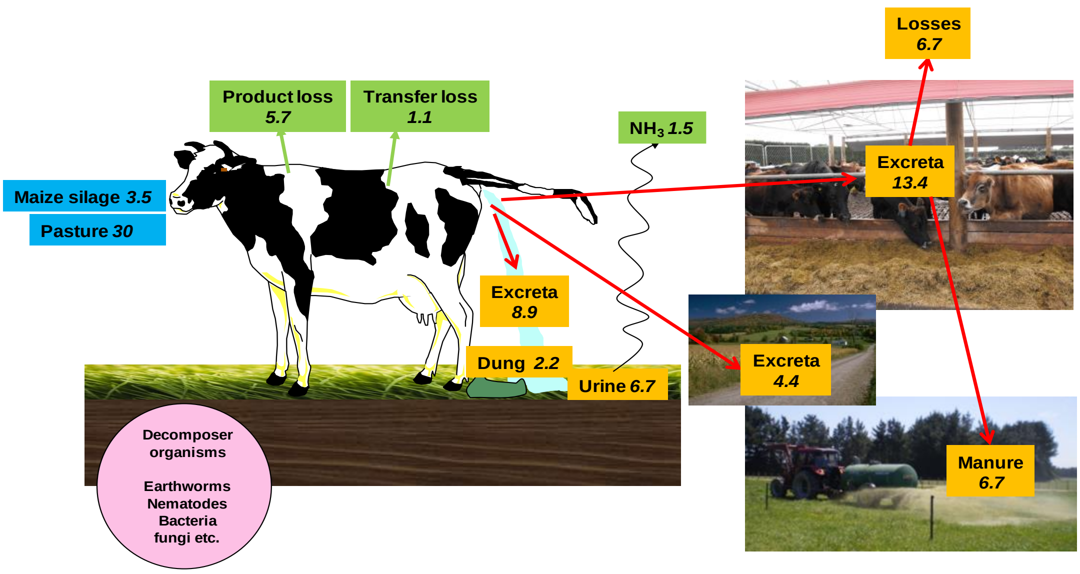

Figure 2.4: Nitrogen (N) transfers in a dairy cow grazing system for one day's grazing with cows being housed indoors for 12 hours of the day (grazed on pasture for 8 hours per day), assuming 100 cows/ha ingesting 1200 kg DM pasture and 350 kg DM maize silage. Numbers in *italics* denote kg of N. Adapted from Care and Hedley (2008).

The time that the cows spend in the standoff facility (12 hours as outlined in Figure 2.4), will also contribute to excreta losses from the cows (12 / 24 hours x 26.7 kg N). Therefore 13.4 kg N will be excreted at the standoff facility from these 100 cows in 12 hours. If this excreta is stored, then losses will be 6.7 kg N (Longhurst *et al.*, 2006). Assuming 9 grazings a year, then there will be 60 kg N available to be spread as manure back on to the paddocks at an even rate when conditions allow (Figure 2.4).

It has been shown that dairy cows consume most of the pasture they are offered in the first four hours of grazing, following milking. This time is followed by rumination, then walking and standing (Draganova, 2009; Draganova *et al.*, 2010). As cows achieve sufficient intakes in the first four hours of grazing, removing them from the pasture can have several benefits including reducing the risk of treading damage (particularly in wet conditions) (Blackwell, 1993). This is especially important during winter months when soil moisture and consequent drainage volumes are high, with increased losses through surface runoff. Furthermore, treading damage has adverse effects on pasture growth, with growth rates reduced by up to 30-50% in treaded areas (Horne and Singleton, 1997).

Treading damage is more prevalent on poorly drained soils, with high clay content. In wet conditions, treading by grazing cows can damage the structure of the soil (Willatt and Pullar, 1983). If cows are removed from these soils, treading damage does not develop (Chadwick *et al.*, 2002). Therefore, particularly on fine textured, poorly drained soils, duration-controlled grazing practices are likely to have favourable results (Blackwell, 1993). However, there may be times when the advantages of removing cows to reduce treading are outweighed by the problems associated with using heavy equipment to apply excreta to paddocks (de Klein, 2001).

Within New Zealand, regulatory measures are being taken in an attempt to reduce the quantity of N entering waterways by leaching. These measures include N leaching caps, or allowances. The regulations are designed to reduce the level of aquatic plant and algal growth in waterways. Water quality deterioration as a result of plant and algal growth can be partly attributed to the N and P lost from dairy farms (Ledgard *et al.*, 2000; Di and Cameron, 2002b). Reducing these losses is important for preserving or improving the quality of waterways in New Zealand (Silva *et al.*, 1999; Di and Cameron, 2002a; Houlbrooke *et al.*, 2004), and it is also imperative for protecting the image of our export products on the global stage (DairyNZ, 2009). The allowances may be based on either historical leaching values (Environment Waikato, 2007) or on Land Use Capability (LUC) class, with allowances decreasing for higher LUC classes, which are lower-producing areas of land (Horizons Regional Council, 2009). In both cases, OVERSEER® Nutrient Budgets are used to determine these leaching values.

If and when these regulations are enforced, dairy farmers throughout New Zealand will need to adjust their systems to comply with the N leaching allowances. One way of being able to do this is by using duration-controlled grazing practices to decrease the amounts of concentrated urine and dung patches distributed to paddocks. An initial capital outlay would need to be made to build a covered wintering pad or animal shelter (de Klein, 2001). However, it may be the most cost effective option if a large decrease in N leaching is required to meet an allowance. Furthermore, the capital cost could be offset by savings made from lower manufactured fertiliser inputs, less wastage when using supplementary feeds, and higher pasture production, which in turn, could lead to increases in milk production.

Several duration-controlled or 'restricted' grazing studies have been conducted in overseas countries (Oudshoorn *et al.*, 2008), but there has been no long-term work performed in New Zealand. Short-term studies and modelling work have been completed in New Zealand surrounding the practice, with favourable outcomes (de Klein and Ledgard, 2001; de Klein *et al.*, 2006), albeit with many assumptions made. A survey of farmers has also been undertaken to understand why farmers invest in animal shelters. A key factor influencing the decision to house cows within this group of farmers was for pasture and soil protection (70-82% of the farmers surveyed) (Care and Hedley, 2008). In addition, farmers agreed that better feed utilization was an important factor. These are key benefits of duration-controlled grazing, a practice which more New Zealand dairy farmers are incorporating into their system.

2.7 Summary

There is a requirement to conduct a large-scale, long-term experiment in New Zealand to quantify the environmental, agronomic and economic benefits of using duration-controlled grazing practices. A long-term study comparing duration-controlled grazing practices to conventional grazing is a way of determining the efficacy and feasibility of this system.

The research objectives of this long term study were developed in light of the literature reviewed in this chapter (and elsewhere in this thesis). These objectives seek to fill the knowledge gaps identified above and to build on this literature by extending understanding of the advantages of duration-controlled grazing to environmental protection and increased productivity of dairy systems. In short, the objectives of this study are to establish a *DC* grazing system and measure the pasture production, cow intakes, and N, P and faecal microbe losses to water through drainage and surface runoff. Comparisons will be made to a standard grazed system, and practical implications for New Zealand dairy farms will be drawn from the conclusions.

2.8 References

ASM, 1999. Microbial pollutants in our nation's water: environmental and public health issues. http://www.asm.org/images/docfilename/0000005987/waterreport[1].pdf.

Auerswald, K., Mayer, F., Schnyder, H., 2010. Coupling of spatial and temporal pattern of cattle excreta patches on a low intensity pasture. Nutrient Cycling in Agroecosystems 88, 275-288.

Betteridge, K., 1986. A survey of the phosphorus content of pastures and the serum inorganic phosphorus content of dairy cows. New Zealand Veterinary Journal 34, 22-26.

Blackwell, M.B., 1993. Experiences with on-off grazing in early lactation on dairy farms in Northland. New Zealand Society of Animal Production, pp. 37-39.

Bolan, N.S., Saggar, S., Luo, J., Bhandral, R., Singh, J., 2004. Gaseous emissions of nitrogen from grazed pastures: Processes, measurements and modelling, environmental implications, and mitigation. Advances in Agronomy 84.

Bussink, D.W., 1994. Relationships between ammonia volatilization and nitrogen fertilizer application rate, intake and excretion of herbage nitrogen by cattle on grazed swards. Fertilizer Research 38, 111-121.

Care, D., Hedley, M., 2008. Housing New Zealand's dairy cows. In: Brookes, I.M. (Ed.), Dairy3. Dairy3 Organising Committee, Rotorua, pp. 167-177.

Castillo, A.R., Kebreab, E., Beever, D.E., France, J., 2000. A review of efficiency of nitrogen utilisation in lactating dairy cows and its relationship with environmental pollution. J. Anim. Feed Sci. 9, 1-32.

Chadwick, D.R., Ledgard, S.F., Brown, L., 2002. Nitrogen Flows and Losses in Dairy Farms in New Zealand and the UK: Effects of Grazing Management. In: Currie, L.D., Loganathan, P. (Eds.), Dairy Farm Soil Management. Fertiliser and Lime Research Centre, Massey University, Palmerston North, pp. 319-332.

Collins, R., 2004. Fecal contamination of pastoral wetlands. Journal of Environmental Quality 33, 1912-1918.

Collins, R., Donnison, A., Ross, C., McLeod, M., 2004. Attenuation of effluent-derived faecal microbes in grass buffer strips. New Zealand Journal of Agricultural Research 47, 565-574.

Cuttle, S.P., Bourne, P.C., 1993. Uptake and leaching of nitrogen from artificial urine applied to grassland on different dates during the growing season. Plant and Soil 150, 77-86.

DairyNZ, 2009. Strategy for New Zealand Dairy Farming 2009/2020.

Day, T.A., Detling, J.K., 1990. Grassland Patch Dynamics and Herbivore Grazing Preference Following Urine Deposition. Ecology 71, 180-188.

de Klein, C.A.M., 2001. An analysis of environmental and economic implications of nil and restricted grazing systems designed to reduce nitrate leaching from New Zealand dairy farms. II. Pasture production and cost/benefit analysis. New Zealand Journal of Agricultural Research 44, 217-235.

de Klein, C.A.M., Ledgard, S.F., 2001. An analysis of environmental and economic implications of nil and restricted grazing systems designed to reduce nitrate leaching from New Zealand dairy farms. I. Nitrogen losses. New Zealand Journal of Agricultural Research 44, 201-215.

de Klein, C.A.M., Paton, J., Ledgard, S.F., 2000. An evaluation of a strategic de-stocking regime for dairying to improve nitrogen efficiency and reduce nitrate leaching from dairy farms in nitrate-sensitive areas., New Zealand Grassland Association, pp. 105-110.

de Klein, C.A.M., Smith, L.C., Monaghan, R.M., 2006. Restricted autumn grazing to reduce nitrous oxide emissions from dairy pastures in Southland, New Zealand. Agriculture, Ecosystems and Environment 112, 192-199.

Decau, M.L., Simon, J.C., Jacquet, A., 2003. Fate of Urine Nitrogen in Three Soils throughout a Grazing Season. Journal of Environmental Quality 32, 1405-1413.

Decau, M.L., Simon, J.C., Jacquet, A., 2004. Nitrate Leaching under Grassland as Affected by Mineral Nitrogen Fertilization and Cattle Urine. Journal of Environmental Quality 33, 637-644.

Dexcel, 2005. Minimising muck, maximising money: Stand-off and feedpads, design and management guidelines URL: http://www.maf.govt.nz/sff/about-projects/search/03-047/03047-design-management-guidelines.pdf.

Di, H.J., Cameron, K.C., 2002a. Nitrate leaching and pasture production from different nitrogen sources on a shallow stoney soil under flood-irrigated dairy pasture. Australian Journal of Soil Research 40, 317-334.

Di, H.J., Cameron, K.C., 2002b. Nitrate leaching in temperate agroecosystems: sources, factors and mitigating strategies. Nutrient Cycling in Agroecosystems 46, 237-256.

Di, H.J., Cameron, K.C., 2005. Reducing environmental impacts of agriculture by using a fine particle suspension nitrification inhibitor to decrease nitrate leaching from grazed pastures. Agriculture, Ecosystems and Environment 109, 202-212.

Draganova, I., 2009. Personal Communication.

Draganova, I., Yule, I., Betteridge, K., Hedley, M., Stevenson, M., Stafford, K., 2010. Activity patterns and nutrient redistribution in an Intensively Managed Dairy Herd. In: Currie, L.D., Christensen, C.L. (Eds.), Farming's Future: Minimizing Footprints and Maximising Margins. Fertilizer and Lime Research Centre, Massey University.

Drewry, J.J., Paton, R.J., 2000. Effects of cattle treading and natural amelioration on soil physical properties and pasture under dairy farming in Southland New Zealand. New Zealand Journal of Agricultural Research 43, 377-386.

During, C., 1972. Fertilisers and soils in New Zealand farming. New Zealand Department of Agriculture Bulletin No. 409.

Environment Waikato, 2007. Protecting Lake Taupo Strategy.

Fonterra, Local Government New Zealand, Ministry for the Environment, Ministry of Agriculture and Forestry, 2003. Dairying and Clean Streams Accord.

Hack-ten Broeke, M.J.D., De Groot, W.J.M., Dijkstra, J.P., 1996. Impact of excreted nitrogen by grazing cattle on nitrate leaching. Soil Use and Management 12, 190-198.

Haynes, R.J., Williams, P.H., 1993. Nutrient cycling and soil fertility in the grazed pasture ecosystem. Advances in Agronomy 49, 119-199.

Hilder, E.J., 1966. Distribution of excreta by sheep at pasture. Proceedings Xth International Grasslands Congress, Helsinki, pp. 977-981.

Holmes, C.W., Brookes, I.M., Garrick, D.J., MacKenzie, D.D.S., Parkinson, T.J., Wilson, G.F., 2007. Milk Production from Pasture - Principles and Practices. Massey University, Palmerston North.

Horizons Regional Council, 2009. Proposed One Plan.

Horne, D.J., Singleton, P.L., 1997. Managing wet soils in wet years: a cautionary tale. Dairy Farming Annual 49, 117-124.

Houlbrooke, D.J., 2005. A study of the quality of artificial drainage under intensive dairy farming and the improved management of farm dairy effluent using 'deferred irrigation'. PhD Thesis. Massey University, Palmerston North.

Houlbrooke, D.J., Horne, D.J., Hedley, M.J., Hanly, J.A., Scotter, D.R., Snow, V.O., 2004. Minimising surface water pollution resulting from farm-dairy effluent application to mole-pipe drained soils. I. An evaluation of the deferred irrigation system for sustainable land treatment in the Manawatu. New Zealand Journal of Agricultural Research 47, 405-415.

Jagusch, K.T., Hollard, M.G., 1974. Maize feeding studies with town-supply cows. II. Effect of variety on the production of maize grown for silage or grain. New Zealand Journal of Experimental Agriculture 2, 67-73.

Ledgard, S.F., De Klein, C.A.M., Crush, J.R., Thorrold, B.S., 2000. Dairy farming, nitrogen losses and nitrate-sensitive areas. New Zealand Society of Animal Production.

Ledgard, S.F., Menneer, J.C., 2005. Nitrate leaching in grazing systems and management strategies to reduce losses. In: Currie, L.D., Hanly, J.A. (Eds.), Developments in fertiliser application technologies and nutrient management. Fertiliser and Lime Research Centre, Massey University, Palmerston North, pp. 79-92.

Ledgard, S.F., Steele, K.W., 1992. Biological nitrogen fixation in mixed legume/grass pastures. Plant and Soil 141, 137-153.

Longhurst, R.D., Luo, J., O'Connor, M.B., Pow, T., 2006. Herd Homes: nutrient management and farmer perceptions of performance. Proceedings of the New Zealand Grassland Association 68, 309-313.

McDowell, R.W., Houlbrooke, D.J., Muirhead, R.W., Muller, K., Shepherd, M., Cuttle, S.P., 2008. Grazed pastures and surface water quality. Nova Science Publishers.

McDowell, R.W., Muirhead, R.W., Monaghan, R.M., 2006. Nutrient, Sediment, and Bacterial Losses in Overland Flow from Pasture and Cropping Soils Following Cattle Dung Deposition. Communications in Soil Science and Plant Analysis 37, 93-108.

Menneer, J.C., Ledgard, S.F., McLay, C., Silvester, W., 2003. The effect of a single application of cow urine on annual N_2 fixation under varying simulated grazing intensity, as measured by four ^{15}N isotope techniques. Plant and Soil 254, 469-480.

Ministry for Primary Industries, 2013. http://www.mpi.govt.nz/news-resources/news/dcd-update-testing-confirms-expectations-of.

Ministry for the Environment, 2010. http://www.mfe.govt.nz/environmental-reporting/freshwater/recreational/monitoring.html.

Monaghan, R.M., 2009. http://www.horizons.govt.nz/assets/horizons/Images/One%20Plan%20Officers%20reports%20water%20hearing/Dr%20Ross%20Martin%20Monaghan.pdf.

Monaghan, R.M., de Klein, C.A.M., Muirhead, R.W., 2008. Prioritisation of farm scale remediation efforts for reducing losses of nutrients and faecal indicator organisms to waterways: A case study of New Zealand dairy farming. Journal of Environmental Management 87, 609-622.

Monaghan, R.M., Hedley, M.J., Di, H.J., McDowell, R.W., Cameron, K.C., Ledgard, S.F., 2007. Nutrient management in New Zealand pastures - recent developments and future issues. New Zealand Journal of Agricultural Research 50, 181-201.

Monaghan, R.M., Paton, R.J., Drewry, J.J., 2002. Nitrogen and phosphorus losses in mole and tile drainage from a cattle-grazed pasture in eastern Southland. New Zealand Journal of Agricultural Research 45, 197-205.

Monaghan, R.M., Paton, R.J., Smith, L.C., Drewry, J.J., Littlejohn, R.P., 2005. The impacts of nitrogen fertilisation and increased stocking rate on pasture yield, soil physical condition and nutrient losses in drainage from a cattle-grazed pasture. New Zealand Journal of Agricultural Research 48, 227-240.

Muirhead, R.W., Collins, R.P., Bremer, P.J., 2005. Erosion and Subsequent Transport State of *Escherichia coli* from Cowpats. Applied and Environmental Microbiology 71, 2875-2879.

Muirhead, R.W., Littlejohn, R.P., 2009. Die-off of *Escherichia coli* in intact and disrupted cowpats. Soil Use and Management 25, 389-394.

NIWA, 2006. Pathogen pathways - best management practices (Technical Paper prepared for MAF Policy).

Oudshoorn, F.W., Kristensen, T., Nadimi, E.S., 2008. Dairy cow defecation and urination frequency and spatial distribution in relation to time-limited grazing. Livestock Science 113, 62-73.

Parliamentary Commissioner for the Environment, 2012. Water Quality in New Zealand: Understanding the Science. Parliamentary Commissioner for the Environment, Wellington.

Roberts, A.H.C., Morton, J.D., 1999. Fertiliser Use on New Zealand Dairy Farms. New Zealand Fertiliser Manufacturers' Association, Auckland, New Zealand.

Russell, E.W., 1973. Soil conditions and plant growth (10th Edition). Longman Group Ltd, London.

Ryden, J.C., Syers, J.K., 1975. Rationalization of ionic strength and cation effects on phosphate sorption by soils. Journal of Soil Science 26, 395-406.

Satter, L.D., Klopfenstein, T.J., Erickson, G.E., Powell, J.M., 2005. Phosphorus and Dairy/Beef Nutrition. In: Sims, J.T., Sharpley, A.N. (Eds.), Phosphorus: Agriculture and the Environment. American Society of Agronomy, Crop Science Society of America, Soil Science Society of America, Wisconsin, USA.

Schnyder, H., Locher, F., Auerswald, K., 2010. Nutrient redistribution by grazing cattle drives patterns of topsoil N and P stocks in a low-input pasture ecosystem. Nutrient Cycling in Agroecosystems 88, 183-195.

Sharpley, A.N., 1977. Sources and transport of phosphorus and nitrogen in a stream draining a dominantly pasture catchment. PhD Thesis in Soil Science. Massey University, Palmerston North.

Sharpley, A.N., 1985. The selective erosion of plant nutrients in runoff. Soil Science Society of America Journal 49, 1527-1534.

Sharpley, A.N., Syers, J.K., 1979. Loss of nitrogen and phosphorus in tile drainage as influenced by urea application and grazing animals. New Zealand Journal of Agricultural Research 22, 127-131.

Shepherd, M., Phillips, P., Snow, V., 2011. The challenge of late summer urine patches in the Waikato region. In: Currie, L.D., Christensen, C.L. (Eds.), Adding to the knowledge base for the nutrient manager, Occasional Report No. 24. Fertilizer & Lime Research Centre, Palmerston North.

Silva, R.G., Cameron, K.C., Di, H.J., Hendry, T., 1999. A lysimeter study of the impact of cow urine, dairy shed effluent, and nitrogen fertiliser on nitrate leaching. Australian Journal of Soil Research 37, 357-369.

Smith, G.S., Middleton, K.R., 1978. Sodium and potassium content of topdressed pasture in New Zealand in relation to plant and animal nutrition. New Zealand Journal of Experimental Agriculture 6, 217-225.

Smith, K.A., Jackson, D.R., Withers, P.J.A., 2001. Nutrient losses by surface run-off following the application of organic manures to arable land. 2. Phosphorus. Environmental Pollution 112, 53-60.

Smith, L.C., Monaghan, R.M., 2003. Nitrogen and phosphorus losses in overland flow from a cattle-grazed pasture in Southland. New Zealand Journal of Agricultural Research 46, 225-237.

Toranzos, G.A., McFeters, G.A., 1997. Detection of indicator microorganisms in environmental freshwater and drinking waters. In: Hurst, C.J. (Ed.), Manual of Environmental Microbiology. ASM Press, Washington DC, pp. 184-194.

van Es, H.M., Sogbedji, J.M., Schindelbeck, R.R., 2006. Effect of Manure Application Timing, Crop, and Soil Type on Nitrate Leaching. Journal of Environmental Quality 35, 670-679.

Vinther, F.P., 1998. Biological nitrogen fixation in grass-clover affected by animal excreta. Plant and Soil 203, 207-215.

Wachendorf, C., Taube, F., Wachendorf, M., 2005. Nitrogen leaching from ^{15}N labelled cow urine and dung applied to grassland on a sandy soil. Nutrient Cycling in Agrosystems 73, 89-100.

Wilcock, R.J., Nagels, J.W., Rodda, H.J.E., O'Connor, M.B., Thorrold, B.S., Barnett, J.W., 1999. Water quality of a lowland stream in a New Zealand dairy farming catchment. New Zealand Journal of Marine and Freshwater Research 33, 683-696.

Wild, A., 1950. The retention of phosphate by soil. A review. Journal of Soil Science 1, 221-238.

Willatt, S.T., Pullar, D.M., 1983. Changes in Soil Physical Properties under Grazed Pastures. Australian Journal of Soil Research 22, 343-348.

Williams, P. H., 1988. The fate of potassium in grazed dairy pastures. PhD Thesis. Massey University, Palmerston North.

Chapter 3 The Duration-controlled grazing system and its impact on pasture accumulation and cow intakes

3.1 Introduction

The dairy industry provides New Zealand with over 20% of its export income (Ministry of Agriculture and Forestry, 2011). In 2012, the national dairy herd was estimated to be 4.6 million cows in milk, occupying 1.64 million hectares. In the last 20 years, cow numbers have increased 89% (Livestock Improvement Corporation and DairyNZ, 2011; Ministry of Agriculture and Forestry, 2011; Livestock Improvement Corporation and DairyNZ, 2012) and the national output of milksolids production has more than doubled. Much of this increase has occurred either through the conversion of sheep and beef farms to dairy farms (particularly in the South Island of New Zealand), the conversion of forestry to dairy farms (in the central North Island) or through intensification of existing dairy farms. Cow numbers per hectare have increased by ca. 15% mainly through greater feeding of purchased off-farm supplements such as cereal grains, maize silage and palm kernel extract, and increased nitrogen (N) fertiliser use (Livestock Improvement Corporation and DairyNZ, 2011). Intensification per hectare from increased supplement use increases farm nutrient loadings which can lead to surplus nutrients leaving the farm in drainage or runoff (Ledgard *et al.*, 2006).

The adverse impact of intensive dairying on surface and ground water quality is of concern to both regional and national government (Monaghan *et al.*, 2008; Parliamentary Commissioner for the Environment, 2012). In addition, the major milk processors are concerned that deteriorating water quality will impact negatively on New Zealand's 'clean green' marketing strategies (Fonterra, 2012). DairyNZ (the national on-farm research body, which is funded by levy payments based on milk production), and the New Zealand government have commissioned research into innovative farm management practices that are able to reduce the adverse impact of intensive dairying on water quality, whilst maintaining or even increasing the productivity of farms.

The major water quality concerns are N leaching through drainage water and phosphorus (P) loss via surface runoff. These pollutants contribute to eutrophication of streams and rivers (Sharpley and Syers, 1979). The majority of N losses from dairy farms can be attributed to excessive N loads (equivalent to 600-1000 kg N/ha) (Haynes and Williams, 1993; Di and Cameron, 2002) in the urine patches that are deposited by cows during grazing (Silva *et al.*, 1999; Ledgard and Menneer, 2005). In most areas of New Zealand annual pasture growth typically ranges from 9 - 17 t DM/ha/yr (Holmes *et al.*, 2007), and N uptake from 180 – 680 kg N/ha/yr, based on an average 2-4% N content in pasture (Ball and Field, 1982). This is insufficient to utilise all N in urine patches. The excess mineral N in the soil profile, primarily nitrate (NO_3^-), is prone to leaching when drainage occurs (Eriksen *et al.*, 1999; Silva *et al.*, 1999; de Klein, 2001; Di and Cameron, 2002; Ledgard and Menneer, 2005), which is mainly over winter and early spring (i.e. May – October).

Phosphorus and faecal microbe losses are predominantly through surface runoff (Collins *et al.*, 2005; Muirhead *et al.*, 2005). Dung pats are the main source of P and faecal microbe return to pasture (Haynes and Williams, 1993), and hence contribute to the susceptibility of P and faecal microbes to loss through runoff (McDowell *et al.*, 2006). Additionally, soil types that are typically wet in winter periods are prone to treading damage and soil surface runoff. As P is strongly bound to hydrous oxides on fine soil particles (Ryden and Syers, 1975), the likelihood of P losses from fine-textured, wetter soils is greater.

Various management strategies are being researched and/or implemented to assist with reducing nutrient and faecal microbe losses (Collins *et al.*, 2005), while maintaining on-farm productivity (Monaghan *et al.*, 2007). An obvious management strategy is to reduce the urine and dung deposition to pastures, which has been well documented as the main driver of nutrient losses from a pastoral grazed system (Sharpley and Syers, 1979; Haynes and Williams, 1993; Ledgard and Menneer, 2005), and also reduce compaction and treading damage of soils to decrease nutrient losses through both drainage and surface runoff. One such strategy is Duration-controlled grazing (*DC* grazing), a practice which involves removing cows from the pasture for a large proportion of time. This system has a number of potential benefits. Although the majority of the cows' diet is pasture, with shorter grazing periods they do not deposit as much urine and dung on paddocks, thereby reducing nutrient losses (de Klein, 2001; de Klein and Ledgard, 2001; Oudshoorn *et al.*, 2008). Furthermore, with shorter grazing intervals, damage to soils and pasture should be markedly reduced.

While it may be obvious that depositing less urine and dung to pasture will reduce nutrient losses, there is a concern that less excreta returns (de Klein, 2001) will lower the amounts of nutrient available for pasture growth (Ledgard *et al.*, 1982). The excreta captured from the stand-off facility used for *DC* grazing should be returned to pasture to retain a balance of nutrients in the system (Monaghan *et al.*, 2008). There is also a risk that increased use of *DC* grazing and associated cow housing will increase the cost of milk production in New Zealand by decreasing pasture intakes and requiring greater feeding of expensive brought-in supplements (Laven and Holmes, 2008).

A three year grazing plot study has been conducted to provide longer-term information on the impacts of *DC* grazing of dairy cows on pasture production, drainage, and runoff water quality. The objectives of this chapter are to present: (i) the methods used for grazing and excreta management and (ii) the comparison of pasture accumulation and estimated pasture intakes by cows in standard and *DC* grazed systems.

3.2 Methods

3.2.1 Trial site

The experiment was established in 2008 on Massey University's No. 4 dairy farm near Palmerston North, Manawatu, New Zealand (40° 23' 46.79" S; 175° 36' 35.77" E). The average annual rainfall is ca. 1000 mm. Average monthly soil temperatures at 10 cm are 7°C in July and 18.1°C in January. The farm, typical of the region, which is 211 ha in area, operates as a system 3-4 (DairyNZ, 2010) seasonal (lactation period from late July to early May) milk supply dairy farm, milking ca. 580 Friesian and Friesian/Jersey cross cows. The cow diet is pasture-based with supplements buffering feed supply when seasonal pasture growth does not meet feed demand (ca. 75% pasture, 25% supplements). The farm is subdivided into 98 paddocks, which have an average area of ca. 2.3 ha. The experimental site was located in one of these paddocks with a slope of <3%. The trial was established on a mole-pipe drained Tokomaru silt loam soil, which is classified as an Argillic-fragic Perch-gley Pallic Soil (Hewitt, 1998). The No. 4 dairy farm was grazed by dairy cattle for ca. 30 years prior to this study taking place (Houlbrooke *et al.*, 2008).

The research area consisted of 14 plots (average area of ca. 850 m^2/plot), each with an isolated mole and pipe drain system (Bowler, 1980). Further details of the topography and soil properties of the site are given by Houlbrooke *et al.* (2004) and Pollok (1975). The plots supported a mixed pasture sward of dominantly perennial ryegrass (*Lolium perenne*) and white clover (*Trifolium repens*).

3.2.2 Experimental design

The 14 trial plots were divided into 7 replicates of two treatments: Standard grazing (*SG*), involving a grazing duration of ca. 7 hours for day grazings and ca. 12 hours for night grazings, and Duration-controlled grazing (*DC*), involving a grazing duration of ca. 4 hours for both day and night grazings (Table 3.1 and Plate 3.1). There was no difference in time spent on races between the two treatments.

Table 3.1: Approximate schedule of daily activity for standard (*SG*) and duration-controlled (*DC*) grazing treatments.

	5-7am	***7-11am***	***11am-2pm***	***2-4pm***	***4-8pm***	***8pm-5am***
DC	Milking	Grazing	Standoff / Housing	Milking	Grazing	Standoff / Housing
SG	Milking	Grazing		Milking	Grazing	

The *SG* treatment was managed as ‘best-practice’ standard grazing, which means grazings were timed to minimise treading damage. This was achieved by ensuring that the soil moisture deficit was at least 5 mm at each grazing, for both treatments (Horne, 2012).

Plate 3.1: Layout of the fourteen ca. 850 m^2, experimental plots (those marked with '*DC*' denote Duration-controlled grazing plots, with the remainder being standard grazed plots; dots denote corners of 50 m^2 runoff plots)

The target feed intake during the lactation season was 16 kg DM/cow/day. Both sets of treatment plots were grazed on the same day with the same average stocking intensity (seasonally ranging from 20 to 30 cows per plot), which was set on the basis of pasture mass estimated using a rising-plate pasture height meter (Jenquip, Feilding, New Zealand). Prior to each grazing the *SG* cows were fed 2-3 kg DM/cow as supplementary feed on a feed pad and then grazed on the *SG* plots to obtain a further 5-6 kg DM/cow. The *DC* cows first received 5-6 kg DM/cow from grazing *DC* plots and then after 4 hours were removed to the feed pad or another area of pasture to simulate their return from grazing to a standoff or housing facility. Here, the *DC* cows received the remainder of their feed requirements (ca. 2-3 kg DM/cow of either pasture or supplement).

Grazings were alternated between 'day' and 'night' grazings to create the annual difference in grazing duration between the two treatments, with a total of 8-10 grazings per year. One 'dry cow' grazing occurred in June or July each year, after the cows had been dried off from lactation. On this occasion, the *SG* and *DC* plots were grazed for 24 and 8 hours, respectively. For these grazings, the targeted pasture intake for each treatment was 8 kg DM/cow/day. Results from the first three years (September 2008 – September 2011) of this study are reported in this chapter.

3.2.3 Estimated pasture dry matter accumulation, herbage quality and cow intakes

A rising-plate pasture meter was used pre- and post-grazing to estimate pasture mass (DM) on all plots at each grazing. Sixty rising-plate measurements were taken per plot. To convert height readings to pasture mass, the 'winter' calibration equation (Compressed height (1/2 cm) x 140 + 500; DairyNZ, 2010) was used for the whole year. Pasture accumulation between grazings was calculated as the difference between pasture mass at a grazing minus the post-grazing pasture mass of the previous grazing (Figure 3.1). Estimated cow intakes were calculated as the difference between pre- and post-grazing pasture mass at each grazing (Abrahamse *et al.*, 2008). Pasture mass was also measured weekly over the entire No. 4 Dairy Farm as part of farm monitoring to assess pasture growth rates for each paddock.

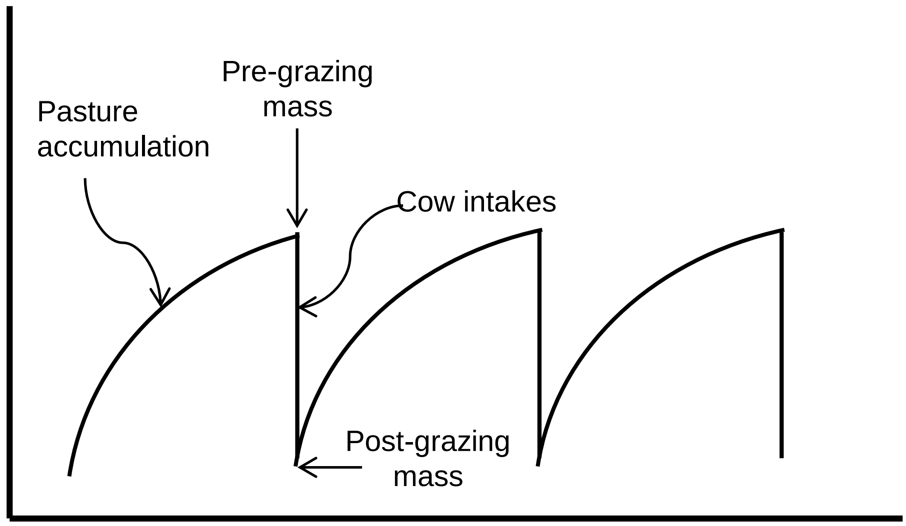

Figure 3.1: Schematic diagram showing measurements taken for each parameter of pasture accumulation and estimated pasture intakes by cows.

The concentration of nutrients in the pasture was measured by collecting 40 hand-plucked samples from each plot prior to each grazing (from January 2009), avoiding herbage directly surrounding dung pats and urine patches. These bulk samples were oven-dried to 60°C, ground to 1 mm, and then underwent basic elemental analysis (Hill Laboratories Ltd, Hamilton, New Zealand).

3.2.4 Soil fertility and fertilisers

All plots were individually soil tested three times throughout the duration of the experiment: at the start of the study (September 2008) and on two other occasions (May 2010 and May 2012) (Table 3.2). Fifteen samples were taken from each plot, at a depth of 7.5 cm. These samples were analysed for pH, Olsen P (µg/g soil), sulphate sulphur (S) (µg/g soil), potassium (K) (quick test (QT)), calcium (Ca) (QT) and magnesium (Mg) (QT). Olsen P levels were not significantly different (P>0.05) between treatments at the beginning of the study. Sulphate S was lower in the first test than in subsequent tests, although this was most probably due to the timing of soil sampling relative to fertiliser application. There was a significant difference (P<0.05) in QT K levels between treatments by May 2010, and this difference will be explored further in Chapter 7.

Table 3.2: Soil test results for the trial site in 2008, 2010 and 2012, averaged for both treatments (Duration-controlled grazing (*DC*) and standard grazing (*SG*)).

	pH		Olsen P (µg/g soil)		Sulphate S (µg/g soil)		QT K		QT Ca		QT Mg	
Soil test date	*DC*	*SG*	*DC*	*SG*	*DC*	*SG*	*DC*	*SG*	*DC*	*SG*	*DC*	*SG*
Sept 2008			32.9	30.3	5.0	5.3						
May 2010	5.8	5.8	30.8	29.7	8.5	12.2	2.7	4.6	12.4	12.2	30.5	30.9
May 2012	5.7	5.7	37.1	39.5	34.3	21.1	4.4	7.1	8.3	7.9	27.8	28.8

The timing and rates of fertiliser applied during the study site are shown in Table 3.3. Both sets of treatment plots received the same fertiliser application rates.

Table 3.3: Fertiliser applications to all plots at the trial site from September 2008 to September 2011.

Fertiliser	2008	2009	2010	2011
Urea (kg N/ha) 46:0:0:0	30 *(September)* 30 *(November)*	25 *(August)* 20 *(November)*	25 *(April)*	
Ammonium Sulphate (kg N/ha) 21:0:0:24		30 *(September)*	30 *(September)* 25 *(October)*	
Superphosphate (kg P/ha) 0:9:0:11	30 *(November)*			
30% Sulphur Superphosphate (kg/ha) 0:7:0:30			200 *(April)*	
Cropmaster 20 (kg/ha) 19:10:0:12				150 *(March)*

3.2.5 Soil water balance

A soil water balance, as described by Scotter *et al.* (1979) was used to predict when drainage would occur. Climate data were sourced from the National Institute of Water and Atmospheric Research (NIWA) site which was ca. 3 km from the research area, except for rainfall data which were measured at a weather station on No. 4 dairy farm, using a Davis II automatic tipping bucket rain gauge. The potential evapotranspiration was calculated for each season using the FAO-56 method as described by Allen *et al.* (1998).

3.2.6 Slurry application to Duration-controlled grazing plots to balance nutrient return

Dung pats were counted on each plot after each grazing in the 2008/2009 lactation season. As dung and urine are returned at similar rates (Haynes and Williams, 1993), the number of dung pats were used to estimate the total amount of excreta returned to the plots. Rate of excretion (dung pats/cow/hour) was also estimated by calculating the number of dung pats deposited in the time that cows had spent on the plots at each grazing. Using average volumes and N concentrations for dung and urine from Haynes and Williams (1993), the amount of N deposited in the plots was estimated (Table 3.4). Nitrogen returns to the plots were also simulated with the model of Salazar *et al.* (2010), using the average N concentrations in pasture and grass silage, the time spent on the different treatment plots, and average product and transfer losses. The return of N via dung and urine to the plots using this model was similar to those estimated in Table 3.4, and is detailed in Chapter 4.

The difference in the number of dung pats between the *SG* and *DC* treatments was used to estimate the amount of dung and urine that the *DC* cows would have deposited on the farm's feed pad. An equivalent amount of slurry was applied to the *DC* plots. During storage of the slurry, a loss of 45% of the original collected excreta N was estimated (Longhurst *et al.*, 2006). The slurry was sourced from the farm's feed pad bunker and farm dairy effluent ponds, and applied using a Williams 'Elephant 5000' slurry tanker (5000 L applied per plot) (Table 3.4 and Plate 3.2). When slurry was applied, a plastic collection tray was placed on each plot to collect 2 x 100 mL samples for analysis of the concentration of total N and P, nitrate, ammonium and dissolved reactive phosphate (DRP), and cations. From this, the amount of each nutrient applied at each application was estimated (N presented in Table 3.4). More details of N, P, faecal microbe and K applications and depositions will be given in Chapters 4, 5, 6 and 7, respectively.

Table 3.4: Amount of nitrogen (N) returned (kg/ha) to *DC* plots in slurry applications, the estimated annual return of N in dung and urine to *DC* and *SG* treatments using dung pat counts, and the reduction in N returned to *DC* plots.

Year	Time of slurry application	Amount total N applied to *DC* plots *(kg/ha)* as slurry[1]	Amount mineral N applied to *DC* plots *(kg/ha)* as slurry	% total N applied as mineral N	Amount N estimated returned as dung and urine *(kg/ha/yr)*[2]		Reduction in N returned to *DC* plots compared with *SG* plots *(kg N/ha/yr)*
					DC	***SG***	
2008/09	Dec 08	212	35.3	16.6	173	318	-67
2009/10					161	297	136
2010/11	Oct 10	15	7.4	50.1	185	339	39
	Nov 10	11	5.1	44.5			
	Feb 11	68	21.1	31.1			
	Apr 11	21	12.1	58.0			

[1] Amount N applied to plots (kg/ha) = N concentration (ppm) x depth (mm) x 10
[2] Amount N estimated returned as dung and urine (kg/ha/yr) = Number of dung pats/ha x 2 x N concentration x volume (Haynes & Williams 1993)

One application of slurry was applied to the *DC* treatment plots at an application depth of 5-10 mm in mid-December 2008 (Chapter 4). This was using 5/6 of a tank of feedpad bunker material, with the remaining 1/6 being sourced from the farm dairy effluent ponds. This applied 67 kg/ha more N than what was required to make up the annual difference between treatments (Table 3.4). Therefore no applications of slurry were made in the 2009/2010 lactation season. Slurry was spread on the *DC* plots four times during the 2010/11 lactation season; each application an average of ca. 6 mm depth, relating to 115 kg N/ha. For these applications, half the material was from the feedpad bunker, and the other half sourced from the farm dairy effluent ponds. In addition, 1.17 kg dissolved granulated urea was added to each tank (1 tank per plot) during filling, to ensure a mineral N content of approximately 40% of total N (TN) was achieved (Chapter 4), and 4.8 kg dissolved granulated KCl was also applied, to increase K levels to what would normally be collected in a standoff facility (Chapter 7).

Plate 3.2: Applying slurry to Duration-controlled grazing plots, using a Williams 'Elephant 5000' slurry tanker, October 2010.

3.2.7 Effect of climate on cow intakes

The heat load index (HLI) was calculated for the duration of all grazings throughout the trial period (Bryant *et al.*, 2010). This was based on the maximum daily air temperature, wind speed, solar radiation, and relative humidity recorded from the NIWA weather station, located ca. 3 km from the experimental site, and was used to investigate the effects of heat load on the actual intakes of the cows while grazing.

3.2.8 Statistical analysis

Pasture accumulation (kg DM/ha) between each grazing and cow pasture intakes at each grazing, were analysed using a generalised linear model procedure (PROCGLM), within the statistical package SAS v 9.1 (SAS Institute Inc., Cary NC, USA) (SAS, 2003). Significance was established using the Kruskal-Wallis test followed by multiple Bonferroni (Dunn) t-tests.

3.3 Results

3.3.1 Climatic conditions

The annual rainfall was 1245, 1032 and 1220 mm/yr for the 2008/09, 2009/10 and 2010/11 years (September to September), respectively (Figure 3.2). These were higher than the 30 year average of 969 mm for Palmerston North. Although the pattern of soil drying varied between the years, the cumulative actual evapotranspiration was similar among all years, being 771, 756 and 722 mm for the 2008/09, 2009/10 and 2010/11 years, respectively.

3.3.2 Pasture accumulation and grazing treatments

Pasture accumulation (Figure 3.3) was not statistically different ($P>0.05$) between treatments for the first four grazings to December 2008 (6560 c.f. 6724 kg DM/ha for *DC* and *SG* respectively). The first application of slurry (average 212 kg N/ha) to the *DC* treatment plots in late December 2008 caused a small but significant ($P<0.05$) increase in pasture accumulation on these plots in the subsequent measurement periods until April 2009. However, for the following grazings, the *SG* treatment showed a greater average accumulation rate than the *DC* treatment. During the first year (2008/09) the *SG* plots accumulated an average of 15213 kg DM/ha, which was similar ($P>0.05$) to the *DC* plots of 14616 kg DM/ha (Figure 3.3).

In the 2009/10 year, annual pasture accumulation on *SG* plots (12778 kg DM/ha), was lower than in the previous year. Compared to the *SG* treatment, the use of *DC* grazing resulted in a 20% lower ($P<0.05$) annual pasture accumulation by the end of the 2009/10 year (Figure 3.3).

In the third year of the study (2010/11), annual pasture accumulation was not statistically different between the two treatments (14990 for *SG* and 13619 for *DC*), although there was statistical evidence ($P<0.05$) for the *SG* treatment having higher accumulation compared to the *DC* treatment between some individual grazings (Figure 3.3).

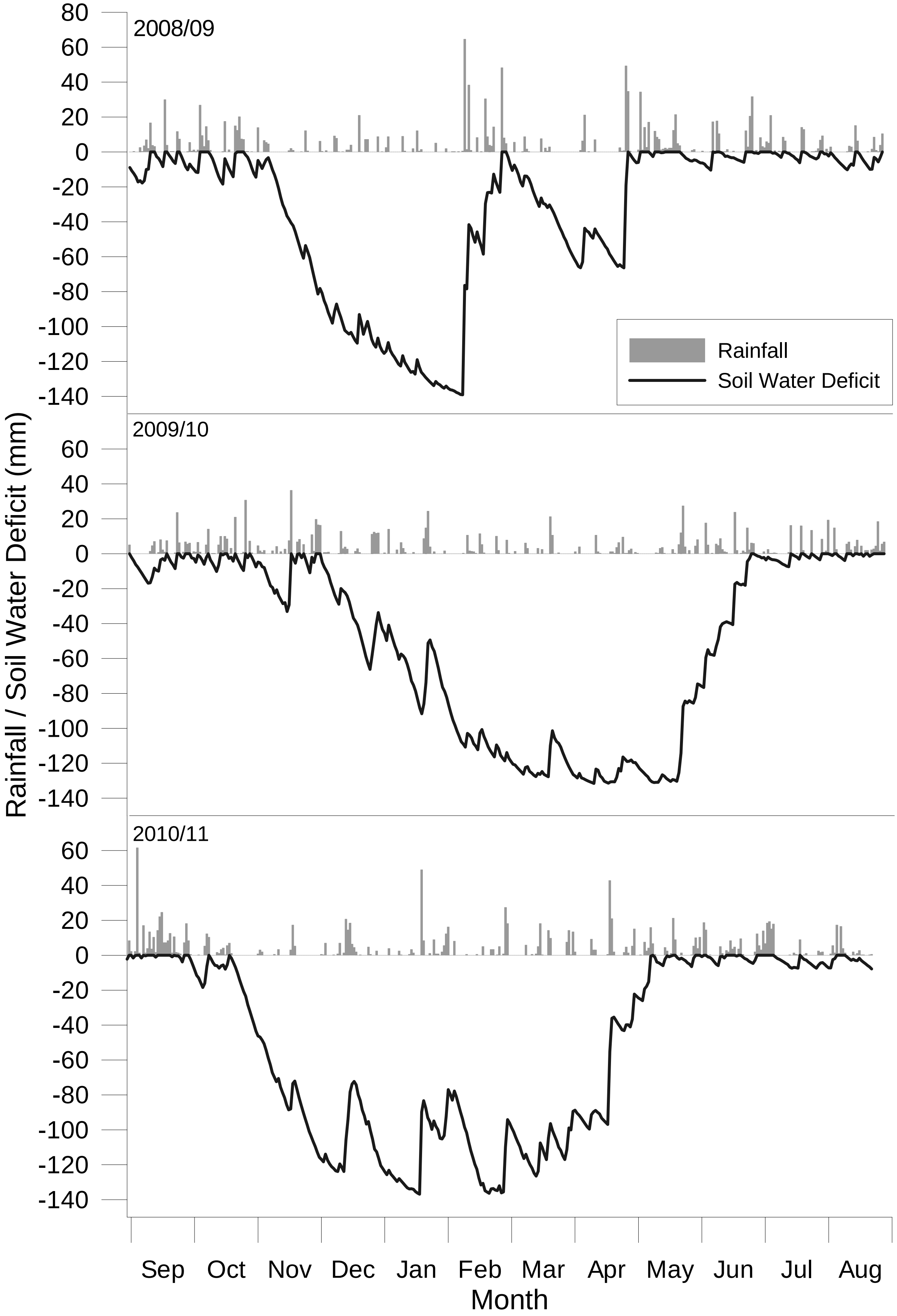

Figure 3.2: Daily rainfall (mm) and soil water deficit (mm) at No. 4 Dairy Farm from September 2008 to September 2011.

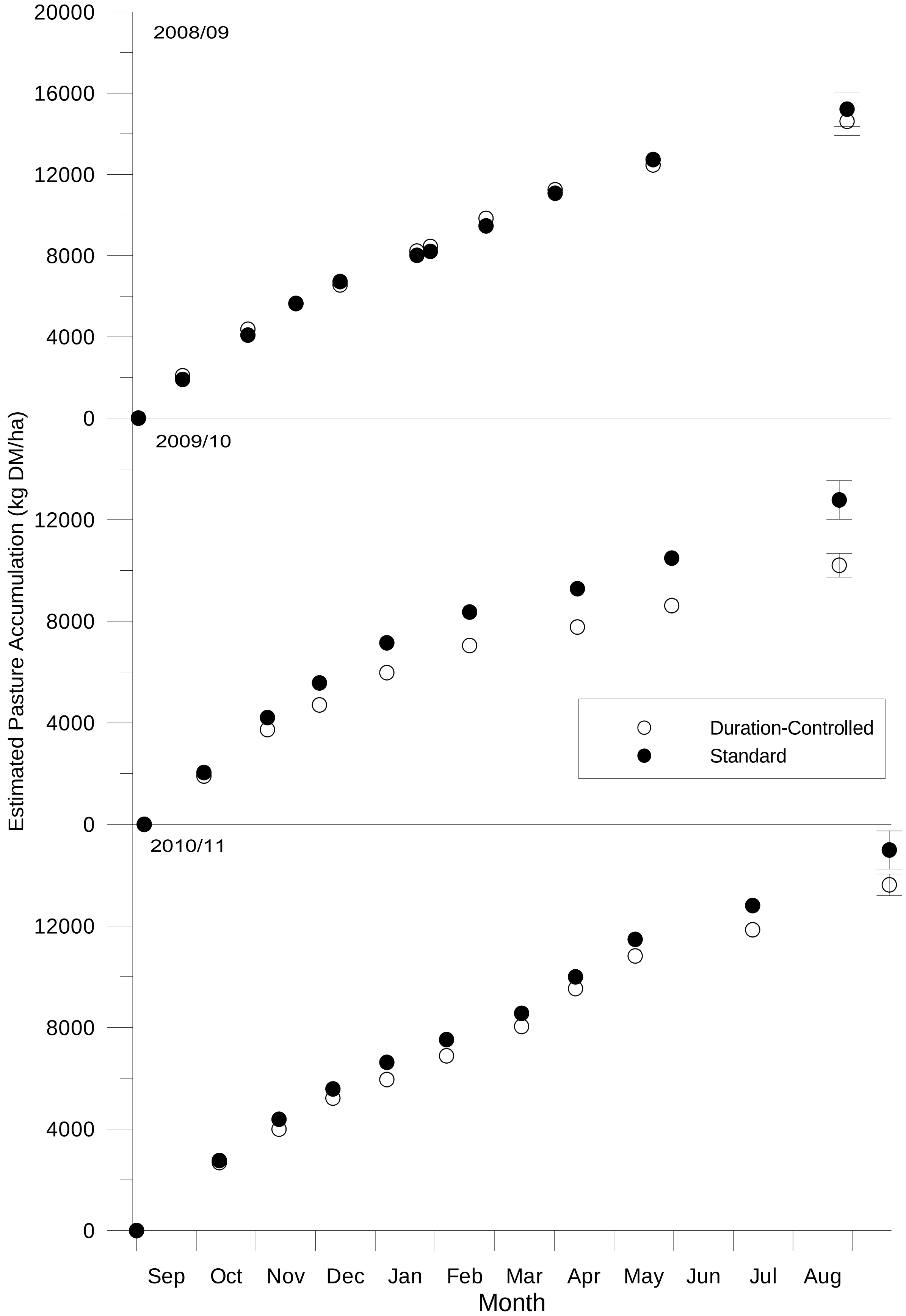

Figure 3.3: Estimated pasture accumulation (kg DM/ha) for each study year, from September 2008 to September 2011. Error bars represent standard error for the total accumulation period.

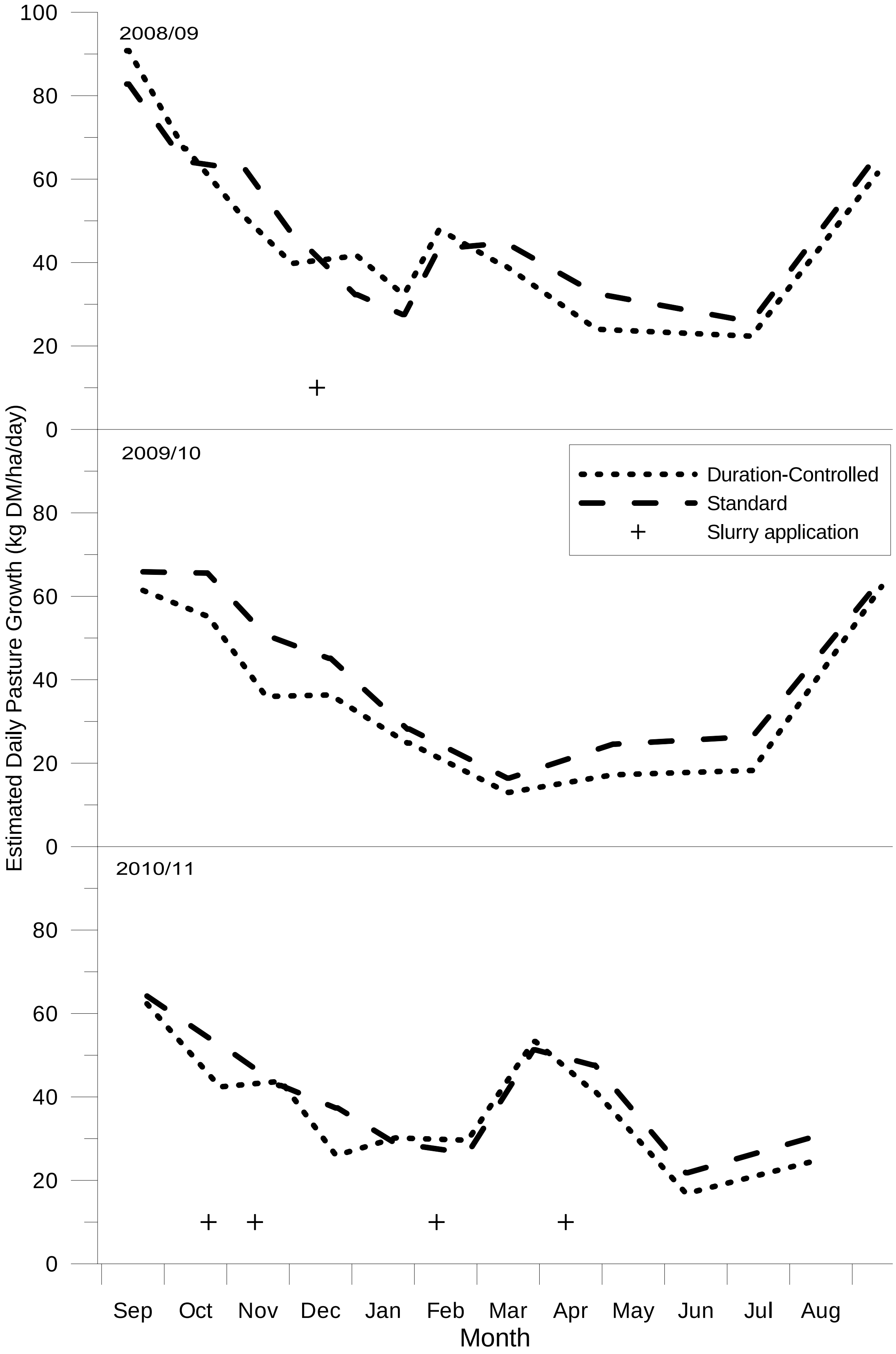

Figure 3.4: Daily average pasture growth rates (kg DM/ha/day) for each study year, from September 2008 to September 2011, for rotationally grazed plots on the two treatments (*DC* and *SG*). Timings of slurry applications to *DC* plots are denoted by black crosses.

3.3.3 Treatment effect on estimated cow intakes

Similar pasture accumulation on the *SG* and *DC* treatments in 2008/09 allowed the same stocking rates between treatments at each grazing (Table 3.5). Estimated cow intakes were similar ($P>0.05$) for the two treatments when averaged over the entire year (Table 3.6). The average pasture intakes were 5-6 kg DM/cow/grazing for both treatments. The season 24-hour grazing (8 hours for *DC* treatment plots) target of 8 kg DM/cow was reached or almost reached for both treatments in all years (Table 3.6).

Table 3.5: Average stocking intensity (cows/ha) and dung pat deposition (per ha and per cow/hour) for *DC* and *SG* grazing treatments in 2008/09; '24hr' denotes 8hr grazing for *DC*, 24hr grazing for *SG*.

Grazing / Time of day	Treatment	Cows/ha	Dung pats/ha	Dung pats/cow/hour
1 Day	*DC*	270	721	0.67
	SG	270	1085	0.61
2 Night	*DC*	350	808	0.56
	SG	350	1689	0.39
3 Day	*DC*	367	813	0.60
	SG	367	1116	0.51
4 Night	*DC*	293	481	0.42
	SG	293	1328	0.37
5 Day	*DC*	226	483	0.52
	SG	226	834	0.50
6 Night	*DC*	372	1046	0.64
	SG	372	1607	0.34
7 Day	*DC*	282	765	0.68
	SG	282	1246	0.60
8 Night	*DC*	270	709	0.66
	SG	270	1755	0.51
9 24hr	*DC*	206	1037	0.65
	SG	206	1958	0.43
Average	*DC*		762	0.60
	SG		1402	0.47

Table 3.6: Average cow pasture intakes (kg DM/cow) for each grazing period in each season, and range of intakes for each season; '24hr' denotes 8hr grazings for *DC*, 24hr grazings for *SG*.

		Pasture intake per cow (kg DM/grazing)		
	Treatment	**2008/09**	**2009/10**	**2010/11**
Day	*DC*	5.3	4.3	4.5
	SG	5.7	5.1	5.5
Night	*DC*	5.0	4.6	5.1
	SG	5.0	5.7	5.2
Range	*DC*	4.0-6.4	3.6-5.5	3.8-6.0
	SG	3.2-6.6	4.2-7.2	3.9-7.4
24hr	*DC*	8.7	8.2	10.1
	SG	7.9	10.1	13.3

In 2009/10, average pre-grazing pasture mass (kg DM/ha) and estimated pasture intakes by cows were lower on *DC* plots at most grazings, compared to the *SG* plots (Figures 3.3 and 3.4, and Table 3.6). There was an 18% reduction in estimated pasture intakes for the *DC* treatment in 2009/10, and a 15% reduction for 2010/11 (Table 3.6).

3.4 Discussion

3.4.1 Pasture accumulation - farm vs. plots

Average pasture accumulation for all three study years was 14327 kg DM/ha/year for the *SG* treatment and 12813 kg DM/ha/year for the *DC* treatment. Both treatment values were higher than that observed and measured on the No. 4 Dairy Farm as a whole (average 11.4 t DM/ha/yr; Taylor 2012, Pers. Comm.). As overall soil fertility and pasture species were similar for the farm and the experimental area, the most likely explanation for the larger accumulation on the experimental plots is that they were grazed only when the soil moisture deficit was > 5 mm to avoid treading damage, whereas the farm did not have adequate facilities to follow this criterion and at times paddocks were grazed when the soil was at or close to field capacity, resulting in treading damage (Horne and Singleton, 1997; Pande *et al.*, 2000).

3.4.2 Pasture accumulation – annual differences

Pasture accumulation from winter to early summer was similar between years. However, there were obvious and significant differences in pasture accumulation rates between years during the summer-autumn period, and so further discussion will be focussed on this part of the year. Larger and more frequent rainfall events in April and May of 2011 (Figure 3.2) reduced the soil moisture deficits and increased daily pasture accumulation rates on both treatments, compared to the previous years (Figure 3.4; Allen *et al.* (1998)). This accelerated daily pasture growth occurred earlier in 2008/09 but did not occur in 2009/10. For instance, the usual ‘autumn flush’ of pasture growth (Holmes *et al.*, 2007) did not occur in autumn 2010 and daily pasture growth rates were small (<30 kg DM/day) from summer through to winter on both treatments. This occurred even though evapotranspiration was similar to the other years.

The rapid increase in pasture accumulation that followed the rewetting of the soil in the autumn of the 2010/11 season (Figure 3.4) was likely due, in part, to the stimulation of net mineralisation of N which was encouraged by warm soil temperatures. On the contrary, the manner in which the soil rewetted in 2009/2010 (Figure 3.2) did not appear to stimulate N mineralisation. It is likely that it was too cold by the time rewetting occurred, which caused lower N mineralisation and pasture growth. This hypothesis is supported by the relatively small quantity of N that was leached in the subsequent drainage season (Chapter 4).

The difference in pasture growth rates between years did not seem to be solely related to water availability because evapotranspiration was the same between years. Therefore it was important to investigate whether a nutrient availability factor was influencing pasture growth, both between years and between treatments.

3.4.3 Pasture accumulation rates – Duration-controlled vs. Standard grazed

Much of the increased pasture accumulation on the *SG* plots compared with the *DC* plots was due to the greater N return from urine to this treatment, as a result of the cows spending longer grazing these plots. In addition, the single slurry application to the *DC* plots in 2008 was not sufficient to increase pasture accumulation on those plots on an annual basis. Increased K returns probably also had an increasing effect on pasture accumulation on the *SG* plots. The *SG* plots, at any one time, had a larger area covered in dung and urine patches, which showed evidence of growing disproportionately more grass compared to inter-dung and urine areas. This observation was consistent with previous studies, where N, and, in some cases K, were contributing directly to greater pasture growth responses under urine deposition (During and McNaught, 1961; Auerswald *et al.*, 2010; Moir *et al.*, 2011). Further discussion on the effect of K in slurry applications will be found in Chapter 7.

Nitrogen concentrations in herbage at all grazings (results not shown) were not significantly different ($P>0.05$) between treatments, but varied significantly ($P<0.05$) between years, and were within the range reported of ca. 2-5% for ryegrass/white clover pasture in a grazing preference study conducted in the Manawatu region of New Zealand (Cosgrove *et al.*, 2002). The lower overall pasture accumulation for 2009/10 may also be attributed to a lower number of grazing days for that season (8 c.f. 9 for 2008/09 and 10 for 2010/11), and thus less urine and dung deposition, compounding to produce the lower pasture accumulation from less N being returned and thus available for plant uptake. This was evident in N concentrations in the herbage for 2009/10, where both treatments had lower N than in 2008/09 and 2010/11 ($P<0.05$), and also lower N leaching in 2010; all of these effects will be discussed in more detail in Chapter 4.

Based on the time spent grazing, it was estimated that, on average, 77% of the N ingested by *SG* treatment cows was returned to the paddock in urine and dung combined. This value is similar to the value given by Haynes and Williams (1993), of 80% for dung and urine combined, but greater than the 52% return calculated using the model of Salazar *et al.* (2010). It was estimated that in 2008/09, 2009/10 and 2010/11 ca. 318, 297 and 339 kg N/ha/yr, respectively, were deposited in excreta on the *SG* plots (Table 3.4). In comparison, an average of only 47% of the N ingested by *DC* treatment cows was returned to the plots in urine and dung. Therefore, in 2008/09, 2009/10 and 2010/11 ca. 385, 161 and 300 kg N/ha/yr were returned on average to the *DC* plots in excreta plus slurry applications, respectively. These estimates suggest that on average, 36 kg N/ha/yr less was returned to the *DC* treatment.

Studies in Europe have also included concepts around *DC* grazing, not only to monitor cow intakes and grazing residuals (‘restricted access time’) (Kennedy *et al.*, 2009), but also to reduce N leaching (Oudshoorn *et al.*, 2008). For example, Oudshoorn *et al.* (2008) used ‘time-limited’ grazing to study urination frequency and spatial distribution of urine. They compared the effects of grazing times of 4, 6.5 and 9 hours per day on urine frequency, and found that frequency did not change with increasing time spent grazing. Therefore urine deposition throughout the day was unaffected by whether cows were being grazed or stood off. This was also consistent with the present study, where the reduction in time spent grazing on the *DC* treatment was related to the reduction in NO_3^- leached, which had a direct relationship with the number of urine patches deposited to the plots (Chapter 4). However in this study, the rate of defecation (dung pats/cow/hour) was slightly lower at night grazings than during the day (0.49 c.f. 0.59 dung pats/cow/hour for night and day grazings in 2008/09, respectively) (Table 3.5). This was mainly due to cows spending more time lying down during night grazings (Draganova *et al.*, 2010).

There was a significant, albeit small and short-lived, increase in pasture production on the *DC* plots following the first slurry application in December 2008 (Figure 3.4). This response may have been small because the slurry that was applied was lower in mineral N (16.6%) than typical slurry from a cow house (ca. 40%, (Houlbrooke *et al.*, 2011)). There was also a relatively high proportion of feed matter in the slurry being applied, and therefore a high carbon content (Clough and Kelliher, 2005), which had the potential to encourage immobilisation of mineral N by transformation into organic forms. In addition, the increased carbon content of the slurry may have led to more N loss by denitrification

(Barton and Schipper, 2001), particularly in the warm wet spring conditions of 2009 that followed.

In 2010/11 the slurry was applied in smaller amounts, with a higher mineral N content (Table 3.4). This mineral N content, with an average of 39.7% of total N, was more comparable to that reported by Houlbrooke *et al.* (2011) for slurry analysed from cow houses (38.7% and 43.3% for HerdHome® and wintering barn slurry, respectively). Slurry application had a positive effect on pasture growth in this year, and this was particularly evident when comparing daily average accumulation rates between treatments (Figure 3.4). Daily pasture growth rates improved on the *DC* plots immediately following slurry application in 2008 and the first 3 applications in 2010/2011. The increased response in growth rates only lasted for one grazing rotation, however. Furthermore, to be effective, the more frequent applications of slurry must occur when soils are moist and temperatures permit pasture growth to actively respond to the addition of nutrients. This was also evident in an earlier study at a nearby site, where the effect of dairy farm effluent application on pasture growth was measured (Bolan *et al.*, 2004). In that study, increasing rates of N via effluent increased pasture response, although it was not possible to distinguish between water and nutrient effects. In the present study, it was evident that single slurry applications did not have long-lasting effects on pasture growth, as demonstrated by the decline in accumulation relative to *SG* plots following the last application in April 2011 to the end of measurements in September 2011 (Figure 3.4), and the lack of a larger response to the first application which added N at such a high rate.

3.4.4 Pasture intake by cows

Within the range of pasture allocation in this study, the intake of pasture per cow at each grazing was linearly related to the amount of accumulated pasture mass allocated per cow, which explained 58% (R^2=0.58) of the variation in actual intakes per cow per grazing (Figure 3.5). There was no relationship between time spent grazing and actual intakes (graph not shown). This was most likely due to the cows having sufficient time to graze in 4 hours on both treatments, before doing other activities such as lying or walking (Draganova *et al.*, 2010). The unexplained variation in intakes, i.e. the residual error of the relationship shown in Figure 3.5, could not be predicted by other known factors such as pre-grazing pasture mass.

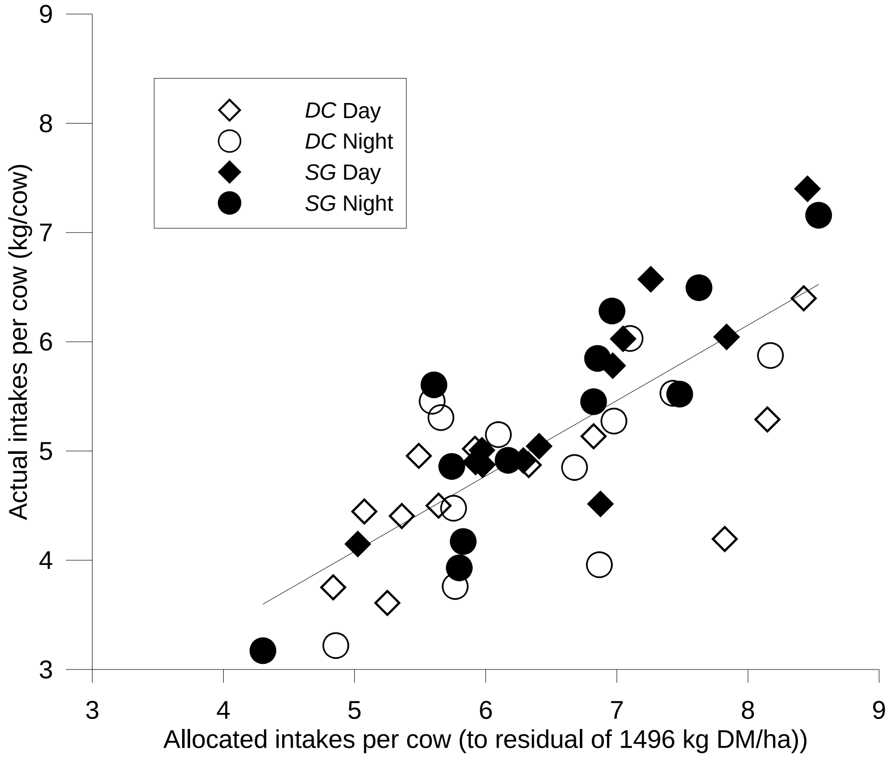

Figure 3.5: Allocated pasture per cow per grazing (accumulated pre-grazing pasture mass – lowest post-grazing pasture mass measured of 1496 kg DM/ha) plotted against the actual pasture intakes per cow per grazing for all grazings in all years. Regression line encompasses every grazing from both treatments.

The cows grazing the *DC* treatment plots were mostly able to ingest the quantity of pasture that was allocated to them (up to 6.4 kg DM, Table 3.6) in the 4 hour grazing period. In the 2009/10 lactation season however, when there was 20% lower pasture accumulation over the year, there were grazing events where the cows did not receive their targeted 5-6 kg DM. This was mostly on the *DC* grazing plots, although there were times when cows grazing the *SG* plots also did not receive their target intakes. This was generally due to the cows not grazing down to the targeted residual pasture cover of ca. 1500 kg DM/ha at many grazings over all 3 years, which was set according to industry standards and previous research (DairyNZ, 2010; Irvine *et al.*, 2010).

It is not known why, when given sufficient pasture to achieve targeted intakes, the cows did not graze to the targeted residual. Some explanation may be due to the climate during grazing. There is some suggestion that pasture intakes decreased with increasing HLI at grazing (Figure 3.6). However, this does not account for the residual variation in cow intake in Figure 3.5 because no trend was observed when HLI was plotted against the deviation from the former relationship. On very hot grazing days (maximum air temperature in excess of 25°C), it was apparent that the actual intakes of cows were much lower than what they were allocated. This is in line with work by Ravagnolo *et al.* (2000), where a review of 134 American farms over 8 years showed that pasture intakes and hence milk production decreased with a temperature-humidity index (THI) over 72 (equivalent to 25°C and 50% relative humidity). This was also seen in a similar New Zealand modelling exercise, where production parameters were measured against THI, although in this case, humidity was higher (60-70%) at similar temperatures (Bryant *et al.*, 2007) which is more comparable to New Zealand outdoor grazing conditions.

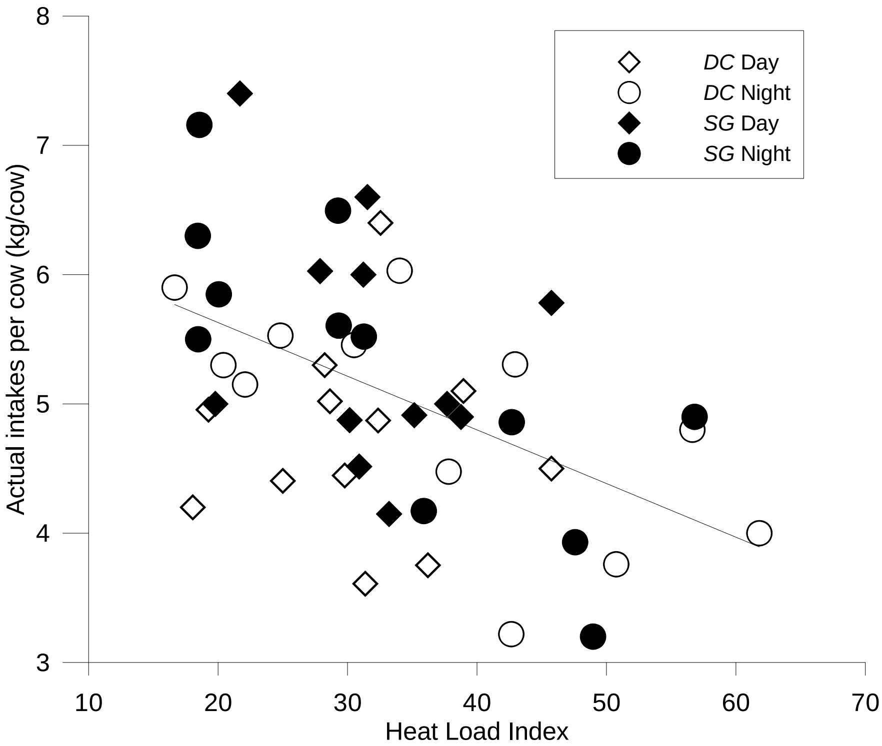

Figure 3.6: Average heat load index for the duration of each grazing plotted against estimated actual intakes per cow (kg dry matter), for every grazing in all years. Regression line emcompasses every grazing from both treatments.

Another possible reason for cows not grazing to the desired residual may have been the higher structural strength of pasture (Hughes *et al.*, 1991), which was in a reproductive state through late spring and summer at the peak of lactation (Irvine *et al.*, 2010). Therefore, it was more difficult for cows to eat and achieve the required intakes during peak lactation in late spring and summer. This may also have been exacerbated by the plate meter recording higher DM yields with drier pasture, and using one calibration equation throughout the entire year.

Having lower stocking intensities on plots may have also resulted in cows avoiding ingesting areas that had dung or urine deposited in previous grazings. Studies of the effects of deposited excreta on grazing behaviour have reported varying conclusions. Dung deposition is a common cause of avoidance or rejection (Marsh and Campling, 1970), while most reports on urine deposition suggest that these areas became more palatable for grazing (Norman and Green, 1958).

In the 2010/11 lactation season, the larger difference in cow intakes between treatments may have been due to a greater number of day grazings, where it was evident that cows grazing the *DC* plots usually ate less during this period than in the night grazings (Table 3.6). Night grazings resulted in intakes between treatments that were more similar (< 2% difference overall) and did not reflect the greater difference in grazing duration between *SG* and *DC* treatments at night. This was probably because after cows had been removed from the *DC* plots, cows on the *SG* plots spent very little time grazing, and instead spent most of their time lying down, as demonstrated by Draganova *et al.* (2010). The 24-hour *SG* grazing (*DC* 8-hr grazing) of dry cows had a large difference between intakes (24% lower for *DC* plots). Notably both intakes on *DC* and *SG* plots were greater than the 8 kg DM allocated for this grazing.

Lower pasture intakes by cows in any year would most likely have resulted in decreased milk production (Baudracco *et al.*, 2010). However, milk production parameters were not measured in this study.

3.5 Conclusion

At times in this three year study, a reduction in the recycling of nutrients in urine, dung and slurry to the *DC* treatment, relative to the *SG* treatment, resulted in lower pasture accumulation rates and subsequent cow DM intakes. This was probably due to lower overall N availability within the *DC* plots. However, it became evident that pasture accumulation, and hence cow pasture intakes, could be markedly improved with careful timing and management of slurry application. The timely re-application of excreta via slurry, at appropriate rates, with a relatively high mineral N content, was important to N dynamics on the *DC* plots. It is recommended that slurry is applied regularly throughout a lactation season when *DC* grazing is practised.

There was no relationship between time spent grazing and estimated intakes. This indicates that generally, cows on the *DC* treatment were able to harvest the allocated pasture in a four-hour grazing period. Care is needed, however, when managing *DC* grazing as for the various reasons that have been discussed, pasture accumulation gains and associated intakes may not be as large as anticipated.

3.6 References

Abrahamse, P.A., Dijkstra, J., Vlaeminck, B., Tamminga, S., 2008. Frequent Allocation of Rotationally Grazed Dairy Cows Changes Grazing Behaviour and Improves Productivity. Journal of Dairy Science 91, 2033-2045.

Allen, R.G., Pereira, L.S., Dirk, R., Smith, M., 1998. Crop evapotranspiration: guidelines for computing crop water requirements. Irrigation and Drainage Paper 56 (FAO: Rome).

Auerswald, K., Mayer, F., Schnyder, H., 2010. Coupling of spatial and temporal pattern of cattle excreta patches on a low intensity pasture. Nutrient Cycling in Agroecosystems 88, 275-288.

Ball, P.R., Field, T.R.O., 1982. Responses to Nitrogen as Affected by Pasture Characteristics, Season and Grazing Management. In: Lynch, P.B. (Ed.), Nitrogen Fertilisers in New Zealand Agriculture. New Zealand Institute of Agricultural Science, Wellington.

Barton, L., Schipper, L.A., 2001. Regulation of Nitrous Oxide Emissions from Soils Irrigated with Dairy Farm Effluent. J. Environ. Qual. 30, 1881-1887.

Baudracco, J., Lopez-Villalobos, N., Holmes, C.W., Macdonald, K.A., 2010. Effects of stocking rate, supplementation, genotype and their interactions on grazing dairy systems: a review. New Zealand Journal of Agricultural Research 53, 109-133.

Bolan, N.S., Horne, D.J., Currie, L.D., 2004. Growth and chemical composition of legume-based pasture irrigated with dairy farm effluent. New Zealand Journal of Agricultural Research 47, 85-93.

Bowler, D.G., 1980. The drainage of wet soils. Massey University, Palmerston North.

Bryant, J.R., López-Villalobos, N., Pryce, J.E., Holmes, C.W., Johnson, D.L., 2007. Quantifying the effect of thermal environment on production traits in three breeds of dairy cattle in New Zealand. New Zealand Journal of Agricultural Research 50, 327.

Bryant, J.R., Matthews, L.R., Davys, J., 2010. Development and application of a thermal stress model. Proceedings of the 4th Australasian Dairy Science Symposium, 360-364.

Clough, T.J., Kelliher, F.M., 2005. Dairy Farm Effluent Effects on Urine Patch Nitrous Oxide and Carbon Dioxide Emissions. J. Environ. Qual. 34, 979-986.

Collins, R., McLeod, M., Hedley, M., Horne, D.J., Hanly, J.A., Close, M., Bagshaw, C., Donnison, A., Ross, C., Matthews, L., 2005. Pathogen Transmission Routes: farm animals to waterways - Best Management Practices. Client Report prepared for MAF. NIWA, New Zealand.

Cosgrove, G.P., Anderson, C.B., Parsons, A.J., Brock, J.L., Tilbrook, J.C., 2002. Can nitrogen-fertilised ryegrass substitute for white clover? Proceedings of the New Zealand Grassland Association 64, 205-209.

DairyNZ, 2010. Facts and Figures for New Zealand Dairy Farmers. DairyNZ.

de Klein, C.A.M., 2001. An analysis of environmental and economic implications of nil and restricted grazing systems designed to reduce nitrate leaching from New Zealand dairy farms. II. Pasture production and cost/benefit analysis. New Zealand Journal of Agricultural Research 44, 217-235.

de Klein, C.A.M., Ledgard, S.F., 2001. An analysis of environmental and economic implications of nil and restricted grazing systems designed to reduce nitrate leaching from New Zealand dairy farms. I. Nitrogen losses. New Zealand Journal of Agricultural Research 44, 201-215.

Di, H.J., Cameron, K.C., 2002. Nitrate leaching in temperate agroecosystems: sources, factors and mitigating strategies. Nutrient Cycling in Agroecosystems 46, 237-256.

Draganova, I., Yule, I., Betteridge, K., Hedley, M., Stevenson, M., Stafford, K., 2010. Activity patterns and nutrient redistribution in an Intensively Managed Dairy Herd. In: Currie, L.D., Christensen, C.L. (Eds.), Farming's Future: Minimizing Footprints and Maximising Margins. Fertilizer and Lime Research Centre, Massey University.

During, G., McNaught, K.J., 1961. Effects of cow urine on growth of pasture and uptake of nutrients. New Zealand Journal of Agricultural Research 4, 591-605.

Eriksen, J., Askegaard, M., Kristensen, K., 1999. Nitrate leaching in an organic dairy/crop rotation as affected by organic manure type, livestock density and crop. Soil Use and Management 15, 176-182.

Fonterra, 2012. http://www.fonterra.com/wps/wcm/connect/fonterracom/fonterra.com/Our+Business/Sustainability/Water/.

Haynes, R.J., Williams, P.H., 1993. Nutrient cycling and soil fertility in the grazed pasture ecosystem. Advances in Agronomy 49, 119-199.

Hewitt, A.E. (Ed), 1998. NZ Soil Classification. Manaaki Whenua - Landcare Research New Zealand Ltd Press, Lincoln.

Holmes, C.W., Brookes, I.M., Garrick, D.J., MacKenzie, D.D.S., Parkinson, T.J., Wilson, G.F., 2007. Milk Production from Pasture - Principles and Practices. Massey University, Palmerston North.

Horne, D.J., 2012. Personal Communication.

Horne, D.J., Singleton, P.L., 1997. Managing wet soils in wet years: a cautionary tale. Dairy Farming Annual 49, 117-124.

Houlbrooke, D., Longhurst, B., Orchiston, T., Muirhead, R., 2011. Characterising dairy manures and slurries. Envirolink tools report AGRX0901.

Houlbrooke, D.J., Horne, D.J., Hedley, M.J., Hanly, J.A., Scotter, D.R., Snow, V.O., 2004. Minimising surface water pollution resulting from farm-dairy effluent application to mole-pipe drained soils. I. An evaluation of the deferred irrigation system for sustainable land treatment in the Manawatu. New Zealand Journal of Agricultural Research 47, 405-415.

Houlbrooke, D.J., Horne, D.J., Hedley, M.J., Snow, V.O., Hanly, J.A., 2008. Land application of farm dairy effluent to a mole and pipe drained soil: implications for nutrient enrichment of winter-spring drainage. Australian Journal of Soil Research 46, 45-52.

Hughes, T.P., Sykes, A.R., Poppi, D.P., Hodgson, J., 1991. The influence of sward structure on peak bite force and bite weight in sheep. Proceedings of the New Zealand Society of Animal Production, pp. 153-158.

Irvine, L.D., Freeman, M.J., Rawnsley, R.P., 2010. The effect of grazing residual control methods on cow intake and milk production in late spring. Proceedings of the 4th Australasian Dairy Science Symposium.

Kennedy, E., McEvoy, M., Murphy, J.P., O'Donovan, M., 2009. Effect of restricted access time to pasture on dairy cow milk production, grazing behavior, and dry matter intake. Journal of Dairy Science 92, 168-176.

Laven, R.A., Holmes, C.W., 2008. A review of the potential impact of increased use of housing on the health and welfare of dairy cattle in New Zealand. New Zealand Veterinary Journal 56, 151-157.

Ledgard, S.F., Menneer, J.C., 2005. Nitrate leaching in grazing systems and management strategies to reduce losses. In: Currie, L.D., Hanly, J.A. (Eds.), Developments in fertiliser application technologies and nutrient management. Fertiliser and Lime Research Centre, Massey University, Palmerston North, pp. 79-92.

Ledgard, S.F., Menneer, J.C., Boyes, M., Jensen, R., Clark, D.A., 2006. Effects of dairy intensification in the resource efficient dairying trial on soil physics, nutrient flows and fertiliser and lime requirements. In: Currie, L.D., Hanly, J.A. (Eds.), Implementing sustainable nutrient management strategies in agriculture, Occasional Report No. 19. Fertilizer & Lime Research Centre, Palmerston North.

Ledgard, S.F., Steele, K.W., Saunders, W.H.M., 1982. Effects of cow urine and its major constituents on pasture properties. New Zealand Journal of Agricultural Research 25, 61-68.

Livestock Improvement Corporation, DairyNZ, 2011. New Zealand Dairy Statistics 2010-11. DairyNZ.

Livestock Improvement Corporation, DairyNZ, 2012. New Zealand Dairy Statistics 2011-12. DairyNZ.

Longhurst, R.D., Luo, J., O'Connor, M.B., Pow, T., 2006. Herd Homes: nutrient management and farmer perceptions of performance. Proceedings of the New Zealand Grassland Association 68, 309-313.

Marsh, R., Campling, R.C., 1970. Fouling of pastures by dung. Herbage Abstracts 40, 123-130.

McDowell, R.W., Muirhead, R.W., Monaghan, R.M., 2006. Nutrient, Sediment, and Bacterial Losses in Overland Flow from Pasture and Cropping Soils Following Cattle Dung Deposition. Communications in Soil Science and Plant Analysis 37, 93-108.

Ministry of Agriculture and Forestry, 2011. http://www.maf.govt.nz/agriculture/pastoral/dairy.

Moir, J.L., Cameron, K.C., Di, H.J., Fertsak, U., 2011. The spatial coverage of dairy cattle urine patches in an intensively grazed pasture system. The Journal of Agricultural Science 149, 473-485.

Monaghan, R.M., de Klein, C.A.M., Muirhead, R.W., 2008. Prioritisation of farm scale remediation efforts for reducing losses of nutrients and faecal indicator organisms to waterways: A case study of New Zealand dairy farming. Journal of Environmental Management 87, 609-622.

Monaghan, R.M., Hedley, M.J., Di, H.J., McDowell, R.W., Cameron, K.C., Ledgard, S.F., 2007. Nutrient management in New Zealand pastures - recent developments and future issues. New Zealand Journal of Agricultural Research 50, 181-201.

Muirhead, R.W., Collins, R.P., Bremer, P.J., 2005. Erosion and Subsequent Transport State of *Escherichia coli* from Cowpats. Applied and Environmental Microbiology 71, 2875-2879.

Norman, M.J.T., Green, J.O., 1958. The local influence of cattle dung and urine upon the yield and botanical composition of permanent pasture. Journal of the British Grassland Society 13, 39-45.

Oudshoorn, F.W., Kristensen, T., Nadimi, E.S., 2008. Dairy cow defecation and urination frequency and spatial distribution in relation to time-limited grazing. Livestock Science 113, 62-73.

Pande, T.N., Valentine, I., Betteridge, K., Mackay, A., Horne, D.J., 2000. Pasture damage and regrowth from cattle treading. Proceedings of the New Zealand Grassland Association 62, 155-160.

Parliamentary Commissioner for the Environment, 2012. Water Quality in New Zealand: Understanding the Science. Parliamentary Commissioner for the Environment, Wellington.

Pollok, J.A., 1975. A comparative study of certain New Zealand and German soils formed from loess., Unpublished PhD Thesis. University of Bonn, Germany.

Ravagnolo, O., Misztal, I., Hoogenboom, G., 2000. Genetic Component of Heat Stress in Dairy Cattle, Development of Heat Index Function. Journal of Dairy Science 83, 2120-2125.

Ryden, J.C., Syers, J.K., 1975. Rationalization of ionic strength and cation effects on phosphate sorption by soils. Journal of Soil Science 26, 395-406.

Salazar, M.E., Hedley, M.J., Horne, D.J., 2010. Using turnips to reduce soil K loading on the effluent block. Proceedings of the New Zealand Grassland Association 72, 247-250.

SAS, 2003. SAS v9.1. SAS Institute Incorporated, Cary NC, USA.

Scotter, D.R., Clothier, B.E., Turner, M.A., 1979. The soil water balance in a Fragiaqualf and its effect on pasture growth in central New Zealand. Australian Journal of Soil Research 17, 455-465.

Sharpley, A.N., Syers, J.K., 1979. Loss of nitrogen and phosphorus in tile drainage as influenced by urea application and grazing animals. New Zealand Journal of Agricultural Research 22, 127-131.

Silva, R.G., Cameron, K.C., Di, H.J., Hendry, T., 1999. A lysimeter study of the impact of cow urine, dairy shed effluent, and nitrogen fertiliser on nitrate leaching. Australian Journal of Soil Research 37, 357-369.

Taylor, B., 2012. Personal Communication.

Chapter 4 The impact of Duration-controlled grazing on nitrogen losses in drainage water and surface runoff

4.1 Introduction

The global demand for dairy products is increasing (Fonterra, 2013) and the New Zealand dairy industry is striving to increase milk production to help satisfy this demand. At the same time, dairy farmers must lower their environmental footprint (Parliamentary Commissioner for the Environment, 2012) to meet the expectations associated with a 'clean/green' image and the more stringent regulations that are being imposed by Regional Councils. To help improve water quality, dairying is seeking to urgently reduce losses of nitrogen (N), phosphorus (P) and faecal microbes to surface and ground waters (Monaghan *et al.*, 2008). The target of the dairy industry is to decrease N leaching losses to water by 30% by the year 2020 (DairyNZ, 2010a). To achieve this, the industry must have access to mitigation strategies such as Duration-controlled (*DC*) grazing (Chapter 3) which can reduce N leaching while maintaining or increasing production.

The urine patch is the largest source of leached N (Silva *et al.*, 1999; Ledgard and Menneer, 2005); the concentration in these patches can be up to 1000 kg N/ha (Haynes and Williams, 1993; Di and Cameron, 2002). The majority of N leaching losses from grazed pastures are in the form of nitrate (NO_3^-) in drainage waters (Haynes and Williams, 1993). Duration-controlled grazing seeks to minimise the quantity of N leached by reducing the amount of urine patches in pastures. It achieves this by moving cows to stand-off facilities to ruminate after a relatively short period (4 hours) of grazing. The excreta collected from the stand-off facility is stored as effluent and then returned to pastures more evenly over a larger area, resulting in a much smaller N application rate, compared to urine and dung patches (Monaghan *et al.*, 2008). Furthermore, the effluent can be stored until conditions are appropriate for application ensuring that plant uptake of nutrients, such as N, is optimal, thus reducing the risk of N leaching from the soil.

In a modelling exercise comparing restricted grazing practices with conventional grazing in the autumn in southern New Zealand, de Klein and Ledgard (2001) predicted a 35-50% reduction in NO_3^- leaching, and a 10% reduction in total N (TN) losses, including volatilisation, under restricted grazing. Given the magnitude of these likely benefits of *DC* grazing, it was deemed important to measure and confirm the effects of year-round *DC* grazing on N losses in grazing trials and to demonstrate whether these predicted reductions in N leaching are achievable in practice.

In this chapter, the experimental objectives are to measure the volume and N concentrations in pipe drainage and surface runoff from grazed plots, and compare the effects of *DC* grazing with *SG* grazing on these N losses.

4.2 Methods

4.2.1 Experimental design

The location and system design has been described in detail in Chapter 3.

The trial consisted of seven replicates of two treatments. One treatment was Standard grazing, (*SG*) which involved a grazing duration of ~7 hours for day grazings and ~12 hours for night grazings. The other treatment was a Duration-controlled grazing (*DC*) treatment, which involved a grazing duration of 4 hours for both day and night grazings. Fertiliser was applied to all plots (both *SG* and *DC*) at various times throughout the course of the experiment. The details of timing and rates of fertiliser application and grazing management were provided in Chapter 3.

The research area consisted of 14 pasture plots (average area ca. 850 m^2/plot). Each plot had an isolated mole and pipe drainage system (Bowler, 1980). Mole channels were ploughed at 2 m intervals at a depth of 0.45 m. The mole channels intercept (perpendicularly) a gravel backfilled trench above a perforated pipe drain (0.11 m diameter, installed at a depth of 0.60 m). A series of sub-plots were installed on the main plots to measure surface runoff (referred to as ‘runoff’ hereafter). These runoff plots (area 50 m^2/plot) were isolated on 3 sides by buried wooden boards (250 mm deep x 25 mm wide, protruding 50 mm above ground). The plots graded (ca. 3 degree slopes) to a gutter on the 4^{th} side which collected the runoff. The mole and pipe drainage plots were established during the summer of 2008 and the runoff plots were added progressively throughout 2008 and the winter of 2009.

4.2.2 Drainage and runoff measurements and water analysis

Runoff and drainage water from plots were channelled through drainage pipes into individual tipping-bucket flow meters located in sampling pits nearby. The flow rate of drainage water was measured with 5 L tipping buckets and the runoff with 2 L tipping buckets. Each tipping-bucket was calibrated dynamically to account for larger tip volumes at higher flow rates. All tipping buckets were instrumented with data loggers to provide continuous measurements of flow rate. During each runoff and drainage event, a proportion (ca. 0.1%) of the runoff and drainage water from every second tip of the tipping bucket flow meter was automatically collected to provide a volume-proportioned, mixed sample for analysis (Plate 4.1).

Plate 4.1: Individual tipping buckets and collection area for drainage and runoff water.

Drainage and runoff samples were filtered through a 0.45 μm filter within 12 hours of collection, and stored frozen until analysis. Filtered samples were analysed for NO_3^--N and ammonium-N (NH_4^+-N) using colorimetric methods on a Technicon Auto Analyser (Blakemore *et al.*, 1987) (Plate 4.2). Total N of unfiltered samples was determined using the persulphate digestion method of Hosomi and Sudo (1986). In this procedure, the various forms of N in the sample are converted to NO_3^- and subsequently analysed using the colorimetric methods described above for NO_3^--N. Total organic N (TON) was estimated from the difference between TN and NO_3^--N.

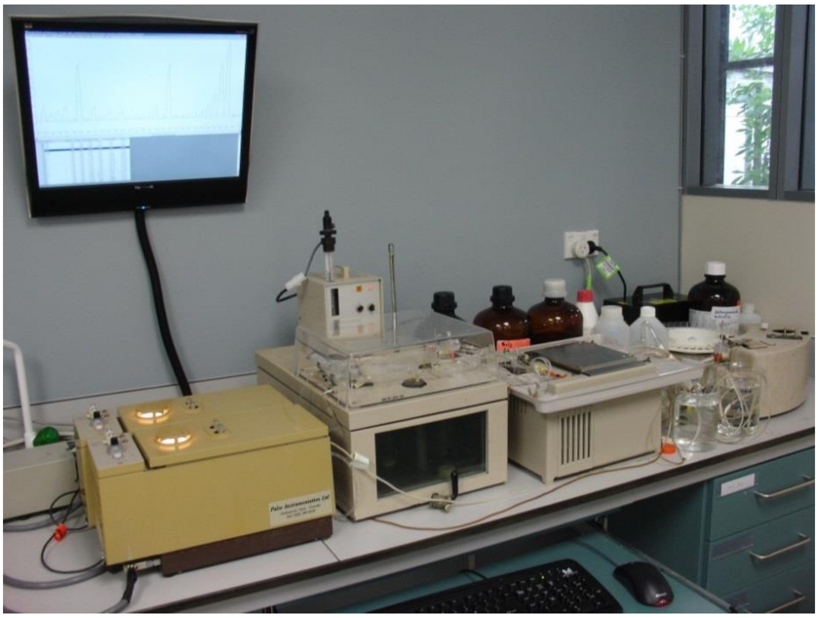

Plate 4.2: Technicon Auto Analyser used for colorimetric nitrate (NO_3^-) and total nitrogen (TN) analysis in the laboratory.

4.2.3 Soil water balance

The soil water balance description has been given in Chapter 3.

4.2.4 Slurry application to Duration-controlled plots to balance nutrient return

The dung pats deposited on each plot were counted in the 2008/09 lactation season to give an indication of the total amount of excreta returned to the plots. The difference in average annual dung depositions to the two treatments was used to estimate the quantity of excreta that would be transferred to the standoff facility and, therefore, the amount of slurry to be applied to the *DC* treatment plots (Chapter 3). The model of Salazar *et al.* (2010), using the average N concentrations in pasture and grass silage, the time spent on the different treatment plots, and average product and transfer losses, (which is based on herbage intake and average N concentration), was also used to estimate the amount of excreta returned to the plots. The estimate of the amount of excreta returned to the plots by the Salazar model (Table 4.1) was similar to the estimate given by counting dung pats, particularly for the *DC* treatment.

Table 4.1: The estimated annual return of nitrogen (N) in dung and urine to *DC* and *SG* treatments, the theoretical target slurry application, and the actual return of N in slurry (kg N/ha/yr). Values in parentheses represent the mineral N percentage applied for each season.

Year	Estimated N returned in dung and urine (model)[1] (kg N/ha/yr)	Estimated N returned in dung and urine (dung counts)[2] (kg N/ha/yr)	Theoretical target slurry application[1,3] (kg N/ha/yr)	Actual N application in form of slurry (kg N/ha/yr)
2008/09 SG	383	318		
DC	155	173	136	212[4] (16.6%)
2009/10 SG	340	297		
DC	137	161	121	nil
2010/11 SG	425	339		
DC	172	185	151	115[5] (39.7%)
Ave SG	383	318		
Ave DC	155	173	136	109

[1]Estimated using modified version of model by Salazar *et al.* (2010) based on number of grazings in each year and average herbage N concentrations.
[2]Estimated using dung counts in 2008/09 lactation season.
[3]Assuming 45% losses of N from stand-off facility, based on Longhurst *et al.* (2006).
[4]A single biennial slurry application strategy, applied in December 2008.
[5]Four applications in October and November 2010, and February and April 2011, at 15, 11, 68, and 21 kg N/ha respectively (Chapter 3).

4.2.5 Statistical analysis

Treatment differences were analysed using analysis of variance (ANOVA) within Microsoft Excel 2010.

4.3 Results

4.3.1 Climatic conditions and drainage volumes

Measured drainage events coincided with periods of zero soil water deficit as predicted by the soil water balance model of Scotter *et al.* (1979) (Figure 4.1). The average (across all 14 plots) annual drainage depth was not statistically different ($P>0.05$) for all three years, with 373, 316 and 329 mm of drainage in 2009, 2010 and 2011, respectively. There was slightly less drainage from the *SG* treatment plots, but this difference was not significant and the degree of variation observed here would be expected for field-scale mole and pipe drainage systems (Houlbrooke *et al.*, 2003).

Although annual drainage depths were similar, the commencement and spread of the drainage seasons varied over the 3 years. The drainage season of 2009 was uncharacteristically long. Unseasonably large rainfall events initiated drainage in February and March. These drainage events were followed by a drier period which created a soil water deficit of up to 70 mm by early April. By early May, the soil had re-wetted and the winter drainage season began. The last drainage event occurred at the end of December (Figure 4.1). The size of drainage events in 2009 ranged from 2.6 - 40.0 mm (averaged across all plots).

Conversely, in 2010 there was less rainfall in February and March, and the soil did not reach saturation until mid-June when drainage began. The drainage season of 2010 was shorter than that of 2009, with the majority of drainage occurring in the month of September, which had a particularly high monthly rainfall of 266 mm (cf. 88 mm and 43 mm in September of 2009 and 2011, respectively). The size of drainage events in 2010 ranged from 2.3 - 25.3 mm (averaged across all plots). The drainage season ended in October (Figure 4.1).

The 2011 drainage season began in late April and finished in November, producing a similar range of drainage event sizes (3.5 - 23.4 mm) to those in 2010 (Figure 4.1). The length of the 2011 drainage season was more ‘typical’ of the site, being comparable to those seasons reported by Hanly (2012), Houlbrooke *et al.* (2003), and Sharpley and Syers (1979).

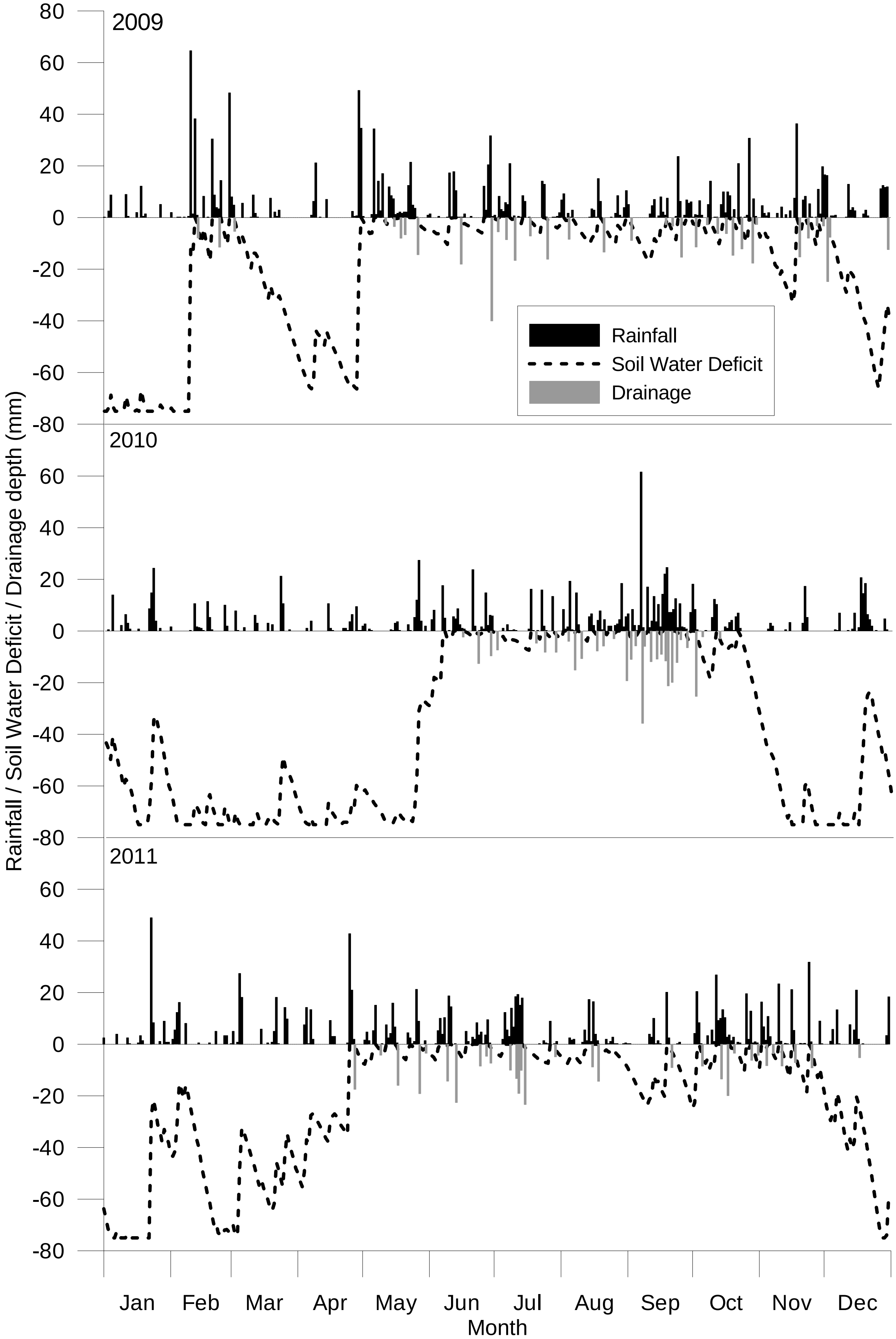

Figure 4.1: Mean drainage (mm) measured for all *DC* and *SG* plots, and rainfall and modelled soil water deficit for each event from 2009-2011. Soil water deficit and drainage depth are absolute values, but are displayed as negative values to add clarity to the figure.

4.3.2 Mineral nitrogen losses in drainage water

Each year, the highest concentrations of NO_3^- in drainage water occurred in the first 6 to 7 drainage events, which was typically the first 50-100 mm of drainage (Figure 4.2). The peak concentrations of NO_3^--N for *SG* grazed plots were 12.9, 9.0 and 19.1 ppm in 2009, 2010 and 2011 respectively, and were significantly ($P<0.05$) higher than the peak NO_3^--N concentrations in drainage from *DC* grazed plots (7.6, 4.2 and 10.8 ppm in 2009, 2010 and 2011, respectively). The magnitude of the concentrations of NO_3^- in these initial drainage events from both treatments varied between years (Figure 4.2).

Irrespective of drainage season and grazing treatment, the large concentrations of NO_3^- in the initial drainage events were followed by a rapid decrease in the concentrations of NO_3^- of subsequent drainage events until approximately 200 mm of drainage had occurred (Figure 4.2). This decreasing trend in NO_3^- concentration was observed by others at the same site (Houlbrooke *et al.*, 2003; Hanly, 2012), and also under similar grazed pasture conditions at other mole and pipe drained sites (Monaghan *et al.*, 2002). By the end of August every year (ca. 200 mm accumulated drainage), NO_3^- -N concentrations were < 2 ppm and similar ($P>0.05$) between both *SG* and *DC* treatments (Figure 4.2). Individual grazing events during the drainage season had no obvious effect on increasing the concentrations of NO_3^- in drainage (Figure 4.2).

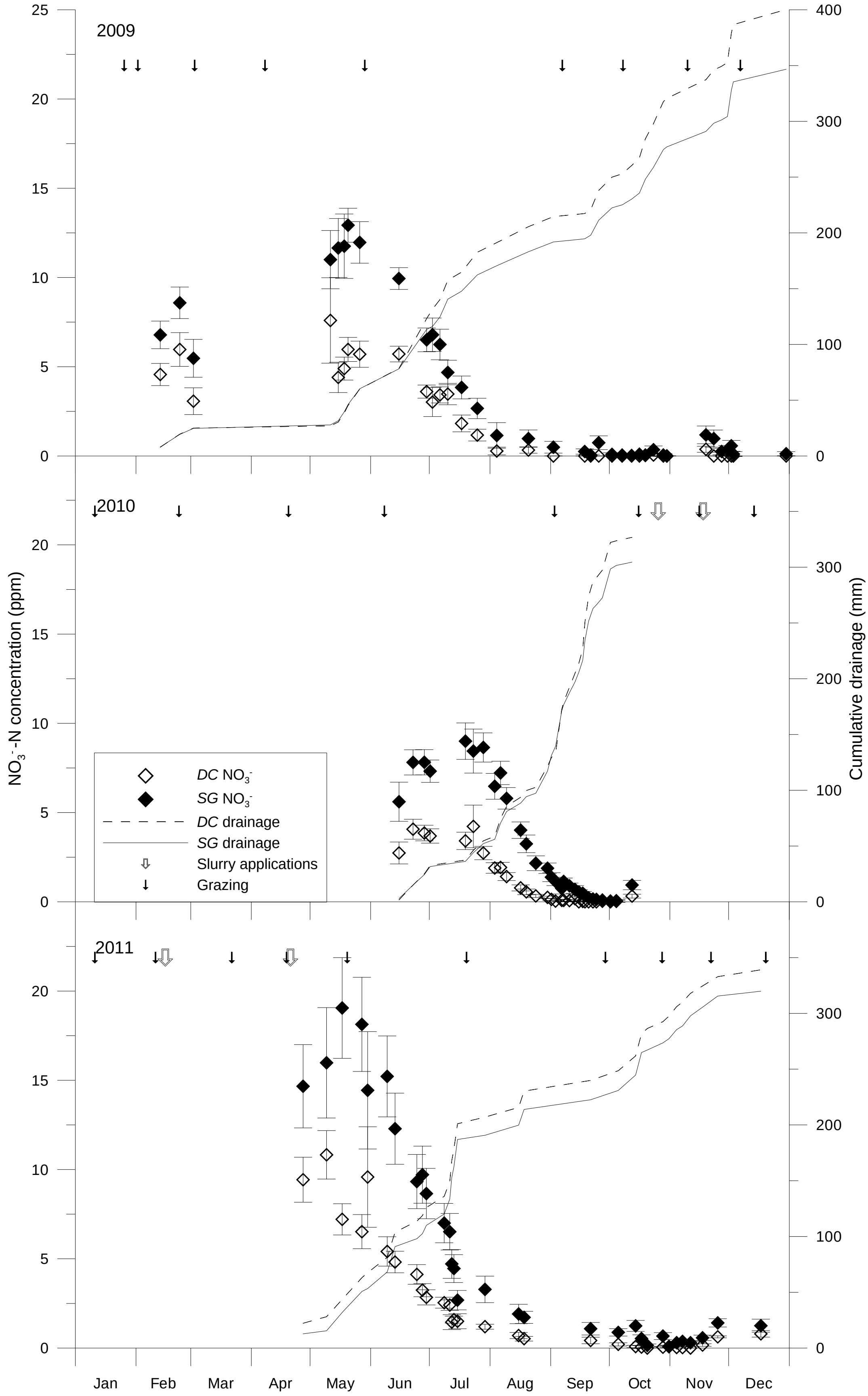

Figure 4.2: Concentration of NO_3^--N (ppm) in drainage water from 2009-2011, and the cumulative depth of drainage (mm) for *DC* and *SG* grazing treatments. (Error bars represent standard error at each sampling event; see legend for pasture management activities).

The cumulative leaching load of NO_3^- (kg N/ha) from the *DC* treatment was lower than the *SG* treatment, with large and significant differences ($P<0.05$) by the end of each drainage season (Figure 4.4). Average NO_3^--N leaching loads from the *SG* plots were 13.11, 8.00 and 20.95 kg NO_3^- -N/ha for 2009, 2010 and 2011, respectively. There was a 43%, 65% and 53% reduction in the annual NO_3^--N load (kg N/ha) from the *DC* treatment plots compared with *SG* plots for 2009, 2010 and 2011, respectively. This gave an average reduction of 52% over the 3 years. The majority (> 70%) of the NO_3^--N leaching load occurred relatively early in the season, i.e. in the first 100 mm of accumulated drainage (Table 4.2). By the time 200 mm of accumulated drainage had occurred, more than 95% of the NO_3^- -N load had leached. This indicates that mitigation strategies targeting substantial reductions in NO_3^- leaching need to be effective within the first 200 mm of drainage. In 2010, a similar pattern of NO_3^- loss occurred as in the other two experimental years, but the loss occurred later in winter due to drainage commencing in June.

The annual amount of NH_4^+ that was leached in drainage was very small, with less than 0.17 kg NH_4^+-N/ha being leached from any treatment in any one year, which represented less than 2.5% of TN. Given the very small contribution NH_4^+ makes to TN losses in drainage, the data have not been presented.

4.3.3 Total nitrogen losses in drainage water

The concentration of TN in drainage water and the cumulative leaching load exhibited similar patterns to those observed for NO_3^- (Figures 4.2 and 4.3). Similar patterns for NO_3^- and TN are expected because a large proportion of TN is comprised of NO_3^- (Table 4.2). There was also a large and significant ($P<0.05$) difference between the TN concentration of drainage from *SG* and *DC* grazing treatments, with this difference decreasing as the drainage season progressed each year. Peak concentrations of TN were 13.8, 13.8 and 21.9 ppm in 2009, 2010 and 2011 respectively, for the *SG* treatment, while the *DC* treatment had peak TN concentrations of 6.5, 7.3 and 11.7 ppm for each of the 3 years, respectively.

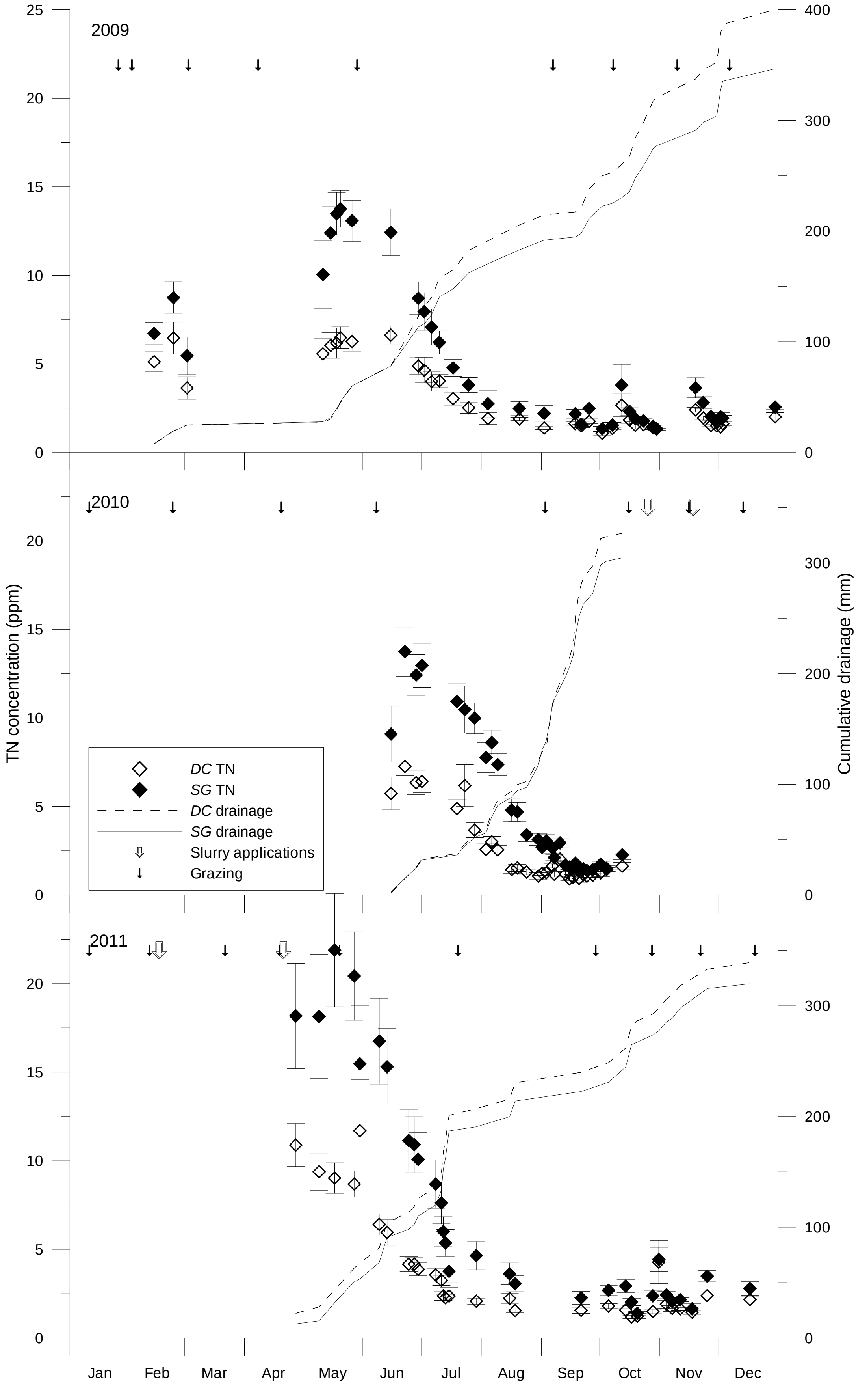

Figure 4.3: Concentration of TN (ppm) in pipe drainage water from 2009-2011, and the cumulative depth of drainage (mm) for *DC* and *SG* grazing treatments. Error bars represent standard error at each sampling event.

The TN concentration stabilised at 1-2 ppm from the end of August after approximately 200 mm of accumulated drainage through to the end of each drainage season (Figure 4.3). The corresponding NO_3^- as well as NH_4^+ concentrations were negligible during this period and, therefore, the majority of the TN losses at this time were likely to be organic forms of N (Table 4.2 and Figure 4.3). There was no relationship between the size of individual drainage events and NO_3^- or TN concentrations at each event (data not shown).

The average TN load in drainage from *SG* plots was 18.37, 13.10 and 26.23 kg N/ha for 2009, 2010 and 2011, respectively. This equated to a 32 - 47% (average 42%) reduction in the annual loss of TN in drainage from *DC* plots compared with *SG* plots (Table 4.2) over the 3 year experimental period. This reduction was slightly smaller than the corresponding decrease in NO_3^- loss under *DC* grazing.

The percentage of TN lost in drainage as NO_3^- varied between years. In 2009, 60% and 71% of the TN leached was as NO_3^- from *DC* and *SG* plots; in 2010 the proportion was 38% and 61%, and in 2011 it was 72% and 80%, respectively (Table 4.2). These values compare favourably with those reported in Hanly (2012) who, in a previous study at this site, found that NO_3^- comprised between 55 – 82% of the annual quantity of TN measured in drainage.

While the annual NO_3^- load in drainage varied widely between treatments and from year to year, the remaining component of TN, referred to as TON, was relatively uniform between treatments and years (Figure 4.5), providing a constant background level of TON, not influenced greatly by grazing treatment. To illustrate this, TON/mm drainage was plotted against cumulative drainage (Figure 4.6). After 100 mm of drainage in each of the three experimental years, the TON load leached was consistent throughout the remainder of the drainage season, being on average 10-15 g TON/mm drainage/ha (Figure 4.6). This rate of loss was similar for both treatments. As each of the three years had similar drainage depths (316-375 mm), the quantity of TON lost each year was also comparable (4-5 kg/ha).

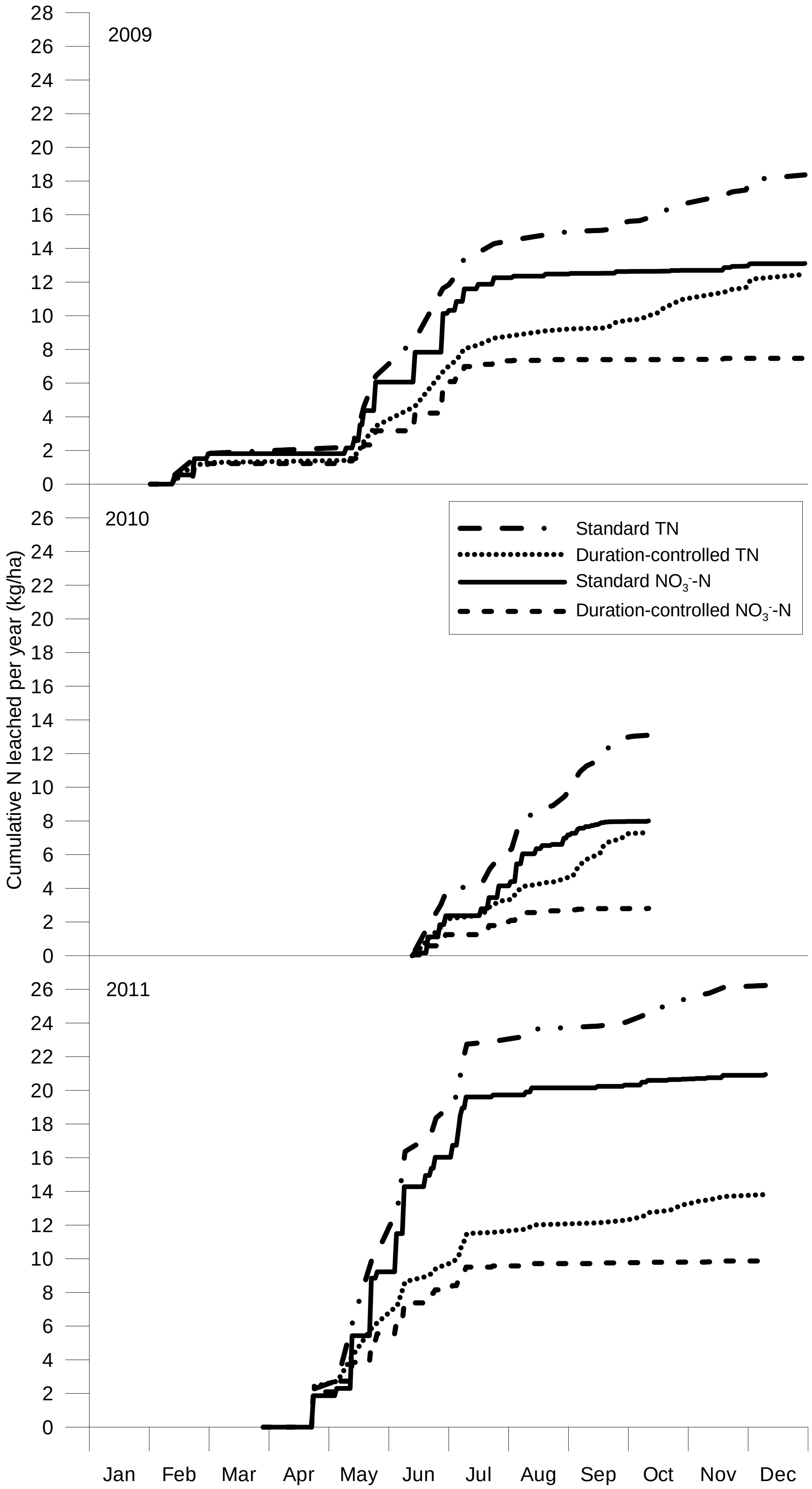

Figure 4.4: Cumulative NO_3^--N and TN leached (kg/ha) in drainage water from 2009-2011, for *DC* and *SG* grazing treatments.

Table 4.2: Total amount of NO_3^--N and TN leached in drainage water from 2009-2011, for *DC* and *SG* grazing treatments, and proportion of N in the first 100 mm of drainage, 100-200 mm drainage, and >200 mm drainage for each year.

			N leached for year (kg N/ha)[1]	% reduction in leaching from DC plots	Ratio NO_3^- to TN (%)	100 mm		100 - 200 mm		> 200 mm	
						N leached (kg)	% year's N leached	N leached (kg)	% year's N leached	N leached (kg)	% year's N leached
2009	**NO_3^--N**	***DC***	7.47	43	60	5.83	78	1.57	21	0.07	1
		SG	13.11		71	10.14	77	2.38	18	0.59	5
	TN	***DC***	12.46	32		6.68	54	2.54	20	3.24	26
		SG	18.37			11.63	63	3.38	18	3.36	18
2010	**NO_3^--N**	***DC***	2.81	65	38	2.66	95	0.13	5	0.02	-
		SG	8.00		61	6.61	83	1.12	14	0.27	3
	TN	***DC***	7.31	44		4.37	60	1.50	21	1.44	20
		SG	13.10			8.91	68	2.51	19	1.68	13
2011	**NO_3^--N**	***DC***	9.92	53	72	7.79	79	1.79	18	0.34	3
		SG	20.95		80	14.95	71	4.78	23	1.22	6
	TN	***DC***	13.82	47		8.95	65	2.62	19	2.25	16
		SG	26.23			17.15	65	5.76	22	3.32	13

[1] NH_4^+-N leaching losses were < 0.17 kg N/ha in any year so have not been included.

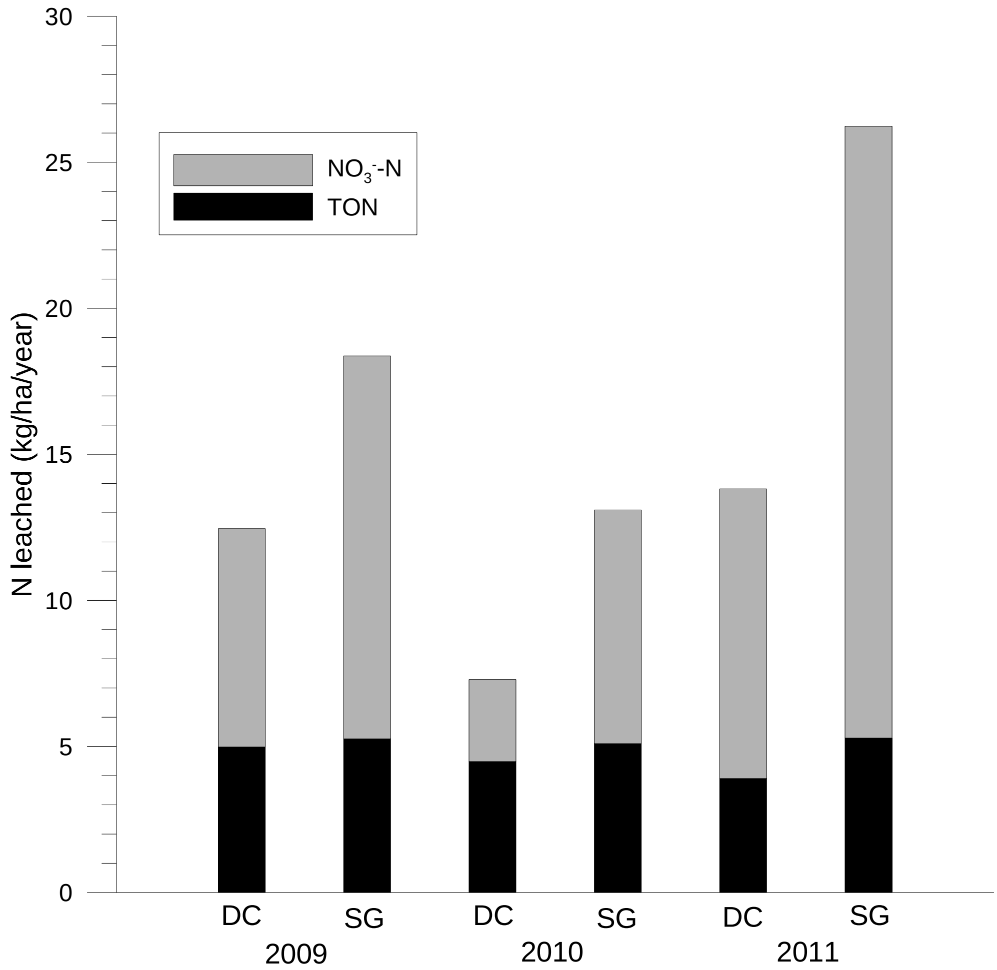

Figure 4.5: Total organic nitrogen (TON) and measured NO_3^- -N leached from *DC* and *SG* treatments, from 2009-2011. The sum of NO_3^- -N and TON is assumed to be total N (TN) leached due to measured NH_4^+ leaching being negligible.

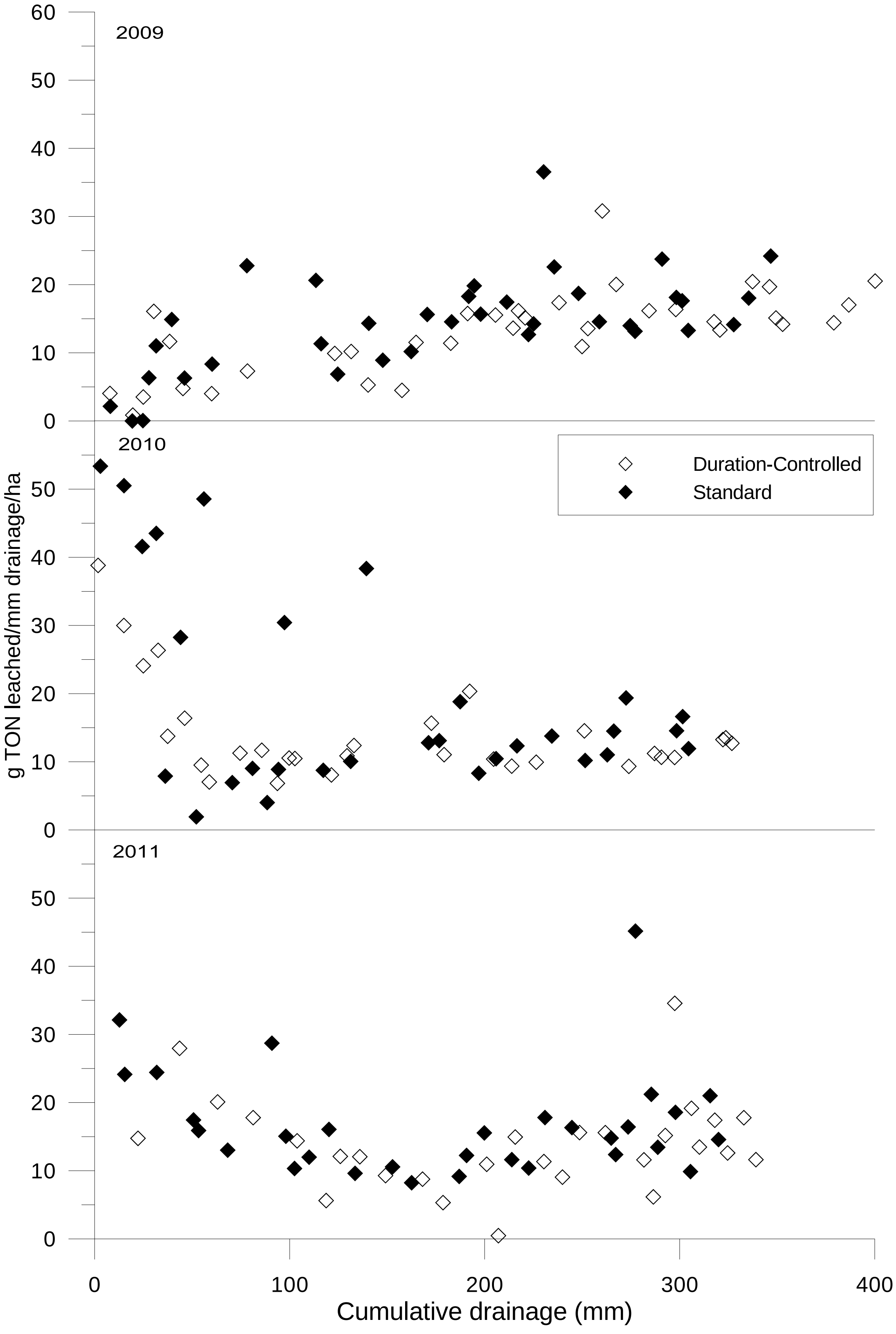

Figure 4.6: Total organic nitrogen (TON) leached (g/mm drainage/ha) with cumulative drainage (mm) from *DC* and *SG* treatments in 2009-2011.

4.3.4 Nitrogen losses in surface runoff

Nitrogen losses in runoff were relatively small over the 3 years of the study. In particular, NO_3^- and NH_4^+ losses were almost negligible with less than 1 kg N/ha lost in either form in runoff from both treatments in all years (Table 4.3). Total N losses were, on average over the 3 years, 2.86 kg N/ha from *DC* plots, and 3.24 kg N/ha from *SG* plots. There was no significant difference (P>0.05) between treatments for TN losses in runoff.

Table 4.3: Total amounts of NO_3^- -N, NH_4^+-N, and TN (kg/ha) lost in surface runoff water from 2009-2011 for *DC* and *SG* grazing treatments.

	NO_3^- -N (kg/ha) in surface runoff		NH_4^+-N (kg/ha) in surface runoff		Total N (kg/ha) in surface runoff	
	DC	*SG*	*DC*	*SG*	*DC*	*SG*
2009	0.23	0.32	0.42	0.69	3.21	4.43
2010	0.02	0.06	0.29	0.44	1.41	1.88
2011	0.77	0.41	0.30	0.39	3.95	3.40

4.4 Discussion

4.4.1 Nitrogen losses in drainage water

Compared with *SG* grazing, *DC* grazing resulted in average reductions of 52% and 42% in NO_3^- and TN leaching, respectively. Duration-controlled grazing commenced in September 2008 and the reductions in the concentrations of NO_3^- and TN in pipe drainage water were observed in the first drainage events of the following season. This suggests that the difference in excreta return between the two grazing systems had an immediate and major impact on the amounts of N lost in drainage water.

The period of high concentrations of NO_3^- at the beginning of each drainage season was the main contributor to annual leaching losses for both treatments (Figure 4.2). These data indicate that NO_3^- accumulates in the soil under urine patches prior to the drainage season, which is a finding similar to that made by Houlbrooke *et al.* (2008) at the same site. Therefore, the difference in N loss between the grazing treatments was mainly due to the differences in NO_3^- concentrations in the early drainage events of each season. At the beginning of each drainage season, when NO_3^- concentrations in drainage were at their peak, NO_3^- concentrations in drainage from the *SG* plots were approximately double those from *DC* plots (Figure 4.2). There was a relatively large and steady decrease in concentrations within the first 100 mm of drainage from both treatments.

The significantly lower NO_3^- concentrations measured in drainage from the *DC* plots is caused by fewer urine depositions on these plots (enumerated using dung pat counts, Table 4.1), which was a result of the shorter grazing durations relative to the *SG* plots. It has been well documented that most NO_3^- leaching comes from urine patches due to the high proportion of excreted N present in urine (Silva *et al.,* 1999; Ledgard and Menneer, 2005). Dung contributes very little leached N, as a higher proportion of the N in dung is in more complex organic forms, which are less readily converted to NO_3^- than urinary N (Haynes and Williams, 1993).

A notable point was that even after grazings in the spring, NO_3^- concentrations in drainage did not increase significantly (Figure 4.2). Although there was drainage occurring right through into December in two of the years (2009 and 2011), NO_3^- concentrations remained < 2 ppm. This was similar to the concentrations reported by Houlbrooke *et al.* (2003) from the same site, where spring grazings had little effect on increasing NO_3^- concentrations in subsequent drainage events. This is due to a combination of a number of factors, including spring being an active period for pasture growth and NO_3^- uptake by pasture, the relatively small area covered by fresh urine spots from individual spring grazings, and possibly insufficient drainage volumes to move significant quantities of NO_3^- into mole channels before the completion of the drainage season.

Denitrification in warm wet soils (Luo *et al.*, 2008) may be another reason for the small NO_3^- concentrations in spring drainage from both treatments (Saggar *et al.*, 2004), and no response in these concentrations to grazings or slurry applications (rates of denitrification were not measured in this study). This has implications for added N by the way of urine, slurry, and manufactured fertiliser. To be getting the most efficient use from the added N for plant uptake, denitrification must be minimised. A decrease in the quantity of N lost to the atmosphere via denitrification from fewer urine spots may be an added benefit of *DC* grazing (de Klein *et al.*, 2006). However, in the spring after significant leaching of N, the amount of N available for plant uptake may be low, requiring fertiliser N or stored N as slurry to be applied to meet this shortage, but much of this N is at risk of denitrification. Furthermore, pasture growth is accelerating through this time and, thus, grazing must occur to maintain pasture quality for future grazings (Holmes *et al.*, 2007). The use of *DC* grazing would also reduce the potential for treading damage, which should reduce the potential for the N returned in urine, slurry and urea to be lost via denitrification. Further work should be conducted to investigate the effects of using a *DC* grazing system on denitrification rates.

The decreasing NO_3^- concentrations over the winter drainage season in each year, in spite of cows grazing plots over the winter period, indicates that there may be minimal benefit, in terms of reduced NO_3^- leaching, from wintering cows off farm. However, grazing cows off over winter, or at least incorporating *DC* grazing, would reduce treading damage (Horne and Singleton, 1997) and, therefore, have a positive effect on decreasing other losses, such as P and faecal microbes, through surface runoff (Chapters 5 and 6).

In 2009, drainage occurred in February, which is not typical. At these February drainage events, the relatively large concentrations of NO_3^--N in drainage water (up to 8.6 ppm from *SG* plots and 6.0 ppm from *DC* plots) suggest that the accumulation of soil NO_3^- had begun (Figure 4.2). Then, when the winter drainage season began in May, the NO_3^--N concentrations in drainage water were even higher (up to 13.0 ppm from *SG* plots and 7.6 ppm from *DC* plots); therefore, further NO_3^- accumulation occurred in the soil between February and May. These results support a hypothesis that the accumulation of NO_3^- in the soil from late-summer grazings onwards is the key source of NO_3^- that leaches during the subsequent drainage season.

Recent studies by Shepherd *et al.* (2011) on free draining soils in the Waikato region of New Zealand have focussed on the relationship between the timing of artificial urine deposition and the subsequent N leaching from those urine patches. These studies found that between 40 and 56% of the N that was deposited in urine from February to May was leached by the end of the drainage season. Not all the N from urine applied in June and July was leached from the root zone by the time drainage had completed in October. In earlier work conducted in France, using ^{15}N labelled urinary N applied to pasture lysimeters, it was found that 16.7% of autumn deposited urine, compared with 0.7% of spring and 7.7% of summer urine applications, was leached (Decau *et al.*, 2003, 2004). Much of the urinary N in the French study was either recovered in herbage (greater in spring than autumn) or in soil organic matter (similar for all seasons).

de Klein *et al.* (2006) studied restricted grazing in the autumn on a poorly drained soil, with a focus on reducing nitrous oxide (N_2O) losses, but also measured NO_3^- leaching. Cows were grazed for 3 hours per grazing, but only during March to May. Similar to the present study, this simulated the use of a feedpad for the remainder of the day. During the 3 years of de Klein *et al.*'s study, an average reduction in NO_3^- leaching of 41% was reported. This was almost as high as the present study, but was achieved by using shorter grazing duration over the autumn months, without the use of year-round *DC* grazing.

In a modelling exercise, de Klein and Ledgard (2001) predicted the effects of short-term nil grazing (between April and August) on NO_3^- leaching. In these studies, a reduction of 35-50% was modelled. In this model, cows grazed 'conventionally' (i.e. *SG*) from September to March, and then feed was cut and carried or conserved while cows were stood off completely from paddocks for the autumn and winter months. Because nil grazing from April to August caused no more than a 50% reduction, it suggests that urinary deposition in earlier months must be accounting for at least 50% of leached NO_3^-, which partly supports the findings from the current study.

Data from the 3 years of the current study have been used in a modelling study to quantify the expected amount of N that would be deposited at each grazing and is likely to remain in the soil under urine patches at the onset of winter drainage. Using the pasture intake data per cow per grazing from Chapter 3, the herbage N concentrations measured at each grazing, and the number of cows grazed at each grazing (129 - 396 cows/ha; Chapter 3), the amount of N eaten as pasture per cow per day was able to be calculated for each treatment (Eq. 4.1). Average DM intakes per cow from each grazing (night or day grazing; Table 3.6, Chapter 3) were doubled to estimate the average pasture intake over a whole day, and it was assumed that cows were receiving the balance of their 16 kg DM/day as maize silage (ca. 4 kg/cow/day, average 1.23% N content (DairyNZ, 2010b)).

N eaten (N_E) (g/cow/day) = $\left(\frac{I_P}{C} \times N \times 1000\right) + \left(\frac{I_M}{C} \times 1.23\% \times 1000\right)$ **Eq. 4.1**,

where: I_P = Intake of pasture (kg DM/ha/day) estimated from each pre-grazing and post-grazing pasture plate measurements from each grazing, then doubled for intake over a whole day,

C = Average number of cows/ha/grazing,

N = Average herbage N concentration measured at each grazing (%N),

I_M = Intake of supplements (kg DM/ha/day) (assumed to be all maize silage at 1.23% N content (DairyNZ, 2010b)).

Using the values generated for N_E, the urine N output equation of Castillo *et al.* (2000), based on a review of 580 dairy cows and 90 treatments, was used to predict the urine N output to plots (g/cow/day) for each averaged whole day grazing (Eq. 4.2). This equation is:

Urinary N output (U_O) (g/cow/day) = 30.4 ($e^{0.0036 \times N_E}$) **Eq. 4.2**.

While milk and faecal N outputs increased linearly with increasing N intakes, the amount of urinary N increased exponentially with an intake greater than 400 g N/cow/day. It was suggested that intakes up to a maximum of 400 g N/cow/day would limit the increases in urinary N output (Kebreab *et al.*, 2001). In this study, the time spent grazing pasture at each grazing as a fraction of the day (T) was used to calculate the area receiving urine (Eq. 4.3). This was based on an average of 11 urinations per cow per day (U_t) (Chapter 2), with an average urine patch area of 0.2 m^2 (U_{PA}) (Haynes and Williams, 1993).

Area (A) receiving urine (m^2/ha) = $(T \times U_t) \times C \times U_{PA}$ **Eq. 4.3**,

where: T = Time spent grazing on pasture (fraction of day),

and

Urinary N output per day (U_{OA}) (kg N/ha/day) = $(U_O \times C \times T)/1000$ **Eq. 4.4**.

There were 8 - 10 grazings per year and each grazing generated a series of U_{OA} for grazings $G_1 - G_n$, where G_n was the final grazing for the season (24 hour grazing in winter; Chapter 3). Uptake of N by pasture growing in urine patches was then estimated (N_{TA}; Eq. 4.5) from the subsequent pasture accumulation in the areas that had received urine at the previous grazing (G_{n-1}). The temporal pattern of herbage N concentrations in urine affected areas were estimated using the results from studies by Menneer *et al.* (2003), assuming pasture in urine and dung patches grew at twice the rate of pasture in inter-urine areas for the first 100 days since deposition, with no soil drainage occurring. At any one time, it was assumed that the urine area from 100 days (3 grazings) covered 8% of the grazed area (based on total ca. 25% urine area per year). Dung spots also covered 8%, giving a total 16% area of excretal deposition from the 3 grazings. The average pasture accumulation measured between each grazing (Figure 3.3, Chapter 3) was an average of the urine areas and inter-urine areas. After 100 days, it was assumed the urine patch area grew at the same rate and at the same herbage N concentration as that measured for the plot average in the present study, until the end of the lactation season.

N uptake from urine area (N_{TA}) (kg N/ha)

$=\left(A \times (\sum P_1\, N_U \ldots\ldots P_3 N_U)\right) + \left(A \times (\sum P_4\, N_4 \ldots\ldots P_n N_n)\right)$ **Eq. 4.5**,

where: $P_1 \ldots P_3$ = Accumulation of pasture (kg/ha) for 90-100 days after urination (assuming 3 grazings) (Menneer *et al.*, 2003) (where P_1 to P_3 = measured pasture accumulation/1.16 x 2; 1.16 is the factor of excretal areas, and 2 is twice the pasture growth rate in those areas),

N_U = Assumed average herbage N concentration in urine patch of 4.5% N (Menneer *et al.*, 2003),

$P_4 \ldots . P_n$ = Accumulation of pasture (kg/ha) after 90-100 days following urination (i.e. after 3 grazings have occurred) until the end of the grazing season,

$N_4 \ldots . N_n$ = Herbage N concentration measured at each grazing.

The difference between urinary N output and the accumulated amount of N taken up from urine patches (Eq. 4.6) between spring and the onset of winter drainage was assumed to be the amount of N that was available either for leaching, immobilisation, volatilisation, or denitrification.

Urinary N not taken up by pasture (U_L) = $U_{OA} - N_{TA}$ **Eq. 4.6.**

Urinary N not taken up (U_L) was then summed for urine patches deposited at all grazings, $G_1 - G_n$ (Eq. 4.7).

Urinary N not taken up for season or year = $\sum\left(U_{L_1} \ldots\ldots U_{L_n}\right)$ **Eq. 4.7**.

A longer period of active growth before the onset of winter drainage results in larger amounts of N taken up from urine patches deposited in the spring, than say, autumn. The estimated amount of urinary N not taken up for the year was plotted against the measured NO_3^- and TN leached (kg/ha/yr). It should be noted that N fertiliser applications to all plots and slurry applications to *DC* plots, which were deemed to be applied uniformly and at low rates (< 212 kg N/ha/yr), were not considered in the calculation of N deposited.

Over the whole season, there was a moderately high correlation between urinary N left after uptake and N leached (R^2=0.77 for NO_3^-; 0.75 for TN). The individual season that showed the highest correlation (R^2=0.96 for NO_3^- and 0.95 for TN) was autumn; i.e., grazings that occurred in March, April and May (Figure 4.7). It was particularly evident that in all years, the autumn grazings contributed the largest overall amount of urinary N left in the soil liable to leach, for both *DC* and *SG* treatments (Figure 4.8). Although some of the urine N deposited in spring grazings was not taken up by pasture over the whole season (Figure 4.8), this was more likely to be immobilised or to be lost as denitrification prior to drainage, as has been discussed. Further research should be conducted to quantify the amount of N being immobilised or denitrified during this time.

The results from this simulation of the amount of urinary-N remaining in the soil prior to winter drainage indicate strongly that *DC* grazing would be most effective in the autumn months for reducing N leaching. The results support the hypothesis that almost all of the NO_3^- leaching comes from urine patches, thus controlling the amount of N returned in those urine patches is imperative to reducing the N leaching in any year.

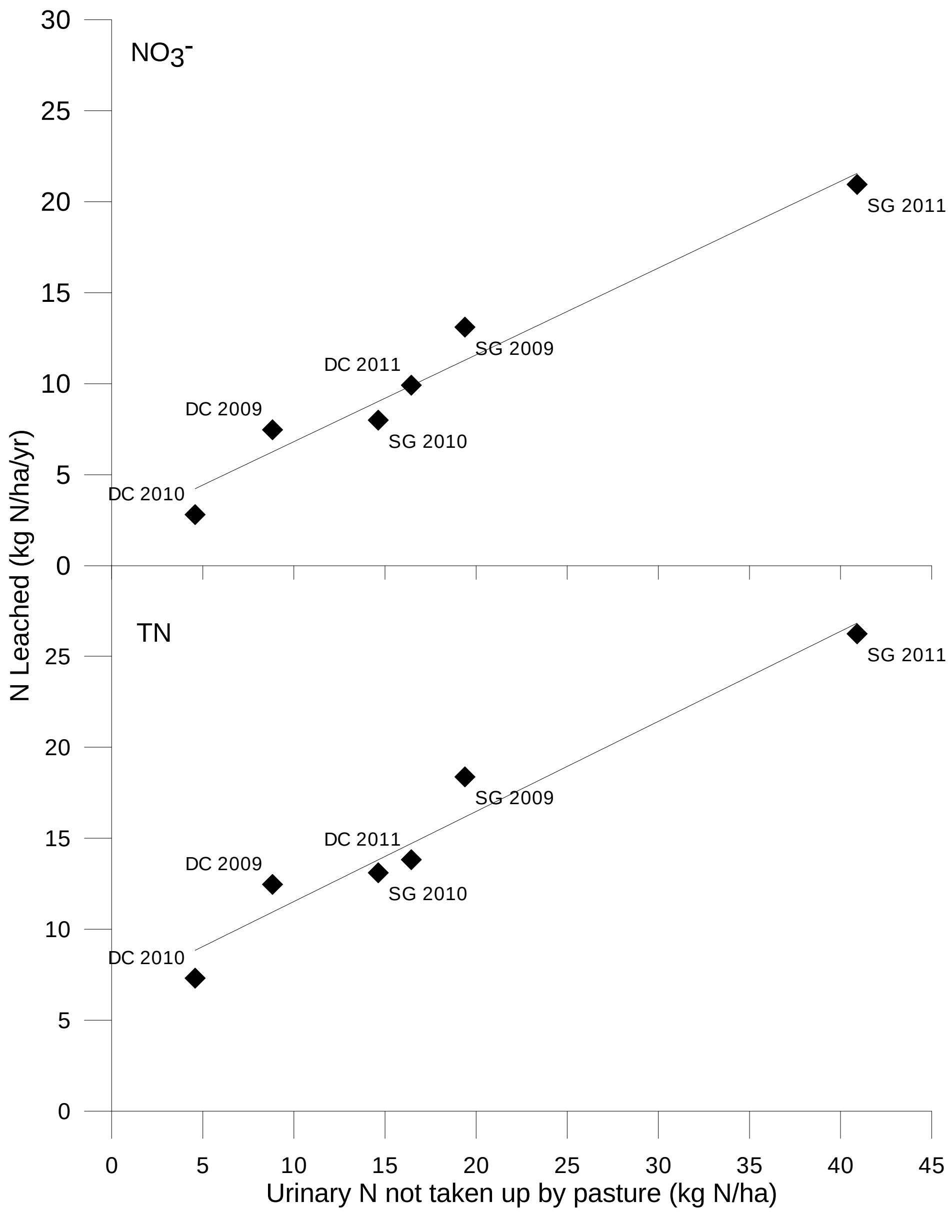

Figure 4.7: Estimated urinary nitrogen not taken up by pasture (kg N/ha) (Eq. 4.7) from autumn grazings and N leached (NO_3^- and TN; kg N/ha) for the subsequent drainage season, for *DC* and *SG* treatment plots in 2009-2011.

Therefore, the results reported around the influences of summer urine patches (Shepherd *et al.*, 2011) and autumn urine patches (Decau *et al.*, 2003, 2004) on subsequent winter grazing both have relevance to the present trial. Furthermore, the results presented from this study go some way towards refining and explaining the importance of the timing of urine deposition on N leaching, particularly on fine-textured, artificially drained soils. This is of practical importance for New Zealand dairy farmers. Where reductions in N leaching are the main objective, one option could be to implement a *DC* management system in the autumn rather than year-round.

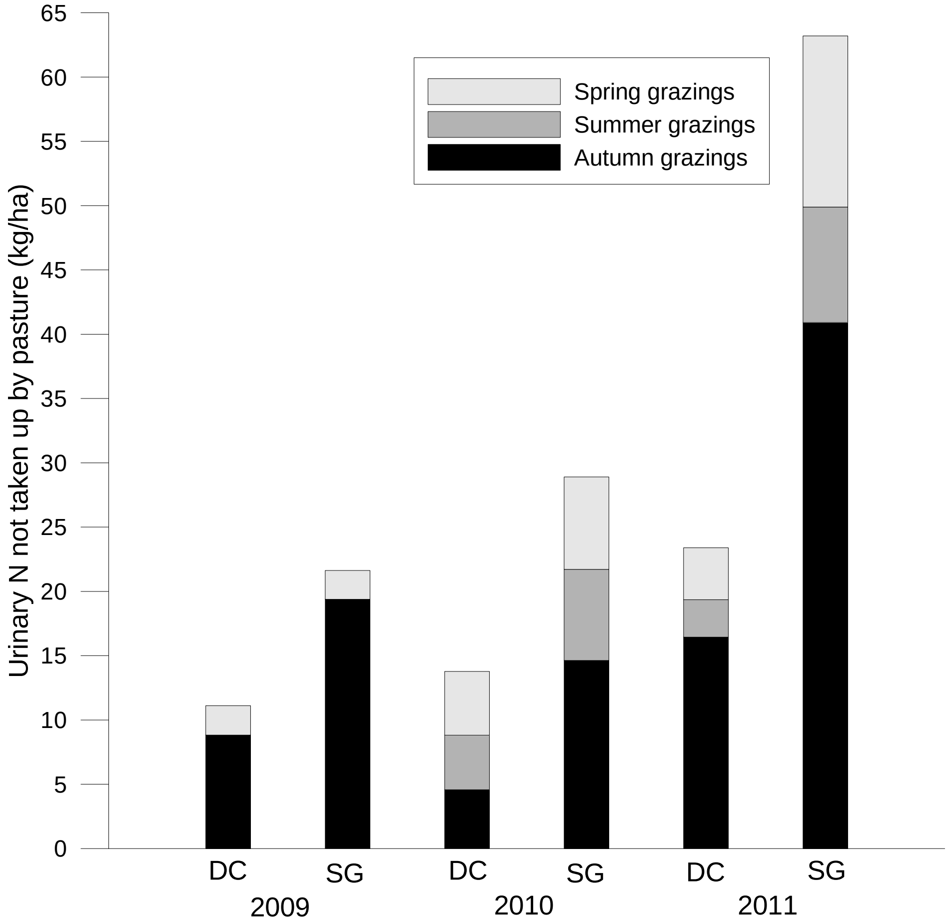

Figure 4.8: Estimated urinary nitrogen not taken up by pasture (kg/ha) for *DC* and *SG* treatments after seasonal grazings from 2009-2011. (Autumn grazings = March-May; summer grazings = December-February; spring grazings = September-November).

In this study, there was relatively high year to year variation in NO_3^- leaching. It is suggested that this variation was due to differences in grazing rotation length and stocking rates during the summer/autumn period, which, in turn, were caused by differences in pasture growth. It is likely that actual farm leaching losses on the No. 4 Dairy Farm would have been higher in some years compared to the *SG* grazing treatment in this study. This reflects differences in the way that the plots and the rest of the farm were grazed. The plots were grazed so that the proportion of grazed pasture in the cow diet was kept relatively constant (ca.75%), year-round. This was achieved by adjusting the grazing rotation length and cow numbers at each grazing. However, the farm typically maintains a shorter grazing rotation and feeds more supplements, particularly in dry conditions. This would result in greater urine returns to pasture during this period.

The current study also found that although slurry was spread evenly over the *DC* plots, the NO_3^- leached was much less than the *SG* plots on average. The form of N in slurry applied to *DC* plots was mostly organic, with ca. 40% as mineral N in 2010/2011. Even in the 2009 drainage season, when 212 kg N/ha was applied as slurry in December 2008 (with a relatively low (16.6%) mineral N content), there was a 43% and 32% reduction in NO_3^- and TN leaching respectively, from the *DC* treatment. Also, all slurry applications were carried out in periods when there was no drainage, in an even manner and with relatively low application depths (6-10 mm depth). Therefore, plant uptake of N and immobilisation in the soil of mineral N from the slurry probably would have resulted in less excess of mineral N in the soil compared with the highly concentrated return of N in urine spots.

The average annual reduction in TN load from *DC* plots (42%) was lower than the reduction in NO_3^- load (52%). It is considered that this results from the relatively constant amount of TON being leached from both treatments, which appeared not to be influenced by grazing treatment. When NO_3^- losses were low for a given volume of drainage, for example in the *DC* treatment in 2010, then NO_3^- became a smaller component of TN. Contributing to the cumulative TN load (Figure 4.4), the TON could have been coming from various sources, such as breakdown of organic matter or direct inputs such as urine or dung that had not yet gone through mineralisation. The consistent TON losses indicate that 4-5 kg/ha/yr of TON would be leached, regardless of the intensity of grazing management and consequent urine deposition. The intercept of the relationship drawn in Figure 4.7 also indicated that if there was no urinary N deposited to be taken up by pasture, there would still be ca. 6 kg TN/ha being leached. This would predominantly be made up of TON (Figures 4.5 and 4.6). The general increase in TON losses with increasing drainage indicates that TON in drainage water can be predicted from the quantity of drainage within a season. For instance, in 2009, where average drainage was 373 mm, then ca. 5 kg TON was leached. In a short drainage season (e.g. 200 mm drainage), it could be expected that ca. 2-3 kg N as TON would be leached. Therefore, TN leaching losses could be estimated by combining the estimated NO_3^- leached (based on urinary return, predominantly in autumn), and TON leached (based on the quantity of drainage in a season).

4.4.2 Nitrogen losses in surface runoff

There were minimal losses of N, both as mineral N and TN in surface runoff over the 3 years of the study (5-9% of total mineral N losses and 14-20% of TN losses). Most of these losses were as TON with the losses of NO_3^- and NH_4^+ being very small. The source of the organic N in surface runoff was most likely dung deposition, which contains mostly organic N (Haynes and Williams, 1993). Total N losses were within the range observed by Houlbrooke (2005) for runoff (3 kg N/ha from non-effluent areas) at the same site.

The overall losses of N in runoff would be dependent on such things as the proximity of dung deposits to the runoff catcher, time since grazing occurred, stocking rate at grazing (number of dung pats), and rainfall intensity. These factors will all be explained in future chapters relating to P and faecal organisms in runoff (Chapters 5 and 6) so are not elaborated on here.

The low levels of N losses in runoff confirm that the loss of N in drainage water, and particularly NO_3^- leaching, is the major pathway of N loss to the aquatic environment and, therefore, should be the main focus of mitigation strategies.

4.5 Conclusion

The implementation of *DC* grazing resulted in marked annual reductions in NO_3^--N leaching (52%) and total N leaching (42%) on a fine-textured, mole and pipe drained soil. The reductions in N leaching were realised in the first season of the study, and hence, the adoption of *DC* grazing by farmers could quickly result in reductions in N losses. Nitrate concentrations in drainage were high at the beginning of all drainage seasons but decreased rapidly to very low levels within the first 200 mm of drainage. This suggests that much of the effect of *DC* grazing on N leaching is due to the decrease in pre-drainage season urine patches.

The quantity of NO_3^- leached in drainage was strongly related to the quantity of autumn-deposited urinary N estimated to not have been taken up by pasture i.e. the fraction that remained in the soil. Urine deposited at spring and summer grazings had less effect on the NO_3^- leaching load. Therefore, implementation of *DC* grazing could be targeted at certain times of the year, namely autumn, rather than practised year-round.

The losses of TN through leaching followed a similar pattern to NO_3^- due largely to the high proportion of NO_3^- in the TN. The remaining component of TN, as TON, was largely related to drainage volume. Therefore, TN losses could be predicted by using drainage volume (TON) and grazing numbers, durations and N concentrations of pasture (NO_3^-).

Although the *DC* grazing treatment plots received slurry applications, they still achieved a 42% reduction in TN leaching compared to the *SG* plots on average. However, timing of this slurry should be managed so as to optimize pasture response (Chapter 3), while avoiding the risks associated with drainage and increasing N intake and urinary output by cows in later grazings, particularly the autumn months.

The minimal losses of N via surface runoff were similar to results that have previously been reported from this site. Most of the runoff losses were as organic N, most likely from dung deposited throughout the year.

Overall, the substantial reductions in N leaching achievable with *DC* grazing exceed the dairy industry’s goal of 30% (DairyNZ, 2010a). Therefore, *DC* grazing provides an effective, proven mitigation strategy that farmers could adopt to reduce N losses to waterways.

4.6 References

Blakemore, L.C., Searle, P.L., Daly, B.K., 1987. Methods for chemical analysis of soils. New Zealand Soil Bureau Scientific Report 80.

Bowler, D.G., 1980. The drainage of wet soils. Massey University, Palmerston North.

Castillo, A.R., Kebreab, E., Beever, D.E., France, J., 2000. A review of efficiency of nitrogen utilisation in lactating dairy cows and its relationship with environmental pollution. J. Anim. Feed Sci. 9, 1-32.

DairyNZ, 2010a. Dairy Industry Strategy - directions to the vision. Hamilton, NZ.

DairyNZ, 2010b. Facts and Figures for New Zealand Dairy Farmers. DairyNZ.

de Klein, C.A.M., Ledgard, S.F., 2001. An analysis of environmental and economic implications of nil and restricted grazing systems designed to reduce nitrate leaching from New Zealand dairy farms. I. Nitrogen losses. New Zealand Journal of Agricultural Research 44, 201-215.

de Klein, C.A.M., Smith, L.C., Monaghan, R.M., 2006. Restricted autumn grazing to reduce nitrous oxide emissions from dairy pastures in Southland, New Zealand. Agriculture, Ecosystems and Environment 112, 192-199.

Decau, M.L., Simon, J.C., Jacquet, A., 2003. Fate of Urine Nitrogen in Three Soils throughout a Grazing Season. Journal of Environmental Quality 32, 1405-1413.

Decau, M.L., Simon, J.C., Jacquet, A., 2004. Nitrate Leaching under Grassland as Affected by Mineral Nitrogen Fertilization and Cattle Urine. Journal of Environmental Quality 33, 637-644.

Di, H.J., Cameron, K.C., 2002. Nitrate leaching in temperate agroecosystems: sources, factors and mitigating strategies. Nutrient Cycling in Agroecosystems 46, 237-256.

Fonterra, 2013. http://www.fonterra.com/nz/en/Financial/Global+Dairy+Industry.

Hanly, J.A., 2012. Management practices and technologies for reducing nitrogen and phosphorus losses from soils receiving farm dairy effluent. PhD Thesis. Massey University, Palmerston North.

Haynes, R.J., Williams, P.H., 1993. Nutrient cycling and soil fertility in the grazed pasture ecosystem. Advances in Agronomy 49, 119-199.

Holmes, C.W., Brookes, I.M., Garrick, D.J., MacKenzie, D.D.S., Parkinson, T.J., Wilson, G.F., 2007. Milk Production from Pasture - Principles and Practices. Massey University, Palmerston North.

Horne, D.J., Singleton, P.L., 1997. Managing wet soils in wet years: a cautionary tale. Dairy Farming Annual 49, 117-124.

Hosomi, M., Sudo, R., 1986. Simultaneous determination of total nitrogen and total phosphorus in freshwater samples using persulfate digestion. International Journal of Environmental Studies 27, 267-275.

Houlbrooke, D.J., 2005. A study of the quality of artificial drainage under intensive dairy farming and the improved management of farm dairy effluent using 'deferred irrigation'. PhD Thesis. Massey University, Palmerston North.

Houlbrooke, D.J., Horne, D.J., Hedley, M.J., Hanly, J.A., Snow, V.O., 2003. The impact of intensive dairy farming on the leaching losses of nitrogen and phosphorus from a mole and pipe drained soil. Proceedings of the New Zealand Grassland Association 65, 179-184.

Houlbrooke, D.J., Horne, D.J., Hedley, M.J., Snow, V.O., Hanly, J.A., 2008. Land application of farm dairy effluent to a mole and pipe drained soil: implications for nutrient enrichment of winter-spring drainage. Australian Journal of Soil Research 46, 45-52.

Kebreab, E., France, J., Beever, D.E., Castillo, A.R., 2001. Nitrogen pollution by dairy cows and its mitigation by dietary manipulation. Nutrient Cycling in Agroecosystems 60, 275-285.

Ledgard, S.F., Menneer, J.C., 2005. Nitrate leaching in grazing systems and management strategies to reduce losses. In: Currie, L.D., Hanly, J.A. (Eds.), Developments in fertiliser application technologies and nutrient management. Fertiliser and Lime Research Centre, Massey University, Palmerston North, pp. 79-92.

Longhurst, R.D., Luo, J., O'Connor, M.B., Pow, T., 2006. Herd Homes: nutrient management and farmer perceptions of performance. Proceedings of the New Zealand Grassland Association 68, 309-313.

Luo, J., Ledgard, S.F., Klein, C.A.M., Lindsey, S.B., Kear, M., 2008. Effects of dairy farming intensification on nitrous oxide emissions. Plant & Soil 309, 227-237.

Menneer, J.C., Ledgard, S.F., McLay, C., Silvester, W., 2003. The effect of a single application of cow urine on annual N_2 fixation under varying simulated grazing intensity, as measured by four ^{15}N isotope techniques. Plant and Soil 254, 469-480.

Monaghan, R.M., de Klein, C.A.M., Muirhead, R.W., 2008. Prioritisation of farm scale remediation efforts for reducing losses of nutrients and faecal indicator organisms to waterways: A case study of New Zealand dairy farming. Journal of Environmental Management 87, 609-622.

Monaghan, R.M., Paton, R.J., Drewry, J.J., 2002. Nitrogen and phosphorus losses in mole and tile drainage from a cattle-grazed pasture in eastern Southland. New Zealand Journal of Agricultural Research 45, 197-205.

Parliamentary Commissioner for the Environment, 2012. Water Quality in New Zealand: Understanding the Science. Parliamentary Commissioner for the Environment, Wellington.

Saggar, S., Andrew, R.M., Tate, K.R., Hedley, C.B., Rodda, N.J., Townsend, J.A., 2004. Modelling nitrous oxide emissions from dairy-grazed pastures. Nutrient Cycling in Agroecosystems 68, 243-255.

Salazar, M.E., Hedley, M.J., Horne, D.J., 2010. Using turnips to reduce soil K loading on the effluent block. Proceedings of the New Zealand Grassland Association 72, 247-250.

Sharpley, A.N., Syers, J.K., 1979. Loss of nitrogen and phosphorus in tile drainage as influenced by urea application and grazing animals. New Zealand Journal of Agricultural Research 22, 127-131.

Shepherd, M., Phillips, P., Snow, V., 2011. The challenge of late summer urine patches in the Waikato region. In: Currie, L.D., Christensen, C.L. (Eds.), Adding to the knowledge base for the nutrient manager, Occasional Report No. 24. Fertilizer & Lime Research Centre, Palmerston North.

Silva, R.G., Cameron, K.C., Di, H.J., Hendry, T., 1999. A lysimeter study of the impact of cow urine, dairy shed effluent, and nitrogen fertiliser on nitrate leaching. Australian Journal of Soil Research 37, 357-369.

Chapter 5 The impact of Duration-controlled grazing on phosphorus losses in surface runoff and drainage water

5.1 Introduction

Large reductions in the leaching of nitrate (52%) and total nitrogen (42%) from dairy pastures has been achieved by using duration-controlled (*DC*) grazing (Chapter 4), compared to standard grazing (*SG*). This was accomplished by standing cows off pasture, thereby reducing the number of urine spots being deposited in the paddocks. As a result, less nitrate (NO_3^-) accumulated in the soil prior to the commencement of seasonal drainage, leading to lower NO_3^- losses in drainage (Sharpley and Syers, 1979; Ledgard and Menneer, 2005).

Whilst successful at significantly reducing N loss from grazed pastures, it is unclear whether *DC* grazing reduces phosphorus (P) losses. This uncertainty arises because the main pathway of P loss is different to N, namely overland flow or surface runoff. The forms of P in surface runoff and drainage include dissolved reactive P (DRP), total dissolved P (TDP) and total P (TP) (Ryden *et al.*, 1973; Sharpley and Syers, 1979; Houlbrooke *et al.*, 2008). Much of the TP lost through surface runoff is as dissolved P, both organic and inorganic, and particulate P which is the difference between TDP and TP (Sharpley and Syers, 1979).

Haygarth and Sharpley (2000) described 'overland flow' as the most correct term for water moving over the soil surface during heavy rain. They defined runoff as a general hydrological term, but stated that it could be qualified when using a term such as 'surface runoff'. Conversely, Ryden *et al.* (1973) used surface runoff and overland flow interchangeably. This chapter will refer to 'runoff' when describing losses of P through water collected from the soil surface.

Phosphorus and N cycles in grazed pasture systems differ in the following respects: firstly, P is mostly re-deposited on pasture via dung, compared to N through urine (Haynes and Williams, 1993); secondly, P is mostly lost in runoff, with comparatively small losses occurring in drainage (Ryden *et al.*, 1973); thirdly, and perhaps most importantly, the major sources of P loss to water include dung, fertiliser and soil (Sharpley, 1977;

McDowell *et al.*, 2008). Dung contributes 30-40% to P losses from a grazed pasture system (this figure being higher if runoff occurs when the dung is wet or freshly deposited and has not skinned over (McDowell *et al.*, 2006)). It has been estimated that fertiliser P makes up about 10% of P losses. This is particularly so when the P fertiliser is in a soluble P form, and when it has been applied directly prior to runoff occurring (McDowell *et al.*, 2008). The remainder of P losses come from soil. This is often as particulate P, and can increase with higher incidences of treading damage of pastures (Sharpley, 1977). However, dissolved P is also generated if the surface soil has been enriched in P by animal camping (excess dung deposition) or fertiliser application (Sharpley, 1977; Curran-Cournane *et al.*, 2010). Soil type (McDowell *et al.*, 2003) and topography (Ahuja *et al.*, 1982) are also factors in soil P loss (McDowell *et al.*, 2008; McDowell and Wilcock, 2008).

When using a *DC* grazing system, the stored effluent from standing cows off is applied to paddocks as a slurry (Chapters 3 and 4). Therefore, any potential reductions in P losses as a result of lower dung deposition on to pastures (McDowell and Sharpley, 2002) from *DC* grazing needs to be compared to any additional P loss caused by slurry applications. The application of farm dairy effluent has been shown to cause accelerated P loss in runoff and drainage water from grazed pastures at the same research site as the current study. In a study by Houlbrooke *et al.* (2008), farm dairy effluent applications were made to pasture in summer, at a total depth of 80 mm (spread over four applications), when soil water deficits were suitable for irrigation to not cause direct drainage. Annual TP losses in the following runoff and drainage season were 0.6 and 0.73 kg P/ha for non-effluent areas, and losses increased by ca. 10% and 100%, respectively, for areas that received effluent (Houlbrooke *et al.*, 2008). Furthermore, one poorly timed effluent application in the spring (25 mm when soil water deficit was 7 mm) resulted in a >300% increase in TP loss, to 1.91 kg P/ha through runoff and 2.30 kg P/ha through drainage. This research highlighted the importance of applying any effluent when there was a low risk of runoff occurring, and the same principle would apply to slurry application in a *DC* grazing system.

The objective of this chapter is to quantify, experimentally, the impact of a *DC* grazing system on P losses in runoff and drainage, and to test the hypothesis that *DC* grazing with low-depth slurry application, but lower dung deposition, will lead to lower P loss to water compared with *SG* grazing.

5.2 *Methods*

5.2.1 Trial site

Detailed descriptions of the trial site, experimental design and grazing management have been reported in Chapters 3 and 4.

The *SG* treatment was managed as ‘best-practice’ standard grazing for the Manawatu region, which means grazing did not occur if significant treading damage was likely. To prevent treading damage, the trial site (both *DC* and *SG* plots) was not grazed when the soil moisture deficit was less than 5 mm (Horne, 2012).

The roughness of the surface soil was measured using the chain method of Saleh (1993). This was conducted twice throughout the duration of the study, in November 2009, and October 2011, on every plot.

5.2.2 Soil fertility and fertilisers

Annual maintenance P and sulphur (S), and strategic N fertilisers were applied to all plots (both *SG* and *DC*) at various times throughout the course of the experiment. Soil test results and details of fertiliser applications have been reported previously in Chapter 3, with the pH and Olsen P values repeated in Table 5.1.

Table 5.1: Soil test results (pH and Olsen phosphorus (P)) for the trial site

	pH		Olsen P (μg/g soil)	
Soil test date	***DC***	***SG***	***DC***	***SG***
Sept 2008			32.9	30.3
May 2010	5.8	5.8	30.8	29.7
May 2012	5.7	5.7	37.1	39.5

5.2.3 Slurry application

A nutrient balance model (Salazar *et al.*, 2010) and the difference in the number of dung depositions between the two treatments were used to guide the rate of return of collected excreta to the *DC* treatment plots in the form of a slurry (more detail has been provided in Chapters 3 and 4). A sub-sample of slurry from each application was taken for analysis of inorganic and total N and P. From this, the quantity of nutrients (N, P and K) applied in each application was estimated (P presented in Table 5.2). In the 2008/09 year the slurry applications began with one large application, with 1.5 - 2 years' worth of nutrient content. In 2009/10, with no slurry application, a 20% decline in pasture accumulation was observed on the *DC* plots compared with the *SG* plots. In 2010/2011, only a 9% decline in pasture accumulation was observed on the *DC* plots when four smaller applications of slurry had occurred (Chapter 3).

Table 5.2: Quantity of phosphorus returned (kg/ha) to *DC* plots in slurry applications

Year	Month applied	Quantity of P applied per application *(kg P/ha)*
2008/09	Dec 2008	48
2009/10		0
2010/11	Oct 2010	5
	Nov 2010	2
	Feb 2011	14
	Apr 2011	5

5.2.4 Rainfall, surface runoff and drainage water volume measurements, and water analysis

Rainfall was measured on site using a Davis Rain Collector II and Odyssey Logger, and this was compared with manual measurements. A water balance simulation model was used to calculate the daily evapotranspiration and soil water deficit, based on Scotter *et al.* (1979) and described in Chapters 3 and 4.

Runoff (from the 50 m^2 subplots) and drainage water (from the 850 m^2 grazing area) was channelled through drainage pipes into individual tipping-bucket flow meters located in sampling pits nearby. Each tipping-bucket (ca. 2 L/tip for runoff and ca. 5 L/tip for drainage collection) was calibrated dynamically to account for larger tip volumes at higher flow rates. All tipping buckets were instrumented with data loggers to provide continuous measurements of flow rate. An analysis tool, written in Microsoft Excel, was used to calculate flow rates from tipping bucket data. During each runoff and drainage event a proportion of the runoff and drainage water from every second tip of the tipping bucket flow meter was automatically collected to provide a volume-proportioned (ca. 0.1%) mixed sample for P analysis.

Runoff and drainage water samples were filtered through a 0.45 μm filter and the filtrate analysed for DRP using the phosphomolybdate method of Murphy & Riley (1962). Total P in unfiltered samples was digested using the alkali persulphate method (Hosomi and Sudo, 1986) and subsequently analysed using the phosphomolybdate method of Murphy & Riley (1962).

5.2.5 Statistical analysis

Treatment differences were tested for significance using ANOVA in Microsoft Excel. Following this, stepwise regression was performed using Minitab (version 16) to consider the most important factors determining the loss of P in runoff.

5.3 Results

5.3.1 Runoff volumes

There was a very large variation in runoff depth and the subsequent volume of water collected from some plots in all years (Figure 5.1). One cause of this is that in 2 of the replicates, runoff flowed from an adjacent paddock and was not a true measure of how much runoff was being generated from those plots. Also, some runoff plot borders were damaged during previous grazings, allowing channelised flow generated in storm events to run through the runoff plots from the larger drainage plots. The extent of this damage was not recorded, thus the number of damaged borders in each year can not be reported.

To quantify this variation, and to verify which plots were to be discounted from further analysis, measured runoff depth was plotted against measured rainfall for each discrete event (Figure 5.1). If the majority of runoff events from a plot exceeded the immediate rainfall, then data from these plots were not used for any further runoff analysis. In the case of 4 of the 14 plots, several runoff depths exceeded rainfall. This resulted in 3 replicates from the *DC* treatment and 1 from the *SG* treatment having large and unreasonable runoff volumes, in all years of sampling. To maintain a balanced replicated scenario, the next 2 most unsuitable plots from *SG* were discounted and 4 replicates for each treatment were used in the remaining analysis (Figure 5.2). The method of discounting obvious anomalies within the data set by comparing runoff with rainfall was also used in a runoff study by Monaghan and Smith (2012). In the present study, for the four replicates in each treatment, runoff was generally less than rainfall, as drainage occurred at the same time (Chapter 4).

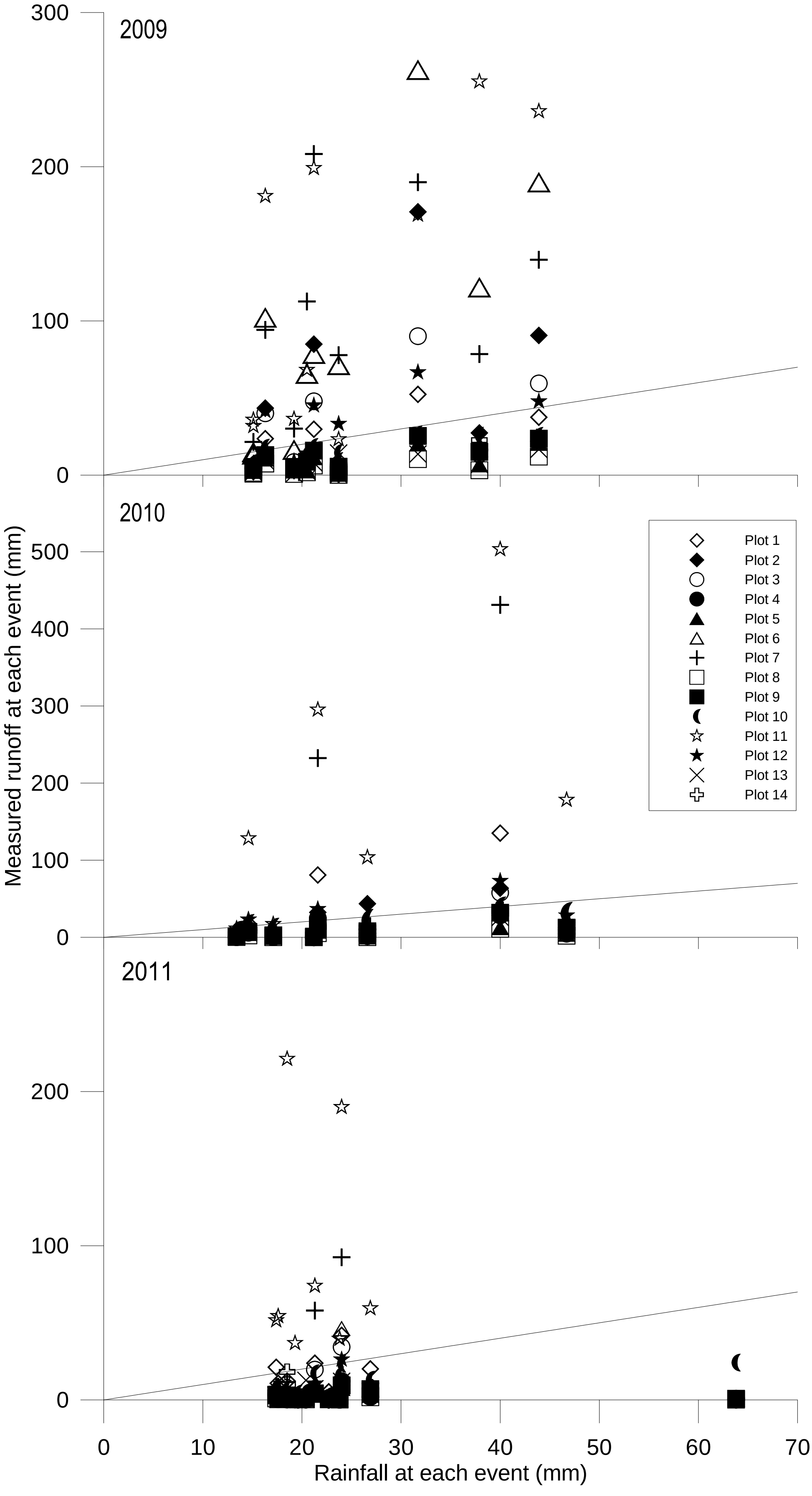

Figure 5.1: The relationship between measured runoff at each event and rainfall (mm) for each of 14 plots. The ruled line represents a 1:1 relationship, where rainfall equals runoff depth.

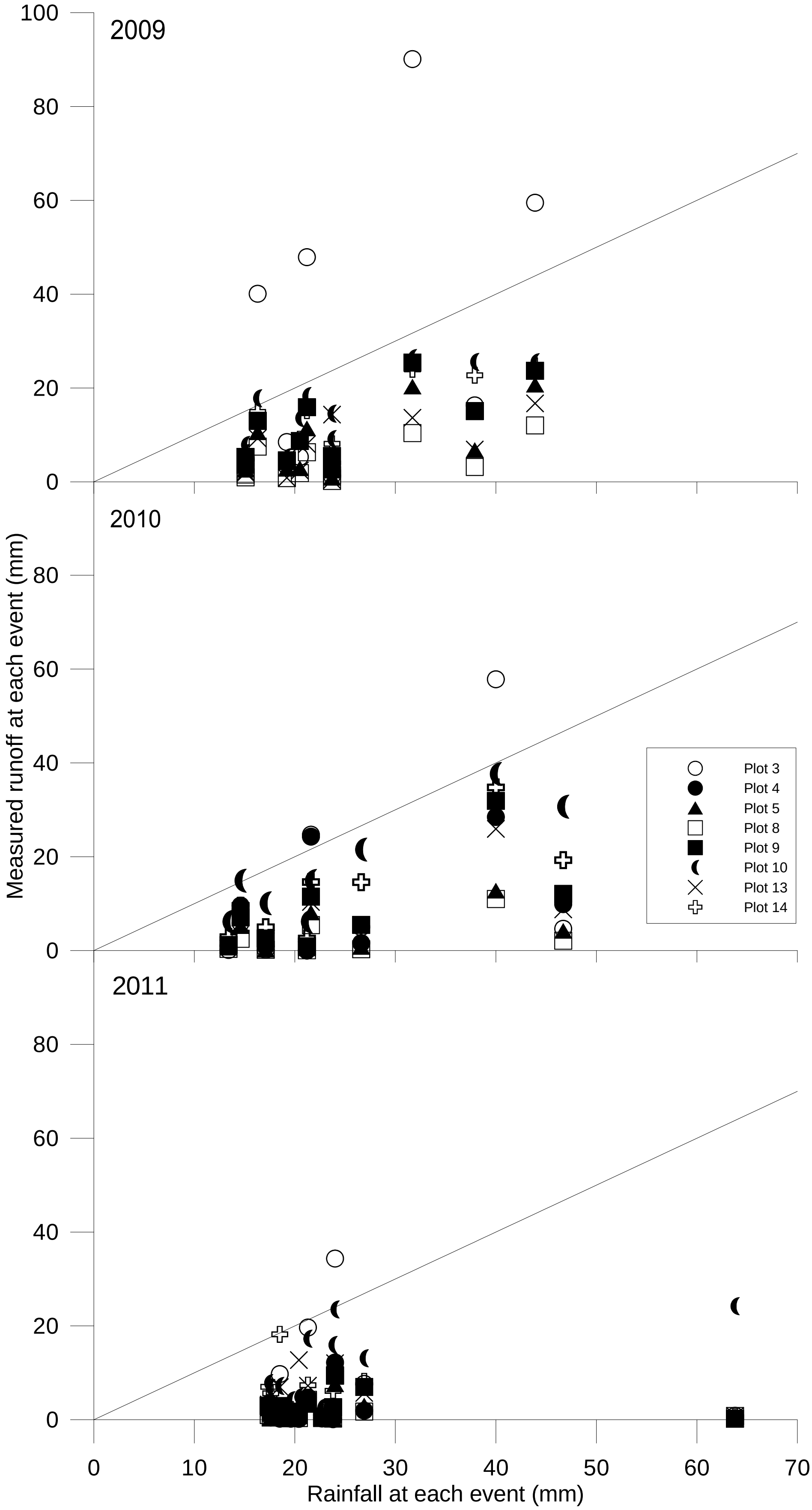

Figure 5.2: The relationship between measured runoff at each event and rainfall for each plot after outliers had been removed. The ruled line represents a 1:1 relationship, where rainfall equals runoff depth.

5.3.2 Climatic factors influencing runoff and drainage

The time of year that runoff occurred varied in each year of the study. The majority of runoff occurred between September and December in 2009 (after measuring devices were in place), between September and October in 2010, and was spread between April and November in 2011 (Figure 5.3). Therefore, all data shown in Figure 5.3 begin in April. In most cases, apart from one very small inconsistent event at the start of 2011, runoff only occurred when the soil was at field capacity (zero water deficit). This was due to the rainfall intensity exceeding the surface soil infiltration rate. For drained Tokomaru silt loam, surface soil infiltration rate has been documented to be a wide range, from 2 – 30 mm/hr (Rennes *et al.*, 1977; Horne, 2012).

Only when the soil was at field capacity or wetter (i.e. zero soil water deficit), and rainfall exceeded 12 mm/day, did rainfall intensity became important in determining whether runoff would occur. For example, over the study period from 2009-2011, when the soil water deficit was zero and rainfall exceeded 12 mm in a 24 hour period, and there was rainfall intensity within that period of at least 0.84 mm/10 minutes, did runoff occur. If any one of these three factors did not occur, then there was no runoff.

The soil was at field capacity for 63, 51 and 56 days in 2009, 2010 and 2011, respectively (Figure 5.3). The total depth of runoff from both treatments averaged 132, 76 and 103 mm/yr in 2009, 2010 and 2011, respectively. This equated to 10.0%, 7.3% and 8.7% of the total annual rainfall for each year, respectively, whereas drainage accounted for 28.3%, 30.4% and 34.0% of the total annual rainfall, respectively. The largest average runoff events in each year were 27.6, 27.0 and 10.2 mm. These events were caused by rainfall depths of 30.7, 61.6 and 17.9 mm, and peak rainfall intensities of 1.96, 1.96 and 3.92 mm per 10 minutes, respectively.

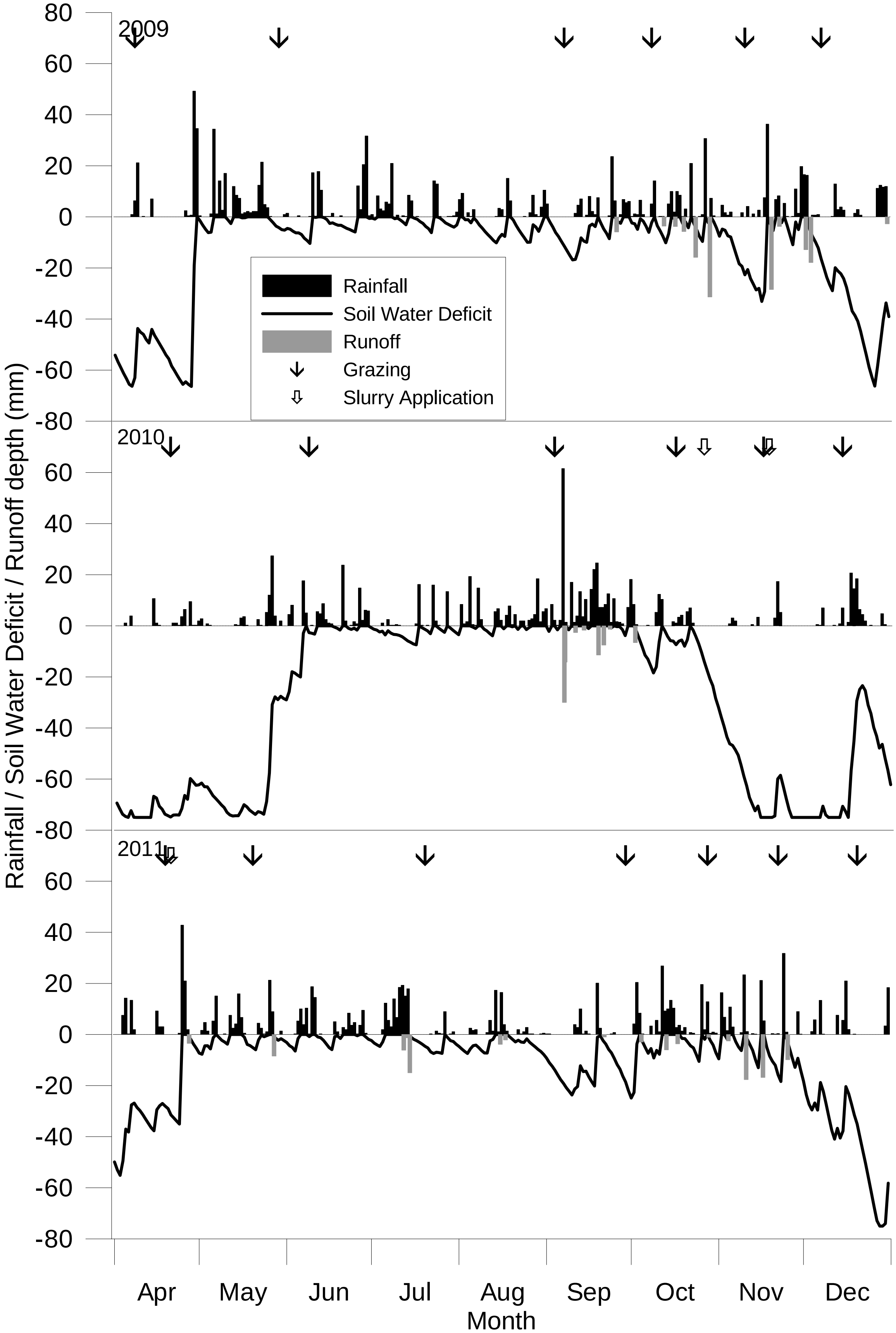

Figure 5.3: Measured rainfall, modelled soil water deficit, and average runoff depth (mm) at each event from *DC* and *SG* treatments, from April to December in 2009-2011. Soil water deficit and drainage depth are absolute values, but have been shown as negative values to add clarity to the figure.

5.3.3 Runoff depth and phosphorus concentrations

Between the 4 replicates, there was still a large variation in the runoff depths observed between plots at different events. This caused high standard errors within each treatment (Figure 5.4), and there was insufficient statistical evidence to support a significant difference between the depths of runoff for the *SG* and *DC* treatments. There was also no relationship between season or time of year and runoff depth (Figure 5.4).

The average concentration of DRP (ppm) in runoff water was generally lower for each event from the *DC* plots but was not always significantly ($P<0.05$) lower than the *SG* plots (Figure 5.5). Weighted average DRP concentrations from the *SG* treatment plots were 0.43, 0.31 and 0.50 ppm in 2009, 2010 and 2011, respectively, and concentrations from the *DC* plots were 0.23, 0.16 and 0.28 ppm, respectively. Although the concentration of DRP between plots was relatively uniform for each treatment at each event, the large variability in runoff depth had a considerable effect, resulting in insignificant differences in DRP load (kg P/ha) in runoff between treatments.

In 2009 and 2010, there was no indication of higher DRP concentrations following P fertiliser, grazing or slurry applications. However, in 2011 there was evidence of the first runoff event, one month after P fertiliser application, having significantly higher concentrations of DRP compared to other events in that year (Figure 5.5). This runoff event in 2011 was also within one week of grazing and slurry application and therefore all three occurrences are likely to have contributed to higher P concentrations.

5.3.4 Dissolved reactive phosphorus loads in runoff

The cumulative DRP load lost as runoff from *SG* plots was 0.51, 0.22 and 0.42 kg in 2009, 2010 and 2011, respectively (Table 5.3). There was a general reduction in DRP losses in runoff from *DC* plots compared with *SG* plots of 37%, 41% and 19% for 2009, 2010 and 2011, respectively (Table 5.3), with an average reduction of 32% over all 3 years. However, this reduction was not statistically significant using analysis of variance ($P>0.15$). Aside from the high variation in runoff volumes between plots, a reason for the small and insignificant reduction between treatments was possibly due to grazing events on both *DC* and *SG* plots being conducted to avoid treading damage.

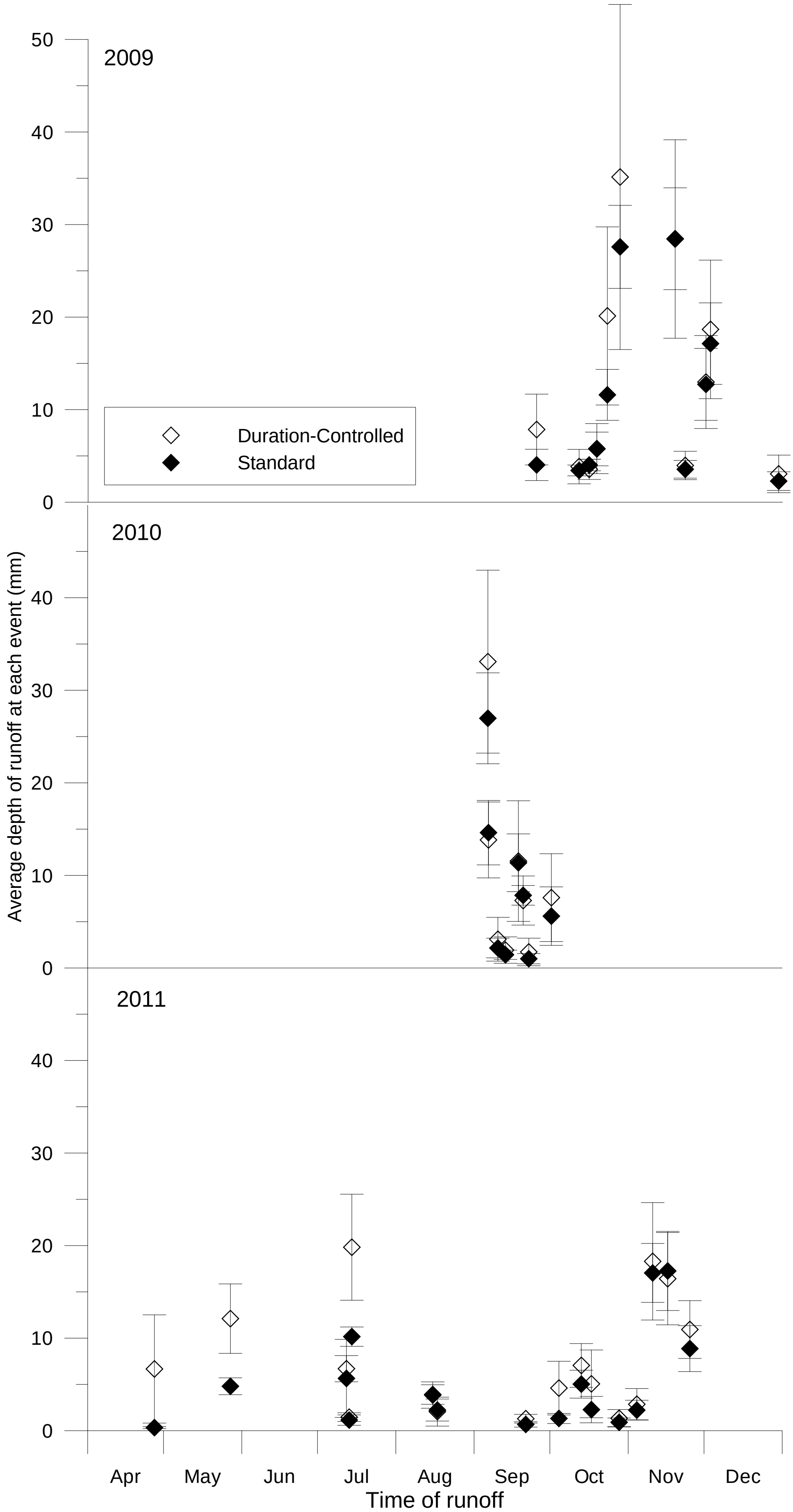

Figure 5.4: Average depth (mm) of runoff at each event for *SG* and *DC* treatments in 2009-2011. Error bars show standard error of the mean.

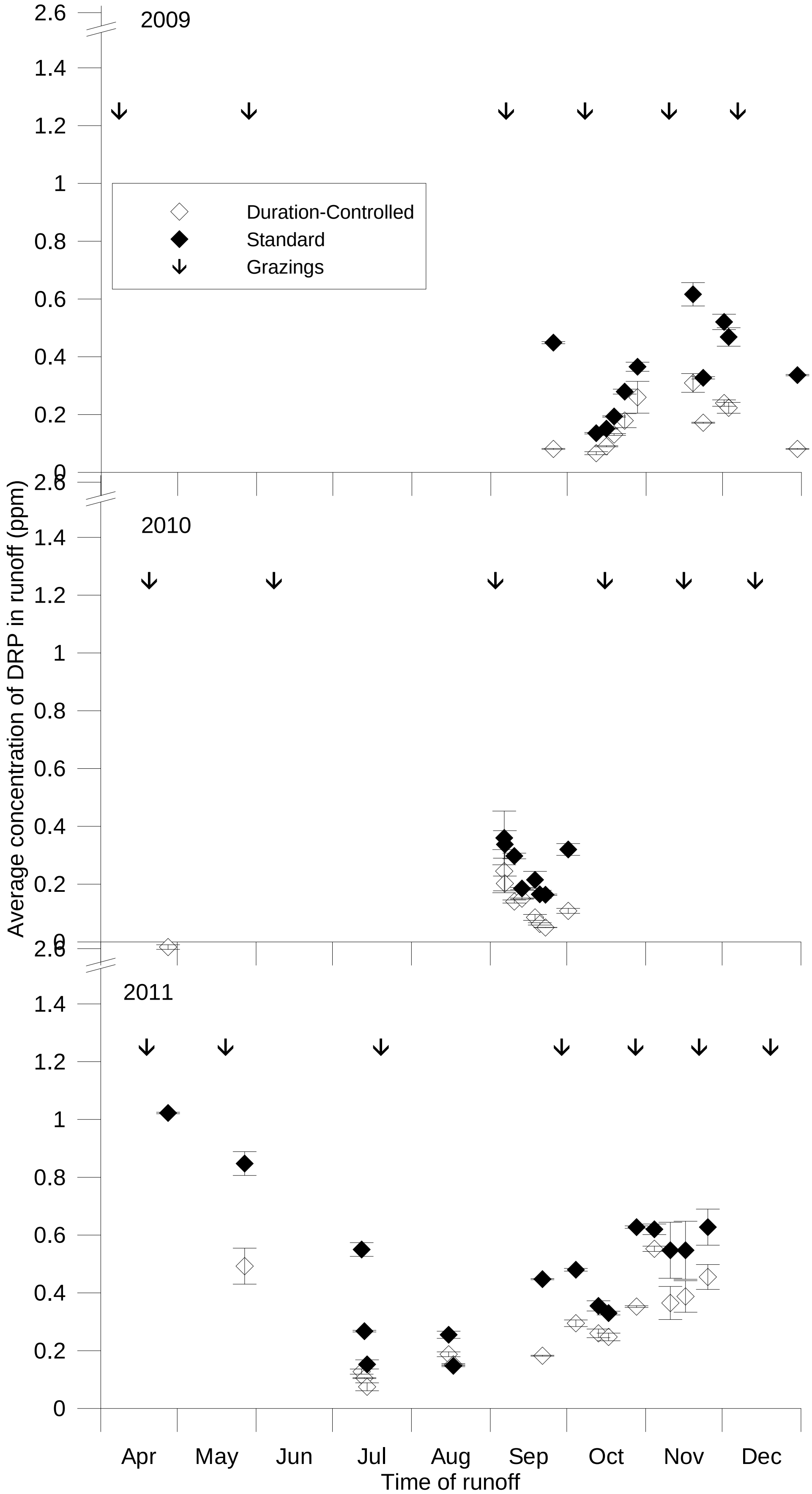

Figure 5.5: Average concentration of dissolved reactive phosphorus (ppm) for *SG* and *DC* treatments at each runoff event in 2009-2011. Error bars show standard error of the mean.

The relationship between runoff depth and DRP load was examined with linear regression. Depth of runoff (mm) in each runoff event was the most influential factor determining DRP load. This is illustrated in Figure 5.6, where the amount of DRP in runoff water increased with the depth of runoff, independent of treatment. This explained 89-98% (R^2=0.89-0.98) of the DRP load lost as runoff for the *DC* treatment in 2009 and 2010 and in all years for the *SG* treatment, and 54% (R^2=0.54) for the *DC* treatment in 2011.

There was no significant effect of the days since grazing (*dsg*) on DRP load (kg P/ha) (data not shown). The number of dung pats deposited was influenced by the time spent by cows grazing and was the main treatment effect.

Stepwise regression was then performed to determine the other parameters that most influenced DRP loads in runoff. This resulted in the following regression (Eq. 5.1) which explained 70% of the variation in DRP load in runoff over the 3 years:

$$\text{DRP load (kg/ha)} = 0.0031*runoff + 0.0128*trt - 0.0002*dsg + 0.00004*cows + 0.0008*hrgr \quad \textbf{Eq. 5.1}$$

where *runoff* = depth of runoff (mm); *trt* = treatment; *dsg* = days since grazing; *cows* = cows grazed/ha at last grazing; and *hrgr* = hours spent grazing at last grazing.

Table 5.3: Amounts (kg P/ha/yr) of dissolved reactive phosphorus and total phosphorus lost in runoff, and average depth of runoff, from *DC* and *SG* treatments for 2009-2011 (TP value in italics includes outlier identified in 2011; values in parentheses show standard error of the mean for each cumulative load).

	DC			***SG***		
Year	**Runoff** *(mm)*	**DRP** *(kg P/ha/yr)*	**TP** *(kg P/ha/yr)*	**Runoff** *(mm)*	**DRP** *(kg P/ha/yr)*	**TP** *(kg P/ha/yr)*
2009	143.4	0.33 (0.15)	0.79 (0.30)	120.6	0.51 (0.08)	1.11 (0.14)
2010	80.1	0.13 (0.03)	0.34 (0.07)	71.0	0.22 (0.07)	0.54 (0.17)
2011	120.5	0.34 (0.07)	0.51 (0.15); *0.83 (0.24)*	83.8	0.42 (0.15)	0.76 (0.23)
Average	114.7	0.27	0.55 *0.65*	91.8	0.38	0.80

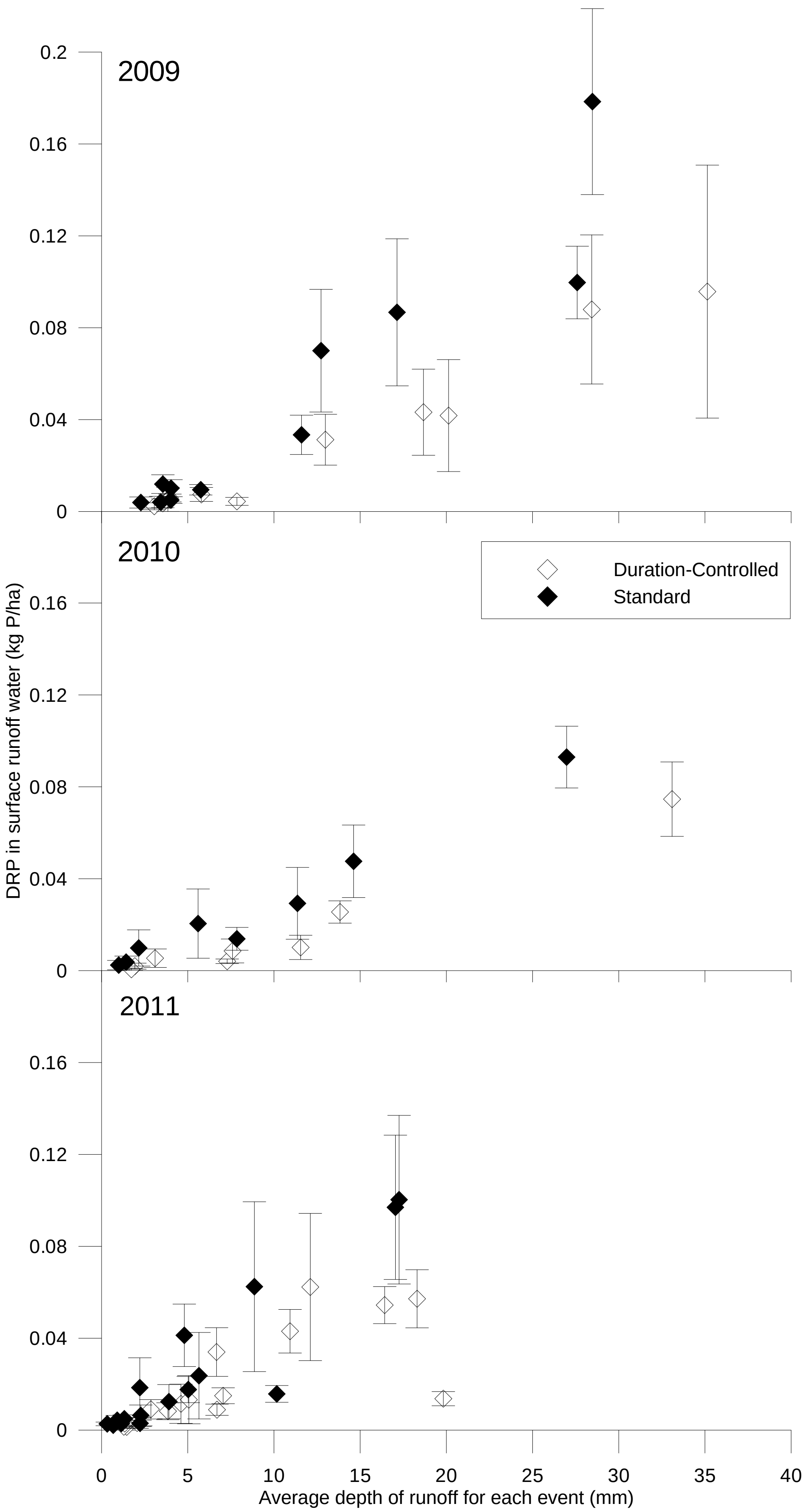

Figure 5.6: Dissolved reactive phosphorus load (kg P/ha) in runoff compared with depth of runoff at each event for *SG* and *DC* treatments from 2009-2011.

5.3.5 Total phosphorus loads in runoff

Total P in runoff from a grazed pasture includes dissolved inorganic, organic and particulate P components. The annual TP load (kg/ha) was linearly related to DRP load (R^2 =0.92-0.99 in each year for each treatment).

The cumulative TP load lost as runoff from the *SG* treatment plots was 1.11, 0.54 and 0.76 kg/ha in 2009, 2010 and 2011 respectively. The comparatively lower losses in 2010 demonstrated that there were less runoff events in that year. There was an overall average reduction in annual TP load leaving *DC* plots compared with *SG* plots for each of the three years, albeit not significant ($P>0.05$) for any of the years. The reductions were 29% and 37% in 2009 and 2010 respectively, and an increase of 9% in 2011 (19% average reduction over all 3 years; Table 5.3).

In 2011, one *DC* plot had a much higher runoff P concentration (4.50 ppm TP; other individual data not shown) than the others analysed. The other replicates all had concentrations < 0.8 ppm at that event. When multiplied by volume, this produced an average TP load of 0.31 kg/ha (Figure 5.7), with a high standard error of 0.19, compared with all other loads having standard errors ≤ 0.05. The cause of this higher concentration outlier could have been due to a number of factors. Grazing had occurred only 6 days prior, so dung pats deposited close to the catcher or disturbance of soil were probably the most likely causes for the higher loss of TP from that replicate at that time. The influence of this outlier was most obvious when comparing the increase in TP load from *DC* plots with *SG* plots in 2011 (Table 5.3). If the outlier was omitted, then the TP loss from *DC* plots was 0.51 kg/ha; a reduction of 33% compared with the *SG* plots in 2011, resulting in a 32% average reduction over all 3 years.

Across both *DC* and *SG* treatments, the TP load (kg/ha) was mostly related to depth of runoff (upward trend; Figure 5.7), as with DRP. When a stepwise regression (Eq. 5.2) was performed on TP for all years, the depth of runoff (mm), treatment and days since last grazing respectively were the factors that were highly significant ($P<0.01$) in predicting TP load. The stepwise regression equation (Eq. 5.2) explained 58% of the variation in annual runoff P load from all treatments:

$$\text{TP load (kg P/ha/yr)} = 0.0062 * runoff + 0.0258 * trt - 0.00076 * dsg \qquad \textbf{Eq. 5.2.}$$

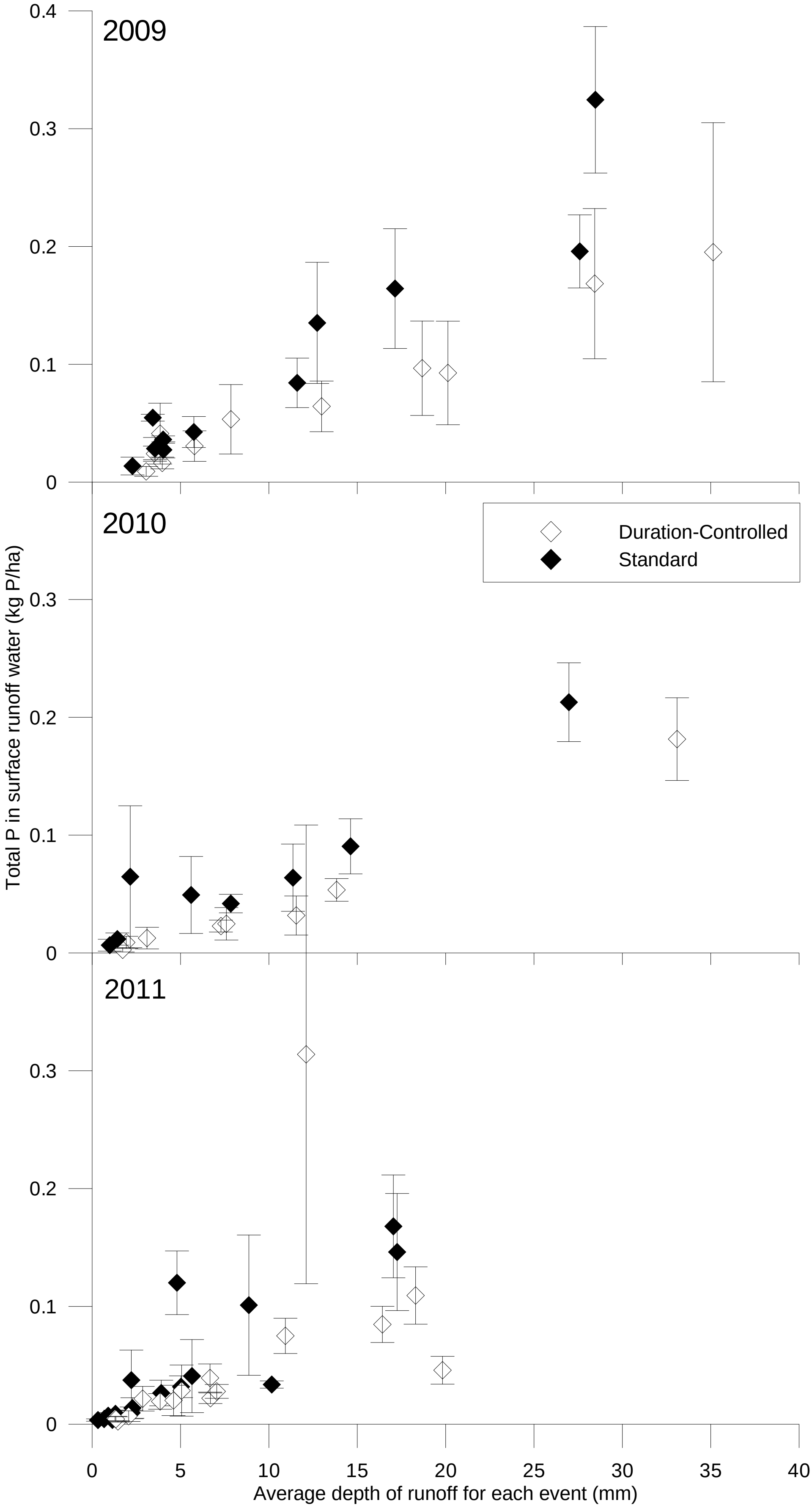

Figure 5.7: Total phosphorus load (kg/ha) in runoff compared with the depth of runoff for each event for *SG* and *DC* treatments, from 2009-2011.

5.3.6 Phosphorus losses in drainage water

Annual DRP and TP loads in drainage water were small compared to P loads in runoff in most years (Tables 5.3 and 5.4), and values from each treatment were not significantly different. The annual DRP loads from *SG* plots were 0.07, 0.04 and 0.06 kg/ha in 2009, 2010 and 2011, respectively (Table 5.4). These are very low losses, so there is limited potential to achieve further reductions from the use of *DC* grazing. The TP loads from *SG* plots were 0.89, 0.28 and 0.21 kg/ha in 2009, 2010 and 2011, respectively, which were similar to the average loads from the *DC* plots (Table 5.4 and Figure 5.8).

Table 5.4: Amounts (kg P/ha) of dissolved reactive phosphorus and total phosphorus lost in drainage, and average drainage depth, from *DC* and *SG* treatments for 2009-2011 (values in parentheses show standard error of the mean for each cumulative load).

	DC			***SG***		
Year	**Drainage** *(mm)*	**DRP** *(kg P/ha)*	**TP** *(kg P/ha)*	**Drainage** *(mm)*	**DRP** *(kg P/ha)*	**TP** *(kg P/ha)*
2009	400	0.07 (0.03)	0.87 (0.17)	347	0.07 (0.01)	0.89 (0.14)
2010	327	0.11 (0.07)	0.39 (0.13)	305	0.04 (0.01)	0.28 (0.05)
2011	339	0.07 (0.03)	0.18 (0.05)	320	0.06 (0.01)	0.21 (0.02)
Average	355	0.08	0.48	324	0.06	0.46

The cumulative P loss (kg P/ha) in drainage showed a different pattern to that of N, where >95% of the N leaching loss occurred in the first 200 mm of drainage (Chapter 4). For DRP, 95% of the load was not lost until more than 379, 300 and 325 mm of drainage (equivalent to 95%, 92% and 96% of the year's drainage in 2009, 2010 and 2011, respectively) from *DC* plots, and more than 248, 251 and 306 mm (equivalent to 71%, 83% and 96% of the year's drainage in 2009, 2010 and 2011, respectively) from *SG* plots. In general, DRP loss was very low and did not differ between years or between grazing treatments. Total P however, increased rapidly in the later part of the drainage season (>200 mm drainage), particularly in 2009 and 2010 (Figure 5.8).

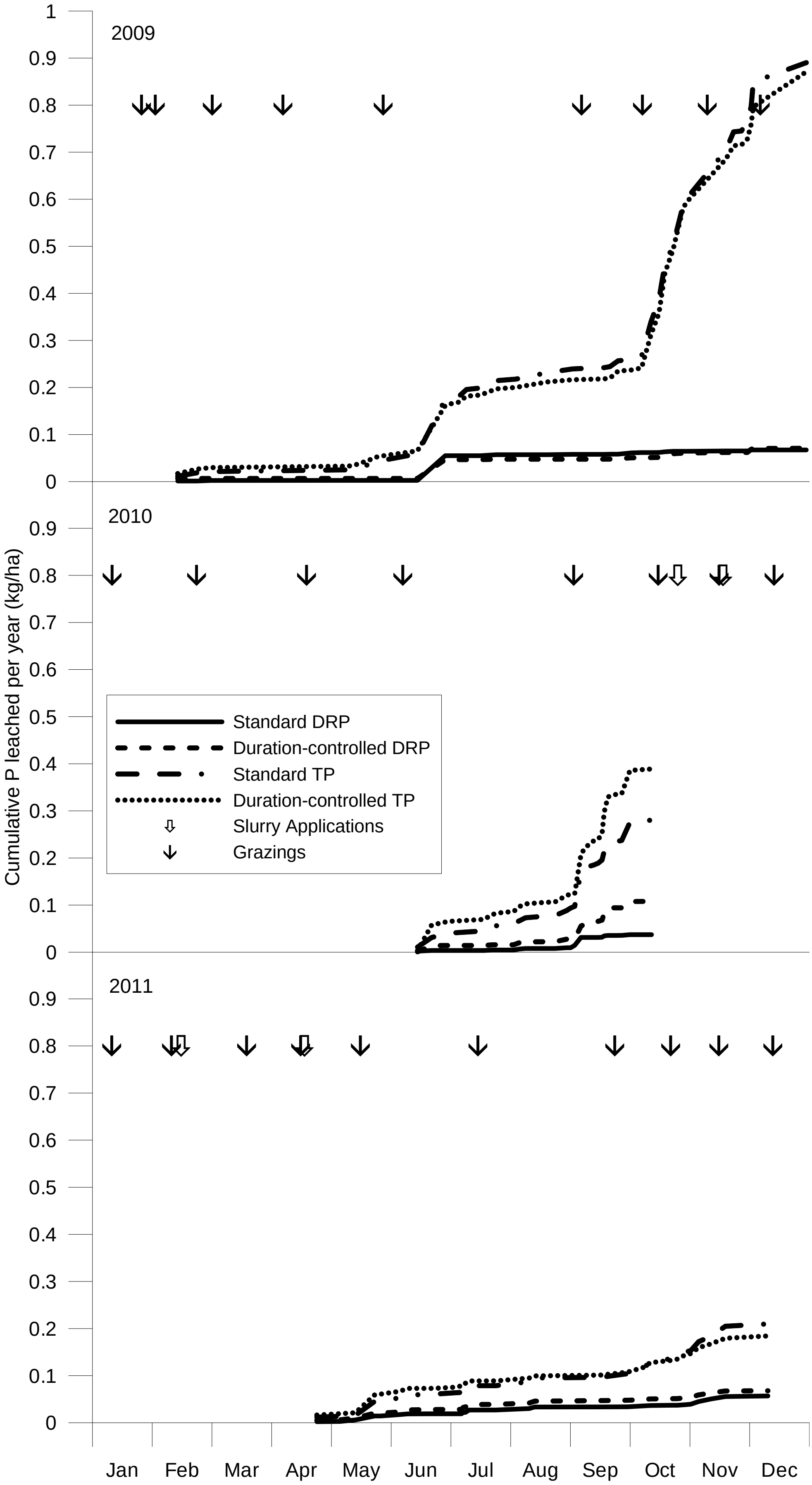

Figure 5.8: Cumulative dissolved reactive phosphorus and total phosphorus load (kg P/ha) in drainage for *DC* and *SG* grazing treatments, from 2009-2011.

5.4 *Discussion*

5.4.1 Phosphorus losses in runoff

The grazing treatments applied in this research were *DC* and *SG*. Duration-controlled grazing included the periodic return of slurry from the standoff facility, which is otherwise deposited *in situ* as dung and urine on the *SG* treatment. Average Olsen P status of the plots ranged from 31-37 mg P/kg soil for *DC* plots and 30-39 mg P/kg soil for *SG* plots over the three years of the trial (Table 5.1). Both sets of treatment plots were grazed on the same day during each grazing event, to ensure that the main treatment difference would be grazing duration and its impact on excreta deposition, rather than the different timings of grazings in relation to soil and climatic conditions. Both treatments were grazed on days when the soil moisture deficit was > 5 mm (Horne, 2012), to minimise treading damage. The *SG* treatment perhaps deviated from a true ‘standard’ dairy system on a commercial dairy farm, in that winter and spring grazings on a commercial farm typically cause treading damage. In this trial the *SG* treatment was run as a ‘best management’ system to avoid treading damage (as described in Section 5.3.1, Trial site). Thus grazing duration and slurry application were the only treatment difference, as surface roughness measurements indicated there was no difference in treading between the two treatments.

Minimising treading damage from grazing resulted in small P losses from both treatments. It has been shown that soil disturbance from treading can contribute to significant increases of P in runoff (Sharpley, 1977) and so differences in P loss between treatments could be expected to be smaller if minimal treading damage occurred.

The soil was at field capacity at this site for a similar number of days in each year (63, 51 and 56 days for 2009, 2010 and 2011, respectively). The occurrence of runoff and variation in runoff depth for each year was dependent on rainfall depth and intensity as indicated by McDowell *et al.* (2008). When large rainfall events occurred at high intensities, then the soil infiltration rate was exceeded and large runoff depths occurred. There was also large variation in runoff depths between plots at each runoff event. This natural variation in runoff volume was often greater than the treatment effect, causing differences in runoff P load to be non-significant between treatments. Some of this variation (Figures 5.1 and 5.2) is also likely to be due to plot to plot differences in micro-topography that influenced ponding and infiltration rates.

The intensity of rainfall causing greater runoff flow did not affect the concentrations of DRP in runoff. This was consistent with the work of Sharpley (1977), where the concentration of dissolved inorganic P (DIP) did not change significantly with increased flow. Even in a high intensity rainfall event from undrained grazed plots of a similar size (55 m^2), where flow rate was up to 120 ml/sec, DIP concentrations remained consistently below 0.2 ppm, while TP concentrations increased with flow rate, ranging from ca. 0.6 ppm and peaking at ca. 2 ppm.

The overall DRP concentrations from both *SG* and *DC* treatments were within the range reported by Sharpley (1977) and Houlbrooke *et al.* (2008) from similar sized plots at the same site. The average concentration over the three years from the *DC* treatment (0.22 ppm) was almost half that of the *SG* treatment (0.41 ppm). The lower concentrations from *DC* plots were likely to be due to less dung deposition, caused by less time spent grazing on the *DC* treatment. Because dung is high in P concentration and it can easily be transported in runoff, P concentrations in runoff are generally higher when there is a dung source contributing to it (McDowell and Sharpley, 2002), particularly soon after grazing. Dissolved reactive phosphorus constituted 38-55% of the TP load over the three years, which was similar to the 46% of TP loss in runoff from pasture reported by McDowell *et al.* (2008).

The first runoff event in 2011 gave the highest concentrations of DRP and TP in runoff from both treatments, and it was particularly high from the *DC* treatment (DRP in Figure 5.5). This runoff event occurred within one month of P fertiliser application (15 kg P/ha), but moreover, within one week of a night grazing (294 cows/ha) and slurry application (5 kg P/ha). It was evident that the grazing on both treatments and slurry application to *DC* plots had increased the potential source of P effect, which could be transported easily in the runoff event (McDowell and Sharpley, 2002; McDowell *et al.*, 2006). These results suggest that the timing of slurry to *DC* plots should be timed or managed to avoid the risk of runoff. It has recently been reported from a small scale trial using soil monoliths, that the highest risk of nutrient loss to water is when manures are applied to pasture within 2 days of rainfall, with the risk decreasing if rainfall occurs 10 or more days after manure application (Houlbrooke *et al.*, 2013).

The effect of Olsen P and P retention on DRP concentration in runoff was studied by McDowell and Condron (2004) using a range of grassland soils. The equation that they generated to predict DRP concentration in runoff was:

DRP in overland flow = 0.024 x (Olsen P/P retention) + 0.024

and was applied to data from the present study. The DRP concentrations in this study (weighted average 0.41 ppm from *SG* plots) were under-estimated by this relationship (DRP concentration <0.06 ppm), using the range of Olsen P measurements from Table 5.1, and a P retention of 24% (Schofield *et al.*, 1981). One reason for this is that the equation by McDowell and Condron (2004) was derived from rainfall simulation experiments on turf slabs, and does not include the effect of fresh dung and treading of soil due to recent grazing events (McDowell *et al.*, 2001). Recent grazing and dung deposition would be expected to increase the P concentrations in runoff (McDowell *et al.*, 2006). Also, different sized runoff plots affect concentration (McDowell and Sharpley, 2002). In the present study, relatively large, 10 m long runoff plots were installed, whereas the equation of McDowell and Condron (2004) was based on 1 m long turf slabs.

The amount of P lost (load), both as DRP and TP, was mostly related to the depth of runoff at each event. As runoff depth increased, so did the P load (kg P/ha). This was similar to results reported by Sharpley (1977) and Sharpley *et al.* (2008) from studies in New Zealand and the USA. In their studies, these authors compared P losses in storm events and base flow events, with clear increases in P losses, particularly as TP and particulate P, when storm flow causing runoff increased. For example, in a 10 year study based in the USA, storm flow mostly contributed about a third of the stream flow (stream flow being storm flow plus subsurface flow (Sharpley, 1977), or runoff plus drainage), but 65% of the dissolved P was transported through storm flow (Sharpley *et al.*, 2008). This was comparable to the current study in which runoff was only 23% of the average volume of runoff and drainage combined, but contributed on average 61% of TP loss from both treatments over all 3 years.

The overall TP losses in runoff from *DC* plots were 32% lower than the *SG* system average annual loss of 0.80 kg P/ha, but this difference was not significant (P>0.10). In this study, experimental variability was large because of the relatively small size of the runoff plots (McDowell and Sharpley, 2002), damage to containment boards, and possibly variations in micro topography, all leading to differences within runoff plots. This means that differences between treatments of much larger than 32% would need to be achieved to be statistically significant. Furthermore, there were relatively low losses of both DRP and TP (<1 kg/ha) from the *SG* treatment, because of the ‘best-management’ grazing system implemented, and thus, minimal treading damage would have minimised the impact of grazings on P losses. Therefore, the *DC* treatment difference of lower grazing duration, and hence less dung deposition, did not provide a statistically significant reduction. This provides some understanding about the impacts that other factors such as treading have on P losses, rather than decreased grazing times. Consequently, further work should be conducted to compare an *SG* system, which has less flexibility in choosing when to graze, to better account for differences in treading damage and the impact on P losses in runoff. This would provide a more comprehensive understanding on how to utilise a *DC* grazing system on-farm to reduce P losses.

5.4.2 Phosphorus losses in drainage water

Although not high (<1 kg/ha/yr), the losses of DRP and TP in drainage from the *SG* treatment plots, ranging from 0.18 – 0.87 kg/ha/yr and 0.21 – 0.89 kg/ha/yr respectively, were similar to those reported by Houlbrooke *et al.* (2008). In their study, on plots that did not have effluent applied (0.15 kg/ha/yr and 0.73 kg/ha/yr for DIP and TP in drainage respectively), Houlbrooke *et al.* (2008) had greater loads of TP being lost through drainage than via runoff (0.60 kg/ha/yr as runoff), although concentrations were higher in the runoff water (1.44 ppm compared with 0.31 ppm in drainage). This was consistent with the present study, with higher volumes of water through drainage (Chapter 4) sometimes leading to higher overall DRP and TP loads, than through runoff. Earlier reports also noted that P losses in drainage could be significant (McDowell *et al.*, 2001), and sometimes higher than in runoff (Ryden *et al.*, 1973). The division between drainage and runoff volumes can be expected to vary dependent on rainfall frequency and intensity.

It is unknown why DRP and TP losses in drainage water were sometimes higher, albeit not significantly, from the *DC* treatment than from the *SG* treatment in 2010. Concentrations of P were similar for both treatments, and therefore slightly higher (but statistically insignificant) drainage volumes from the *DC* plots (Chapter 4) contributed to higher overall loads in this year.

The higher losses of TP compared with DRP (Figure 5.8) indicates that the majority of the P lost in drainage was as organic P or particulate P. The major TP loss occurred late in the drainage season and the increase in loss coincided with the onset of grazing in early spring (August/September). A review by McDowell *et al.* (2001) reported that TP (predominantly made up of organic P) was more likely to be lost in drainage, while DRP was likely to undergo sorption in the surface soil and therefore concentrations of this were much lower in drainage water. This partly explains why TP losses continued to increase in drainage (Figure 5.8), while DRP losses remained low throughout the drainage period. Over the same period, the losses of DRP in runoff continued to increase, presumably due to grazings disturbing the P-enriched layer of the surface soil (McDowell *et al.*, 2001). Increasing TP losses in drainage over time were also similar to that observed by Sharpley (1977), where particulate P had a large bearing on TP lost in drainage, compared with relatively low dissolved P losses. This was most evident within a few days after grazing by cattle, which, as discussed, was also observed in the current study.

Also, the onset of grazings in September of each year coincided with warmer soil temperatures and high soil moisture (Figure 5.1) and thus increased earthworm activity (Sharpley, 1977). The amount of earthworm activity has been related to soil being deposited as castings on the surface. In his early work at a similar site, Sharpley (1977) found that organic P levels in earthworm castings increased over winter to a maximum in August, whereas inorganic P decreased over the same time period. Therefore, this may have influenced the amount of TP in drainage water in the present study. The increased deposition of fresh dung, as well as soil disturbance from grazing, was also probably contributing to the loss of organic P through the drainage system (McDowell *et al.*, 2001). However, this effect cannot be quantified in the current study and future research should be conducted to quantify the movement of P from fresh dung pats through soil to pipe drainage systems. This would help to quantify the losses of TP from dung through to drainage, and P loss from dung could be modelled based on dung distribution and patch intensity (Chapter 3), in a similar way to N losses from urine patches (Chapter 4).

5.5 Conclusion

The TP loads lost in runoff and drainage from the *DC* grazing plots were low over the three years of the study, averaging 0.55 kg P/ha and 0.48 kg P/ha, respectively. These runoff P losses were 32% lower than those from the *SG* treatment, but were highly variable and not significantly different. The average drainage water P losses were similar between the treatments. Both *DC* and *SG* treatments were grazed on “safe grazing” days to minimise treading damage, which may explain why large differences in P losses were not measured between treatments. Overall, this study demonstrates that grazing mole and tile drained soils (<3° slope) using *DC* grazing achieves relatively low TP losses in runoff and drainage of <1 kg P/ha if grazing occurs when the soil moisture deficit is > 5 mm. Also, in these conditions, increasing grazing duration results in minimal increases in P losses.

Surface runoff losses of P were initially of greatest interest because earlier research had shown P loss in runoff to be greater than in drainage. In this study however, P loss in pipe drainage was of equal importance, with similar annual loads of P being lost in runoff and drainage. This study confirmed that depth of runoff, caused by rainfall amount and intensity, had an increasing effect on the P load lost in runoff. Lower concentrations, and therefore loads, from *DC* treatment plots were caused by less deposition of dung from less time spent grazing these plots. The even return of effluent via slurry did not appear to have adverse effects on promoting P losses through runoff. Slurry also did not appear to increase drainage P loads, although the onset of grazings increased TP losses but not DRP losses via drainage. Further work is required to determine the impact of an *SG* grazed system, if grazing when the soil water deficit is > 5 mm is not an option.

There were some limitations around using small runoff plots for measuring runoff volumes and P losses, which resulted in the differences between treatments being non-significant. Firstly, the micro topography within plots affected the division between drainage and runoff between replicates, and secondly, the proximity of dung patches to the runoff catchers was not necessarily treatment related. One way to overcome these limitations would be to increase the height of the boards that border the plots, to completely isolate any water travelling from the greater grazing plots. Also, the size of the plots could be increased to be the same size as each drainage plot. This would add greater expense to research trials, but would reduce the risk of damage by grazing cows as it would be around the perimeter of each grazing area.

5.6 References

Ahuja, L.R., Sharpley, A.N., Lehman, O.R., 1982. Effect of soil slope and rainfall characteristics on phosphorus in runoff. Journal of Environmental Quality 11, 9-13.

Bowler, D.G., 1980. The drainage of wet soils. Massey University, Palmerston North.

Curran-Cournane, F., McDowell, R.W., Littlejohn, R., Condron, L.M., 2010. Effects of stock type on soil physical properties and losses of phosphorus and suspended sediment in surface runoff. Proceedings of the 19th World Congress of Soil Science: Soil solutions for a changing world, Brisbane, Australia, 1-6 August 2010. Symposium 3.1.2 Farm system and environment impacts, 92-95.

Haygarth, P.M., Sharpley, A.N., 2000. Terminology for phosphorus transfer. Journal of Environmental Quality 29, 10-15.

Haynes, R.J., Williams, P.H., 1993. Nutrient cycling and soil fertility in the grazed pasture ecosystem. Advances in Agronomy 49, 119-199.

Horne, D.J., 2012. Personal Communication.

Hosomi, M., Sudo, R., 1986. Simultaneous determination of total nitrogen and total phosphorus in freshwater samples using persulfate digestion. International Journal of Environmental Studies 27, 267-275.

Houlbrooke, D.J., Horne, D.J., Hedley, M.J., Hanly, J.A., Scotter, D.R., Snow, V.O., 2004. Minimising surface water pollution resulting from farm-dairy effluent application to mole-pipe drained soils. I. An evaluation of the deferred irrigation system for sustainable land treatment in the Manawatu. New Zealand Journal of Agricultural Research 47, 405-415.

Houlbrooke, D.J., Horne, D.J., Hedley, M.J., Snow, V.O., Hanly, J.A., 2008. Land application of farm dairy effluent to a mole and pipe drained soil: implications for nutrient enrichment of winter-spring drainage. Australian Journal of Soil Research 46, 45-52.

Houlbrooke, D.J., Laurenson, S., Wilson, T., 2013. Assessing the risk of nutrient and microbial runoff from rainfall following surface application of slurries and manures with varying dry matter content. In: Currie, L.D., Christensen, C.L. (Eds.), Accurate and efficient use of nutrients on farms, Occasional Report No. 26. Fertilizer & Lime Research Centre, Palmerston North.

Ledgard, S.F., Menneer, J.C., 2005. Nitrate leaching in grazing systems and management strategies to reduce losses. In: Currie, L.D., Hanly, J.A. (Eds.), Developments in fertiliser application technologies and nutrient management. Fertiliser and Lime Research Centre, Massey University, Palmerston North, pp. 79-92.

McDowell, R.W., Condron, L.M., 2004. Estimating phosphorus loss from New Zealand grassland soils. New Zealand Journal of Agricultural Research 47, 137-145.

McDowell, R.W., Houlbrooke, D.J., Muirhead, R.W., Muller, K., Shepherd, M., Cuttle, S.P., 2008. Grazed pastures and surface water quality. Nova Science Publishers.

McDowell, R.W., Monaghan, R.M., Morton, J., 2003. Soil phosphorus concentrations to minimise potential P loss to surface waters in Southland. New Zealand Journal of Agricultural Research 46, 239-253.

McDowell, R.W., Muirhead, R.W., Monaghan, R.M., 2006. Nutrient, Sediment, and Bacterial Losses in Overland Flow from Pasture and Cropping Soils Following Cattle Dung Deposition. Communications in Soil Science and Plant Analysis 37, 93-108.

McDowell, R.W., Sharpley, A.N., 2002. Effect of plot scale and an upslope phosphorus source on phosphorus loss in overland flow. Soil Use and Management 18, 112-119.

McDowell, R.W., Sharpley, A.N., Condron, L.M., Haygarth, P.M., Brookes, P.C., 2001. Processes controlling soils phosphorus release to runoff and implications for agricultural management. Nutrient Cycling in Agroecosystems 59, 269-284.

McDowell, R.W., Wilcock, R.J., 2008. Water quality and the effects of different pastoral animals. New Zealand Veterinary Journal 56, 289-296.

Monaghan, R.M., Smith, L.C., 2012. Contaminant losses in overland flow from dairy farm laneways in southern New Zealand. Agriculture, Ecosystems and Environment 159, 170-175.

Murphy, J., Riley, J.P., 1962. A modified single solution method for the determination of phosphate in natural waters. Analytica Chimica Acta 27, 31-36.

Pollok, J.A., 1975. A comparative study of certain New Zealand and German soils formed from loess., Unpublished PhD Thesis. University of Bonn, Germany.

Rennes, A., Tillman, R.W., Syers, J.K., Bowler, D.G., 1977. Effect of mole drainage on surface runoff from a soil under pasture. New Zealand Journal of Agricultural Research 20, 45-49.

Ryden, J.C., Syers, J.K., Harris, R.F., 1973. Phosphorus in runoff and streams. Advances in Agronomy 25, 1-45.

Salazar, M.E., Hedley, M.J., Horne, D.J., 2010. Using turnips to reduce soil K loading on the effluent block. Proceedings of the New Zealand Grassland Association 72, 247-250.

Saleh, A., 1993. Soil roughness measurement – chain method. Journal of Soil Water Conservation 48, 527-529.

Schofield, P.E., Gregg, P.E.H., Syers, J.K., 1981. 'Biosuper' as a phosphate fertiliser: a glasshouse evaluation. New Zealand Journal of Experimental Agriculture 9, 63-67.

Scotter, D.R., Clothier, B.E., Turner, M.A., 1979. The soil water balance in a Fragiaqualf and its effect on pasture growth in central New Zealand. Australian Journal of Soil Research 17, 455-465.

Sharpley, A.N., 1977. Sources and transport of phosphorus and nitrogen in a stream draining a dominantly pasture catchment. PhD Thesis in Soil Science. Massey University, Palmerston North.

Sharpley, A.N., Kleinman, P.J.A., Heathwaite, A.L., Gburek, W.J., Folmar, G.J., Schmidt, J.P., 2008. Phosphorus Loss from an Agricultural Watershed as a Function of Storm Size. Journal of Environmental Quality 37, 362-368.

Sharpley, A.N., Syers, J.K., 1979. Loss of nitrogen and phosphorus in tile drainage as influenced by urea application and grazing animals. New Zealand Journal of Agricultural Research 22, 127-131.

Chapter 6 The impact of Duration-controlled grazing on faecal microbe concentrations in surface runoff and drainage water

6.1 Introduction

In addition to N and P losses (Chapters 4 and 5), contamination of surface waters with zoonotic microbial pathogens such as campylobacter, cryptosporidia, salmonella, and giardia (Connolly *et al.*, 2004; Collins *et al.*, 2005; Collins *et al.*, 2007) derived from grazing animal faeces (Wilcock *et al.*, 1999), is of ongoing concern for regional authorities in New Zealand (Parliamentary Commissioner for the Environment, 2012). Pathogens from animal faeces can have detrimental effects on human health (ASM, 1999; Collins *et al.*, 2007). The loss of faecal microbes from diffuse sources such as grazed pastures to surface water is now an important research issue (McDowell *et al.*, 2008) and the impacts of faecal loss from rural areas on recreational water quality and human health have been reported from New Zealand coastal areas (McBride *et al.*, 1998). Cattle produce a waste output equivalent to 7 people (McBride *et al.*, 1998) and therefore, it is important for the New Zealand dairy industry to reduce waterway contamination from dairy cow faeces.

The measurement of pathogenic coliforms is generally performed using faecal indicator organisms (FIO's) (McDowell *et al.*, 2008). A common FIO used to predict or estimate faecal coliform contamination is *Escherichia coli (E. coli)*, (Collins *et al.*, 2007; Sinton *et al.*, 2007; McDowell and Wilcock, 2008), particularly for freshwater in New Zealand (Parliamentary Commissioner for the Environment, 2012). This organism is relatively easy to measure, and if it is present in water, then it is likely that other faecal-sourced pathogens are present (McDowell *et al.*, 2008). The limit of *E. coli* where action is taken for recreational use in fresh waterways is 550 most probable number (MPN)/100 mL water (Ministry for the Environment, 2013). The number of *E. coli* in a single fresh cow dung pat is at least 10 billion (Collins *et al.*, 2005).

Much of the *E. coli* loss to waterways is via surface runoff (Muirhead and Monaghan, 2012), which is also a major transportation mechanism for P loss (Chapter 5). The source of the *E. coli* is predominately fresh dung, with the soil also acting as a reservoir for *E. coli* populations (Muirhead and Monaghan, 2012). Research conducted over the past 3 decades on the transportation of *E. coli* from pastures to waterways has mainly focussed on the rate of decomposition or erosion of dung pats, and the rate at which protective skins form on the dung pats (Crane and Moore, 1986; Muirhead *et al.*, 2005; Muirhead and Littlejohn, 2009). It has also focussed on the application of faecal organisms to pasture via effluent or slurry (Crane and Moore, 1986; Vinten *et al.*, 2004; Collins *et al.*, 2005). Previous research has also investigated the impact of the period between grazing and the runoff event, with a general trend of lowering numbers of *E. coli* with increasing numbers of days since grazing (Connolly *et al.*, 2004; McDowell *et al.*, 2008).

The transmission pathways for faecal microbes from land to surface water are similar to those of P, which have been discussed in Chapter 5. Therefore, the mitigation strategies for reducing the contamination of waterways are similar for both faecal microbes and P (Collins *et al.*, 2005). These strategies include creating riparian buffer strips or paddocks to capture microbes before they enter waterways (Collins *et al.*, 2007), fencing off streams and shallow wetlands from direct animal access, practicing 'deferred irrigation' when applying farm dairy effluent or slurry from standoff facilities, and reducing grazing time and therefore dung deposition to pasture when the likelihood of surface runoff occurring is high (Chapter 5). Therefore, grazing pastures for shorter durations and using a standoff facility can reduce the number of dung pats and faecal microbe load to pastures (White *et al.*, 2001), thus potentially reducing losses via surface runoff. However, the return of stored slurry (Chapters 3-5) containing faecal organisms, may add to the *E. coli* loading, which could lead to smaller reductions in *E. coli* losses from *DC* grazing.

There have been limited studies from large scale, pastoral grazing areas on the losses of *E. coli* in surface runoff (McDowell *et al.*, 2008). Many studies have been conducted in simulated situations using, for example, artificially-applied cow dung on small areas (Sinton *et al.*, 2007), in laboratory conditions (McLeod *et al.*, 2003), or on losses from manure applied to land (Vinten *et al.*, 2004). The objective of this chapter is to investigate and quantify, using a large scale grazing study, the effect of changing the duration of grazing (*DC* grazing), and therefore the quantity of dung deposited during grazing, on *E. coli* losses in runoff and drainage water.

6.1 Methods

6.1.1 Trial site and trial management

The description of the trial site and management has been detailed in Chapters 3-5, as have dung deposition measurements. As described in Chapter 5, grazings were conducted on days that minimised treading damage on both treatments, and therefore, grazing duration was the only difference being measured.

During the 2009/10 and 2010/11 lactation seasons after most grazings, each runoff plot was photographed immediately following grazing to provide evidence of the spatial distribution of dung pats prior to any runoff events (Plate 6.1). These photographs were used to measure the distance of the edge of the closest dung pat to the catcher measured, as well as the number of dung pats within 1 metre and within 3 metres of the catcher.

Plate 6.1: Example of a runoff plot photographed to observe spatial distribution of dung pats; runoff catching gutter is bottom centre, isolated by white electric fence standards.

6.1.2 Runoff and drainage water volume measurements and water analyses

Runoff and drainage water collection has been detailed in Chapters 3-5. The depth of runoff in each of the 3 years studied has been discussed in Chapter 5, and Figure 5.3 has been repeated in this chapter as Figure 6.1 to allow rapid reference to climatic conditions and the occurrence of runoff.

Randomly selected drainage and almost all runoff samples were collected after each storm event and sent for *E. coli* enumeration within 24 hours of the event (Guy and Small, 1976). These runoff and drainage water samples were analysed unfiltered (approximately 35 mL) for the MPN of *E. coli* per 100 mL (APHA 21st Ed. 9223 B, as per Central Environmental Laboratories, Palmerston North, New Zealand). Data have been presented in MPN/100 mL. The average *E. coli* load lost via runoff in each storm event was also calculated to give context around the dilutions needed for reaching recreational water limits (Ministry for the Environment, 2013).

6.1.3 Statistical analysis

Treatment differences in drainage water were determined using ANOVA in Microsoft Excel. Stepwise regression was performed using Minitab (version 16) to consider the most important factors in determining *E. coli* in surface runoff, and to further analyse treatment differences.

6.2 Results and Discussion

6.2.1 *E. coli* in runoff water

Due to issues associated with external flow entering some plots in runoff events, data from 3 of the 7 replicate runoff plots within each treatment were not included for analysis. Therefore, the statistical analysis has been completed on 4 of the 7 replicate plots within each treatment, as described in detail in Section 5.3.1 (Chapter 5).

Flow rate has been shown to have an effect on the losses of faecal microbes in runoff (Collins *et al.*, 2003; McDowell *et al.*, 2008). This effect was not assessed in this study, due to the proportional sampling methods used, with average concentrations of *E. coli* over the entire runoff event being measured, rather than those in runoff at peak flows. There was no relationship between the highest rainfall intensity during an event (Chapter 5) and *E. coli* concentrations in the proportionally sampled runoff. Therefore, the average *E. coli* concentrations and the total runoff volume for each event were more appropriate metrics for testing effects of treatment or time after grazing.

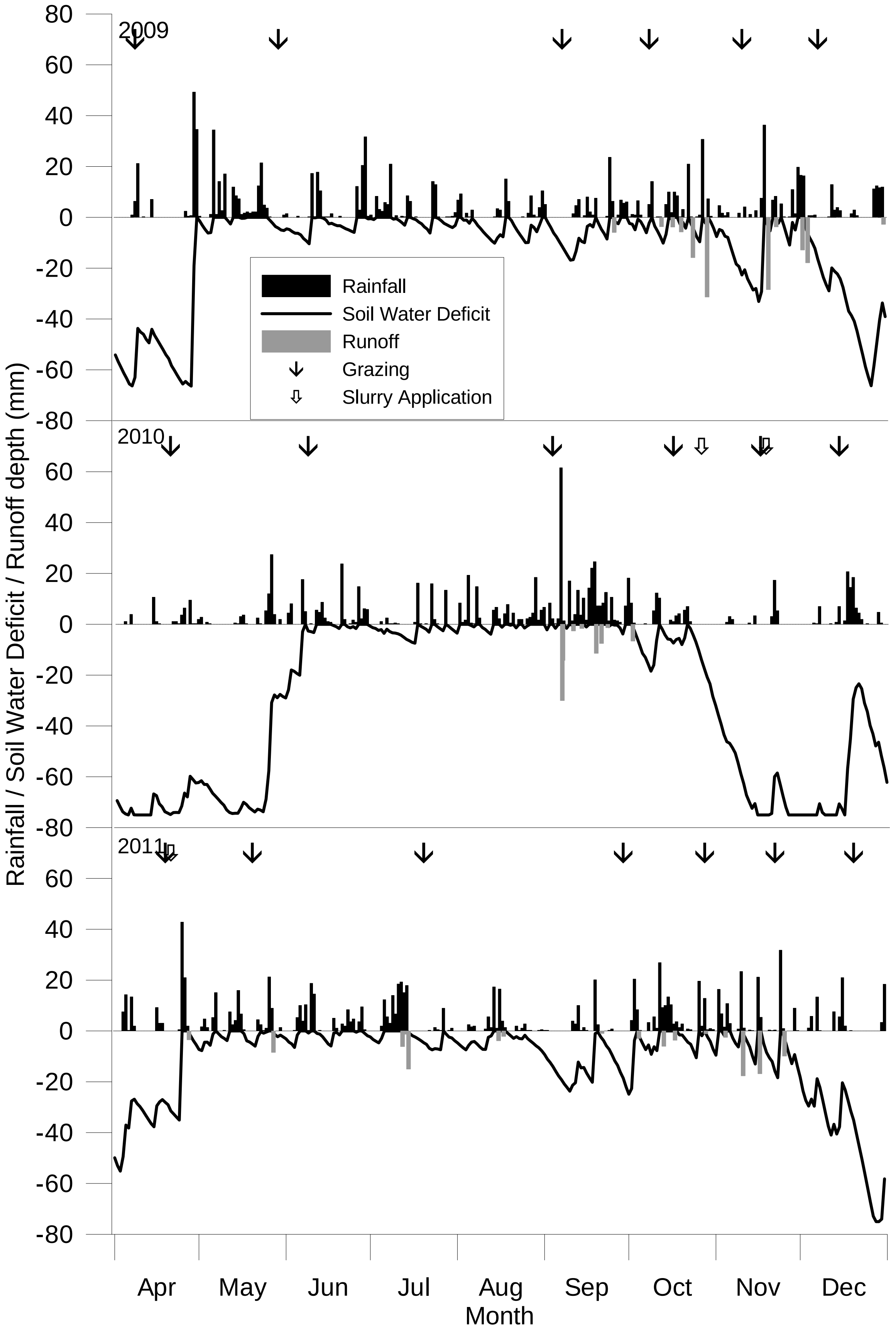

Figure 6.1: Measured rainfall, modelled soil water deficit, and average surface runoff depth (mm) at each event from all grazed plots on *DC* and *SG* treatments, from April-December, 2009-2011. Soil water deficit and drainage depth are absolute values, but have been shown as negative values to add clarity to the figure.

Runoff water from *DC* plots had lower counts of *E. coli* than *SG* plots in some events (Figures 6.2 and 6.3), however they were not significantly different (P>0.05) for all events in any year. There was no pattern in the time of year when significant differences occurred. The maximum average *E. coli* count from *SG* grazed plots was 1 077 500, 852,500 and 1 030 000 MPN/100 mL in 2009, 2010 and 2011, respectively. In comparison, the maximum average from *DC* plots was 360 000, 716 750 and 1 047 500 MPN/100 mL in each of the 3 years, respectively. The annual average concentration in runoff from *SG* plots was 361 770, 203 089 and 261 704 MPN/100 mL in 2009, 2010 and 2011, respectively, and the annual average concentration in runoff from *DC* plots was 138,212, 258 908 and 261 704 MPN/100 mL in each of the 3 years, respectively.

Based on the average concentrations and runoff depths from the *SG* plots, a dilution factor of between 369 and 658 would be required to ensure *E. coli* levels in the receiving waters were below the recreational standard of 550 MPN/100 mL. Therefore, to achieve sufficient dilution, ca. 280 440 – 868 560 m^3 of water, containing no *E. coli*, would be required to achieve the recreational standard. This, for simplicity, is assuming no attrition of the microbes between leaving the grazed area and entering the waterways, which, of course, is likely to be occurring. However, there was a large population of *E. coli* being lost as runoff, and this highlights the importance of reducing the source of faecal coliforms before runoff occurs.

Because of the lack of significance created by variability in *E. coli* concentrations within treatments, it was important to investigate other factors that were contributing to the variation in *E. coli* losses other than treatment. These factors included number of days since the last grazing, climate after grazing (and therefore the rate of the dung pats developing protective skins), rainfall intensity, and runoff volume.

In 2009, the variation in *E. coli* populations in runoff water was not related to the number of days since the last grazing (Figure 6.2). In 2010 there was a strong downward exponential trend, with lower *E. coli* counts occurring at longer time intervals since grazing (Figure 6.2). There was only one grazing in the season that had occurred prior to any runoff occurring. The 2011 runoff period also showed a strong downward trend in concentrations with increasing days since grazing. There were more grazings prior to runoff events in this year than in 2010.

The formation of protective skins over dung pats has been reported to be a factor in reducing the rate of disruption and therefore decreasing the die-off rate of faecal microbes within the pats (Haynes and Williams, 1993; Connolly *et al.*, 2004). Protective skins are formed more quickly if the climate is conducive to this process, for instance sunny, dry environments. For the majority of the runoff events in 2009, there had been a period without significant rain and consequently there was soil drying, prior to runoff but after grazing (Figure 6.1). The formation of a protective surface skin on dung was thus likely in 2009 and this may have led to microbes being less susceptible to exposure to ultraviolet radiation and hence less *E. coli* die-off prior to runoff (Crane and Moore, 1986). In addition, growth of *E. coli* was likely to occur in the protected dung pats (Muirhead, 2009), particularly if they contained above 80% water (Sinton *et al.*, 2007), which may have contributed to the steady, high concentrations of *E. coli* in runoff water in that year if the dung pats were disturbed sufficiently to be lost in runoff.

In comparison in 2010, soil moisture conditions were wet throughout this whole period, with some drying only starting to occur by day 29 after grazing. Therefore, there were less favourable conditions for the quick formation of protective skins. Also, heavy rainfall events were likely to cause greater decomposition of dung pats (Muirhead, 2009) hence the number of *E. coli* susceptible to be lost would have decreased over time (Figure 6.2). A similar trend was observed in 2011 (Figure 6.2), with lower soil moisture deficits prior to runoff (Figure 6.1).

The number of days since grazing was the most important factor in determining *E. coli* concentrations in runoff, using stepwise regression analysis (Eq. 6.1, $P<0.001$). This was also the case with campylobacter and *E. coli* losses in runoff studies conducted by Connolly *et al.* (2004), with decreasing faecal microbe counts with increasing days since grazing. Using stepwise regression, once days since grazing had been accounted for, the other significant factors in the present study, in order of importance, were number of cows grazed per hectare, treatment, and total runoff depth per event:

$$E.\ coli \text{ concentration} = -0.0952 * dsg + 0.594 * runoff + 1.09 * trt + 0.0067 * cows \qquad \textbf{Eq. 6.1}$$

where *dsg* = days since grazing; *runoff* = depth of runoff (mm); *trt* = treatment; and *cows* = cows grazed/ha at last grazing.

Together, these factors explained 60% of the variation ($R^2 = 0.60$). Although this R^2 value is not high, the parameters explaining variation were similar for *E. coli* as for P loss

(Chapter 5). However, there are other parameters that had causal effects on *E. coli* loss. For example, although the number of days after grazing was important, particularly in 2010 and 2011 (Figure 6.2), climate after grazing was a critical factor, as has been discussed. Also, although there was not a significant treatment effect at all times in all years, it was evident that the number of dung pats deposited (caused by treatment and cows grazed per hectare) contributed to the *E. coli* populations likely to be lost in runoff, with more dung pats providing higher overall populations (White *et al.*, 2001).

Spatial distribution of the dung pats in relation to the runoff catchers was also likely to be a contributing factor for the variation in trends. Earlier observations at the same site (unpublished data) indicated that a dung pat very close (less than 1m) to the runoff catcher contributed to extraordinarily high concentrations of faecal microbes in runoff water for the respective plot, which increased the average count over all plots for that runoff event. For the current study, the data using photographs to show proximity of dung pats to the catcher were analysed. Distance of dung pats from the catcher explained little of the residual variance after days since grazing was considered when using stepwise regression analysis. The distance of dung pats from catchers, however, may not have been independent of cow grazing numbers and durations, which were a direct effect of treatment.

The most significant factor in determining P load in runoff was runoff depth (Chapter 5). In the case of P loss in runoff, the soil P status was considered to be an important source of P, whereas fresh dung is essentially the sole source of elevated *E. coli* counts in surface runoff water.

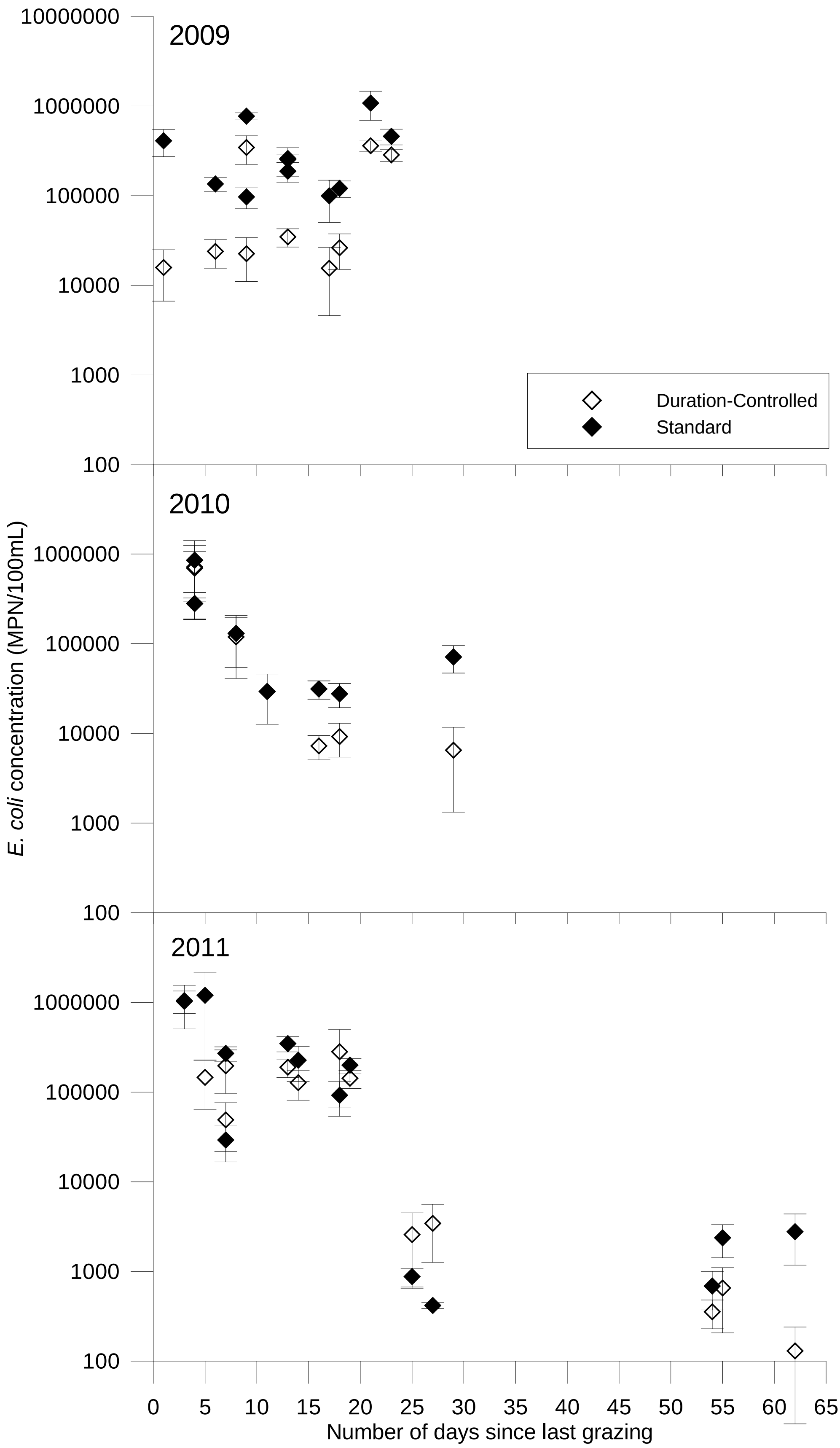

Figure 6.2: Average *E. coli* counts (most probable number/100 mL) in runoff occurring at different times after grazing events from *DC* and *SG* treatment plots, from 2009-2011.

The maximum runoff depth (runoff volume divided by the runoff plot area) from both treatments was 35.1, 33.1 and 19.8 mm in 2009, 2010 and 2011 respectively (Figures 6.1 and 6.3). After other parameters had been accounted for (i.e. days since grazing, number of cows grazed, and treatment) using stepwise regression analysis, runoff depth became a significant factor in predicting *E. coli* losses from the runoff treatment plots (Eq. 6.1). In 2009, runoff depth was not a significant factor in the *E. coli* count in runoff water (Figure 6.3). However 2010 and 2011 showed weak upward trends for *E. coli* with increasing runoff depth. This was probably also due to the climatic conditions prior to runoff. The solar radiation that created greater soil moisture deficits after grazing but before runoff in 2009 was likely to have an effect on faster protective skin formation and therefore less disruption of the dung pat. This resulted in relatively unchanged *E. coli* concentrations with increased amounts of runoff water.

Overall, the plot setup described in this study was useful in investigating the temporal effects of the source of faecal microbe losses in runoff. However, the attenuation of microbes that can occur between dung deposition and runoff occurring was not well simulated (Collins *et al.*, 2007). This was mainly because of the relatively small size of the plots and the large variability in micro topography between plots, and also because of the minimal distance to catchers and rapid movement of runoff water with little riparian buffering. In addition to riparian zones adjacent to paddocks, and therefore reducing runoff flow and faecal microbe transport to waterways, Collins *et al.* (2007) suggested creating riparian zones at a paddock level. This would increase the time and distance for faecal microbes to move through runoff, have a greater effect on the attenuation of the microbes, and thus lower numbers of faecal microbes would reach waterways. This could be expanded on further by reducing treading damage and dung deposition in all paddocks, by using *DC* grazing when the soil moisture deficit is low, as well as using riparian paddock zones for increasing travel times for microbes.

The effects of riparian plantings, treading damage and topography on reducing faecal microbe loss in runoff could not be fully investigated at a plot scale in this study, because of the high variability in climatic conditions between grazings and runoff occurring. To gain a greater understanding about these losses, a catchment scale trial, comparing *DC* and *SG* grazing, would need to be set up and monitored. This could also assist in observing P losses in runoff (Chapter 5).

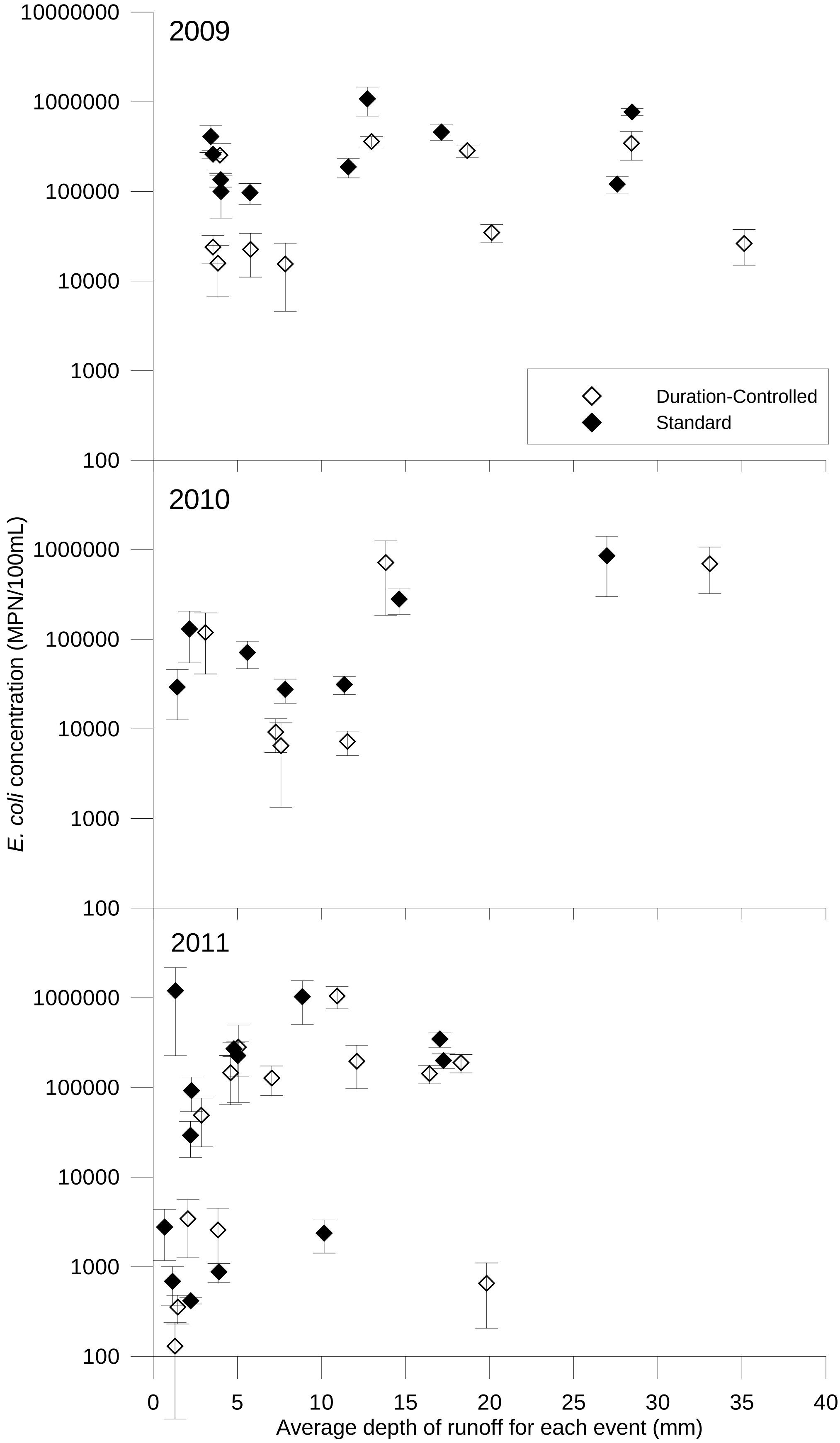

Figure 6.3: Average *E. coli* counts (most probable number/100mL) compared with runoff depth for each runoff event from *DC* and *SG* treatment plots from 2009-2011.

6.2.2 *E. coli* in drainage water

Measurements of *E. coli* in drainage water were made only in 2009 and 2010, for randomly selected drainage events. In 2009, the greatest average population of *E. coli* in drainage water from *SG* grazed plots was 240 000 MPN/100mL, while the greatest average population in 2010 from *SG* plots was 95 000 MPN/100mL (maximum and minimum concentrations from individual plots shown in Table 6.1). There was no significant difference (P>0.05) between treatments in 2009, and in 2010, there was only one drainage event, one day after grazing in September, that *E. coli* concentrations in drainage from *SG* plots were significantly (P<0.05) higher than those from *DC* plots. The transport of faecal coliforms into subsurface waters through fine textured soils that have been mole and tile drained, such as Tokomaru silt loam in this study, has been shown to be rapid and direct (McLeod *et al.*, 2003). Bypass flow is the main form of movement of faecal microbes (Collins *et al.*, 2007; McLeod *et al.*, 2003), which may be a result of fresh dung being directly transported in the drainage system soon after grazing. In addition to P losses from the movement of fresh dung, the rapid loss of faecal microbes in drainage after grazing is an area of research that should be developed further.

The average *E. coli* counts along with the minimum and maximum numbers detected in drainage water in 2010 were generally much lower than in 2009 (Table 6.1). This was most likely due to there being only one grazing event prior to the *E. coli* being measured in 2010, compared with up to three in 2009. The overall population of *E. coli* on the grazing plots would have been higher in 2009 with more dung deposited from a higher number of grazings.

Table 6.1: *E. coli* counts (most probable number/100mL) in drainage from *DC* and *SG* treatment plots in 2009 and 2010.

	2009		2010	
MPN/100mL	***DC***	***SG***	***DC***	***SG***
Average	49 000	61 800	17 000	22 000
Maximum	260 000	350 000	130 000	280 000
Minimum	2 500	3 100	<10	62

There was no consistent pattern between *E. coli* counts in drainage and days since grazing for either year. This is likely due to the relatively high background levels of *E. coli* in the soil and drainage system (Guy and Small, 1976; Sinton *et al.*, 2007; Muirhead, 2009; Muirhead and Monaghan, 2012), as well as in the collection areas. Furthermore, the mole and pipe drainage system, increasing the flow of sub-surface water, would most likely have decreased the attenuation of microbes within the soil (Collins *et al.*, 2007).

6.2.3 *E. coli* in slurry applied to land

The slurry applied to *DC* treatment plots in 2008 and 2010/11, as described in Chapters 3-5, was extremely high in *E. coli* concentrations. Random samples taken directly prior to application ranged from 630 000 to 5 800 000 MPN/100 mL. It is probable that these high concentrations had some impact on adding to the *E. coli* populations within the *DC* plots. However, exposure of the spread slurry to ultraviolet radiation would probably have gone some way to causing die-off of the *E. coli* (Crane and Moore, 1986).

Relative to *SG* plots, the *E. coli* counts in runoff from *DC* plots were larger than expected based on the reduction of dung deposition to these plots. This lower reduction in *E. coli* on *DC* plots may be explained by the application of slurry to these plots. Moreover, in 2009 there was no slurry applied to the *DC* plots (Chapter 3). *E. coli* concentrations from the *DC* treatment were more often significantly lower than those from the *SG* plots in 2009 than in subsequent years (Figure 6.2). Therefore, it is likely that the slurry being applied was contributing to greater *E. coli* populations on the *DC* treatment plots in 2010 and 2011. More comprehensive studies could be undertaken on the effect of slurry on *E. coli* losses if using a *DC* grazing system, particularly with the benefits of reducing treading compared to an *SG* system (Chapter 5).

6.3 Conclusion

Duration-controlled grazing did not produce lower *E. coli* losses in runoff compared with *SG* grazing plots in this study. This supports previous work that concentrations of *E. coli* and other faecal organisms vary greatly according to climatic conditions. In this study, there was a range of different climatic conditions occurring in the interval between grazings and runoff. Climate differences during these intervals, for instance prolonged dry conditions causing faster formation of protective skins on dung pats, was likely to have a greater effect than the treatment differences of reduced grazing time.

Time after grazing and the amount of runoff were important parameters contributing to *E. coli* concentrations in surface runoff. However, there were a number of factors causing large experimental error, including micro topography differences within the plots and large variation in runoff volumes between plots, and due to the variation, direct conclusions cannot be drawn on any one parameter. This was particularly after time since grazing was accounted for. To ensure receiving waters remained below the limit of 550 MPN/100 mL, large dilution factors of up to 658 from stream flow, containing no *E. coli*, would be required for concentrations in excess of 10^6 MPN/100 mL.

Overall, there were higher populations of *E. coli* in runoff than in drainage. This indicates that the prevalence of other pathogenic microbes would also be higher in runoff. Reducing the risk of *E. coli* loss in drainage may also be important because concentrations were much higher than the 550 MPN/100 mL standard set for recreational use.

The treatment difference in this study was reduced grazing time. The *SG* plots were grazed at the same time as *DC* plots using the same soil moisture rules (> 5 mm) to avoid treading damage. An *SG* system without suitable standoff facilities may be expected to cause more treading damage. If treading damage cannot be avoided by *SG* systems, then future research may be required to quantify the periodic effect of treading damage on *E. coli* losses in runoff or drainage.

This study has identified the limitations of using small plots to simulate P and faecal microbe loss in runoff from grazed pastures. Plot (850 m^2) trials usefully measured the ability of *DC* grazing to reduce N losses in drainage. When it comes to P and faecal microbe loss from dung in runoff, the variability in load or concentration delivered in a runoff event is large and is created by a number of interacting factors. A key source are dung pats, deposition of which are a function of stocking rate and animal behaviour creating a large source of spatial variability. Then, in the interval between deposition and runoff event, climatic effects influence source concentration and transportability. At the runoff event rainfall intensity, surface roughness, topography and vegetation influence suspended load in runoff and the attenuation features of the pathway to surface waters. The diversity and complexity of pathways of pathogenic microbe loss from a farm suggest that future studies should be characterised and studied at the catchment scale. Therefore, more work is needed at a catchment scale to quantify these pathways and mitigate the losses of faecal organisms to waterways by using tools such as *DC* grazing.

6.4 References

ASM, 1999. Microbial pollutants in our nation's water: environmental and public health issues. http://www.asm.org/images/docfilename/0000005987/waterreport[1].pdf.

Collins, R., McLeod, M., Hedley, M., Donnison, A., Close, M., Hanly, J., Horne, D., Ross, C., Davies-Colley, R., Bagshaw, C., Matthews, L., 2007. Best management practices to mitigate faecal contamination by livestock of New Zealand waters. New Zealand Journal of Agricultural Research 50, 267-278.

Collins, R., McLeod, M., Hedley, M., Horne, D.J., Hanly, J.A., Close, M., Bagshaw, C., Donnison, A., Ross, C., Matthews, L., 2005. Pathogen Transmission Routes: farm animals to waterways - Best Management Practices. Client Report prepared for MAF. NIWA.

Collins, R., Ross, C., Donnison, A., McLeod, M., 2003. Riparian Attenuation of Faecal Microbes - Objective 2 of the Pathogen Transmission Routes Research Programme. MAF Technical Paper No: 2003/7.

Connolly, J., Hedley, M.J., Ionas, G., Hanly, J.A., Horne, D.J., Collins, R., Donnison, A., Ross, C., 2004. Pathogen Transmission Routes: Farm Animals to Water Bodies (2002-2003). Client Report prepared for MAF. Fertilizer & Lime Research Centre, Massey University.

Crane, S.R., Moore, J.A., 1986. Modeling enteric bacterial die-off: A review. Water, Air and Soil Pollution 27, 411-439.

Guy, E.M., Small, J.A., 1976. Survival of streptococci and coliforms of bovine faecal origin in drainage water and soil stored at different temperatures. New Zealand Journal of Agricultural Research 20, 13-18.

Haynes, R.J., Williams, P.H., 1993. Nutrient cycling and soil fertility in the grazed pasture ecosystem. Advances in Agronomy 49, 119-199.

McBride, G.B., Salmond, C.E., Bandaranayake, D.R., Turner, S.J., Lewis, G.D., Till, D.G., 1998. Health effects of marine bathing in New Zealand. Int. J. Environ. Health Res. 8, 173-189.

McDowell, R.W., Houlbrooke, D.J., Muirhead, R.W., Muller, K., Shepherd, M., Cuttle, S.P., 2008. Grazed pastures and surface water quality. Nova Science Publishers.

McDowell, R.W., Wilcock, R.J., 2008. Water quality and the effects of different pastoral animals. New Zealand Veterinary Journal 56, 289-296.

McLeod, M., Aislabie, J., Ryburn, J., McGilla, A., Taylor, M., 2003. Microbial and chemical tracer movement through two Southland soils, New Zealand. Australian Journal of Soil Research 41, 1163-1169.

Ministry for the Environment, 2013. http://www.mfe.govt.nz/publications/water/microbiological-quality-jun03/html/part-two-e.html#twoe5.

Muirhead, R.W., 2009. Soil and faecal material reservoirs of Escherichia coli in a grazed pasture. New Zealand Journal of Agricultural Research 52, 1-8.

Muirhead, R.W., Collins, R.P., Bremer, P.J., 2005. Erosion and Subsequent Transport State of *Escherichia coli* from Cowpats. Applied and Environmental Microbiology 71, 2875-2879.

Muirhead, R.W., Littlejohn, R.P., 2009. Die-off of *Escherichia coli* in intact and disrupted cowpats. Soil Use and Management 25, 389-394.

Muirhead, R.W., Monaghan, R.M., 2012. A two reservoir model to predict Escherichia coli losses to water from pastures grazed by dairy cows. Environment International 40, 8-14.

Parliamentary Commissioner for the Environment, 2012. Water Quality in New Zealand: Understanding the Science. Parliamentary Commissioner for the Environment, Wellington.

Sinton, L.W., Braithwaite, R.R., Hall, C.H., MacKenzie, M.L., 2007. Survival of Indicator and Pathogenic Bacteria in Bovine Feces on Pasture. Applied and Environmental Microbiology 73, 7917-7925.

Vinten, A.J.A., Douglas, J.T., Lewis, D.R., Aitken, M.N., Fenlon, D.R., 2004. Relative risk of surface water pollution by E-coli derived from faeces of grazing animals compared to slurry application. Soil Use and Management 20, 13-22.

White, S.L., Sheffield, R.E., Washburn, S.P., King, L.D., Green, J.T., 2001. Spatial and time distribution of dairy cattle excreta in an intensive pasture system. Journal of Environmental Quality 30, 2180-2187.

Wilcock, R.J., Nagels, J.W., Rodda, H.J.E., O'Connor, M.B., Thorrold, B.S., Barnett, J.W., 1999. Water quality of a lowland stream in a New Zealand dairy farming catchment. New Zealand Journal of Marine and Freshwater Research 33, 683-696.

Chapter 7 Potassium cycling and the practice of standing cows off: risks and opportunities

C.L.Christensen[1,2], M.J.Hedley[1], J.A.Hanly[1], D.J.Horne[1]

[1] *Fertilizer & Lime Research Centre, Massey University, Private Bag 11222, Palmerston North, 4442, New Zealand*

[2] *C.L.Christensen@massey.ac.nz*

Adapted from a peer-reviewed paper published in the Proceedings of the Joint Australia and New Zealand Soil Science Societies Conference, Hobart, December 2012.

7.1 Abstract

As dairy farmers strive to reduce their environmental footprint while maintaining or improving productivity, the frequency and duration of standing cows off paddocks is increasing. While many of the benefits of standing cows on feedpads are understood, the challenges that this practice presents to nutrient management have not been comprehensively researched. Standing cows on feedpads impacts on nutrient cycles in a number of inter-related ways. The major effects of feedpad use on nutrient dynamics relate to the reduction in excreta deposition to paddocks, and consequently, the capture and management of this excreta as slurry and liquid effluent. While nitrogen (N) has received some attention, there have been very few studies of the impacts of standing cows on feedpads on the potassium (K) cycle (Chapter 2).

The effects of standing cows off paddocks on the K cycle within dairy pastures have been investigated over 3 years at Massey University's No. 4 Dairy Farm as part of a much larger investigation of Duration-controlled grazing (*DC*). As far as K is concerned, the *DC* treatment represented a form of standing cows on feedpads between grazings and returning effluent from a bunker at the end of the pad. Duration-controlled grazing was compared with standard grazing (*SG*) on a series of 14 plots. A range of soil, plant and water parameters were monitored.

Typically, feedpad bunker slurry has low K levels relative to its total solids content. The use of feedpad bunker slurry on the *DC* plots resulted in a net loss of 137 kg K/ha/yr from

these areas compared with the *SG* plots. Consequently, herbage K concentrations and soil QT K were significantly lower in the *DC* plots by the second season. Also, the cumulative annual losses of K in drainage were lower from *DC* plots compared with *SG*. This may have been not only due to less total K return, but also more even return of K, compared to K deposited in urine spots on the *SG* plots.

If *DC* grazing is to be implemented at a farm scale, it is important that the K content of all effluent sources is known and is considered in the farm's nutrient management plan.

7.2 Introduction

Many New Zealand dairy farmers stand their cows on feedpads for prolonged periods in order to protect soils and pasture from treading damage. Standing cows off paddocks also reduces nutrient losses to water ways. In other words, this practice is becoming more common and more important as it is a good fit with the New Zealand dairy industry's objective of increasing productivity while reducing its environmental footprint, with particular reference to nutrient loss to waterways (Parliamentary Commissioner for the Environment 2012).

The management of effluent and/or slurry generated on pads is challenging. Typically, the excreta deposited on feedpads is scraped into a bunker at the end of the pad. Rainwater on the pad also enters the bunker. In order to minimise the size of the bunker, the liquid component of this slurry is sent to the farm's effluent ponds. A weeping wall will often be used to separate the phases.

Standing cows on feedpads is likely to impact on nutrient cycles and nutrient management. While N usually receives the most attention (Chapter 4), standing cows on feedpads may also affect K. Standing cows off paddocks will affect the K cycle in numerous ways; most obviously, there will be less excreta (particularly urine) deposited in the paddocks. While, in theory, it is possible to return K in the excreta placed on the feedpad back to the paddocks, this is often not practical, not least of all because of the draining of much of the urine from the bunker. Also, different systems used for land application of effluent, bunker manure and pond sludges will impact on the K loading to different areas of the farm.

This paper briefly examines the effects of standing cows off (i.e. the *DC* grazing) on herbage, soil and drainage water K concentrations, the total amount of K lost as leaching (kg/ha), and compares these to an *SG* system. Practical implications of these effects are discussed.

7.3 Methods

7.3.1 Trial site

The experiment was conducted on Massey University's No. 4 Dairy Farm near Palmerston North, Manawatu, New Zealand (40° 23' 46.79" S; 175° 36' 35.77" E). The trial site was located in a flat landscape (<3% slope) which receives an average annual rainfall of approximately 1000 mm. The site had a mixed pasture sward of predominantly perennial ryegrass (*Lolium perenne*) and white clover (*Trifolium repens*). The trial was established on a mole-pipe drained Tokomaru silt loam soil, which is classified as a Fragic Perch-gley Pallic Soil (Hewitt 1998).

The research area consisted of 14 plots (~850 m^2/plot), each with an isolated mole and pipe drain system. Mole channels, ~ 40 m long, were installed at a depth of 0.45 m with an interval between moles of 2 m. Drainage from the mole channels was intercepted by a perforated pipe drain (0.11 m diameter) installed perpendicular to the moles at a depth of 0.60 m. Further description of the topography and soil properties of the site can be found in Houlbrooke *et al.* (2004).

7.3.2 Experimental design

The trial consisted of two treatments. The *SG* treatment involved a grazing duration of ~7 hours for day grazings and ~12 hours for night grazings. The other treatment was *DC* grazing which involved a grazing duration of ~ 4 hours for both day and night grazings. All plots were grazed on the same day with the same average stocking rate, which was set according to pasture cover (as estimated using a rising-plate pasture height meter). Grazings alternated between 'day' and 'night' regimes to simulate standard farm practice. There were 8-10 grazings per year. The trial site was established during the summer of 2008. Fertiliser applications, drainage dates and cow grazing targets have been described in Chapters 3 and 4.

7.3.3 Estimated pasture dry matter accumulation, herbage elemental analysis and dung deposition

A rising-plate pasture height meter was used pre- and post-grazing to estimate the quantity of pasture on all plots. Sixty measurements were taken per plot at each pre-and post-grazing event, and were used to estimate pasture accumulation between grazings, the stocking rate required at each grazing and cow intakes at each grazing. Prior to each grazing from January 2009, a total of 40 hand-plucked samples of herbage were taken from each plot, avoiding herbage directly surrounding dung pats and urine patches. This herbage was oven dried to 60°C, ground to 1 mm, and then underwent basic elemental analysis for macro and micro nutrients (K shown in this chapter) (Hill Laboratories Ltd, Hamilton, New Zealand).

The dung pats deposited on each plot were counted to give an indication of the total amount of excreta returned to the plots (described and discussed in Chapters 3 and 4). The difference in average dung deposition between the two treatments was used to estimate the quantity of excreta collected on the feed pad and, therefore, the amount of slurry to be applied to the *DC* treatment, based on N return. Slurry, derived from a feedpad bunker with a weeping wall, was first applied (5 to 10 mm) to the *DC* plots in mid-December 2008 (Chapters 3 and 4). This was the only application of slurry in the 2008/09 season. No slurry was applied during the second lactation season, while four, dilute applications of slurry were applied to *DC* plots in the third season. These applications had 4.8 kg dissolved granular KCl per plot added, to increase the K concentration of the slurry and be more aligned with collected material from a standoff facility (Chapter 3).

7.3.4 Drainage water volume measurements and water analysis

Drainage water collection methods have also been described in Chapters 3 and 4. Drainage water samples were filtered through a 0.45 μm filter. The filtered samples were analysed for K using atomic absorption spectroscopy on a GBC Avanta ∑ spectrometer (Walsh 1955).

7.3.5 Statistical analysis

Herbage K concentrations at each grazing were analysed using repeated measures in the generalised linear model procedure (PROCGLM), using the statistical package SAS v9.3 (SAS Institute Inc., Cary NC, USA) (SAS 2012).

7.4 Results and Discussion

Herbage K concentrations were similar in both treatments at all grazings in 2008/09 (albeit herbage analysis for this season was only measured from January). In 2009/10, the K concentration in *SG* herbage was higher ($P<0.05$) than in *DC* herbage in all but the first and third grazings. This trend continued in 2010/11, where herbage K in *SG* plots was greater than *DC* ($P<0.05$) at all grazings (Figure 7.1).

The K concentrations in drainage remained relatively constant each drainage season (majority of plots leaching < 4 ppm K). This contrasts with the pattern of NO_3^- concentrations in drainage (Chapter 4), which shows a general trend of being higher (8-20 ppm N) at the start of the drainage season and steadily decreases to lower concentrations (<1 ppm N) over the drainage season. Low and constant concentrations of K over the drainage season reflect the slow release of exchangeable soil K, whilst the lower concentrations in drainage from *DC* plots than *SG* plots resulted from less urine return. The total amount of K (kg/ha) lost in drainage water varied over the 3 drainage years after treatments were applied. On average, *DC* plots lost 42% less K than *SG* plots via drainage water, with the difference in losses increasing over time (Figure 7.2).

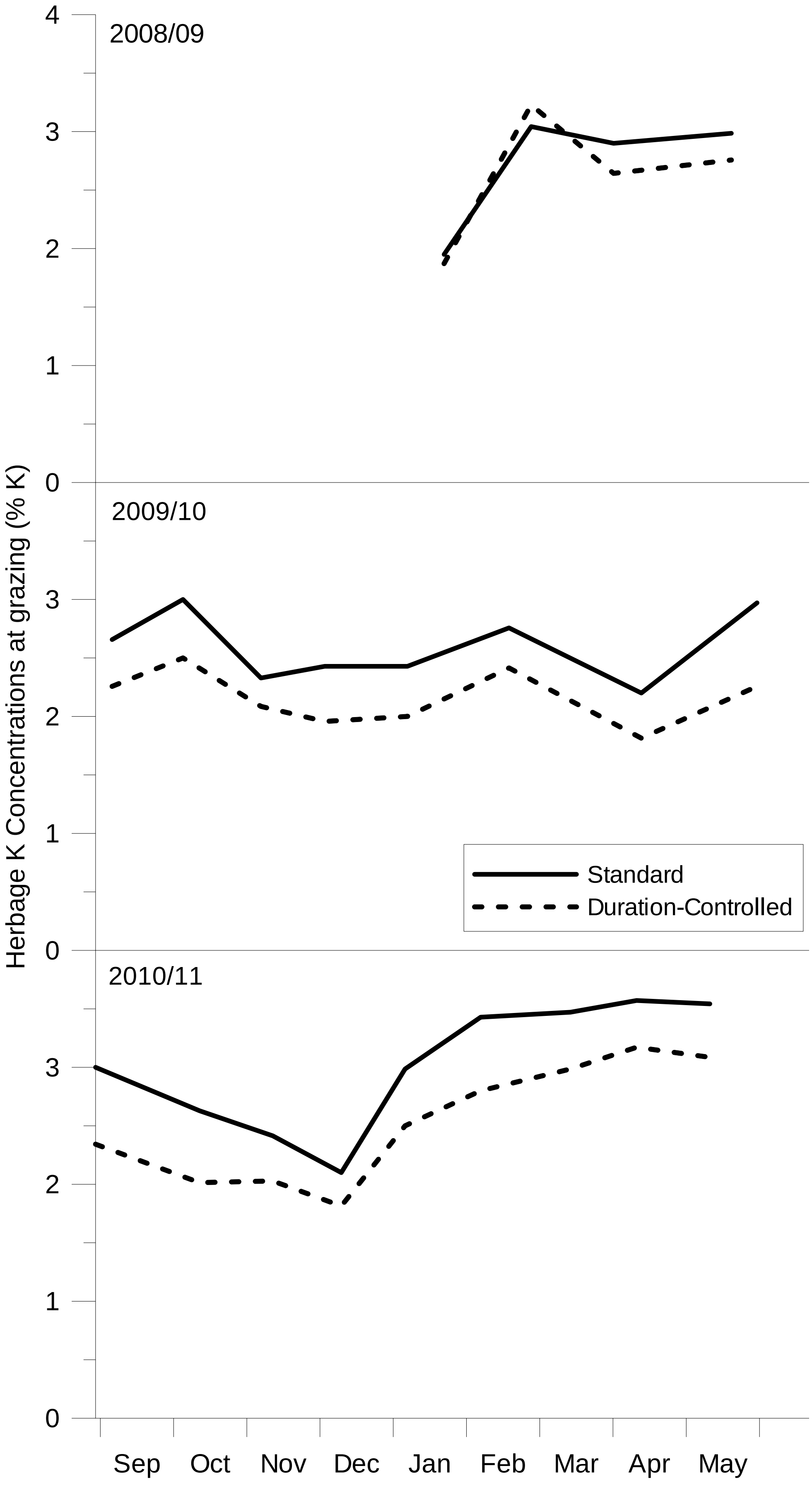

Figure 7.1: Average herbage potassium concentrations (% K) at each grazing for *DC* and *SG* grazing treatments from 2009-2011.

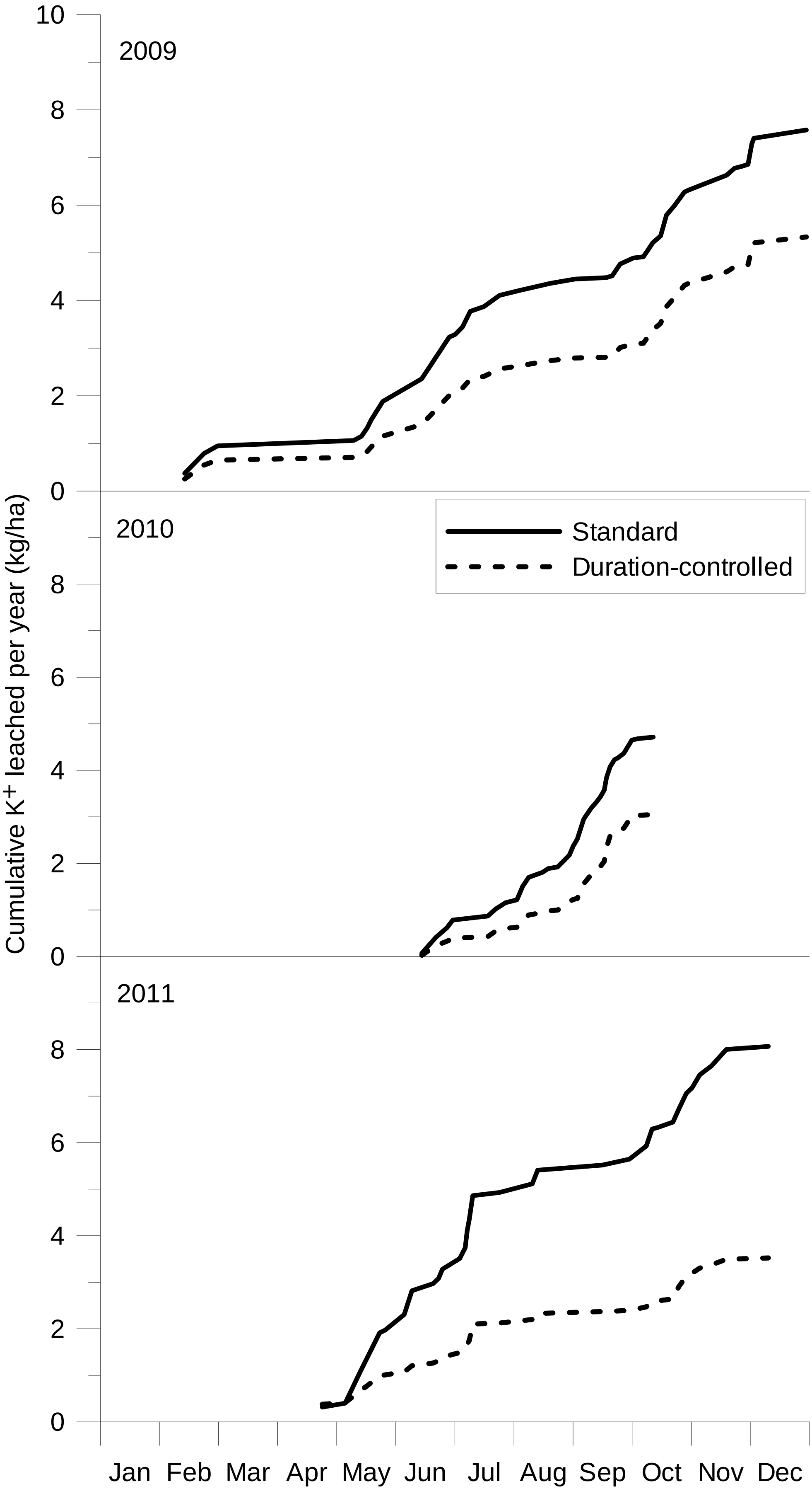

Figure 7.2: Cumulative potassium leached in drainage water for *DC* and *SG* treatments, from 2009-2011.

The widening gap in the quantity of K leached from treatments over time indicates that there was less available (leachable) K in the grazing system on the *DC* plots as time progressed. This trend was also evident in the herbage analyses. In addition, QT K soil test results in May 2010 were on average 4.6 for *SG* plots, while they were only 2.9 for *DC* plots (Chapter 3). Soil tests between treatments were similar at the commencement of the study.

In the 2008/09 season, an average of 66 kg K/ha was applied in slurry in the single application to the *DC* plots. In 2010/11, using the same K:N ratios as the first season for feedpad scrapings and estimates for the K concentrations in pond effluent, it is estimated that 72 kg K/ha was applied in slurry. Using the average K% in the pasture herbage (Figure 7.1) and silage fed on the feedpad, and the total feed mass ingested, the K transfer model of Salazar *et al.* (2010) was used to estimate the amounts of K removed from the plots and transferred to other areas of the farm. Based on the model and calculations from first principles, it was estimated that ~183 kg/ha/yr more K was transferred to the effluent system from the *DC* plots, compared to *SG* plots. The average return of K via slurry to the *DC* plots was 46 kg K/ha/yr, a shortage of ~137 kg K/ha/yr compared with the *SG* plots. A further modelling exercise was conducted using OVERSEER® Nutrient Budgeting software (more detail reported in Chapter 8). The net negative return of K to *DC* plots was similar using both models; however the OVERSEER® software over-predicted the three year average amount of K lost in drainage water from both the *SG* and *DC* plots by ~200% and 100% respectively. Moreover, the deficit of K returned helps to explain the decline in soil tests on the *DC* plots.

It is clear that the slurry applications from the feedpad bunker did not make up the difference (~183 kg K/ha) in the net K budget between the *SG* and *DC* grazing systems. The feedpad slurry was low in K because the liquid stream, containing urine K, was drained away from feedpad slurry to the dairy effluent storage pond. If cows are stood off paddocks regularly, the applied slurry must contain the urine collected from the feedpad if the available K balance in the soil is to be maintained. If it does not, the practice of standing cows off will create a greater K fertiliser requirement. However, areas that have received storage pond effluent in the past, hence with excessive K soil levels, could benefit from less grazing time (associated with standing off) in combination with the addition of only feedpad slurry. These results could be used in further development of models such as OVERSEER® (Chapter 8). If *DC* grazing is to be implemented at a farm scale, it is important that the K content of all effluent streams are known and are considered in the farms nutrient management plan.

7.5 References

Hewitt AE (Ed.) (1998) 'NZ Soil Classification (2 edn).' (Manaaki Whenua - Landcare Research New Zealand Ltd Press: Lincoln).

Houlbrooke DJ, Horne DJ, Hedley MJ, Hanly JA, Scotter DR, Snow VO (2004) Minimising surface water pollution resulting from farm-dairy effluent application to mole-pipe drained soils. I. An evaluation of the deferred irrigation system for sustainable land treatment in the Manawatu. *New Zealand Journal of Agricultural Research* **47**, 405-415.

Parliamentary Commissioner for the Environment (2012) Water Quality in New Zealand: Understanding the Science. Parliamentary Commissioner for the Environment, Wellington.

Salazar ME, Hedley MJ, Horne DJ (2010) Using turnips to reduce soil K loading on the effluent block. *Proceedings of the New Zealand Grassland Association* **72**, 247-250.

SAS (2012) SAS v9.3. SAS Institute Incorporated, Cary NC, USA.

Walsh A (1955) The application of atomic absorption spectra to chemical analysis. *Spectrochim. Acta* **7**, 108-117.

Chapter 8 Nitrogen loss mitigation using duration-controlled grazing: Field observations compared to modelled outputs

C.L. Christensen[1,2], M.J. Hedley[1], J.A. Hanly[1] and D.J. Horne[1]

[1]Fertilizer & Lime Research Centre, Massey University, Private Bag 11222, Palmerston North

[2]C.L.Christensen@massey.ac.nz

Adapted from a peer-reviewed paper published in the Proceedings of the New Zealand Grassland Association, Volume 74, 2012

8.1 Abstract

Dairy farmers in New Zealand are encouraged to adopt a range of management strategies, both well established and emerging, to reduce nitrogen (N) losses to waterways. In most regions the OVERSEER® nutrient budgeting software (Version 6) (hereafter referred to as Overseer) is the tool of choice in the assessment of N losses for both regulatory and monitoring purposes. As part of these processes, Overseer is used to assess the impact of improved farm practices on N leaching and runoff from individual farms. In a 3-year dairy system field trial at Massey University, N losses in leaching and runoff under duration-controlled grazing (*DC*; 4 hours per grazing) were compared with those under standard grazing (*SG*; 7 hours per day-grazing, 13 hours per night-grazing). A 36% reduction in total nitrogen (TN) losses under *DC* grazing was measured (14 kg TN/ha) relative to *SG* (22 kg TN/ha). Due to the impracticalities of measuring TN losses on-farm, farmers adopting *DC* grazing as a mitigation strategy will only be able to claim the reduction in TN losses estimated by Overseer, and thus observations from the field trial were compared with outputs from Overseer. There was good agreement between the Overseer predictions of N leaching and values measured at the trial site for both the *SG* and *DC* grazing treatments. A second Overseer simulation of a *DC* system suggests that while Overseer is able to predict the reductions in N leaching under *DC* grazing reasonably well, some issues such as runoff losses and storage of effluent need further consideration.

Keywords: Duration-controlled grazing; OVERSEER®; N leaching

8.2 Introduction

The New Zealand dairy industry is striving for increased productivity and a smaller environmental footprint, with particular emphasis on decreasing nutrient losses to waterways (Parliamentary Commissioner for the Environment, 2012). A modelling exercise conducted by Monaghan *et al.* (2008) identified a number of farm management strategies that effectively reduce the amounts of nitrate (NO_3^-) leached from dairy farms. One of these strategies involved restricting the grazing hours of cows in autumn and winter. This strategy is effective if it can reduce the deposition of urinary N in concentrated patches in the paddock and instead collect that urine when the cow is stood-off the pasture and return it at a uniformly low rate. The potential of restricted grazing as a strategy to reduce losses of N, phosphorus (P) and faecal microbes from dairy farms to surface water was tested in a field study at Massey University over 3 years (Chapters 3-6). Duration-controlled grazing (*DC* grazing), is a year-round management system that aims to utilise pasture *in situ*, but limits the time that cows spend grazing paddocks to 8 hours per 24-hour period, with the remaining 12 hours spent on stand-off facilities (and 4 hours milking). The effects of this system on pasture production and losses of N, P and faecal microbes have been measured, and reported in Chapters 3-6.

New Zealand farmers have the benefit of having freely available access to OVERSEER® nutrient budgeting software (hereafter referred to as Overseer). The model, which simulates the inputs, transformations and losses from New Zealand's farming systems, has algorithms based on a large number of New Zealand field and farmlet trials. Overseer allows nutrient budget calculations to be estimated from production and farm resource data that is on-hand (Overseer, 2013). Developed by AgResearch Ltd, and owned by the Ministry of Primary Industries, Fertiliser Association of New Zealand, and AgResearch Ltd, Overseer was primarily developed to be used as an on-farm reporting tool. However, it is now widely used for regulatory purposes as well as for modelling potential mitigation strategies.

This paper will compare the field trial results from the *DC* grazing trial with the Overseer simulations of this system. Differences will be discussed, and consideration will be given to how field data can be used for further development of the modelling software.

8.3 Methods

8.3.1 Trial site

The 3-year field trial, established in 2008, was conducted on Massey University's No. 4 Dairy Farm near Palmerston North, Manawatu, New Zealand (40° 23' 46.79" S; 175° 36' 35.77" E). The trial site was located in a flat landscape (*ca.* <3% slope), which receives an average annual rainfall of approximately 1000 mm. The site had a mixed pasture sward of predominantly perennial ryegrass (*Lolium perenne*) and white clover (*Trifolium repens*). The trial was established on a mole-pipe drained Tokomaru silt loam soil, which is classified as a Fragic Perch-gley Pallic Soil (Hewitt 1998).

The research area consisted of 14 plots (*ca*. 850 m^2/plot), each with an isolated mole and pipe drain system. Mole channels, *ca.* 40 m long, were installed at a depth of 0.45 m and spaced 2 m apart. Drainage from the mole channels was intercepted by a perforated pipe drain (0.11 m diameter) installed perpendicular to the moles at a depth of 0.60 m. Further description of the topography and soil properties of the site can be found in Houlbrooke *et al.* (2004).

Within each grazing plot there was also a 50 m^2 surface runoff sub-plot, isolated on 3 sides by buried wooden boards and grading to a gutter on the 4th side to collect storm surface runoff. The runoff was then directed to a tipping bucket water flow recorder and sampler.

8.3.2 Experimental design

The experimental design has been described in Chapters 3 and 4.

8.3.3 Estimated dung deposition and slurry application

The dung pats deposited on each plot were counted to give an indication of the total amount of excreta returned to the plots. The difference in annual average dung depositions between the two treatments was used to estimate the quantity of excreta that would be transferred to the standoff facility and therefore the amount of slurry to be applied to the *DC* treatment. The model of Salazar *et al.* (2010), which is based on herbage intake and N concentration, was also used to estimate the amount of excreta returned to the plots. The estimate of excreta transferred to the standoff facility by the Salazar model (Table 2) was very similar to the estimate given by counting dung pats. Based on the observations of Longhurst *et al.* (2006) that 45% of the dung plus urine N could be lost in storage over a period of 3 months, the long-term annual average slurry application to the *DC* plots was planned to deliver approximately 55% of the difference in N deposited on the *SG* plots compared with the *DC* as urine and dung. In mid-December 2008, slurry, derived from a feedpad bunker with a weeping wall, was first applied to the *DC* plots (212 kg N/ha) at twice the annual N application rate in dry soil conditions (Chapter 4). This represented a biennial application and was the only application of slurry in the 2008/09 season: no slurry was applied during the second lactation season. Four lighter applications totalling 115 kg N/ha were applied to *DC* plots in the third season (Table 8.2).

8.3.4 Drainage water volume measurements and water analysis

Drainage and surface runoff water from plots were channelled through drainage pipes into tipping-bucket flow meters located in sampling pits nearby. Each tipping-bucket was calibrated dynamically to account for slightly larger tip volumes at greater flow rates. All tipping buckets were instrumented with data loggers to provide continuous measurements of flow rate. During each drainage and surface runoff event, a proportion (*ca.* 0.1%) of the water from every second tip of the tipping bucket was automatically collected to provide a volume-proportional sample for water quality analysis.

Unfiltered water samples were digested using the alkali persulphate method of Hosomi and Sumo (1986). They were then analysed using colorimetric methods on a Technicon Auto Analyser (Blakemore *et al.* 1987), to give concentrations of total nitrogen (TN), including organic and mineral components.

8.3.5 Comparison of measured *vs*. modelled losses

Data describing the *SG* management regime were entered into Overseer (Version 6; Web version accessed 13th September 2012), (Table 8.1). A *DC* system was also modelled in Overseer (hereafter called the *MDC1* system). Many of the input parameters for the *MDC1* system were based on the field trial e.g. the inclusion of a covered animal shelter as the standoff area, with 8 hours per day grazing, and 100% of the on-farm cows using the animal shelter year-round. In the Overseer simulation, the pad was scraped (without water) and the effluent kept separate from the farm dairy effluent system. Subsequently, the majority of the '*MDC1*' farm had animal shelter effluent applied to it, the liquid being sprayed infrequently, and the solids being applied four times per year (November, December, February and April). The quantity of imported supplementary feed and manufactured fertiliser, cow numbers and milksolids production were the same as values used in the *SG* system (Table 8.1).

In a second exercise, the inputs to Overseer were modified so as to more closely represent effluent return to the *DC* plots in the field trial (*MDC2*). This allowed for an assessment of the agreement between an Overseer prediction of N losses from a *DC* grazing system and measured values.

Table 8.1: Parameters entered into Overseer (based on Massey No. 4 Dairy Farm) for both *SG* and *MDC* treatment systems

<table>
<tr><td>Cow numbers / Breed / Total farm area</td><td colspan="3">630 / Friesian x Jersey / 211 ha</td></tr>
<tr><td>Total milk solids production</td><td colspan="3">226800 kg/yr</td></tr>
<tr><td>Supplements Imported (all on DM basis)
(SG fed on feed pad for 1 hour per day during lactation;
DC fed in covered animal shelter with 8 hours grazing per day)</td><td colspan="3">Maize silage (180 t/yr)
Triticale silage (90 t/yr)
Pasture silage (120 t/yr)
Hay (15 t/yr)
Palm kernel meal (40 t/yr)</td></tr>
<tr><td>Average rainfall</td><td colspan="3">980 mm/yr</td></tr>
<tr><td>Soil type / drainage</td><td colspan="3">Tokomaru silt loam / mole & pipe drained</td></tr>
<tr><td>Fertiliser inputs (kg/ha; month)</td><td colspan="3">Sulphur super 30 (200 kg/ha; March)
Urea (38 kg/ha; April)
Ammonium sulphate (71 kg/ha; October)
Ammonium sulphate (71 kg/ha; November)
Urea (76 kg/ha; December)</td></tr>
<tr><td>Winter management SG and DC</td><td colspan="3">40% cows off 1st June to 31st July</td></tr>
<tr><td rowspan="2">Soil tests

(Assuming the same values over whole farm, from soil tests performed on trial plots)</td><td>Olsen P (mg/kg)</td><td>QT K</td><td>SO_4-S (mg/kg)</td></tr>
<tr><td>30</td><td>8</td><td>7</td></tr>
</table>

8.4 Results and Discussion

Measured TN leaching losses from the *SG* treatment were 18, 13 and 26 kg N/ha/yr (average of 19 kg N/ha) for the years 2009, 2010 and 2011, respectively. The variation in leaching values across the 3 years is partly explained by differences in drainage patterns. Although total drainage amounts were similar over the 3 years (373, 316 and 329 mm), the initiation and length of the drainage seasons varied. Furthermore, the concentrations of NO_3^--N in drainage water, making up on average 64% of the TN concentration, varied markedly with length of drainage season (Chapter 4). This was expected due to the seasonal nature of N availability and pasture uptake, and the coincidence of grazing and drainage events (Houlbrooke 2005).

Total N leaching losses of 13, 7 and 14 kg N/ha/yr (average 11 kg N/ha/yr) were measured from the *DC* treatment. Compared to the *SG* treatment, this translates into a reduction in TN leaching of 28%, 46% and 46% over the three years, 2009, 2010 and 2011, respectively. This equated to a 42% reduction on average, which exceeds the Dairy Industry's goal of a 30% reduction in N leaching losses (DairyNZ 2010).

When the surface runoff losses (3.2 kg N/ha for *SG* and 2.9 kg N/ha for *DC*) were added to the leaching losses, the average overall TN losses to water for the *SG* treatment was 22 kg N/ha/yr, compared with 14 kg N/ha/yr for the *DC* treatment (a 36% reduction). Therefore, on average 83% of TN losses were through drainage, which is consistent with the values measured by Houlbrooke (2005) at the same site.

When the *SG* treatment data were entered into Overseer, the N loss to water for *SG* grazing was estimated to be 23 kg N/ha/yr, which was slightly higher than the 3-year measured average value for the *SG* plots (22 kg N/ha/yr). While these losses were very close, Overseer predicted that all of the N lost to water was as leaching and no N was lost as runoff, whereas 3.2 kg N/ha was measured in runoff in the field study.

In order to predict N losses to water, Overseer simulates N transfer via dung, urine and effluent streams and predicts the N loss outcome associated with grazing and effluent management. The *DC* grazing strategy reduces the urine deposition to paddocks and uniformly applies the captured excreta as a slurry or effluent – all of these processes can be simulated within Overseer's model framework.

The Overseer simulation of the *MDC1* system predicted an average TN loss to water of 14 kg N/ha/yr, which was, in general terms, similar to the average loss measured in the trial for this treatment. Again, Overseer predicted that all of the loss to water was as leaching, with no runoff losses of N, compared to a 2.9 kg N/ha runoff loss measured in the study.

In the Overseer prediction for the *MDC1* system, the annual average effluent application was 176 kg N/ha/yr. It is notable that the theoretical (Salazar *et al.* 2010) and measured effluent return to the *DC* plots averaged 136 and 109 kg N/ha/yr, respectively (Table 8.2). However, the long-term average return of effluent N predicted by Overseer at 176 kg N/ha/yr was 40 and 67 kg N/yr above the theoretical target and measured slurry returns.

Table 8.2: The estimated annual return of nitrogen in dung and urine to *DC* and *SG* treatments, the theoretical target slurry application, the difference in N returned to *DC* plots, the actual return of N in slurry, and Overseer predictions (kg N/ha/yr).

Year	Amount N estimated returned as dung and urine[1]	Theoretical target slurry application[1,2]	Fertiliser applied (3yr annual average)	Theoretical Total N returned	Theoretical difference in N returned to *DC* plots	Actual N application in form of slurry
2008/09 *SG*	383		60	443		
DC	155	136	60	351	-92	212[3]
2009/10 *SG*	340		100	440		
DC	137	121	100	358	-82	nil
2010/11 *SG*	425		85	510		
DC	172	151	85	408	-102	115
Ave *SG*	383		82	464		
Ave *DC*	155	136	82	372	-92	109
Overseer estimate		176	82			

[1]Estimated using modified version of model by Salazar *et al.* (2010) based on number of grazings in each year and average herbage N concentrations.
[2]Assuming 45% losses of N from animal shelter, based on Longhurst *et al.* (2006).
[3]Biennial strategy of slurry application.

One possible reason for the difference between estimated and measured N returns in effluent/slurry is that Overseer may assume smaller N losses in storage than the Longhurst *et al.* (2006) values, which were used in the theoretical calculations. Nitrogen losses during storage are difficult to predict, not least of all because of variation in storage time between farms.

In the second exercise, the Overseer simulation of the *MDC* system was modified (*MDC2*) to deliver 109 kg N/ha/yr as effluent (i.e. to approximate the *DC* treatment at the field trial). This was achieved by keeping all other parameters the same as the *MDC1* system but increasing the covered storage time of the animal shelter effluent solids to 36 months. With an effluent application of 109 kg N/ha, Overseer predicted TN losses to water from the system at 10 kg N/ha/yr from leaching and 0 kg N/ha/yr from runoff, compared with the field measurements of 11 kg N/ha/yr from leaching and 3 kg N/ha/yr from runoff. This result shows that when the N returned to pasture in effluent was adjusted in Overseer to equal the average returns in the trial (109 kg N/ha/yr), then the N leaching predicted by Overseer was in good agreement with the measured values. However, Overseer did not predict the runoff N losses measured in the trial. Runoff is inherently difficult to model, particularly from annual average inputs, due to the different factors that influence these losses.

There is general agreement between N losses to water in Overseer and the results seen in the field trial. Overseer was able to predict the N leaching loss reductions achievable using the *DC* mitigation strategy, which demonstrates that it has been partially updated to reflect the results from research of mitigation strategies (Overseer 2012), such as using animal shelters. An important point to take from this work is that measured N loss to water varied by *ca.* 50% for both treatments between the three years. Overseer's attempt at predicting the long-term average N loss to water for the *SG* treatment was within 1 kg N/ha/yr, and the model did predict the large reduction from *DC* plots, within 4 kg N/ha/yr. While 4 kg is a large percentage difference in this situation, the important aspect remains the large reduction caused by the mitigation, and reflected in the Overseer predictions.

However, there remains uncertainty surrounding the simulation of N returned as stored effluent and N losses in runoff. It is important for the quantities of effluent N returned to pastures in *DC* grazing systems to be accurately estimated because a much larger proportion of nutrients are returned to pastures as effluent compared to standard grazing practice. Further, farm-scale research of *DC* grazing systems is required to provide improved understanding of nutrient dynamics in stored effluent and the quantity of nutrients actually returned to pastures in these systems.

8.5 Conclusion

In a 3 year field trial at Massey University, *DC* grazing resulted in a 36% reduction in TN losses (leaching plus runoff) relative to a standard dairy grazing system. When a duration-controlled system was modelled using Overseer (*MDC1*), the predicted reduction in TN losses was 39% compared with a standard grazing system. When *DC* grazing as practised at the field trial was simulated with Overseer (*MDC2*), there was good agreement with measured values for N leaching.

Overseer (Version 6) seems to predict N leaching losses under *DC* grazing reasonably well. Issues that require further consideration in Overseer relate to the proportion of TN losses in runoff, and the quantity of N returned to pastures in effluent collected from stand-off facilities.

8.6 References

Blakemore, L.C.; Searle, P.L.; Daly, B.K. 1987. Methods for chemical analysis of soils. New Zealand Soil Bureau Scientific Report 80.

DairyNZ 2010. Dairy Industry Strategy - directions to the vision. Hamilton, NZ.

Hewitt, A.E. (Ed.) 1998. NZ Soil Classification (2 edn). Manaaki Whenua - Landcare Research New Zealand Ltd Press: Lincoln.

Hosomi, M.; Sudo, R. 1986. Simultaneous determination of total nitrogen and total phosphorus in freshwater samples using persulfate digestion. International Journal of Environmental Studies 27(3/4); 267-275.

Houlbrooke, D.J., 2005. A study of the quality of artificial drainage under intensive dairy farming and the improved management of farm dairy effluent using 'deferred irrigation'. PhD Thesis. Massey University, Palmerston North.

Houlbrooke, D.J.; Horne, D.J.; Hedley, M.J.; Hanly, J.A.; Scotter, D.R.; Snow, V.O. 2004. Minimising surface water pollution resulting from farm-dairy effluent application to mole-pipe drained soils. I. An evaluation of the deferred irrigation system for sustainable land treatment in the Manawatu. New Zealand Journal of Agricultural Research 47; 405-415.

Longhurst, R.D.; Luo, J.; O'Connor, M.B.; Pow, T. 2006. Herd Homes: nutrient management and farmer perceptions of performance. Proceedings of the New Zealand Grassland Association 68; 309-313.

Monaghan, R.M.; de Klein, C.A.M.; Muirhead, R.W. 2008. Prioritisation of farm scale remediation efforts for reducing losses of nutrients and faecal indicator organisms to waterways: A case study of New Zealand dairy farming. Journal of Environmental Management 87; 609-622.

Overseer (2012). Changes to the pastoral N model - Technical Note No. 5.

Overseer (2013). www.overseer.org.nz, accessed 26th November 2013.

Parliamentary Commissioner for the Environment 2012. Water Quality in New Zealand: Understanding the Science. Parliamentary Commissioner for the Environment, Wellington, NZ.

Salazar, M.E.; Hedley, M.J.; Horne, D.J. 2010. Using turnips to reduce soil K loading on the effluent block. Proceedings of the New Zealand Grassland Association 72; 247-250.

Chapter 9 Summary and opportunities for future work

9.1 Summary and justification of this study

The expansion and intensification of the dairy industry has contributed to deterioration of fresh water quality in specific regions of New Zealand. Conflicts arise between aspirations for a profitable dairy industry that remains environmentally sustainable. Future dairy industry strategies will need farm management techniques that can maintain profitability whilst reducing nutrient losses to waterways. Reviews of N and P cycles in grazed dairy systems indicate that management strategies to reduce nitrate leaching must focus on decreasing the intensity or concentration of N in urine patches deposited to pasture.

One strategy is to implement *DC* grazing on farms. To evaluate the effectiveness of this on-off grazing technique, a large scale, field plot trial was established, comparing *DC* grazing with 'best-management' *SG* over three years. To simulate *DC* grazing with a standoff facility, cows were removed from pasture after 4 hours of grazing between milkings to reduce the deposition of urine and dung. Stored effluent from a feedpad and farm dairy effluent was re-applied to *DC* plots to complete the return of dung and urine, when soil conditions allowed.

Several data sets were collected from the comparison and have been presented in this thesis. These focused on the impact of *DC* grazing on pasture production and subsequent estimated cow intakes, and nutrient loss through mole and pipe drainage and surface runoff water. Nitrogen, P, K and faecal microbe concentrations were measured. From these, the annual loads of N, P, K and faecal microbes being lost in drainage and surface runoff water were able to be calculated and compared with an *SG* system.

9.2 Main findings from this research

9.2.1 Pasture accumulation and cow intakes (Chapter 3)

From the outset, trial plots were only grazed if the soil moisture deficit was >5 mm so as to minimise treading damage by cows. Therefore, most differences between *DC* and *SG* plots reflect differences in the duration of grazing, and there was no difference in treading between treatments. The number of cows grazed was according to the average pasture cover, to achieve an intake of ca. 5-6 kg DM of pasture per grazing, for both treatments. Pasture accumulation on the *DC* and *SG* plots was on average 12% and 25% higher, respectively, than the annual average accumulation on the No. 4 dairy farm. A 20% reduction in 2009/10 and a 9% reduction in 2010/11 on the *DC* plots compared with the *SG* plots was due to less nutrient return through urine and dung. An important finding is that slurry should not be stored for summer application, but should be returned to the *DC* plots early enough in the lactation season for nutrients to be available for rapid pasture uptake. To simulate a slurry from a cow housing system, the slurry should be at a mineral N content of at least 40% of TN.

9.2.2 Nitrogen losses (Chapter 4)

Large and significant reductions in N losses in drainage were realized within the first season of using *DC* grazing. An average 52% reduction in NO_3^- leached and 42% reduction in TN leached was measured from the *DC* grazed treatment plots over the 3 years of the study. These reductions were mostly from the smaller number of urine patches deposited on to the paddock, even with effluent returned back as slurry.

When the measured data (pasture accumulation and intakes and cow grazing durations) were used to model urine N outputs by cows and uptake by pasture, it was very clear that the urine deposited in the autumn months contributed most to the N leaching in the following winter. There was a strong correlation between urine-deposited N not being taken up by pasture, and the N leaching (both NO_3^- and TN) for both treatments in all 3 years. This modelling demonstrated that the proportion of N leached from a urine patch was the N that had not been utilized for pasture growth or immobilized or denitrified during the grazing season. Therefore, particularly on fine-textured, mole and pipe drained soils *DC* grazing could be used only in the autumn months to achieve large reductions in winter N leaching.

The amount of mineral N (predominantly NO_3^- with negligible levels of NH_4^+) being leached was strongly related to the amount of urine deposited on the paddocks. As TN was mostly comprised of NO_3^-, then this decreased with less urine deposited as well. However, there were always steady levels of TON being leached, with the cumulative load of TON increasing with increasing drainage depths. Therefore, the amount of TON leached was able to be estimated by drainage depth, regardless of grazing duration or intensity, while the NO_3^- leached was able to be predicted using N intakes and outputs, particularly from autumn grazings. In turn, this enabled TN leaching losses to be estimated when comparing a *DC* grazing system with that of an *SG* system.

Surface runoff losses of N were very low and *DC* grazing did not have a significant effect on reducing them.

9.2.3 Phosphorus losses (Chapter 5)

There was an average 32% decrease in DRP and TP in surface runoff water from *DC* grazed plots over the three years of this study. However, the large variation in runoff depth from runoff plots meant that the reduction in P load was not statistically significant. Also, 61% of the TP losses were as surface runoff, while 39% were in drainage water. There were a greater proportion of P losses in drainage than was initially expected; however the P load in drainage was similar to what had been reported from earlier work at the same site.

The losses of TP in drainage increased as the drainage season progressed, particularly after spring grazings occurred, indicating that organic P losses were being influenced by grazing and dung deposition.

9.2.4 Faecal coliform losses (Chapter 6)

Due to the large variation in runoff depth from plots and *E. coli* counts (MPN/100mL water), significant differences in *E. coli* enumeration were also not realised between grazing treatments. There were also many other factors contributing to the loss of *E. coli* in surface runoff water. The factor that explained most variation was the number of days since grazing. This was an expected result, with many other research studies reporting similar findings. The effect of the number of days since grazing was shown clearly in this study, due to 3 years of runoff data, with very different grazing patterns. For example, 2010 only had one grazing prior to all runoff occurring and 2011 had 3 grazings through the runoff season. The climate after grazing, number of cows grazed, treatment (time spent grazing) and runoff depth were also all significant factors in determining *E. coli* concentrations over the 3 years.

9.2.5 Potassium losses (Chapter 7)

It was evident that K status within the soil and pasture changed rapidly depending on management. Duration-controlled grazing caused a rapid decline in QT K soil tests, as well as a significant difference in herbage K measurements. Much of this was because the slurry being added was low in K, as the K-rich liquid effluent (i.e. urine) was drained away. Therefore it is important that when effluent as slurry is applied, the K levels must be adequate to replace the K lost from lower urine returns. For high K areas (e.g. effluent areas) on farms, then *DC* grazing could be a way to reduce the K loading over time. The amount of K leached was overestimated by Overseer®, but the results found in this study will be able to be used to further develop such models.

9.2.6 Validation of Duration-controlled grazing within Overseer® (Chapter 8)

Overseer® is the industry-used standard model for nutrient budgeting and management in New Zealand. As *DC* grazing has proven to be an effective management tool for reducing N leaching, it is important that Overseer® is able to simulate the reductions in N leaching that can be realised using this system. In general, Overseer® was a very good predictor of TN losses to water in both *DC* and *SG* grazed systems. However, there was some disparity between the split of N loss via drainage and runoff within the model. The model over-predicted N loss in runoff, as well as N loss from areas other than urine patches, and under-predicted N leaching losses from urine patches. Results from this study will be able to be used for further development of the Overseer® model.

9.3 Limitations of this research

As described in Chapters 2 and 3, previous modelling by other researchers resulted in the hypothesis that by spreading collected effluent back to paddocks evenly as slurry, then pasture accumulation would increase. This was not the case in this 3 year study, with a 9% reduction in pasture accumulation from 4 applications of slurry applying a total of 115 kg N/ha over a lactation season. The anticipated positive effect of slurry application on subsequent pasture production was not observed within 3 years. However, it did become apparent that timing (i.e. applications early in the lactation season) and mineral N content (i.e. >40%) were critical factors in maximising pasture accumulation under slurry application (Chapter 3). In addition, the K content of the slurry was more important than initially thought, with rapid changes in soil and pasture K on *DC* plots when slurry low in K was applied (Chapter 7). The solution to this issue is to ensure that the urine component is retained during the storage of slurry to be re-applied to pasture.

To gain more accuracy when modelling N intake, output and uptake, the herbage N content of urinary areas and inter-urine areas should be measured. The average N content of the pasture provided a good indication of what the cows were eating (and thus excreting in urine) but for future pasture growth, the urine patch N content would have provided a higher level of detail when conducting this exercise (Chapter 4). This would also assist with future development within the Overseer® model, defining N leaching from the timing of grazings and deposition of urine (Chapters 4 and 8).

There were large differences in runoff depth measured from the plots, in both treatments. Therefore no significant reductions in P loads from *DC* plots were found, even though DRP and TP concentrations were lower from this treatment. The micro topography, which affected temporary ponding volumes and the partitioning of water between runoff and drainage, varied between plots. The use of boards to surround runoff plots was also problematic in some cases. Damaged boards during grazing resulted in flow on from outside areas to the runoff plots. Moreover, channelized flow from adjacent paddocks into the runoff collection areas was an issue. Minimising the outside influence of runoff probably would have assisted with more consistent runoff depths (Chapters 5 and 6). This may have resulted in statistically significant differences between treatments. Also, the transport of *E. coli* in runoff may have been able to be statistically differentiated if runoff depths were more consistent (Chapter 6). Minimising outside flow may be achieved by runoff borders protruding further out of the ground, and/or being made from a more flexible material to suit grazing cows. Larger runoff plots may also assist with reducing runoff volume error.

9.4 Opportunities for future research

The implementation of a *DC* grazing system requires the inclusion of cow housing or a standoff facility to a farm. Because such structures are costly and pose a financial risk to dairying in New Zealand, it would be worthwhile to conduct further research on the effects of limiting *DC* grazing to certain periods of the year, for different outcomes. For example, *DC* grazing could be implemented on a seasonal basis to achieve a desired outcome. Duration-controlled grazing would be practised in autumn to achieve N leaching reductions; winter and spring, for reducing treading damage and P and faecal microbe losses; and summer, to reduce heat stress on cows.

Targeting the return of effluent as slurry to overcome nutrient limitations, for example spring, would assist in creating 'best practice' guidelines for farmers employing *DC* grazing systems. In spring, N is limiting in soil after the drainage season, and therefore larger amounts of N are required for high pasture production. This study has highlighted the need for nutrient return early in the season and therefore there is scope to improve the timing of slurry application in such a system. This needs to be as early as practicably possible for machinery and slurry tankers to drive on paddocks. Also, adding slurry from a purpose-built stand-off facility would enable the correct effluent to be applied, as it would be the excreta collected from standing cows off pasture.

The amount and type of P lost in drainage is an issue of greater magnitude than first anticipated. More quantification is needed of the transport of dung through mole and pipe drainage systems, which appear to be contributing to higher organic P losses, particularly after spring grazings. A short term grazing study should be established in the spring to monitor the transport of dung through drainage systems. This would also assist in quantifying the load of faecal microbes being transported to artificial drainage directly from fresh dung deposition. In addition, a catchment scale study comparing *DC* and *SG* grazing could be established to include such parameters as treading damage and riparian areas to monitor the reductions in P and faecal microbe losses through runoff.

No economic analysis was done as part of this study. If *DC* grazing is to become a formal management strategy on-farm, then a thorough economic analysis should be carried out. This should include costs of suitable stand-off facilities, extra labour, and slurry applications, as well as potential benefits such as increased pasture production and possible lower fertilizer requirements.

There is an opportunity to increase the scope of this research to a farm scale, complete with a cow house, to monitor the effect of *DC* grazing on other parameters. For example, the effects on milk production and cow health (including mastitis and lameness) should be measured. If this research is conducted, then the economic effects would be analysed and reported on a whole farm scale, with the standoff structure being incorporated into the costs of such a system.