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**A STUDY OF THE QUALITY OF ARTIFICIAL DRAINAGE  
UNDER INTENSIVE DAIRY FARMING AND THE  
IMPROVED MANAGEMENT OF FARM DAIRY  
EFFLUENT USING 'DEFERRED IRRIGATION'.**

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## **ABSTRACT**

The last decade has been a period of great expansion and land use intensification for the New Zealand dairy farming industry with a 44% increase in national dairy cow numbers. Intensive dairy farming is now considered to be a major contributor to the deterioration in the quality of surface and ground water resources in some regions of New Zealand. Previous research has demonstrated intensive dairy farming is responsible for accelerated contamination of waterways by nutrients, suspended solids, pathogenic organisms and faecal material. A number of common dairy farming practices increase the risk of nutrient leaching. In particular, farm dairy effluent (FDE) has been implicated as a major contributor to the degradation of water quality. With the introduction of the Resource Management Act in 1991, the preferred treatment for FDE shifted away from traditional two-pond systems to land application. However, on most farms, irrigation of FDE has occurred on a daily basis, often without regard for soil moisture status. Therefore, it has been commonplace for partially treated effluent to drain through and/or runoff soils and contaminate fresh water bodies.

The objectives of this thesis were to design and implement a sustainable land application system for FDE on difficult to manage, mole and pipe drained soils, and to assess the impacts of FDE application, urea application and cattle grazing events on nutrient losses via artificial drainage and surface runoff from dairy cattle grazed pasture. To meet these objectives a research field site was established on Massey University's No.4 Dairy farm near Palmerston North. The soil type was Tokomaru silt loam, a Fragiaqualf with poor natural drainage. Eight experimental plots (each 40 x 40 m) were established with two treatments. Four of the plots represented standard farm practice including grazing and fertiliser regimes. Another four plots were subjected to the same farm practices but without the fertiliser application and they were also irrigated with FDE. Each plot had an isolated mole and pipe drainage system. Four surface runoff plots (each 5 m x 10 m) were established as subplots (two on the fertilised plots and two on the plots irrigated with FDE) in the final year of the study. Plots were instrumented to allow the continuous monitoring of drainage and surface runoff and the collection of water samples for nutrient analyses.

An application of 25 mm of FDE to a soil with limited soil water deficit - simulating a 'daily' irrigation regime - resulted in considerable drainage of partially treated FDE. Approximately 70% of the applied FDE left the experimental plots with 10 mm of

drainage and 8 mm of surface runoff. The resulting concentrations of N and P in drainage and runoff were approximately 45% and 80% of the original concentrations in the applied FDE, respectively. From this single irrigation event, a total of 12.1 kg N ha<sup>-1</sup> and 1.9 kg P ha<sup>-1</sup> was lost to surface water representing 45% of expected annual N loss and 100% of expected annual P loss.

An improved system for applying farm dairy effluent to land called 'deferred irrigation' was successfully developed and implemented at the research site. Deferred irrigation involves the storage of effluent in a two-pond system during periods of small soil moisture deficits and the scheduling of irrigation at times of suitable soil water deficits. Deferred irrigation of FDE all but eliminated direct drainage losses with on average <1% of the volume of effluent and nutrients applied leaving the experimental plots. Adopting an approach of applying 'little and often' resulted in no drainage and, therefore, zero direct loss of nutrients applied.

A modelling exercise, using the APSIM simulation model, was conducted to study the feasibility of practising deferred irrigation at the farm scale on No 4 Dairy farm. Using climate data for the past 30 years, this simulation exercise demonstrated that applying small application depths of FDE, such as 15 mm or less, provided the ability to schedule irrigations earlier in spring and decreased the required effluent storage capacity.

A travelling irrigator, commonly used to apply FDE (a rotating irrigator), was found to have 2-3 fold differences in application depth and increased the risk of generating FDE contaminated drainage. New irrigator technology (an oscillating travelling irrigator) provided a more uniform application pattern allowing greater confidence that an irrigation depth less than the soil water deficit could be applied. This allowed a greater volume to be irrigated, whilst avoiding direct drainage of FDE when the soil moisture deficit is low in early spring and late autumn. A recommendation arising from this work is that during this period of low soil water deficits, all irrigators should be set to travel at their fastest speed (lowest application depth) to minimise the potential for direct drainage of partially treated FDE and associated nutrient losses.

The average concentrations of N and P in both 2002 and 2003 winter mole and pipe drainage water from grazed dairy pastures were all well above the levels required to prevent aquatic weed growth in fresh water bodies. Total N losses from plots representing standard farm practice were 28 kg N ha<sup>-1</sup> and 34 kg N ha<sup>-1</sup> for 2003 and

2004, respectively. Total P losses in 2003 and 2004 were 0.35 kg P ha<sup>-1</sup> and 0.7 kg P ha<sup>-1</sup>, respectively. Surface runoff was measured in 2003 and contributed a further 3.0 kg N ha<sup>-1</sup> and 0.6 kg P ha<sup>-1</sup>.

A number of common dairy farm practices immediately increased the losses of N and P in the artificial drainage water. Recent grazing events increased NO<sub>3</sub><sup>-</sup>-N and DIP concentrations in drainage by approximately 5 mg litre<sup>-1</sup> and 0.1 mg litre<sup>-1</sup>, respectively. The duration between the grazing and drainage events influenced the form of N loss due to a likely urine contribution when grazing and drainage coincide, but had little impact on the total quantity of N lost. Nitrogen loss from an early spring application of urea in 2002 was minimal, whilst a mid June application in 2003 resulted in an increased loss of NO<sub>3</sub><sup>-</sup>-N throughout 80 mm of cumulative drainage suggesting that careful timing of urea applications in winter is required to prevent unnecessary N leaching.

Storage and deferred irrigation of FDE during the lactation season caused no real increase in either the total-N concentrations or total N losses in the winter drainage water of 2002 and 2003. In contrast, land application of FDE using the deferred irrigation system resulted in a gradual increase in total P losses over the 2002 and 2003 winter drainage seasons. However, this increase represents less than 4% of the P applied in FDE during the lactation season.

An assessment of likely losses of nutrients at a whole-farm scale suggests that it is standard dairy farming practice (particularly intensive cattle grazing) that is responsible for the great majority of N and P loss at a farm scale. When expressed as a proportion of whole-farm losses, only a very small quantity of N is lost under an improved land treatment technique for FDE such as deferred irrigation. The management of FDE plays a greater role in the likely P loss at a farm scale with a 5% contribution to whole-farm P losses from deferred irrigation.

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# CHAPTER 1: INTRODUCTION

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## 1.1 REASONS FOR THE STUDY

The New Zealand dairy farming industry has come under increasing criticism in recent times for its role in the deterioration of the quality of New Zealand's surface and ground water resources (Parliamentary Commissioner for the Environment 2004). In particular, artificial drainage networks such as mole and pipe systems have been identified as providing a rapid transport pathway for the water contaminants associated with dairy farming (Sharpley and Syers 1979, Monaghan *et al.* 2002, Monaghan *et al.* 2005). Recent research has demonstrated that drainage and surface runoff water from intensively grazed dairy pasture can contribute to increased nutrient and solids loads to New Zealand water ways. This literature will be further examined in the review of literature (Silva *et al.* 1999; Gillingham and Thorrold 2000; Di and Cameron 2002; Monaghan *et al.* 2002; McDowell *et al.* 2003; Smith and Monaghan 2003).

Twenty-five years ago, Sharpley and Syers (1979) investigated the impacts of dairy farming practises on nutrient losses from a mole and pipe drained Tokomaru silt loam in the Manawatu region of New Zealand. A strong relationship was found between N and P leaching, and suspended solids loss and the incidence of high intensity cattle grazing events (300 cows ha<sup>-1</sup> 12 hour period). Further loss of N was associated with winter applications of urea fertiliser.

Dairy farming has changed considerably since the 1970s. The past 10 years has been a period of intensification and expansion with a 44% increase in national dairy cow numbers (Livestock Improvement 2003). Cattle grazing and excretion behaviour, urea fertiliser and farm dairy effluent (FDE) application have been identified as potential contributors to nutrient losses under dairy farming (Silva *et al.* 1999, Di and Cameron 2002, Monaghan *et al.* 2002, Monaghan *et al.* 2005).

With the advent of the Resource Management Act in 1991, there was a considerable shift in the regulatory authorities' (Regional Councils) preferred management option for FDE from a traditional two-pond treatment system to land application. However, land treatment of FDE has been shown to have a number of adverse environmental effects, including the surface runoff and/or drainage of partially treated (i.e. nutrient-rich)

effluent to fresh water bodies (Macgregor *et al.* 1979; Di *et al.* 1998; Singleton *et al.* 2001, Monaghan and Smith, 2004). Furthermore, high application rates of FDE may result in a decrease in soil quality; in particular surface sealing caused by FDE may decrease surface infiltration and drainage rates (Heatley 1996, Cameron *et al.* 1997). The grazing of wet soils by cattle subsequently increases the risk of soil treading and compaction damage (Drewry *et al.* 2004).

There are currently no criteria for the scheduling of FDE irrigation on New Zealand dairy farms. In addition, many dairy farms operating without a FDE storage capability will have to land apply FDE almost every day irrespective of soil moisture content. Such a practice will increase the risk of drainage and/or runoff of nutrient-rich, partially treated FDE particularly in the spring period when soil moisture is often close to or at field capacity. Furthermore, these problems will be exacerbated where land application occurs on difficult to manage mole and pipe drained soils (Monaghan and Smith 2004).

To help overcome the problems caused by the land application of FDE, the research presented in this thesis aimed to develop a sustainable land treatment system, which has been termed 'deferred irrigation'. This study describes the design, implementation and refinement of a deferred irrigation system. In brief, deferred irrigation involves storing FDE in an existing two-pond treatment system and using a soil water balance to strategically irrigate this effluent to land at times of suitable soil moisture deficit so as to avoid drainage of raw or partially treated FDE.

A field site was established in the Manawatu region of New Zealand on a mole and pipe drained Tokomaru silt loam. This soil is classified as an Argillic-fragic Perch-gley Pallic Soil (Hewitt 1998) or a Typic Fragiaqualf (Soil Survey Staff 1998). Eight plots were installed to allow the investigation of the nutrient losses in winter/spring drainage and runoff associated with urea applications, dairy cattle grazing events and FDE irrigation.

## 1.2 RESEARCH OBJECTIVES

This thesis has the following research objectives;

- To design a sustainable land treatment system called 'deferred irrigation' that minimises nutrient loss from areas that receive FDE and to implement this system on a typical mole and pipe drained soil.
- To compare the environmental impacts of deferred irrigation of FDE with traditional and current effluent treatment options.
- To model pond storage requirements for FDE assuming deferred irrigation is strictly implemented.
- To test the performance of traditional FDE travelling irrigators and new irrigator technology, and to attempt modifications where possible to assess the implications of irrigator performance on associated environmental impacts.
- To measure the impacts of intensive dairy farming on the quality and quantity of drainage and surface runoff water exiting a mole and pipe drained soil. In particular, the effects of the following dairy farm practises on nutrient loss will be assessed;
  - FDE application using deferred irrigation,
  - Dairy cattle grazing events, and
  - Urea fertiliser application.
- To assess the relative contributions of the aforementioned dairy farm practises to nutrient losses at a whole-farm scale.

## 1.3 THESIS STRUCTURE

This thesis is divided into eleven chapters including this 'Introduction'. Chapter 2 is a review of the literature and is split into two parts. Section 2.2 reviews literature on the land treatment of FDE in New Zealand and its impact on water quality. This section has been published in the special 'Farm Effluent' edition of the 'New Zealand Journal of

Agriculture Research', 2004, Vol 47:499-511. Section 2.3 reviews the New Zealand and Australian literature on the impacts of intensive dairy farming on nutrient losses to the aquatic environment and currently is unpublished. Chapter 3 describes the research site, the soil and the experimental design common to all aspects of the thesis.

Chapters 4-8 report and discuss the experimental data collected during the study. Each of these chapters have been written in the format of a journal paper and as such have their own introduction, materials and methods, results and discussion and conclusion sections. While they have been written as discrete papers, these chapters form a coherent, unified argument. Four of the five results chapters presented have been published as peer reviewed, journal or conference papers whilst the fifth results chapter has been submitted for publication.

Chapter 4 presents the design and evaluation of the deferred irrigation system for sustainable land treatment of farm dairy effluent and has been published in the special 'Farm Effluent' edition of the 'New Zealand Journal of Agriculture Research', 2004, Vol 47:405-415. Chapter 5 reports on the performance of travelling effluent irrigators and has been published in the special 'Farm Effluent' edition of the 'New Zealand Journal of Agriculture Research', 2004, Vol 47:587-596.

Chapter 6 presents results on mole and pipe drainage losses of N and P under intensive dairy farming for the 2002 winter/spring drainage season and has been published in the 'Proceedings of the New Zealand Grassland Association', 65:179-184. Chapter 7 presents the results on mole and pipe drainage losses of N and P under intensive dairy farming for the 2003 winter/spring drainage season and has been submitted for publication to the 'Australian Journal of Soil Research'. Chapter 8 reports on nitrogen drainage losses when cattle grazing is immediately followed by a drainage event and has been published in the 'Proceedings of the 3rd Australian New Zealand Soils Conference'.

Chapter 9 is an, as yet, unpublished chapter presenting the results of simulations of the soil water balance and the quantity of FDE storage required in ponds when deferred irrigation is practised at a farm scale. Climate data for the past thirty years was used in this modelling exercise.

Chapter 10 is an extrapolation of plot research data to provide an estimate of whole-farm nutrient losses. The findings from chapters 4-10 are then brought together in a final summary chapter (Chapter 11).

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# CHAPTER 2. REVIEW OF LITERATURE

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## 2.1 INTRODUCTION

This review of literature has been split into two sections. Section 2.2 reviews published literature on the land treatment of FDE in a New Zealand context with particular reference to the effectiveness of nutrient removal and the implications for drainage water quality. Section 2.2 is a published journal paper and is presented in its paper format including its own references list. Section 2.3 is an unpublished review of the literature that addresses the nutrient losses in drainage and surface runoff water associated with grazing dairy cattle and fertiliser application.

A comment about the scope of this literature review is in order. The literature that is reviewed below is deliberately confined to that from New Zealand and Australia. Dairy farms in these two countries differ from most northern hemisphere dairy production systems in numerous ways, not least of all, in the amount of time that cows spend grazing pasture *in situ*. Cows that spend all of the year grazing on paddocks will have very different impacts on nutrient losses in drainage and surface runoff compared to cows that spend prolonged periods indoors (Chadwick *et al.* 2002). In other words, as nutrient losses from patches of excreta play such a significant role in nutrient losses to the aquatic environment from grazed pasture systems, it is not clear what relevance the research conducted where cattle excreta is applied back to the land in a very uniform and artificial manner has to the present study. In addition, the substantial difference in the diets of housed cows and those whose feed is comprised primarily of fresh pasture will also impact on the magnitude of nutrient losses to waterways. There are further dissimilarities in winter and spring drainage patterns associated with differences in climates.

## 2.2 A REVIEW OF LITERATURE ON THE LAND TREATMENT OF FARM DAIRY EFFLUENT IN NEW ZEALAND AND ITS IMPACT ON WATER QUALITY

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### 2.2.1 ABSTRACT

Dairy farming is the largest agricultural industry in New Zealand, contributing 20% of export earnings but providing a challenge for the environmentally acceptable treatment of wastes from dairy farms. Nutrient-rich farm-dairy effluent (FDE), which consists of cattle excreta diluted with wash-down water, is a by-product of dairy cattle spending time in yards, feed-pads and the farm dairy. Traditionally, FDE has been treated in standard two-pond systems and then discharged into a receiving fresh water stream. Changes brought about primarily due to the Resource Management Act of 1991 have meant that most Regional Councils now prefer dairy farms to land treat their FDE. This allows the water and nutrients applied to land in FDE to be utilised by the soil-plant system.

Research on the effects of land treating FDE, and its affects on water quality, has shown that between 2 and 20% of the nitrogen (N) and phosphorus (P) applied in FDE is leached through the soil profile. In all studies, the measured concentration of N and P in drainage water was higher than the ecological limits considered likely to stimulate unwanted aquatic weed growth. Gaps in the current research have been identified with respect to the application of FDE to artificially drained soils, and the lack of research that has taken place with long term application of FDE to land and at appropriate farm scale with realistic rates of application. Whilst the land treatment of FDE represents a huge improvement on the loss of nutrients discharged to fresh water compared to

standard two-pond systems, there is room for improvement in the management of FDE land-treatment systems. In particular it is necessary to prevent the direct discharge of partially treated FDE by taking into account soil physical properties and soil moisture status. Scheduling effluent irrigations based on soil moisture deficits results in a considerable decrease in nutrient loss and may result in a zero loss of raw or partially treated effluent due to direct drainage.

**Keywords** farm-dairy effluent, land treatment, nitrogen, phosphorus, water quality, spray irrigation, pathogens.

## 2.2.2 INTRODUCTION

The period between 1993 and 2003 was one of remarkable expansion and growth for the dairy industry in New Zealand, the country's largest export earner (Statistics New Zealand, 2000). In this ten-year period, national dairy cow numbers increased by 44%, from 2.6 to 3.74 million cows. Likewise stocking rate increased from 2.5 to 2.6 cows hectare<sup>-1</sup>. By 2003 the total effective land area grazed by milking cows had increased to 1.5 million hectares (Livestock Improvement Corporation 2003). These increases in cow numbers, stocking rate and total area in dairy farming generated greater volumes of farm-dairy effluent (FDE) requiring treatment. Poorly-operated effluent treatment systems (two-pond or land application), particularly in areas newly converted to dairying, began to have visible impacts on surface-water and ground-water quality (Monaghan *et al.* 2002b).

To estimate the quantities of FDE being generated, the following approximations can be used. Dairy cows spend approximately 2 hours per day, over the milking season in the farm dairy or yards (i.e. effluent collecting areas) (Cameron & Trenouth 1999). If it is assumed that they deposit excreta at a constant rate over a 24 hour period, then approximately 10% of their excreta will be deposited within the farm dairy and yards. Cameron & Trenouth (1999) state that this percentage ranges from 5 to 15% and depends greatly on cattle management practices. Following milking, the yards are washed down with high-pressure hoses using approximately 50 litres of water cow<sup>-1</sup> day<sup>-1</sup> (Vanderholm 1984). The resulting wash-down water contains a diluted mix of cattle faeces and urine, along with any chemical that may have been used to wash down the milking plant.

The physical, chemical and biological characteristics of FDE are highly variable and change within and between farms due to contrasting management of storm water, feed pads, wash-down waters, chemicals, age and breed of herd, and stock management whilst in the yards (Heatley 1996; Longhurst *et al.* 2000). The average composition of FDE comprises 8% excreta, 4% teat washing and 86% wash down water (Longhurst *et al.* 2000). FDE has a pH of approximately 8 (Heatley 1996). The solid content of FDE from over 63 sites ranged from 0.04 – 5.3%, with an average of 0.9% (Longhurst *et al.* 2000). Longhurst *et al.* (2000) also reported mean nitrogen (N) and phosphorus (P) concentrations for raw FDE, from a number of different research trials, to be 269 mg N litre<sup>-1</sup> (range 181 – 506) and 69 mg P litre<sup>-1</sup>(range 21-82). Using the average concentrations, this suggests that, relative to the 1993 situation, there is now approximately an extra 16 million cubic metres of FDE in New Zealand per year, containing 4300 tonnes of N and 1100 tonnes of P, requiring treatment.

Historically, the common form of treatment for FDE has been the two-pond system combining both an anaerobic and facultative pond (Sukias *et al.* 2001). In the first pond, organic matter in the effluent is digested by anaerobic fermentation, a mixture of partially digested faeces and 'carried-in soil' settles out of suspension and accumulates at the bottom of the pond as sludge. The second pond, which is a facultative pond, is usually larger in surface area and shallower. The bottom layer continues to act as an anaerobic treatment whilst the top layer of the pond provides an aerobic treatment.

The combination of an aerobic and anaerobic pond efficiently removes sediment and biological oxygen demand (BOD), but high concentrations of nutrients remain (Hickey *et al.* 1989; Ledgard *et al.* 1996; Longhurst *et al.* 2000; Sukias *et al.* 2001; Craggs *et al.* 2003). Longhurst *et al.* (2000) reported that effluent discharging from a standard two-pond system to surface waters has approximate concentrations of 91 mg N litre<sup>-1</sup> and 23 mg P litre<sup>-1</sup>. Although the two-pond treated effluent represents a significant improvement compared to the nutrient concentrations of raw FDE, the concentrations of N and P in two-pond treated FDE are still three and two orders of magnitude greater than the levels (0.1 mg N litre<sup>-1</sup> or 0.1 mg P litre<sup>-1</sup>) considered likely to promote aquatic weed growth (Ministry for the Environment 1992).

With the introduction of the Resource Management Act (RMA) in New Zealand in 1991, the two-pond treatment system, with discharge to a stream, began to be phased out by regulatory authorities and is now regarded as a discretionary activity by most regional councils whereby councils retain discretion over all aspects of a consent and all

applications for a consent are generally notified (Parminter 1995; Heatley 1996; Cameron & Trenouth 1999). The RMA provided regional councils with the legislative control to prevent adverse environmental impacts such as discharge of nutrient-rich effluent from two-pond systems. Furthermore, the RMA stated that regional councils must recognise and provide for Maori spiritual and cultural values. Maori advocate that wastes should not be disposed of in water but returned to the land (Cameron & Trenouth 1999). Land application of FDE, taken from either an existing two-pond system or directly from a sump holding the daily wash-down of FDE, became the preferred treatment option for many of the regional councils in the 1990s (Environment Waikato 1994; Parminter 1995; Heatley 1996).

Whereas the impacts of land disposal of industrial and municipal wastes have been widely researched and reviewed (e.g. Cameron *et al.* 1997) and an understanding developed of what constitutes best and poor management practices, research on FDE application to land and the long-term impacts on soil chemical and biophysical quality are less common and have not been extensively reviewed.

The objective of this review is not to provide an analysis of the processes involved in the treatment of effluents when applied to soil, because publications such as Cameron *et al.* (1997) and Bond (1998) have already extensively reviewed the technical and environmental issues associated with land treatment of effluents. This publication aims to review published data on the land treatment of FDE in a New Zealand context with particular reference to the effectiveness of nutrient removal and implications for water quality.

### **2.2.3 SYSTEM DESIGN**

The aim of land application of effluents is to utilise the soil/plant system to absorb, filter and breakdown all waste components of applied effluent, so as to minimise the risk of high nutrient loads and harmful micro-organisms leaching or draining into sources of fresh water (Cameron *et al.* 1997; Bond 1998; Brewer *et al.* 1999; Tillman and Surapaneni 2002). Land treatment allows FDE to be considered a fertiliser resource that supplies N, P, and K as well as trace elements Ca, Mg and Na to the land in a liquid form (Hart & Speir 1992; Parminter 1995; Heatley 1996; Longhurst *et al.* 2000). Effluent can represent a 10 – 12 % saving of a farm's annual fertiliser requirements, however, because FDE is not a balanced fertiliser, additional applications of P are sometimes required (Ledgard *et al.* 1996; Longhurst *et al.* 2000).

The source of FDE for land application is either stored effluent, which may have been partially treated, or effluent which has come directly from the yard via a sump that collects daily wash down liquid. Such effluent is raw with no treatment and little or no settling. Storage facilities vary in size from those with the capability to hold a few days' production of effluent through to those that are able to store upwards of two months' effluent. Many farmers are able to use existing two-pond systems to partially treat and store very large volumes of FDE.

Small, self-propelled travelling irrigators are commonly used to apply FDE to land in New Zealand. These irrigators are generally low maintenance and have a range of operating speeds and, hence, are able to apply effluent at a variety of depths (Heatley 1996). Contract vehicle spreaders can be used to periodically draw FDE from a pond or storage facility and then apply to chosen paddocks. However, the one-off nature of such applications dictates that application rates may need to be high. Slurry stirrers can also be used to remove accumulated sludge from the bottom of ponds and apply it in the same manner (Heatley 1996). Flood irrigation can be used on free-draining soils to apply FDE via furrows and borders. Efficient surface coverage requires a well-graded land surface. Land flooding of FDE can be added to existing border-dyke operations and can therefore be a low cost method of application, however the risk of environmental pollution is high (Heatley 1996). The soil injection method, commonly used in Europe involving the placement of effluent directly into the soil opened up with disks or tines, is not regularly used in New Zealand (Heatley 1996).

Land application of FDE in New Zealand is controlled by regional councils, who determine whether this is considered a permitted (no consent required), controlled (consent required but granted if all appropriate conditions are met) or discretionary activity (consent declined or granted at the discretion of the council). Farmers have to comply with a number of conditions stipulated by their local regional council. Conditions vary between councils but usually take into account hydraulic and nitrogen loading, proximity to water ways, neighbouring properties and system design and maintenance. All of the regional councils rely, to some extent, on outcome-focused rules which specify that land applications should not cause effluent to enter surface water and that application depths should be set accordingly to meet this requirement (Cameron & Trenouth 1999).

The minimum area of land to be set aside for applications of FDE is usually 3 – 5 ha per 100 cows or no less than 10% of the total farm area. This minimum area is set by the maximum permissible annual N loading, commonly 150 - 200 kg N ha<sup>-1</sup> yr<sup>-1</sup> (Environment Waikato 1994; Heatley 1996; Cameron & Trenouth 1999). Selvarajah (1996) builds a case for a maximum annual effluent N loading rate of 150 kg N ha<sup>-1</sup> based on soil mineral N budgeting. He estimated that most Waikato soils would require FDE at a rate of approximately 60 kg N ha<sup>-1</sup> yr<sup>-1</sup> to meet the annual soil mineral N deficit plus another 15 kg N ha<sup>-1</sup> yr<sup>-1</sup> to replace volatilisation and denitrification mineral N losses from the applied effluent. This equates to 75 kg N ha<sup>-1</sup>. Assuming that 50% of total FDE N is in the mineral N form, then FDE at the rate of 150 kg N ha<sup>-1</sup> yr<sup>-1</sup> is required to alleviate the annual mineral N deficit. However, in the long term it was noted that increased N leaching losses could be experienced due to increased mineralisation of N from the increases in the soil organic N pool.

Some Regional Councils do not specify either a maximum daily or per application depth, however many have adopted a range between 15 - 25 mm (Heatley 1996). The Dairying for the Environment Committee (Heatley 1996) recommends a maximum application rate of 10 – 30 mm hr<sup>-1</sup> depending upon soil type.

#### 2.2.4 PASTURE GROWTH RESPONSE

Application of FDE to pastures below optimum fertility (particularly those low in N) can cause marked increases in pasture yield. The nutrients taken up by the extra pasture growth can be considered to be part of the land treatment effect, temporarily associating nutrients with carbon in a non-leachable pool.

An effluent pasture growth study was carried out by Goold (1980) from 1972 – 1976 in the Northland region, near Whangarei, on Waikare clay. Four treatments were set up: control, water-only and two application rates of raw FDE (12 mm and 6 mm every 21 days) irrigated from a sump. Prior to the application of FDE the site was of high fertility (Truog P level - 80). Substantial increases in annual herbage yield were measured under FDE addition (27% for the low FDE application rate and 43% for the higher rate). This increase in pasture growth was attributed to N input from the FDE as the water-only treatment provided no increase in yield compared to the control treatment and the soil already had a high P status. Average nutrient loadings from the low rate of FDE treatment were 156 kg N, 46 kg P and 348 kg K ha<sup>-1</sup> yr<sup>-1</sup>. The high rate of K input from

the high rate of effluent treatment resulted in a significant ( $P < 0.05$ ) increase in soil levels of K.

Roach *et al.* (2001) reported on two pasture growth trials in the regions of Taranaki and the Waikato that received applications of FDE on dairy grazed pasture. The fertility status (Olsen P) was not reported for either trial sites. FDE applications increased pasture yield of all three treatments by up to 17% and was equal to the yield increases gained by the equivalent urea N loading. Such a result suggests that there is more N available for plant uptake than the 50% available mineral N in FDE assumed by Selvarajah (1996). This could be a likely result of mineralisation of N from organic N applied previously in FDE. Measures of pasture nutrient composition indicated that levels of K had increased considerably under high rates of effluent application. Similar results were obtained in the Waikato trial where increased pasture growth as a result of FDE application ranged from 7% ( $75 \text{ kg ha}^{-1} \text{ yr}^{-1}$ ) to 24% ( $375 \text{ kg ha}^{-1} \text{ yr}^{-1}$ ).

Changes in pasture growth rates and nutrient concentration were also noted by Bolan *et al.* (2004) when FDE was applied to a high fertility perennial ryegrass (*Lolium perenne*) and white clover (*Trifolium repens*) pasture (Olsen P 25-30 mg P ml<sup>-1</sup>) at the No. 4 dairy unit of Massey University. FDE (aerobic pond) with N, P, K, Ca, and Mg concentrations of 135, 22.1, 231, 15.2, and 11.5 mg litre<sup>-1</sup>, respectively was applied for a 6-month period at rates of 0, 150 and 200 kg N ha<sup>-1</sup>. Pasture dry matter yield increased with an increasing rate of FDE application. The dry matter response ranged from 4.1 to 7.2 kg dry matter kg<sup>-1</sup>N applied, which was approximately 60% of that achieved with a similar rate of N applied as urea. The concentration of N and K also increased with an increasing rate of FDE application. Analysis of the soil at the end of the trial indicated that increases in the rate of effluent application had decreased the content of exchangeable Ca and Mg in the soil. This was a consequence of the high K loading and leaching of anions, presumably including nitrate. To minimise this accelerated leaching, the rate and frequency of FDE application should not produce a K loading rate that exceeds the maintenance K requirement of the nitrogen-stimulated, increased pasture production.

### **2.2.5 CONTAMINATION OF SURFACE WATER BY PATHOGENS IN FDE**

FDE contains varying concentrations of faecal micro-organisms which originate from dairy cattle excreta. Dairy cattle are asymptomatic carriers of micro-organisms capable of causing gastroenteritis in humans, including *Giardia*, *Cryptosporidium* and two

thermophilic *Campylobacter* subspecies, *C. jejuni* and *C. coli* that are responsible for almost all human cases of Campylobacteriosis. When raw or partially treated FDE reaches fresh-water bodies, micro-organisms potentially within the FDE have the capability of contaminating the water as a drinking source (Donnison & Ross 2003; Heatley 1996). *Campylobacter jejuni* is the principle bacterial hazard for recreational water users or those drinking untreated water (McBride *et al.* 2002).

Aislabie *et al.* (2001) investigated the movement of bacterial indicators through lysimeters containing four differently structured Waikato soil types. FDE was applied at 50 mm hr<sup>-1</sup> followed by simulated rainfall at 5 - 10 mm hr<sup>-1</sup> until one pore volume of leachate had been collected. The two free-draining soils, (Waihou and Atiamuri) with a uniform porous structure which encouraged matrix flow of water, transported considerably fewer microbes than the coarse-structured, poorly drained soils (Te Kowhai and Netherpton) which exhibited preferential bypass flow of drainage through large macropores. The public health impact of such microbe contamination depends on the destination of the drainage water (ground vs. surface water bodies).

McLeod *et al.* (2003) compared the movement of faecal coliforms with a chemical tracer (bromide) in a free draining and poorly drained soil in the Southland region. A volume of 25 mm FDE spiked with a bacteriophage was applied to soil lysimeters followed by one pore volume of rainfall at 5 mm hr<sup>-1</sup>. It was found that phage movement peaked early in the flow (approximately 0.15 pore volume) and then tailed off indicating bypass flow through larger macropores. The bromide tracer moved more uniformly through the soil, entering more of the micropore system profile (matrix flow), and the concentration peaked at approximately 0.5-0.8 pore volume. The results indicated that the rapid transmission of faecal microbes to surface or ground waters could occur when FDE was applied to soils with the potential for larger amounts of bypass flow such as those with installed mole and pipe drainage.

More recently, Donnison & Ross (2003) have reported the occurrence of *Campylobacter jejuni* in water exiting mole and pipe drainage system soils in West Otago. It was found that on the occasions when preferential flow (also termed bypass flow) of raw, spray-irrigated, FDE occurred through the mole and pipe network that the concentrations of campylobacter were similar to those in the applied effluent (10<sup>5</sup> campylobacter 100 ml<sup>-1</sup>).

## 2.2.6 NUTRIENT LEACHING LOSSES RESULTING FROM APPLICATION OF FDE

For a land treatment system to be sustainable it must be efficient in both the storage of effluent in the soil and the subsequent uptake of plant nutrients applied in the effluent. The longer the effluent resides in the soil's active root zone, the greater the opportunity for the soil to physically filter the effluent whilst absorbing nutrients and making them available to plants. Most of the research related to nutrient dynamics under land treatment of FDE has focused on quantifying aspects of nutrient movement, in particular leaching losses. Such losses can be measured in the direct drainage of untreated or partially treated effluent, immediately following irrigation events and/or in the drainage that occurs in the succeeding winter. The studies are reported below.

Research conducted by Macgregor *et al.* (1979) on a mole and tile drained Tokomaru silt loam at Massey University's No. 4 Dairy farm found that approximately 90% of N and 98% of P from partially treated FDE was retained by the soil system. However, nutrient losses were prevalent at times when soil moisture conditions permitted the rapid movement of effluent through the subsurface drainage. The permanent pasture site was spray irrigated with 20 – 30 mm of pond-treated FDE every 10 – 15 days throughout the lactation season. The authors found that when cumulative evaporation usually exceeded cumulative rainfall, no drainage resulted from spray irrigations of FDE. When the soil was close to or at field capacity, up to 30% of the applied effluent volume drained through the mole and pipe network. Annual nutrient inputs of 1125 kg N ha<sup>-1</sup> and 125 kg P ha<sup>-1</sup> were added as FDE. The measured annual nutrient losses were a total of 150 kg N ha<sup>-1</sup> and 1.6 kg P ha<sup>-1</sup>. Of the nutrients lost, 90 kg N ha<sup>-1</sup> yr<sup>-1</sup> and 0.6 kg P ha<sup>-1</sup> yr<sup>-1</sup> exited in winter drainage. Drainage losses from a grazed pasture system without added FDE were 30 kg N ha<sup>-1</sup> yr<sup>-1</sup> and 0.1 kg P ha<sup>-1</sup> yr<sup>-1</sup>.

At the same research site, Cooke *et al.* (1979) found that the quantity of effluent preferentially exiting the soil as drainage was strongly related to soil and climatic conditions. Under saturated soil moisture conditions, when effluent irrigations coincided with natural rainfall (47 mm), they measured 78 mm of drainage following an application of 30 mm of FDE. Under summer conditions, 18 mm of drainage was recovered from an application of 24 mm of FDE when there was a soil water deficit of 6 mm. A total of three irrigation events were intensively monitored and nutrient inputs and drainage losses characterised. Analyses of nutrient loss in the drainage found that the soil was more efficient at removing particulate forms than dissolved forms. The presence of strongly sorbed anions such as P in drainage provided evidence that some

untreated FDE drained via preferential flow pathways. Approximately 20% of both the N and P spray irrigated onto land was leached through the mole and tile drainage system during the drainage events.

Further research investigating an improved method for the land treatment of FDE (deferred irrigation, application only when a suitable soil water deficit exists) was also established on a mole and pipe drained Tokomaru silt loam at the Number 4 Dairy farm of Massey University (Houlbrooke *et al.* 2003). Winter drainage following the summer applications of FDE using the deferred irrigation criteria resulted in no increase in leaching losses of  $\text{NO}_3^-$ -N (Figure 2.1). There was a  $3.3 \text{ kg N ha}^{-1}$  increase in total N lost in winter drainage from soils receiving summer applications of FDE. This represented 3.5% of the N applied in effluent. FDE application also resulted in greater concentrations of dissolved inorganic P being leached throughout the winter drainage period (Figure 2.2). However the annual loss of total P ( $0.52 \text{ kg ha}^{-1}$ ) equated to approximately only 3% of the P applied in summer applications of effluent.

Longhurst *et al.* (1999) applied raw FDE at six loading rates of N (ranging from 0 to  $375 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ ) to un-grazed pastures on a well-drained Horotiu silt loam at the No. 1 Dairy Unit at Ruakura, Hamilton. FDE was applied 17 times, at 4-7-day intervals, from January to April of 1997. Pasture yield over a year was found to have increased by up to 24% above the control. The efficiency of N use decreased with increasing FDE loading rates. At  $75 \text{ kg N ha}^{-1}$ , equivalent N recovery by plant uptake was 85%, however, at the  $375 \text{ kg N ha}^{-1}$  rate the rate of N recovery was only 40%. Nitrate-N leachate was collected using ceramic cup collectors during the subsequent winter drainage period (5th June to 3rd October), which was dryer than average (only 80% of the expected annual 1200 mm rainfall for 1997). The average concentration of leachate in winter drainage from the  $375 \text{ kg N ha}^{-1}$  treatment was  $0.76 \text{ mg NO}_3^- \text{-N litre}^{-1}$  vs.  $0.42 \text{ mg NO}_3^- \text{-N litre}^{-1}$  from the control, with a calculated  $\text{NO}_3^-$ -N loss of  $2.1 \text{ kg N ha}^{-1}$  and  $1.2 \text{ kg N ha}^{-1}$ . This would appear to be very low and is partly attributable to the absence of any cattle grazing effects eliminating the leaching from urine patches and the low volume of winter drainage.

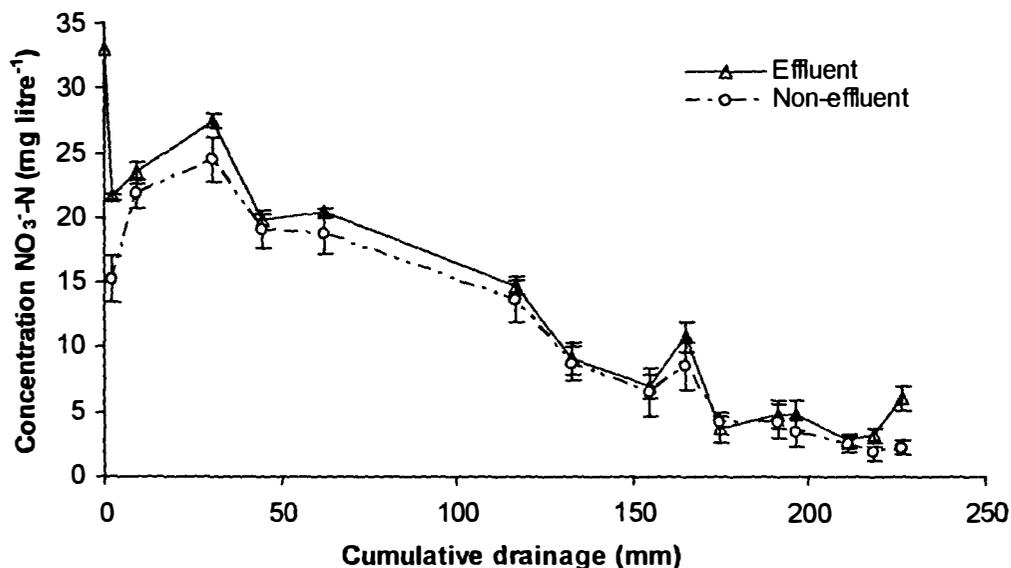


Figure 2.1. The trends in drainage  $\text{NO}_3^-$ -N concentration with total amount of accumulated drainage, in both the non-effluent and effluent plots. Error bars represent two SEM (Houlbrooke et al. 2003).

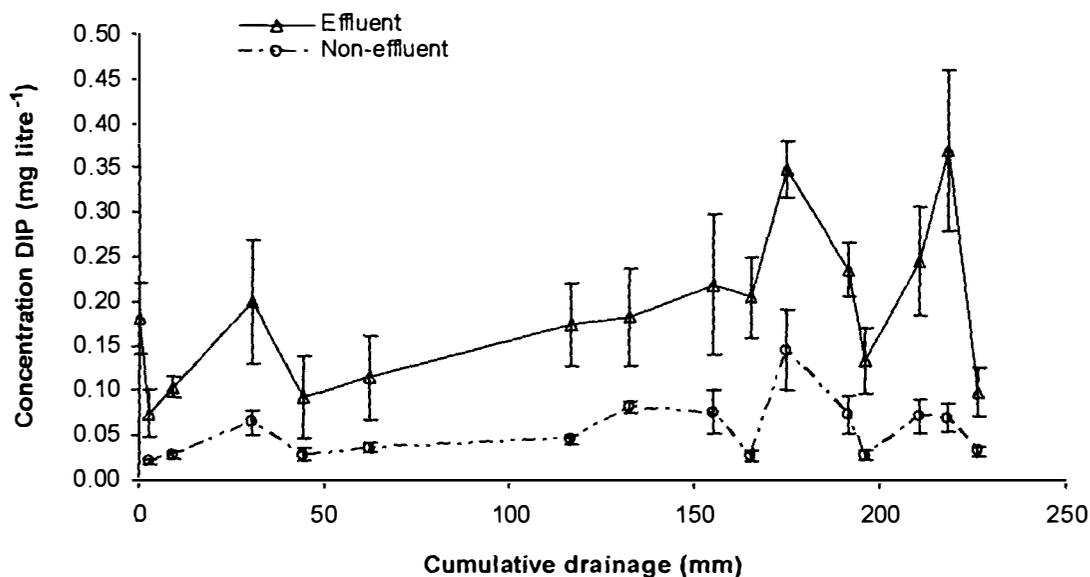


Figure 2.2. The impacts of effluent irrigation on dissolved inorganic P concentrations. Error bars represent two SEM (Houlbrooke et al. 2003).

A laboratory incubation was undertaken by Barkle *et al.* (2001) using Te Kowhai silt loam to investigate soil immobilisation and mineralisation of N from a standard (302 mg N litre<sup>-1</sup>) and high (1011 mg N litre<sup>-1</sup>) rate of FDE application. The high application concentration was achieved by the addition of extra faeces and urine to ensure that substantial microbial growth resulted. At the standard rate of FDE addition, net N

immobilisation persisted throughout the period of the experiment; however, at the high rate of FDE application, net N mineralisation occurred after day 113.

Barkle *et al.* (2000) used lysimeters to investigate the rate at which soil organic matter was mineralised from raw FDE applied to an un-grazed Te Kowhai silt loam soil. The authors suggest that due to increased N mineralisation from the accumulating soil organic matter, N fertilisation should be decreased on FDE application sites. It was found that FDE application to land could only be sustained in terms of N leaching when the supply of inorganic N was continually matched by pasture uptake. However, a constraint to substituting FDE N for fertiliser N is that increased available N is most needed in winter and early spring when climate conditions do not favour rapid N mineralisation. At this time N-boosted pasture growth is required to meet increasing feed requirements at the start of lactation. With FDE application, although the organic-N pool is increased, the larger pool may not increase N availability due to lack of mineralisation during the critical time for pasture requirements.

Roach *et al.* (2001) applied urea and pond-treated FDE to an Egmont brown loam at rates of 100, 200 and 400 kg N ha<sup>-1</sup> yr<sup>-1</sup> to grazed dairy pastures near Hawera, in the Taranaki region. Eight monthly applications were made from September to April. NO<sub>3</sub><sup>-</sup>-N leaching from any direct FDE drainage and winter drainage was measured using ceramic cup samplers in conjunction with lysimeters and showed that the three treatments had average NO<sub>3</sub><sup>-</sup>-N concentrations of 7, 8 and 20 mg N litre<sup>-1</sup> which resulted in annual N losses of 18, 20 and 50 kg N ha<sup>-1</sup> yr<sup>-1</sup>.

A lysimeter study was established by Singleton *et al.* (2001) on a Te Kowhai silt loam soil on the No.1 Dairy Unit at Ruakura, Hamilton, to investigate N leaching of FDE when drainage is managed by controlling the water table. The treatments used were raw FDE and water. The lysimeters were irrigated weekly with raw FDE at a very low instantaneous application rate of 4 mm h<sup>-1</sup> (less than the saturated hydraulic conductivity) to minimise the loss of raw effluent via preferential flow. A weekly total of 17 mm of FDE was applied which corresponded to approximately half the water-holding capacity of the top 20 cm of the soil profile. Drainage was managed using a weir to maintain the water table at depths of 25, 50 or 75 cm below the soil surface. In the first season, a total of 511 kg N ha<sup>-1</sup> yr<sup>-1</sup> was applied as FDE, and leaching losses of approximately 33 kg N ha<sup>-1</sup> were measured. In the medium (50 cm) and high (25 cm) water table treatments, these losses under FDE application were approximately double those receiving the equivalent amount of water only. In the second season, the loading

rate of FDE was increased to  $1518 \text{ kg N ha}^{-1} \text{ yr}^{-1}$  and a total of  $131 \text{ kg N ha}^{-1}$  was leached from the soil, approximately  $100 \text{ kg}$  more than the equivalent water only treatment. The high water table (25 cm below soil surface) treatment resulted in less N being leached in a  $\text{NO}_3^-$ -N form (but a higher organic-N loss) than the lower water table treatments; this was believed to be attributable to the likely enhanced denitrification resulting from the higher water table.

A number of lysimeter studies have been reported from Lincoln University on free draining soils looking at the impacts of dairy farm practices (including the applications of FDE and urea) on  $\text{NO}_3^-$ -N and  $\text{NH}_4^+$ -N leaching losses. Di *et al.* (1998) applied a total of  $400 \text{ kg N}$  as raw FDE by either flood irrigation or spray irrigation over two split irrigations on a Templeton fine sandy loam soil. Throughout the length of the trial (December 1995 to March 1997) a total of  $799$  and  $359 \text{ mm}$  of drainage was collected from the flood irrigation and spray irrigation treatments. After the first application, neither treatment had concentrations of  $\text{NO}_3^-$ -N in drainage any higher than the control ( $< 1 \text{ mg N litre}^{-1}$ ). It was suggested that low  $\text{NO}_3^-$ -N concentrations in the summer should be expected as a result of microbial immobilisation of mineral N, denitrification during irrigation and plant uptake. Raw FDE has only a very low  $\text{NO}_3^-$ -N concentration and FDE moving through soil macro pore systems and recovered as drainage water will carry little  $\text{NO}_3^-$ -N despite having a large total-N concentration. Following the second FDE application (May 1996), the nitrate-N concentration in drainage from the spray irrigation and flood irrigation treatments increased to  $17$  and  $10 \text{ mg N litre}^{-1}$ . The lower concentration measured in drainage from the flood irrigation treatment was attributed to an increased denitrification loss occurring after this method of applying FDE.  $\text{NO}_3^-$ -N losses at the  $400 \text{ kg N ha}^{-1} \text{ yr}^{-1}$  application rate were lower for FDE than from N fertiliser, indicating less mineralisation and nitrification of FDE-N.

Silva *et al.* (1999) applied raw FDE at  $200$  or  $400 \text{ kg N ha}^{-1} \text{ yr}^{-1}$  in four split applications to a Templeton fine sandy loam soil either with or without  $1000 \text{ kg N ha}^{-1}$  of added urine (believed to be the approximate loading of N in a urine patch). A total of  $1328 \text{ mm}$  of water was applied as either rainfall or irrigation and resulted in the collection of  $410 \text{ mm}$  of drainage. Without urine, the total  $\text{NO}_3^-$ -N leaching loss was  $6.3$  and  $10 \text{ kg N ha}^{-1}$  from the  $200$  and  $400 \text{ kg N ha}^{-1} \text{ yr}^{-1}$  treatments. This represented a  $2.5$  to  $3.2\%$  annual loss of the N originally applied as FDE. When urine was applied to the lysimeters, the calculated paddock loss taking into account the likely relative proportion of urine and non-urine patches was  $46.5 \text{ kg N ha}^{-1}$  as  $\text{NO}_3^-$ -N from the  $200 \text{ kg N ha}^{-1}$  treatment ( $14\%$

of the N applied as urine plus FDE) whilst the 400 kg N ha<sup>-1</sup> treatment had a lower loss of 35.8 kg N ha<sup>-1</sup> (10% of the total N applied as both urine and FDE).

Di & Cameron (2002) also flood irrigated a Lismore stony silt loam soil with raw FDE at total loading rates of 200 and 400 kg N ha<sup>-1</sup> yr<sup>-1</sup> over four split applications from April 1998 to March 2000. A total water input (irrigations and rainfall) of 1400 mm for year one and 1700 mm for year two resulted in 610 and 880 mm of drainage. Peak drainage concentrations of NO<sub>3</sub><sup>-</sup>-N following applications of FDE were 21.3 and 34.2 mg N litre<sup>-1</sup> for the 200 and 400 kg N ha<sup>-1</sup> yr<sup>-1</sup> treatments. The annual loss of NO<sub>3</sub><sup>-</sup>-N was considerably different between the two years that the experiment was conducted. Year one had losses of 55 and 78.3 kg N ha<sup>-1</sup> for the 200 and 400 kg N treatments, whilst year two had much smaller losses of 7.6 and 18.7 kg N ha<sup>-1</sup> for the 200 and 400 kg N treatments. These losses represent approximately 10% for year one and 2.5% for year two of the total N originally applied as FDE. Interestingly, smaller NO<sub>3</sub><sup>-</sup>-N losses were measured in the second year despite the fact that year two had a greater cumulative drainage of water than year one. The difference in NO<sub>3</sub><sup>-</sup>-N losses between years was explained by the effects of initial pasture renovation at the beginning of the trial causing a release of mineral-N through the mineralisation of organic-N in decaying plants.

Cameron *et al.* (1999) reviewed a number of different effluent research trials completed at Lincoln University and concluded that rate at which FDE is applied to land influences the quantity and therefore risk of N leaching. Results were reported showing peak drainage water concentrations of 10 to 18 mg N litre<sup>-1</sup> (depending on irrigation conditions) when FDE was applied in two split applications of 200 kg N ha<sup>-1</sup> yr<sup>-1</sup>. The peak concentration was however decreased to 4 mg N litre<sup>-1</sup> when the FDE was applied in four split applications of 100 kg N ha<sup>-1</sup> yr<sup>-1</sup>.

An investigation of the status of soil physical and chemical properties following 6 years of raw FDE applications was reported from a Wairarapa dairy farm under long term dairy pasture by Hawke and Summers (2003). No changes in soil physical properties (saturated hydraulic conductivity, porosity and bulk density) were found. A significant increase in total Kjeldahl-N and cations (K<sup>+</sup>, Na<sup>2+</sup>, Ca<sup>2+</sup>, Mg<sup>2+</sup>) was measured in the top 10 cm of soil. Olsen P levels increased from 56 to 76 µg P g<sup>-1</sup> in a soil that has a low P retention. It was deemed that pasture nutrient requirements could be sustained using FDE. It was, however, noted that the C:N ratio had dropped from 28:1 to 20:1, and that the change in soil N dynamics may have long-term implications for mineral N leaching.

Roygard *et al.* (2001) investigated the effectiveness of several different tree species at removing mineral-N from soils that were receiving applications of pond-treated FDE. A bare ground treatment was used as a control. The research involved a lysimeter study using a Manawatu fine sandy loam soil receiving weekly applications of FDE at a rate of 21.5 mm per application throughout the entire lactation season. The total quantity of N applied in the FDE over the season was 236 kg N ha<sup>-1</sup>. Average NO<sub>3</sub><sup>-</sup>-N concentrations of leachate collected from the lysimeter were higher than the New Zealand drinking water standard (11.3 mg N litre<sup>-1</sup>) for all treatments. However, NO<sub>3</sub><sup>-</sup>-N concentration in the drainage from two of the tree species (*Eucalyptus nitens* and *Salix kinuyangi*) (26 and 19 mg N litre<sup>-1</sup> respectively) was significantly lower than the bare soil treatment (40 mg N litre<sup>-1</sup>). It was found that tree species varied in evapotranspiration rates and their ability to take up N. The deciduous tree species *S. kinuyangi* maintained the lowest N leachate concentration prior to harvest. The lysimeters restricted the root zone laterally but not the canopy, so the projected canopy area exceeded the soil surface area for the root zone and subsequently affected the drainage rates.

### 2.2.7 IMPLICATIONS FOR WATER QUALITY

The consensus view of the research cited above is that land treatment of FDE to trap nutrients in soil is relatively effective, with only 2 - 20% of the nutrients applied in the FDE being transported to fresh water bodies either as immediate drainage or as nutrient leaching from FDE-applied soils. This represents a considerable reduction of the quantity of nutrients that discharge from a two-pond system direct into receiving fresh water bodies. However, in many cases, particularly those with very high loading rates of FDE where there was an increased likelihood of preferential flow of partially treated FDE, the concentration of NO<sub>3</sub><sup>-</sup>-N being leached was above the New Zealand drinking water standard (11.3 mg N litre<sup>-1</sup>). This has particular implications for FDE application to free-draining soils whose drainage waters recharge regional aquifer systems that commonly provide drinking water to rural communities. The health risk posed by drinking water that is high in NO<sub>3</sub><sup>-</sup>-N is greater for babies and young children who may be prone to the blood disorder called methaemoglobinaemia. The result of such a disorder is a reduction of the oxygen-carrying capacity of the blood stream; hence the condition is often known as 'blue baby syndrome'. The health risk of drinking microbial contaminated water can be more severe on human health than that

of water high in  $\text{NO}_3^-$ -N potentially causing severe gastroenteritis. In particular this can be an issue in rural water supplies which are commonly untreated.

In all the reported studies, the concentrations of N and P (if measured) were higher than the levels ( $0.1 \text{ mg N litre}^{-1}$  and  $0.1 \text{ mg P litre}^{-1}$ ) considered likely to promote aquatic weed growth (Ministry for the Environment 1992). Such high levels of N and P are likely to encourage algal blooms and aquatic plant growth that deplete oxygen levels in the water, resulting in a decrease in biodiversity in the stream (Bond 1998; Cameron & Trenouth 1999).

The Resource Management Act (1991) states that effluent application to land should not have any adverse effect on aquatic life (Parminter 1995; Cameron & Trenouth 1999). It appears that standard dairy farming practices of cattle grazing (leaching from urine patches) and fertiliser applications is more responsible (particularly at a farm scale) than is land application of FDE for elevated levels of nutrients leached into groundwater systems, streams and lakes (Di & Cameron 2002; Silva *et al.* 1999; Monaghan *et al.* 2002a; Houlbrooke *et al.* 2003). Of most importance is the careful management of land treatment systems to prevent the direct discharge of raw or only partially treated effluents.

### 2.2.8 CONSIDERATIONS FOR IMPROVED FDE LAND APPLICATION

A number of areas can be identified where there are shortcomings with the current management of FDE application to land. Goold (1980), Roach *et al.* (2001), and Bolan *et al.* (2004) reported high levels of potassium (K) in both soil and pastures following long-term applications of FDE to land. Longhurst *et al.* (2000) reported, in a survey of FDE characteristics, that the concentration of K was usually 80% of the concentration of N. The typical concentration range of N and K found in ryegrass pasture is 2-4% and 1-3% (McLaren & Cameron 1996). Nitrogen is commonly used to determine the maximum nutrient loading to land (often  $150 - 200 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ ). At such rates K will be applied to the soil at loadings of  $120-160 \text{ kg ha}^{-1} \text{ yr}^{-1}$ , which is far in excess of the  $50 - 90 \text{ kg K ha}^{-1} \text{ yr}^{-1}$  required for maintenance of K reserves in the soil. When K is applied to soil at far greater than maintenance requirements, the resulting high K levels in pasture and decrease in calcium and magnesium can induce metabolic disorders in dairy cows, particularly milk fever (hypocalcaemia) and ryegrass staggers (hypomagnesaemia) (Roach *et al.* 2001; Tillman & Surapaneni 2002).

Soil physical characteristics have a considerable impact on the effectiveness of any land treatment system for FDE as the continuity of macropores can have a considerable effect on the preferential flow of water, particularly in soils with mole and pipe drainage. Preferential flow of water (and nutrients) occurs through natural soil macropores such as continuous cracks and earth-worm channels (McLay *et al.* 1991). Furthermore, when mole and pipe drainage has been installed to ameliorate poor drainage, large macropores are created above the ripped mole channel which promotes drainage by by-pass flow. Drainage by-passes a great deal of the soil matrix, exiting the mole system through the pipe drain (Kladivko *et al.* 1991; Kohler *et al.* 2001).

The soil moisture status of the land receiving application must be considered because during periods when rainfall exceeds evapotranspiration (usually May to October) soil water is at, or close to, field capacity and water may move freely through the soil profile. As both Cooke *et al.* (1979) and Macgregor *et al.* (1979) reported, applications of FDE at times of low soil moisture deficit resulted in large quantities of drainage and hence nutrient losses.

The large range of nutrient losses reported for applications of FDE has less to do with the initial nutrient content of the effluent (pond versus sump) and is likely to be related to the quantity of effluent applied, the management of effluent, and soil moisture condition at the time of effluent application (Table 2.1). Many of the research trials reported above applied effluent in a number of split applications across the drier summer months, hence minimising the direct losses of effluent. No trials have reported the common practice of irrigating FDE daily throughout the lactation season (August - May) from a small wash down sump at the farm dairy. Such a practice inevitably involves applying FDE to the soil at times when there is little or no soil moisture deficit, such as early spring and late autumn. The range of percentage losses reported suggests that management of effluent, particularly with regard to the soil moisture status at the time of application, may play a large part in determining the likelihood of nutrients from FDE directly draining, or being retained in the soil profile. In many cases the research reported has applied FDE at extremely high rates of N application, far in excess of the 150-200 kg N ha<sup>-1</sup> commonly stipulated by regional councils in New Zealand. It could be argued that there is a need for more research to be undertaken under more realistic farm conditions for the application of FDE.

Recent research by both Monaghan & Smith (2004) and Houlbrooke *et al.* (2004) has investigated improved methods for land applying effluent to artificially drained soils. FDE should not be applied to soil based simply on its theoretical ability to absorb nitrogen. Effluent applications should be made with regard to current soil moisture status and soil physical characteristics such as soil structure and hydraulic properties. Both research trials found that applications must be applied only at times of suitable soil moisture deficit to avoid the preferential flow of applied FDE and hence the direct drainage of partially treated effluent that is likely to have a significant adverse impact on the receiving water bodies. Adequate pond storage is required to hold FDE at times during the lactation season such as early spring when the soil is commonly close to or at field capacity with low soil water deficit. Such a system has been called 'deferred irrigation' by Houlbrooke *et al.* (2004) and has been successful in minimising nutrient losses as a result of the direct drainage of partially treated FDE and in many cases has eliminated them altogether.

**Table 2.1.** *The amount of N loss from a range of different applications rates and forms of farm dairy effluent.*

Research	FDE source	FDE rate (kg N ha <sup>-1</sup> yr <sup>-1</sup> )	Application criteria	% applied N lost
Macgregor <i>et al.</i> (1979) <sup>a</sup>	Pond	1125	20-30 mm every 10-15 d	10
Di <i>et al.</i> (1998) <sup>b</sup>	Raw	400	2 x 200 kg N ha <sup>-1</sup>	7
Silva <i>et al.</i> (1999) <sup>b</sup>	Raw	200	4 x 50 kg N ha <sup>-1</sup>	2.5
Silva <i>et al.</i> (1999) <sup>b</sup>	Raw	400	4 x 100 kg N ha <sup>-1</sup>	3.2
Roach <i>et al.</i> (2001) <sup>c</sup>	Pond	100	monthly at 12.5 kg N ha <sup>-1</sup>	18
Roach <i>et al.</i> (2001) <sup>c</sup>	Pond	200	monthly at 25 kg N ha <sup>-1</sup>	10
Roach <i>et al.</i> (2001) <sup>c</sup>	Pond	400	monthly at 50 kg N ha <sup>-1</sup>	12.5
Singleton <i>et al.</i> (2001) <sup>d</sup>	Raw	511	17 mm applied weekly	7
Singleton <i>et al.</i> (2001) <sup>d</sup>	Raw	1518	17 mm applied weekly	9
Houlbrooke <i>et al.</i> (2003) <sup>a</sup>	Pond	95	7 applications of ~ 9 mm	3.5

<sup>a</sup> Tokomaru silt loam

<sup>b</sup> Templeton fine sandy loam

<sup>c</sup> Egmont brown loam

<sup>d</sup> Te Kowhai silt loam

It has been identified that there was a lack of research in New Zealand carried out under realistic farm conditions, in particular with regards to the sometimes excessive N and P loadings used. Furthermore it could be argued that there has been a lack of research at a farm scale and even to some extent at a paddock scale from dedicated

long-term effluent blocks within a farm dairy. There is a need for more data to be collected at the farm scale from such blocks on the impacts of long term irrigation of FDE on nutrient leaching and pasture quality and its potential associated animal health impacts. To date there has been a lack of research published on the extra difficulties associated with land treating poorly drained soils that have had artificial drainage such as mole and pipe installed, although there has been some recent research on this carried out in both the Manawatu and Otago regions. Improved strategies are required to assist farmers to land treat FDE in a sustainable manner.

### **2.2.9 SUMMARY**

Traditionally FDE has been treated in standard two-pond systems and then discharged into receiving fresh water streams. Changes brought about primarily due to the RMA of 1991 meant that most regional councils now prefer the land treatment of the FDE. Conditions under the resource consent process often stipulate maximum N loading rates and hydraulic application rates and an effects based clause. FDE is often applied to land by way of small travelling irrigators and is utilised by the soil-plant system for its water and nutrients. Historical research on the effects of land treating FDE on water quality showed that between 2 – 20% of the N and P applied as FDE is directly leached through the soil profile, therefore entering receiving fresh water bodies. In all cases, the concentrations of leachate in drainage waters were higher than the ecological limits considered likely to stimulate unwanted aquatic weed growth, however this is also true of non-irrigated, grazed dairy pastures.

Whilst the land treatment of FDE represents a huge improvement on the loss of nutrients discharged to fresh water from standard two-pond systems, there is still room for improvement in the management of FDE land treatment systems. It has been identified that the applications of FDE at times of zero, or very small, soil water deficit, such as those moisture conditions found in early and late lactation are more likely to result in the direct drainage of only partially treated effluent. Furthermore, free-draining soils and soils with artificial drainage may be prone to the rapid movement of applied effluent by the process of preferential flow through large macropores. Improvement in the management of land applications can be made to prevent the direct discharge of partially treated FDE by taking into account soil physical properties and soil moisture status and by having the capability to store FDE for extended periods when soil moisture conditions are unsuitable for application.

### 2.2.10 ACKNOWLEDGEMENTS

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## 2.3 THE IMPACT OF INTENSIVE DAIRY FARMING ON NUTRIENT LOSSES IN DRAINAGE WATER FROM PASTORAL LAND

### 2.3.1 Introduction

As reported in section 2.2, New Zealand has undergone considerable expansion in intensive dairy farming over the past ten years with a 44% increase in national dairy cow numbers (Livestock Improvement 2003). A number of researchers have found that drainage and surface runoff water from grazed dairy pasture contribute to the increased nutrient and suspended solid loads in New Zealand waterways (Sharpley & Syers 1979; Ledgard *et al.* 1996, Ledgard *et al.* 1998, Silva *et al.* 1999; Ledgard *et al.* 2000; Di & Cameron 2002; Monaghan *et al.* 2002; Smith and Monaghan 2003; Monaghan *et al.* 2005). Accordingly, in some regions, dairy farming has been criticised for its role in the deterioration of the quality of fresh water bodies and streams. However, despite the concern about the negative environmental impacts of dairy farming, there has been relatively little research conducted to quantify the effects of dairy cattle grazing on the quality of drainage and surface runoff from artificially drained soils. In particular, the losses of P under intensively grazed dairy pasture have been poorly documented as much of the research has concentrated on the losses of  $\text{NO}_3^-$ -N.

Fresh surface water bodies such as lakes and streams may be contaminated by way of artificial drainage systems (Sharpley & Syers 1979; Monaghan *et al.* 2002; Monaghan *et al.* 2005) and surface runoff (Nash and Halliwell 1999, Gillingham and Thorrold 2000; McDowell *et al.* 2003a, Smith and Monaghan 2003). Regional groundwater systems may be subjected to increasing contamination when nutrients leach through free draining soils (Silva *et al.* 1999; McLay *et al.* 2001; Di & Cameron 2002). Furthermore, summer base flows in rivers and streams are usually maintained by discharge from groundwater.

The Australian and New Zealand guidelines for fresh and marine water quality (ANZECC 2000) report that nutrient concentrations in fresh water greater than  $0.61 \text{ mg N litre}^{-1}$  and  $0.033 \text{ mg P litre}^{-1}$  are ecologically significant and considered likely to promote aquatic weed growth. The Ministry for Health (1995) has a recommended drinking water level of  $11.3 \text{ mg litre}^{-1}$  for  $\text{NO}_3^-$ -N.

Research carried out in a Waikato hill country catchment (Quinn & Stroud 2002) showed that N and P losses to a stream from a mixed cattle-sheep grazed pasture were approximately 2.5 - 7 fold greater than those from an adjacent native forest catchment. Furthermore, in the Waikato region, Wilcock *et al.* (1999) reported that a small stream draining a pasture catchment intensively grazed by dairy cows received large quantities of N and P (35.3 and 1.16 kg ha<sup>-1</sup> yr<sup>-1</sup>, respectively). Nitrogen discharge was greater in the winter months and was attributed to NO<sub>3</sub><sup>-</sup>-N leaching whilst P concentrations were greater in the summer months and were attributed to the outflow from farm dairy effluent oxidation ponds. Such data clearly shows a strong link between pastoral agriculture and nutrient enrichment of fresh water bodies. This review aims to further investigate some of the New Zealand and Australian research reporting on the loss of nutrients in drainage water from intensive dairy farms to the aquatic environment.

### 2.3.2 Dairy livestock

#### 2.3.2.1 Losses of nitrogen as a result of grazing dairy cattle

The dung and urine deposited by grazing dairy cattle provide a pool of nutrients that is highly susceptible to leaching in subsequent drainage events. Cattle excreta return up to 90% of the nutrients ingested during pasture consumption (Haynes and Williams 1993). Urine patches, in particular, provide large quantities of highly soluble, neutral urea-N. (Silva *et al.* 1999; Di and Cameron 2002). N is deposited in urine patches at a rate of ~ 1000 kg ha<sup>-1</sup> (Silva *et al.* 1999). Urine patches are found fairly randomly across grazed paddocks and, at a stocking rate of 3 cows ha<sup>-1</sup> and cover ~ 25% of the paddock area in one year (Silva *et al.* 1999). Urine rapidly hydrolyses to the plant available NH<sub>4</sub><sup>+</sup>-N form and then is more slowly converted to NO<sub>3</sub><sup>-</sup>-N, which is prone to leaching, usually within 10 days of the initial addition of cattle urine (Condon *et al.* 2004).

A number of studies have identified urine patches as a significant source of the N in leachate from pastoral grazing systems. Park (1996) found that total organic nitrogen (TON) and NH<sub>4</sub><sup>+</sup>-N were the dominant forms of N in leachate from freshly applied cow urine to lysimeters, suggesting the likelihood of preferential flow of applied urine. High concentrations of NH<sub>4</sub><sup>+</sup>-N were to be expected as urea has an estimated half-life of only 3-4 hours as it is rapidly hydrolysed to NH<sub>4</sub><sup>+</sup>-N once applied to the soil surface (Haynes and Williams, 1993).

Condon *et al.* (2004) partitioned the forms of N in soil, at a range of depths, following the application of a urea solution to intact undisturbed soil lysimeters to simulate cow urination on a coarse sandy loam under long term pasture. On the day of the application, urea was the dominant form of N recovered. Between 1 and 16 days after the application,  $\text{NH}_4^+$ -N was the dominant form recovered, whilst between day 16 and day 32,  $\text{NO}_3^-$ -N was the dominant form recovered. No urea was recovered past day 5 whilst no significant quantity of  $\text{NO}_3^-$ -N was recovered in the first week following urine application.

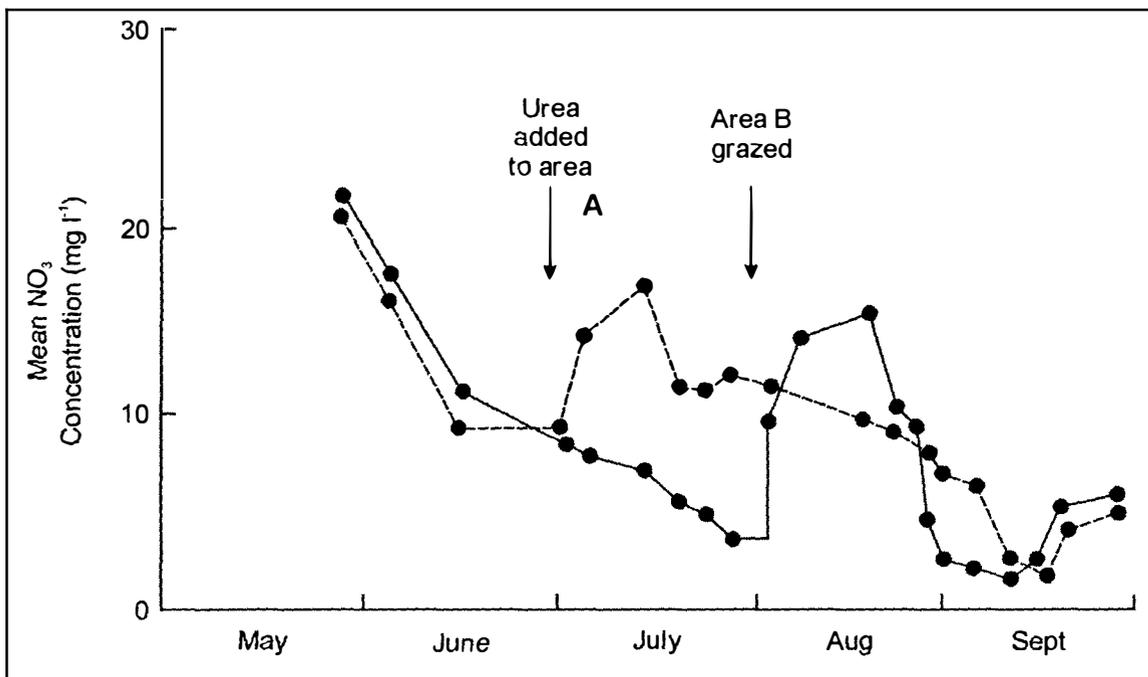
Silva *et al.* (2000), experimenting with intact undisturbed soil lysimeters containing Templeton fine sandy loam, reported that in saturated soil conditions (0 kPa suction), 98% of the N leached below 700 mm from applications of cattle urine was transmitted through macropores greater than 600  $\mu\text{m}$  in diameter. When these large macropores were excluded (0.5 kPa suction), only 0.17% of the N applied in the cattle urine was leached. The main form of N in the leachate from preferential flow in the macropores was  $\text{NH}_4^+$ -N. Urea-N in the leachate from macropores comprised 1.6% of the total N added originally as urine confirming some direct passage of added cow urine via macropore preferential flow.

In South Australia, dairy cattle urine was labelled with  $^{15}\text{N}$  urea and applied to micro-plots and intact undisturbed soil lysimeters containing a sandy loam for an experiment to determine leaching at various depths from irrigated and non-irrigated pasture (Pakrou & Dillon 1995). The irrigated treatment received 25-30 mm of irrigation water every week from December to April. Up to 40% of the applied urine-N was leached below a depth of 150 mm within one day of the urine application to the wetter irrigated paddock. The non-irrigated paddock leached 24% of applied urine in the same time frame. Regardless of the season, all urine-N remaining in the soil was converted to  $\text{NH}_4^+$ -N within one day. During 4 events over one year (one event per season), between 41% and 62% of the applied  $^{15}\text{N}$  was leached from the irrigated site in comparison to 25-51% from the non-irrigated site.

A number of experiments have been conducted at Lincoln University near Christchurch, New Zealand, to examine the leaching losses associated with dairy cow urine application to intact undisturbed soil lysimeters. Silva *et al.* (1999) added the equivalent of 1000 kg N  $\text{ha}^{-1}$  of cow urine to lysimeters containing a free draining Templeton fine sandy loam. A total of 12% of the N applied in urine was leached in c. 400 mm of drainage over one year, predominantly in the  $\text{NO}_3^-$ -N form. As already

mentioned, the authors state that at a stocking rate of 3 cows ha<sup>-1</sup>, urine patches will cover only 25% of a paddock in any one year. On the basis of this area that received urine, they calculated whole paddock losses to be 31 kg ha<sup>-1</sup> yr<sup>-1</sup>. Di and Cameron (2002a) applied 1000 kg N ha<sup>-1</sup> as cow urine to intact undisturbed soil lysimeters containing a free draining Lismore stony silt loam and found that NO<sub>3</sub><sup>-</sup>-N leaching losses comprised 29% of the total N applied in urine during the spring period and between 38-58% during the autumn. As part of the trial, additional applications of urea and farm dairy effluent were made on top of the urine. However, it was the urine patches that were the dominant contributor to NO<sub>3</sub><sup>-</sup>-N loss (c. 75%).

A research trial was established on a mole and pipe drained Pallic soil, the Tokomaru silt loam, near Massey University, Palmerston North by Sharpley & Syers (1979) to assess nutrient losses from a dairy farm. Grazing cattle events at 300 cows ha<sup>-1</sup> for a 12 hour period were found to increase the concentrations of NO<sub>3</sub><sup>-</sup>-N and total-N by 5 fold in subsequent drainage when compared to un-grazed adjacent plots (Figure 2.3). Such results suggest that urine inputs from grazing cattle contribute to the pool of NO<sub>3</sub><sup>-</sup>-N given sufficient time for the processes of urea hydrolysis, mineralisation and nitrification to take place.



**Figure 2.3** Mean NO<sub>3</sub><sup>-</sup>-N concentration in two different mole and pipe drainage plots before and after a cattle grazing and urea application (Area A dotted line; area B solid line). Redrawn from Sharpley and Syers 1979.

Further research was carried out on mole and pipe drained soil by Monaghan *et al.* (2002) on a Pallic Waikoikoi-Arthurton silt loam and Brown Woodlands-Oteramika silt loam mix in Southland to determine nutrient losses in artificial drainage from cattle grazed pasture at a stocking rate of 2.3 cows ha<sup>-1</sup>. An average NO<sub>3</sub><sup>-</sup>-N concentration of 6.9 mg N litre<sup>-1</sup> in 370 mm of drainage resulted in the NO<sub>3</sub><sup>-</sup>-N loss 25 kg N ha<sup>-1</sup> yr<sup>-1</sup>. A further NH<sub>4</sub><sup>+</sup>-N loss of 1.1 kg N ha<sup>-1</sup> yr<sup>-1</sup> was reported (Table 2.2).

**Table 2.2.** Mean concentration and annual loss of different forms of N & P from mole and pipe drained cattle grazed pasture plots on a mixture of Pallic and Brown soil in Eastern Southland. Brackets denote one SEM. (Modified from Monaghan *et al.* 2002)

Nutrient Form	Concentration (mg L <sup>-1</sup> )	Annual loss (kg ha <sup>-1</sup> yr <sup>-1</sup> )
Nitrate-N	6.9	25 (2.4)
Ammonium-N	0.3	1.1 (0.2)
Urea	0.1	0.3 (0.05)
Dissolved reactive P	0.023	0.048 (0.016)
Total dissolved P	0.029	0.059 (0.019)
Total P	0.074	0.152 ( 0.076)

Research by Smith & Monaghan (2003) investigated the impacts of intensive cattle grazing on surface water quality in Southland from 1998-2000. The generation of overland flow was measured as the volume of water travelling over the soil surface. Overland flow losses occurred during the winter-spring period and were greatest within a 1-2 week period of cattle grazing. Overland flow was found to be seven times greater from undrained compared to drained areas. N loss in overland flow was low (0 - 0.38 kg NO<sub>3</sub><sup>-</sup>-N ha<sup>-1</sup> yr<sup>-1</sup> and 0 - 0.32 kg NH<sub>4</sub><sup>+</sup>-N ha<sup>-1</sup> yr<sup>-1</sup>).

Modelling nitrogen leaching losses in the UK, McGechan & Topp (2003) found that the greatest potential for environmental pollution of watercourses from grazing dairy cattle arose from urea transfer to NH<sub>4</sub><sup>+</sup>-N from cow urine through soil macropores during grazing on wet soils. Grazing hotspots, where cows congregate at stocking rates considerably greater than the paddock average, were deemed to leach high levels of NO<sub>3</sub><sup>-</sup>-N in comparison to the rest of the paddock.

A review of N leaching from a range of different agriculture systems by Ledgard and Menneer (2005) reported that nitrate leaching from cattle grazed pasture systems was driven by urine and dung patches and N fertiliser. Cattle urine was identified as the

principle source of leached N however annual N losses were reported to vary in a New Zealand context from 15 -115 kg ha<sup>-1</sup> yr<sup>-1</sup>. The large differences were attributed to differences in farm management (i.e. fertiliser inputs and irrigation), soil type and seasonal and climatic differences (i.e. drainage and plant growth). Previously Ledgard *et al.* (2000) had suggested a whole catchment approach would be required in NO<sub>3</sub><sup>-</sup>-N sensitive areas to decrease NO<sub>3</sub><sup>-</sup>-N concentrations. A range of on farm options were considered including increasing N use efficiency via lower protein feed, limiting external N rich inputs, reducing FDE losses and the use of standoff pads in autumn/winter to avoid direct deposition of cattle excreta to wet soils. Off farm options considered were riparian strips and trenches and in stream removal by aquatic plants.

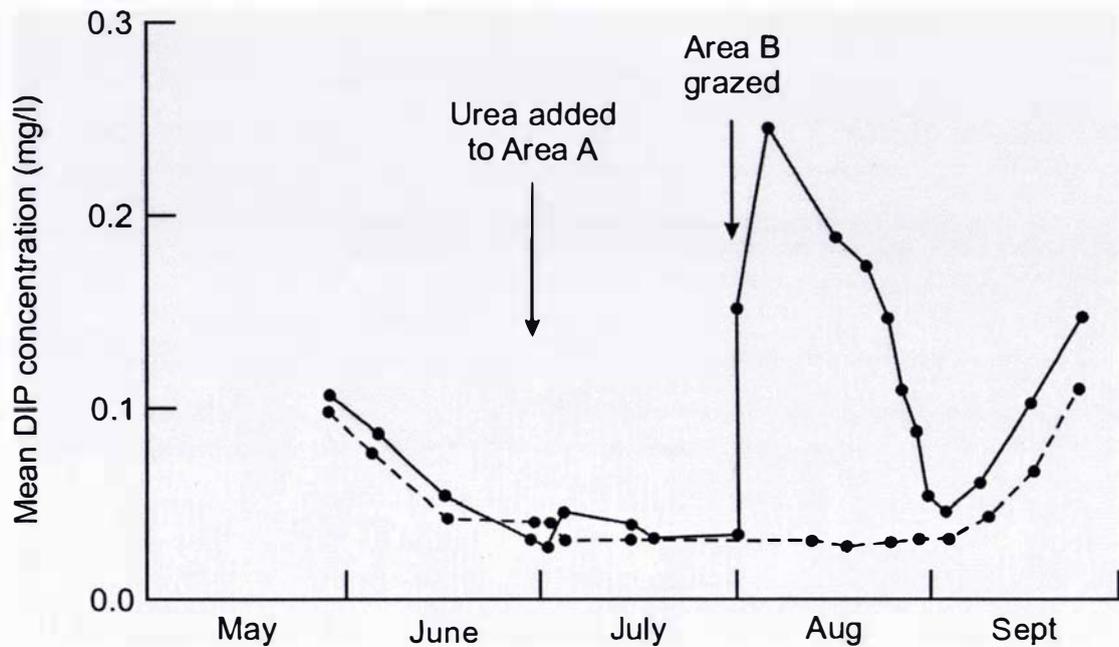
This review of the New Zealand and Australian literature reveals that a number of studies have found that pastoral land grazed with dairy cattle results in high concentrations of N in drainage particularly NO<sub>3</sub><sup>-</sup>-N. The source of this NO<sub>3</sub><sup>-</sup>-N has been linked to the large input of N in cattle urine. Much of the research has been carried out using lysimeters with free draining soils, however, less research has been carried out on artificially drained soils. In particular, mole and pipe networks can promote rapid movement of water and cattle urine via preferential flow paths with limited opportunity for soil treatment and mineralisation processes. Apparently there is currently no research which investigates the simultaneous timing of cattle urination with rainfall induced drainage.

### *2.3.2.2 Losses of phosphorus as a result of grazing dairy cattle*

The addition of cattle dung provides a highly soluble source of P that is susceptible to loss in drainage and runoff (Sharpley and Syers 1979; Smith and Monaghan 2003). Furthermore, treading damage by cattle at the time of grazing disturbs the soil surface resulting in the potential loss of particulate P in sediment transported by surface runoff (Gillingham and Thorrold 2000; McDowell *et al.* 2003a; Smith and Monaghan 2003).

As part of the research carried out on nutrient losses from an artificially drained Pallic soil (low % P retention, McLaren and Cameron 1996) near Palmerston North, Sharpley and Syers (1979) measured the impact of cattle grazing events on P loss. Dramatic increases in the concentrations of dissolved inorganic P (DIP) and particulate P (PP) were measured in the drainage following a single grazing event with 15 and 40 fold increases, respectively (Figure 2.4). The greater loss of PP in drainage following grazing was associated with a 50% increase in sediment load as a result of surface

pugging. Drainage concentrations of DIP and PP returned to pre-grazing levels approximately 4 weeks following grazing.



**Figure 2.4** Mean DIP concentration in two different mole and pipe drainage plots before and after a cattle grazing and urea application (Area A dotted line; area B solid line). Redrawn from Sharpley and Syers (1979).

Further research on P loss under cattle grazed ( $2.3 \text{ cows ha}^{-1}$ ) mole and pipe drained plots in Southland by Monaghan *et al.* (2002) on a Pallic Waikoikoi-Arthurton silt loam (low % P retention, McLaren and Cameron 1996) and Brown Woodlands-Oteramika silt loam (medium to high % P retention, McLaren and Cameron 1996) mix showed that dissolved reactive P concentrations were within the critical limit of  $0.015\text{-}0.03 \text{ mg L}^{-1}$  thought to promote nuisance weed growth in aquatic waters (Ministry for the Environment 1992). The mean annual loss of  $0.152 \text{ kg P ha}^{-1} \text{ yr}^{-1}$  reported was very small in comparison to the  $77 \text{ kg P ha}^{-1}$  applied to the research site as fertiliser in the preceding growing season (Table 3.2). Further results from the same field site are discussed in Monaghan *et al.* (2005) which imposed four N fertiliser treatments and associated cattle stocking rates. Increasing cattle stocking density did not necessarily increase mean total P or dissolved reactive P losses over three seasons (Table 2.3).

Research has been carried out by Smith & Monaghan (2003) at the same research site as above (Pallic Waikoikoi-Arthurton silt loam and Brown Woodlands-Oteramika silt loam mix) on P loss in overland flow from a cattle grazed pasture. As with their N data discussed above, P loss was highest in overland flow within the 1-2 week period

following a spring grazing as a result of added excreta and treading damage. An annual mean P loss from un-drained plots receiving 400 kg N and 45 kg P fertiliser per year was 0.23 kg P ha<sup>-1</sup> yr<sup>-1</sup> which was considerably greater than the 0.03 kg P ha<sup>-1</sup> yr<sup>-1</sup> lost from drained plots receiving the same fertiliser treatment. The authors argue that management strategies should be put in place to reduce soil treading damage on poorly drained soils in the spring.

**Table 2.3.** Annual loss (kg ha<sup>-1</sup> yr<sup>-1</sup>) of total P (TP) and dissolved reactive P (DRP from mole and pipe drained cattle grazed pasture plots on a mixture of Pallic and Brown soil in Eastern Southland. (Modified from Monaghan *et al.* 2005)

Treatment	Stocking density (cows ha <sup>-1</sup> )	DRP loss	TP loss
0 kg (control)	2.4	0.11	0.34
100 kg N ha <sup>-1</sup> yr <sup>-1</sup>	2.8	0.05	0.16
200 kg N ha <sup>-1</sup> yr <sup>-1</sup>	3.0	0.11	0.43
400 kg N ha <sup>-1</sup> yr <sup>-1</sup>	3.3	0.09	0.26
SED		0.05	0.18

A one year study by McDowell *et al.* (2003a) investigated phosphorus and sediment loss in overland flow and changes to soil physical conditions in plots that were cultivated + grazed, ungrazed cultivated and ungrazed pasture. The soil type was a Pallic Waitahuna silt loam (low % P retention, McLaren and Cameron 1996). Following grazing in cultivated soil, there was only a 25% increase in sediment loss, however, there was a >250% increase in total P loss in overland flow. In comparison, P loss in the cultivated ungrazed treatment decreased in subsequent overland flow events. McDowell *et al.* (2003b) investigated sediment and phosphorus loss via overland flow following simulated rainfall on cattle grazed pasture. Treading decreased macroporosity and hence time to ponding and so increased the volume of expected overland flow. Subsequently, sediment and P loss increased following grazing by dairy cattle.

The loss of P in drainage and surface runoff water is clearly an environmental problem. While the loss of particulate P associated with increased suspended solids following soil disturbance during cattle grazing events is well understood, the high concentrations of dissolved P measured in mole and tile drainage water following cattle grazing deserves further investigation.

### 2.3.2.3 Land use intensity

Given the impact that grazing cattle events have on the potential for nutrient losses via drainage and surface runoff, it could be argued that changing land use intensity is an option to reduce such losses. De Klein and Ledgard (2001) compared  $\text{NO}_3^-$ -N leaching losses and total N losses to the environment from three scenarios: nil grazing, restricted grazing and conventional grazing. The analysis was based on measured data, available literature and use of the Overseer® model. The nil grazing scenario assumed that no grazing took place on paddocks i.e. all of the pasture was cut and fed to cattle on a feed pad and all animal excreta was collected and returned evenly to the pasture as FDE. Under these circumstances, the nil grazing scenario decreased  $\text{NO}_3^-$ -N leaching losses by 55-65% compared with conventional grazing. However, the total loss of N to the environment was 35% higher as a result of greatly increased gaseous N losses that occurred due to the storage of dairy effluent. The restricted grazing scenario involved grazing cattle on pasture from September to March and then feeding 'cut and carry' pasture on a feed pad from April to August. Under these circumstances, there was no change in total N losses to the environment but  $\text{NO}_3^-$ -N leaching losses were decreased by 35-50% in comparison to a conventional grazing scenario.

A further analysis of pasture production and a cost benefit analysis from the same comparison of nil grazing, restricted grazing and conventional grazing as discussed above was carried out by De Klein (2001). A nil grazing scenario was deemed to increase pasture production by 18% as a result of lower leaching losses and the elimination of cattle treading damage. However, such a scenario was considered likely to suppress clover growth as clover grows in areas of lower N fertility such as those areas between urine and dung patches. The restricted grazing scenario had an increase in pasture production of 2-8%. The capital and operational costs for the nil grazing scenario were greater than the expected return from the increased production, and resulted in a return on capital of -8 to -10%. For the restricted grazing system, the return on capital was 5-9% or 17% if the farm had an existing effluent application system. Hence, a restricted grazing system was deemed to decrease  $\text{NO}_3^-$ -N leaching losses whilst increasing pasture production and return on capital.

Thorrold *et al.* (2001) reported that the catchment of Lake Taupo in the central North Island of New Zealand has been targeted by local government for decreased input of N into the lake. A simulation study concluded that farming systems seeking to reduce N leaching would require a change in stock density and fertiliser management, and that

these measures would result in a significant economic penalty. It was suggested that enforcing such changes would result in farmers re-optimising their production in an alternative manner with likely equivalent N losses. Furthermore, it was suggested that farmers would not voluntarily take up such changes in land use practise without substantial financial support.

The New Zealand Parliamentary Commissioner for the Environment's report on the sustainability of the country's pastoral agriculture industry entitled 'Growing for good' (Parliamentary Commissioner for the Environment 2004) has received a lot of attention of late. The report suggests that in order to be sustainable, pastoral agriculture systems will have to be redesigned. While the report is short on the specifics of such redesign, it is clear that the farms of tomorrow need to decrease the impacts of N and P losses, faecal contamination and soil erosion on the wider receiving environment. The potential to redesign New Zealand's farming systems to decrease the associated impacts of intensive agriculture merits further research.

### **2.3.3 Fertiliser application**

#### *2.3.3.1 Losses of nitrogen as a result of N fertiliser additions*

In order to help provide the increased pasture required by dairy herds with increasing stocking rates, fertilisers are used to maintain a high soil fertility status on most New Zealand dairy farms. In particular, the addition of urea can be an essential tool for providing increased pasture growth at times of key feed requirement.

Sharpley and Syers (1979) demonstrated that a late June (winter) application of urea ( $60 \text{ kg N ha}^{-1}$ ) resulted in a prolonged period of elevated concentrations of  $\text{NO}_3^-$ -N in mole and pipe drainage from a Tokomaru silt loam in the Manawatu. A total of  $3.14 \text{ kg N ha}^{-1}$  was lost in a four week period following the late June application of urea compared to  $1.75 \text{ kg N ha}^{-1}$  from the control area (Figure 2.3).

A three year trial on four dairy cattle grazed permanent pasture farmlets on a Horotiu silt loam near Hamilton, New Zealand measured nitrogen inputs and outputs (Ledgard *et al.* 1999). Urea was applied (the N being applied in 8 - 10 applications) at rates of 0, 200 and  $400 \text{ kg N ha}^{-1} \text{ yr}^{-1}$  to farmlets with a stocking rate of  $3.3 \text{ cows ha}^{-1}$ . Average leaching losses of 19% and 28% of N applied in the 200 and  $400 \text{ kg N ha}^{-1} \text{ yr}^{-1}$  were recorded with a direct loss of 10% of the  $400 \text{ kg N ha}^{-1} \text{ yr}^{-1}$  treatment in the urea-N

form. The mean leaching losses from this 3 year farmlet trial reported in Ledgard *et al.* (1998) were 40, 81 and 152 kg N ha<sup>-1</sup> yr<sup>-1</sup> from the 0, 200 and 400 kg N ha<sup>-1</sup> yr<sup>-1</sup> farmlets respectively (Table 2.4). Ledgard *et al.* (1998) further reported that groundwater NO<sub>3</sub><sup>-</sup>-N concentrations under the respective 0, 200 and 400 kg N ha<sup>-1</sup> yr<sup>-1</sup> farmlets were 7, 10 and 19 mg litre<sup>-1</sup>.

Eckard *et al.* (2004) reported on NO<sub>3</sub><sup>-</sup>-N leaching losses for three years from a mole and pipe drained soil with and without the application of 200 kg ha<sup>-1</sup> yr<sup>-1</sup> of N fertiliser as either ammonium or urea in 4 split dressings (one per season). The range of annual NO<sub>3</sub><sup>-</sup>-N losses from the three different treatments were; 3.7 - 14.6 kg N ha<sup>-1</sup> yr<sup>-1</sup> from the control, 6.2 - 22.0 kg N ha<sup>-1</sup> yr<sup>-1</sup> from the urea treatment, and 4.3 - 37.6 kg N ha<sup>-1</sup> yr<sup>-1</sup> from the ammonium nitrate treatment (Table 2.4). The season of greatest risk for application of N fertiliser (particularly fertiliser already in the NO<sub>3</sub><sup>-</sup>-N form) was deemed to be winter when there was a substantial likelihood of drainage occurring. This is also a period of slow pasture growth rates and hence low nutrient uptake.

A comparison of NO<sub>3</sub><sup>-</sup>-N leaching losses using ceramic cup samplers from different land uses in Pukekohe, New Zealand on a well drained Allophanic Patumahoe clay loam by Francis *et al.* (2003) showed that dairying had by far the lowest leaching loss of 15 kg ha<sup>-1</sup> yr<sup>-1</sup> from a fertiliser input of 84 kg ha<sup>-1</sup> yr<sup>-1</sup> when compared to potatoes (164 kg ha<sup>-1</sup> yr<sup>-1</sup> from a fertiliser input of 481 kg ha<sup>-1</sup> yr<sup>-1</sup>) and winter green vegetables (132 kg ha<sup>-1</sup> yr<sup>-1</sup> from a fertiliser input of 166 kg ha<sup>-1</sup> yr<sup>-1</sup>). These leaching losses reflected the large differences in N fertiliser inputs.

A number of trials have been conducted at Lincoln University near Christchurch on free draining soils in lysimeters to investigate the impacts of irrigation management and N fertiliser rates on N leaching losses. Di *et al.* (1998) applied 400 kg N ha<sup>-1</sup> yr<sup>-1</sup> of NH<sub>4</sub><sup>+</sup>-N fertiliser (NH<sub>4</sub>Cl) in two even split dressings (December and May) to lysimeters containing Templeton fine sandy loam that received either summer spray or flood irrigation. Total mineral N losses as a result of the added N fertiliser were the same for both irrigation treatments (c. 45 kg ha<sup>-1</sup> yr<sup>-1</sup>) despite the lower concentrations recorded under flood irrigation as a result of the greater volume of water applied (Table 2.4). Further research reported by Silva *et al.* (1999) involved applications of urea at rates of 0, 200 and 400 kg N ha<sup>-1</sup> yr<sup>-1</sup> in 4 split dressings (May, August, November and February) to summer-autumn flood irrigated lysimeters containing a Templeton fine sandy loam. The resulting NO<sub>3</sub><sup>-</sup>-N losses were 8.3 kg N ha<sup>-1</sup> yr<sup>-1</sup> and 17.4 kg N ha<sup>-1</sup> yr<sup>-1</sup>

from the 200 and 400 kg N ha<sup>-1</sup> yr<sup>-1</sup> treatments, respectively (Table 2.4). The no urea scenario resulted in an annual loss of 3.2 kg N ha<sup>-1</sup>.

**Table 2.4** *The amount of N loss from a range of different applications rates and forms of N fertiliser.*

Research	N Input (kg ha <sup>-1</sup> yr <sup>-1</sup> )	Fertiliser	Application (kg ha <sup>-1</sup> yr <sup>-1</sup> )	NO <sub>3</sub> <sup>-</sup> -N Loss (kg ha <sup>-1</sup> yr <sup>-1</sup> )
Di <i>et al.</i> (1998) <sup>a</sup>	400	NH <sub>4</sub> Cl	2 x 200	45
Ledgard <i>et al.</i> (1998) <sup>c</sup>	0	-	-	40
Ledgard <i>et al.</i> (1998) <sup>c</sup>	200	Urea	9 x 22.5	81
Ledgard <i>et al.</i> (1998) <sup>c</sup>	400	Urea	9 x 45	152
Silva <i>et al.</i> (1999) <sup>a</sup>	0	-	-	3.2
Silva <i>et al.</i> (1999) <sup>a</sup>	200	Urea	4 x 50	8.3
Silva <i>et al.</i> (1999) <sup>a</sup>	400	Urea	4 x 100	17.4
Eckard <i>et al.</i> (2004) <sup>b</sup>	0	-	-	6.2 - 22
Eckard <i>et al.</i> (2004) <sup>b</sup>	200	Urea	4 x 50	3.7 – 14.6
Eckard <i>et al.</i> (2004) <sup>b</sup>	200	NH <sub>4</sub> NO <sub>3</sub>	4 x 50	4.3 - 37.6
Monaghan <i>et al.</i> (2005) <sup>c</sup>	0	-	-	30
Monaghan <i>et al.</i> (2005) <sup>c</sup>	100	Urea	2 x 50	34
Monaghan <i>et al.</i> (2005) <sup>c</sup>	200	Urea	4 x 50	46
Monaghan <i>et al.</i> (2005) <sup>c</sup>	400	Urea	8 x 50	56

a = fine sandy loam

b = silty clay loam

c = silt loam

Monaghan *et al.* (2005) reported research on a Pallic Waikoikoi-Arthurton silt loam and Brown Woodlands-Oteramika silt loam mix in Southland to determine nutrient losses in artificial drainage from cattle grazed pasture under a range of N fertiliser rates (control, 100, 200 & 400 kg N ha<sup>-1</sup> yr<sup>-1</sup>). Mean NO<sub>3</sub><sup>-</sup>-N leaching was significantly greater from the 200 and 400 kg N ha<sup>-1</sup> yr<sup>-1</sup> compared to the 0 and 100 ha<sup>-1</sup> yr<sup>-1</sup> treatments (Table 2.4). The authors suggested that N fertiliser rates of up to 170 kg N ha<sup>-1</sup> yr<sup>-1</sup> were sustainable if the New Zealand drinking water standard for NO<sub>3</sub><sup>-</sup>-N (11.3 mg L<sup>-1</sup>) were used to define an acceptable upper limit for N pollution.

The research reported above on N leaching losses as a result of fertiliser application clearly identifies the advantage of splitting applications across the growing season to minimise the risk of NO<sub>3</sub><sup>-</sup>-N leaching. As expected, the greater the input of fertiliser N the greater the magnitude of subsequent NO<sub>3</sub><sup>-</sup>-N leaching (Table 2.4).

### 2.3.3.2 Losses of phosphorus as a result of P fertiliser additions

New Zealand pastoral farming has a long history of applying P fertilisers (particularly superphosphate) to maintain soil P levels (Gillingham & Thorrold 1998). Traditionally, the available P status of soils has been determined in New Zealand by way of the Olsen P test (Cornforth and Sinclair, 1984). McDowell *et al.* (2003c) suggest that the environmental risk of P losses in surface runoff and drainage waters will be reduced if P fertiliser additions are managed so as to keep Olsen P concentrations to agronomic optimum levels. McDowell and Condon (2004) reported a relationship between soil Olsen P and dissolved reactive P (DRP) concentrations from both overland flow and subsurface drainage in small plots (1 m length) with artificial rainfall when pasture soils had not been recently grazed. The concentration of DRP in subsurface flow was estimated by the following equation  $DRP = 0.069 \times (\text{Olsen P/P retention}) + 0.007$ . The equation for overland flow was  $DRP = 0.495 \times (\text{Olsen P/P retention}) + 0.024$ .

On a mole and pipe drained site at Massey University's No. 4 Dairy farm near Palmerston North, Turner *et al.* (1979) found that the addition of P fertiliser resulted in increased losses of dissolved forms of P, particularly DIP, with up to 1.8% of the added P lost in the dissolved form in subsequent drainage events.

A review of P loss in surface runoff under pasture by Gillingham and Thorrold (1998) reported that diffuse sources of P loss from agriculture contribute approximately 90% of the total P entering fresh waters annually in New Zealand. Losses were expected to range from 0.11 to 1.67 kg P ha<sup>-1</sup> yr<sup>-1</sup>. It was reported that up to 20% of the annual P export from a catchment could be accounted for by fertiliser P from aerial top dressing falling directly onto waterways or saturated soils. The accurate placement of fertiliser so as to avoid riparian zones is an important way to decrease the impact of P fertiliser additions. The authors suggested that the optimum time to apply P fertiliser to avoid or reduce the likelihood of P runoff is in late spring and late autumn when there is sufficient soil moisture for a response to the fertiliser, and a reduced risk of surface runoff.

In Victoria Australia, Nash and Murdoch (1997) reported that 69% of P loss in surface runoff under fertile dairy pasture occurred during large storm events. Erosion or movement of surface soil particles was not believed to be the major process by which P loss occurred as 91% of the total 3.2 kg P ha<sup>-1</sup> lost was in the dissolved form. It was suggested that this dissolved form of P was a result of dissolution processes at or near

the surface of this very fertile soil. Further research in Victoria Australia by Nash *et al.* (2000) investigated the relationships between P loss, days since grazing, days since fertilizing and total storm flow from a 3.6 ha dairy pasture receiving 80 kg ha<sup>-1</sup> yr<sup>-1</sup> as P fertiliser in year 1 and 110 kg ha<sup>-1</sup> yr<sup>-1</sup> in year 2 and 3. Days since fertilizing was found to be inversely related to total P concentrations in runoff water where as days since grazing was only weakly related to total P concentrations suggesting that cattle did not mobilise large stores of P relative to fertilization. The total loads exported from the research area were 1.9 kg ha<sup>-1</sup> yr<sup>-1</sup> (year 1) 5.7 kg ha<sup>-1</sup> yr<sup>-1</sup> (year 2) and 9.7 kg ha<sup>-1</sup> yr<sup>-1</sup> in the atypically wet, year three.

### 2.3.4 Summary

Despite the recent expansion of dairy farming in New Zealand, there has been relatively little research of the impacts of intensive grazing of dairy cattle, application of fertiliser, and maintenance of a high soil nutrient status on the quality of artificial drainage and surface runoff waters. In particular, the losses of P under intensively grazed dairy pasture have been poorly documented as much of the research conducted has concentrated on the losses of NO<sub>3</sub><sup>-</sup>-N. Research gaps have been identified, in particular those associated with the likelihood of rapid preferential flow of nutrients in mole and pipe drained soil, and the timing and form of nutrient loss following the coincidence of dairy cattle grazing events and drainage.

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# CHAPTER 3: THE RESEARCH SITE

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## 3.1 INTRODUCTION

This Chapter describes the soil characteristics and the experimental design of the research site where all the field work reported in this thesis was conducted.

## 3.2 SITE AND SOIL

The research site for this study was located on Massey University's No. 4 Dairy farm, near Palmerston North in the Manawatu region (NZMS 260, T24, 312867) of New Zealand. The No. 4 Dairy farm is situated on Tennent Drive immediately south west of the Massey University's Turitea campus.

The soil type at the research site is a Tokomaru silt loam and is classified as an Argillic-fragric Perch-gley Pallic Soil (Hewitt 1998) or a Typic Fragiaqualf (Soil Survey Staff 1998). The soil parent material is derived from deep deposits of non-calcareous, loess-blown river sediments (primarily greywacke origin) deposited on the eastern banks of the Manawatu river which form on a deeply dissected uplifted marine terrace (Pollock 1975, Shepherd 1984, Molloy 1998). The Tokomaru silt loam soil consists of a weakly to moderately developed, brown, silt loam A-Horizon (c. 0 - 250 mm soil depth), a weakly developed, grey, strongly mottled, clay loam B-Horizon (c. 250 - 800 mm soil depth) and a highly compacted, weakly-developed, pale-grey, silt loam fragipan C-Horizon, which acts as a natural barrier to drainage (Sheppard 1984, Scotter *et al.* 1979). Below the fragipan, there is 600-800 mm of compact fine silt loam similar to the overlying B horizon which then overlays 120-200 mm of light grey pumiceous sand (Aokautare Ash). The profiles range from imperfectly to poorly drained (Shepherd 1984). A full soil profile description has been presented by Pollock (1975).

The site is approximately 60 m above sea level in a flat to easy rolling landscape of approximately 3% slope (Pollock 1975). The region receives an average annual rainfall of approximately 1000 mm that is distributed relatively evenly across the winter and spring months but occasionally subjected to fluctuations in the summer and autumn months causing soil water deficits with occasional severe droughts (Pollock 1975, Shepherd 1984). The highest 50 year mean monthly rainfall is in June with 99 mm and

the lowest is in March with 66 mm. Net evapotranspiration is typically greater than net rainfall from November through until May. Palmerston North has a mild sub-humid climate with mean summer (January) temperatures of 18 degrees Celsius and mean winter (July) temperature of 8 degrees Celsius (Pollock 1975, Shepherd 1984).

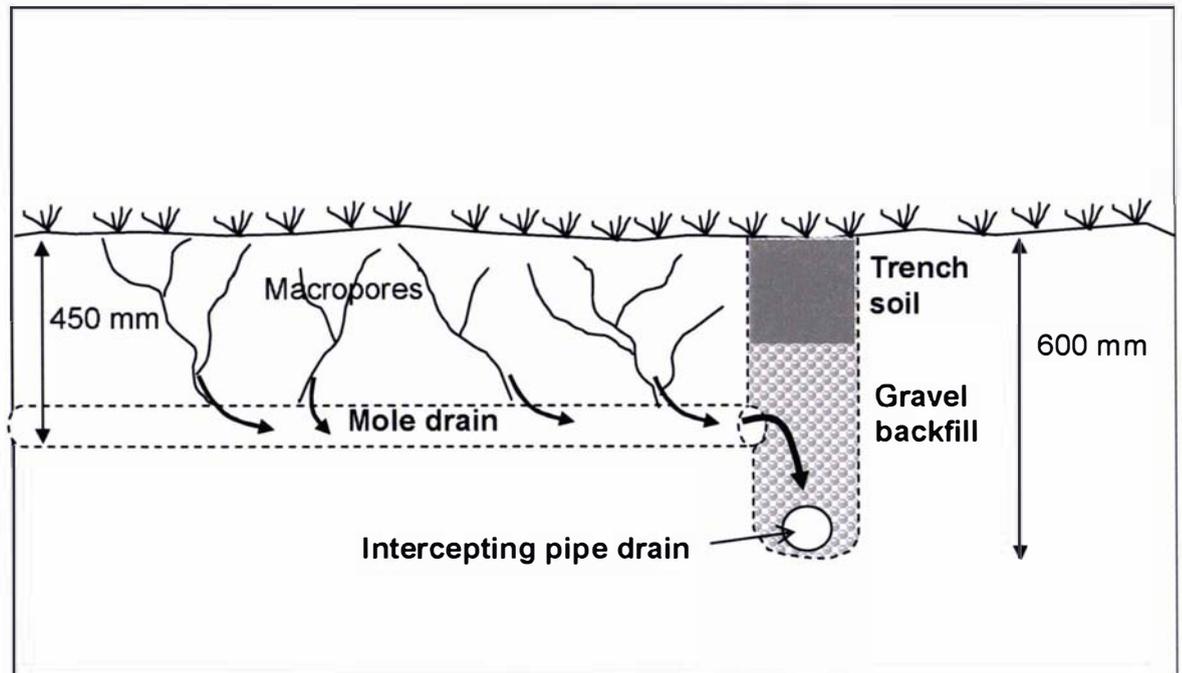
The site supports a mixed pasture of predominantly perennial ryegrass (*Lolium perenne*) and white clover (*Trifolium repens*). Most of the effective plant roots were found above the level of the fragipan (Scotter *et al.* 1979). The vegetation prior to 1880 (pre- European settlement) was a standing forest of indigenous broadleaf and podocarp species (Pollock 1975).

The Tokomaru silt loam was reported by Scotter *et al.* (1979) to have a bulk density of 1.1 Mg m<sup>-3</sup> and a field capacity of 45% v/v at a depth of 0-100 mm. In May 2003, an Olsen P of 40 µg P ml<sup>-1</sup> was measured for the 0-75 mm depth at the research site using the MAF 'Quick Test' method (Cornforth and Sinclair 1984).

### 3.3 EXPERIMENTAL DESIGN

Mole and pipe drainage was installed at the research site approximately 12 months prior to the commencement of initial data collection (January 2000). Mole and pipe drainage provides an artificial subsurface drainage network to alleviate water-logging in naturally imperfectly and poorly drained soils. Mole drains are installed into the soil by a mole plough at approximately 450 mm depth (Figure 3.1). Mole ploughing should take place when the subsoil is in a reasonably plastic state; such conditions allow for the extrusion of a stable mole channel, and for the creation of a shattered zone above the mole channel (Bowler 1980, Cornforth 1998). The resulting system of cracks and channels then act as large macropores effectively removing surplus drainage water to the mole channels. The mole channels are spaced at approximately 2 m intervals and drain water into an intercepting pipe drain at approximately 600 mm depth. The location of an impermeable fragipan at approximately 800 mm depth means that the mole and pipe drainage systems at the No.4 Dairy farm provide an effective way of capturing nearly all the excess water at one point source location.

The mole and pipe drainage system was established with a large drainage coefficient. This meant that flow rates measured were as a result of the mole cracks and channels ability to drain water to the collecting pipe drain rather than flow rates being kept in check by the pipe drains' ability to move water as was reported by Horne (1985).



**Figure 3.1** Diagram showing a typical mole and pipe drained soil profile.

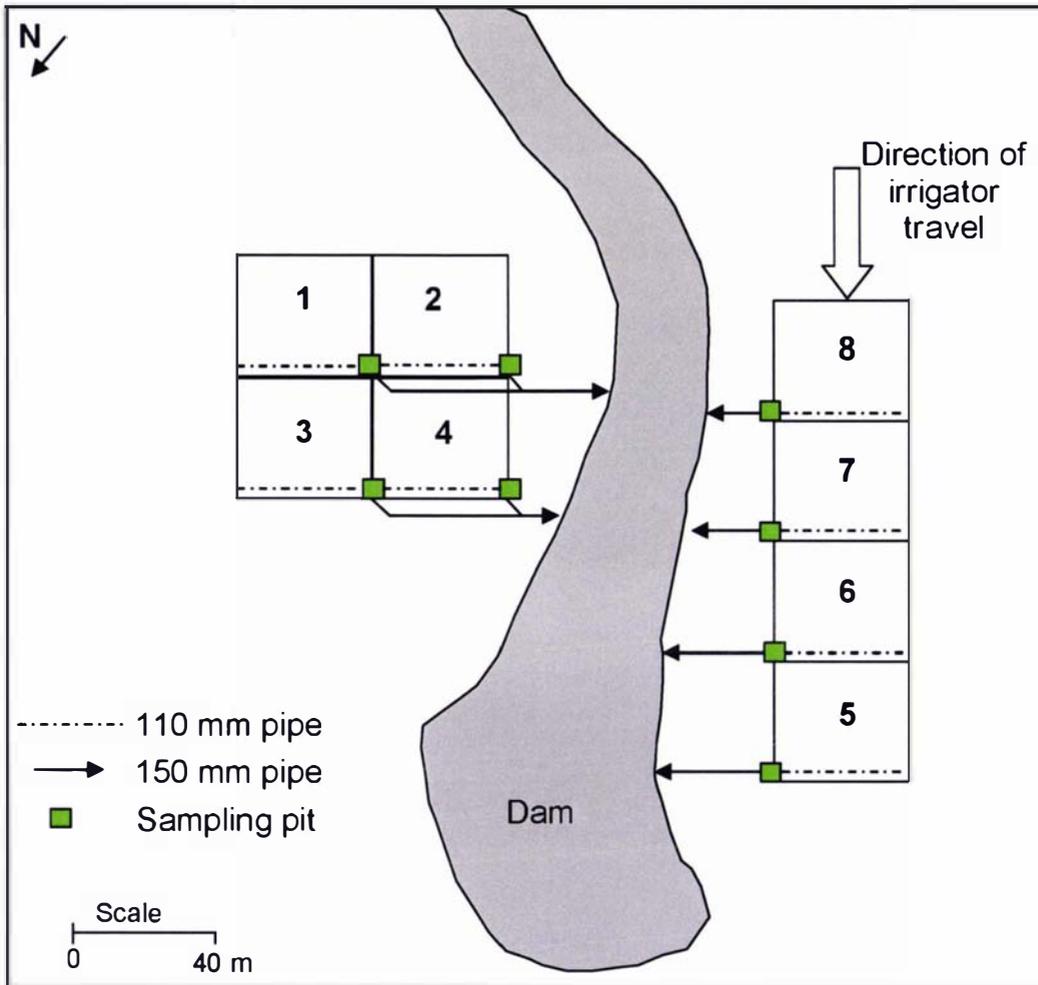
The research area consists of eight hydrologically isolated mole and pipe drained plots; each plots is 40 m by 40 m. Each plot has a collecting pipe drain at the downslope perimeter. All drainage water passes through a v-notch weir in an excavated sampling pit (Plate 3.1, Figure 3.2). The v-notch weirs are fitted with data loggers (Odyssey capacitance probes or NIWA hydrologgers) for continuous measurement of flow rate.



**Plate 3.1** A v-notch weir placed within an excavated pit and fitted with a data-logger to measure stage height of any exiting pipe drainage.

Two treatments were imposed upon the eight experimental plots. Plots 1-4 represent the standard farm practises of the No.4 dairy farm including grazing and fertiliser

regimes. Plots 5-8 represent the standard farm practises of the No.4 dairy farm plus the irrigation of farm dairy effluent (FDE) but excluding the application of nitrogen fertiliser.



**Figure 3.2** A schematic diagram of the field research site. Plots 1-4 represent standard farm practise at No 4 dairy farm whilst plots 5-8 represent standard farm practise plus FDE applications.

During the autumn of 2003, four hydrologically isolated sub-plots were installed on plots 2, 4, 7 & 8 in order to capture surface runoff from two of the four plots from each treatment (Plate 3.2). Each of these runoff plots was 5 m by 10 m. All surface runoff collected was channelled into the nearby pit that had been excavated to sample drainage water. In this pit, runoff water flowed through a one litre tipping bucket fitted with a data logger to provide continuous flow rate (Plate 3.3).



**Plate 3.2** *Installation of a 50 m<sup>2</sup> surface runoff plot. The plots were hydrologically isolated with buried boards. Runoff was collected by guttering placed at the down-slope edge of the plot. Runoff water was conveyed to the adjacent pit where it was measured.*



**Plate 3.3** *A one litre tipping bucket fitted with a data logger to measure flow rate from the surface runoff plots.*

### 3.4 RESEARCH REPORTING

The field experimentation was reported as results chapters/publications from chapter 4 to chapter 8. Research reported in chapters 4 & 5 was carried out on the four effluent plots (plots 5-8) over three lactation seasons from November 2000 to April 2003. Chapters 6 & 7 report on drainage quality research carried out on all experimental plots for the consecutive winter/spring periods of 2002 and 2003 respectively. Chapter 8 reports on two field experiments that took place during the 2003 drainage season. Experiment 1 took place in June on both the effluent and non effluent plots whilst experiment 2 took place on only the effluent receiving plots in October.

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# CHAPTER 4: MINIMISING SURFACE WATER POLLUTION RESULTING FROM FARM DAIRY EFFLUENT APPLICATION TO MOLE-PIPE DRAINED SOILS. AN EVALUATION OF THE DEFERRED IRRIGATION SYSTEM FOR SUSTAINABLE LAND TREATMENT IN THE MANAWATU.

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## 4.1 ABSTRACT

There is little information available on the magnitude of nutrient losses to surface water from the two-pond and daily irrigation treatment systems for farm dairy effluent (FDE). A research site has been established on a mole-pipe drained Tokomaru silt loam at Massey University's No 4 Dairy farm (475 cows) to investigate some of these issues. The site consists of four plots (40 x 40 m) that have been instrumented to allow the continuous monitoring of drainage and surface runoff. The research was conducted over three lactation seasons (2000/01-2002/03). Based on data collected at the study farm over one season, it was calculated that in the past 1500 kg N yr<sup>-1</sup> and 250 kg P

yr<sup>-1</sup> were potentially discharged from the two-pond system directly to a stream. A simulation exercise suggests that approximately 108 kg N yr<sup>-1</sup> and 18 kg P yr<sup>-1</sup> would be lost to surface waters if daily irrigation was practised at the farm. The problems of daily irrigation, particularly those related to surface runoff, were further quantified in an experiment in which a single 25 mm FDE irrigation was applied to a soil near field capacity. Approximately 40% of the applied effluent left the soil profile as mole and pipe drainage and 30% as surface runoff. These losses equated to 12 kg N ha<sup>-1</sup> and 2 kg P ha<sup>-1</sup>. To minimize nutrient losses from land application of FDE, a system called “deferred irrigation” was designed. Deferred irrigation involves storing effluent in a two-pond treatment system and then applying it strategically when there is a suitable soil water deficit i.e. the irrigation volume does not exceed the potential soil-water storage. The evaluation of deferred irrigation over three lactation seasons showed that direct losses of nutrients to surface waters were almost eliminated and resulted in the drainage of only approximately 1% of the total effluent nutrients applied. The successful adoption of the deferred irrigation system would require only the capability to store effluent and model or measure soil moisture status within the active root zone.

**Keywords:** farm dairy effluent, spray irrigation, nutrient loss, soil water balance, mole and pipe drainage, preferential flow

## 4.2 INTRODUCTION

The impact of dairy farming on the aquatic environment has come under increasing scrutiny in recent times. It is widely believed that intensive dairy farming is responsible for accelerated contamination of waterways by: nutrients, suspended solids, pathogenic organisms and faecal matter (Sharpley & Syers 1979; Monaghan *et al.* 2002; Houlbrooke *et al.* 2003 (*Chapter 6*)). In particular, farm dairy effluent (FDE) is implicated as a major contributor to the degradation of surface water quality (Ledgard *et al.* 1996; Sukias *et al.* 2001). Poorly managed FDE land treatment systems may generate nutrient-rich surface runoff and drainage waters which have the potential to pollute surface and ground waters (Macgregor *et al.* 1979; Di *et al.* 1998; Singleton *et al.* 2001). In addition, high effluent loads may decrease soil quality (Cameron *et al.* 1997).

The standard aerobic-anaerobic two-pond treatment system efficiently reduces the sediment load due to settling and the biological oxygen demand of FDE where excess oxygen produced by algae is used by bacteria to breakdown effluent, but discharge

from ponds to surface water typically has a high concentration of nutrients (Longhurst *et al.* 2000; Sukias *et al.* 2001). Consequently, the direct discharge of effluent from the two-pond treatment system to surface water is currently being phased out by regulatory authorities (Cameron & Trenouth 1999). Spray irrigation of FDE to land is considered to be a major improvement on the two-pond system which discharges directly to surface water, and accordingly, land treatment is the preferred treatment option of regulatory authorities (Heatley 1996).

In most cases, regulatory authorities attach conditions to land application of FDE, including setting minimum sizes for treatment areas based on cow numbers and maximum nitrogen loading rates (usually 150 - 200 kg N ha<sup>-1</sup> yr<sup>-1</sup>). In addition, some attention is given to the hydraulic loading of the soil to restrict or prevent surface water ponding (Heatley 1996). However, the conditions imposed by regulatory authorities are too general and lack soil-specificity to help farmers schedule spray irrigation of FDE and minimise the risk of nutrient loss in surface runoff and drainage.

Many farmers need to spray irrigate raw (untreated) FDE daily, irrespective of soil moisture conditions, because they have no or limited storage facilities. Under these circumstances, direct drainage of effluent is likely in spring and late autumn when the soil is close to, or at, field capacity. These problems can be further exacerbated when FDE applications are made to soils with drainage limitations where there is the risk of generating runoff. Also, when FDE is applied to soils that are artificially drained with mole and pipe systems, such as is common in many of the dairy-farmed areas of the Manawatu and other regions of New Zealand, there is the risk that FDE will move quickly through the preferential flow paths above mole channels before exiting pipes into surface waters (Scott *et al.* 1998).

To date, there has been no published research in New Zealand documenting the direct loss of spray-irrigated FDE to fresh water bodies via surface runoff. The high instantaneous application rates of many traveling irrigator's means that, at larger application depths, ponding and surface runoff of effluent may occur. Also, the nutrient composition of effluent lost as surface runoff is likely to be greater than that of effluent lost as drainage because there will be less opportunities for chemical (sorption) and biological (plant uptake) processes to remove nutrients from effluent that runs across the soil surface. Although we refer to the 'drainage and runoff of FDE' throughout this paper, we acknowledge that the drainage and runoff that results from FDE irrigation is not likely to be undiluted FDE but will be the product of a complex set of interactions -

involving mixing, dilution and displacement - between applied FDE and resident soil moisture.

Land treatment of FDE is still a relatively new technology and many farmers experience difficulties implementing effective systems. There are currently no rigorous criteria (based on soil water status) for the day-to-day management of effluent irrigation, particularly in difficult situations, such as those presented by imperfectly drained soils with established mole and pipe drainage. Therefore, it is not surprising that anecdotal evidence suggests that current land treatment practices for FDE are causing nutrient enrichment of surface waters via runoff or rapid movement through artificial drainage systems. It is clear that when FDE irrigations are carried out frequently in the late-winter/spring period, irrespective of soil water status, there is a clear risk to the aquatic environment from nutrient loss.

As stated above, the maximum effluent irrigation rate is commonly set so that the nitrogen loading of land does not exceed 150 – 200 kg N ha<sup>-1</sup> yr<sup>-1</sup>. When the maximum amount of FDE is applied to soils, K is added at rates of 120-160 kg ha<sup>-1</sup> yr<sup>-1</sup>, which is far in excess of the 50 - 90 kg K ha<sup>-1</sup> yr<sup>-1</sup> required for maintenance of K reserves in the soil (Roach *et al.* 2001). When K is applied to dairy farm soils at rates that are far greater than maintenance requirements, the increase in K levels in spring-time pasture, and the concomitant decrease in calcium and magnesium in the cows diet can induce metabolic disorders in cows, particularly milk fever (hypocalcaemia) and ryegrass staggers (hypomagnesaemia) (Roach *et al.* 2001; Tillman & Surapaneni 2002).

To help overcome the problems associated with the spray irrigation of FDE to artificially drained soils and soils with drainage limitations, an improved treatment system called 'deferred irrigation' has been developed. Deferred irrigation is a synthesis of current scientific knowledge and best management practices into a comprehensive package that allows farmers to manage effluent irrigation in a sustainable manner. Deferred irrigation utilises existing resources while streamlining and simplifying the irrigation procedure.

Deferred irrigation involves storing effluent in the aerobic pond of a two-pond treatment system, then irrigating it strategically from the aerobic pond when there is a suitable soil water deficit. Scheduling effluent irrigation when sufficient soil water deficits exist avoids the risks of generating surface runoff or direct drainage of effluent. When applied effluent adds to the pool of plant available water and not to the pool of drainage

water, then the soil-plant system's ability to remove soluble nutrients via plant uptake and immobilisation processes in the soil is maximised. Thus, the application criteria for spray irrigation of FDE if drainage is to be avoided are:

$$E_i + \theta_i Z_R \leq \theta_{FC} Z_R \quad \text{eq. 1}$$

$$E_i \leq Z_R (\theta_{FC} - \theta_i) \quad \text{eq. 2}$$

Where  $E_i$  is the depth of FDE (mm) applied on day  $i$ ,  $Z_R$  is the effective rooting depth (mm),  $\theta_{FC}$  is the soil water content at field capacity ( $\text{m}^3 \text{m}^{-3}$ ), and  $\theta_i$  is the soil water content on day  $i$  ( $\text{m}^3 \text{m}^{-3}$ ).

In many areas of New Zealand, soil water deficits greater than 10 mm usually only occur between the months of October and May, however the generation of FDE starts at the beginning of lactation in late winter (August). Consequently, having sufficient storage for FDE is essential to ensure that spray irrigation only occurs during times when the soil water deficit is adequate.

The first objective of this study was to estimate annual losses of nutrients (N and P) to surface waters from two common FDE treatment systems: a two-pond system and a daily spray irrigation system. The second objective was to quantify the nutrient composition of surface runoff and drainage resulting from a single irrigation of FDE at a rate in excess of the soil water deficit, as might be the case when FDE is applied irrespective of soil water status in a daily land application system. The final objective was to demonstrate the effectiveness of the deferred irrigation system at minimising the risk of direct losses of FDE to surface waters.

## 4.3 MATERIALS AND METHODS

### 4.3.1 Site and soil

A research site was established in January of 2000 on a mole-pipe drained Tokomaru silt loam soil on Massey University's No. 4 Dairy Farm near Palmerston North, New Zealand. The soil, which is classified as an Argillic-fragic Perch-gley Pallic Soil (Hewitt 1998) or a Typic Fragiaqualf (Soil Survey Staff 1998), is derived from deep deposits of loess-blown river sediments which form on a deeply dissected uplifted marine terrace (Molloy 1998). The Tokomaru silt loam soil consists of a weakly to moderately developed, brown, silt loam A-Horizon (c. 0 - 250 mm soil depth), a weakly developed,

grey, strongly mottled, clay loam B-Horizon (c. 250 - 800 mm soil depth) and a highly compacted, weakly-developed, pale-grey, silt loam fragipan C-Horizon, which acts as a natural barrier to drainage (Scotter *et al.* 1979a). Most of the effective plant roots were found above the level of the fragipan. The site is located in a flat to easy rolling landscape (c. 3% slope) which receives an average annual rainfall of approximately 1000 mm. The site supports a mixed pasture of perennial ryegrass (*Lolium perenne*) and white clover (*Trifolium repens*). The Tokomaru silt loam was reported by Scotter *et al.* (1979a) to have a bulk density of 1.1 Mg m<sup>-3</sup>, a saturated hydraulic conductivity of 32 mm day<sup>-1</sup> and a field capacity of 45% v/v at a depth of 0-100 mm. In May 2003, an Olsen P of 40 mg P ml<sup>-1</sup> was measured for the 0-75 mm depth using the MAF 'Quick Test' method (Cornforth and Sinclair 1984).

The research area consisted of four plots (each 40 m x 40 m). Each plot had an individual mole-pipe drain system. Mole drains were installed at 2-m intervals at a depth of 450 mm. Drainage from the mole system was intercepted by a collecting 110 mm perforated pipe drain perpendicular to the mole drains at 60 cm depth. At the corner of each plot, a pit was excavated and a V-notch weir placed at the exit of the pipeline to monitor drainage flow rates. All pits were instrumented with data loggers to provide continuous measurements of flow rate. Any drainage water occurring as a result of the application of FDE was sampled manually from the pipe outlet for water quality analyses. For the 2003 winter drainage season, two of the plots each had an isolated sub-plot (50 m<sup>2</sup>) installed for the measurement of surface runoff. Surface runoff water collected from each sub-plot passed through a tipping bucket instrument to measure flow rate and to provide a flow-proportioned mixed sample for water quality analyses.

#### **4.3.2 Experimental procedure**

Aerobic pond effluent was irrigated to the plots, in accordance with the deferred irrigation scheduling criteria, for the three lactation seasons (c. 280 days from August to May) between 2000 and 2003. Farm dairy effluent was applied to the plots by travelling irrigators. In the first two lactation seasons (2000/01 and 2001/02), a 'Briggs' Model 15 rotating travelling irrigator was used and in the third season (2002/03) a 'Spitfire Mark I' oscillating travelling irrigator was used. Adjustable speed settings on the irrigators allowed the depth of the effluent applied to be varied. Irrigations varied in timing and quantity between seasons.

The quantity of direct drainage resulting from the land application of FDE using the deferred irrigation system was monitored immediately following application, and the nutrient (N and P) concentrations in drainage waters were determined during the three lactation seasons. The nutrient concentration of the applied FDE was also measured at each irrigation. The quantity and nutrient concentration of surface runoff was also monitored, where it occurred, in the 2002/2003 season. In the simulation study, the concentrations of N and P in effluent exiting the aerobic pond were the mean concentrations of N and P in effluent samples collected under the travelling irrigator during the 2001/02 lactation season.

Near the start of the 2003/04 season, an additional effluent irrigation was made to provide quantitative data on the nutrient composition of effluent lost in surface runoff and drainage in a case when deferred irrigation criteria was not followed e.g. daily irrigation systems where FDE is applied irrespective of soil moisture conditions. In September 2003, a single irrigation (25 mm depth) of FDE was applied (at a rate of approximately  $2.7 \text{ mm min}^{-1}$ ) to the plots when the soil water deficit was only 6 mm.

Another dictate of the deferred irrigation system, the annual removal of a cut of grass silage, was also practised. For the rest of the time, the plots were grazed according to the farm's normal management programme i.e. with an average grazing density of approximately  $80 \text{ cows ha}^{-1}$  for a 12-hour period.

Soil water deficits were computed using the soil water balance for Tokomaru silt loam described by Scotter *et al.* (1979b). The soil water balance was used to predict the likelihood of drainage of FDE when it was applied to land at  $30 \text{ mm day}^{-1}$  on a daily basis throughout the lactation season. Scotter *et al.*'s (1979b) model uses the Priestly and Taylor method to calculate evapotranspiration and requires the key parameters of daily rainfall, solar radiation and air temperature in order to predict, soil water deficits, drainage and runoff.

#### 4.3.3 Nutrient analyses

The following analyses was conducted on samples of the FDE applied at each irrigation, and on any resulting drainage samples (and runoff on one occasion) captured from each of the experimental plots. The analyses included: suspended solids, total phosphorus (P) and total dissolved P (TDP), dissolved inorganic P (DIP), total nitrogen (N), total dissolved nitrogen (TDN), nitrate-N ( $\text{NO}_3^-$ -N), and ammonium-N

( $\text{NH}_4^+\text{-N}$ ) . Total N and TDN were calculated by adding  $\text{NO}_3^-\text{-N}$  values to those of Kjeldahl N and dissolved Kjeldahl N. Analyses were carried out calorimetrically on a Technicon Auto Analyser II using the following methods: Kamphake *et al.* (1967) using a hydrazine reduction for  $\text{NO}_3^-\text{-N}$ ; Searle (1975) for ammonium-N; McKenzie & Wallace (1954) for Kjeldahl N and dissolved Kjeldahl N following a Kjeldahl acid digest; and Murphy & Riley (1962) for analyses of DIP. Total P and TDP were determined by the Vanadomolybdate method (AOAC 1975) following a Kjeldahl acid digest as described by McKenzie & Wallace (1954). All measures of dissolved nutrients were determined on samples passed through a 1.2  $\mu\text{m}$  glass filter paper. The variability in measured data was quantified using the standard error of means (SEM) from four replicates.

## 4.4 RESULTS AND DISCUSSION

### 4.4.1 Theoretical analysis of two effluent treatment systems

Two methods of treating FDE, the two-pond system with direct discharge to surface water and daily irrigation, vary in their ability to prevent effluent-derived nutrients from contaminating surface waters. Provided below are estimates of losses of nutrients from these types of treatment systems on an annual basis.

#### 4.4.1.1 Two-pond treatment of FDE and discharge to surface water

The two-pond treatment system at Massey University's No 4 Dairy Farm receives approximately 10 000  $\text{m}^3 \text{ year}^{-1}$  of waste water. This waste water consists of the farm dairy cleaning and wash-down water containing the dung and urine associated with the milking of 475 cows (assumed to be 50 litres  $\text{cow}^{-1} \text{ day}^{-1}$ ; Vanderholm 1984), plus rainfall landing on the farm dairy, yards and a large feed pad. The average N and P concentrations of the effluent in the aerobic pond (second pond) were 150 mg N  $\text{litre}^{-1}$  (range 110 -151) and 25 mg P  $\text{litre}^{-1}$  (range 16-34) In the past this effluent was discharged directly to a stream, adding approximately 1500 kg N  $\text{yr}^{-1}$  and 250 kg P  $\text{yr}^{-1}$  to the aquatic environment.

#### 4.4.1.2 Daily spray irrigation of FDE

Due to limited, or lack of, storage facilities for FDE on many dairy farms, FDE may need to be applied on a daily basis irrespective of climate and soil moisture conditions.

Consequently, immediate losses of nutrient-rich FDE to surface waters, in either direct drainage or surface runoff could be expected.

To identify the potential for daily FDE applications to generate direct drainage or runoff, the soil water balance for No 4 Dairy Farm was simulated using past climate data for the ten-year period from 1991 to 2001 (Scotter *et al.* 1979b). A typical lactation period for No 4 Dairy Farm, 280 days, was assumed. In this simulation, a standard rotating travelling irrigator was assumed to apply a mean depth of 30 mm of effluent at each application. This is a common application depth used by farmers. The poor application uniformity under the rotating irrigator (Houlbrooke *et al.* 2004 (*Chapter 5*)) is accounted for in the simulation using the procedure described by Houlbrooke *et al.* 2004.

Results from the simulation suggest that under daily FDE irrigation, on average, 16% of the total annual volume of FDE reaches surface water as either direct drainage or surface runoff. Losses of FDE under daily irrigation would have been less if a lower application depth than 30 mm had been used in the simulation (Houlbrooke *et al.* 2004 (*Chapter 5*)). Typically, the majority of these losses under daily irrigation occur in the late-winter/spring period when soil water deficits tend to be less than 10 mm or non-existent. If nutrient concentrations in the FDE were undiluted during its passage across the soil surface or through macropore drainage channels, then an annual loss of 16% of FDE to surface waters would add approximately 240 kg N yr<sup>-1</sup> and 40 kg P yr<sup>-1</sup> to the aquatic environment. However, it is unlikely that the FDE lost in direct drainage or surface runoff will have nutrient concentrations identical to the applied FDE because some dilution or treatment can occur as it passes through or across the soil profile.

The extent of treatment, mixing and dilution undergone by effluent as it moves through and across the soil is described below. This study (data presented below) showed that the average concentrations of TN and TP of the effluent volume that directly drained were 45% of the original effluent applied. In comparison the nutrient concentration in surface runoff was c. 80% of concentration of TN and TP originally applied as FDE. As the model (Scotter *et al.* 1979b) does not differentiate between drainage and runoff volumes for the purpose of calculating nutrient losses here, it is assumed that all the excess effluent drains through the soil. For this best case scenario of zero surface runoff, the annual losses of TN and TP from the simulated daily application of FDE would be 45% of the applied effluent concentrations, which equates to 108 kg N yr<sup>-1</sup> and 18 kg P yr<sup>-1</sup>. If these total losses were spread over an area of 16 ha (the area irrigated with FDE at No. 4 Dairy Farm), they would equate to about 7 kg N ha<sup>-1</sup> yr<sup>-1</sup> and

1 kg P ha<sup>-1</sup> yr<sup>-1</sup>. If it had been possible to simulate the surface runoff that accompanied daily irrigation, these nutrient losses would have been considerably greater.

The losses estimated here for daily irrigation are significantly smaller than the calculated nutrient losses to surface waters from the No. 4 dairy two-pond system. However, daily irrigation still poses an unacceptable risk of contamination to the aquatic environment, which potentially could be reduced by scheduling FDE land application to coincide with soil water deficits.

#### 4.4.2 Spray irrigation of FDE to wet soils

To demonstrate the potential effect of applying FDE to wet soils, the impact of applying 25 mm of FDE to soil with a soil water deficit of 6 mm was measured. As a consequence of a single irrigation of 25 mm of FDE, there was 10 mm of drainage and 8 mm of surface runoff (Table 4.1). Approximately 70% of the applied FDE left the experimental plots (about 30% as surface runoff and 40% as direct drainage). With the exception of NO<sub>3</sub><sup>-</sup>-N, concentrations of different forms of both N and P in drainage were less than half (40-45%) of the corresponding concentrations in applied FDE (Table 4.1). The concentrations of N and P in drainage were still 550 and 50 times greater, respectively, than the ecologically significant levels (0.1 mg N litre<sup>-1</sup> & 0.1 mg P litre<sup>-1</sup>) deemed likely to promote aquatic weed growth (Ministry for the Environment 1992). The greater concentration of NO<sub>3</sub><sup>-</sup>-N measured in drainage water than was applied in FDE may be a result of leaching of NO<sub>3</sub><sup>-</sup>-N resident in the soil mineral pool at the time of application (Table 4.1). NO<sub>3</sub><sup>-</sup>-N leaching has been measured at the experimental site in winter-spring drainage induced by natural rainfall (Houlbrooke *et al.* 2003 (*Chapter 6*)).

Concentrations of different forms of both N and P in surface runoff were approximately 80% of the corresponding concentrations in the applied FDE (Table 4.1). This suggests that where effluent runoff enters surface waters, it does so in a more-or-less unchanged state.

In this single irrigation, 30 kg N ha<sup>-1</sup> and 4 kg P ha<sup>-1</sup> was applied to the soil. From this application, over 12 kg of N ha<sup>-1</sup> (5.1 kg N ha<sup>-1</sup> in drainage and 7.5 kg N ha<sup>-1</sup> in surface runoff) and nearly 2 kg of P ha<sup>-1</sup> (0.8 kg P ha<sup>-1</sup> in drainage and 1.1 kg P ha<sup>-1</sup> in surface runoff) was lost to surface water (Table 4.2). These nutrient losses from a single, badly-managed irrigation are significant, particularly when compared to annual drainage

losses of around 27 kg N ha<sup>-1</sup> and less than 0.4 kg P ha<sup>-1</sup> reported by Monaghan *et al.* (2002) and Houlbrooke *et al.* (2003) from pastures grazed by dairy cattle without effluent irrigation. In other words, N losses from a single FDE irrigation event to wet soil equate to about 45% of the expected annual drainage loss from grazed dairy pasture, while P losses were equivalent to the expected annual drainage loss from grazed dairy pasture.

**Table 4.1** *The range and average flow weighted nutrient concentrations of the applied FDE, drainage, and surface runoff associated with an irrigation of FDE when the soil was close to field capacity in September 2003. Values in brackets indicate SEM from four replicates. Mean drainage and runoff concentration data is presented in bold. Range is in small font italics.*

Nutrient Form	Concentration (mg litre <sup>-1</sup> )		
	Applied FDE	Mole and pipe drainage	Surface runoff
TN	122.4 (3.5)	<b>55.8 (3.5)</b> 20.3 - 92.2	<b>95.3 (1.9)</b> 93.4 - 97.2
TDN	84.7 (3.2)	<b>43.2 (6.1)</b> 12.3 - 66.6	<b>69.4 (1.9)</b> 67.3 - 71.4
NH <sub>4</sub> <sup>+</sup> -N	71.7 (2.1)	<b>27.1 (3.6)</b> 3.4 - 52.0	<b>55.9 (1.3)</b> 54.7 - 57.2
NO <sub>3</sub> <sup>-</sup> -N	0.08 (0.04)	<b>0.67 (0.27)</b> 0 - 2.2	<b>0.06 (0.02)</b> 0.0 - 0.2
TP	17.9 (0.5)	<b>8.5 (0.6)</b> 3.4 - 13.4	<b>15.7 (1.3)</b> 14.0 - 17.4
TDP	12.0 (0.6)	<b>5.0 (0.7)</b> 1.0 - 8.8	<b>9.4 (1.2)</b> 8.3 - 10.6
DIP	8.8 (1.4)	<b>3.0 (0.4)</b> 0.5 - 4.9	<b>6.8 (0.5)</b> 6.3 - 7.4
Volume	25 mm	10 mm	8 mm

**Table 4.2** *Nutrient inputs and drainage and surface runoff losses resulting from a single irrigation of FDE in September 2003 when the soil was close to field capacity.*

Nutrient form	Effluent input	Nutrient Quantity (kg ha <sup>-1</sup> )		
		Total mole and pipe drainage loss	Total surface runoff loss	Total loss to fresh water body
TN	30.1	5.1	7.5	12.6
TDN	20.8	3.2	5.4	8.6
NH <sub>4</sub> <sup>+</sup> -N	17.6	2.0	4.4	6.4
NO <sub>3</sub> <sup>-</sup> -N	0.02	0.1	0.0	0.1
TP	4.4	0.8	1.1	1.9
TDP	3.0	0.3	0.7	1.0
DIP	2.2	0.22	0.5	0.72

#### 4.4.3 Principles of deferred irrigation

Soil water and nutrient balance models were used to define the key design parameters of deferred irrigation. The key elements of the deferred irrigation system are:

- Effluent is stored in the two-pond system during winter and early lactation when the soil is at or near field capacity.
- Daily weather records and computer simulation models are used to track storage pond volumes and soil moisture deficits (Scotter *et al.* 1979b) to identify opportunities for irrigation.
- Effluent irrigation events are strategically scheduled on occasions when soil water deficits are sufficient to prevent drainage of applied FDE (October – May).
- Four to six irrigation events per year ranging from 10 to 25 mm depth per event are applied at the appropriate soil water deficit. Early season events are likely to be smaller than late summer events as soil water deficits are usually small (<10 mm) in the late winter-spring period.
- Soil and pasture quality is maintained by removing excess nutrients (particularly K) from the soil-plant system in conserved pasture (hay or silage).

One of the most important requirements for the successful implementation of deferred irrigation is the capacity to store effluent produced in early-winter/spring period when soil moisture deficits are small (<10 mm) or non-existent. Soil water balances for the Tokomaru soil were computed for the period 1991 to 2001. In most years, a soil water deficit suitable for the application of FDE (for example, 25 mm) would not develop until at least October. For the period from commencement of lactation to the middle of October, approximately 4000 cubic meters of aerobic pond storage would be required at Massey University's No 4 Dairy farm to accommodate the wash-down water (475 cows requiring approximately 50 litres of water per cow per day) plus the rainfall onto yards, feed pad and ponds minus evaporation from the ponds. This assumes that the ponds are empty at the end of the previous lactation season.

A further key to the successful adoption of a deferred irrigation system is the accurate monitoring of soil moisture status or the soil's ability to store effluent. The alternative to modelling the soil water deficit is the direct measurement of soil moisture contents. There are a number of instruments available - for example, probes using time domain reflectometry measure the soil moisture content *in situ* and can assist farmers in scheduling irrigation of FDE.

Bond (1998) argues that there is likely to be a gap between the design of an effluent irrigation scheme and its implementation. He suggests that effluent irrigation is unlikely to be implemented as prescribed because of imprecision in scheduling techniques, rain interruptions, problems associated with irrigation hardware performance and maintenance, and spatial variability of infiltration rates. Deferred irrigation is a simple tool that seeks to address some of these problems. It is based on the soil water status, and allows for flexibility in the scheduling of irrigation events.

#### **4.4.4 Measuring drainage and nutrient losses under the deferred irrigation system.**

For three lactation seasons (2000/01 to 2002/03), deferred irrigation was implemented at the trial site on Massey University's No 4 Dairy farm for the purposes of both research and paddock-scale demonstration. In the 2000/01 season, 130 mm of FDE was irrigated over six events with an average application depth of 22 mm per application (Table 4.3). This resulted in nutrient applications of approximately 236 kg N ha<sup>-1</sup>yr<sup>-1</sup> and 32 kg P ha<sup>-1</sup>yr<sup>-1</sup>. Whilst some irrigation events generated nutrient enriched drainage from the mole and pipe drainage network, drainage volumes and nutrient losses were greatly reduced by the use of deferred irrigation. Except for two occasions, irrigation of FDE normally generated drainage volumes less than 1% of the total effluent applied (Table 4.3). The average volume of drainage leaving the mole and pipe network as a result of FDE applications was 2.6% of the total effluent applied and resulted in the nutrient loss of approximately 3 kg N ha<sup>-1</sup> yr<sup>-1</sup> and 0.5 kg P ha<sup>-1</sup> yr<sup>-1</sup>.

Where drainage occurred following irrigation of FDE, concentrations of Total N and Total P in the drainage were high. For example, mean concentrations were 44 mg N litre<sup>-1</sup> and 6 mg P litre<sup>-1</sup> on the 6th of December 2000. However, as the practice of deferred irrigation reduced drainage volume, it also minimised the quantity of nutrients that were leached to surface waters. For example, an application of 31 mm of FDE on the 6th of December 2000 resulted in an average loss per plot of 0.61 kg N ha<sup>-1</sup> and 0.08 kg P ha<sup>-1</sup> from an input of 56.5 kg N ha<sup>-1</sup> and 7.7 kg P ha<sup>-1</sup>.

In the 2001/02 season, higher than average spring and summer rainfall meant that soil moisture deficits were smaller than what would be considered typical during the summer-autumn period (i.e. deficit < 100mm) and, therefore, the potential to apply large quantities of effluent in a single irrigation event was very limited (Figure 4.1). In the 2001/02 season, 63 mm of effluent was irrigated over seven events at an average

of 9 mm depth. This resulted in nutrient applications of 95 kg N ha<sup>-1</sup> and 16 kg P ha<sup>-1</sup>. The strategy of irrigating smaller quantities of FDE, more frequently, resulted in zero drainage of applied effluent through the mole and pipe drainage system, and therefore, no direct loss of nutrients.

**Table 4.3** *The mean depth of FDE applied by the irrigator at each of the irrigations, the soil moisture deficit at the commencement of irrigation, the average drainage from all plots over three lactation seasons (2000/01 – 2002/03), and the maximum drainage from any single plot as a proportion of the applied FDE (i.e. representing the 'worst case' scenario).*

Irrigation date	Effluent applied (mm)	Estimated soil moisture deficit at irrigation (mm)	Average drainage (mm)	Maximum drainage from a single plot as proportion of irrigation (%)
23/11/00 <sup>a</sup>	27	66	nd	nd
6/12/00 <sup>a</sup>	31	63	1.4	14
10/04/01 <sup>a</sup>	8	195	0	0
10/04/01 <sup>a</sup>	12	187	0	0
11/04/01 <sup>a</sup>	25	175	0.1	1
16/05/01 <sup>a</sup>	26	117	1.2	6
20/08/01 <sup>b</sup>	9	30	0	0
24/01/02 <sup>b</sup>	7	40	0	0
20/02/02 <sup>b</sup>	9	34	0	0
01/03/02 <sup>b</sup>	6	53	0	0
11/03/02 <sup>b</sup>	10	63	0	0
28/03/02 <sup>b</sup>	13	48	0	0
22/04/02 <sup>b</sup>	9	47	0	0
19/11/02 <sup>c</sup>	9	29	0	0
20/02/03 <sup>c</sup>	24	193	0.2	1.7
13/03/03 <sup>c</sup>	29	220	<0.1	<1
03/04/03 <sup>c</sup>	18	230	<0.1	<1

nd = no data

<sup>a</sup> = 2000/01 lactation season

<sup>b</sup> = 2001/02 lactation season

<sup>c</sup> = 2002/03 lactation season

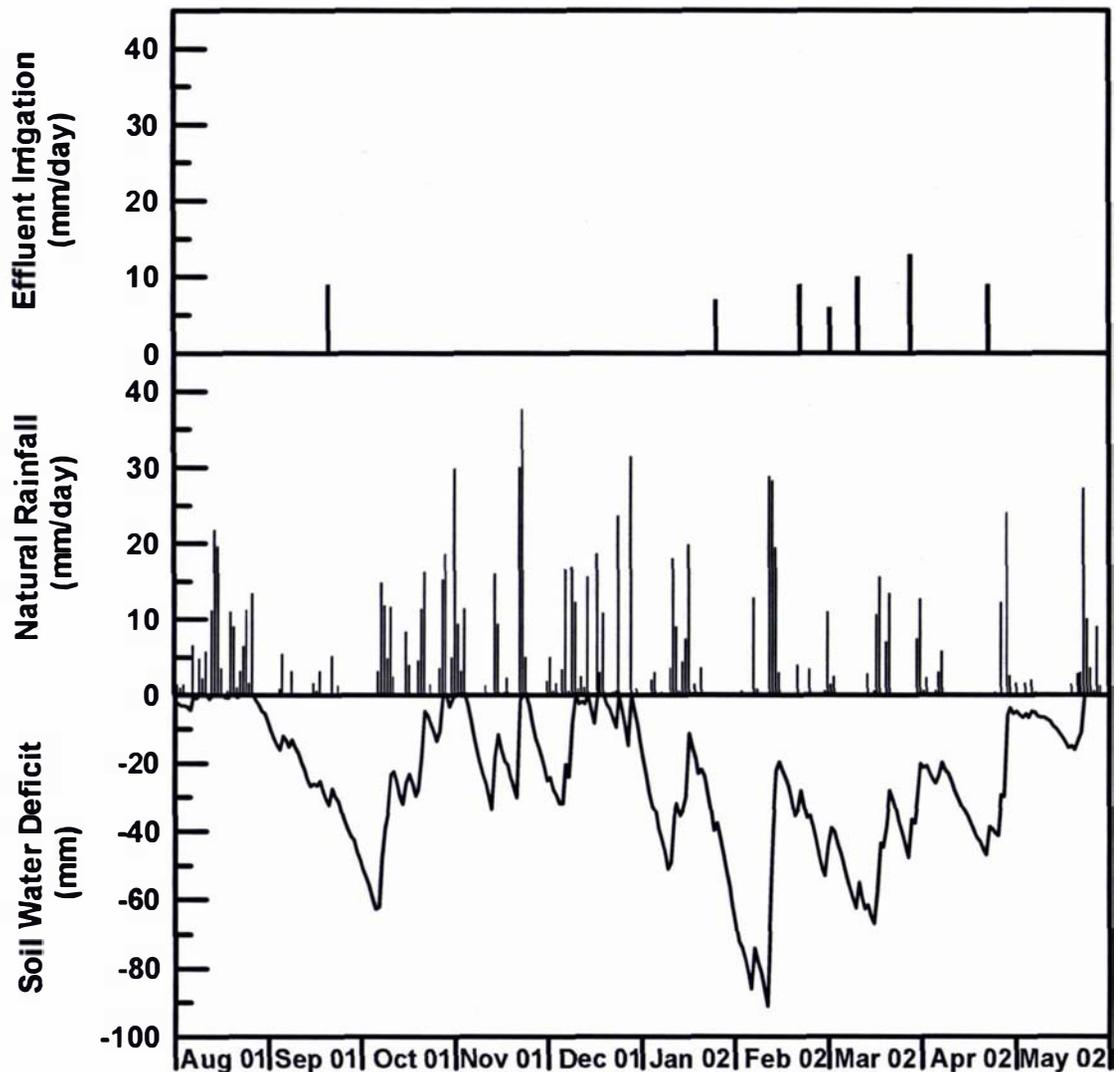


Figure 4.1 The timing of irrigations during the 2001/02 season when regular spring and summer rainfall meant that soil moisture deficits remained small throughout the summer.

In the 2002/03 season, soil moisture deficits were greater than 200 mm due to a prolonged drought. New irrigation technology was being tested and so it was decided to experiment with higher application rates again (Houlbrooke *et al.* 2004). A total of 80 mm of FDE was applied in four separate FDE irrigations at an average application depth of 20 mm per application. FDE irrigation in 2002/03 resulted in nutrient applications of 154 kg N ha<sup>-1</sup> and 31 kg P ha<sup>-1</sup>. On one occasion, a drainage volume equivalent to 1.7% of the effluent applied left the mole and pipe network from one of the plots, and on two further occasions a small amount (< 1%) of the total effluent applied drained from two of the plots (Table 4.3). This small amount of drainage is attributable to preferential flow, most likely through cracks immediately above the

collecting pipe. The average volume of drainage exiting the mole and pipe network as a result of FDE applications was 0.1 % of the total effluent applied for the 2002/03 season. This resulted in a total loss of 0.4 kg N ha<sup>-1</sup> and 0.08 kg P ha<sup>-1</sup>, which was <1% of the TN and TP applied in effluent.

During the 3-year research trial, small amounts of drainage occurred on occasions when the soil moisture deficit at the commencement of the irrigation event was considerably larger than the depth of applied effluent. This drainage can be explained by reference to the variation in application depths under the rotating travelling irrigator and, to a lesser extent, preferential flow under effluent that ponds due to the mismatch between the instantaneous application rate of the irrigator and the soil infiltration rates (Houlbrooke *et al.* 2004 (*Chapter 5*)).

When averaged over all three lactation seasons (2000/01 to 2002/03), FDE application to the soil using the deferred irrigation criteria generated drainage equivalent to 1.1% of the total effluent applied. The average nutrient loss as a result of the direct drainage of FDE following irrigations using the deferred irrigation criteria over three lactation seasons was c. 1.1 kg N ha<sup>-1</sup> and 0.2 kg P ha<sup>-1</sup>. This shows that an improved FDE land application system, such as deferred irrigation, can minimise the environmental risk associated with a daily application system. However, if insufficient storage is available to fully implement deferred irrigation practice, then FDE irrigations should be applied at the lowest rates possible during the critical times of the season to reduce the risk to the aquatic environment.

#### **4.5 CONCLUSIONS**

Daily spray irrigation of FDE can considerably decrease nutrient loss to surface waters compared to a direct discharge of FDE from the two-pond system to a stream. However, daily application of FDE still represents an unacceptable level of nutrient loss to surface waters. Storage of FDE and spray application only when adequate soil water deficits exist (deferred irrigation) has the potential to prevent direct contamination of drainage and surface runoff with FDE. During the three lactation seasons that deferred irrigation was evaluated, direct losses of nutrients to surface waters were almost eliminated, being on average 0.7% of the total N and 0.3% of the P applied as effluent nutrients. The successful adoption of the deferred irrigation system would require only the capability to store effluent and the ability to model or measure soil moisture status.

## 4.6 ACKNOWLEDGEMENTS

The authors are grateful for financial assistance from Dairy InSight, the C. Alma Baker Trust, Marley NZ Ltd, Spitfire Irrigators Ltd and Horizons Regional Council. The support of staff from Massey University's Institute of Natural Resources, No. 4 Dairy Farm and Drainage Extension Service is also gratefully acknowledged.

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# CHAPTER 5: THE PERFORMANCE OF TRAVELLING EFFLUENT IRRIGATORS: ASSESSMENT, MODIFICATION AND IMPLICATIONS FOR NUTRIENT LOSS IN DRAINAGE WATER

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## 5.1 ABSTRACT

Land application of farm dairy effluent (FDE), the treatment option preferred by most regional councils, is commonly practised with the use of small travelling irrigators. Field observations indicate that FDE application by rotating irrigators to artificially drained soils can generate drainage contaminated with partially treated FDE, even when the set application depth is less than soil water deficit. The uniformity of FDE application from rotating, modified-rotating, and oscillating travelling irrigators was determined for a range of application depths and wind conditions.

The rotating irrigator produced a bimodal application profile with a two-to three-fold difference between the highest and lowest application depths. These high application depths are likely to result in drainage of partially treated FDE in late winter and spring when soil moisture deficits are often small. A rotating irrigator was modified with splash

plates or an irrigation bar that diverted more FDE to the centre of the application profile. Neither modification improved the uniformity of application. The most uniform application profiles were obtained using a new technology oscillating irrigator. The measured application profiles and a soil water balance were used to simulate the drainage and nutrient loss under each irrigator type. In early spring and late autumn, when soil water deficits were low, the more uniform application profile of the oscillating irrigator, set at its lowest application depth of 10 mm, created less risk of partially treated FDE reaching pipe drains. The simulation model estimated that when operating at a set average application depth of 25 mm the rotating irrigator and oscillating irrigator required soil water deficits of 44 mm and 32 mm, respectively, to avoid generating drainage. When FDE was applied at 25 mm depth with a deficit of only 18 mm, substantial quantities (30%) of partially treated effluent were estimated to have been drained from the soil no matter which irrigator was used. When application depth equalled the moisture deficit, the more uniform oscillating irrigator had a lower drainage loss (7%) compared to the rotating irrigator (14%). With only a small buffer of 7 mm between soil moisture deficit and application depth it was estimated that the oscillating irrigator achieved zero drainage. When set at their fastest travel speeds, the peak application depths of the rotating and the oscillating irrigators were similar (13 mm) and therefore these irrigators have the same number of operational irrigation days at times when soil water deficits are low.

**Keywords:** farm dairy effluent, travelling irrigators, irrigator uniformity, mole and pipe drainage, preferential flow, nutrient loss

## 5.2 INTRODUCTION

Land application of farm dairy effluent (FDE) is the treatment option preferred by regional councils in New Zealand (Heatley 1996; Cameron *et al.* 1997). However, poorly managed FDE land-treatment systems can have a range of adverse impacts on the aquatic environment: nutrient-rich surface runoff and drainage waters have the potential to pollute surface and ground waters, and high FDE loads may decrease soil quality (Macgregor *et al.* 1979; Heatley 1996; Cameron *et al.* 1997; Bond 1998; Di *et al.* 1998; Singleton *et al.* 2001).

The use of small travelling irrigators is the most common method of applying FDE to land (Heatley 1996). To date, there has been limited research of the performance characteristics of these irrigators, ways to improve the design of these irrigators, and

the operational guidelines required to minimise the risk of irrigated FDE generating polluted drainage and surface runoff. The development of criteria to help in the management of small travelling irrigators has not been straight forward. For example, it has been claimed that in order to maximise the use of nutrients supplied in FDE and minimise the loss of nutrients to receiving fresh water bodies it is important that application depths do not exceed the soil water deficit. However, despite scheduling FDE irrigation when the soil water deficit exceeded the application depth setting of the rotating irrigator, Houlbrooke *et al.* (2004) (*Chapter 4*) and Monaghan & Smith (2004) have observed drainage of partially treated FDE. Also, several farmers have observed discoloured pipe drainage when such irrigators are operating seemingly within appropriate guidelines (D Horne pers.comm).

The objectives of this study are: firstly, to measure and determine the uniformity of FDE application under a standard rotational travelling irrigator and an oscillating travelling irrigator of the types being used to apply FDE on dairy farms in New Zealand, secondly, to explore and quantify the implications of poor application uniformity to nutrient contamination of receiving waters, and thirdly, to describe an attempt to modify a rotational travelling irrigator to improve its application uniformity.

### **5.3 MATERIALS AND METHODS**

The description of the field sites experimental design and site and soil characteristics are already described in this issue by Houlbrooke *et al.* (2004) (*Chapter 4*).

#### **5.3.1 Experimental procedure**

In conjunction with the deferred irrigation research (Houlbrooke *et al.* 2004 (*Chapter 4*)), the performance of two types of small FDE travelling irrigators has been assessed. Travelling irrigators are propelled forward by the ejection of FDE from the rotating arms. In the first two seasons of the study (2000/01 & 2001/02), a 'Briggs Model 15' standard rotating travelling irrigator (hereafter referred to as a 'rotating irrigator') was evaluated. In the third season of this study (2002/03), a 'Spitfire' (mark I) oscillating boom travelling irrigator (hereafter referred to as an 'oscillating irrigator') was evaluated. During the second of these first two seasons the rotating irrigator was modified in an attempt to improve its application uniformity.

During FDE applications, each irrigator was set up to move along a continuous path of travel down the length of the plots, which were arranged in a single row. Adjustable speed settings on each of the irrigators allowed the FDE application depth to be set to a depth within the range of 6 – 36 mm for the rotating irrigator and 10 - 48 mm for the oscillating irrigator. FDE was pumped to the travelling irrigator from the aerobic pond of a standard two pond treatment system.

The depth and the uniformity of application under the irrigators were measured using a series of catch containers (each 430 mm long x 300 mm wide), which were arranged at 2-m spacing in a single row across the pathway of the irrigator. The containers were arranged so as to cover the entire distance from the irrigator to the outer edge of the 'throw', on both sides of the irrigator. Irrigator travel speed was assessed by timing how long it took the irrigator to travel 40 m. The average application rates ( $\text{mm min}^{-1}$ ) of the different irrigators used were calculated using average travel speeds and the longitudinal diameter of the irrigation swath. For example an application depth of 15 mm would be divided by the time required for the irrigator to pass over an irrigation swath width.

A standard rotating irrigator (Heatley, 1996) was used to apply FDE to the experimental area in the 2000/01 season. The irrigator requires an irrigation pressure between 150-250 kPa. A rubber sleeve was fitted to the end of the booms to create a 10 mm nozzle.

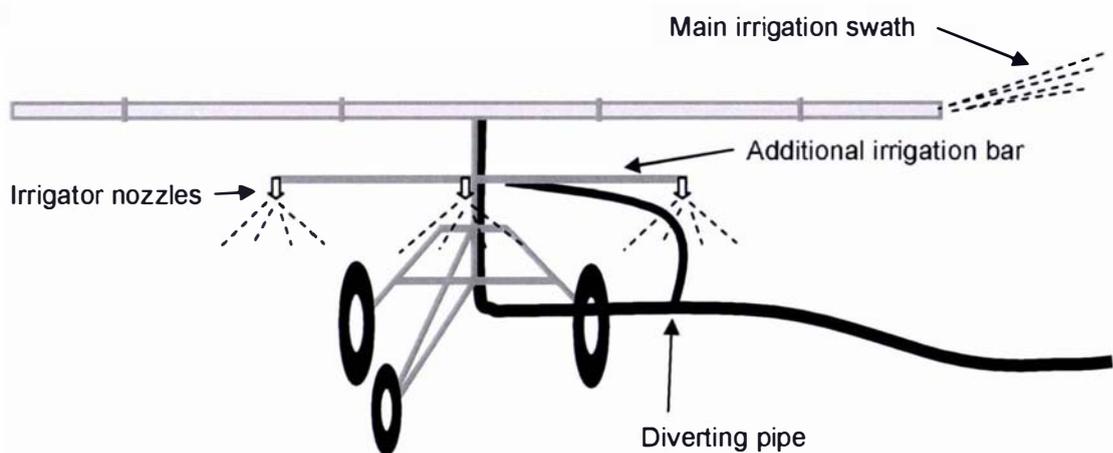
In the 2001/02 season, modifications were made to the rotating irrigator in an attempt to improve its application uniformity. These modifications included using irrigator splash plates or a custom-built irrigation bar, which was designed to divert more FDE to the centre of the application profile. The splash plates were approximately 450 mm long and 80 mm wide with a 45 degree change in angle at the end. They were attached to each end of the rotating booms of the irrigator to help disperse FDE as it came out of the nozzles, thereby widening the application swath width (the width that is instantaneously wet from one pass of the irrigation nozzle) (Plate 5.1). The use of splash plates on the rotating irrigator to disperse or enlarge the irrigation swath, and to improve the application uniformity, was investigated during the 2001/02 irrigation season. A number of different splash plate configurations were studied.

The irrigation bar used in the modifications was attached to the centre frame of the rotating irrigator at a height of 1 m above the ground (Figure 5.1). The bar was 4 m long and had three low-pressure emitters (2 m spacing), one emitter in the middle of

the bar and the other two at each end. Each emitter had an outside spray radius of 2 m, which provided the irrigator bar with a total spray coverage width of 8 m. Approximately  $0.5 \text{ litre sec}^{-1}$ , which equated to 10% of the irrigator's total flow, could be diverted through the irrigator bar without seriously compromising the travel speed of the irrigator.



**Plate 5.1** *Splash plate strapped on to the end of a rotating boom arm of a standard rotating irrigator.*



**Figure 5.1** *Standard travelling irrigator with an additional irrigation bar strapped to the middle of the irrigator. Water is diverted to the bar from the main pipe line.*

The application pattern of a new technology, oscillating travelling irrigator was evaluated, in the 2002/03 season. The oscillating irrigator that was used has a long-

range, flick-over nozzle on a single oscillating 18 mm boom that allows for 180° coverage (Plate 5.2). The irrigator is self propelled, has electronic control of application depth and operates between pressures of 150 to 280 kPa. Because the oscillating irrigator's boom flicks from side to side, without doing a full circumference, it is likely to have a different application pattern to that observed for the rotating irrigator. A governor added to the irrigator was designed to eliminate longitudinal variation in application depth.



**Plate 5.2** Mark 1 'Spitfire' oscillating irrigator in operation with single boom and flick nozzle to achieve 180° coverage.

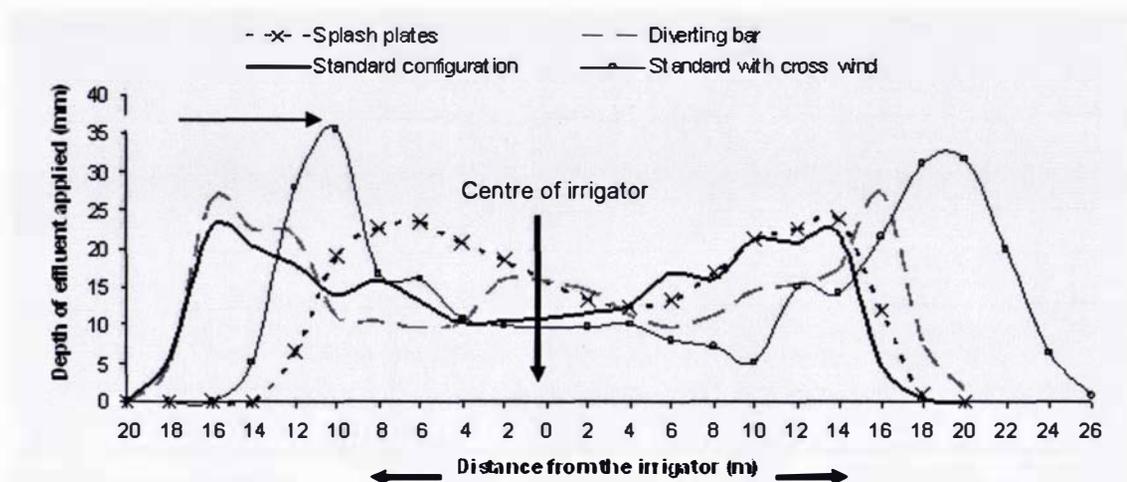
The authors acknowledge that the method used to evaluate the application uniformity of the irrigators did not conform to standardised engineering testing protocols (BS EN ISO 11545:2001). This is because the aim was to operate the irrigators as close to farmer practice as possible in order to assess the influence of application uniformity on direct drainage of partially-treated FDE.

## **5.4 RESULTS AND DISCUSSION**

### **5.4.1 Assessing irrigator performance**

During the first season of this study (2000/01) the rotating irrigator was evaluated. Testing showed that the application depth profile across the path of the rotating irrigator displayed a bimodal distribution with two- to three-fold differences between the highest

and lowest FDE application depths (Figure 5.2). For example, while the average application depth for one irrigation event was 27 mm, the application depths ranged from 17 to 46 mm.

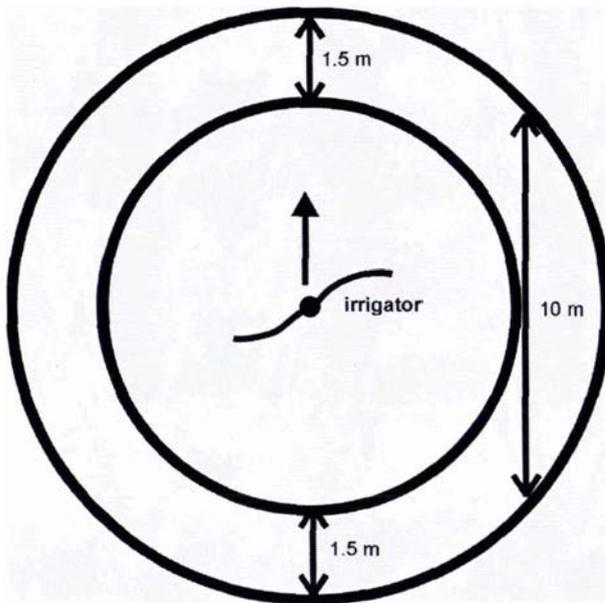


**Figure 5.2** Profiles for the depth of FDE applied by a standard rotating irrigator under relatively calm conditions and cross wind conditions and after modifying the irrigator with either vertical splash plates or a diverting bar. Thin horizontal arrow indicates wind direction for  $4 \text{ m sec}^{-1}$  cross wind profile. Bold vertical arrow indicates the axis of the irrigator.

The bimodal distribution of FDE under the rotating irrigator is explained by reference to the circular pattern of FDE application under this irrigator. FDE emitted from the rotating nozzles is deposited on the soil surface in a circular swath approximately 1.5 m wide (Figure 5.3). Areas at the edge of the irrigators throw (furthest from the line of travel of the irrigator in particular at the inside of the circular swath) are under the irrigators throw pattern (i.e. the swath) for longer periods than areas closer to the irrigator. As a consequence, areas around the circular pattern parallel to the line of travel receive greater depths of FDE.

The bimodal application peaks may result in saturated soil and the drainage of nutrient-rich, partially-treated FDE, which may pass directly to groundwater when soils are free draining, or to surface waters, when soils have artificial drainage systems such as mole and pipe networks. This may occur when the average application depth is less than the soil water deficit. For example, an irrigation with an average application depth of 25 mm of FDE during December 2000 had peak application depths of 55 mm: on this occasion the quantity of drainage of partially treated FDE was equivalent to 14% of the FDE applied (Houlbrooke *et al.* 2004 (Chapter 4)). Therefore, the peak application depth,

rather than the average application depth, should be compared to the soil water deficit when scheduling FDE irrigation by rotating irrigators. This is particularly critical in early spring and late autumn when soil moisture deficits are often low or non-existent and drainage is most likely.



**Figure 5.3** *Circular pattern of FDE application under the rotating irrigator, showing how areas at the outer edge (furthest from the line of travel of irrigator) are under FDE spray swath for longer periods and so receive more FDE.*

Wind causes the rotating irrigator to slow, or stall, which in turn exaggerates peaks in application depth. Most of the testing of the rotating irrigator was conducted under relatively calm conditions. In strong cross winds (perpendicular to the direction of travel), a greater quantity of FDE was deposited down-wind with a resulting four fold difference between troughs and peaks (Figure 5.2). This effect of wind on the application depth means that in certain climates, wind conditions should be a further consideration in the scheduling of effluent application by rotating irrigators.

The problem of variability in application depth is compounded by the high average rate at which the rotating irrigator applied FDE (up to  $5 \text{ mm min}^{-1}$ ). In contrast, the infiltration rate at the site was typically  $0.5 \text{ mm min}^{-1}$ , or smaller where treading damage had occurred. When applying FDE at an average application depth of 15 mm with the rotating irrigator, the average rate of application is equivalent to  $2.2 \text{ mm min}^{-1}$  (Table 5.1). In this situation, a medium amount of surface roughness is required to prevent runoff.

**Table 5.1** Swath width, travel speed and average rate of application for a range of irrigator configurations and types applying farm-dairy effluent at a mean depth of 15 mm.

Irrigator type	<sup>A</sup> Approx. swath width (m)	<sup>B</sup> Approx. travel speed (m min <sup>-1</sup> )	<sup>C</sup> Average rate of application (mm min <sup>-1</sup> )
Rotating irrigator (RI)	3	0.7	2.2
RI plus splash plates	12	0.56	0.8
Oscillating irrigator	2	0.36	2.7

Application rate is calculated by the following equation:  $C=D/(A/B)$ . Where A, B & C are marked columns and D = depth at point of reported swath width (A).

The longitudinal variation in travel speed and application depth was assessed over 160 m and it was found that travel speed and average application depth decreased by 21% and 16% respectively. This is considered minor in comparison to the 300% variation measured in transverse direction. The decrease in travel speed and increase in average application depth is associated with the resistance to travel caused by the increased load from the hose. The wetted width resulting from rotational jets varies inversely with rotational speed (air resistance effect) and the drum diameter on to which the wire cable is being wound increases with distance travelled so that the gear ratio between the 'turbine' and cable vary with distance.

To minimise the occurrence of drainage of partially treated FDE, irrigator application uniformity should be known and application depth should be set so that peak depths do not exceed the soil water deficit.

#### 5.4.2 Modification of existing hardware to improve uniformity of application

Changing the orientation of the plates from horizontal to vertical, so that FDE sprayed upward, decreased the maximum wetted footprint of the irrigator from c. 36 m to 30 m due to the decreased throw capacity of the nozzles. The swath widened from c. 2 m to 6 m (Figure 5.2). However, there was still a two- to three-fold difference between the application depths at the trough and the peaks. The orientation of the splash plates had a marked effect on the travel speed of the irrigator, as the use of splash plates decreased the ground speed of the travelling irrigator. This, in turn, resulted in an increase in the average depth of application at any particular speed setting. Therefore, a major disadvantage of splash plates is that they will reduce the capability of the

irrigator to apply the shallow depths of FDE that are necessary when soil water deficits are low.

Splash plates reduced the average rate of application by the rotating irrigator. An average rate of application of  $0.8 \text{ mm min}^{-1}$  for the rotating irrigator with splash plates is considerably lower than the value of  $2.2 \text{ mm min}^{-1}$  measured under a standard rotating travelling irrigator (Table 5.1). This difference is due to the wider swath created by the use of splash plates.

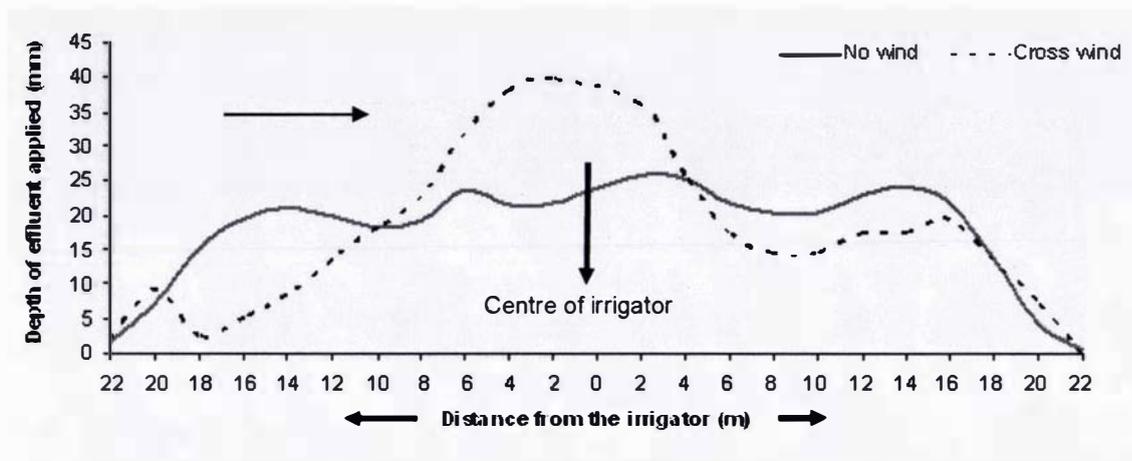
The effect of splash plates on drainage of partially treated FDE is not clear cut. As noted, splash plates did not markedly improve application uniformity. However, as a consequence of lower average application rates, they may reduce the risk of surface runoff and redistribution, and therefore localised ponding and subsequent drainage.

To address the problem caused by low application depths in the centre of the rotating irrigator's application pattern and high depths near the perimeter, a static irrigation bar was added to the centre of the irrigator. FDE was diverted from the delivery hose, before it entered the rotating irrigator arms, to a central horizontal irrigation bar attached to the rotating irrigator's frame, and applied to the centre of the circular application pattern (Figure 5.1). Approximately 10% ( $0.5 \text{ litre sec}^{-1}$ ) of the irrigator's total flow could be diverted through this bar before the travel speed of the irrigator was seriously reduced. Whilst the diverting bar did fill in a modest portion of the gap between the two application peaks (approximately 8 metres), application peaks were still approximately two-to three-fold higher than the lowest point of the trough (Figure 5.2). Fitting the bar did not decrease the peak application depths and hence did not make the irrigator more suitable for applying low depths of FDE at times of low soil moisture deficits. Extending the bar further may have helped decrease the amplitude between peaks and troughs, however this would have made the irrigator unpractical for moving easily around many dairy farms and would have further slowed down the irrigator travel speed resulting in an increase in the minimum application depth.

#### **5.4.3 New technology to improve uniformity of application**

Under most of the wind conditions encountered during testing of the oscillating irrigator, its application pattern was more uniform than that of the rotating irrigator. In relatively calm wind conditions, or when the wind was blowing in the same direction as the line of travel of the irrigator, the application under the oscillating irrigator was relatively

uniform. In calm conditions, the total wetted diameter was approximately 44 metres and the difference between the highest and lowest application depths across the application profile was 25% (Figure 5.4). However, as with the rotating irrigator, the application uniformity under the oscillating irrigator was sensitive to wind conditions. The single oscillating arm slows down on the upwind direction of its movement when a cross wind is blowing. Under such circumstances, greater application depths occur in the middle of the irrigator's application path giving a less uniform profile with a greater than two-fold difference in application depth between the highest and lowest depths (Figure 5.4).



**Figure 5.4** Profile for the depth of FDE applied by the oscillating irrigator under contrasting wind conditions. Thin horizontal arrow indicates wind direction for  $6 \text{ m sec}^{-1}$  cross wind profile. Bold vertical arrow indicates the axis of the irrigator.

During still to light wind conditions, the oscillating irrigator provided a relatively even application pattern and, therefore, if the application depth setting is less than the soil water deficit then the likelihood of direct drainage of applied FDE is low. Compared to the rotating irrigator, the oscillating irrigator had an appreciably smaller swath width, suggesting that the average rate of application for the oscillating irrigator would be substantially higher. However, due to the differences in the mode of application and the slower travel speed, the oscillating irrigator's average application rate ( $2.7 \text{ mm min}^{-1}$ ) was only slightly higher than for the rotating irrigator. This suggests that the likelihood of surface runoff generation from FDE applications with the oscillating irrigator is similar to that for the rotating irrigator.

#### 5.4.4 Predicting drainage of partially-treated FDE induced by different irrigators.

The irrigation patterns measured for a rotating and an oscillating irrigator were used in conjunction with a simple soil water balance to predict the quantity of drainage of partially treated FDE. A range of FDE application depths and soil moisture deficits were studied. In the model, the irrigator application profiles, and consequently the soil profile, were divided into 2-m sections and the application depth of each section was compared independently with the soil moisture deficit. If the application depth in any one section exceeded the soil water deficit, then this difference was counted as drainage. The sum of drainage from all sections provided an estimate of the total volume of drainage resulting from the FDE application. By accounting for the variability in application depth under each irrigator, this model provides a more realistic estimate of the drainage of partially treated FDE than that given by the use of the average application depth.

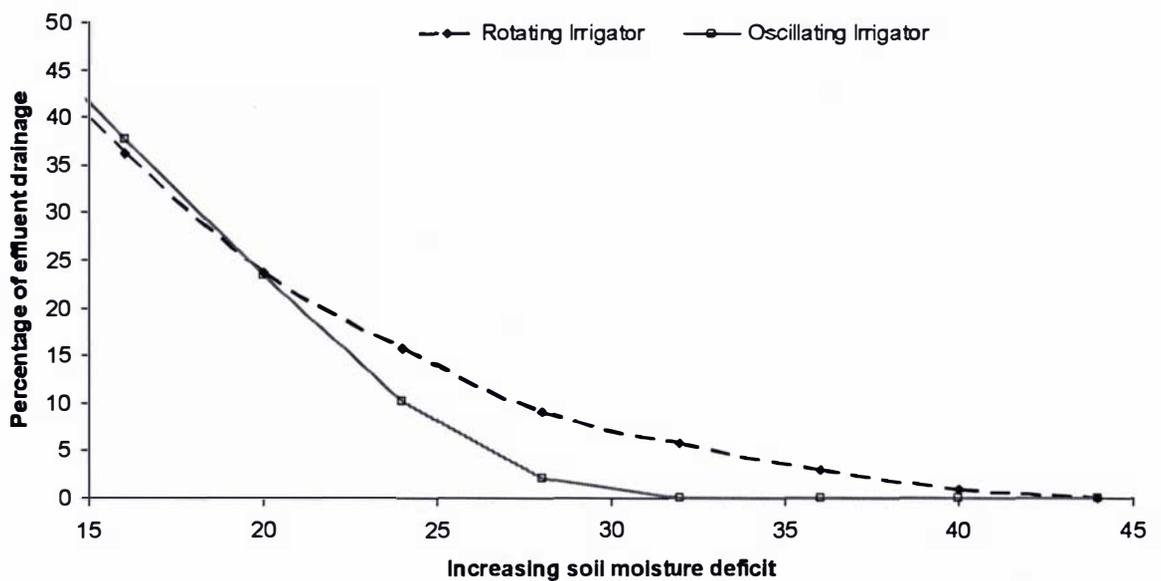
For any application depth, the resulting drainage will depend on both the soil water deficit and the variation in the irrigator's application depth profile. The two irrigator types were compared in the model at an average application depth of 25 mm. Three antecedent soil water deficits (18 mm, 25 mm and 32 mm) were considered in these simulations. The losses of nutrients (Total N and P) in direct drainage of partially treated FDE were predicted using average concentrations of N and P that had previously been measured in direct drainage of FDE by Houlbrooke *et al.* (2004) (*Chapter 4*). Nutrient losses are presented as kg nutrient ha<sup>-1</sup>, even though the size of the area affected is likely to be considerably less than a hectare itself. The drainage simulation presented here is based on a simple soil water balance and so predicts the likelihood of exceeding the stipulated soil moisture deficit at different points across the application profile. It does not account for the possibility of preferential flow through large macropores when high average application rates cause redistribution and surface ponding of FDE. Neither does it distinguish between surface runoff and drainage, therefore, the predicted "drainage losses" incorporate both losses from surface runoff and soil drainage.

The simulation showed that around 30% of FDE drains through the soil when the average FDE application depth (25 mm) exceeds the soil moisture deficit (18 mm), regardless of irrigator type (Table 5.2). Approximately one-third of the applied FDE was predicted to have drained from the soil profile with the associated loss of nutrients.

When the average application depth (25 mm) is equal to the soil moisture deficit of 25 mm, the quantity of drainage is directly related to the relative uniformity of the different irrigator scenarios. Under these circumstances, the drainage volume under the rotating irrigator was equivalent to 14 percent of the applied FDE (Table 5.2). Under the 25-mm soil water deficit, the oscillating irrigator generated drainage equivalent to 7% of the FDE volume applied. The smaller quantity of direct drainage estimated for the oscillating irrigator is due to its relatively uniform distribution pattern.

**Table 5.2** *The predicted direct drainage loss of farm-dairy effluent volume and nutrients under a range of different irrigator and soil moisture scenarios for an average application depth of 25 mm, under relatively calm wind conditions.*

Irrigator scenario	Soil moisture deficit (mm)	% of applied FDE that drains	Predicted drainage loss of N (kg ha <sup>-1</sup> )	Predicted drainage loss of P (kg ha <sup>-1</sup> )
Rotating irrigator	18	29	4	0.62
Rotating irrigator	25	14	1.9	0.29
Rotating irrigator	32	6	0.8	0.12
Oscillating irrigator	18	30	4.1	0.64
Oscillating irrigator	25	7	1	0.16
Oscillating irrigator	32	0	0.0	0.0



**Figure 5.5** *Percentage of FDE drainage likely under a range of different soil water deficits (SWD) following a mean application of 25 mm of FDE from rotating and oscillating irrigators.*

When the soil moisture deficit is increased to 32 mm i.e. 7 mm greater than the average application depth (25 mm), the simulated results for the oscillating irrigator estimated that there would be zero drainage and, therefore, no direct nutrient losses from the soil profile. Under the same circumstances the rotating irrigator resulted in drainage equivalent to 6% of the FDE volume applied.

As a result of its lower application uniformity, the rotating irrigator requires a greater difference between average application depth and soil moisture deficit to achieve zero drainage compared to the oscillating irrigator. For example, at an average application depth of 25 mm, the model estimates that the rotating irrigator requires a soil moisture deficit of 44 mm in order to achieve zero drainage. However, the oscillating irrigator only requires a soil moisture deficit of 32 mm at the same average application depth (Figure 5.5).

#### **5.4.5 Quantifying operational days during early lactation for different irrigators.**

Most irrigators used for the application of FDE have a range of application depths at which they can operate. An assessment was made of the number of days at the start of the milking season (August, September, October) that the rotating and oscillating irrigators could successfully apply FDE, at their respective lowest application depths (i.e. fastest travel speed), while not exceeding the soil water deficit. Calculations were based on modelled water deficits from 2000 to 2003 (the period within which the experimental testing took place).

The oscillating irrigator's lowest application setting provides an average application depth of 10 mm, a maximum application depth of 13 mm, and a minimum depth of 9 mm. The daily soil water balance for the period August to October in the years 2000-2003, indicates that without accounting for wind, there would be a total of 34 days of the 92 in this three month period when FDE could be applied by the oscillating irrigator without exceeding the soil water deficit (Table 5.3). In comparison, the rotating irrigator when set at the lowest application setting provides an average application depth of 6 mm. However, as the application uniformity under the rotating irrigator is more variable, its application depth can also be as high as 13 mm in places. This means that it can not be used on any more days than the oscillating irrigator; however, it would have to irrigate an area 67% larger to apply the same quantity of FDE as the oscillating irrigator at its lowest setting. Whilst the diverting bar option was also similar to both the

oscillating and rotating irrigator, the splash plate option required a considerably higher soil moisture deficit to safely irrigate and hence only 19 days on average were estimated to be suitable for FDE irrigation during the early lactation period.

**Table 5.3** *The estimated minimum soil water deficit (SWD) required to avoid drainage from different irrigator scenarios set at their fastest travel speed and hence lowest application depth. The average days above the minimum SWD is calculated for the period during early lactation (August-October) from deficits calculated for the springs of 2000 - 2003.*

Irrigator scenario	Lowest mean application depth (mm)	Minimum SWD required to achieve zero drainage (mm)	Average days above minimum SWD in early lactation
Rotating irrigator (RI)	6	13	34
RI plus splash plates	15	23	19
RI plus diverting bar	8	15	30
Oscillating irrigator	10	13	34

The overall drawback of the poor application uniformity of the rotating irrigator, compared to the oscillating irrigator, is that at low soil water deficits, the rotating irrigator will need to be set at a lower average application depth (faster travel speed) than the oscillating irrigator to avoid direct drainage of FDE. This means that the rotating irrigator will cover a larger area for the same amount of FDE being applied, compared to the oscillating irrigator.

In general, operating irrigators at their fastest speed (i.e. the smallest application depth) is the best management practice during the early lactation period from August until the end of October as it will minimise the risk of losses of nutrients in the direct drainage of partially treated FDE.

## 5.5 CONCLUSIONS

The spatial variability in the depth of FDE applied by the rotating irrigator was very high and its circular throw pattern creates a 2-3 fold difference between the highest and lowest application depth. Operators unaware of this uneven application pattern are more likely to generate drainage of partially treated FDE. Attempts to improve the performance of these irrigators was made by adding splash plates to the end of the irrigators booms, and by diverting ten percent of the flow of FDE into the low

application area in the middle of the profile. Neither of these modifications to the rotating irrigator significantly improved the bimodal distribution pattern. New technology in the form of an oscillating irrigator was evaluated and found to have a relatively uniform distribution pattern under calm or light wind conditions. The variability in application depth under travelling irrigators needs to be taken into consideration when applying FDE irrigation. A simple model simulating the daily soil water balance, irrigator throw patterns and application depths was used to compare the quantity of direct drainage resulting from FDE applications using the rotating and oscillating irrigator. The more uniform application pattern of the oscillating irrigator offered the best chance of avoiding direct drainage of FDE when the soil moisture deficit was low in early spring and late autumn. During this period of low soil water deficits, all irrigators should be set to travel at their fastest speed (lowest application depth) to minimise the potential for direct drainage of partially treated FDE and associated nutrient losses.

## 5.6 ACKNOWLEDGEMENTS

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# CHAPTER 6: THE IMPACT OF INTENSIVE DAIRY FARMING ON THE LEACHING LOSSES OF NITROGEN AND PHOSPHORUS FROM A MOLE AND PIPE DRAINED SOIL.

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## 6.1 ABSTRACT

While it is widely believed that intensive dairy farming is a major contributor to the increased nutrient loads in surface waters, there is little current research quantifying the magnitude of nutrient loss from dairy farms to waterways, particularly from artificial drainage. An experimental site has been established on a Pallic soil (Tokomaru silt loam) to measure the impacts of intensive dairying on the quality and quantity of drainage water exiting from an artificial drainage system. A key component of this study is the development and evaluation of a land-based treatment system for farm dairy effluent (deferred irrigation). The research site has eight replicated plots, each with an isolated mole and pipe drain network. All the plots are subjected to the farm's standard grazing management. Four of the plots receive fertiliser according to the farm's fertiliser programme, while the other four plots receive applications of farm dairy effluent.

Measurements of drainage flows during year one of this study showed that the average concentrations of total nitrogen (12.9 mg N litre<sup>-1</sup>) and total phosphorus (0.15 mg P litre<sup>-1</sup>) in drainage water for the winter of 2002 under standard dairy farming practices were

all well above the levels necessary to prevent aquatic weed growth in fresh water bodies.

Adherence to the scheduling criteria prescribed by the 'deferred irrigation' system prevented the direct loss of nutrients during irrigation of farm dairy effluent in the summer of 2001/2002. Summer applications of farm dairy effluent did not increase N loss in subsequent winter drainage. Effluent irrigation increased P loss during the subsequent winter drainage period by 0.52 kg total-P ha<sup>-1</sup> (0.38 kg P ha<sup>-1</sup> as DIP). However, this increase in total loss corresponds to less than 4% of the P (16 kg ha<sup>-1</sup>) applied as effluent. Deferred irrigation proved to be a very successful tool for minimising nutrient losses from effluent irrigated areas in direct drainage of effluent at the time of irrigation and subsequent winter drainage. Dairy cattle grazing events also increased nutrient concentrations in drainage waters following grazing by approximately 5 mg total-N litre<sup>-1</sup> (nearly all in the NO<sub>3</sub><sup>-</sup>-N form) and 0.1 mg total-P litre<sup>-1</sup> (nearly all in the DIP form). The effect of an application of urea in spring on NO<sub>3</sub><sup>-</sup>-N concentrations in drainage water was minimal.

**Keywords:** dairying, effluent irrigation, mole and pipe drainage, nitrogen, nutrient leaching phosphorus, water quality.

## 6.2 INTRODUCTION

It is widely believed that intensive dairy farming is a major contributor to the increased nutrient and suspended solid loads in waterways (Sharpley & Syers 1979; Monaghan *et al.* 2002). Accordingly, the dairy farming industry has come under substantial criticism recently for its potential role in the deterioration of the quality of fresh water bodies and streams. In this context, farm dairy effluent is often cited as a serious problem. Poorly managed effluent treatment systems can have a range of adverse environmental impacts: nutrient-rich surface runoff and drainage waters have the potential to pollute surface and ground water and high effluent loads decrease soil quality (Cameron *et al.* 1999; Heatley 1996; Silva *et al.* 1999). In particular, land treatment of dairy shed effluent on imperfectly drained soils is proving a major challenge for many farmers.

To help overcome the problems associated with the disposal of farm dairy effluent, a land based treatment system called 'deferred irrigation' has been developed. Deferred irrigation involves storing effluent in a two-pond treatment system and then applying it

strategically from the aerobic pond when there is a suitable soil water deficit. Soil water balances (e.g. the model developed by Scotter *et al.* (1979)) may be used to determine when this soil water deficit is reached. Scheduling effluent irrigation in this manner reduces the risk of direct surface runoff or drainage, therefore allowing time for the removal of effluent applied nutrients via plant uptake and soil immobilization. In addition, a forage crop is harvested from the irrigated area to minimize the accumulation of nutrients, particularly potassium (K), in the soil-pasture system. Normal farm grazing applies for the rest of the season.

It has been 25 years since Sharpley & Syers (1979) and Turner *et al.* (1979) investigated the impacts of fertilisation and animal grazing events on the quality of water exiting mole and pipe drainage systems on the No.4 Dairy Unit at Massey University, Palmerston North. There has been little work since then despite; the expansion of dairy farming on artificially drained soils, the more intensive nature of modern dairy production system, and the heightened awareness of the risks that polluted drainage water poses for both the environment and on-going market access for dairy produce. The aim of the current research is to measure the impacts of modern, intensive dairying on the quality and quantity of drainage water exiting from artificial drainage systems. In particular, the impacts of deferred irrigation of farm dairy effluent, grazing events and fertiliser additions are assessed.

### 6.3 MATERIALS AND METHODS

A research site to investigate the impacts of intensive dairy farming on drainage water quality has been established on a naturally poorly drained Pallic Soil, the Tokomaru silt loam, at Massey University's No. 4 Dairy Farm, Palmerston North. As part of this study, a sustainable land-based treatment system for farm dairy effluent, called 'deferred irrigation', is being developed and evaluated. The site has 8 replicated plots (40 m x 40 m). Each plot has an isolated mole-pipe drain network. Four of the plots (hereafter called the 'non-effluent' plots) are fertilised and grazed according to the farm's normal management programme (plots 1-4), whilst the other four plots (hereafter called the 'effluent' plots) receive aerobic pond effluent in accordance with the deferred irrigation scheduling criteria (plots 5-8). These plots are also grazed, and have a crop of forage removed annually. At the corner of each plot, a pit has been excavated and a v-notch weir placed at the exit of the pipeline to monitor drainage flow rates, and help facilitate the sampling of drainage events for subsequent measurements of water quality. All pits have been instrumented with data loggers to provide continuous

measurements of flow rate. Comparisons are made of drainage water quality from the two different areas.

Following two seasons of farm dairy effluent application under the deferred irrigation system (2000/01 and 2001/02), monitoring of winter drainage water commenced in late April 2002. Water samples were collected manually. This sampling was scheduled strategically so as to get representative samples from all phases of the flow events. A suite of analyses was carried out including: suspended solids, total phosphorus (P) and total dissolved P (TDP), dissolved inorganic P (DIP), total nitrogen (N),  $\text{NO}_3^-$ -N, and ammonium-N. Analyses were carried out on a Technicon Auto Analyser using the following methods, Kamphake *et al.* (1967) for  $\text{NO}_3^-$ -N, Searle (1975) for ammonium-N, McKenzie & Wallace (1954) for Kjeldahl N and dissolved Kjeldahl N, AOAC (1975) determined by the Vanadomolybdate method for total P and TDP following a Kjeldahl acid digest as described by McKenzie & Wallace (1954) and Murphy & Riley, (1962) for analyses of DIP. Total N and TDN were calculated by adding Kjeldahl N and dissolved Kjeldahl N to  $\text{NO}_3^-$ -N.

On three occasions (mid-June, late-July, and late-September) grazing took place on all the plots approximately 7 to 10 days prior to a drainage event. On a fourth occasion in early-October, only the effluent plots were grazed. The stock density was 80 cows  $\text{ha}^{-1}$  and the grazing period was 12 hours. On 6 September 2002, the effluent plots received 80  $\text{kg ha}^{-1}$  of urea (37  $\text{kg N ha}^{-1}$ ). Additions of P fertiliser were made to both the effluent and non-effluent plots outside of the drainage season, with the effluent plots receiving 22  $\text{kg P ha}^{-1}$  and the non effluent plots 49  $\text{kg P ha}^{-1}$ . MAF quick test values for K and Olsen P were 8 and 41  $\mu\text{g ml}^{-1}$  from the non-effluent plots, and 13 and 48 for the effluent plots.

In the summer of 2001/02, 63 mm of farm FDE was irrigated over the effluent plots. At an average N concentration of 150  $\text{mg N litre}^{-1}$  and an average P concentration of 25  $\text{mg P litre}^{-1}$ , effluent irrigation resulted in nutrient applications of 95  $\text{kg N ha}^{-1}$  and 16  $\text{kg P ha}^{-1}$ .

Deferred irrigation is being implemented and evaluated at the farm system level: use of the plots described above to gather detailed information is an integral part of this larger initiative. The requirement to gather data relevant to the farm scale necessitated the use of a standard travelling irrigator. Therefore, it was impracticable to randomise the 'effluent' and 'non-effluent' treatments across the plots. In other words, the effluent

plots are adjacent to one another. Given this, the variability in measured data is quantified using the standard error of means (SEM).

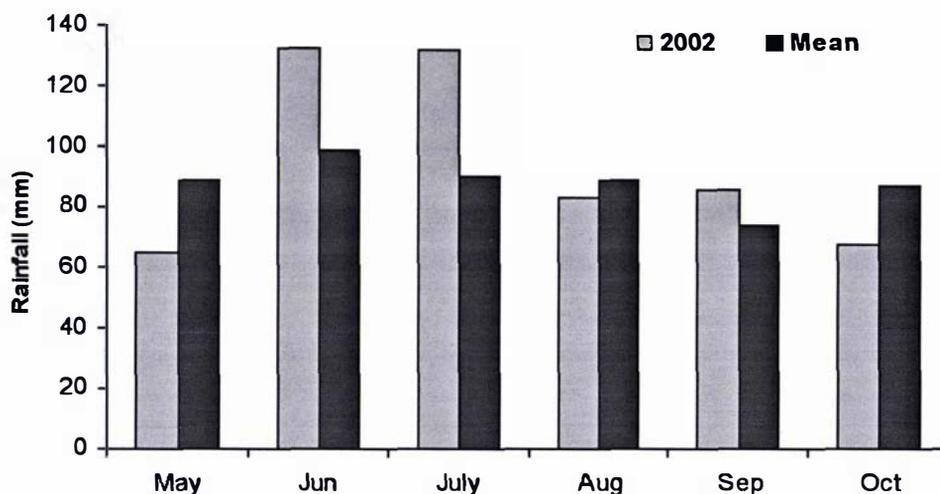
## 6.4 RESULTS AND DISCUSSION

### 6.4.1 Effluent irrigation

The poor uniformity of distribution from a standard rotating travelling irrigator means that the maximum effluent application depth can be no more than half of the soil water deficit. The summer of 2002 was particularly wet and, therefore, soil water deficits were relatively small (between 20 and 80 mm). Consequently, effluent irrigation was spread out over seven events, at an average of 9 mm depth per application. This strategy of irrigation resulted in zero drainage of applied effluent from the mole-pipe system.

### 6.4.2 Drainage quantity

The winter of 2002 was very wet, particularly the months of June and July (Figure 6.1). Throughout the season there were 16 drainage events from the mole-pipe drainage system with a mean total of 220 mm of cumulative drainage. Large drainage events were measured through the wet June and July period, followed by smaller drainage events through spring (Figure 6.2). This was due to the fact that increased evapotranspiration in spring created small soil water deficits that had to be replenished before drainage commenced.



**Figure 6.1.** Fifty year mean monthly rainfall during the winter/early spring period for the Manawatu compared with winter/early spring rainfall during 2002.

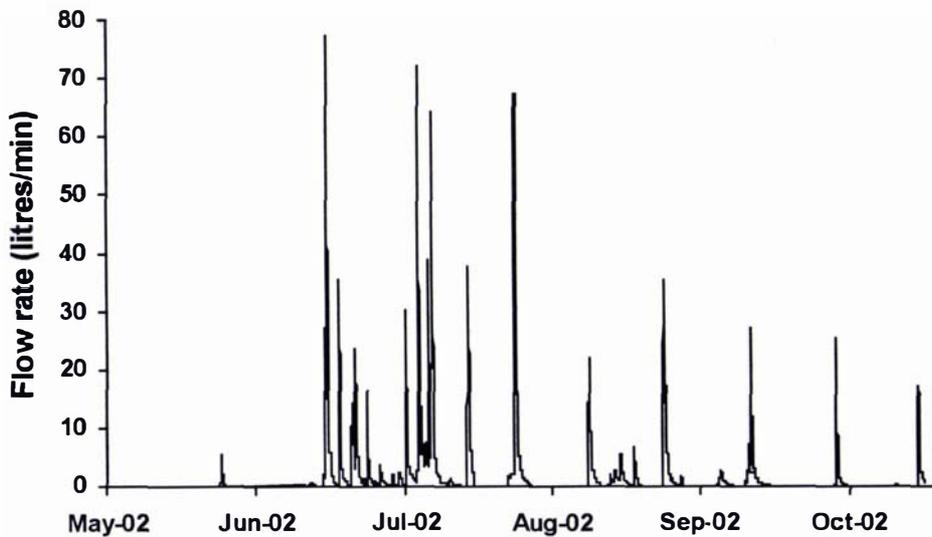


Figure 6.2. Typical hydrograph for one of the plots during the winter/spring period of 2002.

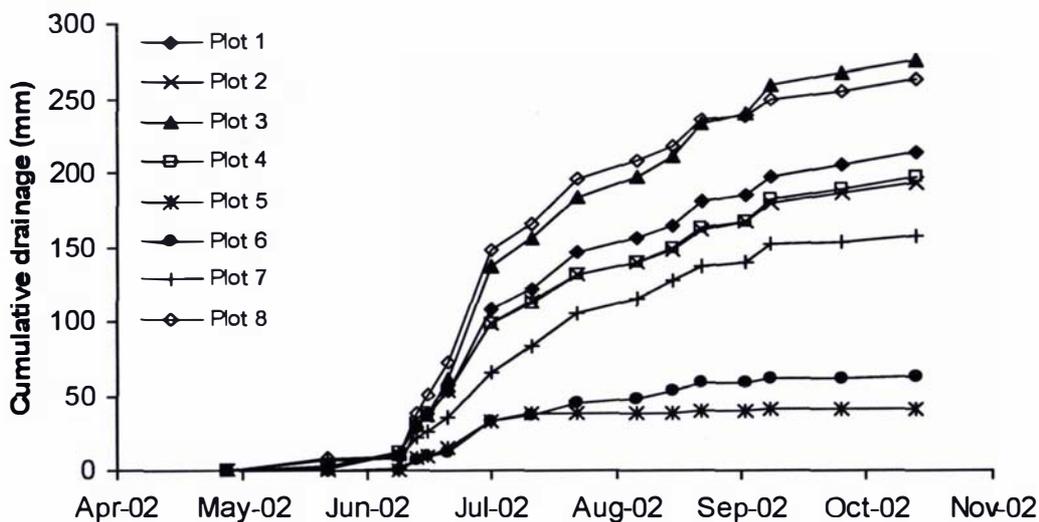


Figure 6.3. Cumulative drainage from the plots in the winter/spring period of 2002.

Flow rates and hence cumulative seasonal drainage from six of the eight plots (plots 1-4 and 7, 8) were variable but were at least 150 mm in total. In comparison, much smaller quantities of drainage were recorded for plots 5 and 6 (Figure 6.3). Analyses of flow patterns indicate that drainage water is being lost from the moles on plots 5 and 6, at all but the greatest of flow rates, before it reaches the pipe drain. One possible cause of this loss is the interception of drainage water in the current mole drains by an

old tile drain. Plots 5 and 6 have been omitted from the calculation of cumulative drainage, and so mean annual loss of nutrients (and SEM) for the effluent treatment is calculated using data from plots 7 and 8. Concentration data was not affected by the low cumulative drainage from plots 5 and 6 and as such were included in the calculation of mean concentrations both within drainage events and throughout the season.

#### 6.4.3 Drainage losses of N

The concentrations of total N in drainage water from all plots were ecologically significant as they were two orders of magnitude greater than the level ( $0.1 \text{ mg N litre}^{-1}$ ) likely to promote aquatic weed growth (Ministry for the Environment, 1992). As much as 99% of the total-N measured in drainage water was in the dissolved form, most (88%) of which was  $\text{NO}_3^-$ -N (Table 6.1). Concentrations and losses of N in the winter drainage of 2002 were slightly greater from the effluent plots than from the non-effluent plots. The increase in total N lost in winter drainage from soils receiving summer applications of farm dairy effluent ( $3.3 \text{ kg N ha}^{-1}$ ) was 3.5% of the N applied in effluent. Nitrate-N concentrations in drainage water displayed a marked trend (Figure 6.4). The first few drainage events of the winter had  $\text{NO}_3^-$ -N concentrations greater than  $20 \text{ mg N litre}^{-1}$ . With increasing cumulative drainage,  $\text{NO}_3^-$ -N levels steadily declined to concentrations of less than  $5 \text{ mg litre}^{-1}$ . A similar pattern of  $\text{NO}_3^-$ -N concentrations in artificial drainage waters was observed by Magesan *et al.* (1994), Heng *et al.* (1991) and Monaghan *et al.* (2002).

The average  $\text{NO}_3^-$ -N concentration of all the drainage water from all plots was greater than the recommend drinking water level of  $11.3 \text{ mg litre}^{-1}$  (Ministry of Health 1995). Approximately 60% of the cumulative drainage water was in excess of the recommended drinking water level.

The summer application of farm dairy effluent, as part of the deferred irrigation system, did not increase the amount or concentration of  $\text{NO}_3^-$ -N in winter drainage water (Figure 6.4 and Table 6.1). The consistent manner in which the mean values for the effluent plots lie above the mean values for non-effluent plots in Figure 6.4 is interesting, and may warrant more detailed investigation in the future.

Grazing resulted in an increase in  $\text{NO}_3^-$ -N concentration of drainage water of approximately  $5 \text{ mg litre}^{-1}$  (Figure 6.4). Sharpley & Syers (1979) found that  $\text{NO}_3^-$ -N

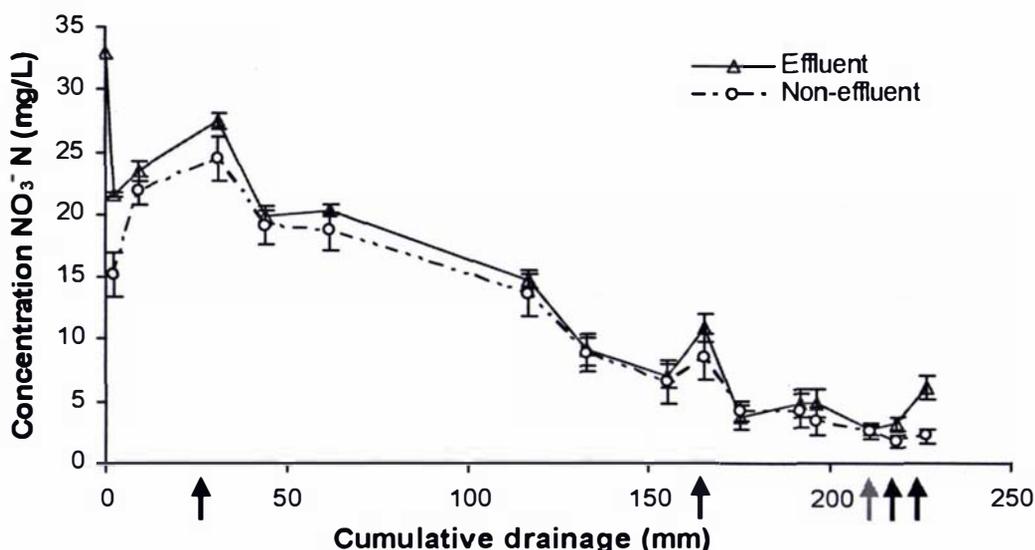
levels in drainage water were elevated by approximately 10 mg litre<sup>-1</sup> following grazing by dairy cattle at a stocking rate of 300 cows ha<sup>-1</sup>. The difference between the results reported here and Sharpley & Syers (1979) suggest that decreasing cattle grazing density may be one way of reducing the concentration of NO<sub>3</sub><sup>-</sup>-N leaching in drainage waters following grazing. The two spring grazing events did not increase the NO<sub>3</sub><sup>-</sup>-N concentrations of drainage water to the same extent as the two mid-winter grazing events. This difference is most likely due to greater plant uptake of NO<sub>3</sub><sup>-</sup>-N in spring compared with winter.

**Table 6.1.** *The amounts and forms of N lost in winter drainage. SEM of annual loss and concentration data are presented in brackets.*

N Form	Treatment	Annual loss (kg ha <sup>-1</sup> )	Concentration (mg litre <sup>-1</sup> )
Nitrate- N	Effluent	26.4 (9.4)	13.7 (1.1)
	Non-effluent	24.5 (2.2)	11.4 (1.4)
Tot Dissolved N	Effluent	30.0 (10.2)	15.8 (1.4)
	Non-effluent	27.2 (2.3)	12.6 (1.5)
Total N	Effluent	31.1 (10.1)	15.9 (1.1)
	Non-effluent	27.8 (2.3)	12.9 (1.5)

There was no increase in the NO<sub>3</sub><sup>-</sup>-N concentrations of drainage water in the drainage event that occurred five days after the application of urea (Figure 6.4). However, as only five days had elapsed between urea application and drainage, most of the N supplied in the urea is likely to have been in the ammonium form and, therefore, less available for leaching. Also, at this time of the year, plant uptake of N would have been increasing as temperatures increased in early spring. It is possible that, along with the early October grazing, the urea application contributed to the increase in NO<sub>3</sub><sup>-</sup>-N concentration in the final drainage event. The five-week period from urea application to the final drainage event would have given sufficient time for most of the ammonium to be converted to NO<sub>3</sub><sup>-</sup>-N.

Ammonium-N results are not reported as in most cases ammonium-N was not found at detectable levels.



**Figure 6.4.** The trends in drainage  $\text{NO}_3^-$ -N concentration with total amount of accumulated drainage, showing the impact of cattle grazing events in both the non-effluent and effluent plots (bold arrows) and an application of urea at a rate of  $37 \text{ kg N ha}^{-1}$  to the effluent plots (grey arrow). Error bars represent two SEM.

#### 6.4.4 Drainage losses of P

The concentrations of total P in drainage water from all plots were ecologically significant, as they were all above the minimum concentration ( $0.1 \text{ mg P litre}^{-1}$ ) considered to stimulate aquatic weed growth (Ministry for the Environment, 1992). Furthermore, considerably more P was leached from the plots that received summer applications of farm dairy effluent (Table 6.2). The increase in total P loss in winter drainage from effluent plots ( $0.52 \text{ kg P ha}^{-1}$ ) equated to approximately 3% of the P applied in summer applications of effluent.

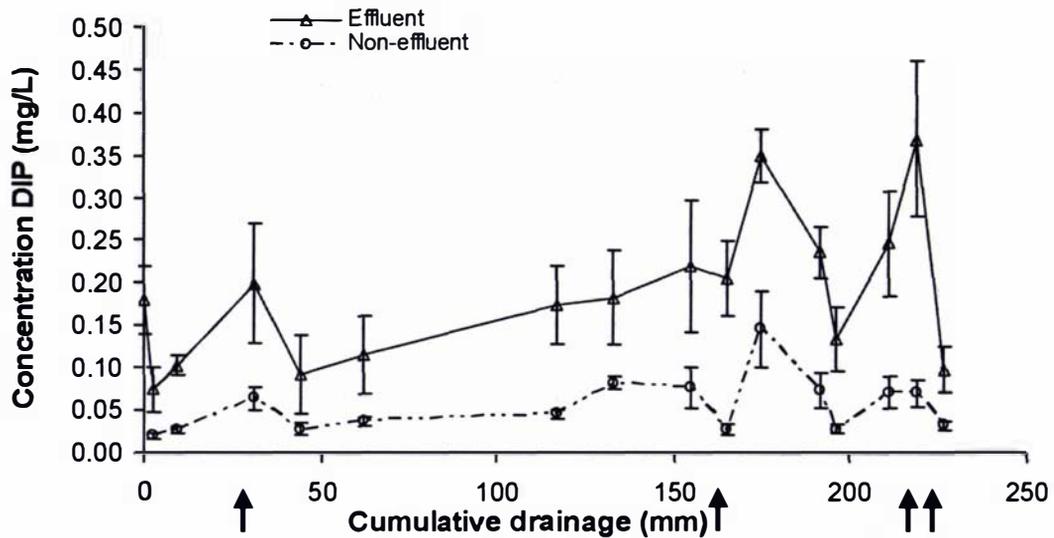
**Table 6.2.** The amounts and forms of P lost in winter drainage. SEM of annual loss and concentration data are presented in brackets.

P Form	Treatment	Annual loss ( $\text{kg ha}^{-1}$ )	Concentration ( $\text{mg litre}^{-1}$ )
DIP	Effluent	0.51 (0.001)	0.18 (0.05)
	Non-effluent	0.13 (0.026)	0.06 (0.01)
Tot Dissolved P	Effluent	0.66 (0.003)	0.27 (0.05)
	Non-effluent	0.24 (0.022)	0.11 (0.01)
Total P	Effluent	0.86 (0.04)	0.35 (0.06)
	Non-effluent	0.34 (0.04)	0.15 (0.01)

Dissolved P made up more than 80% of total P losses (Table 6.2), consequently, less than 20% was in the particulate form, which is the form more commonly found in drainage waters from mole-pipe networks (Sharpley & Syers, 1979; Monaghan *et al.* 2002). This result was supported by the low levels of suspended solids observed in the drainage water (data not presented).

DIP concentrations in drainage water did not exhibit a seasonal flushing trend like that observed for  $\text{NO}_3^-$ -N (section 6.4.3). Monaghan *et al.* (2002) made a similar observation. The average concentration of DIP in drainage waters from the non-effluent plots ( $0.06 \text{ mg litre}^{-1}$ ) is the same as that reported by Sharpley & Syers (1979), and a little higher than concentrations reported by Monaghan *et al.* (2002). Land application of farm dairy effluent consistently increased the concentration of DIP in drainage waters (Figure 6.5). This result is unexpected as the non-effluent plots received  $49 \text{ kg of P ha}^{-1}$  as fertiliser, while the effluent plots received  $38 \text{ kg P ha}^{-1}$  ( $22 \text{ kg P ha}^{-1}$  as fertiliser and  $16 \text{ kg P ha}^{-1}$  as effluent). The difference in Olsen P levels ( $7 \text{ } \mu\text{g P ml}^{-1}$ ) between the effluent and non-effluent plots was not sufficient to account for the difference in DIP concentrations between the two treatments. This suggests that the increase in leaching of DIP from the effluent plots is related, in some way, to the application of farm dairy effluent. Perhaps it is due to the mineralisation of P from residual effluent in the soil-mole drainage network. Further monitoring over time is required.

The impact of cattle grazing on P losses was evident throughout the drainage season. In most cases, the concentration of DIP in drainage water increased by approximately  $0.1 \text{ mg litre}^{-1}$  following a grazing event (Figure 6.5). However, compared to the  $\text{NO}_3^-$ -N data, increases in DIP concentrations in drainage water following grazing were not as consistent in either magnitude or timing. The effects of the June and September grazing events on DIP concentrations were evident in the following drainage event, whilst the effect of the July grazing event was not evident until the second drainage event following grazing. The October grazing did not increase DIP concentrations. However, the low flow rate measured during this event would have provided an increased opportunity for P absorption.



**Figure 6.5.** *The impacts of cattle grazing events and effluent irrigation on DIP concentrations. (Bold arrows positioned on the x-axis represent the timing of grazing events relative to cumulative drainage). Error bars represent two SEM.*

Sharpley & Syers (1979) found that DIP concentrations in drainage waters increased approximately 15 fold following intensive grazing (300 cows ha<sup>-1</sup> for 12 hours) of pastures by dairy cattle. Sharpley & Syers (1979) measured peak concentrations of 0.25 mg P litre<sup>-1</sup> as DIP, which were considerably higher than the peak concentration of 0.15 mg DIP litre<sup>-1</sup> measured in the grazed (80 cows ha<sup>-1</sup> for 12 hours) non-effluent plots in the current study. As for NO<sub>3</sub><sup>-</sup>-N, these results suggest that decreasing cattle grazing density during winter may be an effective way of decreasing the impact of grazing events on DIP concentrations in drainage water.

## 6.5 SUMMARY AND CONCLUSIONS

The average concentrations of N and P in winter mole-pipe drainage water from grazed dairy pastures were all well above the levels required to prevent aquatic weed growth in fresh water bodies. A number of common dairy farm practices increased the losses of N and P in the artificial drainage water. Recent grazing events increased NO<sub>3</sub><sup>-</sup>-N and DIP concentrations in drainage by approximately 5 mg litre<sup>-1</sup> and 0.1 mg litre<sup>-1</sup>, respectively. The impact of an early spring application of urea on drainage water quality was minimal, suggesting that careful timing of urea applications at this time of year may result in little nitrogen leaching.

Effluent irrigation in the previous summer caused no real increase in total N concentration or loss in winter drainage water. Effluent irrigation resulted in a substantial increase in total P losses in winter drainage (0.52 kg P ha<sup>-1</sup>). However, this increase represents less than 4% of the P applied in the effluent. By adhering to stringent scheduling criteria, such as those developed in the deferred irrigation system, it is possible to prevent the direct loss of nutrients following irrigation of farm dairy effluent and thus minimise nutrient leaching losses during winter.

## 6.6 ACKNOWLEDGMENTS

The research team is grateful for financial assistance from the C. Alma Baker Trust, Fonterra Research through the Dairying for the Environment Scheme, Marley NZ Ltd, Spitfire Irrigators Ltd and Horizons Regional Council. The support of staff from Massey University's Institute of Natural Resources, No. 4 Dairy Unit and Drainage Extension Service is also gratefully acknowledged.

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# CHAPTER 7: THE EFFECTS OF DAIRY FARM MANAGEMENT PRACTICES ON NUTRIENT ENRICHMENT OF MOLE AND PIPE DRAINAGE AND SURFACE RUNOFF.

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## 7.1 ABSTRACT

It is often claimed that dairy farming is a significant contributor to the nutrient enrichment of New Zealand's aquatic environment. However, there is relatively little quantitative information on nutrient loss in either drainage or surface runoff from dairy farms on artificially drained soils. A research trial was set up in the Manawatu region, New Zealand to investigate the impact of some dairy farm management practices on the quantity and quality of water exiting a soil as artificial drainage or surface runoff. The experimental site was established on a Pallic soil (Tokomaru silt loam) at the No. 4 dairy farm at Massey University, Palmerston North. There were eight plots (each 40 m x 40 m): each with an isolated mole and pipe drainage network. Four of the plots received fertiliser according to the farm's fertiliser programme, while the other four plots received applications of farm-dairy effluent (FDE). Four surface runoff plots (each 5 m x 10 m) were established as subplots (two on the fertilised plots and two on the plots irrigated with farm dairy effluent (FDE)). All of the plots were subject to the farm's standard grazing management.

A number of dairy farm practices increased the losses of N and P in both artificial drainage and surface runoff in the winter-spring of 2003. The average concentrations of N and P in the drainage (236 mm) and the surface runoff (41 mm) from both the fertilised and effluent irrigated plots were well above the threshold concentrations that stimulate aquatic weed growth in fresh water bodies. Annual nutrient losses of 34.1 kg N ha<sup>-1</sup> yr<sup>-1</sup> and 0.73 kg P ha<sup>-1</sup> yr<sup>-1</sup> in drainage and 3.0 kg N ha<sup>-1</sup> yr<sup>-1</sup> and 0.6 kg P ha<sup>-1</sup> yr<sup>-1</sup> in surface runoff were recorded for fertilised plots. Increases in NO<sub>3</sub><sup>-</sup>-N concentration of approximately 5 mg litre<sup>-1</sup> were recorded for two separate drainage events occurring ten days after cattle grazing events. A urea application (37 kg N ha<sup>-1</sup>) in mid June, when plant uptake of mineral N was low, resulted in increased concentrations of NO<sub>3</sub><sup>-</sup>-N (ranging from 3 to 9 mg N litre<sup>-1</sup>) in the following 80 mm of cumulative drainage.

Application of FDE to near-saturated soil in mid September resulted in the direct drainage and surface runoff of partially treated effluent and hence N and P losses that were six to ten fold greater than would normally be expected from surface runoff and drainage events induced by winter-spring rainfall. This illustrates the importance of scheduling FDE irrigation. Applying FDE in the preceding summer according to the deferred irrigation criteria resulted in no increase in the loss of N in winter drainage. In contrast, FDE irrigation did result in a substantial increase (50%) in total P losses in winter drainage (an additional 0.74 kg P ha<sup>-1</sup>). Although this increase is relatively high in relation to total P loss, it represents less than 2.5% of the P originally applied in the summer FDE applications.

**Keywords:** effluent irrigation, artificial drainage, nitrogen, phosphorus, nutrient leaching.

## 7.2 INTRODUCTION

Intensive dairy farming has undergone considerable expansion in New Zealand over the past ten years with a 44% increase in national dairy cow numbers (Livestock Improvement 2003). Much of this expansion has occurred on artificially drained soils. While there is considerable data showing that surface runoff water and drainage from free draining soils under grazed dairy pasture can contribute to accelerated nutrient enrichment of waterways (Nash and Murdoch 1997; Di and Cameron 1998; Silva *et al.* 1999; Di and Cameron 2002a; Di and Cameron 2002b; Gillingham and Thorrold 2000; McLay *et al.* 2001; Smith and Monaghan 2003) there has been relatively little research of the impacts of land application of farm dairy effluent (FDE), intensive grazing of dairy

cattle, application of urea fertiliser, and maintenance of a high soil nutrient status on the quality of artificial drainage water. In particular, the losses of P under intensively grazed dairy pasture have been poorly documented as much of the research conducted has concentrated on the losses of  $\text{NO}_3^-$ -N.

Excreta from grazing dairy cattle provide the greatest potential pool of nutrients for leaching. Cattle excreta can return up to 90% of the nutrients ingested in pasture (Haynes and Williams 1993). Urine patches, in particular, provide large quantities of highly soluble urea-N (Silva *et al.* 1999; Di and Cameron 2002a). The N in urine patches is deposited at rates up to  $1000 \text{ kg N ha}^{-1}$  (Silva *et al.* 1999). Urine patches are deposited randomly across grazed paddocks and, at a stocking rate of 3 cows  $\text{ha}^{-1}$ , cover ~ 25% of the paddock area in one year (Silva *et al.* 1999). Urine N rapidly hydrolyses to the plant available ammonium-N form which may, in turn, be converted to highly mobile  $\text{NO}_3^-$ -N within 10 days (Condon *et al.* 2004). Likewise, cattle dung provides a source of soluble P that is susceptible to drainage and runoff (Sharpley and Syers 1979, McDowell and Sharpley 2002). Furthermore, treading damage by cattle at the time of grazing disturbs the soil surface resulting in the potential loss of particulate P in sediment transported by surface runoff (Gillingham and Thorrold 2000; McDowell *et al.* 2003a, Smith and Monaghan 2003).

On New Zealand dairy farms, it is common practice to maintain a high soil nutrient status using fertilisers. In particular, urea is applied at key times to increase pasture growth. Sharpley and Syers (1979) found that an application of urea in late June resulted in a prolonged period of increased levels of  $\text{NO}_3^-$ -N in drainage from a Tokomaru silt loam soil in the Manawatu. At a nearby site, Turner *et al.* (1979) found that the addition of P fertiliser resulted in increased losses of dissolved forms of P, particularly dissolved inorganic P (DIP). They found that up to 1.8% of the applied P was lost in dissolved form in subsequent drainage events (Turner *et al.* 1979). McDowell *et al.* (2003b) suggest that the environmental risk of P loss in surface runoff and drainage waters will be reduced if P fertiliser additions are managed so that Olsen P concentrations are at or below agronomic optimum levels. In Victoria Australia, Nash and Murdoch (1997) reported that 69% of P loss in surface runoff on fertile dairy pasture occurred during large storm events. Erosion or movement of surface soil particles was not believed to be the major process by which P loss occurred as 91% of the total  $3.2 \text{ kg P ha}^{-1}$  lost was in the dissolved form.

Land treatment of FDE, whilst the preferred option of New Zealand regional councils (Heatley 1996, Cameron and Trenouth 1999), can generate nutrient-rich surface runoff and drainage waters if poorly managed (Houlbrooke *et al.* 2004a (*Chapter 4*)). High effluent loads have the potential to pollute surface and ground waters and degrade soil quality (Heatley 1996; Cameron *et al.* 1997; Sukias *et al.* 2001; Tillman and Surapaneni 2002; Houlbrooke *et al.* 2004b (*Chapter 2*)). In particular, application of FDE to imperfectly drained soils is proving a major challenge for many farmers (Houlbrooke *et al.* 2004a (*Chapter 4*); Monaghan and Smith 2004) as a result of rapid transport of contaminants through mole and pipe drainage systems or via surface runoff. To help farmers minimise the quantity of partially treated FDE that is lost from artificially drained soils, a system called deferred irrigation has been developed (Houlbrooke *et al.* 2004a (*Chapter 4*)).

This paper reports research designed to measure the influence of intensive dairy farming on the quality and quantity of water exiting grazed dairy pasture as surface runoff or artificial drainage. The influence of grazing events, urea additions and the land application of FDE using deferred irrigation are assessed and compared with data collected for different management and climatic conditions (Sharpley and Syers 1979, Monaghan *et al.* 2002, Houlbrooke *et al.* 2003 (*Chapter 6*)).

## **7.3 MATERIALS AND METHODS**

### **7.3.1 Site and soil**

The research site was established on a mole-pipe drained Tokomaru silt loam, a Fragic Perch-gley Pallic Soil (Hewitt 1998) or Typic Fragiaqualf (Soil Survey Staff 1998) on Massey University's No. 4 dairy farm in the Manawatu, New Zealand (NZMS 260, T24, 312867). A more detailed description of the soil characteristics is given in Scotter *et al.* (1979a).

### **7.3.2 Experimental design**

Eight grazed pasture plots (40 m x 40 m) were established in 2000 to measure drainage and surface runoff volumes. Each plot had a hydrologically isolated mole-pipe drain network, whilst half of the plots had an additional surface runoff subplot. Mole drains were installed in January 2000 at 2-m intervals and at a depth of 0.45 m.

Drainage from the mole system was collected by a perforated pipe drain (0.11 m in diameter) that was perpendicular to the mole drains at 0.6 m depth.

As part of a larger research program, a sustainable land-based treatment system for FDE, called 'deferred irrigation', was developed and evaluated (Houlbrooke *et al.* 2004a (*Chapter 4*)). Four of the plots (plots 5-8; hereafter called the 'effluent' plots) received aerobic pond effluent in accordance with the deferred irrigation scheduling criteria. These plots were grazed according to the farm's normal management programme and, as part of the deferred irrigation system, had a crop of baleage removed annually. Another four of the plots (plots 1-4, hereafter called the 'non-effluent' plots) received conventional fertilisers (mostly urea in 2003) and were also grazed according to the farm's normal grazing regime. Each treatment of four plots was contained within one paddock to simplify grazing management.

At the corner of each plot, a pit was excavated and a v-notch weir placed at the exit of the pipeline to monitor drainage flow rates and facilitate the sampling of drainage events for subsequent measurements of water quality. All pits were instrumented with data loggers to provide continuous measurements of flow rate. Isolated sub-plots (each 50 m<sup>2</sup>) were installed on two of the plots from each treatment (plots 2, 4, 7, 8) for the measurement of surface runoff. Surface runoff water collected from each sub-plot passed through a tipping bucket instrument to measure flow rate and to provide a flow proportioned, mixed sample for water quality analyses. All pits and runoff collector trays were fenced off from stock.

### 7.3.3 Experimental procedure

The research area at the No.4 dairy farm has been grazed by lactating dairy cattle for 25 years prior to this research. The effluent plots had received applications of FDE in accordance with deferred irrigation criteria (Houlbrooke *et al.* 2004a (*Chapter 4*)) in the three summers prior to the 2003 winter drainage season. In the summer of 2002/03, a total 80 mm of FDE was irrigated over the effluent plots. Summer irrigation of FDE applied 154 kg N ha<sup>-1</sup> and 31 kg P ha<sup>-1</sup>. The concentrations of N and P in the applied FDE ranged from 124-234 mg N litre<sup>-1</sup> and 29-43 mg P litre<sup>-1</sup>.

In an attempt to quantify the impact of a poorly timed effluent application on the quantity and quality of drainage and surface runoff water, 25 mm of FDE was applied to the effluent plots on the 15 September 2003 when the soil moisture deficit was only 7

mm for the top 800 mm. In this single irrigation event, 30 kg N ha<sup>-1</sup> and 4 kg P ha<sup>-1</sup> were applied to the soil. The day after this poorly timed FDE irrigation there was 12 mm of rainfall which produced further drainage.

Cattle grazing took place at the research site throughout the summer/autumn period of 2002-2003 with an average frequency of approximately 21 days. During the winter/spring drainage period of 2003, grazing events lasted 12 hours with an average stocking rate of 100 cows ha<sup>-1</sup> (range 70-129 cows ha<sup>-1</sup>). One exception was on the 29 August 2003 (non-effluent plots) when 20 cows ha<sup>-1</sup> grazed for 72 hours. Urea (35 kg N ha<sup>-1</sup>) was applied to non-effluent plots on the 23 April and 26 May 2003 before drainage had commenced for the season. On 12 June 2003, the non-effluent plots received 37 kg N ha<sup>-1</sup> (80 kg ha<sup>-1</sup> of urea) and a further 35 kg N ha<sup>-1</sup> (75 kg ha<sup>-1</sup> of urea) on the 9 October 2003. No P fertiliser was applied to effluent or non-effluent plots in the 2002/2003 season. Average Quick Test values measured in autumn 2003 (Cornforth and Sinclair 1984) for K and Olsen P were 8 and 41 µg ml<sup>-1</sup> for the non-effluent plots, and 13 and 48 for the effluent plots, respectively.

Deferred irrigation of FDE was implemented and evaluated at the farm system level (Houlbrooke *et al.* 2004a (*Chapter 4*)). The plots described above were an integral part of this larger study. The requirement to gather data relevant to the farm scale necessitated the use of a standard travelling irrigator. Therefore, it was impracticable to randomise the 'effluent' and 'non-effluent' treatments across the plots. In other words, the effluent plots are adjacent to one another (as are the non-effluent plots). Given this, the variability in measured data is quantified using the standard error of means (SEM).

Soil water deficits and the sum of drainage and surface runoff (i.e. no differentiation between these two pathways of water movement) for Tokomaru silt loam were predicted using the soil water balance model and parameters of Scotter *et al.* (1979b).

#### **7.3.4 Nutrient analyses**

A suite of water quality analyses were conducted on drainage and runoff samples captured from each of the experimental plots throughout the 2003 drainage season. The analyses included: suspended solids, total phosphorus (P) and total dissolved P (TDP), dissolved inorganic P (DIP), total nitrogen (N), total dissolved nitrogen (TDN), nitrate-N (NO<sub>3</sub><sup>-</sup>-N), and ammonium-N (NH<sub>4</sub><sup>+</sup>-N). Total N and TDN were calculated by adding NO<sub>3</sub><sup>-</sup>-N values to those of Kjeldahl N and dissolved Kjeldahl N. Analyses were

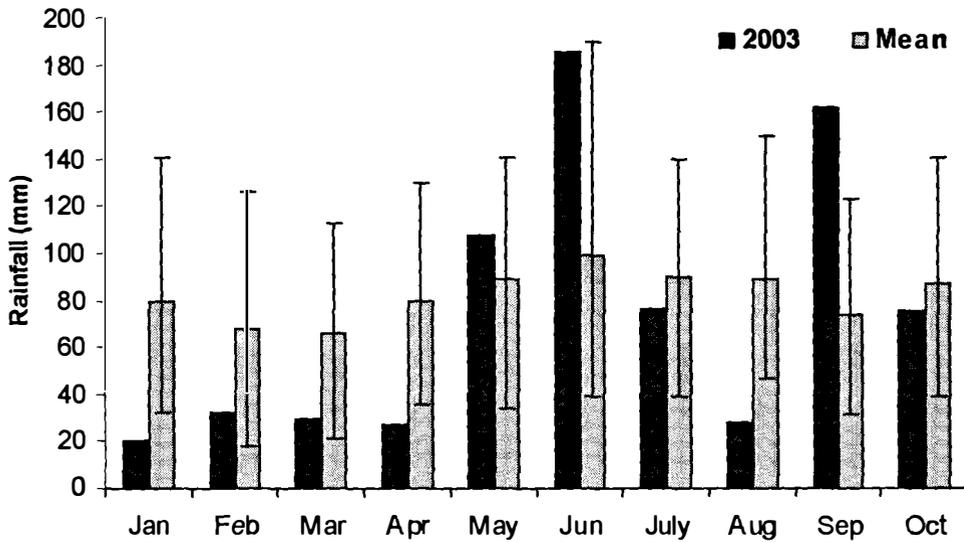
carried out colorimetrically on a Technicon Auto Analyser II using the following methods: Kamphake *et al.* (1967) using a hydrazine reduction for  $\text{NO}_3^-$ -N; Searle (1975) for ammonium-N; McKenzie & Wallace (1954) for Kjeldahl N and dissolved Kjeldahl N following a Kjeldahl acid digest; and Murphy & Riley (1962) for analyses of DIP. Total P and TDP were determined by the Vanadomolybdate method (AOAC 1975) following a Kjeldahl acid digest as described by McKenzie & Wallace (1954). All measures of dissolved nutrients were determined on samples passed through a 1.2  $\mu\text{m}$  glass filter paper.

All treatment concentrations presented on an event basis are derived from the mean of each independent plot, calculated from a series of flow weighted samples collected from each event. Total nutrient losses presented are calculated using individual plot concentrations and the mean volume of drainage for all plots concerned. This allowed the effects of treatment on concentrations to be the dominant factor in calculations of nutrient loss rather than inconsistencies in plot drainage volumes. Further justification based on variable flow data is presented in appendix 1.

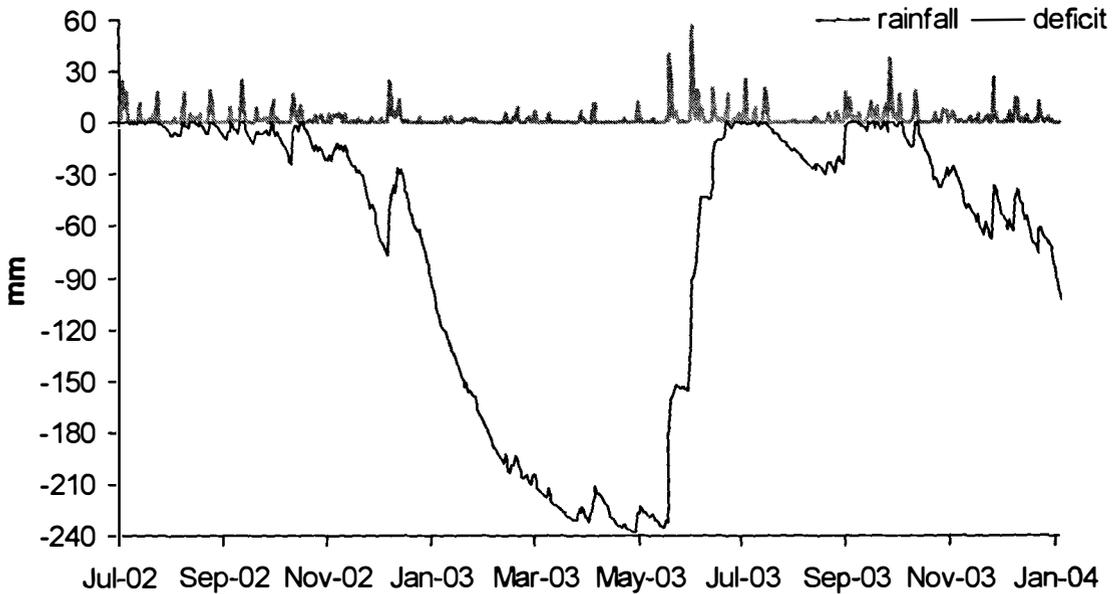
## **7.4 RESULTS AND DISCUSSION**

### **7.4.1 Drainage and runoff quantity**

The summer-autumn period of 2003 was one of the driest on record (Figure 7.1): the soil water balance predicted a soil water deficit in excess of 200 mm (Figure 7.2). A series of very large storms (40 mm on 21 May, 29 mm on 22 May and 57 mm on 3 June) in late autumn rewet the soil (Figure 7.2). In June and July, there were substantial quantities of rainfall, drainage and runoff (Figure 7.1, 7.3). This wet period was followed by a 6 week interval - from mid July to the end of August - in which there was very little rainfall and no drainage or surface runoff. September had more than twice the average rainfall (Figure 7.1), and frequent rain events continued into early October (Figure 7.2).



**Figure 7.1.** Rainfall at the research site during 2003 compared with the fifty-year mean monthly rainfall for the January - October period for the Manawatu. Error bars represent the 90% upper percentile and 10% lower percentile of mean monthly rainfall.



**Figure 7.2** Simulated soil water deficit using a soil water balance model (Scotter et al. 1979b) at the research site for an 18 month period from July 2002 to January 2004.

Flow rates and hence cumulative seasonal drainage from six of the eight plots (plots 1-4 and 7, 8) were variable but relatively uniform and consistent with the predicted water balance. In comparison, plots 5 and 6 generated smaller quantities of drainage (Figure 7.4). Analyses of cumulative drainage from each plot indicates that drainage water was



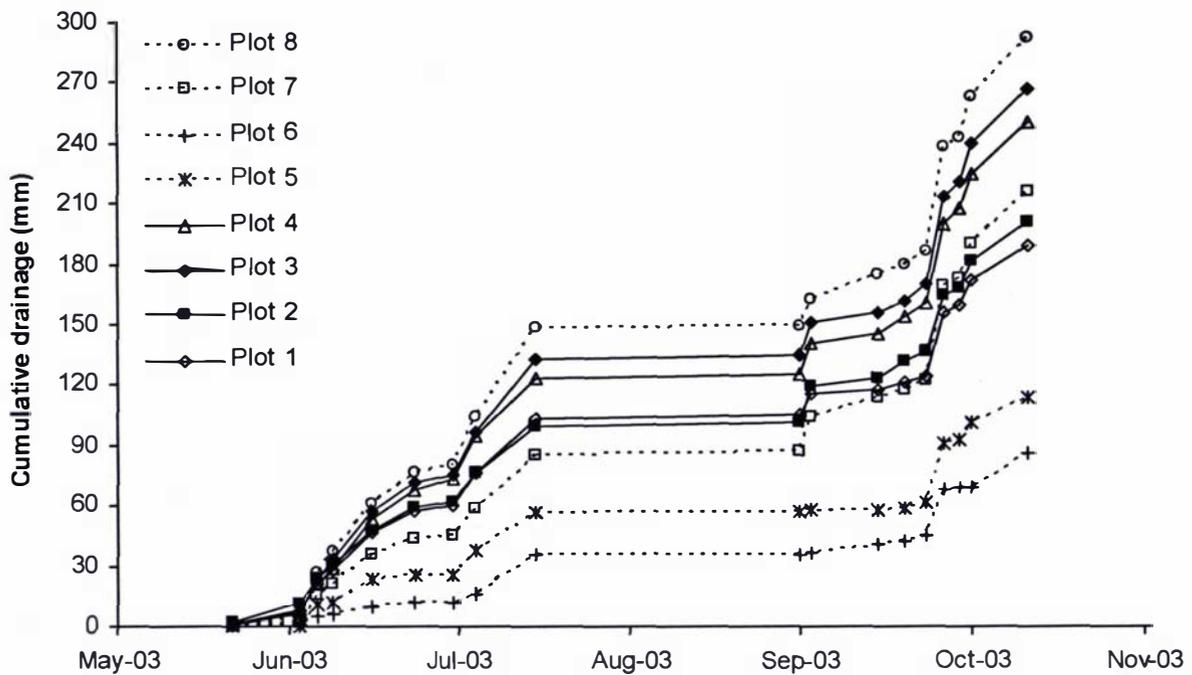


Figure 7.4 Cumulative drainage from the plots in 2003.

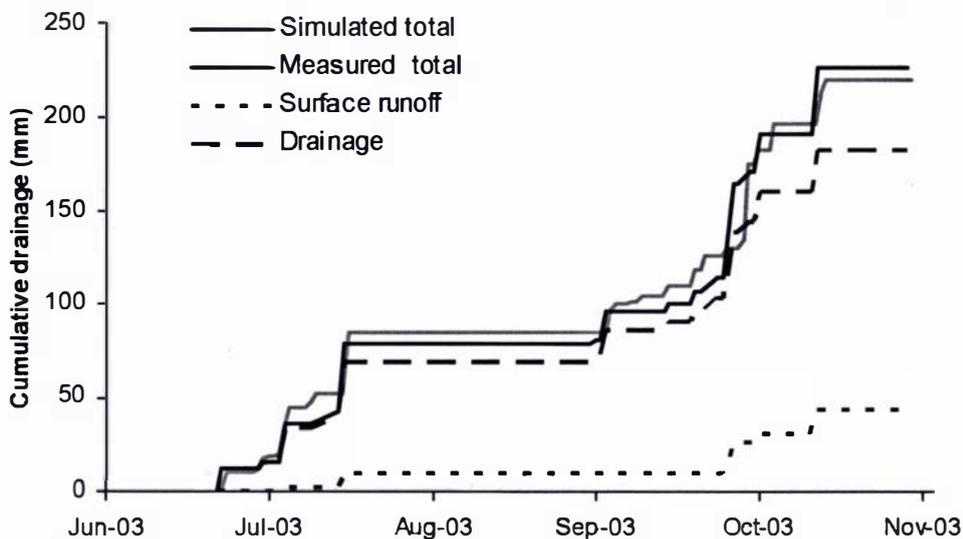
#### 7.4.2 Observed and predicted drainage volumes

The total of 236 mm of subsurface drainage recorded in 2003 was 16 mm greater than the volume of drainage measured during 2002 (Houlbrooke *et al.* 2003 (Chapter 6)). The 41 mm of measured surface runoff in 2003 made up 15% of the total volume of water exiting the soil profile. Although the proportion of discharge exiting as surface runoff will vary between years depending upon, among other factors, rainfall intensity. In the winter of 1975, Sharpley and Syers (1979) found similar partitioning between drainage and surface runoff volumes for a mole and pipe drained soil. The Tokomaru silt loam has an impermeable fragipan ( $K_{sat} < 0.1$  mm/day at 300 mm depth). In contrast, mole and pipe drainage systems in this soil typically have a drainage coefficient of approximately 10 mm/day (Horne 1985). Artificial drainage, therefore, decreases the proportion of discharge from these soils that is surface runoff providing greater stock intensification does not decrease soil hydraulic properties.

The daily soil water balance procedure (Scotter *et al.* 1979b) gave a good prediction of drainage plus runoff volumes (Figure 7.5) once the soil had been recharged to field capacity (as determined by the model). However, the daily soil water balance was a poor predictor of drainage volumes during the recharge period. The following observations help to explain the poor prediction. Heavy rainfall (70 mm over two days)

in late autumn resulted in preferential rewetting of the very dry soil above mole channels. This was visible in the soil profile and in the small amounts of drainage that were recorded despite the large soil water deficit (as predicted by the soil water balance model). With each successive rainfall event, a greater percentage of the rainfall exited as drainage until field capacity was reached in mid June. As the simple soil water balance assumes uniform rewetting of dry soil, and that drainage plus runoff can only commence on the attainment of field capacity. The model is unable to predict preferential wetting and the associated drainage.

The relationships between rainfall intensity and quantity, soil moisture deficit and infiltration rate all dictate the commencement of drainage and surface runoff. An example of the response in runoff and drainage rates to rainfall intensity and quantity is given in Figure 7.6. Between 2 am and 7am on 28 September, 6 mm of rainfall brought the soil back to field capacity from a predicted 5 mm deficit. After field capacity was attained the drainage and runoff response to rainfall was rapid. Drainage was essentially complete within 12 hours of the conclusion of the last major rainfall event at 7am on the 29 September.



**Figure 7.5** A comparison of actual cumulative drainage and surface runoff with simulated cumulative drainage and runoff using a soil water balance following the attainment of field capacity in June 2003.

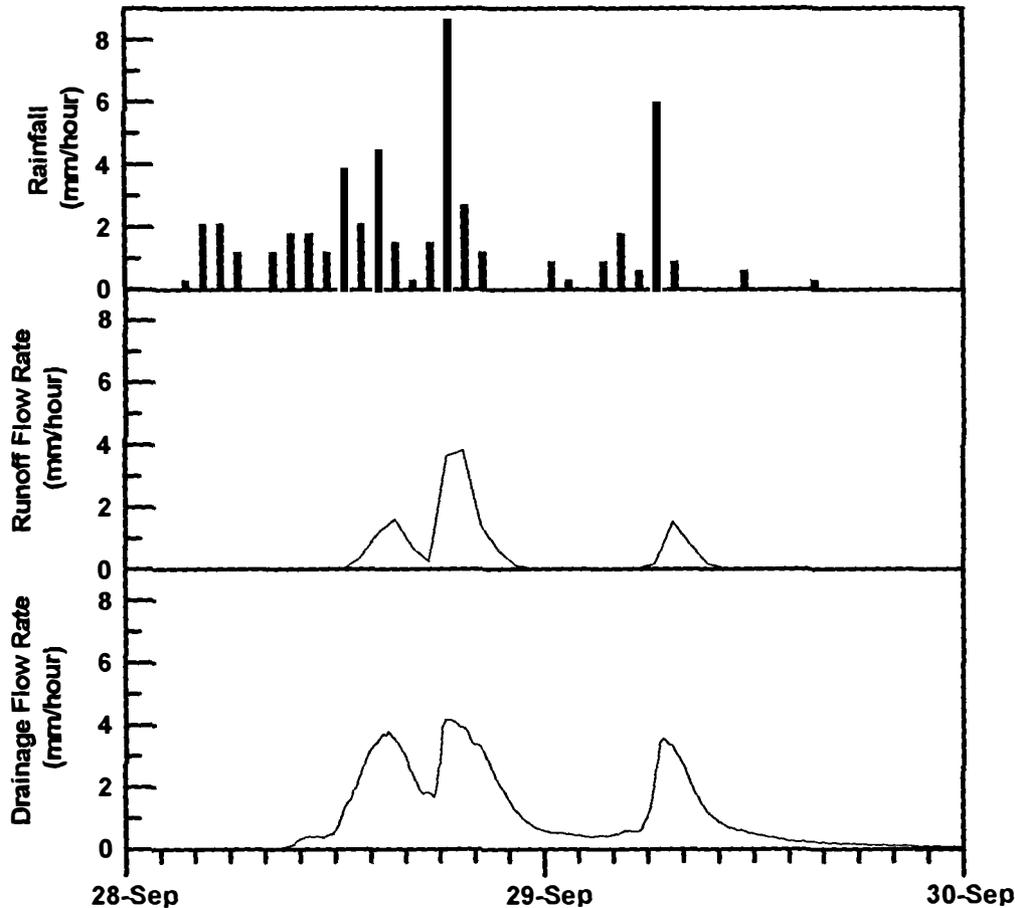


Figure 7.6 An example of the influence of rain intensity and quantity on runoff and drainage rates when the soil was at field capacity.

### 7.4.3 Nutrient loss from non-effluent plots

The mean concentrations of both TN ( $14.5 \text{ mg N litre}^{-1}$ ) and TP ( $0.3 \text{ mg P litre}^{-1}$ ) in the 2003 mole and pipe drainage water (Tables 7.1 & 7.2) under standard dairy pasture (i.e. non-effluent plots) were considerably greater than the levels ( $0.61 \text{ mg N litre}^{-1}$  &  $0.033 \text{ mg P litre}^{-1}$ ) considered likely to promote aquatic weed growth in lowland streams (ANZECC, 2000). Total losses of N and P in mole and pipe drainage for the season were  $34.1 \text{ kg N ha}^{-1}$  and  $0.73 \text{ kg P ha}^{-1}$  (Tables 7.1 & 7.2). These values were greater than those measured at the same research site for the previous (2002) winter-spring drainage season (Houlbrooke *et al.* 2003 (Chapter 6)).

Nearly all the N lost in drainage was in the dissolved form (c. 96%); 76% of the dissolved N was  $\text{NO}_3^-$ -N, 18% was dissolved organic N, and 2% was  $\text{NH}_4^+$ -N (Table

7.1). On the non-effluent plots, approximately 75% of P loss was in the dissolved form (20% as DIP; Table 7.2); consequently, only 25% of P loss was in the particulate form.

**Table 7.1.** *The amounts and forms of N lost in 2003 winter drainage and surface runoff. SEM of annual loss and concentration data are presented in brackets. SEM is calculated from four plots on the non-effluent drainage plots, two plots on the effluent drainage plots and non-effluent surface runoff plots and not calculated for the un-replicated effluent surface runoff plot.*

N Form	Treatment	Drainage		Surface runoff	
		Concentration (mg/L)	Annual loss (kg/ha)	Concentration (mg/L)	Annual loss (kg/ha)
NO <sub>3</sub> <sup>-</sup> -N	Effluent <sup>A</sup>	8.8 (0.4)	21.7 (0.9)	0.49	0.26
	Effluent <sup>B</sup>	11.1 (1.8)	21.6 (0.9)	0.49	0.26
	Non-effluent	9.2 (0.4)	26.3 (4.3)	1.68 (0.51)	0.69 (0.21)
NH <sub>4</sub> <sup>+</sup> -N	Effluent <sup>A</sup>	0.96 (0.01)	2.36 (0.03)	8.8	4.7
	Effluent <sup>B</sup>	0.11 (0.01)	0.27 (0.03)	0.47	0.19
	Non-effluent	0.28 (0.04)	0.67 (0.1)	0.42 (0.11)	0.17 (0.04)
TDN	Effluent <sup>A</sup>	12.9 (0.3)	31.6 (0.7)	13.7	7.3
	Effluent <sup>B</sup>	12.0 (0.3)	28.4 (0.7)	4.2	1.7
	Non-effluent	13.8 (1.8)	32.5 (4.2)	4.8 (0.45)	2.0 (0.19)
TN	Effluent <sup>A</sup>	14.2 (0.25)	35.0 (0.6)	18.9	10.1
	Effluent <sup>B</sup>	12.6 (0.3)	29.8 (0.6)	5.9	2.4
	Non-effluent	14.5 (2.0)	34.1 (4.6)	7.3 (1.2)	3.0 (0.49)

<sup>A</sup> Data from effluent treatment plots includes the drainage and runoff from a poorly timed application of FDE in mid September.

<sup>B</sup> Data from effluent treatment plots excludes the drainage and runoff from an application of FDE in mid September (i.e. deferred irrigation only applications).

Total losses of N and P in surface runoff were 3 kg N ha<sup>-1</sup> and 0.6 kg P ha<sup>-1</sup> with average concentrations of 7.3 mg N litre<sup>-1</sup> and 1.4 mg P litre<sup>-1</sup> (Table 7.1 & 7.2). The greater concentration of P in surface runoff water compared to mole and pipe drainage is commonly because surface runoff contains P enriched surface sediment and dung, particularly following cattle grazing which would usually be adsorbed by the soil before the water drained. The proportion of particulate P in surface runoff water (40%; Table 7.2) was greater than that in the drainage water (27%). Whilst this increase in proportion of PP made a contribution to the higher concentration of TP in surface

runoff, it is also due to a near 4 fold increase in TDP concentration. In contrast to the forms of N in drainage, only 20% of the N lost in runoff water was in the  $\text{NO}_3^-$ -N form (average concentration of 1.68 mg N litre<sup>-1</sup>; Table 7.1). This difference is likely to be due to the limited opportunity that surface runoff has to leach and transport  $\text{NO}_3^-$ -N.

**Table 7.2.** The amounts and forms of P lost in 2003 winter drainage. SEM of annual loss and concentration data are presented in brackets. SEM is calculated from four plots on the non-effluent drainage plots, two plots on the effluent drainage plots and non-effluent surface runoff plots and not calculated for the un-replicated effluent surface runoff plot.

P Form	Treatment	Drainage		Surface runoff	
		Concentration (mg/L)	Annual loss (kg/ha)	Concentration (mg/L)	Annual loss (kg/ha)
DIP	Effluent <sup>A</sup>	0.37 (0.06)	0.90 (0.14)	1.60	0.85
	Effluent <sup>B</sup>	0.3 (0.06)	0.68 (0.14)	0.74	0.31
	Non-effluent	0.07 (0.01)	0.15 (0.03)	0.54 (0.04)	0.23 (0.02)
TDP	Effluent <sup>A</sup>	0.59 (0.06)	1.44 (0.14)	2.27	1.21
	Effluent <sup>B</sup>	0.46 (0.06)	1.09 (0.14)	1.10	0.45
	Non-effluent	0.22 (0.01)	0.53 (0.03)	0.85 (0.1)	0.35 (0.04)
TP	Effluent <sup>A</sup>	0.93 (0.11)	2.30 (0.27)	3.59	1.91
	Effluent <sup>B</sup>	0.63 (0.11)	1.47 (0.27)	1.58	0.66
	Non-effluent	0.31 (0.02)	0.73 (0.04)	1.44 (0.11)	0.60 (0.05)

<sup>A</sup> Data from effluent treatment plots includes the drainage and runoff from a poorly timed application of FDE in mid September.

<sup>B</sup> Data from effluent treatment plots excludes the drainage and runoff from an application of FDE in mid September (i.e. deferred irrigation only applications).

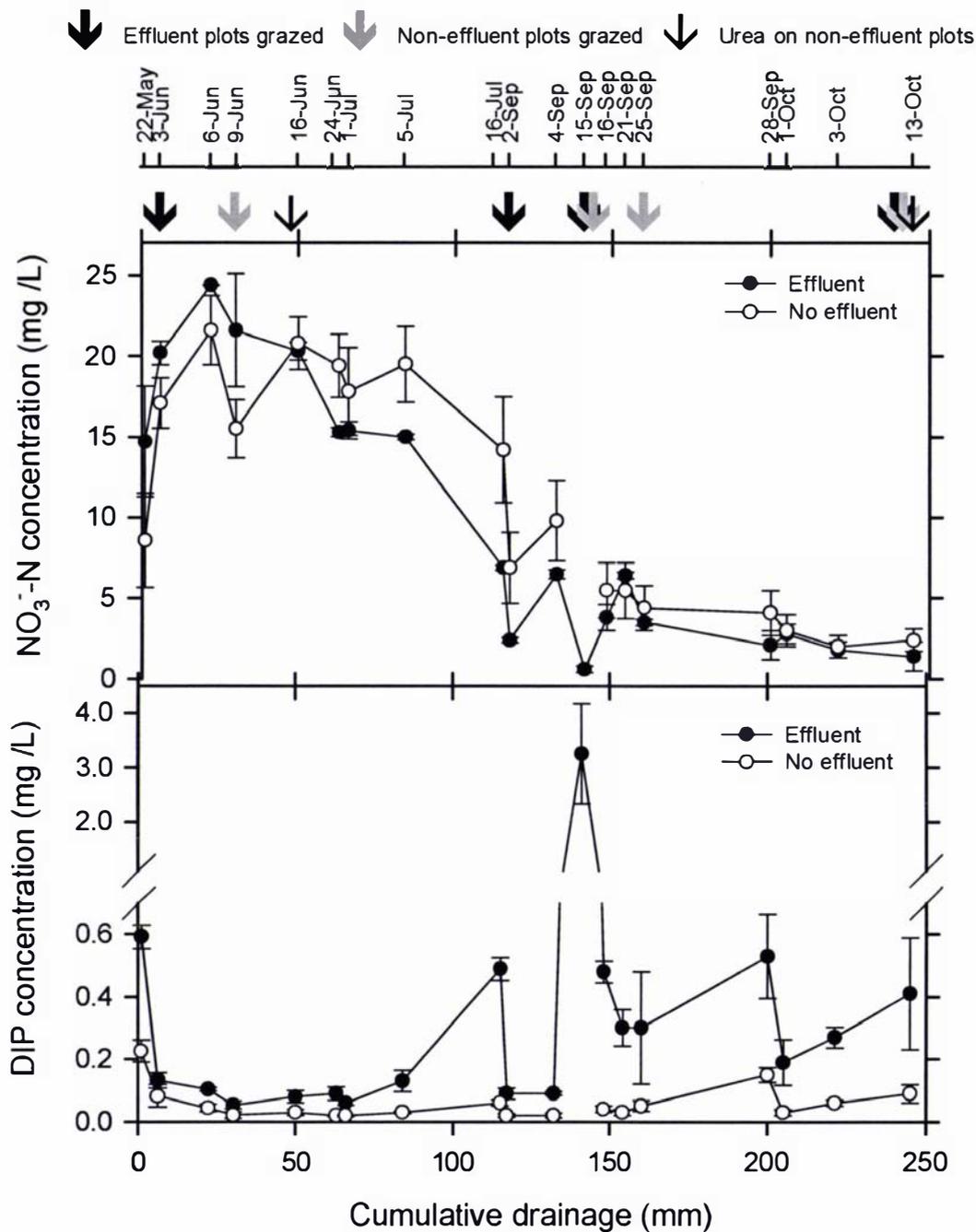
Over the drainage season,  $\text{NO}_3^-$ -N concentrations of the mole and pipe drainage showed a marked trend of decreasing concentration (Figure 7.7). The early drainage events had  $\text{NO}_3^-$ -N concentrations up to 20 mg N litre<sup>-1</sup>. However, with increasing cumulative drainage,  $\text{NO}_3^-$ -N concentrations steadily declined to concentrations less than 5 mg litre<sup>-1</sup>. A similar pattern of  $\text{NO}_3^-$ -N concentrations in artificial drainage waters was observed in 2002 (Houlbrooke *et al.* 2003 (*Chapter 6*)) and previously by Sharpley and Syers (1979), Monaghan *et al.* (2002) under cattle grazing and by Heng *et al.* (1991) under sheep grazing. This pattern in N leaching is generally attributed to the flushing, in the early drainage events, of the pool of mineralised  $\text{NO}_3^-$ -N that has accumulated during the warmer summer-autumn period. The seasonal flushing effect

and decreasing trend in  $\text{NO}_3^-$ -N concentrations were not evident for surface runoff water (data not presented) indicating different N sources contribute to surface runoff than drainage. The  $\text{NO}_3^-$ -N concentration in the first drainage event following the six week dry period at the end of August (117 mm of accumulated drainage) was smaller than that in the previous drainage event prior to the dry period (Figure 7.7).

Whilst the average  $\text{NO}_3^-$ -N concentration of  $9.2 \text{ mg litre}^{-1}$  (Table 7.1) in the mole and pipe drainage water was less than the recommended drinking water standard of  $11.3 \text{ mg litre}^{-1}$  (Ministry of Health 1995), nearly 50% of the cumulative drainage water had  $\text{NO}_3^-$ -N concentrations in excess of this level (Figure 7.7). To bring this value down below the recommended level considered likely to promote aquatic weed growth in lowland streams (ANZECC, 2000) a receiving pristine water body would have to dilute this drainage at a ratio of approximately 150:1.

The concentration of DIP throughout the winter and spring mole and pipe drainage exhibited no seasonal pattern (Figure 7.7). Changes in P concentrations were largely attributed to management and will be discussed below. Similar results showing no seasonal influence on DIP concentrations have been reported by Monaghan *et al.* (2002) and Houlbrooke *et al.* (2003(Chapter 6)).

In comparison to previous research at the same site, the concentrations of both  $\text{NO}_3^-$ -N and DIP were very similar to those reported 25 years earlier by Sharpley and Syers (1979) even though they used a stocking density of  $300 \text{ cows ha}^{-1}$  for 12 hours compared to the average stocking density of  $100 \text{ cows ha}^{-1}$   $12 \text{ hours}^{-1}$  from our research. Unfortunately, as Sharpley and Syers only present data for two intervals of four weeks each, no comparison can be made of seasonal nutrient losses. Research reported by Monaghan *et al.* (2002) on a similar mole and pipe drained Pallic soil type in Southland reported lower nutrient losses ( $26.5 \text{ kg N ha}^{-1}$  and  $0.15 \text{ kg P ha}^{-1}$ ) than those reported above. These differences can be attributed to the Southland research sites previous long term history of less intensive sheep farming, the current grazing of non-lactating cattle at an average stocking rate of  $2.3 \text{ cows ha}^{-1}$ , a low Olsen P of  $14 \mu\text{g ml}^{-1}$  and the lack of any N fertiliser application during the 5 years prior to the research or during the year that the research was reported for.



**Figure 7.7** The trends in the NO<sub>3</sub><sup>-</sup>-N and DIP concentration of drainage water with the total amount of accumulated drainage. The poorly timed mid-September application of FDE occurred at c. 140 mm cumulative drainage. The broken line within the non-effluent plots covers the period of drainage within the effluent plots due to the mid September application. Error bars represent two SEM.

Comparisons of nitrogen leaching losses in drainage water between soil types are difficult. There has been a number of studies (Silva *et al.* 1999, Di and Cameron 2002a, Di and Cameron 2002b) arising from free draining Brown soils in the Canterbury region of New Zealand. However, this research often involves different management practises

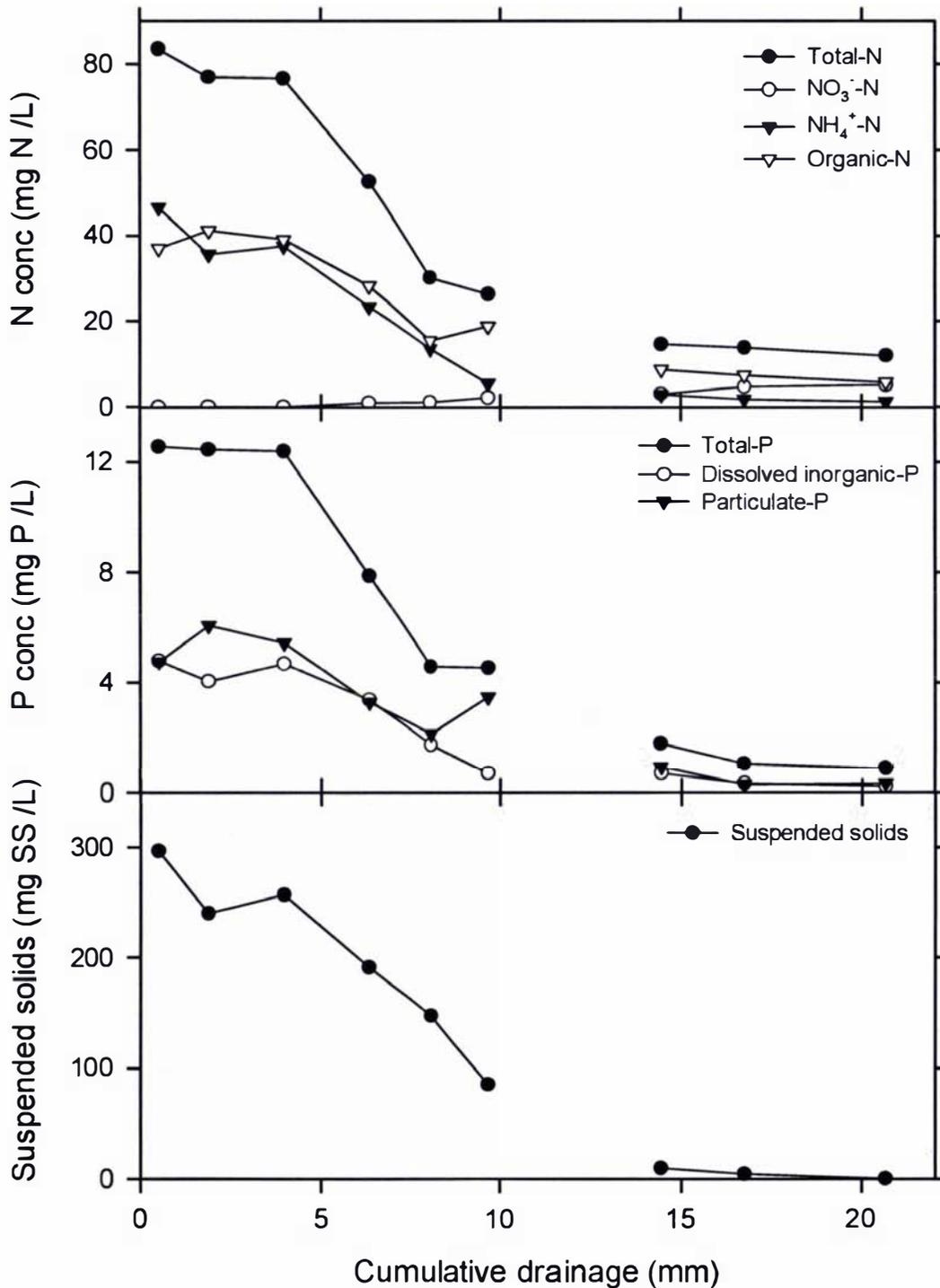
such as flood irrigation applications and/or high rates of N fertiliser, both of which tend to increase total N loss. An unfertilised control treatment reported by Silva *et al.* (1999) leached a total of 31 kg N ha<sup>-1</sup> of NO<sub>3</sub><sup>-</sup>-N. This value is c. 5 kg ha<sup>-1</sup> greater than the data reported above for the non-effluent plots which received urea. One possible reason for the difference is that the wetter Pallic soil type is leaching smaller volumes of NO<sub>3</sub><sup>-</sup>-N due to increased denitrification.

#### 7.4.4 Direct effects of FDE on drainage and surface runoff water quality

The impacts of deferred irrigation and a FDE application when the soil was near field capacity, on drainage and surface runoff water quality at the research site during the 2002-2003 lactation season are described by Houlbrooke *et al.* (2004a) (*Chapter 4*) and Houlbrooke *et al.* (2004c) (*Chapter 5*). For deferred irrigation, a total of 80 mm of FDE was applied in four separate irrigations at an average depth of 20 mm per application during the spring to autumn period of 2002/2003. The average volume of drainage exiting the mole and pipe network as a result of FDE applications was equal to 1% of the total accumulated effluent applied, and was likely to be attributable to the preferential flow of effluent through the large macropores leading to the moles. This drainage resulted in a total direct loss of 0.4 kg N ha<sup>-1</sup> and 0.08 kg P ha<sup>-1</sup>, which was <1% of the TN and TP applied in the effluent. When 25 mm of FDE was applied to the effluent plots on the 15 September 2003 (7 mm deficit) approximately 70% of the applied FDE left the plots (8 mm as surface runoff and 10 mm as direct drainage). The direct nutrient losses associated with this poorly timed irrigation were 13.2 kg of N ha<sup>-1</sup> (5.6 kg N ha<sup>-1</sup> in drainage and 7.6 kg N ha<sup>-1</sup> in surface runoff) and over 2 kg of P ha<sup>-1</sup> (0.9 kg P ha<sup>-1</sup> in drainage and 1.2 kg P ha<sup>-1</sup> in surface runoff) (Houlbrooke *et al.* 2004a (*Chapter 4*)).

In this paper, the details of changes in concentrations of N, P and suspended solids in the drainage caused by the poorly timed irrigation event (15 September) and the rainfall of the following day are presented. The concentrations of TN in the first 4 mm of cumulative drainage generated by the poorly timed FDE irrigation were relatively constant at c. 80 mg N litre<sup>-1</sup> before declining steeply to c. 30 mg N litre<sup>-1</sup> after nearly 10 mm of cumulative drainage (Figure 7.8). This suggests that the early drainage was likely to be partially treated or diluted FDE percolating rapidly through the large macropores above the mole channels. As drainage continued, the remaining FDE was likely to have infiltrated medium sized pores and mixed with the resident soil solution providing greater opportunity for dilution and adsorption of nutrients. Drainage N loss

was predominantly in the dissolved form with almost equal contributions of  $\text{NH}_4^+$ -N and total organic N (TON).  $\text{NO}_3^-$ -N made only a very small contribution to TN loss of less than 2 mg N litre<sup>-1</sup> (Figure 7.8).



**Figure 7.8** The concentrations and forms of N, P and suspended solids with cumulative drainage (0-10 mm) from a poorly timed application of FDE on 15/09/03 and cumulative drainage (10-21 mm) induced by rainfall one day later on 16/09/03.

Very large concentrations of TN in the drainage were associated with the poorly timed application of FDE: at nearly 60 mg N litre<sup>-1</sup>, this concentration was approximately 6 fold greater than concentrations measured for other drainage events at that time of year. The same dramatic six-fold increase was also evident in the concentration of DIP in drainage from the effluent plots resulting from the same FDE application (Figure 7.7).

On the day following the poorly timed application of FDE (16/09/03), 12 mm of rainfall induced 11 mm of drainage. The average N concentration of this drainage water was c. 14 mg N litre<sup>-1</sup> compared to c. 8 mg N litre<sup>-1</sup> from the non-effluent plots (1-4). This difference indicates that soil water enriched with FDE during the previous day's application contributed to drainage (7.8). During this drainage event, NO<sub>3</sub><sup>-</sup>-N made an increasing contribution to TN loss. This would be expected as the drainage generated by the lower intensity rainfall had greater opportunity to leach resident soil NO<sub>3</sub><sup>-</sup>-N from smaller pores than the high intensity FDE irrigation.

The TP concentrations of the drainage event induced by the poorly timed application of FDE followed a similar trend to that of TN. The concentration of TP was relatively constant at c. 12 mg P litre<sup>-1</sup> during the first 4 mm of cumulative drainage before declining steeply to c. 5 mg P litre<sup>-1</sup> after nearly 10 mm of cumulative drainage (Figure 7.8). Concentrations of particulate P (PP) and DIP were similar throughout the drainage event. One day after the poorly timed FDE irrigation, TN and TP concentrations in the natural drainage were still relatively large suggesting further flushing of nutrients applied in FDE (Figure 7.8).

Suspended solids (SS) concentrations declined from c. 300 mg litre<sup>-1</sup> at the start of drainage to c. 90 mg litre<sup>-1</sup> by the time drainage finished; a pattern similar to TN and TP concentrations (Figure 7.8). Greater SS concentrations indicate that preferential flow was the dominant mechanism for rapid drainage of FDE through the mole and pipe system. SS concentrations fell from 10 to 1 mg litre<sup>-1</sup> following the rainfall induced drainage one day after the effluent application (Figure 7.8), indicating a decrease in the amount of preferential flow.

#### 7.4.5 Indirect effects of deferred irrigation of FDE on drainage and surface runoff water quality

Land application of FDE using deferred irrigation had minimal impact on N loss in winter-spring drainage water. The difference between the effluent and non-effluent plots was only  $0.9 \text{ kg N ha}^{-1} \text{ yr}^{-1}$  or 2.5% of total N loss (Table 7.1). As described above, considerable portions of the losses of nutrients from the effluent plots were as a result of the single, poorly-timed September application of FDE. When the N loss from the September FDE irrigation is subtracted from annual N loss, the total loss of N from the effluent plots is less than that from non-effluent plots although this is not likely to be a significant difference (Table 7.1). Comparison of  $\text{NO}_3^-$ -N concentration from the effluent and non-effluent plots subsequent to the aberrant FDE application suggest that there was no long term carryover leaching effect.

In contrast to N loss, deferred irrigation of FDE in summer resulted in increased P loss in winter-spring drainage. When the P losses associated with the poorly-timed irrigation are excluded, then the P loss from effluent plots was  $1.47 \text{ kg P ha}^{-1} \text{ yr}^{-1}$  compared with  $0.73 \text{ kg P ha}^{-1} \text{ yr}^{-1}$  from the non-effluent plots. However, this additional loss of P under deferred irrigation ( $0.74 \text{ kg P ha}^{-1}$ ) represents only c. 2.5% of the P applied in FDE in the summer (Table 7.2), a result very similar to that observed in the previous season (Houlbrooke *et al.* 2003 (*Chapter 6*)). It is likely that the increased loss of P in drainage as a result of summer application of FDE is related to its application in liquid form and intensity of application. It is suggested that phosphorus in 25 mm application of FDE will percolate to a greater soil depth than P diffusing from surface applied fertiliser granules. Subsequently, winter drainage is likely to cause P to desorb from macropore surfaces previously enriched by P from FDE.

Similar research conducted in west Otago, New Zealand by McDowell *et al.* (2005) found that up to 56% of P loss from a plot irrigated with FDE took place when drainage events occurred within one week of either a FDE application or cattle grazing event. Data over three years showed a mean P loss of  $0.4 \text{ kg ha}^{-1}$  compared for  $0.2 \text{ kg ha}^{-1}$  from un-irrigated plots. These values are considerably lower than the P loss presented above, however the concentrations of P were similar suggesting that the large difference is attributable to a near 100% increase in drainage volume from our plots.

Tables 7.1 and 7.2 show that the total losses of N and P in surface runoff water from the effluent plots ( $10.1 \text{ kg N ha}^{-1} \text{ yr}^{-1}$  &  $1.91 \text{ kg P ha}^{-1} \text{ yr}^{-1}$ ) were considerably greater

than that from the non-effluent plots ( $3.0 \text{ kg N ha}^{-1} \text{ yr}^{-1}$  &  $0.6 \text{ kg P ha}^{-1} \text{ yr}^{-1}$ ). Again, this is mostly attributable to the effect of the poorly timed FDE irrigation in September. There were no extra losses of N or P in winter-spring surface runoff water from the effluent plots associated with summer applications of FDE using deferred irrigation (Table 7.1 & 7.2). This is to be expected as the nutrients in the FDE applied in the summer would have likely been adsorbed in the soil profile and, therefore, would be at less risk to loss in winter runoff.

Deferred irrigation of FDE increased DIP losses in subsequent winter-spring drainage. The concentration of DIP in drainage from the effluent plots was greater than that from the non-effluent plots (Figure 7.7). These differences become more evident as the drainage season progressed. A similar trend was observed in the previous winter drainage season (Houlbrooke *et al.* 2003 (*Chapter 6*)). The average concentration of DIP in the effluent plots (excluding the mid Sept FDE application) was  $0.3 \text{ mg P litre}^{-1}$  compared to the  $0.18 \text{ mg P litre}^{-1}$  for these same plots reported by Houlbrooke *et al.* (2003). In contrast, the concentration of DIP from the non-effluent plots was the same over the two drainage seasons. An increase in effluent P input from  $16 \text{ kg P ha}^{-1} \text{ yr}^{-1}$  in the 2001/2002 summer to  $31 \text{ kg P ha}^{-1} \text{ yr}^{-1}$  in the summer preceding the 2003 winter-spring drainage is likely to have contributed to the greater concentration of DIP in 2003. Furthermore some of the increase could be attributed to the residual effect of the poorly timed FDE application half way through the drainage season.

The large DIP loss on the effluent plots is the chief component of the increased loss of TP from these plots (Table 7.2) discussed above. This is in contrast to the findings of McDowell *et al.* (2005) who found that the primary component of P loss under dairy pasture receiving applications of FDE was in the dissolved organic P (DOP) form. Losses of DOP were nearly 4 times greater from their effluent plots than from non-effluent plots. Differences in the partitioning of P loss may be attributable to P sorption due to disparity in the age of the respective mole pipe systems and the history of FDE application. Possible reasons for the enhanced loss of P from FDE irrigated soils included a build up of mobile organic P forms, decreased sorption capacity of the soil and/or preferential pathways due to blockage of P-rich FDE colloids.

Toor *et al.* (2004) also measured increased losses of P two to three times greater from lysimeters on a free draining Lismore stony silt loam amended with FDE at approximately  $30$  and  $60 \text{ kg ha}^{-1} \text{ yr}^{-1}$  and P fertiliser ( $45 \text{ kg ha}^{-1} \text{ yr}^{-1}$ ) then from soil

receiving P fertiliser only. Greater losses of particulate P during the irrigation season are attributed to the high intensity of flood irrigation.

Whilst summer applications of FDE had no impact on either N or P losses in surface runoff water in winter, there was a dramatic increase in the concentration of both N and P as a result of the poorly timed application of FDE in mid September with c. 10 fold increases in concentration. These peak concentrations of N and P were 95 and 16 mg litre<sup>-1</sup>, respectively (data not presented).

#### **7.4.6 Effects of winter cattle grazing on drainage and surface runoff water quality**

For the winter-spring drainage season of 2002, Houlbrooke *et al.* (2003) (*Chapter 6*) reported increases in NO<sub>3</sub><sup>-</sup>-N concentration of c. 5 mg litre<sup>-1</sup> in drainage events that occurred approximately 10 days after a cattle grazing event. In the 2003 drainage season, such clear trends were not so evident due to variations in the timing of grazing and the duration of the period between grazing and subsequent drainage. Grazing on the effluent plots on 30 May was followed by an increase in NO<sub>3</sub><sup>-</sup>-N concentration of c. 4 mg litre<sup>-1</sup> in drainage that occurred seven days following grazing. The response to this urine addition, occurring at a very early stage in the drainage season (< 10 mm of cumulative drainage), may have been obscured by the rapid increase in NO<sub>3</sub><sup>-</sup>-N concentrations that was measured on all plots as part of the flushing trend discussed above (Figure 7.7).

Houlbrooke *et al.* (2004d) (*Chapter 8*) reported on a drainage event on the 6 June 2003 (at 20 mm of cumulative drainage) that coincided with cattle grazing the non-effluent plots. In this case, the NO<sub>3</sub><sup>-</sup>-N concentration in the drainage was typical of an event for this time of year (Figure 7.7) but the levels of NH<sub>4</sub><sup>+</sup>-N and TON were unusually high, particularly in the first 2-3 mm of drainage. This suggests that some of the urine and ammonium-N produced by rapid hydrolysis is preferentially drained through large macropores into the mole and pipe drainage system. There was no increase in the NO<sub>3</sub><sup>-</sup>-N concentrations of drainage from the non-effluent plots on the 9 June, three days after the simultaneous grazing and drainage of 6 June. However, a further drainage event 10 days following the initial grazing resulted in c. 5 mg litre<sup>-1</sup> greater NO<sub>3</sub><sup>-</sup>-N concentrations in drainage (Figure 7.7). Ten days is considered sufficient time for the conversion of urine inputs to NO<sub>3</sub><sup>-</sup>-N via the processes of hydrolysis and nitrification (Condon *et al.* 2004).

Further cattle grazing events on both the effluent (24 August) and non-effluent plots (29 August) resulted in increases in  $\text{NO}_3^-$ -N concentrations of 5 and 3 mg litre<sup>-1</sup> in drainage that occurred 11 and 7 days after the grazing events, respectively. A light grazing (91 cows for a 12 hour period) on the non-effluent plots on the 25 September (c. 160 mm of cumulative drainage) did not result in an increase in  $\text{NO}_3^-$ -N concentration in subsequent drainage. Grazing events on both treatments late in the drainage season occurred only seven and six days prior to the last drainage event on the 13 October. Although, this period should have been sufficient for some nitrification to occur, higher rates of plant growth and mineral N uptake at this time of year may have reduced the availability of  $\text{NO}_3^-$ -N for leaching (Figure 7.7).

Research carried out 25 years earlier on the same soil type at the No. 4 dairy by Sharpley and Syers (1979) reported a five fold increase in  $\text{NO}_3^-$ -N concentration in mole and pipe drainage events following cattle grazing. The approximate 12 mg litre<sup>-1</sup> increase in  $\text{NO}_3^-$ -N concentration in drainage was considerably greater than the approximate 5 mg litre<sup>-1</sup> increase reported following cattle grazing events above. This could largely be attributed to stock grazing density with Sharpley and Syers (1979) grazing their research plots 300 cows ha<sup>-1</sup> for a 12 period compared to the average stocking density of 100 cows ha<sup>-1</sup> for a 12 period for the current research.

Cattle grazing events in the 2003 winter-spring did not appear to effect the concentration of DIP in drainage. The early grazing of both treatments in late May and early June resulted in no increase in DIP in drainage at a time when concentrations seemed to be decreasing or stable (Figure 7.7). Likewise, cattle grazing in the late August grazing round did not impact on DIP concentrations in drainage. A small increase in DIP of c. 0.1 mg litre<sup>-1</sup> was evident in the drainage from non-effluent plots 10 days following a cattle grazing on the 25 September. However, DIP concentration in drainage from the effluent plots that had not received such a recent cattle grazing showed an even more marked increase (c. 0.25 mg litre<sup>-1</sup>) over the same time period. This is in contrast to Sharpley and Syers (1979) who reported a 40 fold increase in DIP concentration following a 12 hour grazing at a density of 300 cows ha<sup>-1</sup> for a 12 hour period compared to the average stocking density of 100 cows ha<sup>-1</sup> for a 12 period from the above reported data.

TN and TP concentrations in surface runoff water increased by 10.8 and 0.65 mg litre<sup>-1</sup>, respectively, when grazing coincided with surface runoff on 6 June. This surface runoff is likely to have contained relatively large quantities of fresh dung and sediment

disturbed by the hooves of the cattle. The same trend was evident for both TN and TP when surface runoff occurred three days following a cattle grazing event on the non-effluent plots on 28 September.

#### 7.4.7 Effects of winter and spring urea applications on drainage water quality

The impact of urea applications on  $\text{NO}_3^-$ -N leaching can be assessed by comparing  $\text{NO}_3^-$ -N concentrations in drainage water from the non-effluent plots with those from the effluent plots for the periods immediately following addition of the fertiliser to the non-effluent plots. Urea was applied to the non-effluent plots at  $37 \text{ kg N ha}^{-1}$  on the 12 June 2003, three days prior to a drainage event that occurred when cumulative drainage for the season had reached c. 50 mm. Up until this time, the average concentrations of  $\text{NO}_3^-$ -N in drainage from the effluent and non-effluent plots were very similar (Figure 7.7). The urea application did not affect mean  $\text{NO}_3^-$ -N concentration in drainage water three days after its application. However, at the subsequent drainage event, nine days after application of urea, the mean  $\text{NO}_3^-$ -N concentration in the drainage from non-effluent plots was c.  $5 \text{ mg N litre}^{-1}$  greater than the concentration in drainage from the effluent plots. Presumably, this period was sufficient for hydrolysis and nitrification of the added urea (Condon *et al.* 2004). The increase in  $\text{NO}_3^-$ -N concentration in drainage, attributed to the urea application, was evident for the next five drainage events, representing c. 80 mm of cumulative drainage (Figure 7.7). The total  $\text{NO}_3^-$ -N loss of  $26.3 \text{ kg N ha}^{-1} \text{ yr}^{-1}$  from the non-effluent plots was  $4.7 \text{ kg N ha}^{-1} \text{ yr}^{-1}$  greater than the  $\text{NO}_3^-$ -N loss from the effluent plots (Table 7.1). This result supports the view that applications of urea in winter when pasture growth and N uptake are slow can result in considerable increases in  $\text{NO}_3^-$ -N leaching. It is acknowledged that Appendix one demonstrates that most of the increase in  $\text{NO}_3^-$ -N leaching following urea application is attributable to large increases from 2 of the 4 plots that received urea fertiliser.

A similar result was reported by Sharpley and Syers (1979) who applied 62% more urea ( $60 \text{ kg N ha}^{-1}$ ) in late June. They recorded an approximate  $7 \text{ mg N litre}^{-1}$  increase in  $\text{NO}_3^-$ -N concentration in the drainage event following urea application. Sharpley and Syers (1979) also measured increased  $\text{NO}_3^-$ -N concentrations in drainage for a prolonged 2 month, 10 event period following the application of urea. In contrast, Houlbrooke *et al.* (2003) (*Chapter 6*) reported no increase in  $\text{NO}_3^-$ -N concentrations following a urea application in early spring, a time when plant uptake would have accounted for most of the mineral N in the soil. A further urea application on the 9 October 2003 did not increase the  $\text{NO}_3^-$ -N concentration in drainage because there was

insufficient time between application and the last drainage event (13 October) for it to be converted to  $\text{NO}_3^-$ -N (Condon *et al.* 2004). In addition, an application at this time of the year would not pose the same leaching risk due to greater plant uptake.

## 7.5 SUMMARY AND CONCLUSIONS

The average concentrations of N and P in 2003 winter mole-pipe drainage and surface runoff water from a grazed dairy pasture in the Manawatu were all well above concentrations that will stimulate increased aquatic weed growth in fresh water bodies. A number of dairy farm practices (not necessarily best practice) increased the losses of N and P in both artificial drainage water and surface runoff. In particular, application of FDE to wet soil in mid September resulted in the drainage and surface runoff of partially treated effluent. N and P losses in drainage and surface runoff associated with this poorly timed FDE irrigation were six to ten fold greater than would normally be expected from drainage and surface runoff events induced by spring rainfall. In contrast, applying FDE effluent in the proceeding summer according to the deferred irrigation criteria resulted in no increase in the loss of N in winter drainage. Deferred irrigation of FDE did not prevent a substantial (50%) increase in total P losses in winter drainage. However, this increase represents less than 2.5% of the P applied in the effluent. This result highlights the importance of being able to store FDE and schedule it for land application in accordance with deferred irrigation criteria.

Both grazing and the application of urea increased nutrient loss in drainage and surface runoff. Drainage events occurring c. ten days after cattle grazing events resulted in an increase of  $\text{NO}_3^-$ -N concentration of c. 5 mg litre<sup>-1</sup>. A urea application in mid June at a time when plant uptake of mineral N was low resulted in an increase in the loss of  $\text{NO}_3^-$ -N for up to 80 mm of subsequent cumulative drainage.

To decrease nutrient loss in mole and pipe drainage and surface runoff water on winter wet Pallic soils such as the Tokomaru silt loam, best farm management practise would avoid winter cattle grazing or the application of urea when soil temperature, pasture growth and hence plant uptake are all low. This is particularly applicable in the months of June and July when dairy cows are dry and many are grazed off the farm. The potential benefits of providing supplementary feed on covered feed-pads vs. pasture and forage winter grazing on both soil and water quality requires further investigation.

## 7.6 ACKNOWLEDGEMENTS

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# CHAPTER 8: NITROGEN LOSSES IN ARTIFICIAL DRAINAGE AND SURFACE RUNOFF FROM PASTURE FOLLOWING GRAZING BY DAIRY CATTLE

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## 8.1 ABSTRACT

Dairy cattle are significant contributors to the nutrient enrichment of New Zealand's aquatic environment. However, there have been very few field studies that quantify the accelerated loss of nutrients that occurs when grazing coincides with or is immediately followed by drainage and/or surface runoff. More detailed information is required on the environmental risk associated with cows grazing at times when rainfall is likely to generate drainage and surface runoff. Nitrogen loss occurring in drainage and surface runoff immediately after grazing was quantified at a research site established on an artificially drained, fine-textured soil, at Massey University's No. 4 Dairy Farm in the Manawatu.

The direct effect of an early winter grazing, which was immediately followed by a drainage event, on total N concentrations of drainage water was evident only during the first ~3 mm of a single drainage event. During this drainage episode there was a large initial increase in total N concentration of drainage water (~20 mg N litre<sup>-1</sup>), which quickly decreased to values comparable to drainage water from plots that had been grazed 7 days prior to the drainage occurring. The greater initial total N concentration

from the freshly grazed plots was attributable to greater initial concentrations of both total organic-N and ammonium-N. Although the total quantity of N lost in this single drainage event was similar for both treatments, being  $\sim 4 \text{ kg N ha}^{-1}$ , the forms of N lost differed. From the freshly grazed plots, the non  $\text{NO}_3^-$  forms of N (total organic-N and ammonium -N) were  $\sim 0.5 \text{ kg N ha}^{-1}$  greater, and were likely to be associated with the direct drainage or preferential flow of cattle urine. This loss attributed to direct leaching of urine-N is relatively small compared to the total annual quantities of N typically lost from dairy pastures in drainage ( $25\text{-}30 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ ).

**Keywords:** Nitrogen, cow urine, mole and pipe drainage, surface runoff, cattle grazing

## 8.2 INTRODUCTION

The number of field studies quantifying the contribution of intensive dairy farming to increased nutrient and suspended solid loads in New Zealand surface waterways is relatively small (e.g. Sharpley & Syers 1979; Monaghan *et al.* 2002, Houlbrooke *et al.* 2003(*Chapter 6*)). These studies have established that grazing dairy cattle are a cause of nutrient enrichment of artificial drainage water. Artificial drainage, such as the mole-pipe systems commonly used in poorly drained soils in New Zealand, may exacerbate the risk of nutrient leaching. They can promote rapid macropore flow to the tile drain ensuring that the drainage water bypasses a great deal of the soil matrix (Kladivko *et al.* 1991, Kohler *et al.* 2001). However, there is still limited information on what happens to the forms and quantities of nutrients lost when grazing coincides with drainage and/or runoff.

Approximately 70% of the N consumed by grazing dairy cattle is returned to the soil surface as dung and urine. With N concentrations in urine ranging from 8000 to 15000  $\text{mg N litre}^{-1}$  (Haynes and Williams 1993), urine patches provide large quantities of highly soluble, uncharged urea-N molecule which is prone to leaching (Silva *et al.* 1999; Silva *et al.* 2000; Di and Cameron 2002). The quantity of N deposited in a urine patch ( $\sim 0.4 \text{ m}^2$ ; Williams and Haynes 1994) is equivalent to a rate of  $\sim 1000 \text{ kg N ha}^{-1}$  (Silva *et al.* 1999). Urine patches occur randomly across grazed paddocks and at a stocking rate of 3 cows  $\text{ha}^{-1}$ , may cover  $\sim 25\%$  of the paddock area in one year (Silva *et al.* 1999).

A few studies have measured the forms of N in leachate under patches of cattle urine. Park (1996) found that total organic nitrogen (TON) and ammonium-N ( $\text{NH}_4^+$ -N) were

the dominant forms of N in leachate. The collected TON was likely to be predominantly derived from cow urine suggesting the likelihood of preferential flow of the urine applied. High concentrations of  $\text{NH}_4^+\text{-N}$  are to be expected as urea has an estimated half-life of only 3-4 hours (Haynes and Williams, 1993) and is rapidly hydrolysed to the ammonium form once applied to the soil surface. Silva *et al.* (2000) demonstrated that in water saturated soil, applied cattle urine leached through the soil in the form of unhydrolysed urea. Condon *et al.* (2004) partitioned the forms of N in soil, at a range of depths, following the application of a urea solution to simulate cow urine. On the day of the application, urea was the dominant form of N recovered. Between 1 and 16 days after the application,  $\text{NH}_4^+\text{-N}$  was the dominant form recovered, whilst between day 16 and day 32,  $\text{NO}_3^-\text{-N}$  was the dominant form recovered. Consequently, the form of N in drainage under urine patches will depend on how soon drainage occurs after grazing. If drainage occurs within 24 hours of grazing, then the form of urine N lost in drainage is expected to be predominately urea. However, when drainage occurs > 10 days after grazing then the form of urine derived N lost in drainage will mostly be as  $\text{NO}_3^-\text{-N}$  (Sharpley and Syers 1979; Houlbrooke *et al.* 2003 (*Chapter 6*)). Although,  $\text{NH}_4^+\text{-N}$  is expected to be the dominant form of urine derived N in the soil between 1 and 10 days after a grazing, this form of N is less susceptible to leaching in drainage compared to both the urea and  $\text{NO}_3^-$  forms of N.

While there has been no work on runoff of urine-N, the form of N lost in surface runoff tends to be dominated by TON and  $\text{NH}_4^+\text{-N}$  rather than  $\text{NO}_3^-\text{-N}$ . Smith and Monaghan (2003) found that losses of  $\text{NH}_4^+\text{-N}$  in surface runoff were greater in the spring period which was attributable to increased grazing pressure creating soil treading damage during moist soil conditions. The increased volumes of surface runoff water that are induced by rainfall occurring after grazing by cattle have been attributed to compaction caused by cattle treading during grazing (Nguyen *et al.* 1998; Tian *et al.* 1998 Drewry & Paton 2000).

The major aim of this research was to determine changes in the forms and concentrations of nitrogen in drainage and surface runoff when drainage occurs immediately following grazing of pasture with dairy cattle.

## 8.3 MATERIALS AND METHODS

### 8.3.1 Site and soil

The site was established on a mole-pipe drained soil in the Manawatu region of New Zealand (NZMS 260, T24, 312867) to investigate nutrient leaching losses from grazed dairy pasture. The site was located on a Tokomaru silt loam soil on Massey University's No. 4 Dairy Farm. The soil was classified as an Argillic-fragic Perch-gley Pallic Soil (Hewitt 1998) or a Typic Fragiaqualf (Soil Survey Staff 1998), and was derived from loess blown from adjacent river sediments to be deposited on a deeply dissected uplifted marine terrace (Molloy 1991). The Tokomaru silt loam soil consists of a weak to moderately developed, brown, silt loam A-Horizon (~ 0 - 250 mm soil depth); a weakly developed, grey, strongly mottled, clay loam B-Horizon (~ 250 - 800 mm soil depth); and a highly compacted, weakly-developed, pale-grey, silt loam fragipan C-Horizon. This fragipan acts as a natural barrier to drainage (Scotter *et al.* 1979a). The site is located in a flat to easy rolling landscape (~ 3% slope), which receives an average annual rainfall of ~ 1000 mm. The site supports a mixed pasture of perennial ryegrass (*Lolium perenne*) and white clover (*Trifolium repens*).

The research area consisted of 8 plots (each 40 m x 40 m). Each plot had an individual mole-pipe drain network. Mole drains were installed at a depth of 45 cm and at 2 metre intervals. Drainage from the mole network on each plot was intercepted by a perpendicular collecting pipe drain at a depth of 60 cm at one edge of the plot. At the corner of each plot a pit was excavated and a V-notch weir placed at the exit of the pipe drain to continuously monitor drainage flow rates with data loggers. Drainage water was sampled manually from the pipe outlet for determination of N concentrations. Isolated sub-plots (each 5 m x 10 m) were installed in four of the research plots for the measurement of surface runoff. Surface runoff water collected from each sub-plot passed through a tipping bucket instrument to measure flow volume and to provide a flow proportioned and mixed sample for water quality analysis.

### 8.3.2 Experimental procedure

During the 2003 winter drainage season, two drainage events that occurred immediately following grazing events were monitored (Experiment 1 & Experiment 2). In Experiment 1, a natural rainfall event (36 mm) occurred on the 6<sup>th</sup> of June (beginning

of the drainage season) which caused drainage to start 2 hours after cows had grazed four of the experimental plots (grazing intensity of 90 cows ha<sup>-1</sup> for 12 hours). The other four adjacent plots were not grazed at this time, but had been grazed 7 days prior to rainfall and subsequent drainage, so were used as a comparison for the freshly grazed plots.

Experiment 2 was conducted on two plots at the end of the drainage season, and involved two drainage events (Experiment 2a & 2b) generated using a rainfall simulator (with a coverage of 150 m<sup>2</sup>), which reduced the effective drainage plot area to 150 m<sup>2</sup>. Surface runoff also occurred at each of these drainage events. The drainage events occurred 2-4 hours following grazing (Experiment 2a; 7<sup>th</sup> of October), and two days after grazing (Experiment 2b; 9<sup>th</sup> of October). The pasture was grazed with 130 cows ha<sup>-1</sup> for 12 hours. The quantity of rainfall required to cause drainage was estimated using a soil water balance model developed by Scotter *et al.* (1979b).

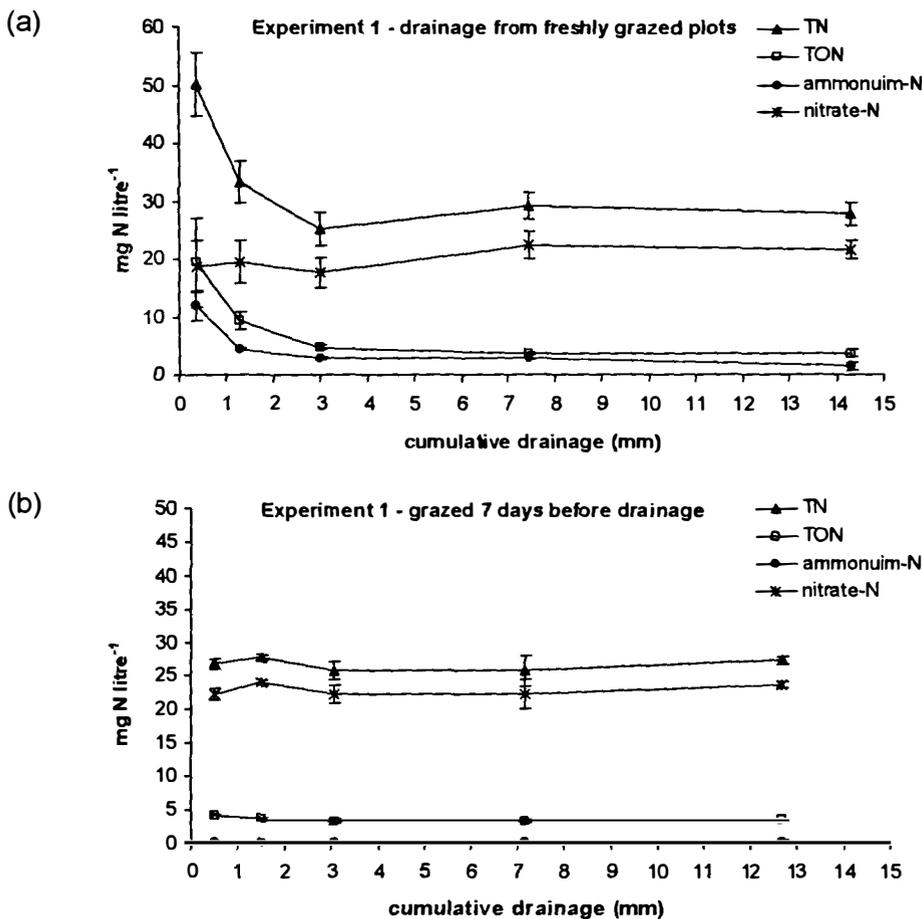
### 8.3.3 Laboratory analyses

Drainage and surface runoff water samples were analysed with a Technicon Auto Analyser II using the following methods: Kamphake *et al.* (1967) for nitrate-N (NO<sub>3</sub><sup>-</sup>-N); Searle (1975) for ammonium-N (NH<sub>4</sub><sup>+</sup>-N) and McKenzie & Wallace (1954) for Kjeldahl N. Total N (TN) was calculated by adding NO<sub>3</sub><sup>-</sup>-N concentrations to both the Kjeldahl N concentration and the dissolved Kjeldahl N concentrations. Total organic N (TON) concentration was calculated by subtracting NH<sub>4</sub><sup>+</sup>-N from Kjeldahl N. All measures of dissolved nutrients were determined on samples passed through a 1.2 µm glass filter. The variability in nutrient concentrations is quantified using the standard error of means (SEM) from 4 replicates (Experiment 1) or 2 replicates (Experiment 2).

## 8.4 RESULTS AND DISCUSSION

The 36 mm rainfall event occurring in Experiment 1 resulted in an average of 13.5 mm of drainage and no surface runoff. The TN concentration of water draining near the beginning of the event (~ 0.3 mm of accumulated drainage) from freshly grazed pasture was 50 mg N litre<sup>-1</sup> compared with 26 mg N litre<sup>-1</sup> from plots that had been grazed 7 days earlier (Fig 8.1a & 8.1b). Total N concentrations in drainage from the freshly grazed plots steadily decreased to about 25 mg N litre<sup>-1</sup> during the first ~3 mm of drainage, and remained at 25-30 mg N litre<sup>-1</sup> for the duration of the drainage event (Figure 8.1a). The large decrease in TN concentration during the first ~3 mm of

drainage from the grazed plots was due primarily to the marked decrease in TON and  $\text{NH}_4^+$ -N concentrations. Total organic N and  $\text{NH}_4^+$ -N made up the majority (about 60%) of TN at the start of the drainage event, but provided less than 30% of TN after the initial 3 mm of drainage. After 3 mm of drainage, the concentrations of all forms of N, except for  $\text{NH}_4^+$ -N, in drainage water were similar for both grazing treatments, with  $\text{NO}_3^-$ -N being the dominant form of N in drainage water. These relatively high levels of  $\text{NO}_3^-$ -N are consistent with other studies which show that the levels of  $\text{NO}_3^-$ -N in drainage tend to be high (typically 15-30  $\text{mg N litre}^{-1}$ ) early in the winter drainage season (Houlbrooke *et al.* 2003) (Figure 8.4). The flushing pattern evident in the first 3 mm when drainage immediately follows grazing may not be evident when grazing and drainage events are simultaneous.



**Figure 8.1.** The change in concentrations of different forms of N with cumulative drainage from a natural rainfall event (Experiment 1) on 06/06/03 for (a) freshly grazed plots and (b) plots that had been grazed 7 days before. Bars represent two SEM.

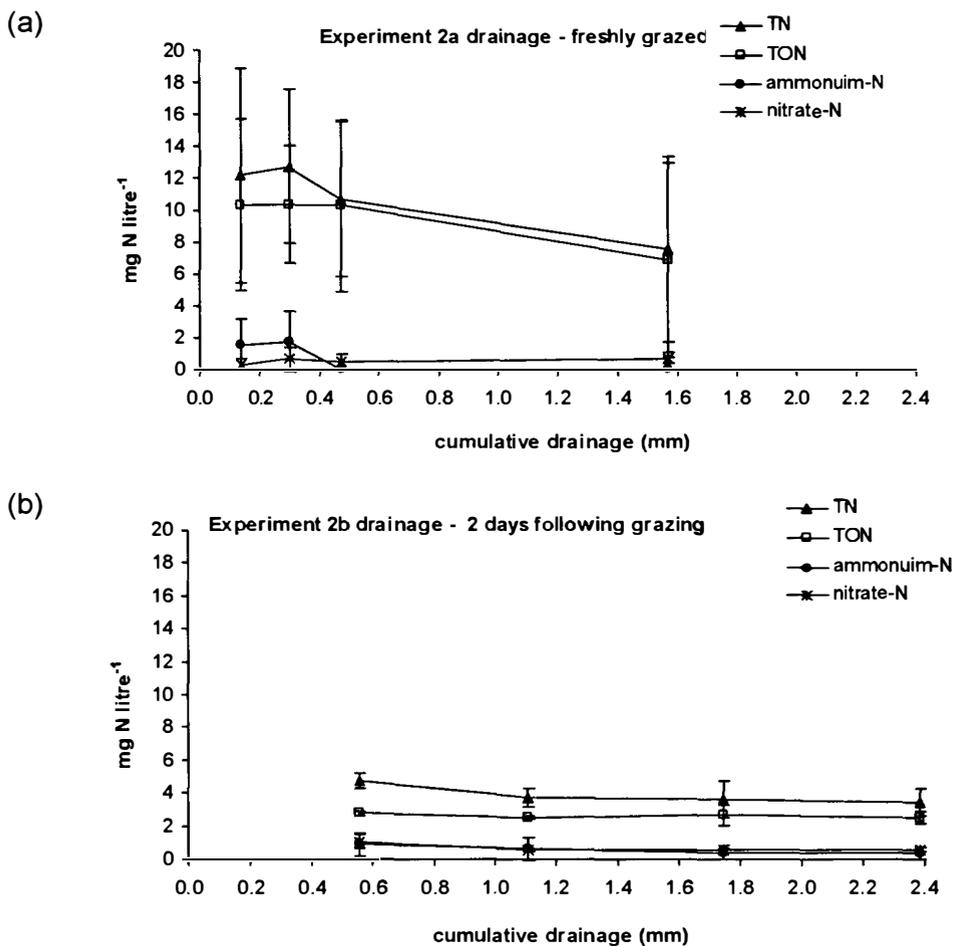
The total N drainage loss was similar for both grazing treatments being  $\sim 4 \text{ kg N ha}^{-1}$  (Table 8.1). However the treatments differed in the forms of N lost. The combined TON and  $\text{NH}_4^+\text{-N}$  drainage loss was  $\sim 0.5 \text{ kg N ha}^{-1}$  higher from the freshly grazed plots compared with plots grazed 7 days before drainage occurred. This greater loss of TON and  $\text{NH}_4^+\text{-N}$  from the freshly grazed treatment suggests that these losses were associated with the direct loss of urine-N. Some further loss of TON may be attributable to organic carbon priming effect from urea releasing organic N. This loss is relatively small compared to the total annual quantities of N typically lost from dairy pastures in drainage ( $25\text{-}30 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ ; Houlbrooke *et al.* 2003 (*Chapter 6*)). On the plots grazed 7 days prior to the drainage event,  $\text{NO}_3^-\text{-N}$  loss represented a larger proportion of total N drainage loss. This apparent increased loss of  $\text{NO}_3^-\text{-N}$  may be due to the occurrence of nitrification during the 7 days between grazing and drainage.

**Table 8.1.** Mean concentrations and losses of different forms of N in drainage on 06/06/03 (Experiment 1) from plots that had been freshly grazed compared with plots that had been grazed 7 days before the drainage event. Standard errors of means (SEM) are presented in brackets. TN = total N, TON = total organic N.

Nitrogen form	Concentration ( $\text{mg litre}^{-1}$ )		Nitrogen Loss ( $\text{kg ha}^{-1}$ )	
	Freshly grazed	Grazed 7 days before	Freshly grazed	Grazed 7 days before
TN	28.8 (2.0)	26.5 (0.9)	4.1 (0.1)	4.0 (0.7)
$\text{NO}_3^-\text{-N}$	21.6 (1.8)	22.8 (1.0)	3.1 (0.1)	3.5 (0.6)
$\text{NH}_4^+\text{-N}$	2.34 (0.41)	0.18 (0.09)	0.3 (0.04)	0.02 (0.004)
TON	4.79 (0.25)	3.43 (0.04)	0.7 (0.06)	0.5 (0.09)

In Experiment 2, 12 mm of artificial rainfall was applied at a rate of  $1.3 \text{ mm min}^{-1}$  to two drainage plots on 7 October (Experiment 2a) and again on 9 October 2003 (Experiment 2b). At the first artificial rainfall event, the predicted soil moisture deficit was 5.5 mm, and 2.7 mm of drainage and 3.9 mm of surface runoff were measured. At the second rainfall simulation, the predicted soil moisture deficit was 2.9 mm, and 5.8 mm of drainage and 3.7 mm of runoff were recorded. The N composition of drainage from the freshly grazed plots in this experiment (Figure 8.2a) suggested a similar trend to that measured for drainage from the freshly grazed plots in Experiment 1. In both of these experiments, TON,  $\text{NH}_4^+\text{-N}$ , and TN concentrations decreased steadily over the initial stage of the drainage event ( $<3 \text{ mm}$ ). As with Experiment 1, the  $\text{NO}_3^-\text{-N}$  concentrations in Experiment 2 remained relatively constant throughout the duration of the drainage event. Figure 8.2b shows the concentrations of N in drainage following artificial rainfall

applied two days after grazing. The main differences, compared with the freshly grazed scenario, were reductions in TN and TON concentrations.

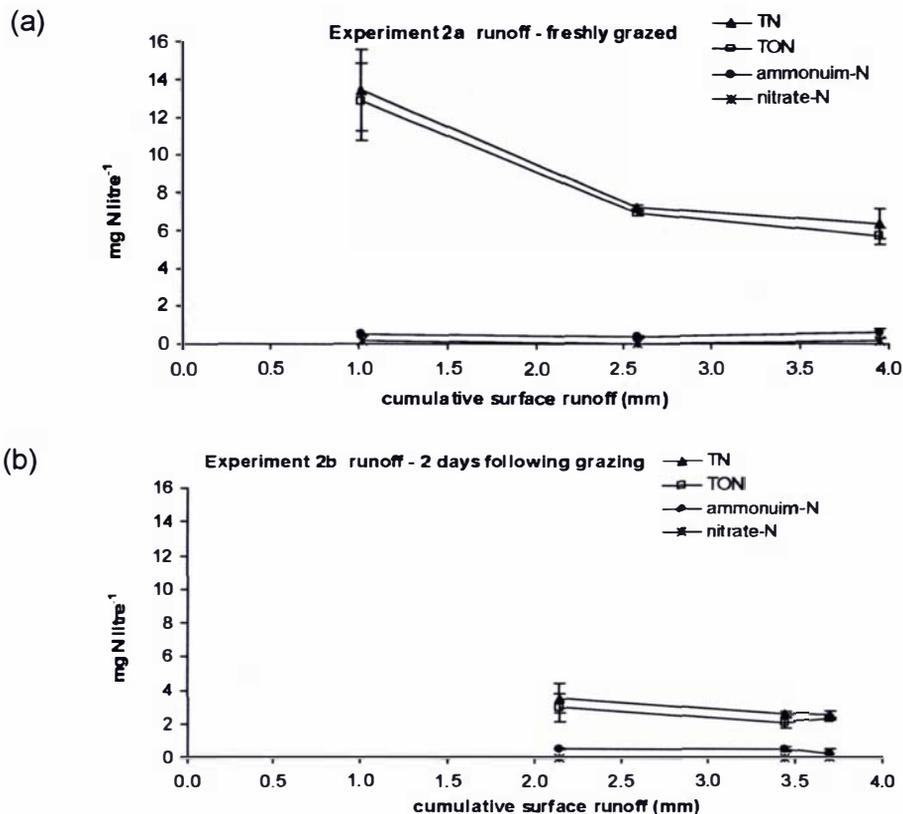


**Figure 8.2.** The change in concentrations of different forms of N with cumulative drainage from two artificial rainfall events (Experiment 2) on (a) 07/10/03 for freshly grazed plots and (b) 09/10/03 for the same plots two days following grazing. Bars represent two SEM.

During Experiment 2, surface runoff occurred with both artificial rainfall events. TN concentrations in runoff from the freshly grazed plots were predominantly in the form of TON. Also, both TN and TON concentrations decreased over the first ~3 mm of runoff (Figure 8.3a). Figure 3b shows that when artificial rainfall caused surface runoff two days after grazing, TN was lower as a result of lower TON concentrations. However, because no samples were collected at the very beginning of these two surface runoff events, it was not possible to determine what N levels were like in the initial runoff. Concentrations and forms of N in surface runoff from freshly grazed plots were very

similar to those in drainage water with the exception of  $\text{NO}_3^-$ -N which was either very low or not detectable (Figure 8.2a & 8.3a).

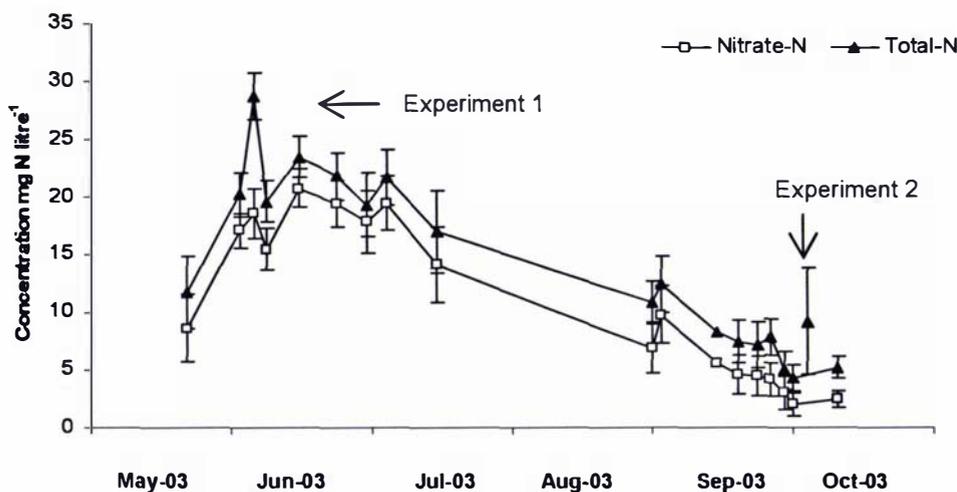
In Experiment 2, the concentrations of TON from the freshly grazed plots were considerably lower than those in initial drainage in Experiment 1, even though samples were collected for the first 0.1 mm of drainage. One possible explanation for the lower TON levels in Experiment 2 was the small plot size. In Experiment 1, the size of each of the plots was 1600 m<sup>2</sup> which was considerably larger than the 150 m<sup>2</sup> plot area used in Experiment 2 (i.e. under the rainfall simulator). As only a small area (< 5%) of a paddock is likely to receive urine at any one grazing, it is conceivable that by chance the smaller plots did not receive urine at rates comparable to the rest of the paddock.



**Figure 8.3.** The change in concentrations of forms of N with cumulative runoff from two artificial rainfall events (Experiment 2) on (a) 07/10/03 for freshly grazed plots and (b) 09/10/03 for the same plots two days following grazing. Bars represent two SEM.

The contributions of  $\text{NO}_3^-$ -N towards TN concentration in Experiments 1 and 2 were very different (TN ~ 85% for Experiment 1 and ~ 20% for Experiment 2). These differences can be largely explained by the seasonal variation in  $\text{NO}_3^-$ -N leaching (Figure 8.4). At the time of Experiment 1 in early June, only approximately 20 mm of

cumulative drainage had occurred and the average  $\text{NO}_3^-$ -N concentration in drainage was  $\sim 18 \text{ mg N litre}^{-1}$ . Experiment 2 was conducted in early October after approximately 200 mm of cumulative drainage had occurred over winter and the average  $\text{NO}_3^-$ -N concentration of drainage water was only  $\sim 3 \text{ mg N litre}^{-1}$ . The relative contributions of TON to TN in drainage associated with grazing events are higher later in the winter-spring season when  $\text{NO}_3^-$ -N concentrations are lower.



**Figure 8.4** The change in concentrations of total-N and  $\text{NO}_3^-$ -N (averaged for each drainage event) throughout the 2003 winter-spring drainage season. Arrows indicate when Experiments 1 & 2 occurred in relation to the winter-spring drainage period. The average total-N concentration of drainage from freshly grazed plots in Experiment 2 is indicated by an isolated solid triangle, as it involved an artificial rainfall event. Bars represent two SEM.

## 8.5 SUMMARY AND CONCLUSIONS

When subsurface drainage occurs immediately following grazing, drainage water may initially have higher total N concentrations due to the contribution of TON and  $\text{NH}_4^+$ -N from direct drainage or preferential flow of cattle urine. However, these elevated concentrations are short-lived and disappear following  $\sim 3 \text{ mm}$  of cumulative drainage. Although the total quantity of N lost in a single drainage event was similar whether drainage occurred either immediately or 7 days following grazing ( $\sim 4 \text{ kg N ha}^{-1}$ ), the forms of N lost differed. The non  $\text{NO}_3^-$  forms of N (total organic-N and ammonium-N) were  $\sim 0.5 \text{ kg N ha}^{-1}$  greater from the freshly grazed plots. This loss, which was attributed to direct leaching of urine-N, is relatively small compared to the total annual quantities of N typically lost from dairy pastures in drainage ( $25\text{-}30 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ ).

## 8.6 ACKNOWLEDGEMENTS

The authors are grateful for financial assistance from Dairy InSight, the C. Alma Baker Trust, Marley NZ Ltd, Spitfire Irrigators Ltd and Horizons Regional Council. The support of staff from Massey University's Institute of Natural Resources, No. 4 Dairy Farm and Drainage Extension Service is also gratefully acknowledged.

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# CHAPTER 9: IMPLEMENTING DEFERRED IRRIGATION OF FARM DAIRY EFFLUENT USING APSIM.

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## 9.1 INTRODUCTION

This chapter reports on a modelling exercise that simulated the soil water balance under a range of different FDE irrigation scenarios; the aim of which is to further explore the issues of FDE irrigation scheduling from a whole-farm perspective which was not considered in the earlier chapters (chapters 4 & 5). Specifically, the modelling exercise sets out to use the deferred irrigation criteria established in Chapter 4 to simulate the effect of deferred FDE irrigation at a whole-farm scale on potential pond overflow rates and storage requirements.

## 9.2 METHODOLOGY

### 9.2.1 Model setup

The model APSIM (Agricultural Production Systems Simulator, Keating *et al.* 2003) has been developed to run different soil-water simulations. APSIM is a process-based model with a modular framework that has been developed by the Agricultural Production Systems Research Unit (APSRU) from Australia. It was created to simulate biophysical processes in farming systems. The APSIM system uses a 'plug in' approach to adding or removing; modules intended to simulate either biological or physical farming system processes, data modules to provide information input, and management modules that specify rules used to control the simulation. A full description of APSIM can be found in Keating *et al.* (2003) with aspects particularly important to this work presented in Appendix 2.

For the simulations presented here, the soil water module is the most important module. The soil water module used in these simulations was APSWIM, a module based on the SWIM model (Soil Water Infiltration and Movement) developed by Peter Ross (Ross *et al.* 1992) and further described by Verburg *et al.* (1996). SWIM was

developed to simulate infiltration, evapotranspiration and redistribution, and more recently the ability to simulate the effect of sub-soil drainage has been included into APSIM-SWIM (N. I. Huth, CSIRO Sustainable Ecosystems, *pers com.*). See Appendix 2 for more detail.

Soil physical properties for the Tokomaru silt loam (as described in section 3.1) were taken from Scotter *et al.* (1979a) and Wilde (2003) and are given in Table A1. APSWIM was set up to account for mole drains at 2 m intervals and at 45 cm depth with a 35 mm diameter. A fragipan was present beyond 800 mm depth; therefore most of the winter drainage loss was via surface runoff or subsurface artificial drainage with very little deep drainage.

In order to simulate the effect of pasture growth on the soil water deficit, APSIM-Slurp was used. Slurp does not account for pasture growth as such but supplies user-set values for leaf area index, light extinction coefficient, and root length density to other modules in the simulation so as to estimate crop water demand and uptake. The weather data needed to drive the simulation were obtained from AgResearch's Palmerston North meteorological station situated at 34 m elevation above sea level with latitude 40 23S and longitude 175 37E. Thirty years of data, from 1974-2003, were used in the simulations. APSIM used internal calculations to calculate potential and actual evaporation. More detailed information on these settings is given in Appendix 2.

### **9.2.2 Model validation**

The simulation model APSIM was set up to estimate; soil water extraction by pasture, drainage through the fragipan, mole-pipe drainage, and evaporation from the soil surface based on soil hydraulic properties derived from the literature and inputs of plant characteristics and weather. Comparison was made with the soil water deficit data of Scotter *et al.* (1979a; 1979b) and Barker *et al.* (1985) and the drainage and runoff data presented in Chapters 6 and 7. Appendix 2 gives the results of these comparisons. The main deficiency of the APSIM model was its inability to predict the water contents that developed at depth under dry conditions. Such deficits are larger than those that could be accounted for by water extraction by pasture. These large deficits are likely to result from enhanced soil water evaporation caused by cracking of the soil, a process that is not represented in APSIM. Despite this, it is evident that the model performs

well in winter and spring, when it is required to schedule the irrigation of farm dairy effluent, and therefore can be considered to be suitable for this work.

### 9.2.3 Model scenarios

The objective of the modelling exercise is to run a number of different FDE irrigation scenarios to simulate the soil water balance, drainage, FDE generation and storage in an attempt to improve scheduling of FDE at a whole-farm scale with minimal impact on the aquatic environment. Irrigation depths of 15 mm, 30 mm and 45 mm were chosen for use in the model. As part of the conservative approach adopted here, a value of 15 mm was selected as the lowest application depth because it is the peak application depth that would be expected when a rotating machine was set at its fastest travel speed (chapter 5). Hence the 30 mm application depth represents a medium travel speed whilst the 45 mm scenario represents the depth likely if the irrigator was operated at its slowest travel speed. Whilst a 45 mm application depth appears excessive this is the depth applied on numerous farms where the farmer tries to reduce labour inputs to FDE irrigation by maximising the time between irrigator shifts.

Irrigations were scheduled to take place during the lactation season (1 August - 15 May) whenever a specified trigger soil water deficit was achieved and there was a suitable volume of FDE available in the pond. As a further safeguard against the direct drainage of partially treated FDE, the specified trigger soil water deficits were set at 5 mm more than the irrigation depth. Therefore, providing that there was sufficient FDE in the pond, irrigation occurred when the deficit was 20 mm, 35 mm and 50 mm (or greater) in the 15 mm, 30 mm and 45 mm irrigation scenarios, respectively. Irrigation rates were set in accordance with the recommended maximum rate for a silt loam of 10 mm hr<sup>-1</sup> (Heatley, 1996).

The simulations were run using the common setup described above to specifically represent the conditions present at the No. 4 dairy farm which has a large rainfall collection area due to the presence of the farm dairy yard and a large feed pad. In an attempt to identify a best management scenario, a fourth simulation was added. Here, 15 mm of FDE (low application depth) was applied and rainfall was collected only on the pond area. In this case, it was assumed that the dairy shed and feed-pad were roofed or rainfall on these areas was diverted from the pond. Any FDE remaining in the ponds at the end of the lactation season (15 May) was irrigated across the effluent block as it is an important criterion for implementation of deferred irrigation that the ponds are empty at the end of the season to enable storage of all the FDE generated in

winter from rainfall and to optimise the potential to store FDE generated in early lactation. All of the simulations are run over a 30 year period from 1974 to 2003.

Appendix 3 displays a copy of the pond manager logic used to simulate FDE volume, pond storage and irrigation. The pond manager logic was setup to implement 'deferred irrigation' criteria as described in Section 4.3.3. In brief, the pond stores all the FDE generated in winter and continues to store throughout spring until either a suitable soil water deficit exists to irrigate to the FDE block or the maximum pond storage is exceeded. Further irrigations of FDE are made throughout the lactation season as both soil water deficit and the generation of suitable volumes of FDE to fulfil the specified irrigation depth allow. The 2000 m<sup>3</sup> storage capacity of the 2<sup>nd</sup> pond at the No. 4 dairy has been used to define any pond overflow. When the storage of the pond is exceeded, but the deficit is insufficient for irrigation (i.e. smaller than the trigger value), FDE is allowed to overflow from the pond rather than being irrigated to land. The latter is more likely to happen on most farms in New Zealand as Regional Councils frown on unauthorised discharge from ponds. Pond overflow was allowed and calculated in this study in order to define the maximum storage capacity required under a range of FDE application depths whilst implementing the deferred irrigation criteria.

A number of reported outcomes from each irrigation scenario are presented as results:

- The first potential irrigation day for the lactation season.
- Pond overflow due to incoming effluent exceeding available pond storage.
- Total irrigation depth for each season.
- Total number of irrigations per season.
- Total FDE volume per season.
- Additional drainage from applications of FDE.

APSIM is a one dimensional model and therefore cannot represent the whole 16 ha irrigation area, which needs a number of days to irrigate in its entirety. The model was, therefore, set up to represent the 4 ha that would be irrigated in one day, and which equates to 25% of the total FDE receiving area. Four "irrigation days" are required to apply FDE to the whole effluent block. Irrigation days 2-4 of any application round still require suitable soil moisture deficit and FDE availability to initiate an irrigation using deferred irrigation criteria. The area represented by irrigation days 2-4 was not modelled specifically but an approximate water balance was calculated to indicate when irrigation might take place and the volume of effluent extracted from the pond for the irrigation was accounted for. See Appendix 3 for details on how this was done. Following the first application of irrigation to the modelled area (irrigation day 1), further

applications of FDE could not be made to the modelled area until there was sufficient FDE generated and stored in the second pond to cover the rest of the 16 ha area at the required depth. It is acknowledged that this does not necessarily represent what would happen on a farm where part irrigations of smaller areas are likely to take place.

### 9.3 RESULTS AND DISCUSSION

Following model validation against both historical and measured data (see Appendix 2), the simulation was run for the 30 years between 1974 to 2003 using weather data from this period. The modelled summer soil water deficits range from approximately 160 to 120 mm with the soil returning to field capacity most seasons. During 6 of the 30 years, the soil water deficit fails to return to field capacity during the winter/spring period as a result of low rainfall.

In the first irrigation scenario, the impacts of applying 15 mm of FDE, a moderate application depth for a travelling irrigator, were simulated. Applications of 15 mm of FDE took place on a regular basis during the lactation season (August to May) with little individual effect on either spring time, or maximum summer, soil water deficits due to the regular but only small shift in soil water storage of 15 mm per event (typically equivalent to only 3-5 days of evapotranspiration (Figure 9.1 and 9.2).

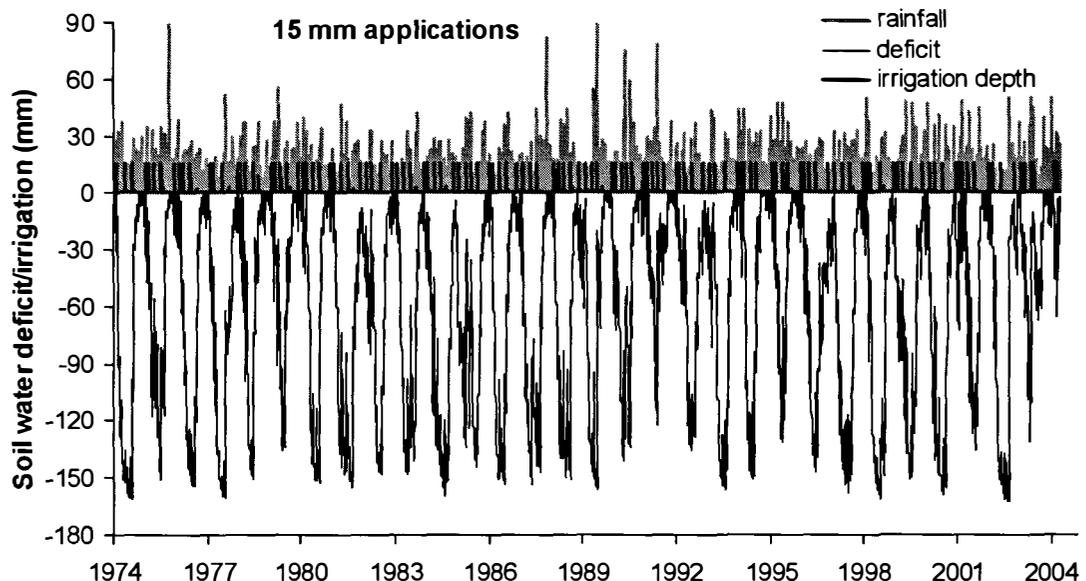
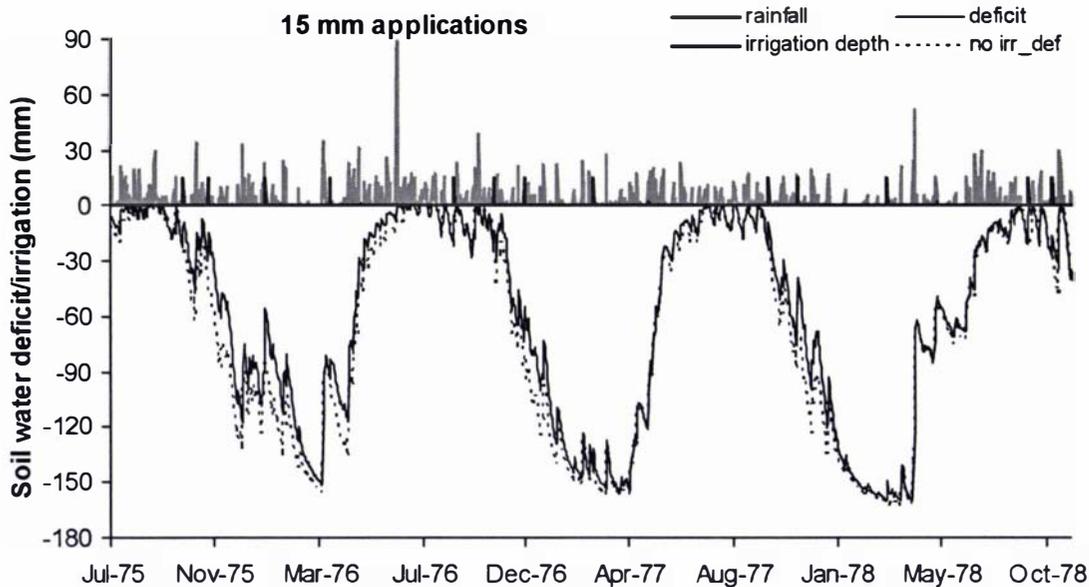


Figure 9.1. Simulated soil water deficits and 15 mm irrigation events over 30 years.



**Figure 9.2.** Simulated soil water deficits and 15 mm irrigation events over 3 years (1975-1978).

For the 15 mm irrigation scenario, the first spring (new lactation season) irrigation day, as determined by the soil water deficit, ranged from the 2 August to the 28 October with a mean first irrigation date of 7 September (Table 9.1). The average irrigation depth was 64 mm per lactation season with a range of 46 - 77 mm and an average of 4.2 irrigation events throughout the season (Table 9.1). In 20 out of 30 seasons, the 2000 m<sup>3</sup> ponds overflowed. The greatest volume of FDE that overflowed from the ponds was 3463 m<sup>3</sup> with an average overflow of 856 m<sup>3</sup> per season. This would suggest that, on average, a minimum storage pond of 3000 m<sup>3</sup> would be required to successfully practise deferred irrigation with an application depth of 15 mm. To prevent all overflow of FDE from the storage pond in this scenario would require a storage pond of 5500 m<sup>3</sup>, however, a pond size of 4000 m<sup>3</sup> would prevent overflow in all but 3 of the 30 years simulated (Table 9.1).

Additional drainage was calculated by subtracting seasonal deep drainage, mole and pipe drainage and surface runoff from a nil FDE applied scenario. The average extra drainage under applications of 15 mm of FDE was 20 mm per year (Table 9.1). The increase in drainage was not due to the direct loss of irrigated FDE as applications were only made when there was a suitable soil water deficit. The exception to this is when the ponds were emptied at the end of the season. Increased drainage was a result of changes to the soil water balance: as irrigation decreased the soil water deficit, drainage was more likely under subsequent rainfall events. In particular, the addition of 64 mm of FDE throughout the lactation season may bring the soil back to

field capacity earlier in late autumn/early winter and, therefore, result in additional winter drainage. This is most likely during wetter summer periods when plants are under little water stress or when a large application of FDE is made near the end of the lactation season when daily evapotranspiration rates are decreasing.

**Table 9.1.** Yearly irrigation simulation data from applications of 15 mm of FDE.

<b>15 mm FDE applications</b>						
<b>Season</b>	<b>Total Rainfall (mm)</b>	<b>Date of first irrigation</b>	<b>Irrigation depth (mm)</b>	<b>Total irrigations</b>	<b>Pond overflow (m<sup>3</sup>)</b>	<b>Additional drainage (mm)</b>
1974	1078	7/10/1974	62	4	1713	0
1975	1182	3/10/1975	62	4	1904	13
1976	1079	10/09/1976	62	4	1508	17
1977	714	15/10/1977	48	3	1783	22
1978	1131	10/09/1978	69	4	256	17
1979	969	22/10/1979	63	4	1406	33
1980	1070	26/08/1980	75	5	0	17
1981	840	8/09/1981	62	4	651	4
1982	868	11/08/1982	61	4	0	22
1983	856	19/08/1983	63	4	0	8
1984	795	20/08/1984	63	4	0	6
1985	983	7/08/1985	75	5	0	6
1986	980	27/09/1986	61	4	1284	28
1987	851	18/08/1987	61	4	0	27
1988	1130	28/10/1988	62	4	3463	26
1989	915	3/08/1989	63	4	0	23
1990	1054	11/09/1990	75	5	760	24
1991	968	7/09/1991	77	5	464	27
1992	1065	7/10/1992	61	4	1535	32
1993	802	3/08/1993	60	4	0	27
1994	1182	31/08/1994	76	5	151	23
1995	1279	28/10/1995	62	4	2700	26
1996	1003	24/09/1996	61	4	1349	12
1997	876	3/08/1997	63	4	0	18
1998	970	17/09/1998	63	4	1293	7
1999	889	3/09/1999	62	4	358	24
2000	869	4/08/2000	60	4	0	24
2001	1196	6/09/2001	77	5	490	44
2002	925	11/10/2002	46	3	2138	21
2003	1222	2/08/2003	76	5	462	33
<b>Mean</b>	<b>991</b>	<b>7-Sep</b>	<b>64</b>	<b>4.2</b>	<b>856</b>	<b>20</b>

The 30 mm FDE irrigation scenario represents a typical medium application depth for a travelling irrigator. The 30 mm FDE application scenario resulted in an average of only two irrigations per season with an average irrigation depth of 65 mm (Figure 9.3 and 9.4, Table 9.2). Using the deferred irrigation criteria, irrigation events were typically scheduled in mid to late spring and late summer with an often obvious effect on the soil water balance (Figure 9.4).

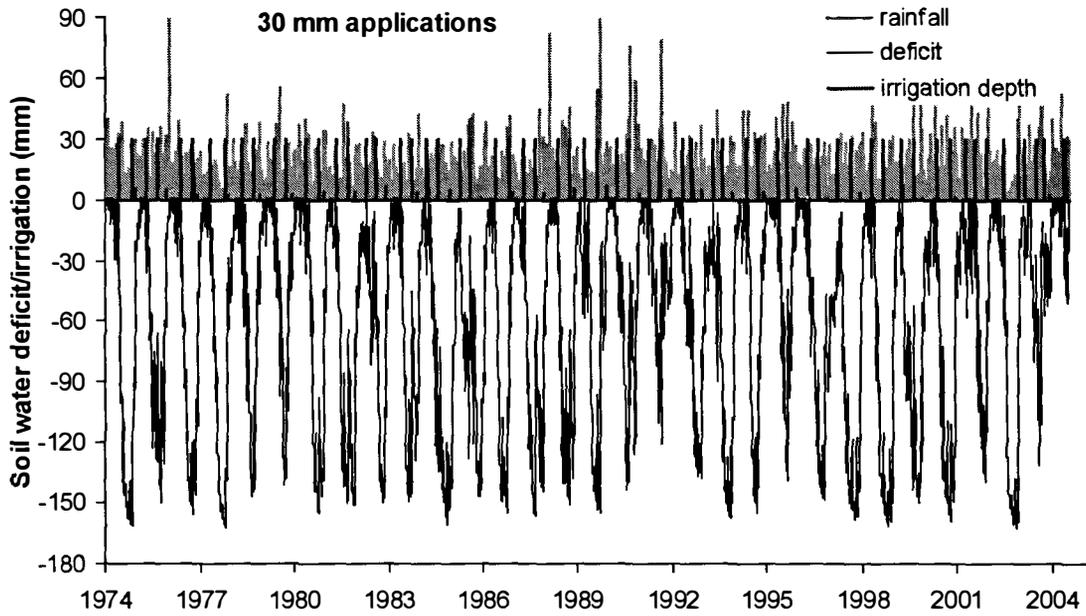


Figure 9.3. Simulated soil water deficits and 30 mm irrigation events over 30 years (1974-2003).

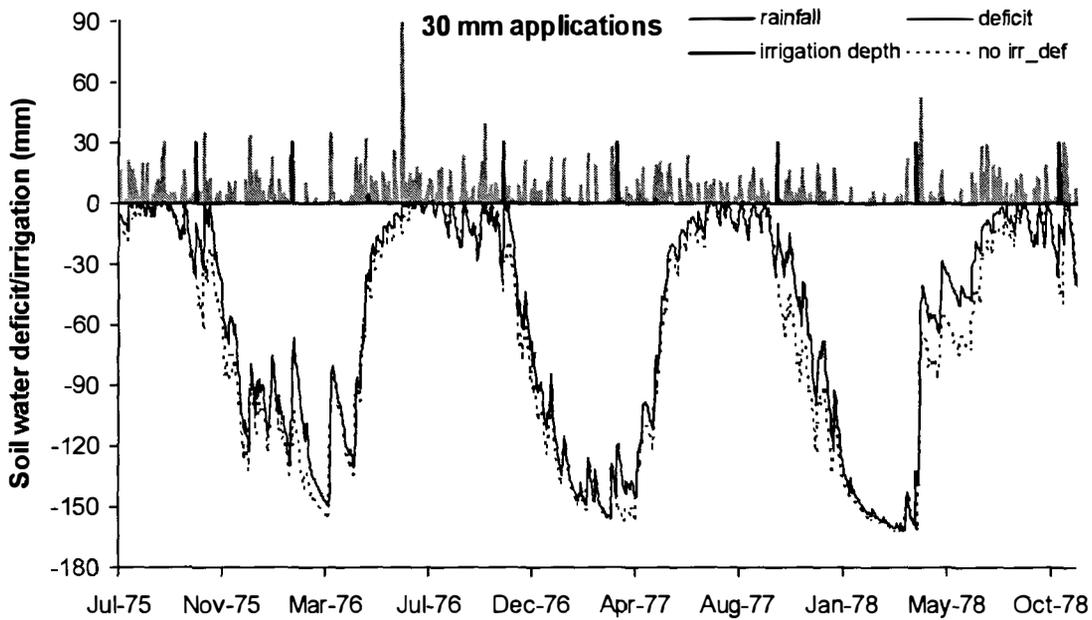


Figure 9.4. Simulated soil water deficits and 30 mm irrigation events over 3 years (1975-1978).

For the 30 mm irrigation scenario, the average first day of irrigation for the lactation season was the 6<sup>th</sup> of October with a range from the 7<sup>th</sup> of August to the 20<sup>th</sup> of November (Table 9.2). The average volume of pond overflow of FDE was 1541 m<sup>3</sup>.

The current pond (2000 m<sup>3</sup>) was breached in all but two of the 30 years. A storage pond of 4000 m<sup>3</sup> would prevent overflow in 20 of the 30 years. The greatest annual volume lost to pond overflow was 3870 m<sup>3</sup>. Therefore, 6000 m<sup>3</sup> of storage would be required to successfully practise deferred irrigation with an application depth of 30 mm. The additional drainage occurring as a result of applications of 30 mm of FDE was on average 5 mm per year (Table 9.2).

**Table 9.2.** Yearly irrigation simulation data from applications of 30 mm of FDE.

<b>30 mm FDE applications</b>						
<b>Season</b>	<b>Total Rainfall (mm)</b>	<b>Date of first irrigation</b>	<b>Irrigation depth (mm)</b>	<b>Total irrigations</b>	<b>Pond overflow (m<sup>3</sup>)</b>	<b>Additional drainage (mm)</b>
1974	1078	8/11/1974	65	2	3013	8
1975	1182	9/10/1975	64	2	2032	1
1976	1079	5/11/1976	62	2	3870	5
1977	714	20/10/1977	62	2	1865	1
1978	1131	13/10/1978	62	2	1312	3
1979	969	29/10/1979	62	2	1554	11
1980	1070	29/10/1980	64	2	2379	0
1981	840	23/10/1981	63	2	2300	0
1982	868	26/09/1982	66	2	325	0
1983	856	31/10/1983	63	2	1765	4
1984	795	10/09/1984	65	2	251	8
1985	983	19/09/1985	60	2	538	0
1986	980	8/11/1986	63	2	2859	0
1987	851	4/10/1987	63	2	293	27
1988	1130	2/11/1988	66	2	3563	1
1989	915	28/08/1989	67	2	0	4
1990	1054	17/09/1990	63	2	871	0
1991	968	14/09/1991	65	2	605	0
1992	1065	12/10/1992	64	2	1625	0
1993	802	23/09/1993	63	2	102	0
1994	1182	26/10/1994	64	2	2605	0
1995	1279	31/10/1995	66	2	2745	24
1996	1003	27/09/1996	61	2	1393	14
1997	876	1/09/1997	63	2	0	4
1998	970	3/10/1998	60	2	1758	0
1999	889	13/09/1999	61	2	598	5
2000	869	24/10/2000	61	2	1649	0
2001	1196	15/09/2001	90	3	719	22
2002	925	20/11/2002	60	2	3466	5
2003	1222	7/08/2003	92	3	189	20
<b>Mean</b>	<b>991</b>	<b>6-Oct</b>	<b>65</b>	<b>2.1</b>	<b>1541</b>	<b>5</b>

The 45 mm irrigation scenario represents a high rate of application of FDE by a travelling irrigator. On average, there were only 1.2 applications per season for this scenario with a mean total depth of 57 mm (Table 9.2). The main application typically

occurred in late spring with a further smaller application to empty the pond at the end of the season (Figures 9.5 and 9.6). In between these times, the required large volume of FDE is irrigated on the remaining 75% of the FDE block which still counts as the same rotation. Whilst this is also true of the two proceeding scenarios, the 45 mm application depth requires a greater generation of FDE between irrigations, hence increasing the rotation time between different irrigation days.

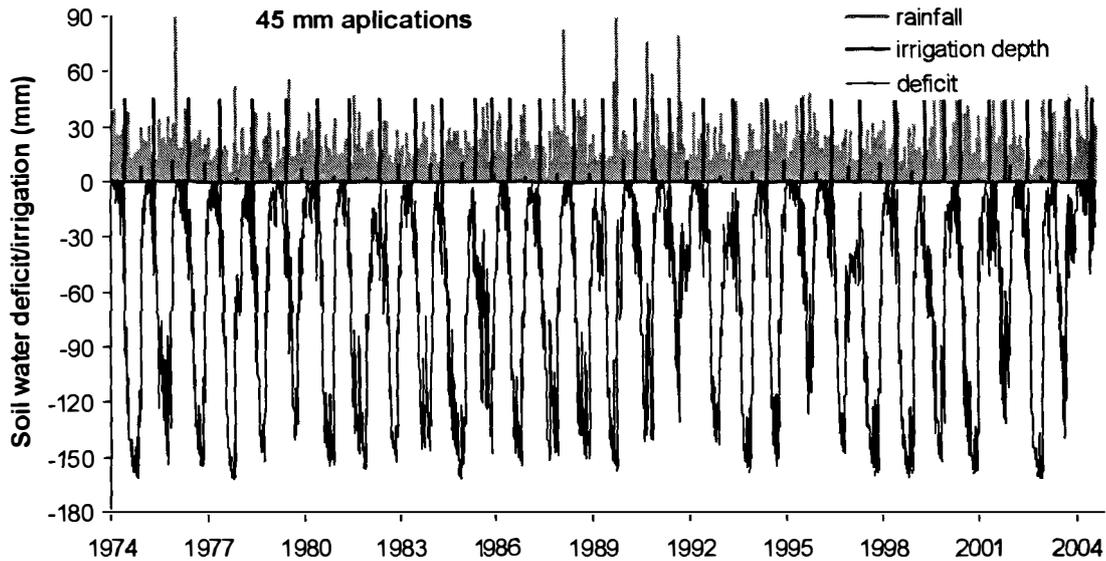


Figure 9.5. Simulated soil water deficits and 45 mm irrigation events over 30 years.

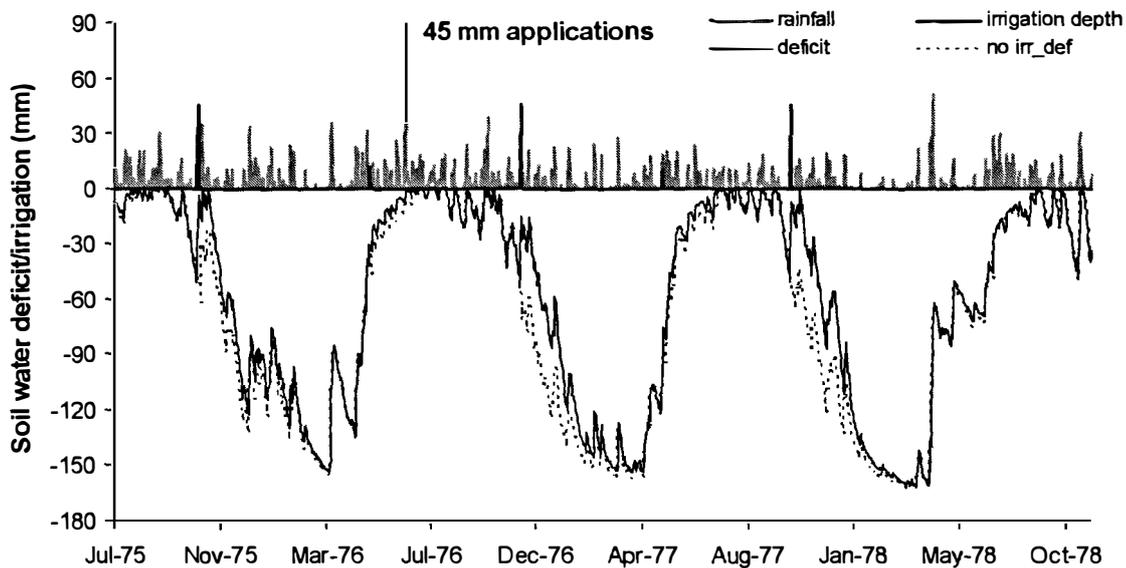


Figure 9.6. Simulated soil water deficits and 45 mm irrigation events over 3 years (1975-1978).

The average first day of irrigation of 45 mm depth for the lactation season was the 23<sup>rd</sup> of October with a range from the 19<sup>th</sup> of August to the 6<sup>th</sup> of December (Table 9.3). The average volume of pond overflow was 2132 m<sup>3</sup>. The current 2000 m<sup>3</sup> pond was

breached every year of the 30 year long simulation with the exception of 1997. A storage pond of 4000 m<sup>3</sup> would prevent overflow in only 14 of the 30 years. The greatest volume lost to pond overflow was 4432 m<sup>3</sup>. Therefore, 6500 m<sup>3</sup> of storage would be required to successfully practise deferred irrigation with an application depth of 45 mm. The additional drainage occurring as a result of applications of 45 mm of FDE was on average 8 mm per year (Table 9.3).

**Table 9.3.** Yearly irrigation simulation data from applications of 45 mm of FDE.

<b>45 mm FDE applications</b>						
<b>Season</b>	<b>Total Rainfall (mm)</b>	<b>Date of first irrigation</b>	<b>Irrigation depth (mm)</b>	<b>Total irrigations</b>	<b>Pond overflow (m<sup>3</sup>)</b>	<b>Additional drainage (mm)</b>
1974	1078	17/11/1974	52	1	3264	0
1975	1182	14/10/1975	56	1	2121	16
1976	1079	22/11/1976	55	1	4432	0
1977	714	24/10/1977	47	1	1936	3
1978	1131	9/11/1978	55	1	2402	0
1979	969	11/11/1979	52	1	1978	4
1980	1070	5/11/1980	48	1	2549	0
1981	840	2/11/1981	47	1	2557	0
1982	868	18/10/1982	55	1	964	0
1983	856	20/11/1983	56	1	2429	0
1984	795	23/09/1984	55	1	594	21
1985	983	25/09/1985	94	2	640	0
1986	980	12/11/1986	47	1	2924	0
1987	851	29/10/1987	49	1	1264	0
1988	1130	10/11/1988	49	1	3767	0
1989	915	23/09/1989	57	1	401	17
1990	1054	23/09/1990	96	2	979	30
1991	968	28/10/1991	56	1	2037	0
1992	1065	16/11/1992	47	1	3130	0
1993	802	20/10/1993	50	1	922	11
1994	1182	31/10/1994	56	1	2697	9
1995	1279	6/12/1995	50	1	4090	0
1996	1003	31/10/1996	52	1	2719	0
1997	876	23/09/1997	55	1	0	32
1998	970	17/11/1998	51	1	3871	0
1999	889	4/10/1999	90	2	1227	30
2000	869	29/10/2000	46	1	1737	0
2001	1196	26/09/2001	99	2	1122	28
2002	925	26/11/2002	48	1	3593	0
2003	1222	19/08/2003	91	2	1604	40
<b>Mean</b>	<b>991</b>	<b>23-Oct</b>	<b>59</b>	<b>1.2</b>	<b>2132</b>	<b>8</b>

The 15 mm irrigation scenario was modelled again with a 'rainfall diverted' option. Diverting the rain that falls on the farm dairy and feed-pad from the pond by the use of such means as a roof is considered best management as it decreases the volume of effluent generated. For the 15 mm irrigation/with diversion scenario, FDE applications

were spread throughout the lactation season (Figure 9.7 and 9.8), however, with less rainfall collected there were, on average, only 3 irrigations per season for a total of 48 mm applied. The average date for the first application was the 14 September. This is on average 7 days later than the original 15 mm irrigation scenario due to the time required to generate enough FDE to trigger an irrigation rather than soil water deficits not being suitable for irrigation scheduling.

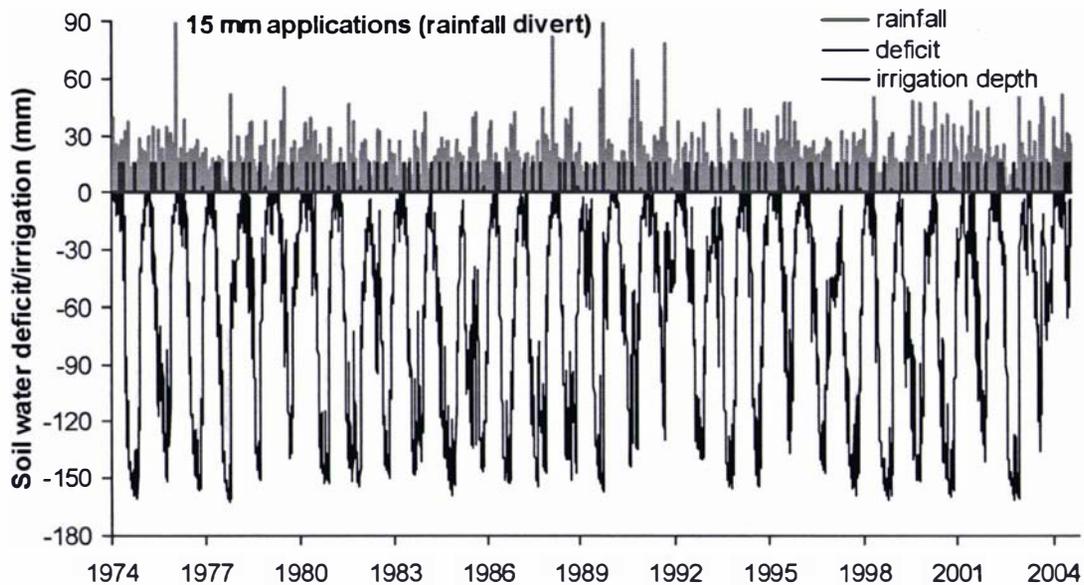


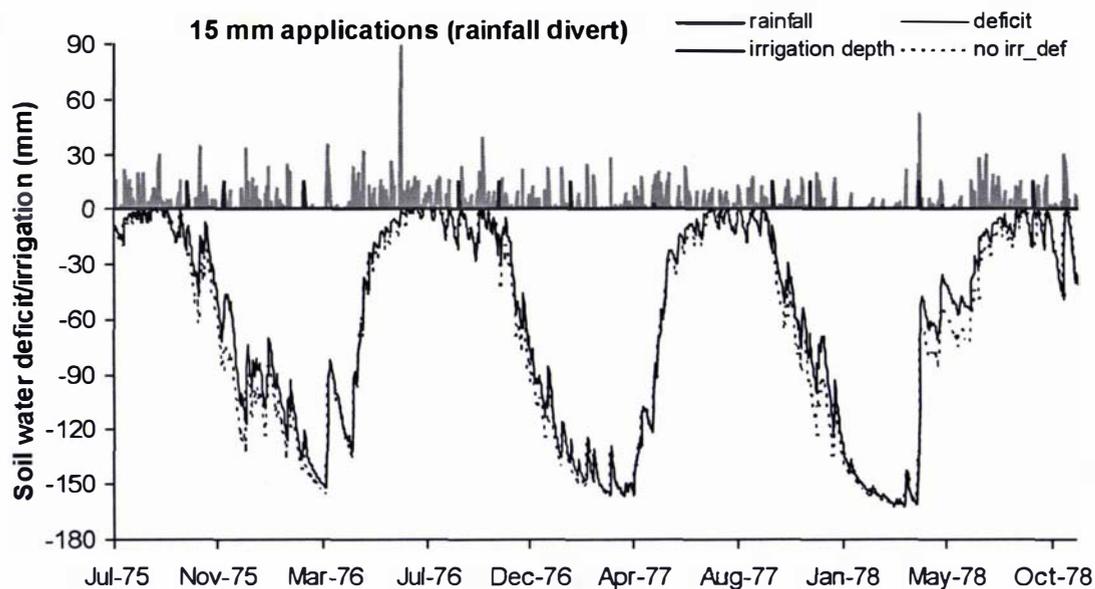
Figure 9.7. Simulated soil water deficits and 15 mm irrigation events with feed-pad and dairy shed rainfall diverted over 30 years.

With less FDE generated, there was less pond overflow: the average overflow was only 133 m<sup>3</sup> per season, approximately one-sixth of the overflow simulated for the original 15 mm irrigation scenario. The maximum overflow was 1250 m<sup>3</sup> in 1988. The current pond of 2000 m<sup>3</sup> would be breached in only 9 of the 30 seasons. For this scenario, a pond with storage of 3300 m<sup>3</sup> would be required to prevent all overflow of FDE. There was no additional drainage from the 15 mm application depth with rainfall diverted simulation.

**Table 9.4.** Yearly irrigation simulation data from applications of 15 mm of FDE with all rainfall on the dairy shed and feed-pad diverted.

<b>15 mm FDE applications (rainfall diverted)</b>						
<b>Season</b>	<b>Total Rainfall (mm)</b>	<b>Date of first irrigation</b>	<b>Irrigation depth (mm)</b>	<b>Total irrigations</b>	<b>Pond overflow (m<sup>3</sup>)</b>	<b>Additional drainage (mm)</b>
1974	1078	7/10/1974	45	3	265	0
1975	1182	3/10/1975	46	3	265	0
1976	1079	10/09/1976	48	3	0	0
1977	714	15/10/1977	47	3	325	0
1978	1131	10/09/1978	48	3	0	0
1979	969	22/10/1979	47	3	219	0
1980	1070	26/08/1980	48	3	0	0
1981	840	8/09/1981	48	3	0	0
1982	868	23/08/1982	47	3	0	0
1983	856	20/10/1983	45	3	230	0
1984	795	20/08/1984	45	3	0	0
1985	983	7/08/1985	48	3	0	0
1986	980	27/09/1986	45	3	0	0
1987	851	28/08/1987	45	3	0	0
1988	1130	28/10/1988	48	3	1250	0
1989	915	28/08/1989	46	3	0	0
1990	1054	11/09/1990	47	3	0	0
1991	968	7/09/1991	47	3	0	0
1992	1065	7/10/1992	46	3	160	0
1993	802	27/08/1993	48	3	0	0
1994	1182	31/08/1994	60	4	0	3
1995	1279	28/10/1995	48	3	847	0
1996	1003	24/09/1996	47	3	0	0
1997	876	30/08/1997	48	3	0	0
1998	970	17/09/1998	47	3	0	0
1999	889	3/09/1999	45	3	0	0
2000	869	22/09/2000	47	3	0	0
2001	1196	6/09/2001	60	4	0	0
2002	925	11/10/2002	47	3	416	0
2003	1222	2/08/2003	60	4	0	0
<b>Mean</b>	<b>991</b>	<b>14-Sep</b>	<b>48</b>	<b>3.1</b>	<b>133</b>	<b>0</b>

The current recommendation for an aerobic pond receiving FDE from a 500 cow herd is only 1440 m<sup>3</sup> (Heatly, 1996), falling well short of the simulated requirement to prevent pond overflow. However, this figure was not set to provide the storage required to implement deferred irrigation, but rather to allow the aerobic pond enough residence time to operate efficiently.



**Figure 9.8.** Simulated soil water deficits and 15 mm irrigation events with feed-pad and dairy shed rainfall diverted over 3 years (1975-1978).

When comparing the 4 different irrigation scenarios it can be seen that three of the four scenarios had similar total irrigation depths per season (Table 9.5). Total irrigation depth was limited by the volume of FDE generated, not the soil's ability to store the applied FDE. The 4<sup>th</sup> and contrasting scenario was the rainfall diverted scenario which produced less FDE, lower seasonal total application depths and, therefore, less additional drainage. The small differences in total effluent volume simulated between application scenarios were caused by random differences in the depth of irrigation applied at the end of the season when the ponds were emptied. For all of the four scenarios to receive exactly the same volume of FDE, the final irrigation must be the 4<sup>th</sup> of the 4 irrigations required to complete an irrigation rotation across the effluent block.

The most important differences between the four different irrigation scenarios are in the volume of pond overflow and in the first irrigation date, hence pond storage required to implement deferred irrigation. At the whole-farm scale, both of the 15 mm scenarios have considerably earlier first irrigation dates (on average, mid-September) than the 30 and 45 mm scenarios, which do not apply FDE until, on average, mid to late October. Subsequently, considerably greater volumes of FDE overflow from the current 2000 m<sup>3</sup> effluent storage pond in both the 30 and 45 mm FDE scenarios. The small difference of, on average, 4 days between the rainfall-diverted and rainfall-included 15 mm application scenarios is caused by the lack of FDE available to fulfil the required irrigation volume in the rainfall diverted scenario. However, the small volume of FDE

produced in the rainfall diverted scenario results in a much decreased volume of FDE overflowing the existing pond storage of 2000 m<sup>3</sup>. The practice of applying FDE little and often in 15 mm applications with all rainfall (with the exception of the pond area) diverted will result in the least pond overflow, lower pond storage requirements and the least subsequent winter drainage as a result of the FDE applications.

**Table 9.5.** Summary of irrigation data from four different simulations. 15 mm rf represents the rainfall divert scenario.

Effluent depth (mm)	Date of first irrigation	Range of first irrigation	Average irrigation depth (mm)	Average pond overflow (m <sup>3</sup> )	Average effluent volume (m <sup>3</sup> )
15 mm	7-Sep	02/08 - 28/10	64	856	11096
30 mm	6-Oct	07/08 - 20/11	65	1541	12041
45 mm	23-Oct	19/08 - 06/12	59	2132	11572
15 mm rf	14-Sep	02/08 - 28/10	48	133	7813

#### 9.4 CONCLUSIONS

The scheduling of FDE irrigation to land has a number of challenges and potential environmental impacts. This modelling exercise suggests that it will be difficult to successfully apply FDE in accordance with strict soil water based criteria, such as those used for 'deferred irrigation', while using ponds of limited storage capacity. This modelling exercise set out to determine the storage capacities required on a typical dairy farm (No 4 Dairy farm, Massey University) for a range of irrigation depths (trigger deficits). It was found that the most practicable way to manage deferred irrigation was by the adoption of a little and often policy, that is, by irrigating only a low application depth such as 15 mm. Irrigating this low depth in combination with diverting all rainfall from the farm dairy shed and feed-pad to decrease the potential volume of FDE generated was deemed to be best management practice.

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# CHAPTER 10: WHOLE-FARM NUTRIENT LOSS

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## 10.1 INTRODUCTION

The practice of deferred irrigation of FDE to grazed dairy pasture can markedly reduce nutrient losses in winter drainage and surface runoff when compared to daily irrigation of FDE. The experimental evaluation and detailed discussion of the impacts of FDE irrigation on nutrient losses to water ways were presented in Chapters 4, 6 and 7. In these Chapters, where comparisons were made between nutrient losses from the irrigated plots with those from non-irrigated plots, the data for losses was reported and discussed on a 'per ha' basis. Such comparisons run the risk of giving a false impression about the size of the contribution of land application of FDE to the loss of nutrients from the whole-farm. In thinking about the role of FDE irrigation in whole-farm nutrient losses, it is important to remember that only relatively small areas of the farm receive effluent. As a 'rule of thumb' only approximately 10% of a dairy farm's area will be irrigated with FDE (Heatley 1996, Cameron & Trenouth 1999).

The aim of this Chapter is to further explore the nutrient loss in artificial drainage and surface runoff (Chapters 6 & 7) from dairy cattle grazed pasture with and without FDE applications from a whole-farm perspective.

## 10.2 METHODOLOGY

To further explore issues related to FDE irrigation, including its contribution to whole-farm nutrient losses, the drainage and runoff plot losses reported in Chapters 4 and 7 were extrapolated to the whole-farm scale by scaling each nutrient loss component by the appropriate number of ha. The No. 4 dairy farm - 195 ha in area with a 16 ha effluent block - was used for this holistic assessment. N and P losses in winter drainage and surface runoff were taken from the 2003 winter-spring period (Chapter 7) and from deferred irrigations of FDE in the previous summer-autumn period (Chapter 4). Two-pond effluent loss data was taken from Chapter 4.

Three scenarios for the No 4 dairy farm are considered. In the first scenario, FDE is applied to land using the deferred irrigation system. The farm is divided into two areas – the main block (179 ha) and the effluent treatment block (16 ha). Measurements of nutrient losses from the non-effluent treatment plots (Chapter 7) were used to calculate losses for the main block (on a per ha basis) where standard dairying is practiced (Table 10.1). Nutrient loss measurements for the effluent treatments were used to calculate losses from the effluent block (standard dairy farming practice plus deferred irrigation of FDE). In the second scenario, FDE is treated using a two-pond system and then discharged to a stream (Table 10.1).

For comparative purposes, a third scenario, the farm with zero FDE, has been fabricated (Table 10.1). This scenario, called 'no-effluent' (grazing only), poses a hypothetical version of the farm where cows do not produce FDE and so there are no losses of nutrients to the aquatic environment from this farm as a result of FDE production. This scenario is similar in some ways to the two-pond case except that no account is made of the nutrients discharged from the pond. The purpose of this scenario is to allow a comparison between the relative contributions from a two-pond FDE treatment system, deferred irrigation of FDE and standard dairy farm practices to nutrient losses in drainage and runoff from the whole farm.

### 10.3 RESULTS AND DISCUSSION

Without FDE (grazed pastures only), the expected annual loss of TN in drainage from the entire farm (195 ha) would be 7235 kg N yr<sup>-1</sup> with a further 585 kg N yr<sup>-1</sup> contribution from surface runoff (i.e. 34.1 kg TN ha<sup>-1</sup> yr<sup>-1</sup> and 3 kg TN ha<sup>-1</sup> yr<sup>-1</sup>, respectively (section 7.4.3)). Application of FDE according to the deferred irrigation criteria would result in no additional loss of N from the farm in winter drainage (Chapter 7, section 7.4.5) and only 0.4 kg TN ha<sup>-1</sup> yr<sup>-1</sup> from direct losses at the time of application (Chapter 4, section 4.4.4). In comparison, discharge from a two-pond treatment system into a small stream would result in a contribution to whole-farm loss of 1500 kg N yr<sup>-1</sup>. This loss from ponds makes up 17% of the total nitrogen lost to the aquatic environment from the farm (Table 10.2). These simple calculations suggest that whilst a considerable quantity of N can be prevented from leaving a farm in drainage by utilising improved land treatment techniques for FDE, it is standard dairy farming practises (like intensive cattle grazing) that produce the majority of the N that is lost to the aquatic environment from the entire farm.

**Table 10.1.** *Source of nutrient losses reported for three different FDE management scenarios.*

Source of loss	Two-pond discharge to stream	Deferred irrigation land application	No Effluent Grazed only
Drainage - N	34.1 kg N ha <sup>-1</sup> x 195	34.1 kg N ha <sup>-1</sup> x 179 + 29.8 kg N ha <sup>-1</sup> x 16	34.1 kg N ha <sup>-1</sup> x 195
Drainage - P	0.73 kg P ha <sup>-1</sup> x 195	0.73 kg P ha <sup>-1</sup> x 179 + 1.47 kg P ha <sup>-1</sup> x 16	0.73 kg P ha <sup>-1</sup> x 195
Surface runoff - N	3.0 kg N ha <sup>-1</sup> x 195	3.0 kg N ha <sup>-1</sup> x 179 + 2.4 kg N ha <sup>-1</sup> x 16	3.0 kg N ha <sup>-1</sup> x 195
Surface runoff - P	0.6 kg P ha <sup>-1</sup> x 195	0.6 kg P ha <sup>-1</sup> x 179 + 0.66 kg P ha <sup>-1</sup> x 16	0.6 kg P ha <sup>-1</sup> x 195
Pond discharge - N	150 mg/L x 10000m <sup>3</sup>	-	-
Pond discharge - P	25 mg/L x 10000m <sup>3</sup>	-	-
Direct loss FDE - N	-	0.4 kg N ha <sup>-1</sup> x 16	-
Direct loss FDE - P	-	0.08 kg N ha <sup>-1</sup> x 16	-

**Table 10.2.** *Whole-farm losses of N in drainage and from treatment ponds under three different FDE management scenarios.*

Nitrogen loss (kg)	Two-pond discharge to stream	Deferred irrigation land application	No Effluent Grazed only
N lost in winter drainage	6650	6580	6650
N lost in winter surface runoff	585	575	585
Direct loss of FDE N	1500	6	0
Total farm N loss	8735	7161	7235
% of farm N loss from FDE	17	0.1	0

By using simulated data from Chapter 4 (section 4.4.1.2) a further scenario could be added looking at the possible contribution of FDE to whole-farm N loss under daily land application of FDE. This simulation presumed that there was no surface runoff loss of applied FDE. However, drainage of raw or partially treated FDE alone resulted in N loss of approximately 108 kg N yr<sup>-1</sup> or 1.5% of the total nitrogen lost in drainage and runoff from the farm.

In contrast to losses of N, the management of FDE has a significant effect on whole-farm P losses in drainage (Table 10.3). Without FDE (grazed pastures only), the expected annual loss of P in drainage from the farm would equate to 142 kg yr<sup>-1</sup> (i.e. 0.73 kg ha<sup>-1</sup> yr<sup>-1</sup>, Section 7.4.3). In addition, the expected annual loss of P in surface from the farm would equate to 117 kg yr<sup>-1</sup> (i.e. 0.6 kg ha<sup>-1</sup> yr<sup>-1</sup>, Section 7.4.3). By using the 'deferred irrigation' method for land application, FDE losses of 0.8 kg ha<sup>-1</sup> yr<sup>-1</sup> could be expected with a total of 13 kg yr<sup>-1</sup> from the 16 ha effluent block plus direct losses of 0.08 kg ha<sup>-1</sup> yr<sup>-1</sup> (1.3 kg yr<sup>-1</sup>) as a result of drainage at the time of FDE application. This represents 5% of the total P losses in drainage from the farm. The use of a two-pond treatment system would result in the discharge of 250 kg P yr<sup>-1</sup> into a small stream. This P exiting the ponds makes up nearly 50% of the total P lost from this scenario (Table 10.3). When a more sustainable FDE treatment system such as 'deferred irrigation' is utilised then it appears that the predominant contribution to P loss at a whole-farm scale is from standard dairy farming practise (like intensive cattle grazing). However, if a traditional two-pond treatment system discharges directly into a stream then P loss from FDE will be nearly equal to the losses of P expected under standard grazed dairy pasture.

**Table 10.3.** *Whole-farm losses of P in drainage and from treatment ponds under three different FDE management scenarios.*

Phosphorus loss (kg)	Two-pond discharge to stream	Deferred irrigation land application	No Effluent Grazed only
P lost in winter drainage	142	154	142
P lost in winter surface runoff	117	118	117
Direct loss of FDE P	250	13	0
Total farm P loss	509	285	259
% of farm P loss from FDE	49	5	0

Once again, by using simulated data from Chapter 4 (section 4.4.1.2), a further scenario could be added looking at the possible contribution of FDE to whole-farm P loss under daily land application of FDE. As for N loss, this simulation presumed that there was no surface runoff of applied FDE. In this scenario, direct drainage of raw or partially treated FDE alone resulted in P loss of approximately 18 kg P yr<sup>-1</sup> or 7% of the total P lost in drainage and runoff from the farm. Added to expected P loss in winter drainage and runoff, the total contribution of FDE- P to farm loss to waterways would be 12%.

Some caution is advised when P runoff data such as that presented here is extrapolated to the farm scale. Research from south-eastern Australia (Barlow *et al.* 2005) reported large differences between paddock and plot scale P runoff data when compared to whole-farm data. This suggests that caution is needed when interpreting P loss results generated from a scaling up from the paddock to the farm. The authors suggest that differences in climate, geomorphology and management are likely to impact on potential runoff generation processes. However, with this in mind, the exercise presented here was interesting in that it allowed a first assessment of the likely contribution of surface runoff loss to whole-farm loss.

#### 10.4 CONCLUSIONS

Standard dairy farm practices such as the grazing of cows make the greatest contribution to nutrient losses from dairy farms. The management of FDE plays a significant but more minor role in the likely loss of nutrients in drainage from the whole-farm. Under a sustainable FDE land treatment operation such as deferred irrigation, N loss could be expected to make a zero contribution to total N losses in drainage and runoff from the farm. In contrast, where a two-pond system is employed to treat FDE, 17% of the farm's N losses is accounted for in the pond discharge. The management of FDE plays an even greater role in the likely loss of P from the farm in drainage. Deferred irrigation of FDE would contribute 5% of the P losses in drainage and runoff from the farm. If the farm had a two-pond treatment system then nearly 50% of whole-farm P losses would be in the discharge from the ponds. As has been described in Chapter 4, the use of pond storage and strategic application of FDE at times of suitable soil moisture (deferred irrigation) is not standard practice on New Zealand dairy farms. Many dairy farms have no storage and are forced to apply FDE on a daily basis. Such a practise would increase the contribution of both N and P from FDE application at the whole-farm scale.

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# CHAPTER 11: SUMMARY

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## 11.1 INTRODUCTION

While it is widely acknowledged that intensive dairy farming in New Zealand is a major contributor to the increased nutrient loads in surface waters, there is little current research quantifying the magnitude and cause of nutrient loss from dairy farms to waterways from artificial drainage and surface runoff. Mole and pipe drainage provides an artificial subsurface drainage network to alleviate water-logging in naturally imperfectly and poorly drained soils. However, such mole and pipe systems can provide large macropores and preferential flow paths for rapid transport of nutrients to receiving water bodies. It has been 25 years since Sharpley and Syers (1979) demonstrated the impact of dairy cow grazing and urea application on the N and P concentrations in mole and tile drainage and runoff from a research area on the Massey No.4 Dairy Farm. At that time, the farm was only 2-3 years out of low fertility sheep pasture. In the intervening 25 years, N, P and S fertiliser inputs have increased as has grazing intensity and milk production. All of these developments, which are typical of the intensification of dairy farms throughout much of New Zealand, are likely to have increased nutrient losses to the aquatic environment.

The major objectives of this thesis were to; develop a sustainable land application treatment system for FDE (deferred irrigation), and; investigate the quantity and quality of artificial drainage and surface runoff water leaving artificially drained (mole and pipe system) soil. In particular the impact on water quality of three standard dairy farming practices has been investigated:

- Farm dairy effluent application,
- Dairy cattle grazing, and
- Urea application.

## 11.2 FDE MANAGEMENT USING DEFERRED IRRIGATION

A deliberately, poorly-timed application of 25 mm of FDE - simulating a 'daily' irrigation regime - to a soil which had an estimated deficit of only 6 mm, resulted in considerable drainage and runoff of partially treated FDE (Section 4.4.2). Approximately 70% of the

applied FDE left the experimental plots with 10 mm of drainage and 8 mm of surface runoff. The concentrations of N and P in this drainage were approximately 45% of those in the original FDE while the runoff had concentrations of N and P that were approximately 80% of those in the irrigated FDE. From this single irrigation event, a total of 12.1 kg N ha<sup>-1</sup> and 1.9 kg P ha<sup>-1</sup> was lost to surface water: these losses represent 45 % of expected annual N loss and 100% of expected annual P loss from normal, dairy grazed pasture.

In order to improve the current practice for land application of FDE, a new system was devised called 'deferred irrigation'. By eliminating the drainage of raw or partially treated, nutrient-rich FDE to receiving waterways, deferred irrigation affords greater environmental protection and ensures that land treatment is more sustainable (Section 4.4.3). In brief, the on farm implementation of deferred irrigation involves:

- Storage of FDE in a two-pond system throughout early lactation when the soil is at or near field capacity.
- Collection of daily climate data for simulation of the daily soil moisture deficit.
- Strategic scheduling of FDE irrigation when suitable soil water deficits exist.
- Removal of excess nutrients (particularly K) by the conservation of an annual pasture crop.

Deferred irrigation was successfully implemented at the field research site over three lactation seasons (2000 to 2003). Direct losses of nutrients to surface water via mole and pipe drainage and/or surface runoff were almost eliminated (Section 4.4.4). On average, 0.7 % of the total N and 0.3 % of the total P applied in FDE was lost per irrigation whilst the drainage volume was less than 1% of the total irrigation volume applied. In the 2001/2002 lactation season, the adoption of a 'little and often' policy resulted in no drainage and therefore none of the nutrients applied in FDE were lost directly.

The successful scheduling of FDE irrigation at a farm scale faces a number of challenges. For example, it would be difficult to successfully apply FDE at No 4 Dairy farm (475 cow herd) in accordance with strict soil water based criteria, such as those used for 'deferred irrigation', without exceeding the farm's existing pond storage volume (approx. 2000 m<sup>3</sup>) in some years. A modelling exercise – using the APSIM simulation model (Chapter 9) - was undertaken to look at the quantity of effluent that would overflow from the ponds at No 4 dairy farm under deferred irrigation. The impact of a range of FDE application depths on pond overflow was simulated for the past 30

years (Section 9.3). This modelling exercise found that applying a low irrigation depth of 15 mm allowed a first irrigation day of on average the 7<sup>th</sup> of September compared to the 6<sup>th</sup> of October for applying a depth of 30mm. Further more it was found that a 'little and often' policy for irrigating FDE (i.e. 15 mm or less) minimised overflow from the ponds. Best management practise was represented by irrigating this small depth in combination with diverting all rainfall from the farm dairy shed and feed-pad to decrease the potential volume of FDE generated.

### 11.3 FDE IRRIGATOR PERFORMANCE

The performance of travelling FDE irrigators has major implications for the successful implementation of the deferred irrigation system and, therefore, the potential nutrient losses in drainage and runoff water. The spatial variability in the depth of FDE applied by a standard rotating irrigator was very high; its circular throw pattern created a bimodal distribution pattern with a 2-3 fold difference between the highest and lowest application depth (Section 5.4.1). Attempts to improve the performance of these irrigators was made by adding splash plates to the end of the irrigators booms, and by diverting an additional ten percent of the flow of FDE into the low application area in the middle of the profile. Neither of these modifications to the rotating irrigator significantly improved the bimodal distribution pattern (Section 5.4.2). New technology in the form of an oscillating irrigator was evaluated and found to have a relatively uniform distribution pattern under calm or light wind conditions (Section 5.4.3). The variability in application depth under travelling irrigators needs to be taken into consideration when scheduling FDE irrigation.

A simple model that simulated the daily soil water balance, irrigator throw patterns and application depths was used to compare the quantity of direct drainage resulting from FDE applications using the rotating and oscillating irrigator (Section 5.4.4). The more uniform application pattern of the oscillating irrigator offered the best chance of avoiding direct drainage of FDE when the soil moisture deficit was low in early spring and late autumn. During this period of low soil water deficits, all irrigators should be set to travel at their fastest speed (lowest application depth) to minimise the potential for direct drainage of partially treated FDE and associated nutrient losses.

## 11.4 NUTRIENT LOSSES UNDER INTENSIVE DAIRY FARMING

The average concentrations of N and P in both 2002 (Chapter 6) and 2003 (Chapter 7) winter mole-pipe drainage water from grazed dairy pastures were all well above the levels required to prevent aquatic weed growth in fresh water bodies. Total nutrient losses from plots representing standard dairy farm grazing practice (i.e. no FDE applied) were 28 kg N ha<sup>-1</sup> and 34 kg N ha<sup>-1</sup> for the 2002 (Section 6.4.3) and 2003 (Section 7.4.3) seasons, respectively. Total P losses for 2002 (Section 6.4.4) and 2003 (Section 7.4.3) were 0.35 kg P ha<sup>-1</sup> and 0.7 kg P ha<sup>-1</sup>, respectively. Surface runoff, measured in 2003 (Section 7.4.3), contributed a further 3.0 kg N ha<sup>-1</sup> and 0.6 kg P ha<sup>-1</sup>.

A number of common dairy farm practices increased the losses of N and P in the artificial drainage water. Cattle grazing increased N losses in drainage. The increased quantities of N lost in a single drainage event (06/06/03) as a result of grazing (~ 0.5 kg N ha<sup>-1</sup>) were similar whether drainage occurred immediately after, or 7 days following grazing (100 cows ha<sup>-1</sup> for 12 hours), however, the interval between grazing and rainfall influenced the form of N lost (Section 8.4). When drainage occurred immediately following grazing, drainage water initially had large total N concentrations due to the contribution of TON and NH<sub>4</sub><sup>+</sup>-N from the preferential flow of cattle urine. However, these very high concentrations were short-lived and disappeared following ~3 mm of cumulative drainage. This increase in N loss attributed to direct leaching of Urine -N is relatively small compared to the total annual quantities of N lost in drainage (Section 6.4.3 & 7.4.3). A grazing event of approximately 100 cows ha<sup>-1</sup> for a 12 hour period occurring approximately 10 days prior to a drainage event increased NO<sub>3</sub><sup>-</sup>-N and DIP concentrations in drainage by approximately 5 mg N litre<sup>-1</sup> (Section 6.4.3 & 7.4.6) and 0.1 mg P litre<sup>-1</sup> (Section 6.4.4 & 7.4.6), respectively.

The impact of an early spring application of urea in 2002 (section 6.4.3) on drainage water quality was minimal, suggesting that careful timing of urea applications at this time of year may result in little nitrogen leaching. In contrast, a urea application (at a rate of 37 kg N ha<sup>-1</sup>) in mid June of 2003 (Section 7.4.7), when plant uptake of mineral N was low, resulted in an increased loss of NO<sub>3</sub><sup>-</sup>-N of approximately 5 mg litre<sup>-1</sup> throughout 80 mm of cumulative drainage (an increased loss of ~4 kg N ha<sup>-1</sup>).

There were no significant increases in total N concentrations or losses in the 2002 (Section 6.4.3) and 2003 (Section 7.4.5) winter drainage water when FDE was applied in the previous summer to soils with suitable water deficits (Deferred irrigation).

Deferred FDE irrigation, however did cause increases in total P losses in 2002 (0.52 kg P ha<sup>-1</sup>) and 2003 (0.74 kg P ha<sup>-1</sup>) winter drainage. However, these increases in P losses represents less than 4 % and 2.5 % of the P applied in the effluent during the 2001/2002 and 2002/2003 lactation seasons, respectively.

At the whole-farm scale, it is standard dairy farming practise (mostly the intensive grazing of cattle) rather than the land application of FDE that results in the greatest nutrient losses (Section 10.3). N loss from a sustainable FDE land treatment operation could be expected to contribute as little as 0.1% of the total N loss from a dairy farm. However, if a two pond system was used to treat FDE, then the N loss associated with FDE treatment would amount to 17% of whole-farm N loss. This serves to illustrate the advantages of improved land treatment of FDE to dairy production systems. The management of FDE plays a greater role in the likely whole-farm loss of P with a 5% contribution from an efficient FDE land treatment system and nearly 50% contribution to whole-farm losses of P from a two-pond system.

To decrease nutrient loss in mole and pipe drainage and surface runoff water on winter wet Pallic soils such as the Tokomaru silt loam, best farm management practise would avoid winter cattle grazing or the application of urea when soil temperature, pasture growth and hence plant uptake are all low. This is particularly applicable in the months of June and July when dairy cows are dry and many are wintered off farm.

In comparison to previous research on the same soil type (and on the same farm), the concentrations of both NO<sub>3</sub><sup>-</sup>-N and DIP and the seasonal patterns of these concentrations were very similar to those reported 25 years earlier by Sharpley and Syers (1979). However, they reported a five fold increase in NO<sub>3</sub><sup>-</sup>-N concentration in mole and pipe drainage events following cattle grazing (12 mg litre<sup>-1</sup> increase in NO<sub>3</sub><sup>-</sup>-N concentration in drainage vs. 5 mg litre<sup>-1</sup> increase reported above) following cattle grazing events. This is likely explained by the difference in stocking rate between the two studies: Sharpley and Syers used a stocking density of 300 cows ha<sup>-1</sup> for 12 hours compared to the average stocking density of 100 cows ha<sup>-1</sup> 12 hours<sup>-1</sup> in this research. Furthermore, Sharpley and Syers (1979) recorded an approximate 7 mg N litre<sup>-1</sup> increase in NO<sub>3</sub><sup>-</sup>-N concentration in the drainage events (for a two month period) following urea application in late June. A slightly smaller increase of approximately 5 mg N litre<sup>-1</sup> in NO<sub>3</sub><sup>-</sup>-N concentration was recorded in the current research, however, Sharpley and Syers (1979) applied 62% more urea (60 kg N ha<sup>-1</sup>) in their June application. Unfortunately, as they only present data for two intervals of four weeks

each, no comparison can be made of total nutrient losses. The similar nutrient concentrations reported 25 years earlier by Sharpley and Syers (1979), despite the then low fertility status and short-term history of dairy cattle intensification suggest that nutrient losses are driven on an event basis, in their case that of cattle grazing and urea application in mid winter.

## 11.5 FURTHER WORK

The research described above suggests a number of areas that warrant further investigation. In particular:

- Further investigation of the relationship(s) between the instantaneous application rate of travelling irrigators, the soil moisture deficit and the incidence of runoff and preferential flow of partially treated FDE.
- Investigation of surface runoff of FDE from rolling land where the use of travelling irrigators in the early and late stages of the lactation season when soils are wet and increase the risk of nutrient losses via surface runoff.
- An economic analysis of the costs and benefits of operating an optimal deferred irrigation system including; construction of adequate pond storage, irrigator hardware, soil moisture monitoring hardware and/or computer simulation capability, labour requirements, pasture production and FDE nutrient value.
- Improved irrigation technology for applying FDE to land needs to be developed: this would make the implementation of deferred irrigation easier for farmers. It would require irrigator technology capable of irrigating large areas at a slow rate with small depths of FDE within small windows of time i.e. whilst suitable soil water deficits exist. The potential to use K line or long lateral irrigators with appropriate solid separation or filtering deserves further evaluation. These types of irrigators may have a role in combating the problem of increased losses of P in winter drainage water following deferred irrigation of FDE. The ability to apply small depths of FDE may help confine the movement of FDE in the soil to the surface layer(s) (150 mm or so) where plant roots are more active and can take-up the P from FDE before it is leached.

- Modelling the use and benefits of deferred irrigation for other climates and dairy production systems is likely to enhance the uptake of this improved practise for FDE treatment.
- The simulation model APSIM could be used to further investigate a number of questions that arise from this study. It could be used to predict the drainage and runoff of partially treated FDE for a range of climate and irrigation scheduling scenarios (as opposed to simulating pond overflow to evaluate storage requirements whilst implement deferred irrigation). It could also be adapted to study the effect that a constraining drainage coefficient (e.g. 10 mm d<sup>-1</sup>) would have on drainage and runoff rates and the proportioning of excess water between these two pathways.
- A better understanding is required of the seasonal pattern of P loss, in particular the trend of increasing concentrations of DIP in winter/spring drainage from areas receiving summer applications of FDE.
- Research into effective solutions for further improving drainage water quality by removing N and in particular P either within or at the end of mole and pipe drainage systems.
- Research into the potential environmental benefits of using standoff pads, covered feed pads and wintering homes to protect soil health and reduce nutrient losses attributable to winter grazing.

## 11.6 REFERENCES

Sharpley AN, Syers JK (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.

# APPENDIX 1: PLOT VARIABILITY

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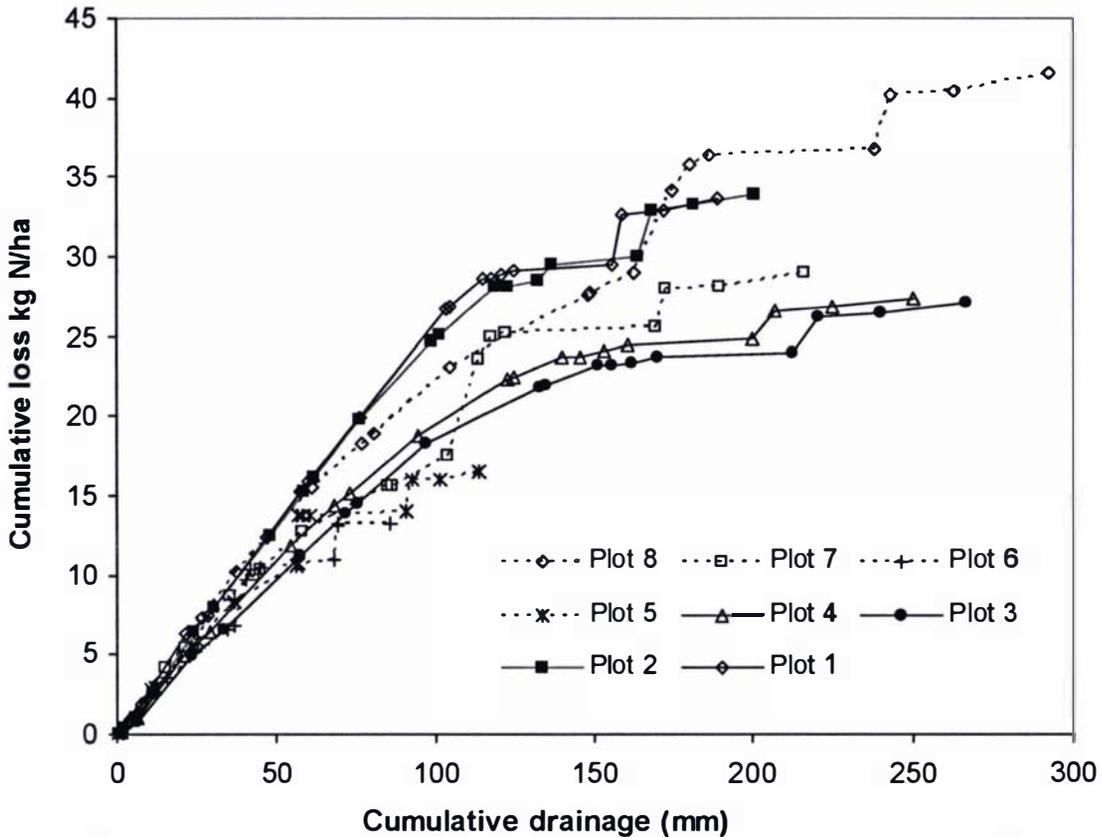
This appendix sets out to further clarify some issues surrounding the variability in drainage rates measured for the plots and the quantification of this variability (chapters 6 and 7). Figures 6.3 and 7.4 illustrate the large variability in annual drainage volumes between plots. As stated in these chapters, the moles installed on plots 5 and 6 were not conveying as much water to the collecting pipe at the plot boundary as would be expected. It is possible that an old tile line may have been intercepting the water in these moles. In addition, a detailed topographic survey showed that there were areas on these plots that sloped away from the collecting pipe at the plot edge. This would also have encouraged water movement away from the collecting pipe. A further decline in the performance of the mole and pipe drainage system being studied on plots 5 and 6 was observed in the very wet year that followed the present study. In this year (2004) of record rainfall, there was an average of 388 mm of drainage from plots 1 to 4 and 7 and 8 but only minimal drainage was measured for plots 5 and 6. It is clear that either the moles on these two plots were not functioning correctly or that that water in the moles was not getting into the collecting pipe at the plot boundary and thence to the v-notch weirs.

## A1.1 CALCULATING ANNUAL NUTRIENT LOSSES

The quantity of a nutrient (X) lost in a drainage event is calculated by multiplying the concentration of the nutrient in drainage ( $\text{mg X litre}^{-1}$ ) by the total volume of drainage (litres). It is common to then scale this loss so that it can be presented on a  $\text{kg ha}^{-1}$  basis. To calculate the nutrient losses from each drainage event in the 2002 and 2003 winter-spring drainage season, a mean drainage volume was calculated for plots 1 to 4 and 7 and 8. This common drainage volume was then used to calculate nutrient losses from each plot. The nutrient losses from each drainage event were aggregated to give total annual nutrient losses. Use of a single, common drainage volume means that any differences in nutrient losses between treatments is due to differences in concentration of the nutrients in drainage.

Figure A1, a plot of cumulative nitrogen (TN) loss for each plot verse cumulative drainage for each plot, demonstrates both plot drainage variability and its potential impact on calculated annual loss

Given that the plots used in the present study were large, a high degree of variability in drainage volumes between the plots would not have been anticipated. Both Sharpley and Syers (1979) and Horne (1985) observed relatively uniform drainage from plots that were in the same soil type, artificially drained in the same manner and similar in size to those used here.



**Figure A.1.** Cumulative drainage from the plots in the winter/spring period of 2003 plotted against calculated cumulative loss.

### A1.2 ANALYSIS OF VARIANCE

Figures A2, A3, A4 and A5 demonstrate that the standard error of means (SEM) was a suitable method for demonstrating data variance from the plots. Concentrations of DIP and  $\text{NO}_3^-$ -N verse cumulative drainage have been plotted from all 6 correctly functioning plots for the 2002 and 2003 drainage seasons. Figures A2 and A4 bear out the argument made in chapters 6 and 7; there was no real difference between the concentrations of  $\text{NO}_3^-$ -N in drainage from the effluent and non-effluent treatment plots in either year. Conversely figures A3 and A5 show that in both 2002 and 2003 a treatment difference is apparent: DIP concentrations in winter-spring mole and pipe

drainage are greater in the effluent treatment than non-effluent plots as reported in chapters 6 and 7.

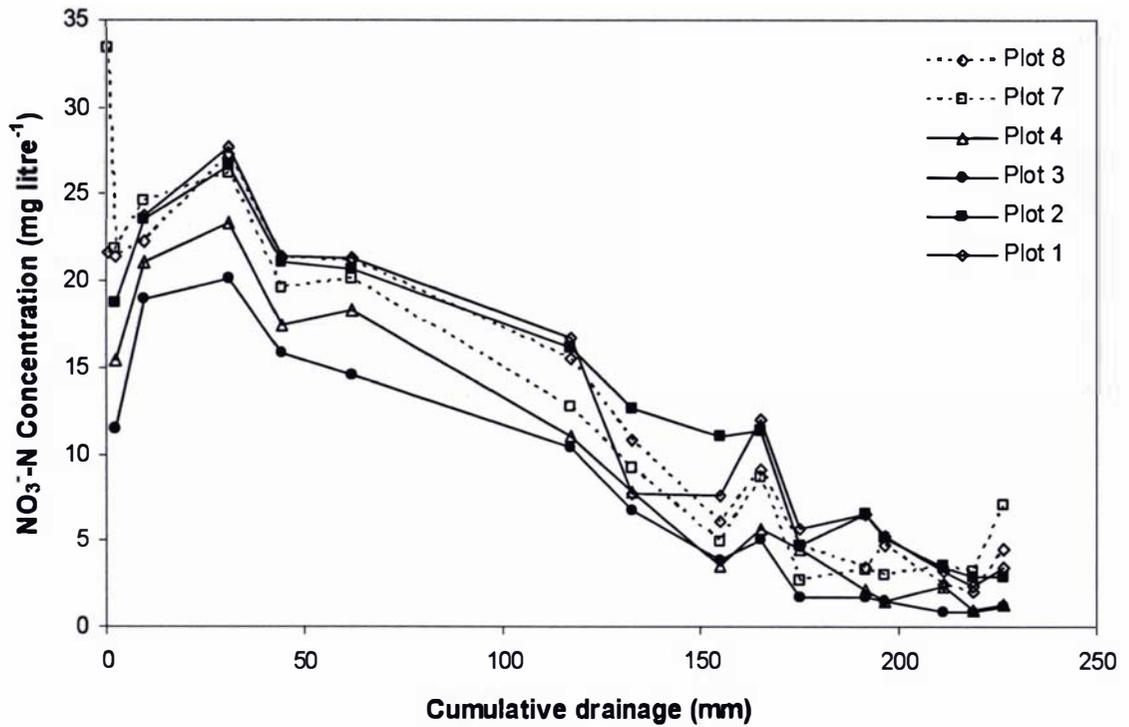


Figure A.2.  $\text{NO}_3^-$ -N concentration from individual plots vs. cumulative drainage for the winter/spring period of 2002.

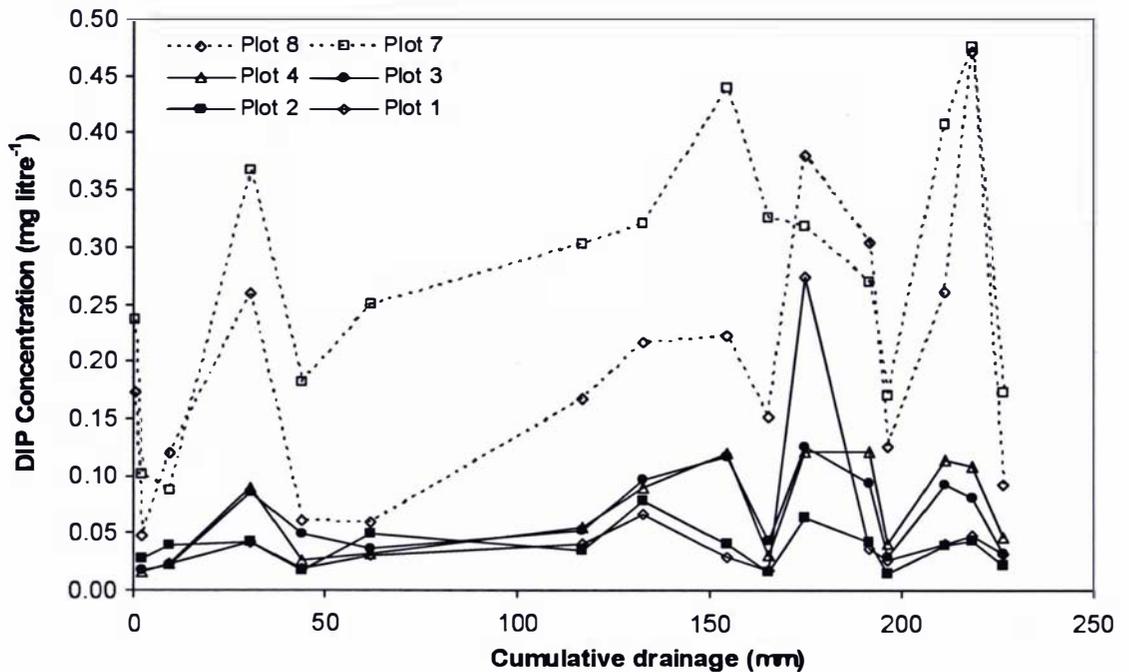


Figure A.3. DIP concentration from individual plots vs. cumulative drainage for the winter/spring period of 2002.

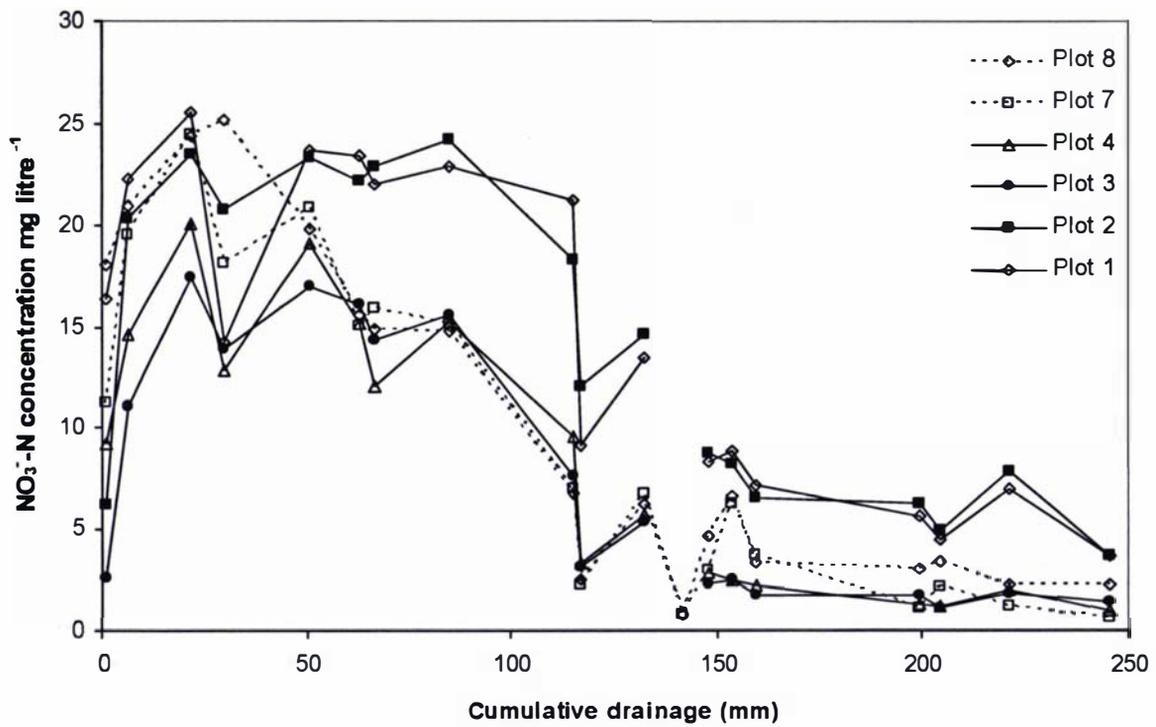


Figure A.4.  $\text{NO}_3^-$ -N concentration from individual plots vs. cumulative drainage for the winter/spring period of 2003.

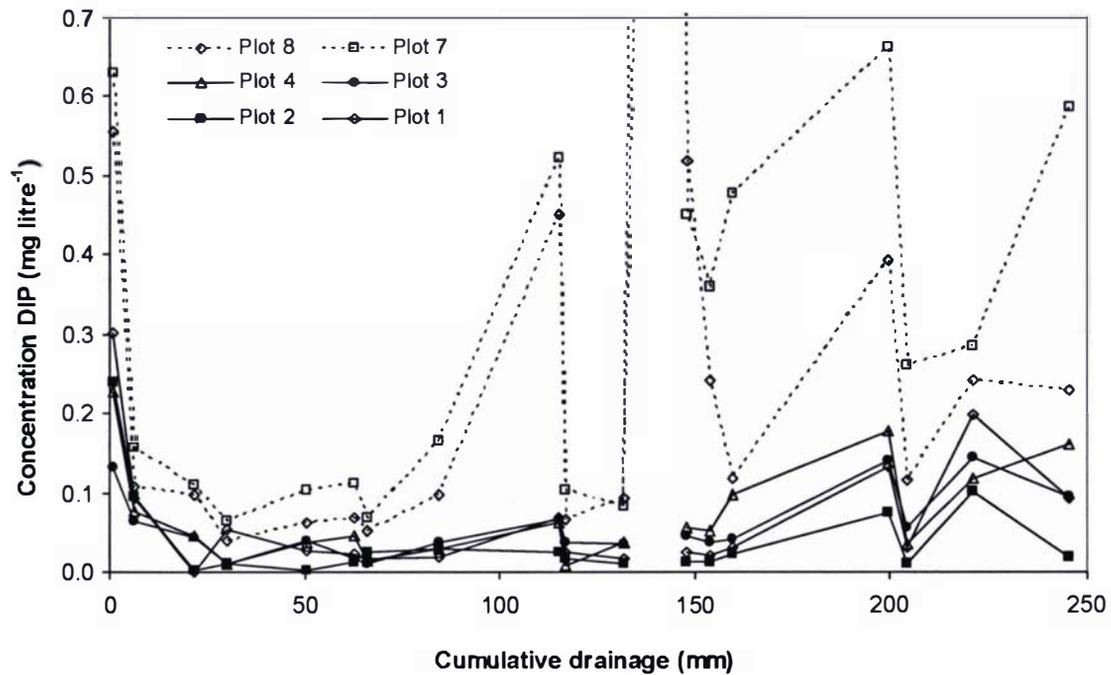


Figure A.5. DIP concentration from individual plots vs. cumulative drainage for the winter/spring period of 2003.

# APPENDIX 2: APSIM DESCRIPTION AND TESTING

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V. O. SNOW AND D. J. HOULBROOKE

## A2.1 INTRODUCTION

Experimentation is a vital part of understanding natural and managed systems and developing new management systems, such as deferred irrigation, with the intention of creating improved environmental or productive outcomes. It is however limited in the range of weather, soil, and other variables that can feasibly be examined and in the detail at which some processes can be measured. It is here that simulation modelling can enhance understanding of the responses and interactions observed during the experimental period and also extend the experimental conditions to a range of differing soil, climate, and management conditions. This is particularly important in developing and evaluating new management techniques. Here the simulation model APSIM was used to predict the soil water balance of the effluent-irrigated system in order to evaluate the likely long-term implications of deferred irrigation.

This appendix describes the APSIM model, how it was set up and parameterised for use in this study, and finally a combination of data from previously published studies and this work were used to test the model predictions. How the model was used for scenario analysis is described in Chapter 9.

## A2.2 APSIM DESCRIPTION AND SETUP

### A2.2.1 *APSIM description*

APSIM, Agricultural Production Systems Simulator, (Keating *et al.* 2003), is a modular simulation framework developed by the Agricultural Production Systems Research Unit (APSRU, [www.apsru.gov.au](http://www.apsru.gov.au)). APSIM has been used extensively to simulate productivity (see Keating *et al.* 2003 and [www.apsru.gov.au/apsru/Publications/publicat1.htm](http://www.apsru.gov.au/apsru/Publications/publicat1.htm) for many examples), as well as water (e.g. Snow *et al.* 1999a; Keating *et al.* 2002) and nitrogen balances (e.g. Jones

*et al.* 1996; Snow *et al.* 1999b). This work has primarily involved cropping systems but has been extended to agroforestry systems (Huth *et al.* 2001).

For the simulations presented here the ability of APSIM to simulate plant growth was not used as the aim was to thoroughly examine the soil water balance and no tested growth module currently exists for New Zealand dairy pastures. Apart from the basic APSIM modules controlling input, output, and simulation management the modules APSWIM, MICROMET, and SLURP were used. APSWIM is described below. MICROMET (Snow and Huth, 2004) calculates the Penman-Monteith estimate of potential soil water use from inputs of solar radiation, temperature, humidity, and wind speed from a weather file in combination with leaf area index from a crop module. SLURP ([www.apsim.info/apsim/Apsim/apsim/slurp/docs/slurp\\_science.htm](http://www.apsim.info/apsim/Apsim/apsim/slurp/docs/slurp_science.htm)) is a very simple crop module that does not simulate crop growth but provides user-set values of leaf area index and root distribution to other modules in the simulation. This option was used rather than a dynamic crop module because productivity issues are not important here and this allows focus on the soil water balance parts of the model.

### **A2.2.2 APSWIM and simulation of mole-tile drainage**

APSWIM (Huth *et al.* 1996) is an APSIM version of the SWIM (Ross *et al.* 1992; Verburg *et al.* 1996) model of water and solute movement in soil. SWIM uses Richards' equation to simulate water infiltration, movement, redistribution, evaporation from the soil surface and uptake by plants. Richards' equation combines Darcy's law for movement of water under saturated and unsaturated conditions (which states that the flux of water is proportional to the hydraulic gradient multiplied by the hydraulic conductivity) with the continuity equation (describing conservation of mass of water) to describe transient (saturated-unsaturated) flow of water. Consistent with the assumptions implicit in Richards' equation, the soil is assumed to be horizontally uniform. The flow equations are solved using the soil hydraulic conductivity and water retention function in combination with calculated matric and gravitational potential gradients.

More recently the ability to simulate the effect of artificial drainage has been added (N. I. Huth, CSIRO Sustainable Ecosystems, *pers com.*) by including a drainage term as a sink in Richards' equation. The equation describing the drainage rate (Bouwer and van Schilfgaarde, 1963; Skaggs 1991) is:

$$q = \frac{4K_e m(2d_e + m)}{CL^2},$$

where  $q$  is the soil water flux from the drains (cm /hr),  $K_e$  is the effective lateral hydraulic conductivity of the soil (cm /hr),  $m$  is the height of the water table above the drain depth (cm),  $d_e$  is the equivalent depth from the drains to the impermeable layer (cm),  $C$  is the ratio of average flux to the flux at a point midway between the drains and following Skaggs (1991) is assumed equal to 1.0, and  $L$  is the distance between the drains (cm). The equivalent depth to the drains,  $d_e$ , is calculated according to Skaggs (1991) as:

$$d_e = \begin{cases} \frac{d}{1 + \frac{d}{L} \left[ \frac{8}{\pi} \ln\left(\frac{d}{r}\right) - \alpha \right]}, & \alpha = 3.55 - 1.6 \frac{d}{L} + 2 \frac{d^2}{L} \quad 0 < \frac{d}{L} \leq 0.3 \\ \frac{L\pi}{8 \left[ \ln\left(\frac{L}{r}\right) - 1.15 \right]} & \frac{d}{L} > 0.3 \end{cases},$$

where  $d$  is the actual depth to the drains (cm) and  $r$  is the drain radius (cm). In calculating the simulated drainage the user sets the descriptive parameters  $d_e$  and  $L$  while  $K_e$  and  $m$  are calculated dynamically within the simulation based on soil hydraulic properties and simulated height of the water table.

### A2.2.3 Soil hydraulic parameters and APSWIM settings

SWIM uses an innovative sums-of-functions (Ross and Smettem, 1993; Verburg *et al.* 1996) approach to describe soil hydraulic properties in a way that both represents the effects of particularly high hydraulic conductivity due to macropores near saturation (Smettem and Ross, 1992) while retaining the characteristics needed for the function to be differentiable as required in the solution to Richards' equation. In application to the Tokomaru silt loam, two smoothed Brooks-Corey functions (Brooks and Corey, 1964; Campbell, 1974) were used to describe the soil water retention and hydraulic conductivity properties. One function with relatively high conductivity near saturation and very low conductivity below saturation (pore interaction index = 100), and containing no effective soil water retention, was used to represent the macropores. A second function was used to represent the soil matrix and all the soil water retention.

Soil physical properties for the Tokomaru silt loam were taken from Scotter *et al.* (1979b) and the National Soils Database (Wilde, 2003) and are given in Table A1 presented as inputs to the programme 'HYPROPS' (Verburg *et al.* 1996). HYPROPS

generates a hydraulic property table based on a piecewise spline approximation of the sum of the two Brooks-Corey functions.

In describing the drainage system to APSWIM, it was assumed that the mole drains were responsible for the drainage and that the tile system did not limit drainage. Given these assumptions the parameters for the Hooghoudt equation were a drain spacing of 2 m at 0.45 m deep with a diameter of 35 mm. The fragipan, at 0.8 m deep, was assumed to form the impermeable layer. The lateral hydraulic conductivity was, somewhat arbitrarily, assumed to be 1 m/day. There was no data to base this parameter on, but with such an intensive mole system it was thought that the network of cracks formed when the moles were pulled should provide a relatively high conductivity.

Other settings for APSWIM included that:

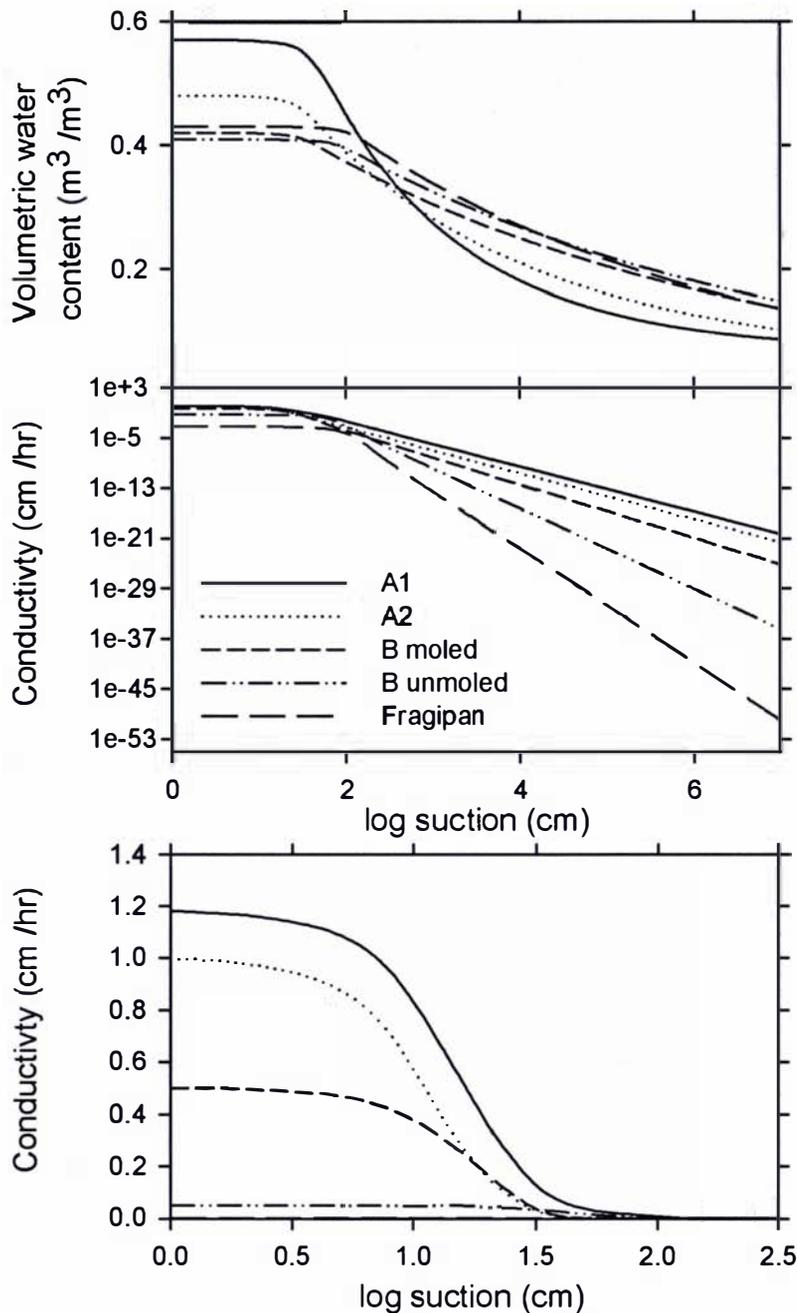
- any rainfall or irrigation unable to infiltrate because of a saturated soil profile or because the rainfall intensity was greater than the soil could accept was assumed to become runoff with no surface detention; and
- the bottom boundary condition was set to drainage under unit gradient.

**Table A1.** Soil hydraulic properties for the Tokomaru Silt Loam.

Description	Depth range (cm)	Bulk density (Mg /m <sup>3</sup> )	$\Theta_r$	$\Theta_s$	$\psi_e$ (cm)
Surface	0 - 2	1.20	0.069	0.57	-35.5
A1	2 - 100	1.20	0.069	0.57	-35.5
A2	100 - 250	1.35	0.053	0.48	-23.9
B - above moles	250 - 450	1.53	0	0.42	-25.9
B - below moles	450 - 800	1.56	0	0.41	-68.6
Fragipan	800 - 1500	1.65	0	0.43	-92.3

Description	Matrix			Macropores
	b	$K_s$ (cm /hr)	P	$K_s$ (cm /hr)
Surface	3.8	0.182	4	0.5
A1	3.8	0.182	4	1.0
A2	6.0	0.091	8	0.909
B - above moles	11.5	0.045	24	0.455
B - below moles	11.7	0.005	50	0.046
Fragipan	10.1	0.00011	70	0.0005

$\Theta_r$  – residual water content;  $\Theta_s$  – saturated water content;  $\psi_e$  air entry potential; F – fractional contribution of component to soil water retention, b – slope parameter;  $K_s$  – saturated hydraulic conductivity; P – pore interaction index. See Verburg *et al.* 1996 for further detail.



**Figure A6.** Soil water retention and hydraulic conductivity and wet-end hydraulic conductivity as calculated by Hyprops from the data in Table A1.

#### **A2.2.4 Weather data**

The weather data needed to drive the simulation were obtained from the Grasslands Palmerston North meteorological station situated at 34 m elevation above sea level with latitude 40 23S and longitude 175 37E and less than 1 km from the experimental site. Thirty-two years of data, from January 1972 to December 2004, were used in the

simulations. The data used in the simulations were the daily temperature extremes, average humidity for the day, solar radiation calculated from sunshine hours, average wind speed, and rainfall. Rainfall was measured at the experimental site during 2002 and 2003 and these data were substituted for the data from the Grasslands meteorological station for that period. The rainfall intensity was assumed to be 5 mm/hr based on simulation analysis of 10-minute rainfall intensity data for the period 1-July-2001 to 30-June-2004. APSIM uses internal calculations by MICROMET to calculate potential evaporation while actual evaporation is based on that potential combined with the ability of the root system to extract water from the soil.

### **A2.2.5 Other settings**

Other settings used in the simulations included:

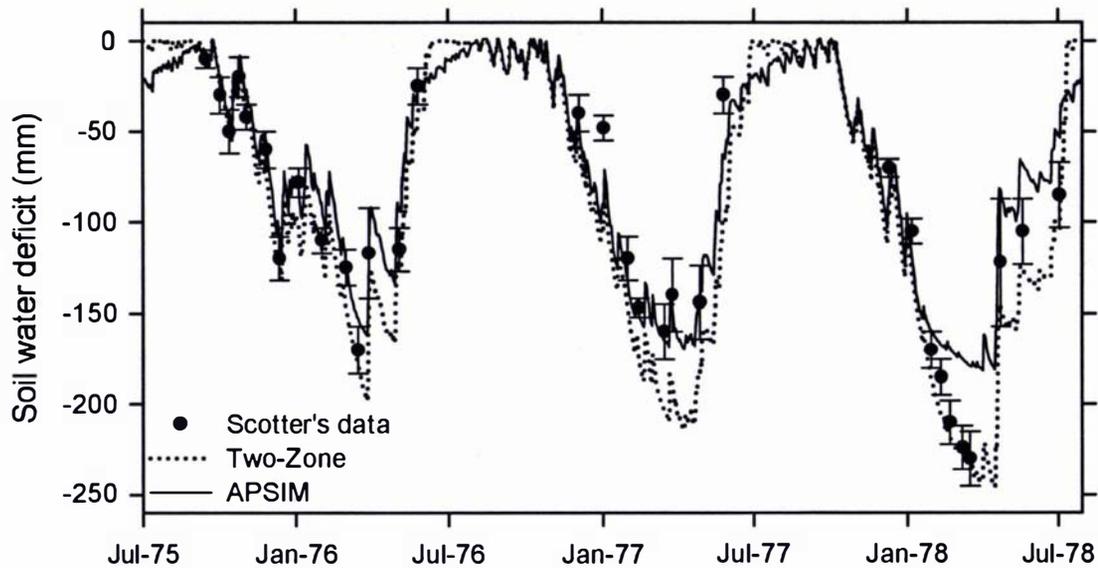
- the pasture was assumed to be able to lower its leaf water potential to -1.5 MPa to extract water from the soil;
- the leaf area index and light extinction coefficient were set so that 92% of the incident solar radiation was intercepted;
- the albedo of the system was set to 0.23;
- most of the pasture roots were constrained to the top 0.3 m with a total rooting depth set to 1.5 m consistent with observations by Scotter *et al.* (1979b).

## **A2.3 MODEL COMPARISON AGAINST MEASUREMENT**

To test the predicative capability of APSIM three data sets were used. The following sections describe the data and compare the model predictions against measurements.

### **A2.3.1 Scotter's 1975-78 Data**

Scotter *et al.* (1979a; 1979b) measured soil water deficit using a neutron moisture meter during the summers between 1975 and 1978. The experimental site was on the Tokomaru silt loam soil and was near Houlbrooke's site. Drainage and runoff were not measured.



**Figure A7.** Soil water deficit to 1.7 m deep as measured by Scotter *et al.* (1979b) data (points and 1 standard deviation) and predicted by the Two-Zone model (dotted line) and APSIM (solid line).

Figure A7 shows APSIM's predictions against Scotter *et al.*'s (1979b) data and empirical soil water balance model that has been calibrated to the data, later called "Two-Zone" by Woodward *et al.* (2001). In comparison to the data, APSIM and the Two-Zone seem to predict the data equally well in 1975/76, APSIM provides the better simulation in 1976/77, and the Two-Zone better mimics the data in 1977/78.

One noticeable feature is that the Two-Zone routinely allows greater soil water deficits to develop than does APSIM. Part of this is attributable to the description of the soil properties in APSIM. The Tokomaru silt loam on which Scotter *et al.* (1979b) did their experimental work contained an approximately 100 mm thick ash layer within the fragipan at approximately 1.5 m depth. Under normal conditions this ash layer does not supply water to the pasture and, because it is enclosed by the very poorly conductive fragipan, does not influence drainage. For these reasons, in combination with the lack of information about the hydraulic properties of the ash and the likelihood that this thin permeable layer within the fragipan would cause numerical instabilities in Richards' equation simulation, the ash layer was not included in the APSIM representation of the soil profile. In the first two years of Scotter *et al.*'s (1979b) experiment, the assumption that the ash layer did not provide water to the pasture was reasonable but in the dry year Scotter *et al.*'s (1979a) data clearly shows a decrease in water content in the ash layer. This contributed approximately 10 mm to the measured

soil water deficit, approximately 20% of the total difference between measured and simulated maximum soil water deficit. Much of the remaining difference is attributable to the development of water contents well below the plant extractable limit in Scotter *et al.*'s (1979a) data and this will be discussed further below.

Another feature evident in A7 is that the soil water deficit takes longer to climb to zero in APSIM than in Two-Zone. This happens because APSIM predicts some tile-drain flow as the soil approaches field capacity when the soil wets up and the perched water table rises, perhaps only transiently during the day, above the mole-drain level. In Two-Zone, excess water (the model cannot differentiate between slow drainage through the fragipan, tile drainage, or runoff) is not generated until the model predicts the soil is at field capacity.

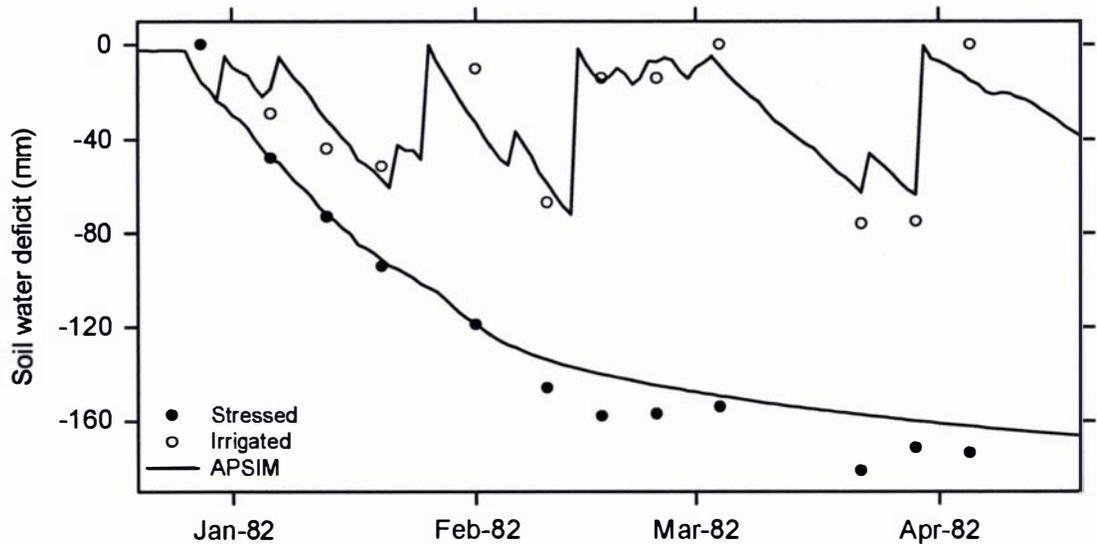
### **A2.3.2 Barker's 1982 Data**

In the summer of 1982, Barker *et al.* (1985) conducted an experiment near Houlbrooke's site on the Tokomaru silt loam. There were two treatments; one treatment received rainfall and was irrigated to avoid water stress whenever the estimated soil water deficit fell below -60 mm, and the second was conducted under a rain shelter so no water was added to the soil after the experiment began. Both treatments were irrigated to excess before the treatments were imposed. Further detail can be found in Barker *et al.* (1985). Figure A8 compares Barker *et al.*'s soil water deficit data against the APSIM predictions and Figure A9 shows data and predictions for selected depth profiles. Note that the soil water deficits associated with Barker's data are to 1.0 m deep in contrast to those associated with Scotter's data which are to 1.7 m deep.

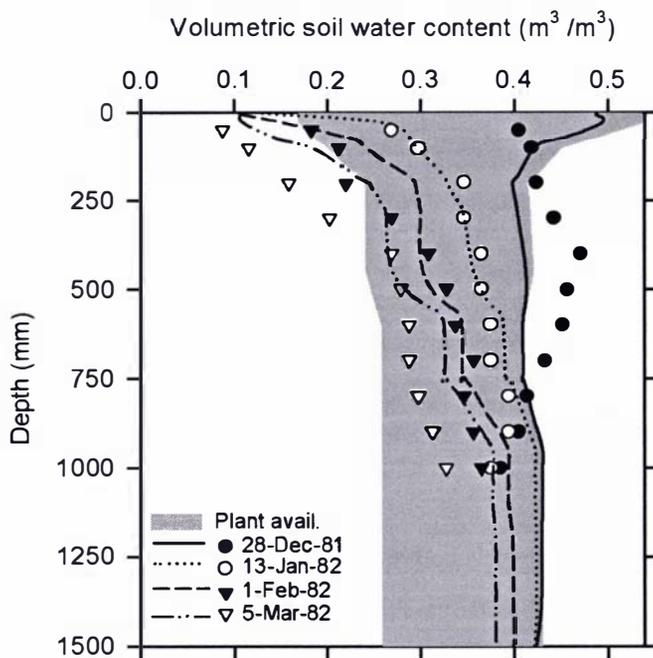
APSIM predicts the irrigated treatment well and the early part of the stressed treatment when soil water deficit was greater than -140 mm but would not generate the low soil water deficits measured by Barker *et al.* (1985) in the stressed treatment. Examination of the depth profiles of water content (Figure A9) reveals that although the data and simulation had similar total soil water storage at the start of the experiment the pattern of water content was rather different. The data suggest that at the surface and at depth the soil was not wet to field capacity at the start of the experiment, perhaps a result of difficulty properly wetting drained soil when evaporation rates are high. The bulge in water content data at between 0.25 and 0.75 m is above laboratory measured saturation and may indicate that there was a gap between the access tube and soil at

those depths. Any such gap would have little effect on water content once the excess water drained away.

Barker *et al.*'s (1985) data also show water contents well below the plant-extractable water at the end of the stressed experiment. This will be further discussed below.



**Figure A8.** Soil water deficit measured to 1.0 m deep as measured by Barker *et al.*'s (1985) irrigated (open points) and stressed data (solid points) with the APSIM predictions (lines).



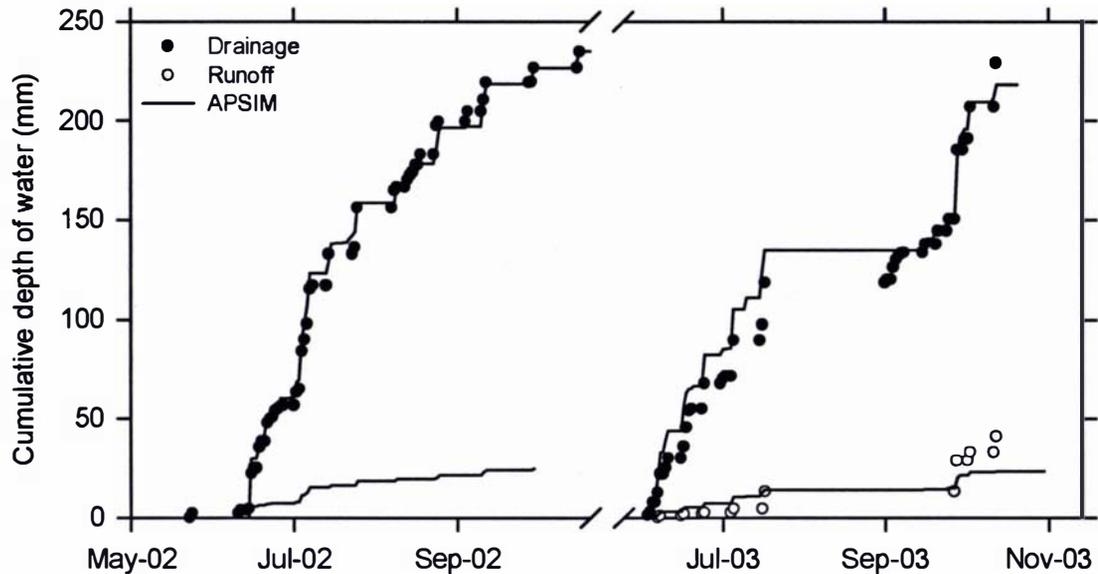
**Figure A9.** Selected depth profiles from Barker *et al.*'s (1985) stressed data (points) with the APSIM predictions (lines) and the limits of plant-available soil water (grey shaded area).

### **A2.3.3 Houlbrooke's 2002-2003 Data**

Houlbrooke's drainage (2002 and 2003) and runoff (only measured in 2003) data which have also been presented in Chapters 6 and 7 are shown in Figure A10 with the APSIM predictions. APSIM's prediction of total drainage was excellent for both seasons although the pattern in time was distorted in the middle of the 2003 season. The late onset of drainage after the dry period in August-September 2003 would be consistent with excessive estimation of plant water use and/or soil water evaporation during this period, however there are no soil water deficit data to confirm or refute this hypothesis.

Surface runoff was somewhat under-predicted, with APSIM estimating 24 mm whereas 40 mm was measured. Although the difference is large in comparison to the measured runoff, it is small, 6%, compared to the total excess water for the season. Table A2 gives measured and predicted runoff for individual runoff events. Several small, less than 0.2 mm, events predicted by APSIM have been excluded as these are unlikely to be measured with the equipment used. For every measured runoff event there is corresponding predicted surface runoff and most of the events agree within a few mm of water. The exceptions are the large event, 15 mm, measured on 28/9/2003 when only 7 mm was predicted and the final event on 13/10/2003 with 8 mm measured and 0.4 mm estimated.

Despite these deficiencies it is clear that given the limited specific information supplied to APSIM, good predictions of drainage and surface runoff are possible.



**Figure A10.** Cumulative drainage (solid points) and runoff (open points) from Houlbrooke's plots and APSIM's predictions (lines).

**Table A2.** Measured and predicted runoff events.

Date	Measured	Predicted
7/06/2003	0.8	2.9
16/06/2003	1.0	1.8
24/06/2003	0.7	2.0
5/07/2003	2.1	3.4
17/07/2003	8.7	3.3
28/09/2003	15.4	6.7
3/10/2003	4.2	1.5
13/10/2003	7.8	0.4

#### A2.3.4 Summary

Before using the model for predictive purposes it was necessary to examine the ability of APSIM, and the input variables used, to predict the water balance of mole-drained pasture growing on the Tokomaru silt loam. This was done by comparison against two existing data sets where soil water deficit was measured. Unfortunately no existing data with measured drainage and runoff were found so it was necessary to use the data measured here for that purpose. A further comparison to test the model robustness was made by comparing APSIM's prediction of drainage against the drainage measured from the experimental site presented in chapters 6 and 7. During

this testing, all the parameters here held constant with the exception of those needed to impose the irrigation and rain shelter treatments in Barker *et al.*'s (1985) data.

The APSIM predictions closely matched Scotter *et al.*'s (1979a) measured neutron probe data at small and intermediate deficits and at large deficits on two of the three summers measured (Figure A7). The notable exception to this was for the summer of 1977/78 which was particularly dry with only 32 mm of rainfall for the months of January, February and March compared to an expected average of 214 mm. The neutron probes installed down to 1.7 m measured soil water deficits in excess of 200 mm. A similar pattern was noted in testing against Barker *et al.*'s (1985) data where the data showed about 15 mm more extraction than APSIM's predictions. It is possible that the neutron probe data had over-estimated the soil water deficit under the extremely dry conditions, perhaps the measurements were influenced by shrinkage of the dry soil away from the access tubes. However it is more likely that the model failed under these conditions in two specific aspects: inadequate specification of the soil hydraulic properties in the fragipan, and the potential for the soil to dry out from the exposed surfaces of the cracks that form in dry conditions.

The effect of the ash layer is only relevant in comparison against Scotter *et al.*'s (1979a) data, because the deficits were calculated to below the ash layer, and have been discussed above.

It is usually assumed that pasture can only extract water from soil to a soil water potential of about -1.5 MPa (Reeve and Carter 1991) . Soil water contents below this potential are likely to be caused by direct evaporation and are normally observed only near the soil surface, generally in the top 100-200 mm, because the drying is limited by the ability of the soil to transport water to the soil surface. Examination of figure 9 in Scotter *et al.* (1979a) clearly shows that the soil dried to water contents well below the -1.5 MPa water content to a depth of 700 mm. In figure A9, Barker *et al.*'s (1985) data shows soil drier than -1.5 MPa to a depth of 400 mm. In contrast, APSIM predicts that only the soil in the top 200 mm will dry below -1.5 MPa.

Such low water contents are usually attributable to direct evaporative drying of the near-surface soil, usually only down to a depth 100-200 mm as predicted by APSIM, but here are evident in the data to 400 and 700 mm deep. One possible explanation for this is that the cracking that occurs in this soil as it dries (Scotter *et al.* 1979a) effectively increases the exposed surface area of the soil. In turn this decreases the

distance water has to be transported before it evaporates and speeds the rate and depth of drying. This is a process that APSIM does not represent. The enhanced drying of soil below -1.5 MPa accounts for all of the difference between model and prediction in Barker *et al.*'s (1985) data and, together with the effect of the ash layer, 85% of the difference in the Scotter *et al.* (1979a) data.

It is interesting to compare these APSIM simulations against the soil water deficit models of Scotter *et al.* (1979b) and Woodward *et al.* (2001) which were calibrated against the data rather than attempting to derive the parameters independently of the data. In both of these studies the very high soil water deficit was predicted better in the dry year than with APSIM their simulations. The converse was the case for the only moderately dry year (1976/77) when APSIM predicted the maximum soil water deficit better than the calibrated models.

While the performance of APSIM in the very dry year is interesting, these high soil water deficits are of little consequence to the current modelling aims which seek to simulate irrigation scheduling of farm dairy effluent, particularly in the spring time when soil water deficits are typically small or non-existent. For this purpose APSIM was accurate at predicting soil water storage at times of low and middle range soil water storage.

## A2.4 CONCLUSIONS

The simulation model APSIM was set up to estimate soil water extraction by pasture, drainage through the fragipan, mole-tile drainage, and evaporation from the soil surface based on soil hydraulic properties derived from the literature and inputs of plant characteristics and weather. Comparison was made against the soil water deficit data of Scotter *et al.* (1979a; 1979b) and Barker *et al.* (1985) and the drainage and runoff data presented in Chapters 6 and 7. The main deficiency of the APSIM was its inability to predict the very dry, less than that extractable by pasture, water contents that developed at depth under dry conditions. This is likely to result from enhanced soil water evaporation caused by cracking of the soil, a process that is not represented in APSIM. Despite this, it is evident that the model does perform well in winter and spring and the time of year it is required for irrigation scheduling of farm dairy effluent and therefore can be considered to be suitably parameterised for this work.

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## APPENDIX 3: APSIM POND MANAGER

---

Copied below is a simulation description of the Apsim pond manager used in Chapter 9 to provide rules for FDE volume, pond storage and irrigation. Below each section a small description is in italicised writing explaining what the pond manager is describing or attempting to simulate.

```
[Pond_15mm.manager.init]
irrig_dep_def = 15 ! mm
irrig_trigger = -20 ! mm deficit to trigger irrigation
```

```
[Pond_30mm.manager.init]
irrig_dep_def = 30 ! mm
irrig_trigger = -35 ! mm deficit to trigger irrigation
```

```
[Pond_45mm.manager.init]
irrig_dep_def = 45 ! mm
irrig_trigger = -50 ! mm deficit to trigger irrigation
```

*<DEFINITION OF 3 OF THE 4 IRRIGATION SIMULATION DEPTHS AND THEIR TRIGGER DEFICIT TO APPLY FDE IRRIGATION>.*

```
[Pond.manager.init]
Ana_Pond_volmax = 4000 !m3
Ana_Pond_vol = 4000 !m3
Ana_Pond_evap_area = 800 !m2
Ana_Pond_rain_area = 900 !m2
Aer_Pond_volmax = 2000 !m3
Aer_Pond_vol = 100 !m3
Aer_Pond_evap_area = 1800 !m2
Aer_Pond_rain_area = 5400 !m2
irrig_area1day = 40000 ! m2
days2irrig = 4 !number of days to irrigate all the area once - always make a whole number
irrig_dep = 0.0 ! mm
irrig_vol = 0.0 ! mm
irrig_vol_def = irrig_area1day * irrig_dep_def / 1000 !the minimum irrigation needed to irrigate for 1 day
irrig_day = 0
overflow = 0
irrig_season = 0
```

*<LIST OF POND PARAMETERS INCL. POND SIZE, RAINFALL AND EVAPORATION AREAS AND THE CALACULATION OF IRRIGATION VOLUME>.*

```
[Pond.manager.start_of_day]
if (irrig_day = days2irrig) then
  irrig_day = 0 !finished irrigating the rest of the area
endif
if (day < 136) then !15 May
  Ana_incoming = 475 * 50 / 1000 !m3 incoming effluent
  irrig_season = 1 !OK to irrigate
```

```

elseif (day < 214) then !1 Aug
  Ana_incoming = 0.0 !m3 incoming effluent
  irrig_season = 0 !no irrigation even if enough deficit
elseif (day < 245) then !1 Sep
  Ana_incoming = (1 - (245 - day) / (245 - 214)) * 475 * 50 / 1000 !m3 incoming effluent
  irrig_season = 1 !OK to irrigate
else !after 1 Sep
  Ana_incoming = 475 * 50 / 1000 !m3 incoming effluent
  irrig_season = 1 !OK to irrigate
endif

```

*<EXPLANATION OF THE DAY IRRIGATION IS TO TAKE PLACE. FDE CAN ONLY BE APPLIED BETWEEN AUGUST 1 AND MAY 15. FDE VOLUME IS CALCULATED BASED ON 50 L/COW/DAY DURING THE PERIOD SEPTEMBER 1 AND MAY 15>.*

```

!first calculate the movement of effluent from the anaerobic pond to the aerobic pond -
based on overflow
Ana_Pond_vol = Ana_Pond_vol - (Ana_Pond_evap_area * apswim_eo_yesterday * 0.9
/ 1000) + (Ana_Pond_rain_area * rain / 1000) + Ana_incoming
if (Ana_Pond_vol > Ana_Pond_volmax) then
  Aer_incoming = Ana_Pond_vol - Ana_Pond_volmax
  Ana_Pond_vol = Ana_Pond_volmax
else
  Aer_incoming = 0.0
Endif

```

*<CALCULATION OF POND VOLUME IN THE ANAEROBIC AND AEROBIC INFLOW>.*

```

!if end of season - day 136 - empty the pond otherwise figure out the irrigation from the
aerobic pond
if (day = 136 ) then !empty the aerobic pond to 100 m3
  if (Aer_Pond_vol < 100) then !Nothing to irrigate anyway
    irrig_vol = 0.0
    irrig_dep = 0.0
  else
    irrig_vol = Aer_Pond_vol - 100
    irrig_dep = irrig_vol / 160000 * 1000 !depth should be in mm
    irrigate.amount = irrig_dep
  endif
  irrig_day = 0
elseif (irrig_season = 0) then !no irrigation to happen at this time of year
  irrig_vol = 0.0
  irrig_dep = 0.0
  irrig_day = 0.0
elseif (irrig_day >= 1) then !need to irrigate the rest of the area
  if (Aer_Pond_Vol < irrig_vol_def) then
    irrig_vol = 0.0 !basically do nothing - wait until there is enough effluent
    irrig_dep = 0.0 !basically do nothing - wait until there is enough effluent
  else
    if (est_deficit < irrig_trigger) then
      irrig_vol = irrig_vol_def !take it out of the pond but do not need to actually irrigate
      this though - going to the other paddocks
      irrig_day = irrig_day + 1
      irrig_dep = 0.0 !basically do nothing - wait until there is enough effluent
    else

```

```

    irrig_vol = 0.0 !basically do nothing - wait until there is enough effluent
    irrig_dep = 0.0 !basically do nothing - wait until there is enough effluent
  endif
endif
elseif (deficit < irrig_trigger ) then !mm
  if (Aer_Pond_Vol < irrig_vol_def) then
    irrig_vol = 0.0 !basically do nothing - wait until there is enough effluent
    irrig_dep = 0.0 !basically do nothing - wait until there is enough effluent
  else
    irrig_vol = irrig_vol_def
    irrig_dep = irrig_dep_def
    irrigate.amount = irrig_dep
    irrig_day = 1
  endif
else
  irrig_vol = 0.0
  irrig_dep = 0.0
endif

```

*<RULES FOR FDE APPLICATION. IF THERE IS GREATER THEN 100 CUBIC METERS OF FDE AT THE END OF THE LACTATION SEASON (15<sup>TH</sup> MAY) THEN ALL REMAINING FDE WILL APPLIED EVENLY ACROSS APPLICATION AREA. OTHERWISE FDE WILL BE APPLIED ONCE BOTH THE DESIRED IRRIGATION TRIGGER DEFICIT HAS BEEN MET AND A SUITABLE VOLUME OF STORED FDE HAS BEEN GENERATED TO IRRIGATE THE DESIRED IRRIGATION DEPTH>.*

```

!Aerobic pond water balance
Aer_Pond_vol = Aer_Pond_vol - (Aer_Pond_evap_area * apswim_eo_yesterday * 0.9 /
1000) + (Aer_Pond_rain_area * rain / 1000) + Aer_incoming - irrig_vol
if (Aer_Pond_vol > Aer_Pond_volmax) then
  Overflow = Aer_Pond_vol - Aer_Pond_volmax
  Aer_Pond_vol = Aer_Pond_volmax
else
  overflow = 0.0
endif

```

*<CALCULATION OF POND VOLUME IN THE AEROBIC POND TAKING INTO ACCOUNT INPUTS AND OUTPUTS OF AEROBIC POND INFLOW, EVAPORATION AND RAINFALL AND IRRIGATION. IF THE POND VOLUME IS GREATER THEN THE MAXIMUM STORAGE THEN POND OVERFLOW WILL OCCUR>.*

```

[Pond.manager.end_of_day]
deficit = - 642.4 + sw_dep()
est_deficit = deficit - irrig_dep_def !attempt to estimate the deficit on the 'other'
paddocks
apswim_eo_yesterday = apswim.eo
Ana_perc_full = Ana_Pond_vol / Ana_Pond_volmax * 100
Aer_perc_full = Aer_Pond_vol / Aer_Pond_volmax * 100
report do_output

```

*<CALCULATION OF SOIL WATER DEFICITS WITHIN THE 4 IRRIGATED ZONES (IRRIGATION DAYS) >.*