DECISION SUPPORT TOOLS TO REDUCE THE RISK OF GROUNDWATER NITRATE CONTAMINATION AT LAND TREATMENT SYSTEMS

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ABSTRACT

One of the main environmental concerns of a land treatment system (LTS) for effluent disposal is the risk of groundwater contamination. Although there have been many studies on the leaching of nitrate-nitrogen (NO_3-N) from soils, there is still a need for procedures by which engineers can design and operate a LTS in ways that minimise NO_3-N leaching into the groundwater. The goal of this study was to develop decision support tools to assist managers reduce groundwater contamination at a LTS.

An exploratory field scale investigation was undertaken at a land treatment site belonging to Carterton District Council of Wellington Region, New Zealand. The field scale investigation demonstrated that increasing effluent application rates resulted in increased groundwater NO_3-N concentrations. During summer and autumn the NO_3-N groundwater concentrations remained below the maximum permissible limit (MPL) of 11.3 mg/L, but during winter, when significant rainfall events caused leaching, the groundwater NO_3-N concentrations increased and exceeded the MPL.

Based on this finding a simple hydrological model was developed to predict the occurrence of leaching events. This model was capable of predicting the timing and amount of leaching, but it was not able to quantify the nitrogen (N) loss that would occur during these leaching events. The model may be a useful tool to manage effluent application during the drier months of the year to avoid leaching and maximise the plant uptake of effluent nutrients prior to the inevitable onset of leaching during the wetter winter months.

A decision support system (DSS) based on LEACHN was developed to predict the movement of N in a LTS. The LEACHN model was parameterized using data from the Carterton site and was then used to explore alternative effluent disposal strategies at the site. The ability of the OVERSEER® model to describe N leaching at the Carterton site was compared with that of the LEACHN model. Finally the parameterized LEACHN model was successfully tested against a separate data set from an effluent disposal area on a commercial dairy farm. The knowledge gained by using this DSS would enable managers to monitor and manage a LTS in a way that minimises the impact of contaminant leachate.

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DEDICATED TO MY PARENTS

MR. & MRS. MUHAMMAD HUSSAIN (Late)

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LIST OF ABBREVIATIONS

AET	*Actual Evapotranspiration*
AEV	*Air Entry Value*
ANIMO	*Agricultural Nutrient Model*
APSIM	*Agricultural Production Systems Simulator*
BCAM	*'b' Exponent of Campbell's Equation*
BOD	*Biochemical Oxygen Demand*
C:N	*Carbon Nitrogen Ratio*
Ca	*Calcium*
CALF	*CALculates Flow*
CDC	*Carterton District Council*
CDE	*Convection Dispersion Equation*
CEC	*Cation Exchange Capacity*
Cl	*Chloride*
CLUES	*Catchment Land Use and Environmental Sustainability*
DBH	*Diameter at Breast Height*
DCD	*Dicyandiamide*
DOC	*Dissolved Organic Carbon*
DON	*Dissolved Organic Nitrogen*
DSE	*Dairy Shed Effluent*
DSS	*Decision Support System*
EC	*Electrical Conductivity*
EF	*Modelling Efficiency*
E-nitens	*Eucalyptus Nitens*
E-ovata	*Eucalyptus Ovata*
EPA	*Environmental Protection Agency*
ET	*Evapotranspiration*
EU	*European Union*
FC	*Field Capacity*
FDE	*Farm Dairy Effluent*
FPS	*Fine Particle Suspension*
GLEAMS	*Groundwater Loading Effects of Agriculture Systems*
K	*Hydraulic Conductivity*

KCl	*Potassium Chloride*
K_d	*Distribution Coefficient for Nitrate Nitrogen*
LEACHB	*Microbial Population Sub-Model of LEACHM*
LEACHC	*Heavy Metals Sub-Model of LEACHM*
LEACHM	*Leaching Estimation and Chemistry Model*
LEACHN	*Nitrogen Sub-Model of LEACHM*
LEACHP	*Leaching Estimation and Chemistry Model – Pesticides*
LEACHW	*Water Transport Sub-Model of LEACHM*
LLC	*Land Limiting Constituent*
LTS	*Land Treatment System*
LUCI	*Land Use Change and Intensification*
MEDLI	*Model for Effluent Disposal using Land Irrigation*
Mg	*Magnesium*
MPL	*Maximum Permissible Limit*
NCSWAP	*Nitrogen and Carbon Transformation in Soil-Water-Air-Plant systems*
NH_3	*Ammonia*
NH_4-*N*	*Ammonium Nitrogen*
Ni	*Nickel*
NIWA	*National Institute of Water and Atmosphere*
NLE	*Nitrogen Leaching Estimation*
NO_3-*N*	*Nitrate Nitrogen*
NPLAS	*Nitrogen and Phosphorus Load Assessment System*
P	*Phosphorus*
PET	*Potential Evapotranspiration*
PNM	*Precision Nitrogen Management*
PRZM	*Pesticide Root Zone Model*
Q_{10}	*Base Temperature at which rate constants change for each* 10^0C *increase*
RMA	*Resource Management Act*
RMSE	*Root Mean Square Error*
RNG	*Random Number Generator*
ROTAN	*ROtorua and TAupo Nitrogen*
RZWQM	*Root Zone Water Quality Model*
S1	*Just rainfall and no effluent irrigation*

S2	*Rainfall and irrigation without N*
S3	*Rainfall and irrigation with effluent that contains N*
SMC	*Soil Moisture Content*
SMS	*Soil Moisture Storage*
SNC	*Soil Nutrient Content*
SPARROW	*Spatial Referenced Regression On Watershed attributes*
SPASMO	*Soil-Plant-Atmosphere System Model*
SSF	*Site Specific Factors*
TDR	*Time Domain Reflectrometry*
TKN	*Total Kjeldhal Nitrogen*
VS2DT	*Variable Saturated 2 Dimensional Transport*
WAVE	*Water and Agrochemicals in the soil, crop and Vadose Environment*
WHC	*Water Holding Capacity*
WHO	*World Health Organisation*
Zn	*Zinc*

CHAPTER 1

INTRODUCTION AND OBJECTIVES

1.1 Introduction

Application of wastes onto land is an approach that is at the interface of waste management, land use and water quality management (Heinzmann and Sarfert, 1995). The main objective of a land treatment system (LTS) is to provide a sustainable process for the treatment and disposal of effluent (sewage, industrial waste, or agricultural waste) through natural processes occurring in the plant-soil-water matrix. Sustainability generally implies a high degree of waste management and non-degradation of the receiving environment. Although land application of wastewater is a good management option (Feigin et al., 1991), it may lower soil and water quality if not properly managed and maintained.

For many years, soil was thought to be the 'perfect filter' - nutrients in the wastewater being confined to and renovated in the uppermost soil horizon. Recently it has been realized that these nutrients can move through the soil, resulting in groundwater contamination (McLeod et al., 1998).

One of the main environmental concerns about land treatment of effluent is the risk of groundwater contamination. It is appropriate to consider at the outset what is meant by the term "*land treatment*" as that will fundamentally affect the design ethos. *Land treatment is the intimate mixing or dispersion of wastes into the upper zone of the soil-plant system with the objective of microbial stabilization, adsorption, immobilization, selective dispersion or crop recovery, leading to an environmentally acceptable assimilation of the waste* (Overcash and Pal, 1979). Waste treatment normally refers to the transformation of a material such that it no longer represents a risk to human health or the environment. Where an action has resulted in no risk reduction, that management approach could be regarded as a disposal activity. This rather simplistic definition is given by Riddell-Black (1995).

Land application of wastes is becoming more widespread as regulatory authorities move to protect water quality by restricting waste disposal into rivers, lakes, and the marine environment. It is not clear however, that soil is in fact an appropriate dumping ground for all our wastes (Cameron et al., 1997). The pressure to dispose of wastes onto land rather than into water often results in engineers being forced to design land treatment systems with little rigorous scientific information to guide them. One of the main problems is that there is a wide range of waste materials with different physical, chemical and biological characteristics such that it is inappropriate and indeed risky to transfer guidelines from one waste disposal system to another.

Groundwater is a resource of significant social value; people rely on it for sustaining life and livelihoods. Recognition that this resource is subject to degradation has heightened interest in formulating management practices that reduce the risk of groundwater contamination. Development of such management practices requires two steps: firstly design and secondly management of land treatment systems.

1.1.1 Design and Management of a Land Treatment System

The design of a LTS should aim to balance the loading rates of water, nutrients, and toxins with the ability of the site and vegetative cover to safely convert, absorb, use or store them. A fundamental principle emerges as extremely important in all future discussions of land treatment of effluents: i.e. the design and subsequent management of a plant-soil receiver system for any type of waste is absolutely site-dependent. The specific site must first be identified, monitored, and the data used to determine the rate of application of a waste or constituents within the waste. This has been learned from experiences with successful and unsuccessful systems, in which the critical difference was in accounting for site specific properties (Overcash and Pal, 1979).

Land based treatment design criteria are generally not transferable from location to location - only the methods of data collection and design calculations are general. To some, this limitation is severe and may have contributed to the slower acceptance of land application by engineers and regulating bodies. However, this design requirement is the essence of the need for engineering in this field and is more than offset by economic and technical advantages of land based treatment of wastes.

To operate a system within safe limits, it is necessary to make some site specific measurements that provide guidelines as to when (what time of the year) and how much (kg/ha/year) effluent to apply. From information collected during the field investigation, the engineer or manager can confirm the suitability of the site for the identified land treatment processes.

The amount and timing of effluent is also influenced by seasonal conditions, day to day weather, soil physical characteristics and current soil moisture levels. Other factors include the allowable nitrogen (N) loading rate and the occasional need to leach salts from the root zone.

In New Zealand, little information is available on environmentally sustainable N loading rates for effluent application onto land. Despite intensive research on soil reactions of N fertilizers in New Zealand and world wide, loading rates for fertiliser-N are yet to be determined for New Zealand soils (Selvarajah, 1996). A review of national loading rates is given in Chapter 2. Soil and groundwater reaction to effluent application is a more complex and dynamic process than that of fertilizer-N. Consequently, intensive research is required to determine suitable N loading rates for a wide range of effluents, different soil types and prevailing climatic conditions in New Zealand.

1.2 Rationale

New Zealand generates large quantities of agricultural, industrial, and municipal wastes. As authorities move to protect the environment by regulating waste disposal practices, environmentally sound methods of waste disposal are being sought. In particular, land application of wastes as a means of disposal, nutrient re-cycling, and water conservation is becoming increasingly popular. In New Zealand, the disposal of wastes into the environment is controlled by regional councils pursuant to the Resource Management Act 1991 (RMA). The purpose of the Act is to promote sustainable management of natural and physical resources, and relies on safe-guarding the environment by avoiding, remedying or mitigating any adverse effects of activities on the environment. Although the RMA has not placed any higher emphasis on land treatment and disposal than the previous legislation, the requirement that the proposal is 'sustainable', and that alternative treatment and disposal systems are considered, results in most circumstances in consideration of land treatment and disposal along with more traditional treatment and disposal options. The lack of suitable alternative options has brought land treatment and disposal into serious consideration in a large number of applications.

The land application of treated effluent, as opposed to direct discharge to local rivers or streams, is the preferred option for disposal and treatment in New Zealand as regulatory authorities move to protect the environment and meet Maori spiritual and cultural values. In New Zealand, there is increasing government, social and cultural pressure to recognise Maori cultural values and practices, particularly regarding the use of natural resources (i.e. land, water, and air). Under the RMA (1991) regional councils are required to recognise and provide for the relationship of Maori and their cultural and traditions with their ancestral lands, water, waahi tapu (i.e. a place that is spiritually sacred and culturally important to a local iwi or hapu) and other toanga (i.e. treasured things which could be tangible and intangible). Especially, a particular regard must be given to the ancestral Kaitiaki or guardian role of tangata whenua (means people of land), and the principles of the Treaty of Waitangi need to be taken into account. When planning to undertake an activity such as land application of effluent, local iwi may need to be consulted on the proposal.

Land application recognises the value of nutrients in the effluent. However, it is not without risks. At present the management of effluent-irrigated land treatment systems is primarily concerned with potential nitrate nitrogen (NO_3-N) contamination of groundwater. The health hazards of NO_3-N in drinking water are well documented, although not universally agreed (Heathwait et al., 1993). The links between different land uses and the contamination of groundwater have not been well established quantitatively, either in New Zealand, or in other countries. All regional councils and unitary authorities in New Zealand have policies and practices in place to deal with NO_3-N contamination of groundwater related to land management. Those policies and practices need periodic review and revision.

There appears to be a severe shortage of information on the fate of wastes applied to New Zealand soils; particularly considering the wide variety of wastes and disposal conditions that exists. Land disposal of wastes has nevertheless been identified as a potential source of pollution of New Zealand aquifers (Burden, 1982; Barton et al., 2005; Cichota and Snow, 2009) and regional authorities restrict the rate and conditions of disposal of most wastes.

Land application, however, can still pollute receiving waters and there is a need to develop better management options for a LTS that reduce pollution by providing for more efficient monitoring/management. The question is - can we manage a LTS to reduce the risk of groundwater contamination? The fundamental goal of this study is to develop tools that will assist managers to reduce the risk of groundwater contamination at a land treatment system.

1.3 Aim of the Study

To investigate approaches to managing land treatment systems in a way that the risk of groundwater contamination is minimized.

The study was divided into five interrelated objectives:

1. To carry out a preliminary investigation on a LTS to identify factors that are important in causing groundwater contamination.
2. To develop a simple hydrological model to describe the short terms effects of applying effluent to a LTS and to identify the parameters that are practicable and feasible to monitor at a LTS to enable better short-term management of the disposal site.
3. To investigate the potential of the LEACHN (Nitrogen version of Leaching Estimation and Chemistry Model) model to describe and predict the leaching of NO_3-N from effluent applied to a soil at a laboratory scale level.

4. To use the LEACHN model to describe the field-scale movement of NO_3-N through soil to groundwater at a LTS.
5. To undertake a critical analysis of the usefulness of the LEACHN model in 'predictive mode' using past weather forecasts to provide information on the design and operation of a LTS.
6. To undertake a detailed evaluation of the LEACHN model in order to assess its usefulness as a decision support tool for managing a LTS.

The philosophy underlying this study was that if a LTS is appropriately monitored and managed, the quality of groundwater can be protected from the effects of contaminant leachate. The tools investigated in this study will assist the manager of a LTS and other local authorities to make better informed decisions in the future as to whether land treatment is feasible and if so, to determine the land area required to meet the limiting load factor of the effluent. This may assist the regional councils to process resource consent applications to discharge effluent onto lands.

1.4 The Structure of the Study

The present study is based on field and laboratory scale experiments (Chapters 3, 4 and 5) followed by modelling studies (Chapters 6, 7 and 8) with the aim of developing decision support tools, or a management approach, that minimises undesirable environmental impacts at a LTS. The research undertaken in this study is, in part, presented here in the form of peer-reviewed journal papers. Six papers have been published in international refereed journals. These papers are included as Appendices 1 - 6. Some changes to the format and content of the papers have been made to achieve consistency, provide better linkage throughout the thesis, and to reduce repetition. The basic structure of several of the papers however remains intact. Abstracts of the papers have been removed and the references have been moved to the end of the thesis. All papers have multiple authors except one (Chapter 7). The contribution of the authors is as follows:

Babar Mahmood:	**Principle Author/Investigator**
Carried out:	All the planning, designing and execution
	Data collection, collation and analysis

	Physical and chemical analysis
	Development and testing of the decision support model
	Manuscript preparation, writing, and editing
Gavin L Wall:	**Advisor**
Aided the study by:	Discussing data collection, methodology and results
	Discussing the decision support model development and testing at field and laboratory scales
John M Russell:	**Advisor**
Aided the study by:	Discussing the decision support model testing at laboratory scale
Russ W Tillman:	**Advisor**
Aided the study by:	Discussing the detailed evaluation of LEACHN model

The details of the published papers are as follows:

1. Mahmood, B., and Wall, G. L. (2001). Land Treatment Practices of Wastes and Their Implications on Groundwater Quality in New Zealand: A Review. *International Agricultural Engineering Journal*, 10(3&4), 121-150. The Asian Association of Agricultural Engineering (AAAE). Asian Institute of Technology (AIT), Bangkok, Thailand.
2. Mahmood, B., and Wall, G. L. (2001). The Environmental Impact Assessment of Effluent Irrigation onto the Land - A Case Study in New Zealand. *International Agricultural Engineering Journal*, 10(3&4), 209-230. The Asian Association of Agricultural Engineering (AAAE). Asian Institute of Technology (AIT), Bangkok, Thailand.
3. Mahmood, B. Wall, G. L., and Russell, J. M. (2002). A New Management Technique to Reduce the Risk of Groundwater Contamination at a Land Treatment System (LTS). *International Agricultural Engineering Journal*, 11(2&3), 157-171. The Asian Association of Agricultural Engineering (AAAE). Asian Institute of Technology (AIT), Bangkok, Thailand.
4. Mahmood, B. Russell, J. M., and Wall, G. L. (2002). Field Scale Nitrate Simulation. *Transactions of the ASAE* (American Society of Agricultural

Engineers), 45(6), 1835-1842. Department of Food, Agricultural and Biological Engineering. The OHIO State University, Columbus, USA.

5. Mahmood, B. Wall, G. L., and Russell, J. M. (2003). A Physical Model to Make Short-term Management Decisions at Effluent-irrigated Land Treatment Systems. *Agricultural Water Management – An International Journal*, 58, 55-65. Elsevier Science B. V., Molenwerf 1, 1014 AG Amsterdam, The Netherlands.

6. Mahmood, B. (2005). Analysis of a LEACHN-Based Management Technique in Predictive Mode. *Agricultural Water Management*, 75, 25-37. Elsevier Science B. V., Molenwerf 1, 1014 AG Amsterdam, The Netherlands.

<u>Thesis Outline:</u>

Chapter 1 - Introduction and Objectives

This chapter provides a brief introduction to the design and management of land treatment systems, site specific factors, the problem statement, the aim and specific objectives of this study, and the structure of the thesis.

Chapter 2 - Land Treatment Practices of Wastes and Their Implications on Groundwater Quality in New Zealand – A Review.

This chapter provides the literature review of land application of wastes and their impact on soil, plant and groundwater, the NO_3-N leaching losses, and also summarises the further developments in the area (Appendix 1).

Chapter 3 - The Environmental Impact of Sewage Effluent Irrigation onto Land – A Case Study in New Zealand (relates to objective 1)**.**

This chapter describes an exploratory field scale study that was undertaken at a land treatment site of Carterton District (Appendix 2).

Chapter 4 - A Physical Model to Make Short Term Management Decisions at Effluent-Irrigated land Treatment System (relates to objective 2)

This chapter introduces the concept of a physical model to check the short term effects of land application of wastewater on groundwater quality at a LTS (Appendix 3).

Chapter 5 – Laboratory Scale Testing of the LEACHN Model (relates to objective 3)

This chapter investigates using a LEACHN-based model to describe and predict leaching of NO_3-N following application of effluent to soil in a glasshouse experiment (Appendix 4).

Chapter 6 – A Long-term Modelling Approach to Assist Management of a LTS (relates to objective 4)

This chapter uses a LEACHN-based model to describe the leaching of NO_3-N to groundwater at a LTS (Appendix 5).

Chapter 7 - Analysis of the LEACHN-Based Management Technique in 'Predictive Mode' (relates to objective 5)

This chapter covers the critical analysis of the decision support model in 'predictive mode' (Appendix 6).

Chapter 8 – Solute Transport Modelling (relates to objective 6)

This chapter covers a detailed analysis of the LEACHN model to investigate the usefulness of this model as a decision support tool. This work has not been published yet.

Chapter 9 summarise the results and insights gained during the study.

CHAPTER 2

LITERATURE REVIEW

2.1 Introduction

Land application of wastes can have a variety of beneficial or detrimental effects on groundwater quality, and plant and soil physical conditions, depending on the characteristics of the waste and the soil. Research and monitoring/management programmes are therefore necessary to ensure that waste application systems are sustainable and that they do not damage soil and groundwater quality.

Over the last decade, regulatory pressures and public concerns about the protection of water quality has increased the need for land disposal of municipal, industrial and agricultural effluents. Land treatment systems can have considerable positive effects on the local environment. Up to 80-90% of the nutrients in the effluent can be trapped by the soil in the land treatment system resulting in a considerable reduction in the quantity of nutrients reaching lakes and rivers.

2.2 Structure of Literature Review

This literature review explores the topic of land treatment practices and their implications for groundwater quality in terms of NO_3-N contamination. This review is broken down into three main topics. The first topic (Sections 2.3-2.5) is devoted to land application of wastes and its impact on groundwater quality in terms of NO_3-N contamination; reasons for land applications of wastes; information needed to design and manage a LTS; the need for groundwater monitoring; and parameters to consider for monitoring and management of a LTS. The second topic (Section 2.6-2.7) is devoted to NO_3-N leaching losses from soils; factors affecting NO_3-N leaching losses; N cycling and the use of nitrification inhibitors; and dissolved organic nitrogen (DON). This section also includes a review of effluent application rates and N leaching losses at LTS in New Zealand. The last topic (Section 2.8) is devoted to the development, use and application of water and solute transport models; and modelling of water and nutrient movement in a LTS.

2.3 Land Application of Wastes and its Impact on Groundwater Quality in terms of NO_3-N contamination

Sewage and dairy farm effluents are full of nutrients and their application onto land (instead of disposing into local rivers and streams) is the preferred way of disposal in New Zealand, as regulatory authorities move to protect the environment and meet Maori cultural and spiritual values. Recent studies have involved a number of different effluents applied to a range of plant species and land uses. However, in this section, the main focus of the literature reviewed is on the environmental impact of sewage, dairy farm, meatworks and piggery effluents, biosolids and land use. Information on the environmental impacts of pesticides, herbicides, fertilisers, and other industry wastes can be found in SNZ (1993), Cameron (1993), Cameron et al. (1997), and Muller et al (2007).

2.3.1 Land Disposal of Sewage Effluent

Because of the usefulness of LTS in many ways (e.g. recovery of nutrients, maintenance of Maori cultural values, crop irrigation and replenishment of groundwater, etc.), the land application of wastewater has been promoted and used in many rural areas of New Zealand since the implementation of RMA (1991). However, there is still a risk of groundwater contamination.

A number of earlier studies have shown that land disposal of sewage effluent may contaminate underlying groundwater (Quin, 1978; Lucas and Reeves, 1981). The extent to which this occurs depends on a range of factors, (Close, 1984), which affect the irrigated effluent as it percolates through the soil and gravel strata and disperses in the groundwater system. These factors include soil depth, soil temperature, the amount of drainage to groundwater, and dispersion.

Soil depth is important in determining the total cation exchange capacity (CEC). Ho et al. (1980) constructed a model for nitrogen removal from effluent and found that CEC and hydraulic conductivity were the most significant soil properties for achieving nitrogen removal. Soil temperature, which is seasonal, may affect the percolating effluent in a number of ways. Low soil moisture conditions in summer can affect the mode of drainage, causing more cracks to become available for rapid drainage, thus reducing the proportion of effluent coming in contact with the soil. This also reduces the effective time delay achieved by the soil system. Drainage will also be affected by the seasonal variation in evapotranspiration. Dispersion in the groundwater attenuates the concentration of ions as they move away from the disposal area, and has a major influence on downstream concentrations (Thorpe et al., 1982).

Application of sewage effluent to sandy soil under lucerne at 480 kg N/ha/year increased leaching losses from about 60 kg N/ha/year to about 240 kg N/ha/year. The response was limited by poor winter growth and only 47% of the applied N was removed by lucerne - compared with 50% removed by Nui ryegrass, 60% by prairie grass and 84% by tall fescue (Stevenson and Wilcock, 1979).

Another Canterbury study determined the effects of treated sewage effluent applied by flood irrigation on the groundwater N concentration under a grazed ryegrass/clover pasture on a Templeton silt loam (Close, 1984). Effluent irrigation with a N content of 116 kg N/ha/year increased the mean groundwater N concentration from a background value of 7.8 g/m^3 to 9.0-10.4 g/m^3 NO_3–N, downstream of the irrigation area. Shallow groundwater NO_3–N concentrations were greater in winter than summer, as shallow groundwater in winter could have been influenced by drainage from effluent irrigation. Estimated leaching losses from this site were 180-200 kg N/ha/year and represented most of the N applied in the effluent.

Quin (1984) studied New Zealand's oldest existing sewage effluent irrigation scheme, situated at Templeton, Christchurch that was commenced in 1958. This Templeton scheme has demonstrated that effluent irrigation is a practicable method of utilising both the water and nutrients contained in the effluent. The major problems that may be associated with land disposal of sewage effluents and sludge include contamination of groundwater by the leaching of the chemicals through the soil profile, the build-up of biotoxic elements in soils and plants, and the survival and distribution of disease organisms. He concluded that some contamination of groundwater (either chemical or microbiological or both) is almost certain to occur, and the installation of monitoring wells is an essential part of any scheme. If wells show that the level of contamination is unacceptable, effluent management can be altered by whatever means necessary (e.g. pre-treatment, reduced application rates, pasture harvesting) to reduce contamination to acceptable levels. He mentioned that there was no technological reason why land treatment of effluent could not be very much more widely used than it was at that time. The reason it was not, despite the savings in capital and running costs compared to conventional sewage treatment, was largely a result of the practical difficulty potential scheme designers would face in trying to assimilate information and advice from representatives of the many different disciplines involved, including engineers, irrigation scientists, soil scientists, agronomists, hydrologists and water quality researchers.

Quin (1978) used detailed chemical analysis of percolating soil water to interpret the effects of the effluent irrigation on groundwater quality. Due to biological conversion and chemical displacement processes in the soil, the percolating soil water contained higher concentrations of NO_3-N, Calcium (Ca) and Magnesium (Mg) than the effluent, but lower concentrations of NH_4-N and K^+ (potassium). The ratio of the chemical composition of the groundwater to that of the percolating soil-water was much closer to unity for the downstream bores than for the upstream control bore.

In the early nineties, a LTS was established in the Whakarewarewa forest of Rotorua, New Zealand, to reduce nutrient loading to Lake Rotorua, which was affected by eutrophication (Smith, 1996). On average, 71 mm/week of treated effluent was applied to the forest. The N and P (phosphorus) loading rates were approximately 406 kg/ha/year and 98 kg/ha/year, respectively, (Tomer at al., 2000). The land was planted with *Pinus Radiata*. The area has a mild climate and the effluent was applied throughout the year on weekly basis. The study showed that soil pH, invertase activity (i.e. an enzyme that catalyzes the hydrolysis of sucrose into glucose and fructose), mineralisable N, and extractable NO_3-N were increased by effluent irrigation. No change in total N & C, basal respiration, microbial mass and extractable ammonium was observed (Schipper et al., 1996). Overall the effects of treated effluent on soil biological properties were not as significant as those observed at other land treatment systems (Ross et al., 1982).

Since 1991, considerable research has been undertaken into the performance of the Whakarewarewa forest based LTS (Cook et al., 1994; Schipper et al., 1996; Tomer et al., 1997; Magesan et al., 1998; Barton et al., 1999; McClay et al., 2000; Thorn et al., 2000; Tomer et al., 2000; Wall et al., 2008), and the major findings of most of the studies were summarised by Magesan and Wang (2003). According to Magesan and Wang (2003), irrigation with tertiary treated wastewater at the Rotorua LTS significantly increased macro porosity, but did not greatly change most other soil physical properties. The LTS was very effective at retaining P with little or no added P moving below the pine root-zone.

The LTS was, however, less successful at retaining N. Annual N application rates averaged approximately 406 kg N/ha and although generally <150 kg/ha was lost through leaching of NO_3-N, this was close to the permitted limit for the LTS (Magesan and Wang, 2003). Only 9% of the added N was taken up by the trees and denitrification and volatilisation losses were thought to be low. Therefore storage as organic N in the soil and understory seems to have been significant at this LTS. This supports the conclusion of Snow et al. (1999b) quoted below.

In Australia, the Wagga Wagga Effluent Plantation Project (Myers et al., 1994) investigated the effects of applying secondary-treated sewage effluent to plantations of *Pinus Radiata* and *Eucalyptus Grandis*. The results of this major study and the associated modelling have been reported by Falkiner and Smith (1997) and Snow et al. (1999a and b). Snow (1999b) concluded from the predicted N balance that there was:

"…a high potential for N leaching if the amount of N applied in the effluent is greater than the accumulation of N in the trees. On permeable soils, despite the frequent irrigation and large amounts of N and C in soil water, denitrification is likely to account for a relatively small fraction of the amount of N added in the effluent".

In India, Singh and Bhati (2003) reported on the application of municipal effluent to *Eucalyptus* seedlings, and in the United Kingdom Moffat et al. (2001) studied the application of sewage sludge and wastewater to short rotation hybrid poplar. An interesting alternative to the use of a forested LTS as a way of disposing of effluent was the work of Candela et al. (2007) who conducted an assessment of the soil and groundwater impacts resulting from treated urban wastewater reuse on a golf course in Girona, Spain.

A number of workers (e.g. Clapp et al., 1999; Sharma and Ashwath, 2006; Tzanakakis et al., 2009) have compared the performance of different plant species at effluent disposal sites and concluded that *"... substantial improvement in the performance of the LTS in terms of nutrient removal can be achieved through the selection of appropriate plant species".*

In another study, Barton et al. (2005) examined the effects of land application of domestic effluent onto four types of soils i.e. well drained Allophanic soil (Typic Hapludand), a poorly drained Gley soil (Typic Endoaquept), a well drained Pumice soil formed from rhyolitic tephra (Typic Udivitrand), and well drained Recent soil formed in sand dunes (Typic Udipsamment). They also examined the impact of land application of effluent on plant uptake and nutrient leaching. The effluent was applied onto grassed intact soils cores (500 mm diameter and 700 mm deep) at a rate of 50 mm/week for two years. The pasture was cut and removed. The irrigated soils received between 746 and 815 kg N/ha and unirrigated treatments received 200 kg N/ha of dissolved inorganic fertiliser over the two years of irrigation. They found that the plant uptake of N was between 186 and 437 kg N/ha/year and was significantly increased from all soil types. The N leaching losses were increased for all soils and ranged between 17 and 184 kg N/ha. Specifically, Gley and Recent soils had a significant increased N leaching losses after two years in comparison to control treatments. The N leaching losses from effluent treatments were mainly (69-87%) in organic form. They reported that N leaching losses from the Gley soil were due to preferential flow that reduced contact between the applied effluent and soil particles. Also, the greater N leaching losses from the Recent soil were due to the increased leaching of native soil organic N because of high hydraulic loading from effluent irrigation.

Recently, Treweek et al. (2010) commenced a field scale experiment to study the N leaching from effluent irrigated pasture on a vitrand (pumice soil), Taupo, New Zealand. The study aimed to quantify the N leached from the soil beneath the LTS under four different effluent loadings (i.e. 650, 550, 450, and 0 kg N/ha/year). The resource consent was granted to apply 550 kg N/ha/year at the site; however a higher rate of application was also sought from the Taupo District Council. Twelve lysimeters (300 mm wide and 450 mm deep) were installed in each treatment i.e. forty eight in total. Each lysimeter had a leachate collection system beneath the undisturbed soil core and the leachate was collected on a monthly basis and the amount of leachate was recorded. The initial results of the study showed that 4.5, 1.2, 1.1, and 0 kg N/ha was leached from 650, 550, 450 and 0 kg N/ha treatments, respectively, over a period of two months. The trial is in progress and will run for another four years.

2.3.2 Land Disposal of Farm Dairy and Other Effluents

In New Zealand there has been a considerable focus on the land disposal of farm dairy effluent (FDE) as Regional Councils have actively discouraged the discharge of treated effluent directly into waterways. The effect of land disposal of FDE on the environment and soil properties has been reviewed extensively by a number of authors (Houlbrooke et al., 2004; Wang et al., 2004; Hawke and Summers, 2006). The following statement by Houlbrooke et al. (2004) at the end of their review provides a good summary of the current state of knowledge.

"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. ...Improvement in the management of land applications can be made to prevent 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".

Following on from this observation, Hedley et al. (2005) reviewed the recent research into the effectiveness of "deferred irrigation" of FDE as a management strategy to decrease the nutrient enrichment of drainage water from artificially drained dairy pastures. This research trial illustrated how scheduling of FDE irrigation events to avoid drainage and runoff (storage and deferred irrigation) could maximise soil treatment of FDE and cause minimal nutrient loss in drainage. In contrast, the application of a single, poorly-timed FDE irrigation event, which was well in excess of the current soil moisture deficit, could cause relatively large losses of nutrients. The research showed that the accelerated nutrient loss in drainage water from paddocks used for FDE application could be avoided but this would require FDE management systems that included customised pond storage, effluent block nutrient budgets and a soil water balance monitoring system to guide irrigation scheduling. One of the consequences of meeting the proposed new deferred irrigation criteria would be that there would be fewer safe irrigation days and FDE pond storage capacity would have to increase on farms.

Application of FDE to pasture can affect the mineral composition of the pasture, which may then have implications for animal health and production. Bolan et al. (2004) found elevated K concentrations in pasture irrigated with FDE and pointed out that this may have implications for the calcium and magnesium nutrition of grazing dairy cows.

In the Waikato region of New Zealand, the land application of FDE is a very common practice. The percentage of dairy farmers in the Waikato region, who apply effluent to land has increased from 35% in 1993 to 100% in 2004 (Hawke and Summers, 2006). While land application of effluent diverts the nutrients from waterways, it has the potential to increase recharge to the groundwater accompanied by salts and NO_3-N. It can also result in an accumulation of salt and other minerals, such as P, in the soil with a resulting increased risk of runoff of these contaminants to rivers, streams and lakes.

A comprehensive review of land application of FDE and its impact on soil properties is also given by Hawke and Summers (2006). In their review, they mentioned that changes to soil properties are slow and gradual and are not always easy to assess. The application of FDE is the application of different forms of nutrients, which may only become available to plant slowly and may also leach slowly. However, there are instances where the capacity of the soil to store nutrients is reached and the soil subsequently acts as a point source of pollution (Phillips, 2002). Hawke and Summer (2006) reported that land application of FDE appears to increase the concentrations of nutrients such as N and P, organic C and organic matter, and plant available nutrients such as potassium and calcium. Over time, the application rate of these may exhaust the soil-plant system's ability to store and use them thus leaching is inevitable. They also reported that soil chemical properties are affected by soil physical properties. For example, an increase in soil organic matter content can affect the infiltration capacity, hydraulic conductivity and porosity of soil. The effects of FDE on soil physical properties are quite variable. Both increases and decreases in soil hydraulic conductivity have been observed. However, long term fertility in terms of total N and P concentrations and plant available nutrients has generally improved due to application of FDE. Also, land application of FDE has resulted in greater and more diverse microbial biomass.

Similarly, the land application of dairy factory effluent affected a range of physical properties of Horotiu and Te Kowhai soils of Waikato region of New Zealand. The hydraulic conductivity of both soils was increased. The bulk density of Horotiu soils was decreased but there was no change of bulk density for the Te Kowhai soils. The macoporosity of both soils was not changed (Sparling et al., 2001).

The effects on groundwater NO_3-N contamination of 15 years of casein wastewater irrigation were investigated on a silt loam soil under pasture in the Manawatu (McAuliffe et al., 1979). Annual N loading was approximately 3000 kg N/ha/year, with individual applications of 250 kg N/ha. The mean NO_3-N concentration in soil solution was about 23 g/m^3 but NO_3-N concentrations below the water table were only about 1.5 g/m^3. Complete N budgets were not calculated for this site, but it appears that about one third of the applied N was retained as soil organic N. The low NO_3-N concentrations in groundwater were attributed to dilution by rapidly flowing groundwater.

In contrast, the application of dairy factory waste at an average annual loading of 1200 kg N/ha/year to a sandy loam soil in Hamilton Basin increased groundwater NO_3-N concentrations under the study area from approximately 10 g/m^3 to 40-70 g/m^3 (Selvarajah et al., 1994). Groundwater NO_3-N concentrations outside the study area boundaries were also affected, with values of >40 g/m^3 NO_3-N recorded. Levels in soil solution at 30 cm depth were higher (400 g/m^3 NO_3-N) than those in the groundwater, with this reduction at depth being attributed to dilution and denitrification.

Although in New Zealand FDE is the major agriculture-derived effluent applied to land, there are others. Cameron and Di (2004) conducted a lysimeter experiment that compared the N leaching losses from a number of different farm effluents. They concluded that at similar rates of application N leaching losses decreased in the order cow urine>pig slurry>FDE>dairy pond sludge.

2.3.3 Land Disposal of Other Industrial Wastes

In New Zealand, land application of industry wastewater, especially from the meat industry, has increased over the past 30 years (Khan and Irvin, 2011). Land application of meat industry wastewater is one of the preferred disposal activities because it is considered to be a dilute organic fertiliser that contains significant quantities of plant nutrients such as N and P. Khan and Irvin (2011) examined the past and current practices of land disposal of wastewater and their effects on the environment. They reported that current wastewater irrigation practices could increase the NO_3-N concentrations in the groundwater even if the N loading rates were within acceptable levels. Khan and Irvin (2011) mentioned that the assessment of LTS should include an assessment of how much N is taken up by plants seasonally rather than annually, the hydraulic or nutrient limiting factors for the plants, and the ability to reduce N loading rates through land treatment and/or other seasonal options to dispose of treated wastewater.

Hawkes Bay is another area of New Zealand, where partially treated wastewater has been disposed of to land since the establishment of a meat plant in 1981. Initially, the wastewater, from the large meat processing plant in Hawkes Bay, was applied using a border-dyke irrigation system but later changed to low pressure spray irrigation. On average, the N loading rate was 2400 kg/ha/year but was reduced to 1300 kg/ha/year when the land disposal area was increased from 19 ha to 69 ha. During 1995, when the land disposal area was increased to 212 ha, the average N loading rate was 400 kg/ha/year. The results showed that NO_3-N concentrations in groundwater increased due to the historical wastewater irrigation at the site. Since 2000, the increase was observed in groundwater down-gradient of heavily loaded blocks, within the deeper intermediate aquifer. For shallow groundwater (i.e. 15 m deep), the NO_3-N concentrations increased sharply to 30 mg/L during April and December, 2000 before falling between December 2000 and February 2007 to pre 2000 levels of 1 mg/L. For the deeper intermediate aquifer (i.e. 45 m deep), the NO_3-N increased from 2.8 mg/L (during 2001) to 17 mg/L in March 2009 (Khan and Irvin, 2011). The increase in NO_3-N was attributed to the border-dyke irrigation system as the primary method of wastewater irrigation and also to the different travelling times of the groundwater in the shallow and intermediate aquifers.

Khan and Irvin (2011) also reported that a simple groundwater modelling for the Hawkes disposal site showed that peak NO_3-N concentration in the shallow and intermediate aquifers took 15 and 43 years, respectively, from the time of deposition of N onto the land and the plume effect could take up to 36 years to pass the monitoring location (i.e. 1900 m from the furthest extent of the disposal area). The estimated peak NO_3-N concentration was 26 mg/L. There is a possibility that the peak plume NO_3-N concentration has either been reached in the intermediate aquifer, with current concentrations at 17 mg/L, or is expected to be observed within the next few years (because of plume length and low velocity) before declining. For this Hawkes Bay site, the treatment of wastewater to reduce N loading rates to plant uptake rates needs to be considered. This integrated approach was also adopted by one of the sites in Waikato, where biological N removal was used to reduce the N concentrations in the wastewater prior to land application.

Guo and co-workers carried out a detailed study of the application of meatworks effluent to three different *Eucalyptus* short rotation forest species at Oringi, Dannevirke, New Zealand (Guo et al., 2002, 2006; Guo and Sims, 2003) and supported this work with more detailed studies of tree biomass production and leaf litter decomposition in growth cabinets (Guo and Sims, 2000, 2001).

A five-year monitoring programme of unconfined and semi-confined groundwater around three piggeries sited on shallow, stony soils was undertaken in Christchurch. The results suggested that application of piggery effluent at 200 kg N/ha/year did not appear to increase NO_3-N concentrations in either shallow or deep aquifers (Casey and Cameron, 1995), although these results were considered skewed by inadequate location of the monitoring wells. Nitrate-N concentrations in wells were extremely variable, with annual peaks recorded in shallow groundwater in late spring/early summer that reflected the seasonal pattern of soil drainage and water table levels. Nitrate-N concentration in groundwater declined with depth, as deep wells are less influenced by soil drainage recharge and NO_3-N leaching from soil than shallow wells. In some instances, elevated NO_3-N concentrations were recorded in groundwater, but it was not possible to conclusively determine the source of the leached NO_3-N.

Smith et al. (2003) conducted a study on forest land application of pulp and paper industrial effluent, as part of Forest Research Institute of New Zealand research programs. The study showed that land application of two types of effluents (i.e. thermochemical pulp mill and chemi-thermochemical pulp mill) was highly efficient in removing organic contaminants from the effluents. More than 92% and 96% of the total chemical oxygen demand and BOD was removed, respectively.

2.3.4 Land Disposal of Biosolids

In recent years, the public attitudes towards LTS have changed. Effluent is regarded as a resources rather than a waste. Also, land application of sludge is an acceptable activity as it can improve the site productivity by increasing the soil organic matter, which can lead to improved soil structure, decreased soil bulk density, and increased soil porosity, soil moisture retention, and hydraulic conductivity (Smith, 1996; Osborne and Michalk, 2000).

In New Zealand, sewage, dairy farm, meatworks, piggery, paper and pulp mill effluents are being applied onto forest based LTS (as mentioned above in Sections 2.3.1 - 2.3.3). The sludge (i.e. biosolids) from municipal wastewater treatment plants is also being applied to forest based LTS. For example, Christchurch city biosolids were applied to 1000 ha of forest in Nelson (Magesan and Wang, 2003). Municipal sludge usually contains high concentrations of N and this is a limiting factor when a forest based LTS is designed.

Magesan and Wang (2003) reviewed the research issues associated with land application of liquid and solid residuals on forest based LTS in New Zealand. They presented four case studies of liquid and solids application on forested lands. They reported that most biosolids in New Zealand have low concentrations of heavy metals.

It is well known that most of N in biosolids is in organic form and this needs to be mineralised before it is available for plant uptake. The ammonium may be oxidised to NO_3-N through the nitrification process. Therefore, it is important to match the pattern of biosolids applications and N mineralisation to the natural capacity of soil and water of the ecosystems in order to reduce the risk of contamination of water bodies from residual derived N. The strategies to minimise the amount of NO_3-N available for leaching from biosolids applied sites may include system design, crop selection, the method of injecting or incorporating of biosolids into soil, and management of vegetation (Magesan and Wang, 2003). Wang et al. (2003) reported that N mineralisation of solid residuals varies greatly from one source to another and is influenced by temperature and soil properties. Excessive leaching of NO_3-N may result from high rates of application of biosolids (Harrison et al., 1994).

Christchurch City Council planned a trial to apply biosolids to 1500 ha of *Pinus Radiata* plantation forest at a rate of 400 kg N/ha/year. Nitrogen in the biosolids was ^{15}N-enriched and the concentration of N ranged between 4.18 and 5.74% (Clinton et al., 2002). The biosolids were applied at three rates (0, 400, and 800 kg N/ha). The results of the study showed that despite the high N content, the soil concentrations of NH_4-N, NO_3-N, DON, total N, phosphate, Ni, and Zn were not increased above the control levels during the initial period after the applications. The results also showed that almost 57% of the total N applied remained in residual biosolids 15 months after application (Clinton and Leckie, 2002).

Another trial of forest based land application of biosolids was conducted in Nelson, New Zealand. Aerobically digested liquid biosolids from Nelson wastewater treatment plant were applied to 1000 ha of land planted with *Pinus Radiata*. The soil was low fertility sand at Rabbit Island (Kimberley et al., 2002). The ^{15}N in biosolids from Nelson varied between 5.00 and 8.71%. Initially, a 4 ha research trial was established to examine the possible effects of biosolids applications on tree growth, nutrition, soil and groundwater quality. The biosolids were applied in 1997 and 2000 at three rates i.e. 0 (control), 300, and 600 kg N/ha to six year old trees (Robinson and Wilks, 2001). The study showed that the tree growth was enhanced significantly (in terms of increase in tree basal area and volume) by the biosolids treatment (Kimberely et al., 2002), and this increase was due to the improved N supply (as high N concentrations in pine foliage were observed after biosolids applications). The average tree basal areas of the high treatment (600 kg N/ha) and the standard treatment (300 kg N/ha) were 48% and 30% greater than the tree basal area in the control treatment. The live volumes of the high treatment and standard treatment were 52% and 34% greater than the control treatment. Soil analysis showed that there were no significant changes in soil pH and concentrations of C, N, calcium, magnesium, potassium and heavy metals. Also, the groundwater quality was not affected by the land application of biosolids at Rabbit Island (Wang et al., 2002).

2.3.5 Land Use and Irrigation Effects on Groundwater

Providing a definitive link between land use and NO_3-N concentration in groundwater is difficult unless one land use type is dominant. In the Waikato basin, leaching losses from intensively farmed land have been estimated to be at least 60 kg N/ha/year (Selvarajah et al., 1994). Most of this leaching occurs during autumn and winter as shallow aquifers are recharged by infiltrating rainfall. These leaching losses have resulted in elevated levels of NO_3-N in shallow (<30 m) bores. The NO_3-N levels decreased with aquifer depth mainly due to dilution.

When irrigation is applied at optimum rates for plant growth, plant N uptake can be increased, so decreasing the amount of NO_3-N in the soil that can be leached. However, when irrigation causes additional drainage, N losses may be increased. If the additional drainage amount is large (as in some border-dyke irrigation schemes), irrigation may dilute the N concentrations transported to groundwater. Excessive irrigation that causes wet soil conditions over extended time periods can increase denitrification losses and reduce the N concentration in solution (Roberts et al., 1996). On the other hand, increased dry matter production under irrigation leads to higher stocking rates and more N returned in urine patches, resulting in a greater potential of leaching loss from the greater urine returns. Consequently, the effect of irrigation on NO_3-N concentrations in leachate and groundwater is unclear (Roberts et al., 1996).

In arid areas, more irrigation water than is needed for evapotranspiration must be applied to the soil to avoid accumulation of salts in the root zone (Bouwer, 1987). This creates a downward flow of water from the root zone to the underlying groundwater. The resulting deep percolating water not only contains the salts that were in the irrigation water, but also fertiliser and pesticide residues that are a potential source of contamination of the groundwater. The potential for groundwater contamination under such conditions is extensive and serious. Soluble salts, nitrates, and pesticides, are the chemicals in deep percolated water that are of greatest concern. For example, extensive pollution of shallow groundwater in parts of the San Joaquin Valley has been caused by the use of the pesticides (Schmidt and Sherman, 1987).

Much attention has been paid to the effects of irrigation on increased leaching of solutes from the soil to the groundwater system and the resulting deterioration in ground water quality (Khan, 1980). Salt leaching is of most importance in arid environments, but in humid areas attention is focused on the leaching of nitrate and other soil nutrients. North American studies indicate that increases in ground water NO_3-N concentration have been related to fertiliser application (Saffigna and Keeney, 1977; Hubbard et al., 1984). In late seventies, it was not very common to apply nitrogenous fertiliser on irrigated pastures in New Zealand, which rely instead on N fixation by clovers. In this situation, leached N is mainly derived from N concentrated in urine patches from grazing animals. Quin and Burden (1979) in a study of irrigation in Mid-Canterbury, New Zealand, suggested that leaching from border strip irrigation had raised the NO_3-N concentration in ground water 30 m below from 1.6 g/m^3 in 1961 to 7.5 g/m^3 in 1976.

Close (1987) carried out a study to monitor the ground water quality in a shallow unconfined aquifer before and after border strip irrigation was implemented, to assess the effects of irrigation on ground water quality. He found that the variation of water quality with depth allowed estimation of ground water affected by percolating drainage. Nitrate, sulphate, and potassium concentrations decreased with depth, indicating that the majority of these inputs to ground water came from leaching. By contrast, sodium, calcium, bicarbonate, and chloride concentrations increased with depth, an indication that the ground water was gaining significant amounts of these chemicals from the aquifer materials.

2.3.6 Reasons for Land Application of Wastes

In addition to the phasing out of waste disposal into waterways and the ocean, the renewed interest in land application of wastes in the past 20 years is partly because of the need to conserve water and nutrient resources and to use them efficiently, and also due to the Maori cultural values that require human waste to be returned to the land rather than be allowed to pollute natural water resources.

Discharge of sewage waters, sludges, and other wastes (e.g. dredged soils, hazardous wastes) into the marine environment (rivers, lakes, and sea) is practised in many countries (UNEP, 1993). In New Zealand, about 60% of sewage is discharged to coastal waters after secondary treatment. However, the treated effluents still retain high concentrations of organic matter, suspended solids, nutrients (N and P), and other contaminants (Hauber, 1995). It is estimated that Australia produces about 100,000 tonnes of N and 10,000 tonnes of P in sewage effluent annually and much of this is discharged to coastal waters (Brodie, 1995). In low river-flow conditions, sewage effluent may be the major source of nutrients for many rivers.

The discharge of sewage and other nutrient-rich wastes to waterways can result in depletion of dissolved oxygen, eutrophication, chemical toxicity, and salinity. Eutrophication and salinity are considered to be the two major water quality problems in

Australia (Sumner and McLaughlin, 1996). Eutrophication is produced by an excess concentration of nutrients in the water leading to accelerated plant growth and changes in plant species composition. The critical nutrients responsible for eutrophication are N and P, although other nutrients, e.g. iron, molybdenum, manganese and silicon, may also contribute to the process (AEC, 1987).

Although the dynamic coastal waters of New Zealand generally disperse the discharge effectively, serious eutrophication problems are becoming more common (Cullen, 1996). For example, in January and February 1993, there were wide spread incidences of algal blooms and shellfish poisoning around New Zealand's coastlines which resulted in the temporary shut-down of the entire coastline from shellfishing, and the temporary cessation of shellfish exports. The direct economic consequences and remediation costs of eutrophication are very high; it is estimated that eutrophication may cost Australia $A10 - 50 million per year (Cullen, 1996).

In many parts of the world, land application of organic waste is not only an economic imperative but also a management necessity in order to stem the degradation and erosion of soil. In the mid-nineties, the world population was estimated to be 5.8 billion and was predicted to rise to about 10 billion by the year 2050 (Mink, 1994). It is likely that in many parts of the world the use of inorganic fertilisers alone will not ensure adequate agricultural production levels and that land application of wastes will be a necessity (Obi and Ebo, 1995). The environmental impact of twice as many people living on the earth is likely to be substantial and one of the main problems will be the safe disposal of all the extra waste that is generated. Although the nutrient content of wastes makes them attractive as fertilisers, land application of many industrial wastes and sewage is constrained by the presence of heavy metals, hazardous organic chemicals, salts, and extreme pH values (Cameron et al., 1997).

2.3.7 Information needed to Design and Manage a LTS

The design of a land treatment system should aim to balance the loading rates of water, nutrients, and toxins with the ability of the site and vegetative cover to safely convert, absorb, use or store them. This requires careful selection of site, species, and calculation of the area needed, combined with determination of acceptable hydraulic and nutrient loading rates (kg/ha/year). The hydraulic loading rate is the volume of wastewater applied per unit area of land over at least one loading cycle. Hydraulic loading rate is commonly expressed in mm/wk or m/year and is used to compute the land area required to apply the estimated volume of effluent. The hydraulic loading rate used for design is based on the more restrictive of two limiting conditions—the capacity of the soil profile to transmit water (soil permeability) or the limiting constituent concentration in the effluent applied. In municipal wastewater applied to land treatment systems, N is usually the limiting constituent when protection of potable groundwater is a concern. If percolating water/effluent enters potable groundwater, then the system should be designed such that the concentration of NO_3-N in the receiving groundwater at the project boundary does not exceed 10 mg/L (WHO, 1984). A separate case is considered for those systems in arid regions where crop revenue is important and the wastewater is used as a valuable source of irrigation water. For such systems, the design hydraulic loading rate is usually based on irrigation requirements of the crop.

The prime objective of management of land treatment systems is to apply effluent to the vegetative cover in as close as possible to optimum quantities so that the receiving water bodies are not going to be contaminated. Management means the scheduling of effluent irrigation i.e. amount and timing or frequency of irrigation. Soil characteristics as well as the vegetation’s capacity to take up the effluent will influence effluent irrigation frequency and duration. Management must avoid creating a serious risk of run-off, erosion, soil damage or unacceptable leaching. Other factors, for example, a need to avoid reductions in wood quality due to excessively rapid tree growth, where forestry is the vegetative cover, may be additional considerations.

The ability to develop acceptable land disposal systems for effluents requires an understanding of the reactions and transformations that take place when effluents are applied to the soil (Loehr, 1974). The priority pollutants in sewage effluent are N, P, and pathogenic organisms. These are transported away from the site to receiving waters (groundwater) by a combination of diffuse and preferential flow (Addiscott et al., 1991). Persistent pollutants will therefore accumulate in the system until a dynamic balance is reached between the rate of input of the pollutants to the system and the rate of discharge of the pollutants from the system (Jackson, 1980).

When a pollutant is released from a LTS, it migrates downward through the unsaturated zone to the water table, and then moves laterally in the direction of the hydraulic gradient in the saturated zone. Throughout this transport, its fate is controlled by a myriad of physical, chemical and biotic processes. These include the physical processes of advection, diffusion, dispersion, and capillarity, and the biotic and abiotic processes of bio-accumulation, degradation, immobilisation, retardation and volatilisation. Quantification of these various processes at field level is very difficult (Charbeneau et al., 1992). Assessment of subsurface pollutant fate and transport must address questions of source characterisation (what is released, where, when, how much, etc.), vadose zone transport processes, groundwater transport, and exposure and dose assessment.

2.4 Groundwater Monitoring

This section covers the need for monitoring and the parameters to consider for groundwater monitoring at a LTS.

2.4.1 Why Monitoring?

The goal or objective of groundwater monitoring is to serve as a check on potential or actual leachate contamination, and establish continuing evidence of groundwater quality (Cheremisinoff et al., 1984). Groundwater monitoring at land disposal sites is usually performed with the ostensible objective of determining how well the land disposal system is functioning and to ensure that contamination of groundwater is not occurring. One of the main environmental concerns about land treatment of effluent is the risk of groundwater contamination. The objective of land treatment of wastes is to utilise the chemical, physical, and biological properties of the soil/plant system to assimilate the waste components without adversely affecting soil quality or causing contaminants to release into water or the atmosphere (Loehr, 1984). A number of processes will affect the contaminant within the unsaturated zone before it enters the groundwater systems. Several studies have shown that underlying groundwater is being contaminated due to land disposal of effluents from sewage, industrial and agricultural wastes (Quin, 1978; Lucas and Reeves, 1981; Magesan et al., 1998; Tomer et al., 2000; Mahmood and Wall, 2001; Khan and Irvin, 2011). The extent to which this occurs depends on a range of parameters, as studied by Close (1984), which affect the applied effluent as it percolates through the soil and gravel strata and disperses in the groundwater system.

Often the monitoring and management of land treatment systems are considered to end at the soil surface where effluent is applied. However, renovation can occur throughout the landscape in surface soils, the vadose zone, the groundwater, and in nearby riparian soils. Poorly designed land disposal systems can have serious consequences for the groundwater beneath the treatment sites (Barkle et al., 1993). Research into land treatment has been focused on application to pasture. But more recently studies by the New Zealand Forest Research Institute and Massey University have shown the suitability of short rotation forest energy crops for waste treatment (Roygard et al., 1998). These crops have high dry matter productivity, and can take up considerable amounts of nutrients and transpire large volumes of wastewater. Energy benefits can accrue through the production of an environmentally sustainable and renewable energy source.

If a LTS was to always operate at or near 100% efficiency, the possibility of groundwater contamination from applied effluent would be minimal and groundwater monitoring would be of little importance. However, there are so many variable parameters controlling the performance of LTS, such as meteorological conditions, the amount and concentration of effluent applied, and the capacity of the soil to degrade contaminants, that monitoring of groundwater should be an essential part of the LTS design and management. Unfortunately, this is not always the case. In a survey of 117 land treatment facilities in USA, only 46% responded that they had some form of groundwater monitoring, 20% had no groundwater monitoring, and the remainder did not respond (Tedaldi and Loehr, 1991). In addition, most of those that did have groundwater monitoring programmes had initiated these in response to regulatory requirements. It was apparent from the studies that monitoring of groundwater was not considered a high priority for most land treatment systems at the time. In the mid to late 1970's the USA government and Environmental Protection Agency (EPA) encouraged the use of land treatment, as an alternative method for disposal of municipal waste. This lead to a discussion amongst water practitioners as to whether land treatment was a step forward or backward in the treatment of wastewater and our ability to protect groundwater quality (Wright and Rovey, 1979). A step forward because land treatment meant less wastewater being disposed of directly into surface water supplies such as rivers and lakes, or a step backward because it allowed degradation of the groundwater.

2.4.2 Parameters to Consider for Monitoring/Management of a LTS

There are number of parameters that may be relevant for the effective monitoring/management of a LTS. This may include, for example, type of effluent applied, type of contaminant in the effluent, physical & chemical properties of soil-water matrix, effect of climatic factors in flushing NO_3^- & PO_4^- ions out of the soil into groundwater below, microbial activities, etc. (Reichard et al., 1990). Further, the knowledge of groundwater conditions and how these conditions change as a result of climatic conditions and effluent application are also important for the effective management of a LTS (Rosen, 1995).The identification and prioritisation of these parameters requires a wealth of knowledge of soil and soil-plant processes, pollution and toxicity issues, and system operations. The appropriateness of these parameters for a particular land treatment situation can be defined by specific system characteristics.

Rosen (1995) stated that there are a number of characteristics to consider when designing a groundwater monitoring system for a land treatment facility. The most important of these are (a) the hydraulic properties of the aquifer, (b) the geology of the area, (c) the type of waste that is being applied, and (d) the distance of the proposed site to surface discharges (streams, lakes, etc.). A brief description of each is given below. The most important hydraulic properties of the aquifer below a land treatment facility are the depth to water table, hydraulic conductivity of the aquifer, direction of groundwater flow, and whether the area is in a groundwater discharge or recharge area.

The depth to groundwater table under a land treatment facility must permit aeration of the soil (Wright and Rovey, 1979), and generally about 1.5 m is considered sufficient. However, Taylor (1981) considered 1.2 - 3.0 m to be adequate, but greater than 3 m to be optimal. Seasonal variation in the water table must be determined, and the effect of waste loading on the position of the water table must also be considered. The hydraulic conductivity (K) of the aquifer (its ability to transmit water through the porous media), will determine how quickly contaminants will move in the aquifer and how much dilution can be expected in the aquifer. Although an aquifer with a high K may dilute a contaminant more than one with a low K, the contaminant will also move much faster and farther in the aquifer with a high K value. The direction of groundwater movement, determined from an accurate survey of the water table, is obviously important in determining where monitoring wells should be sited. If the direction of groundwater is not known, monitoring wells may be placed in the wrong positions. Whether an area under a LTS is a groundwater discharge zone or recharge area can be determined with a survey of wells constructed to different depths that will determine the vertical hydraulic gradient. If the area is a discharge zone, the water table will be high and possibly prone to frequent flooding. This type of area may not be suitable as a site for a LTS. The geology of the area under a land treatment facility will determine if the aquifer is confined or unconfined, if fracture zones should be avoided that could short circuit treatment, or if sinkhole collapse in karst areas may be a problem. Other considerations, such as depth to bedrock, may be important if the aquifer under the facility is thin.

The types of wastes being applied to a site are important, not only in determining what parameters in the groundwater at the site should be monitored, but also how the monitoring wells should be constructed (Rosen, 1995). For example, if the waste being applied contains heavy metals and the major concern to groundwater is metal contamination, monitoring wells should not be constructed using stainless steel components. Conversely, if pesticides or organic compounds such as hydrocarbons are a major component of the waste, monitoring sites should not be constructed with regular PVC material, and sampling devices should also be thoroughly checked for possible contaminating components. For normal sewage and agricultural wastes the important parameters to monitor are: chloride (Cl^-), NO_3^-, ammonia (NH_3), BOD and bacteria, but depending on the type of waste, other parameters may be necessary. It is also recommended that a yearly complete analysis of major ions in solution (with an ion balance) be performed as a quality assurance procedure. The distance from receiving surface waters such as lakes or streams is an important parameter to consider in siting a LTS. Sites that are close to surface water discharges may influence the water table and may cause short-circuiting of the waste directly into the surface water body (Rosen, 1995).

2.5 Nitrate-N Leaching Losses from Soils

Regional councils in New Zealand generally limit the N loading in effluent applied to land to 150-200 kg N/ha/year, unless special circumstances apply. The basis for an effluent loading rate of 150 kg N/ha/year as an environmentally acceptable, but agronomically practicable and sustainable activity for a grazed clover-based dairy pasture system in the Waikato has been reviewed by Selvarajah (1996). Selvarajah (1996) reported that this loading rate was obtained from a detailed budget of N inputs and N outputs based on information on N transformations in New Zealand soils. However, values given to some of the N flows may vary from the actual values. In addition, the fate of N after leaving the root zone was not considered in this budgeting approach. Consequently, if there is extensive denitrification in the unsaturated zone, or recharge of groundwater is dominated by river seepage, then acceptable N loading rates with respect to groundwater NO_3-N contamination may be markedly different. Thus, from the limited scientific evidence that is available, in general it does not seem possible to accurately differentiate between the effects of loading rates of 150 and 200 kg N/ha/year on groundwater NO_3-N contamination (Selvarajah, 1996).

Main et al. (1996) questioned whether an effluent loading rate of 200 kg N/ha/year would prevent groundwater contamination in Canterbury. Based on N flows for a high producing sheep-grazed pasture in Canterbury (Quin, 1982), leaching losses of 100 kg N/ha/year was predicted. To be under the New Zealand drinking water standard of 11.3 g/m^3, this amount of nitrate needs to be leached by at least 885 mm of drainage, but in the absence of irrigation the annual drainage is only 200 - 300 mm for the Canterbury plains. However, Quin's (1982) estimate is based on sheep grazing and the N flows in pasture receiving effluent may be quite different. In particular, denitrification losses may be higher than predicted by Quin, as the soil conditions following effluent disposal (high soil moisture content, high concentration of soil NO_3-N and soluble carbon) can encourage denitrification. Furthermore, dilution from other groundwater sources is not considered in this calculation.

Some of the more common wastes and their typical N concentrations are shown in Table 2.1 (Cameron et al., 1996a). It should be noted that organic wastes have extremely variable nutrient contents, requiring a regular monitoring programme to establish nutrient application rates.

Many studies have shown that soil fertility is increased after the land application of wastes (e.g. Keeley and Quin, 1979; Hart and Spear, 1992). Land application of wastes can produce significant increases in soil N, with the rapid mineralisation of organic N following the application of wastes. For example, up to 70% of organic N in poultry litter was mineralised within 140 days (Bitzer and Sims, 1988). In contrast, the organic N in sewage sludge may require a long time for net N mineralisation (Cameron et al., 1996b). This variation in mineralisation rate between the different types of organic wastes further complicates the definition of the optimum rate of waste application.

Table 2.1: Typical N concentrations in different types of wastes. (*Source: Painter et al. (1997), Lincoln Environmental, Draft Report No 2776/1, August 1997).*

Waste type	**N-concentrations (g/m^3)**
Farm dairy effluent	190
Dairy oxidation pond slurry	800
Piggery effluent	1300 - 1400
Poultry cage manure	14000
Milk powder/butter factory waste water	70
Dairy factory effluent	1400
Meatworks secondary effluent	40 - 200
Municipal sewage effluent	20
Municipal sludge (dewatered)	5300
Dairy pond sludge	1400 - 1600
Grass silage effluent	10000 - 40000

A number of studies have shown the beneficial effects on plant production of applying effluents to land. Increased plant production is desirable in cut-and-carry systems as it removes N and water from the waste-treated soil, thus lowering the potential for NO_3-N leaching. However, unless the cut plant material is removed from the property, such systems may merely transfer potential NO_3-N leaching losses from one paddock to another. The effects of effluent applications on plant production in the second and subsequent years after application are generally low (Cameron et al., 1996a), although the long-term effects of repeated applications are unknown. Further research is required to determine such long-term effects.

The amount of N applied in the effluent has a considerable influence on N leaching losses. When effluent is applied at a rate that matches plant N requirements, groundwater N concentrations are not adversely affected. However, excessive N application rates can pose a significant threat of groundwater contamination (Cameron et al., 1996b). Some typical NO_3-N leaching losses following the land application of a range of effluents are presented in Table 2.2.

Table 2.2: Typical NO_3-N leaching losses following the land application of a range of effluents. (*Source: Painter et al. (1997), Lincoln Environmental, Draft Report No 2776/1, August 1997).*

Effluent type	**N applied (kg N/ha/year**	**N leaching loss (kg N/ha/year)**	**Soil texture**
Farm dairy	1200	150	Silt loam
Dairy pond sludge	300	4 - 22	Silt loam
Piggery	200 600	11 - 12 116	Shallow, stony Silt loam
Sewage	480	240	Sandy
Sewage sludge	250	<211	Sandy
Sewage sludge	800-1600	9 - 44	Sandy
Animal processing	500-2000	85 - 660	Sandy loam

Leaching of NO_3-N from the soil root zone is often taken as evidence that NO_3-N contamination of groundwater will result. However, NO_3-N concentrations are also affected by the origin of the groundwater, aquifer hydrology and the geochemical processes occurring within the aquifer (Smith, 1993c). Thus, in aquifers where either recharge is dominated by river seepage rather than drainage, or denitrification is extensive in the presence of high iron concentration, groundwater NO_3-N concentrations are low. In contrast, where recharge to unconfined aquifers predominantly occurs through soil drainage and the aquifer flow rate is a low, groundwater NO_3-N concentration can be high. In addition, in certain circumstances, confined groundwater may be protected from such leaching and should contain lower NO_3-N levels. Results from Canterbury study (Smith, 1993c) show that shallow groundwater in the west has not transported its higher NO_3-N concentrations into the deeper groundwater in the east either because insufficient time has elapsed, or because the shallow groundwater in the west does not directly recharge the deeper aquifer in the east.

In some cases, either deep soil core sampling (Ledgard et al., 1996d) or soil solution samples have shown similar trends in NO_3-N concentrations under different land management practices to those observed in groundwater samples. In the study by Ledgard et al. (1996d), groundwater NO_3-N concentrations were much less than those present in the soil or soil solution. Lower NO_3-N concentrations in groundwater than in soil solution may be due to dilution of drainage water with the groundwater. Alternatively, lower groundwater NO_3-N concentrations can result from extensive denitrification losses in the unsaturated zone between the soil and the water table. The residence time of water in the unsaturated zone also needs to be considered as there is often a time lag of several months to several years between water draining from the soil root zone and its arrival in the groundwater. In addition, NO_3-N concentrations may also be reduced through denitrification in the groundwater (Korom, 1992). Higher apparent NO_3-N concentrations in the soil solution than the groundwater may also be because the soil solution samplers can remove highly concentrated water from within soil aggregates, whilst groundwater recharge consists of water that may have by-passed this resident soil NO_3-N (i.e. by macropore flow).

Overall, NO_3-N leaching losses from the soil can be used as indicators of potential groundwater NO_3-N contamination although they may not necessarily provide an exact value for the concentrations that will occur. They are useful in giving early warning signals of potentially environmentally-damaging land use practice (Ledgard et al., 1996d). However, research is needed to establish the relationship between the amounts of NO_3-N leached from soil and the amount of NO_3-N arriving in the groundwater.

The influence of flow rate on the concentration of indigenous (resident) and applied solutes in mole-pipe drainage was studied by Magesan et al. (1995). The field experiment was conducted on two mole and pipe drained plots (1250 m^2 each) on Tokomaru silt loam soil in New Zealand. Sodium bromide was applied onto one plot at 200 kg Br/ha, and NO_3-N was taken to be the indigenous solute. The urea was applied, as a solute, at 120 kg N/ha onto the other plot, and chloride was the indigenous solute. They found that the concentration of applied solute was increased with the increase of flow rate for the drainage events that occurred during the early stage of drainage period, but then the concentrations were decreased with increased flow rate because of preferential flow. There was a time lag between the peak flow rate and the highest concentration and that could be due to the time taken by the surface-applied solutes to reach the drains at 450 mm depth. However, there was no time lag between the highest flow rate and the lowest concentration. They also found that there was an inverse relationship between indigenous solute concentration and flow rate. They concluded that if the time interval between the solute application and drainage event is longer then it is difficult to distinction between an applied and resident solute.

2.5.1 Factors Affecting Nitrate Leaching Losses

A wide range of factors affect NO_3-N leaching losses from fertilisers and soils. These have been reviewed in detail by Cameron and Haynes (1986) and Juergens-Gschwind (1989) and can be summarised as follows:

Crops: The amount of leaching loss from fertiliser depends on the rate of application relative to the plant requirement. The 'safe rate' to apply depends on the crop, soil and climatic conditions. Research in Europe has indicated that for arable crops application above 160 kg N/ha/year can result in increased leaching losses, however it is not known if this applies to New Zealand conditions. Cereals generally use fertiliser N efficiently and leaching losses are therefore small at optimum fertiliser rates (Prins et al., 1988). In New Zealand, Mohammad et al. (1986) recorded a short term increase in NO_3-N concentration in tile drain effluent when urea was applied to barley. Adams and Pattinson (1985) concluded that fertiliser applied at between 25 and 50 kg N/ha had little effect on NO_3-N leaching from wheat grown on Wakanui soil in Canterbury. Fertiliser N recovery by field vegetables is generally low (e.g. lettuce 11%; spinach 31%; red beet 52%; potatoes 50%) due to their shallow rooting depth and the high rates of irrigation often used (Prins et al., 1988).

Pasture: Extensive pasture systems and pasture cut for hay or silage generally have low leaching losses (Cameron and Haynes, 1986). Relatively large leaching losses can however occur under intensively managed pasture, particularly when high stocking rates are combined with high fertiliser inputs. Animal urine patches contain large amounts of N (the equivalent of up to 500 kg N/ha for sheep and 1000 kg N/ha for cattle - Steel, 1982). These amounts are greater than the plant can assimilate and therefore significant leaching losses can occur. Lysimeter studies in New Zealand have found that between 8 and 20% of urine-N may be leached (Field et al., 1985b; Frase et al., 1994). There are very few measurements of fertiliser leaching losses from grazed pasture in New Zealand. Sharpley and Syers (1979) reported that the application of urea at 60 kg N/ha/year resulted in a temporary increase in the concentration of NO_3-N in drainage water from a Tokomaru silt loam at Palmerston North. Field et al. (1985a) reported large leaching losses from sheep grazed pasture in the Manawatu region.

Soil properties: Sandy soils usually retain less water, and leaching is therefore more rapid and extensive on lighter soils. Split applications of fertiliser on light soils reduce leaching loss. Denitrification, immobilisation and ammonium fixation can reduce the loss whilst mineralisation can increase the loss. Nitrate-N leaching following mineralisation of ploughed pasture and clover residues can be a major source of NO_3-N contamination of groundwater (Cameron and Wild, 1984; Francis et al., 1992).

The rate of water and solute moving through the soil-water matrix depends on many factors e.g. soil type, climatic conditions, soil conditions, and amounts and types of agricultural chemicals and nutrients applied. Macropores (i.e. cracks and biologically created channels in the soil-water matrix) can induce preferential flow paths into the soils systems. These macropores only make up a small proportion of the total pore space. However, these macropores can allow the bulk of water and solute movement through soil when the soils' water flow conditions are near saturation (Clothier and White, 1981).

Plant uptake: As it is well known that plant uptake is a primary factor affecting the amount of NO_3-N leached. The greater the uptake, the lower the concentration of NO_3-N in the soil solution, and thus the lower the potential for leaching.

Irrigation: Irrigation applied at optimum rates for plant growth can enhance N uptake and reduce NO_3-N leaching; however excessive irrigation rates can increase the loss of NO_3-N. Irrigation of pasture permits an increased stocking rate and thus there is a potential for a higher leaching loss due to greater urine returns. Irrigation may also increase denitrification losses and thus reduce the concentration of NO_3-N in solution. The influence of irrigation on groundwater contamination in New Zealand is unclear with conflicting opinions (e.g. Quin and Burden, 1979; Burden, 1980; Close, 1987) due to different irrigation and soil conditions.

Organic wastes: Application of organic wastes (e.g. piggery waste) at rates that can be utilised by a pasture or crop (say 200 kg N/ha/year) appears to pose no greater threat to groundwater quality than many other agricultural practices (e.g. ploughing pasture) (Cameron and Rate, 1992). Application at higher rates results in larger leaching losses and higher NO_3-N concentrations in drainage water (Cameron and Rate, 1992).

Weather and season: In New Zealand, most N leaching occurs from late autumn to early spring when plant N uptake is low and rainfall exceeds evaporation. However, leaching can also occur at other times of the year if the soil is wet and heavy rainfall or irrigation is received. The pattern of the rainfall over autumn/winter has a major impact on the extent of leaching losses, especially from cultivated soils. Thus, for about the same total amount of winter drainage, leaching losses can be twice as great when drainage occurs late in winter rather than earlier in winter (Francis et al., 1998). This difference is due to the greater accumulation of mineral N in the soil profile later in the winter, following a longer period for net N mineralisation of soil organic N and plant residues.

The amount of drainage in the spring can also be important, and it can determine the amount of leaching from recently applied fertilisers, animal returns and mineralised NO_3-N (Roberts et al., 1996). Rainfall intensity and amount are both important in determining leaching losses. Heavy rainfall may induce macropore flow and a proportion of NO_3-N in the soil may be rapidly leached to depth (Cameron et al., 1995).

However, studies have also shown that intermittent rainfall can be more effective than continuous rainfall at leaching NO_3-N from soil due to 'holdback' of NO_3-N in soil macropores in response to slow diffusion rates (e.g. Francis et al., 1988). A dry summer can cause an additional accumulation of NO_3-N in the soil profile due to poor crop uptake and this may result in enhanced leaching losses over the following winter.

2.6 Nitrogen Cycling and the Use of Nitrification Inhibitors

Research on the cycling of N in forests and grazed pasture systems has gained impetus in recent years as environmental concerns have broadened to include not only the deleterious effect of leached NO_3-N on the quality of surface and sub-surface water, but also the important contribution of various oxides of N to New Zealand's greenhouse gas inventory. A number of authors have studied transformations of soil N following effluent application (Bhandral et al., 2007; Linsley et al., 2007). In a LTS the large quantities of added water, C and N should theoretically provide ideal conditions for denitrification but many authors have concluded that denitrification losses from a LTS are usually small (Snow et al., 1999b; Meding et al., 2001; Magesan and Wang, 2003). Even inserting layers of organic matter as a "denitrification layer" below the topsoil of a site irrigated with dairy factory effluent failed to generate large losses of N through denitrification (Schipper and McGill, 2008).

In an interesting recent development, measurements of the natural abundance of N-15 have been used to identify the source of leached N from an LTS. Using this technique Tozer et al. (2005) confirmed that the majority (88%) of the NO_3-N in the stream draining Whakarewarewa Forest LTS was directly attributable to effluent N.

In New Zealand the concern about leaching and gaseous losses of N from dairy farms has prompted a large research programme on the potential for nitrification inhibitors to mitigate these environmental impacts. One such nitrification inhibitor is dicyandiamide (DCD) which inhibits the first stage of nitrification, the oxidation of NH_4^+ to NO_2^- (Amberger, 1989). Dicyandiamide has been used in the past to increase the efficiency of

N supply from fertilizers or manures with variable results (e.g. Davies and Williams, 1995).

In New Zealand the major focus has been on reducing the leaching of NO_3-N from urine patches in grazed dairy pastures. In these urine patches the effective N application rate can be as high as 1000 kg N/ha and therefore many workers (Blard et al., 2006; Di and Cameron, 2002, 2004 and 2005; Di et al., 2009; Meneer et al., 2008a and 2008b; Singh et al., 2005) have studied the effect of DCD on N leaching from simulated urine patches containing very high rates (598 – 1000 kg N/ha) of added urine N. Williamson et al. (1998) studied the effect of DCD on NO_3-N leaching from FDE (rather than urine), but again the rate of application (1100 kg N/ha) was very high - much higher than would normally be considered for an LTS.

Most of these studies were conducted in lysimeters, or similar controlled environments, and most reported reductions in the amounts of NO_3-N leached and some of these reductions were large. For example, Di and Cameron (2004) reported a decrease in leaching of NO_3-N from 85 to 20-22 kg N/ha/year following treatment with DCD. Other workers however, reported somewhat smaller decreases in NO_3-N leaching following addition of DCD. Meneer et al. (2008b) reported 50% reductions in NO_3-N leaching during the early stages (30-55 days) of a trial investigating the effect of DCD on NO_3-N leaching from an autumn urine application, but from day 84 greater quantities of NO_3-N were leached from the +DCD treatment. At the end of the trial the quantities of NO_3-N leached from the +DCD treatment were still less than from the –DCD treatment but the difference was not statistically significant.

Singh et al. (2009) also reported reduced potential for NO_3-N leaching from urine applied to intact soil cores as a result of adding DCD, but at somewhat lower rates (144 – 570 kg N/ha) than those used by most other workers, and Monaghan et al. (2009) demonstrated that DCD can reduce NO_3-N leaching from dairy pastures on a field scale, where the urine-effected areas occupy only a small percentage of the paddock.

During the early experimental work (Di and Cameron, 2002, 2004) the DCD was usually fully dissolved in water prior to application. In later work however, Di and Cameron (2005) demonstrated that DCD applied in the form of a fine particle suspension (FPS) specifically developed to enable the DCD to be applied by commercial spray contractors using small volumes of water (e.g. 10 kg/ha in 100 L of water) was still highly effective in reducing the leaching of NO_3-N from grazed pasture soils.

Nitrate leaching from a drained, sheep grazed pasture was also studied by Magesan et al. (1996). They reported that the amount of NO_3-N leached could be reduced by reducing the soil solution NO_3-N concentrations and this could be achieved by reducing the fertiliser rates. The study showed that during spring there was no substantial increase in the rate of NO_3-N leaching from an application of 50 kg urea-N/ha. The leachate NO_3-N concentration increased and exceeded the 10 mg/L limit during early drainage samples from Tokomaru silt loam soil. It was suggested that diversion and treatment of early drainage water could possibly reduce the contamination of surface and subsurface waters. The study also showed that only 8% of applied 120 kg urea-N/ha was leached during the 1990 drainage season and about 44 kg/ha of NO_3-N leached from unfertilised area. This clearly shows that much of the N leached was coming from the mineralisation of the soil organic N. The cumulative drainage volume was more in 1991 than 1990 but the amount of NO_3-N leached in 1991 was almost half of that leached in 1990. This was due to the lesser amount of NO_3-N present in the soil at the start of 1991 drainage season, which was due to slower mineralisation during the wetter summer and autumn periods.

It is interesting that there seems to have been little work on the potential for nitrification inhibitors to reduce NO_3-N leaching from effluent applied to a LTS, where the N is applied evenly across the whole area at much lower rates than are found in urine patches from dairy cows. There has been work in Europe on the effect of DCD on the leaching of NO_3-N from animal manures (e.g. Wadman and Neetson, 1992) and as noted above, Williamson et al. (1998) did investigate the effect of DCD on the leaching of NO_3-N from FDE in a lysimeter study - but the effective rate of application was very high (1100 kg N/ha). In an earlier glasshouse study Williamson et al. (1996) did also demonstrate the effectiveness of DCD in reducing the leaching of NO_3-N from mixtures of FDE and urea applied at more typical rates (160 kg N/ha).

In New Zealand, nearly all the work on nitrification inhibitors such as DCD has been conducted in the context of commercial dairy farming operations. This has restricted the timing and frequency of DCD applications being investigated to those that could reasonably be afforded by a dairy farmer. There might be less financial constraint on the operators of an LTS, if the regular application of DCD could be demonstrated to reduce the leaching of NO_3-N. This area merits further study.

2.7 Dissolved Organic Nitrogen (DON)

Dissolved organic N can be a significant contributor to N leaching from agricultural ecosystems (Campbell et al., 2000; Siemens and Kaupenjohann, 2002; Murphy et al., 2000. However, little research has been undertaken to assess the impact of agricultural land use systems on the concentration of DON in soil (Christou et al., 2005). Perhaps because of this reason, most nutrient budget studies for agricultural systems have not either considered or measured DON (van Kessel et al., 2009; Ghani et al., 2007), and most of the NO_3-N leaching models do not contain a sub-model to simulate leaching of DON (Korsaeth et al., 2003). Another reason for the little attention given to leaching losses of DON could be that when organic N is applied onto land it is converted to NO_3-N following mineralisation and subsequent nitrification. This means that NO_3-N is the dominant form of soluble N in the system. Because of its high solubility and very mobile nature, it has become a common understanding that most of the N leaching will occur as NO_3-N.

However, some studies showed that DON losses from forest based land treatment systems could be important and could pose a threat to the health of the local environment (e.g. Campbell et al., 2000; Perakis and Hedin, 2002).

Christou et al. (2005) undertook a study to quantify the relative amounts of dissolved organic C (DOC), DON and soluble inorganic N for different agricultural land use systems at sites throughout England, Wales and Greece. Seven different land use systems (intensive citrus orchards, vegetable production, arable, monoculture forestry, high and low intensive grassland of mixed species, low intensive wetland, and low intensive grazed land) were included in the study. They found that DON was a significant proportion of total dissolved N in all agriculture ecosystems but its concentration was less sensitive to land use system than dissolved inorganic N. The dissolved inorganic N varied between land use with intensive agricultural systems being dominated by NO_3-N and low input systems dominated by NH_4-N. Willett et al. (2004) also reported that inorganic N can be more responsive to biotic factors than DON.

Nitrogen losses from agricultural systems were also reviewed by van Kessel et al. (2009). The focus of this review was to examine the impact of DON losses from agricultural systems and its impact on human and environment. This review was performed using the ISI-Web of Science research database and only focused on losses of DON from agricultural systems. They (van Kessel et al., 2009) found that losses of DON from agricultural systems varied between 0.3 kg DON/ha/year in a pasture to a maximum loss of 127 kg DON/ha/year in a grassland following the application of urine. They concluded that the losses of DON increased with increasing of rainfall, irrigation, N inputs and sand contents. They recommended that the models used to simulate N losses from agricultural systems should be able to simulate DON losses as it could be an important N loss from the system. Further details of concentrations of DON and NO_3-N in leachates collected from different agricultural systems with various rates of N inputs can be found in van Kessel et al. (2009).

The loss of DON from forest ecosystems has also been reviewed elsewhere (e.g. Neff et al., 2002, 2003; Cooper et al., 2007). The contribution of DON to N leaching from four German soils was also studied by Siemens and Kaupenjohann (2002). They monitored the soil solution concentration and fluxes of DON and mineral N at two cropped sites (Plaggic Anthrosols) and two fallow sites (a Plaggic Anthrosol and a Gleyic Podzol) from November 1999 till May 2001. The median DON concentration of soil solution samples varied between 0.4 and 2.3 mg/L for cropped sites and between 0.6 and 1.8 mg/L for fallow sites. The DON concentrations found in this study were higher than those in subsoil horizons of temperate forests (0.08-0.8 mg DON/L; Michalzik et al., 2001). It is likely that the higher concentrations of DON reported in this study (Siemens and Kaupenjohann, 2002) were because Plaggic Anthrosols are characterised by high contents of soil organic matter and N. However, Siemen and Kaupenjohann (2002) concluded that DON contributes significantly to N leaching from arable soils. The DON concentrations varied less over the study period than NO_3-N concentrations. They suggested that soil N budget studies should include leaching of DON, and also there is a need to develop an analytical tool for the direct measurement of DON.

A similar observation was reported by Murphy et al. (2000). They reviewed the methods of extraction, the pool size and the functions of both soluble organic N extracted from soils and DON present in soil solution and drainage water in arable agricultural soils. They found that soluble organic N was of the same order as of magnitude as mineral N and of equal pool sizes in many cases. The soluble organic N was present in a wide range of arable agricultural soils from England and the quantities varied between 20 and 30 kg/ha. Mineralisation, immobilisation, leaching and plant uptake affected the dynamics of DON in the same way as those of mineral N. However, the pool size of DON was more constant than that of mineral N. Murphy et al. (2000) also reported that a significant amount of DON was leached but this comprised only about 10% of the soluble organic N extracted from the same soil. The leached DON may take with it nutrients, complex metals and pesticides. They mentioned that soluble organic N extracted from soils and DON present in the soil solution and leachate are clearly an important pool in N transformation and plant uptake, and therefore more work is needed to enhance our understanding of the processes.

2.8 The Development and Application of Water and Solute Transport Models

2.8.1 Background

As reported by Hutson and Wagenet (1992) that "*mathematical modelling is an accepted scientific practice, providing a mechanism for comprehensively integrating basic processes and describing a system beyond what can be accomplished using subjective human judgements. As our understanding of basic processes deepens, it is possible to construct models that better represent the natural system and to use these models in an objective manner to guide both our future research efforts and current management practices".*

Different approaches have been used to describe the movement of water and solute movement in field soils. Few models have been developed to study the solute leaching patterns. The models that have been developed so far vary widely in their conceptual approach and degree of complexity, and are strongly influenced by the environment, training, and biases of their developers. Many of the models have been produced as the result of research into the basic physics and chemistry of salt, N or pesticide transport and transformation in agricultural soils (Hutson and Wagenet, 1992; Vanclooster et al., 1995; Kroses and Roelsma, 1998; Abrahasen and Hansen, 2000; Leaonard et al., 1987; Eckersten et al., 1994; Close et al., 2003, 2005; Nolan et al., 2005).

Over the past four decade, the development and use of simulation models for predicting nutrient and pesticide behaviour in the root zone of agricultural land systems, and in the underlying unsaturated zone, has received considerable attention. The models currently available for predicting the fate and transpiration of pesticides and nutrients in soils and groundwater have been critically reviewed by, among others, (Addiscott and Wagenet, 1985; Donigian and Rao, 1990; Wagenet and Rao, 1990; Cichota and Snow, 2009). There has been a considerable debate over the appropriate conceptual representation of environmental processes used in nutrients and/or pesticides simulation models and as well as the formulation of key criteria for evaluation and comparing the ability of models to predict observed pesticide or nutrients behaviour. Although considerable efforts have put in to develop nutrient simulation models (Wagenet and Rao, 1990), comparatively little work has been done to validate these models using independent data sets. This is primarily due to the fact that considerable physical and financial resources are required to conduct field studies. In addition, many of the field studies conducted to date lack adequate measurement of input parameters required to run the model, and field data required evaluating the models. As highlighted earlier, the need for efficient use of agricultural chemicals and their potential adverse impact on critical water resources have increased the use of simulation models of the soil and plant system. Nevertheless, there is currently little or no agreement concerning model validity and applicability in varied soils and environments (Sogbedji et al., 2001).

Hutson and Wagenet (1992) reported that *"models have not been used for management purposes in the past. A major reason is that there is apparently little recognition in most model development efforts that different types of models are developed for different purposes. The quantity of required input data, depth of consideration of basic processes, and sensitivity and accuracy of simulations all depend upon whether the modeller intends to approach the simulation from a research or management perspective. A short discussion or review on the range of modelling experiences on water and solute movement in soil may establish the logic behind the development of a model*. A review of modelling approaches (Addiscott and Wagenet, 1985; Cichota and Snow, 2009) identified a number of research and management models that have been reported in the scientific literature. The research models were identified as generally intended to provide quantitative estimates of water and solute movement, but with comprehensive data demands regarding the system to be simulated. Additionally, few of either type of model have been tested against field data, and little attention has been paid to the use of the so-called management models for the actual purposes of managing applications to soil of salty irrigation water, effluents, fertiliser, amendments, pesticides or other solute. Appreciation of the strengths and weaknesses of these models is a necessary preliminary step before proceeding to use a model.

Deterministic, mechanistic models of solute movement based upon miscible displacement theory (Nielsen and Biggar, 1962) have been the most widely used modeling approach in soil science over the past 50 years. Hutson and Wagenet (1992) mentioned that *"these models presume that soil-water and solute displacement processes operate so that the occurrence of a given set of physical and chemical events leads to a uniquely definable water or solute distribution in the soil profile. Water flow is assumed to be describable as the product of hydraulic gradient and water content dependent hydraulic conductivity. It is assumed that physical convection (mass flow) and chemical diffusion combine to displace a solute in porous media".* This type of solute model is summarised in the Convection-Dispersion Equation (CDE), which has been derived in detail in Kirkham and Powers (1972), and solved analytically for a variety of particular initial and boundary conditions (reviewed by van Genuchten and Alves, 1982). These analytical solutions, representing particular research models of water and solute movement have been successfully applied to the results of laboratory scale studies in which water flow is a constant (i.e. steady state rate). Based on these studies, the CDE is a well-established solute modeling approach for such cases (Hutson and Wagenet, 1992).

The movement of both water and solute under field conditions varies with depth and time; require numerical (rather than analytical) solution of the CDE to properly represent the influence of changing water contents, fluxes and solute concentrations. Further, the Richard's equation (i.e. the transient water flow equation) must be numerically solved to describe the water flow regime. Research models that have been constructed based on Richard's equation are numerous, including those that simulate plant growth in response to transient water regimes (Nimah and Hanks, 1973), nitrogen movement and transformation (Tillotson and Wagenet, 1982), management of irrigation water and salts of wastewater (Iskander and Selim, 1981). The LEACHM model is similar in structure to these models, and has evolved from modeling experiences of the last two decades. It has already been successfully used to describe pesticide movement in field soils (Wagenet et al., 1989; Close et al., 1999) and is used by many research groups in New Zealand, United States and other countries (Hutson and Wagenet, 1992;

Pearson et al., 1996; Mahmood et al., 2002a, 2002b; Mahmood, 2005; Close et al., 2003, 2005).

As we know that it is impractical to directly measure the nutrient losses to the environment and therefore simulation models could be the best alternative to asses the potential loss of nutrients to the local environment. However, there are a number of models that have been used in New Zealand to simulate nutrient loss from different sources to the natural environment. Cichota and Snow (2009) presented an overview of some models that are relevant to New Zealand pastoral farms. They reviewed the models for nutrient loss estimation that are used in New Zealand at farm scale (including OVERSEER®, NPLAS - Nitrogen and Phosphorus Load Assessment System, SPASMO - Soil-Plant-Atmosphere System Model, EcoMod, LUCI - Land Use Change and Intensification, and APSIM - Agricultural Production Systems Simulator) and at large scales i.e. catchment level (including NLE - Nitrogen Leaching Estimation, SPARROW - Spatial Referenced Regression On Watershed attributes), ROTAN - ROtorua and TAupo Nitrogen, CLUES - Catchment Land Use and Environmental Sustainability, and AquiferSim).

Cichota and Snow (2009) also reviewed some soil process models (e.g. GLEAMS (Groundwater Loading Effects of Agriculture Systems), LEACHM, and HYDRUS). In their review, they mentioned that nutrient management models are being extensively used in New Zealand. They said that most of the models have shown their usefulness to estimate nutrient losses in order to prevent environmental impacts at a small or larger scale levels. However, there is a lack of information about how these models work and what is their main focus, and therefore, it is important to know the main purpose, strengths, and weaknesses of a model so that the most appropriate model could be selected.

2.8.2 Application of Water and Solute Transport Models

This section mainly covers the application of N version of LEACHM and others water and solute transport models.

Application of LEACHN: Long term impacts of alternative citrus N and water management practices implemented at grower cooperative sites on the Central Florida Ridge were studied by Harrison et al. (1999) using the LEACHN model. A bromide tracer test was conducted to calibrate and validate the model. They mentioned that to model field scale solute transport in the unsaturated zone is an ambitious goal – a statement also supported by Pennell et al. (1990). However, Harrison et al. (1999) suggested that accurate predictions of field scale NO_3-N concentration distribution may be unrealistic due to high variability of soil, climate, and source characteristics affecting solute transport. The study showed that reducing the application rate and increasing the frequency of N application, and improving irrigation management can increase the ability of the plant to take up N and reduce groundwater NO_3-N concentration below the EPA's limit of 10 mg/L.

Harrison et al. (1999) reported that the validity of LEACHN predictions is highly dependent on the rate of N uptake by the plant in the model. In this study (Harrison et al., 1999), the potential N plant uptake was calibrated so that the modelled crop would remove 200 to 240 kg N/ha/year. The LEACHN model does not simulate crop growth. The plant uptake of N is constant and is not adjusted for root growth, shoot growth, and tree development. Due to this and other model limitations (as reported by Hutson and Wagenet, 1992), they suggested putting more confidence on the relative effectiveness of best management practices rather than the precise NO_3-N concentrations as predicted by the model.

Sogbedji et al. (2001) conducted a study to calibrate the LEACHN (nitrogen version of LEACHM) model for nitrification, denitrification, and volatilization rate constants using measured data from a 3-year field study involving sod plow down followed by maize production under three N fertilizer rates on clay loam and loamy sand soils. In this study (Sogbedji et al., 2001), LEACHN was calibrated for each N treatment-year combination and soil type. The results showed that model satisfactorily predicted growing season soil profile NO_3-N distributions. Slight discrepancies occurred however between measured and predicted data, primarily as a result of the model's incapacity to accurately simulate maize N uptake and the high initial soil NO_3-N of the experimental sites due to sod plow down. The results also showed that the N application rates (22, 100, and 134 kg N /ha) used in this study minimally affected the calibrated N transformation rates, while cropping history and soil type have greater effects on the N transformation rates. This implies that single N transformation rate constants can be applied to estimate N fate and transport within a given soil type and cropping practice. However, rate constants need to be adjusted for different soil types and when crop conversions occur.

Jabro et al. (1993) used LEACHM and NCSWAP (Nitrogen and Carbon transformation in Soil-Water-Air-Plant systems) models to simulate NO_3-N leaching and compared them with the field data collected from a 3-year NO_3-N leaching experiment in central Pennsylvania on Hagerstown silt loam soil. Despite calibration of model parameters to site specific soil conditions, both models generally did not predict accurately NO_3-N leaching below the 1.2 m depth for most of the treatments for the validation periods. However, the calibration results for both models were reasonably accurate.

Soulsby and Reynolds (1992) used LEACHM to model soil water flux for an Aluminium leaching study. After calibrating the model (i.e. optimising parameters such as saturated hydraulic conductivity, K_{sat}) using in situ tensiometer data, they compared model predictions against measured tensiometer data for the remainder of the year. Despite good agreement between simulated and measured matrix potentials throughout their calibration period, LEACHM predicted greater summer drying than actually occurred and required further optimisation of K_{sat}. In all of these studies, poor model performance was thought to result in part from the inability of LEACHM to simulate preferential flow.

Application of other Water and Solute Models: The N movement in the soil-water-plant system is a very dynamic and complex process and it varies in space and time within the system. Mathematical models are required as a DSS to predict the effects of different management strategies on water quality. There are many existing models. Examples that have been used to simulate NO_3-N leaching from agricultural soils include WAVE - Water and Agrochemicals in the soil, crop and Vadose Environment (Vanclooster et al., 1995), Agricultural Nutrient Model - ANIMO (Kroses and Roelsma, 1998), Danish Simulation model DAISY (Abrahamsen and Hansen, 2000), GLEAMS (Leonard et al., 1987), SOIL-N (Eckersten et al., 1994), and LEACHN (Hutson and Wagenet, 1992).

Wallis et al. (2011) conducted a study to simulate NO_3-N leaching under potato crops in the Mediterranean area (Majorca, Spain). They focused on the importance of frost prevention irrigation management strategy, its effect on the transport of N below the plant root zone and its impact on groundwater quality. They used the GLEAMS model, which is a one dimensional model, deterministic and physically based numerical model. This model can be used to simulate ET, runoff, erosion, drainage and N and pesticide leaching on a daily time step. However, it has commonly been used to assess the effects of management strategies on NO_3-N leaching at field scale level. The model was calibrated using visual inspection and statistical measures as used by (Loague and Green, 1991; Mahmood et al., 2002a). A systematic sensitivity analyses was also performed by increasing or decreasing the parameters by 20% while keeping all other input parameters constant. They found that GLEAMS was able to predict water content at different depths and leachate NO_3-N concentrations below 900 mm with reasonably accuracy, providing a good match between the measured and predicted values. The RMSE values for measured and predicted leachate volume ranged between 0.13 and 0.25 for the calibration and validation periods, respectively. The statistical results for chemical transport comparing leachate NO_3-N concentrations gave a correlation of 0.82 and 0.60 for calibration and validation, respectively. They concluded that the additional water that was applied to prevent frost caused an increase in N loading to the local groundwater.

It is a general knowledge that different soil-plant systems show different leaching patters due to differences in the soil physical, chemical and biological behaviour and fertiliser history. Ducheyne et al. (2001) conducted a study to examine the potential impact of fertiliser measures taken by the Flemish government, Belgium. They assessed parameters of a mechanistic model (WAVE) using historic data from experimental field sites in Belgium. The model was calibrated and validated before being used to analyse different scenarios to examine the factors affecting the amount of NO_3-N leached at the depth of the bottom of root zone. The study showed that the amount of NO_3-N leached at the bottom of the root zone is not only controlled by the fertiliser practice of the crop during that year, but also by the rainfall depth and pattern, the soil texture, the soil mineralisation capacity, and the past fertilisation practice.

Vogeler et al. (2004) conducted a field experiment close to Rotorua, New Zealand, to quantify the water dynamics from an effluent irrigated plantation of *Pinus Radiata*. There were two types soils used in the experiment - the native volcanic soil and coastal dune sand. The secondary and tertiary treated effluent or fresh water was applied at three different rates (0, 30 and 60 mm/week). The annual average rainfall at Rotorua is 1335 mm/year and the reference ET is about 880 mm/year. Therefore, it was likely that large drainage losses would occur through the soil profile. Vogeler et al. (2004) found that increasing effluent irrigation increased the amount of drainage under all treatments. In the first years, drainage occurred even during the summer, which means that the application was greater than the demand of the growing trees. However, during the third year no drainage was observed in the summer period, even when the application rate was 30 mm/week to the ash soil. This indicates that irrigation was beginning to match to tree's water demand or in other words tree water demand had increased to the stage that it matched the irrigation application rate. They used the SPASMO model to predict the partitioning of rainfall and irrigation water to plant uptake and drainage beyond the root-zone of the pine trees. The predicted and measured cumulative drainage values were in good agreement for the different treatments. They suggested that SPASMO could be used to estimate the water balance under effluent irrigated land treatment systems planted with pine trees and therefore, SPASMO model could be used as a decision support tool for designing the effluent irrigated land treatment systems.

The Denitrification-Decomposition model is one of the biogeochemical models that include a relatively complete suite of N transformation processes under both aerobic and anaerobic conditions. Li et al. (2006) attempted to improve this model for estimating NO_3-N leaching for crop fields with tile drainage systems. They modified the model by using the observations from nine drainage tiles with three different fertilizer treatments in four years. Initial comparisons of the observed and predicted discharge flow indicated that the original Denitrification-Decomposition model lacked the water leaching recession character. They added a new water retention feature to the model by adopting a simplified recession curve to regulate the gravity drainage flow in the simulated soil profile (i.e. 0 - 500 mm), and by introducing a virtual water pool for the specie between the bottom of the modelled soil profile and the tile lines depth (i.e. 1450 mm) in order to control the tile drainage flow. Li et al. (2006) defined the recession curve as *"…drainage rate reaches its maximum when the soil is saturated during a rainfall event, and gradually decreases as the depth of saturation decreases when there is little or no precipitation".* A simplified recession curve that was implemented in Denitrification-Decomposition model to describe water discharge during and after rainfall events for each layer can be found in Li et al. (2006).

This modification improved the model predictions of water leaching fluxes from the tile drainage lines. They also created an adsorbed N pool in the model to simulate the buffering effects of soil on the amount of NO_3-N available for leaching. Li et al. (2006) reported that "*this soil buffering effect can be constituted by several mechanisms including N assimilation/dissimilation by the soil micro-organisms and NH_4^+ adsorption/desorption by the soil adsorbents (e.g. clay minerals or organic matter)"*. They (Li et al., 2006) highlighted that creation of N pool in this model increased its ability to simulate free ammonium dynamics, nitrification and NO_3-N leaching. They also reported that this model (Denitrification-Decomposition model) could be used to predict NO_3-N leaching (with some modifications in the hydrology section) and also could be a useful tool for sustainable management of cropping systems.

Nolan et al. (2005) tested seven solute transport models using two data sets in order to select models for use by the Agricultural Chemical Team of the US Geological Survey's National Water Quality Assessment Programme. In this evaluation, they used both complex and simple models based on their water flux algorithm. The complex models (e.g. HYDRUS2D, LEACHP, RZWQM – Root Zone Water Quality Model, and VS2DT – Variable Saturated 2 Dimensional Transport) use Richard's equation to simulate water flux and are suitable to understand the field processes. The simple models (e.g. CALF – CALculates Flow, GLEAMS and PRZM) use a tipping-bucket algorithm and required less data to run. In this study the models were run without calibration to predict the transport of bromide and the transport and fate of atrazine, and three of its transformation products. They reported that RZWQM, HYDRUS2D, and VS2DT were easy to use and had quite user friendly graphical interfaces and extensive documentation. RZWQM could simulate water and transport in macropores, which is a potentially important feature of structured soil sites. HYDRUS2D and VS2DT could simulate water and solute flux in two dimensions, which may be required for sites that are closer to streams or drains. Nolen et al. (2005) also reported that LEACHP does not have the graphical interface as compared to the other models used in this study. The Richard's equation version does not have a preferential flow component. Further detail of the models can be found in the USGS evaluation report (Nolan et al., 2005).

The Danish Simulation model DAISY (one dimensional mechanistic, deterministic model for the soil plant system) was used by Djurhuus et al. (1999) to evaluate the use of effective hydraulic parameters for coarse sandy and sandy loam soils. They used different approaches (i.e. the geometric mean, arithmetic mean, estimated arithmetic mean from lognormal distribution, and mean estimated from a stochastic large scale model for water flow – which is similar to one dimensional Richard's equation) to estimate the effective means of hydraulic conductivity for simulation of NO_3-N leaching at field scale. The mean value of field data from 57 soil columns was used. The predicted NO_3-N concentrations were compared with the measured NO_3-N concentrations from 57 ceramic cups (installed at 250 mm and 800 mm depths) for the winter periods of 1989 and 1990. The study showed that the NO_3-N concentrations, at both depths, simulated by the geometric mean of hydraulic conductivity, the stochastic approach and the mean of 57 simulations matched the measured NO_3-N concentrations. However, the other approaches did not give reliable results for the coarse sand. They concluded that the geometric mean approach could be used at field scale level to simplify the calculations.

Gärdenäs, et al. (2005) presented a two dimensional modelling approach to improve fertigation practices. Fertigation allows the controlled placement of nutrients near the plant roots, which reduces fertiliser's leaching losses into the groundwater. Four different micro-irrigation systems (i.e. drip tape, subsurface drip tape, surface drip emitter, and micro-sprinkler) were used in this study to examine the effect of fertigation management and soil type on NO_3-N leaching. Four different soil types (sandy loam, loam, silty clay, and anisotropic silty clay) and five fertigation strategies were used for the purpose. HYDRUS2D was used to model the water flow and fertigation scenarios. The study showed that fertigation in the beginning of the irrigation cycle increased seasonal NO_3-N leaching losses for coarse textured soils. However, fertigation at the end of the irrigation cycle potentially reduced NO_3-N leaching losses. The seasonal leaching was lowest for subsurface drip tape and highest for the surface tape system. They concluded that leaching potential increases as the differences between the extent of the wetted soil volume and rooting zone increases. For fine textured soils, the lateral spreading of water and NO_3-N was increased by surface ponding i.e. causing the water to spread across the surface with subsequent infiltration downward and also causing the horizontal spreading of soil NO_3-N near the soil surface.

It is also well known that pastoral ecosystems with grazing animals could be a major contributor to the degradation of water quality in rivers, streams and lakes. There are others factors such as soil type, climate, type and number of animals, and application of N fertiliser that could be responsible for the movement of N through the soil-water matrix into the local surface and subsurface water bodies. Field scale experiments may not be sufficient to explore the effects of these factors, and use of models could be the way forward. Bryant et al. (2011) used an agro-systems model (EcoMod – Johnson et al., 2008) to examine the impact of situational variability in climate, soil, animal type, and N fertilisation on N leaching from pastoral farming systems in the Lake Taupo area of New Zealand. In their simulations, they used different combinations of soil types (Oruanui and Waipahihi), climate (low, medium and high rainfall), animal types (sheep, beef, and dairy), and N fertilisation rates of 0 and 60 kg N/ha/year for sheep and beef systems, and 50-200 kg N/ha/year for dairying systems. The study showed that the timing and amount of rainfall had an impact on N leaching. For the two soil types, N applied as fertiliser increased N leaching. This could be due to either increased N return to the soil in dung and urine from grazing animals as a consequence of greater N uptake by plants or due to the direct leaching of fertiliser N through the soil-water matrix. Soil type had a considerable impact on the amount and timing of N leaching, and also on the pasture production. The plot with beef cattle had the most N leaching followed by dairy and sheep. The simulation results suggested that sheep farming systems with limited N fertiliser inputs are likely to decrease N leaching from Taupo farms. They suggested that farm systems should be designed carefully for the Lake Taupo catchment areas in order to control N leaching from the farm system.

2.8.3 Modelling Water and Nutrient Movement in a LTS

Several authors have modelled water and nutrient use in an LTS in recent years, but there is no single modelling approach that is used by the majority of researchers. Snow et al. (1999a, b) used APSIM for Effluent to model the water balance and N dynamics in a Eucalypt plantation irrigated with bore water. APSIM is described by Snow et al. (1999b) as *"…a simulation environment that can be configured with modules suitable for the simulation of many different agricultural production systems. APSIM for Effluent is a particular configuration of APSIM which includes the modules required for the simulation of effluent-irrigated plantations."*

Al-Jamal et al. (2002) developed a daily growth-irrigation scheduling model for *Eucalyptus* tree plantations irrigated with treated sewage water, and Sharma and Ashwath (2006) used the MEDLI (Model for Effluent Disposal using Land Irrigation) model to schedule effluent irrigation in assessment of the performance of different agroforestry systems. MEDLI is a software package developed by the Queensland Department of Natural Resources, Mines and Energy for scheduling irrigation for effluent treated cropping systems.

Kunjikutty et al. (2007) chose the LEACHN model to simulate nitrate leaching through soils to identify the best wastewater application strategies. The model was calibrated and tested against experimental data derived from lysimeter experiments prior to the final simulations. The LEACHN model is a module of the LEACHM model (Kunjikutty et al., 2007) designed to simulate leaching of nutrients. Other modules of the LEACHM model simulate the movement of water, pesticides and other chemicals. Kunjikutty et al. (2007) chose the LEACHN model for their simulations because they thought it would be "more robust and simpler" than some other models, and also because it had a well-described N-simulation algorithm.

Although there have only been a limited number of modelling studies on LTSs there have been a number of modelling studies on manure-fertilised cropping systems. One such study was that of Sogbedji et al. (2006), who evaluated the ability of the Precision Nitrogen Management (PNM) model to simulate drain flow NO_3-N concentrations under manure-fertilized maize. The PNM model is based on the LEACHN model, but with a more sophisticated crop growth and N uptake component. Sogbedji et al. (2006) concluded that the PNM model appeared promising in its ability to predict N dynamics under manure-fertilised maize.

In contrast to the process-driven models (such as LEACHN) described above, Wheeler et al. (2006) have developed the OVERSEER® nutrient budget model, which is on-farm decision support software designed to assist users to develop nutrient budgets for New Zealand farms. The model has the capacity to predict leaching losses from different areas of farms, including those areas used for land disposal of FDE. The model predicts annual losses of NO_3-N and, similarly, the input variables are annual values. The underlying algorithms are based on research done in NZ conditions, and can provide good predictions of leaching losses of NO_3-N from “effluent blocks” on New Zealand dairy farms, but the model has not been applied to other LTSs.

Over the past 10 years, the need to safely apply effluents to LTSs has continued to increase. There is now a better understanding of some of the processes controlling the movement of N in soil and there may be the opportunity in the future to manipulate the N cycle to achieve better environmental outcomes through the use of new technologies such as nitrification inhibitors. Despite the passage of time however, many of the fundamental principles outlined earlier in this review still apply and in a very recent publication Tzanakakis et al. (2009) still felt it necessary to point out that:

“Efficient planning, design, operation and management of land treatment systems (LTS) is therefore required to reduce nutrient losses to the environment.”

2.9 Summary and Conclusions

- This literature review shows that although there is more information on NO_3-N leaching losses from soils, the links between these losses and contamination of groundwater has not been well established either in New Zealand or overseas.
- The application of N at various rates in the form of N fertiliser, meatworks effluent, dairy effluent, and sewage effluent in all cases increased NO_3-N concentrations in groundwater. However, there is insufficient information linking the application rate of N from any source with groundwater contamination of NO_3-N to establish definitive recommendations for N loading rates.
- To operate a system within safe limits, it is necessary to make some site specific measurements that provide the guidelines to make short and longer term management decisions about when (what time of the year) and how much (kg/ha/year) effluent should be applied to the land. From information collected

during the field investigation, the engineer or manager can confirm the suitability of the sites for the identified land treatment processes. Based on the loading rate estimates, land area, pre-application effluent treatment, storage, and other system requirements can be estimated.

- Climate, site area, and the water and nutrient use capacities of the vegetative cover determine how much effluent can be applied on a particular day. Effluent loading rate also depends on a different set of factors—notably seasonal conditions, day to day weather, soil physical characteristics and soil moisture levels. Other factors that might influence the amount and timing of effluent irrigation include the allowable N loading rate and the occasional need to leach salts from the root zone.
- The design and management of land treatment systems should focus on ensuring that they retain their ability to deal with effluent over the long term and have minimal adverse environmental impacts. Particularly important goals are maintaining the quality of ground and surface water resources and avoiding soil contamination or structural change that could reduce vegetative cover growth or constrain land use in the future.
- The use of nitrification inhibitors to reduce NO_3-N leaching losses at effluent irrigated land treatment sites has so far received little attention in New Zealand. This area merits further study.
- The leaching losses of DON are normally small but could be a significant contributor to N leaching from agricultural ecosystems. Therefore, it may be important to measure DON while conducting nutrient budget studies, and also to include a sub-model (within an N leaching model) to simulate leaching of DON.
- Improved models of the fate of land-applied effluent need to be developed and tested to better describe the processes involved and to assist with the prediction of environmental impacts. This will require an integrated knowledge of the processes operating in the soil-plant-water (and occasionally animal) system. This can only be achieved by interdisciplinary teams, including pedologists, soil physicists, soil chemists, soil biologists, plant scientists, hydrologists, engineers, and farm managers.
- Most of the models developed so far have been tested by their developers and have shown their usefulness to estimate nutrient losses to the environment at a small or

larger scale. Cichota and Snow (2009) reported that it is important to maintain the development and testing of available models as well as continuing laboratory or field scale studies of actual processes involved in the loss of nutrients. They pointed out that:

"There is, in general, a lack of organised information about how several of these models work and what their main purposes are…….Knowing their strengths and deficiencies will help to select models properly and understand better their results".

They also pointed out that:

"There is much misunderstanding and lack of information about the potential of existing tools and that may be limiting their use and acceptance…..It is important to maintain the development and testing of these models, as well as the studies on the actual processes involved on nutrient loss, so that the descriptions can be better understood and improved. How well we communicate and deal with uncertainty, whether arising from measurements or from modelling, is a challenge that deserves a close look in the near future".

CHAPTER 3

AN EXPLORATORY FIELD SCALE INVESTIGATION

3.1 Background

In the previous chapter, it was noted that the application of N at various rates in the form of N fertiliser, meat works effluent, dairy effluent, and sewage effluent is often associated with increased NO_3-N concentrations in groundwater. However, there is only limited information linking the application rate of N from any source with groundwater contamination by NO_3-N in a way that could be used to establish definitive recommendations for N loading rates.

This chapter describes an exploratory field scale investigation that was undertaken at a land treatment site belonging to Carterton District Council, and relates to Objective 1 of the study (Section 1.3). The Carterton site was used because there was an existing research contract that utilised this site. The goal of the study was to examine how the irrigation of sewage effluent on trees and pasture at different application rates affected the concentration of NO_3-N in ground water. As part of the study, an assessment was made of what site specific factor(s) (SSF) could and should be measured in order to indicate NO_3-N leakage from the LTS into the groundwater, in both the short and long term. This information could then be used when developing monitoring protocols at a LTS, and devising appropriate management strategies for operating a LTS. This chapter is based on the published paper reproduced below. A number of changes have been made to the original text of the paper to provide clarity and consistency within the thesis, and to avoid undue overlap with the material presented in Chapter 4.

3.2 The Environmental Impact of Sewage Effluent Irrigation onto Land - A Case Study in New Zealand

by Babar Mahmood and Gavin L Wall

Agricultural Engineering Journal, 2001, 10(3&4), 209-230.

3.2.1 Introduction

Application of wastes onto land is an approach that is at the interface of waste management, land use and water quality management (Heinzmann and Sarfert, 1995). The main objective of a land treatment system is to provide a sustainable process for the treatment and disposal of effluent (e.g. sewage, industrial waste, agricultural waste) through natural processes occurring in the plant-soil-water matrix. Sustainability generally implies a high degree of waste treatment and non-degradation of the receiving environment.

One of the main environmental concerns arising from land application is the risk of groundwater contamination. Prior to application, the effluents may still contain large concentrations of organic matter, nutrients such as N and P, and other contaminants (Hauber, 1995). Discharging sewage effluent to waterways can result in water quality degradation such as eutrophication, depletion of dissolved oxygen, chemical toxicity, and salinity. In New Zealand, land application of wastewater is becoming more widespread as regulatory authorities move to protect water quality by restricting sewage effluent (i.e. treated wastewater) disposal into rivers, lakes, and the marine environment. The factors contributing to increased use of land for waste application include the requirements of the Resource Management Act 1991, and cultural concerns related to beliefs of the indigenous Maori population that all sewage waste should pass through the earth prior to entering fresh water. Although lysimeter studies (e.g. Roygard et al., 1998; Magesan et al., 1998) have been conducted to monitor the environmental impacts of NO_3-N leaching into groundwater and nutrient uptake by crops, NO_3-N leaching in effluent-treated forest soils has received relatively little attention (Magesan et al., 1998). The study reported here was conducted to assess the environmental impacts – particularly contamination of groundwater with NO_3-N - of sewage effluent irrigation, at different hydraulic loading rates, onto land planted with trees and pasture.

3.2.2 Methodology

The experimental site was located at Carterton district of Wellington region, New Zealand. The current population of Carterton is close to 4200. There was a wastewater treatment plant that consists of fine screen, primary sedimentation tank with sludge digestion, and secondary and tertiary oxidations ponds at the time of experiment. The treated effluent was discharged to nearby Mangatarere stream. There was no history of applying treated effluent onto land before this experiment. At the time, the Carterton Council has the permission to discharge the treated effluent into the stream. The experimental site was very close to the oxidation ponds.

The soil was a stony silt loam from the recent fluvial deposits associated with the Mangaterere River and tributary streams. The land treatment area was located immediately adjacent to the oxidation ponds of the effluent treatment plant (Fig. 3.1). The study area comprised 0.162 ha, planted with either of two tree species *Eucalyptus Nitens (E-N) and Eucalyptus Ovata (E-O)*) or with pasture. At the commencement of the experiment the trees were approximately 2 years old.

There were three blocks, each of 540 m^2 (Fig. 3.2). Each block consisted of 3 plots (2 plots of trees with the species in each plot randomly assigned and one plot of pasture). The area of each plot was 180 m^2. The stocking density of each tree species was 5000 stems/ha. Previously, the land had been periodically grazed by sheep and had never been fertilised or irrigated with effluent (Carterton District Council staff). The topography within each irrigated plot was not leveled. Five monitoring wells were installed at the site – with one (wells 1, 2 and 3) in each effluent irrigation treatment

(Fig.3.2). The fourth well was installed in the pasture plot receiving the high effluent treatment, and the fifth (well 5) was installed on the upstream side of the experimental site, between the oxidation pond and plantation area (Figs. 3.1 and 3.2).

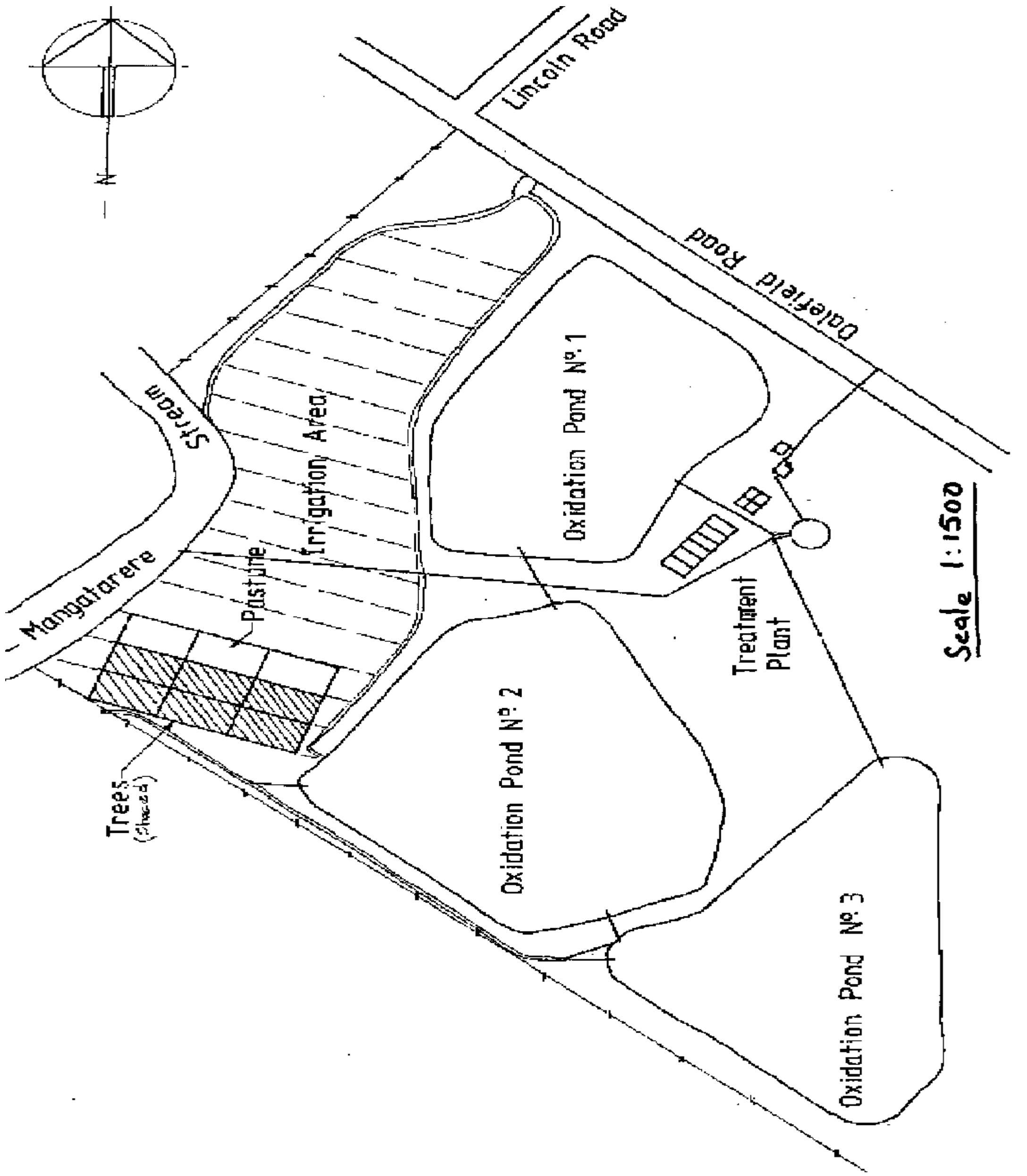


Fig. 3.1: Existing site layout.

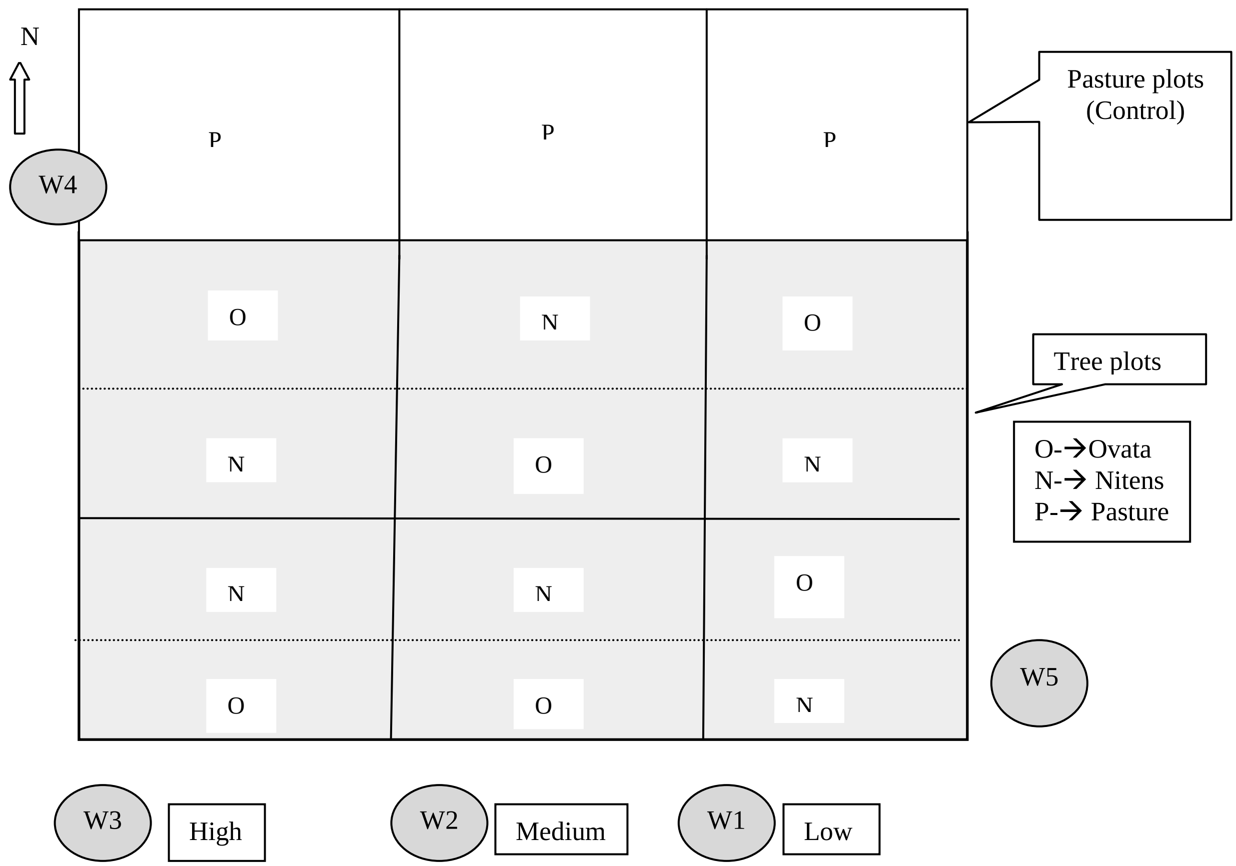


Fig. 3.2: A layout map showing the tree plantation and pasture areas, the locations of monitoring wells, and the areas receiving the low, medium and high rates of effluent application at the site.

The casings of all the monitoring wells were extended 1m above ground level, with the screened section starting 1m below ground level. Monitoring wells 3 and 4 on the downstream (river) side of the plantation extended only 3 m below ground level due to caving in of material during excavation, but the rest extended to a depth of 5 m. The location of these wells was determined on the basis of a piezometric survey conducted by Good Earth Matters Consultants.

A trickle irrigation system was designed to apply sewage effluent at three hydraulic loading rates of 30, 45, and 100 mm per week (designated as low (L), medium (M), and high (H) treatments, respectively). The lowest hydraulic loading rate was chosen on the basis of N as the land limiting constituent (LLC). The hydraulic loading rate used for design is based on the more restrictive of two limiting conditions - the capacity of the soil profile to transmit water (soil permeability) or the limiting constituent concentration in the effluent applied. In municipal wastewater land treatment systems, N is usually the limiting constituent when protection of potable groundwater is a concern. If percolating water/effluent enters a potable groundwater, then the system should be designed such that the concentration of NO_3-N in the receiving groundwater at the project boundary does not exceed 10 mg/L (WHO, 1984).

The concentration of total N in the sewage effluent was 15.5 g/m^3, and the maximum allowable N loading rate according to the Wellington Regional Council Rule 11 (Annual Wellington Regional Council Plans, 1997, pp. 65) was 150 kg N/ha/year. The designed N loading rates for the low, medium, and the high irrigation treatments were 150, 225 and 525 kg N/ha/year, respectively.

The trial began in December 1997 and continued through to August 1998. Monitoring of site specific factors relevant to the climate, effluent, soil, plants, and groundwater was undertaken.

The daily rainfall, and maximum and minimum daily temperatures were measured at the site. Daily evaporation was not measured at the site. Instead, daily pan evaporation data recorded at Martinborough and crop (i.e. pasture) evapotranspiration (ET) data recorded at East Taratahi were collected from the National Institute of Water & Atmospheric Research, New Zealand. The two sites were on different sides, but close (approximately 10 km) to the experimental site.

The daily pan evaporation data recorded at the Martinborough station was used to calculate the potential evapotranspiration (PET) for pasture and tree crops for the experimental site. A crop factor of 0.75 was selected for pasture. This figure was chosen after noting that crop factors in agriculture typically range from 0.6 to 1.0 (e.g. Israelsen and Hansen, 1962; Stewart et al., 1988). The crop factor was then multiplied by the pan evaporation data to estimate the pasture PET. The results of the estimated pasture PET (from Martinborough) and pasture ET (from East Taratahi station) were compared. A t-test was carried out to show the statistical comparison between the ET values from both stations.

Because of limited data on the water use of natural *Eucalyptus Nitens* and *Ovata* stands, and less information on water used by trees under effluent-irrigated conditions, a final crop factor of 1.5 for both tree species was selected to calculate the tree PET. This was based on evidence that forest uses more water than grassland (e.g. Holmes and Colville, 1970; Stewart et al., 1988).

A bi-weekly climatic water balance approach was adopted to estimate the actual evapotranspiration (AET) as reported by Myers et al. (1994). The input parameters used in the climatic water balance were; precipitation (PPT) measured at the site, PET estimated for the site, and the potential soil water storage (SMS) which can be calculated by multiplying the water holding capacity (WHC - mm/m) of the soil by the effective plant root depth (m).

The soil texture at the site was silt loam and therefore a WHC of 200 mm/m was assumed (Department of Agricultural Engineering, 1983). The effective root depths for pasture and trees were assumed to be 0.6 m and 1.0 m respectively (Bryan Myers, personal communication, 1999). Similar root depths are also reported by Myers et al. (1994).

The calculated soil water storage values for pasture and trees (i.e. 120 mm and 200 mm) were used in the climatic water balance in order to determine the AET for both crops (pasture and trees). The estimated AET values were used in the water balance to calculate the drainage losses in all treatments on the tree and pasture plots. The data on drainage losses are reported in Chapter 4. A Kent flow meter was used to measure the amount of effluent added to the system during each irrigation cycle. Flow meter readings were taken to check that the system was running at the designed application rates.

Volumetric soil moisture content (SMC) measurements were made using Time Domain Reflectometry (TDR) probes at three depth ranges (0 - 150, 0 - 300, and 0 - 500 mm) prior to irrigation, and then fortnightly in all treatments on the tree and pasture areas. Soil temperature measurements were made using soil temperature probes at soil depths of 150, 300 and 500 mm prior to irrigation, and then fortnightly.

Soil-water samples were collected from suction cups. These porous ceramic cups, 25 mm in diameter and 60 mm long, were installed at depths of 150, 300 and 500 mm in all plots (pasture and tree) by boring holes in the soil to the required depth, and then placing sand around the ceramic cups, and filling the holes with the original soil from the site. Two suction cups were installed for each depth per tree plot per irrigation treatment – giving a total of 36 suction cups in the tree plots. One suction cup was installed for each depth per pasture plot per irrigation treatment – giving a total of 9 suction cups in the pasture plots.

The effluent, soil-water and groundwater samples were collected prior to irrigation and then fortnightly. Fifteen soil core samples were collected at the beginning of the experiment from each tree and pasture plots for 0 - 500 mm depth using an electric soil core sampler. Three soils samples were taken from each soil core at 150, 300 and 500 mm depth (i.e. 15 samples for each soil depth). Another fifteen samples were collected at the conclusion of the experiment using the same method, and the soil samples were collected at the respective depths from each core. The samples collected before and at the conclusion of the experiment were analysed separately for each depth. The effluent, soil pore water, and groundwater samples were stored at 4^oC and then analysed for total Kjeldahl nitrogen (TKN), NO_3-N and NH_4-N using the standard methods of water analysis (Gillian, 1984). Soil total N and available N (i.e. NO_3-N and NH_4-N) were determined using the standard Auto Analyser method (Kamphake et al., 1967 and Gillian, 1984) and expressed on a dry weight basis for the 150, 300 and 500 mm depths in all the tree plantation and pasture treatment areas, before and at the conclusion of the experiment. The total and available N contents in the top 500 mm of soil were calculated by multiplying the soil bulk density at different depths with the N concentration at those depths, and then adding the values for all depths up to 500 mm. Soil bulk density, soil pH, and soil organic carbon were determined for the 150, 300 and 500 mm depths in all the tree plantation and pasture treatment areas before and at the conclusion of the experiment. The soil bulk density and pH samples were all bulked. A survey was carried out to determine the levels of the bed of the effluent channel (which was carrying the tertiary treated effluent from the oxidation pond to the river), the river bed, and the ground surface levels of well 1, well 2 and well 3.

Four trees of each species were selected from each tree plot, around the suction cup and TDR probe sampling points. Height and diameter at breast height (DBH) measurements were taken prior to and at the end of experiment. In each plot the diameter of selected trees was measured 15 cm above the ground. This is called the basal diameter. Trunk wood (stem wood) sub-samples of both tree species were collected at the end of experiment when the trees were almost three years old. Tree leaf samples were collected from the selected trees in each tree plot before and at the end of the experiment.

The pasture was cut at a height of 10 mm before irrigation and removed from the site. The next pasture cut was made to the same height 10 weeks after the first irrigation, then the final pasture cut was made at the end of the experiment to determine the total dry matter production of pasture (kg/ha/38 weeks). The leaf and pasture samples collected before and at the end of the experiment, and stem-wood sub samples collected at the conclusion of irrigation were analyzed for N using the standard methods of plant analysis (Gillian, 1984).

3.2.3 Results and Discussion

3.2.3.1 Climate

The mean annual rainfall of 700 mm during the study (Dec 1997 - August 1998) was close to the long-term average of 713 mm. The annual pan evaporation was 900 mm and was strongly seasonal, varying from a monthly low of 20 mm in June and July to a peak of 195 mm in January and February (Fig. 3.3). The mean minimum temperature of the coldest month was -2 ^{0}C and mean maximum temperature of the hottest month was 32 ^{0}C. The months of June and July (early winter) were considerably wetter and less evaporative at the site. The rainfall was consistently less than the pan evaporation during the months from December to April.

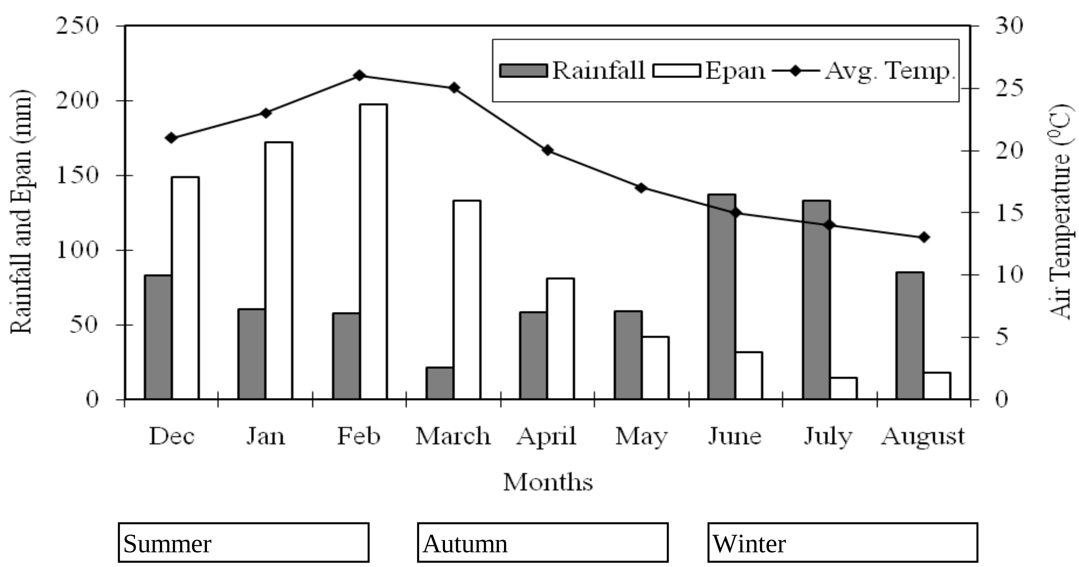


Fig. 3.3: Monthly pattern of rainfall, pan evaporation and air temperature data for the site.

3.2.3.2 Effluent Monitoring

Overall the experiment went well. Generally speaking, no ponding or surface runoff was observed during effluent irrigation at the site. The trees and pasture were growing well in all irrigation treatments at the site. The effluent applied was generally alkaline (pH 6.65 - 8.36) (Table 3.1) and of low to medium salinity (Electrical Conductivity - EC) 250 - 470 μS/cm). Overall the quality of the effluent applied was satisfactory for irrigating tree crops in terms of the salinity hazard (EC) and pH (U.S. EPA, 1980). The cumulative N inputs (loading) for the whole period of irrigation, in all tree plantation and pasture treatment areas, are shown in Table 3.2.

Table 3.1 - Chemical characteristics of the effluent applied.

Parameter	**Mean**	**Range**
pH	7.95	6.65 - 8.36
EC (μS/cm)	371	250 - 470
TKN (mg/L)	16	15 - 45
NH_4-N (mg/L)	10.57	3.9 - 18.56
NO_3-N (mg/L)	0.83	0.25 - 3.00

Table 3.2 - Cumulative nitrogen loading rates in all treatments on the tree and pasture plots.

Treatments	**Nitrogen Loading (kg N/ha)**	
	38 weeks	**year**
L	130	178
M	195	267
H	433	593

Organic N and NH_4-N were the principal forms of N in the effluent. Most of the N present in the effluent was as NH_4-N - ranging from 70% to 90% (Table 3.1). The monthly trends of TKN, NO_3-N and NH_4-N concentrations in the effluent applied, for the whole irrigation period, are shown in Fig. 3.4. The concentration of TKN in the effluent applied fluctuated between 15 and 45 mg/L (Table 3.1). Other field studies (e.g. Pound and Crites, 1973; Asano et al., 1985) have shown similar TKN concentrations in secondary effluent applied to land. Although the TKN concentrations given in Table 3.1 are representative of most sewage effluents used for irrigation, higher or lower concentrations are possible, and concentrations below 10 mg/L are not uncommon (Idelovitch et al., 1976). Moreover, seasonal fluctuations are normal and should be taken into account (Feigin et al., 1991).

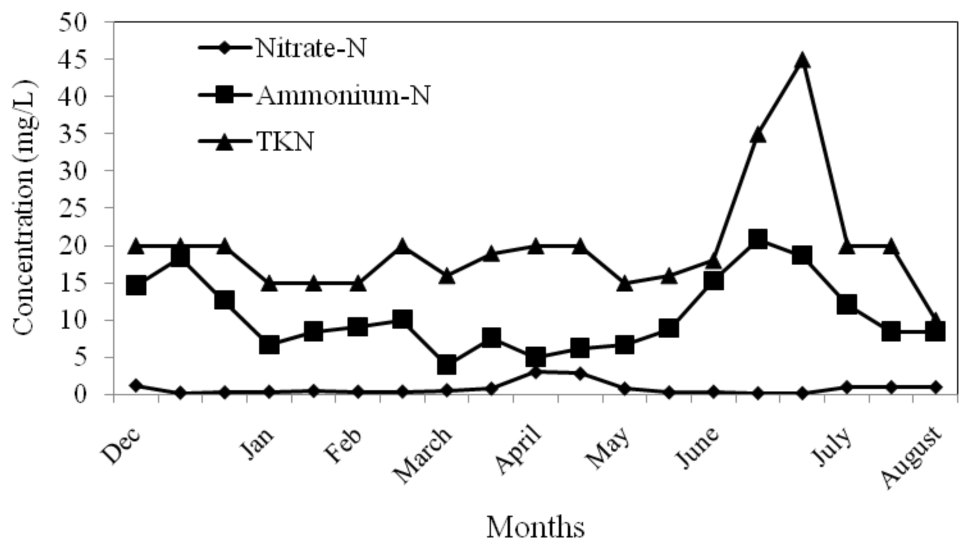


Fig 3.4 - Monthly trends of TKN, NO_3-N and NH_4-N concentrations in the effluent applied (Dec 97-Aug 98).

3.2.3.3 Estimation of Actual Water Use

A t-test indicated that there was no significant difference between the potential ET values from the two weather stations located in opposite directions from the experimental site. This suggested that it was reasonable to use the Martinborough estimated PET data for the experimental site. The patterns of the estimated AET for tree and pasture crops, on a daily and monthly basis, are shown in Fig. 3.5. The estimated AET values for both tree species were assumed to be the same because there was very little difference in the SMC between them throughout the experimental period (Fig. 3.6).

The daily estimated water use for *Eucalyptus* trees varied from 9.2 mm/day (during summer) to 1 mm/day (during winter), and from 5.2 to 0.5 mm/day for pasture (Fig. 3.5).

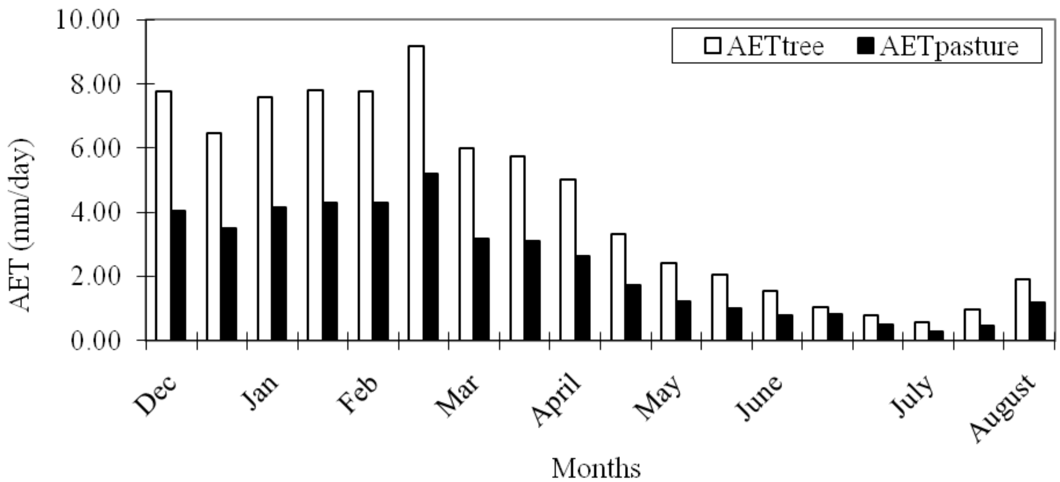


Fig. 3.5: Estimated daily evapotranspiration rates for *Eucalyptus* trees and pasture.

The fluctuation in water use was due to the seasonal climatic variation (rainfall and temperature) at the site. During summer and early autumn, the monthly estimated AET of the trees was between 180 - 250 mm. This compares with the monthly effluent irrigation applied in the L, M, and H treatment blocks of 120, 180 and 400 mm respectively.

3.2.3.4 Soil Moisture Content Measurements

The volumetric soil moisture contents measured to 500 mm depth, for each tree species and the pasture plots, are presented in Fig. 3.6. The soil moisture content (mm) was calculated from the measured SMC using the following equation. The effective depth used was 500 mm.

$$SMC\ (mm) = [SMC\ (\%)\ X\ Effective\ depth]\ /\ 100 \qquad (3.1)$$

There was a consistent increase in SMC at times of significant rainfall, and a general drying over the dry period (Fig. 3.6). During summer, two high peaks of SMC were observed in response to rainfall in all treatments. During winter (June - August), when the evaporative demands were less and rainfall was high, the SMC increased to 30 - 46% in all treatments of tree plantation and pasture areas (Fig. 3.6).

On the tree plantation area it was observed that the SMC was relatively stable at 38% after irrigation and did not go above this level (Fig. 3.6). It suggests that this is the upper limit of the soil's ability to hold water and can be called the field capacity (FC) of the soil. In this case the term FC can be defined as the SMC when the application of water has stopped, the water in the largest pores has drained out rapidly, and the SMC is relatively stable. Often it takes 1 to 2 days to drain the water from the macro-pores, depending on the texture, structure and hydrological conditions of the soil.

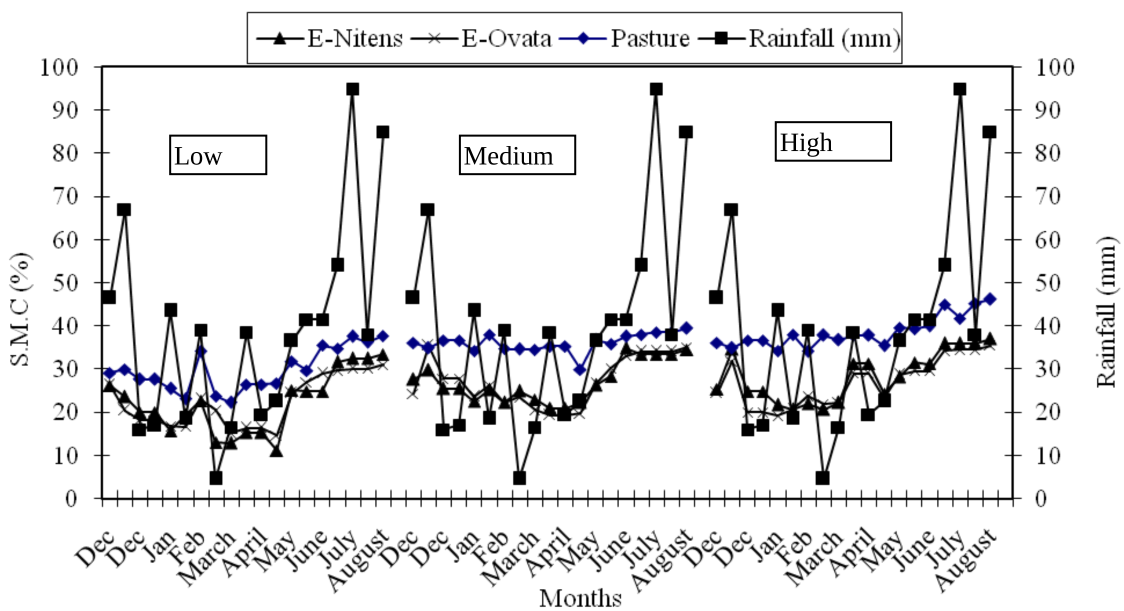


Fig. 3.6: The pattern of soil moisture content in the 0 - 500 mm soil depth in the L, M, and H treatments of tree and pasture plot areas.

As originally defined (Veihmeyer and Hendrickson, 1949), FC "*is the amount of water held in soil after excess water has drained away and the rate of downward movement* *has materially decreased, which usually takes place within 2 or 3 days after a rain or irrigation in soils of uniform structure and texture*". A later definition (Rich, 1971), although still stating or implying that the profile should be well drained, omits the reference to uniform structure or texture. Although it is not possible to define FC precisely, it can be described as the state of the soil after rapid drainage has effectively ceased and the soil water content has become relatively stable (David Scotter, personal communication, 1998).

The SMC in the pasture plots was higher throughout the irrigation period than the SMC in the tree plots. During summer and autumn, the SMC in all treatments of the tree plantation and pasture areas was between 10 - 25% and 22 - 38% respectively. The reason for the lower SMC for the tree plantation area during irrigation was probably that the daily water extraction by trees was higher than by pasture, although during winter, when drainage was occurring on all plots, the SMC was still higher in all treatments on the pasture area than on the tree plantation area. It is difficult to explain this difference, except to say that the root development of the 3 year old trees may have an effect. This effect warrants further investigation.

3.2.3.5 Soil Characteristics

Soil temperature: The soil temperature in the tree plantation area was 2 - 4 ^{0}C (on average) lower than the soil temperature in the pasture area, possibly due to shading effects. During summer and autumn (high air temperature and fewer rainfall events) soil temperature was high, ranging from 15 to 25 ^{0}C in all treatments for the tree and pasture plots. In winter the soil temperature dropped to 10 ^{0}C in all treatments. It is well known (McLaren and Cameron, 1990) that soil temperature has a large effect on soil biological and chemical activities. The rate of biological and chemical transformations of nutrients increases as the soil temperature increases, and they slow down or stop at low temperature.

Soil total nitrogen concentration: There was considerable variability in total nitrogen level between samples taken before and after the experiment (Table 3.3). This was due in part to the different sampling locations within each plot before and at the end of the experiment. This variability was exacerbated by the spatially variable soil type, which consisted of layers of varying thickness of undifferentiated alluvium. This variability meant that few conclusions could be drawn about differences in total soil N between treatments, or over the course of the trial (Table 3.3).

Soil available (or 2M KCl-extractable) nitrogen (NO_3-N and NH_4-N): The average soil available N levels measured at different depths in all treatments on the tree plantation and pasture areas are given in Table 3.3. At the end of the experiment the concentrations of soil mineral N under the trees were considerably higher than at the start of the experiment, and the major part (80 - 90%) of the extractable N was NO_3-N, indicating a high nitrification rate. This high nitrification rate is consistent with the high SMC prior to the final sampling and the favorable soil pH (Table 3.3). The optimal pH for nitrifying bacteria is in the range from 4.5 - 7.5 (McLaren and Cameron, 1990). They also reported that the maximum rate of nitrification occurs in soil at or around field capacity. In wetter soils, the rate slows down and in saturated soils it is almost zero because there is little or no oxygen. In soils drier than permanent wilting point, the rate of nitrification is also slow. In general, nitrification can proceed when there is sufficient water stored for plant growth. Interestingly, there was no such increase in available N in the pasture plots. This may result from the increased leaching in the pasture plots (see Chapter 4).

Soil bulk density and soil pH: The bulk densities of the soil samples taken at the end of the experiment were lower for all depths than soil samples taken prior to irrigation. The t-test suggested that this difference was statistically significant (at 95% confidence interval). The reduction in bulk density value could perhaps be attributed to increased root growth and organic matter additions resulting from the effect of effluent addition on plant growth.

Table 3.3 - Soil physical and chemical properties at different depths in all treatments of tree and pasture areas before and after the experimental period.

Tree plots

Treatment	Parameters	150 mm		300 mm		500 mm	
		Before	After	Before	After	Before	After
Low	Total N (mg/kg)	2060	2380	2290	2670	2790	1660
	2MKCl NO_3-N (mg/kg)	6.28	49.70	6.19	39.8	5.72	22.9
	2MKCl NH_4-N (mg/kg)	2.74	1.70	3.12	2.90	2.98	2.38
	pH	5.10	5.13	5.12	5.20	5.14	5.40
	Bulk density (kg/m^3)	1051	885	1382	1162	1438	1272
Medium	Total N (mg/kg)	2170	2080	1600	1250	1270	900
	2MKCl NO_3-N (mg/kg)	6.38	20.50	3.85	15.5	4.88	10.3
	2MKCl NH_4-N (mg/kg)	3.56	1.10	2.63	1.84	2.63	1.68
	pH	5.10	5.13	5.12	5.20	5.14	5.40
	Bulk density (kg/m^3)	1051	885	1382	1162	1438	1272
High	Total N (mg/kg)	2810	3090	1480	2260	2310	3250
	2MKCl NO_3-N (mg/kg)	3.66	44.7	3.00	25.3	3.00	20.8
	2MKCl NH_4-N (mg/kg)	5.72	4.30	2.53	2.70	3.10	2.20
	pH	5.10	5.13	5.12	5.20	5.14	5.40
	Bulk density (kg/m^3)	1051	885	1382	1162	1438	1272

Pasture plots

Treatment	Parameters	150 mm		300 mm		500 mm	
		Before	After	Before	After	Before	After
Low	Total N (mg/kg)	3270	1960	2380	2860	970	1450
	2MKCl NO_3-N (mg/kg)	12.75	12.8	9.00	12.00	5.63	4.00
	2MKCl NH_4-N (mg/kg)	12.38	3.20	5.63	1.68	3.38	2.40
	pH	5.30	5.70	5.3	5.70	5.30	5.80
	Bulk density (kg/m^3)	1051	885	1382	1162	1438	1272
Medium	Total N (mg/kg)	2880	2370	1640	870	880	670
	2MKCl NO_3-N (mg/kg)	10.88	8.80	4.13	4.00	4.13	4.00
	2MKCl NH_4-N (mg/kg)	4.35	2.00	2.85	1.92	5.63	1.68
	pH	5.30	5.70	5.30	5.70	5.30	5.80
	Bulk density (kg/m^3)	1051	885	1382	1162	1438	1272
High	Total N (mg/kg)	1730	130	900	1090	690	630
	2MKCl NO_3-N (mg/kg)	5.63	4.00	2.63	4.00	1.88	2.00
	2MKCl NH_4-N (mg/kg)	31.88	4.00	3.75	2.40	3.00	2.00
	pH	5.30	5.70	5.30	5.70	5.30	5.80
	Bulk density (kg/m^3)	1051	885	1382	1162	1438	1272

3.2.3.6 Soil Solution Monitoring (Suction Cup Samples)

No soil solution samples were obtained before irrigation - presumably because the soil was dry at the time of sampling, due to high temperature and evaporative demands. Immediately following the start of irrigation in December, soil solution samples were obtained from all treatments on both the tree and pasture plots. From then on however, it proved difficult to obtain soil solution samples from many treatments. It appears as if this difficulty was related to SMC, as more soil solution samples were obtained from plots receiving the high rate of irrigation, and from the pasture plots – which were wetter than the tree plots. However it is not at all clear why soil solution samples were able to be collected from some treatments in December, but not again for the duration of the trial – even when the SMC was higher than it had been in December.

The concentrations of NO_3-N and NH_4-N, pH and electrical conductivity of all soil solution samples were determined. Nitrate-N in the soil solution samples taken at the 500 mm depth was considered to be beyond the influence of the root zone and lost from the site. The NO_3-N concentration of soil solution samples collected from the tree plantation area and the pasture area were compared with each other, as shown in Fig. 3.7 and 3.8. The results show that, generally, NO_3-N concentration in soil solution decreases with an increase in soil depth and there was no consistent difference between tree and pasture plots.

Very high NO_3-N concentrations (19 - 137 mg/L) were observed in the soil solution samples taken from the low irrigation treatment at the commencement of the trial in December. These high concentrations of NO_3-N were attributed to the flushing through of NO_3-N (already resident in the soil) from the top soil layer to the sampling depths in response to a rainfall event of 67 mm plus the first irrigation. Much lower NO_3-N concentrations (2 - 17 mg/L) were found in soil solution samples collected from the medium and high irrigation treatments (Fig. 3.8). One possible reason for this was that the combination of higher rates of effluent irrigation and the large rainfall event had leached the initial "plug" of NO_3-N beyond the sampling zone. This is supported by the high concentrations of NO_3-N observed in the groundwater at this time (Section 3.2.3.7).

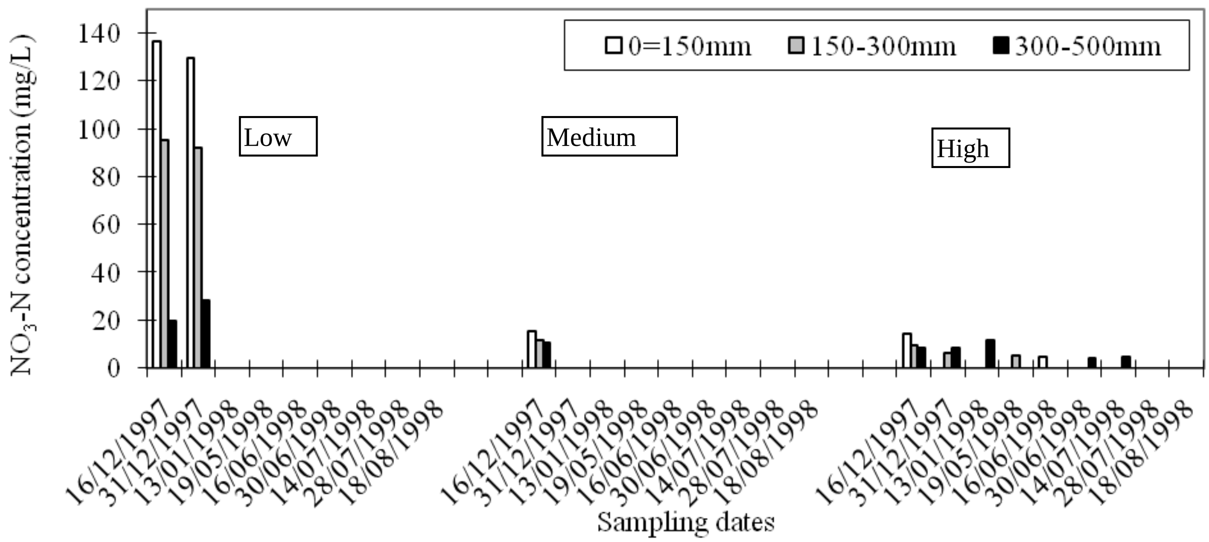


Fig. 3.7: NO_3-N concentrations in soil solution samples taken from the treatments receiving low, medium and high rates of effluent irrigation on the tree plantation area.

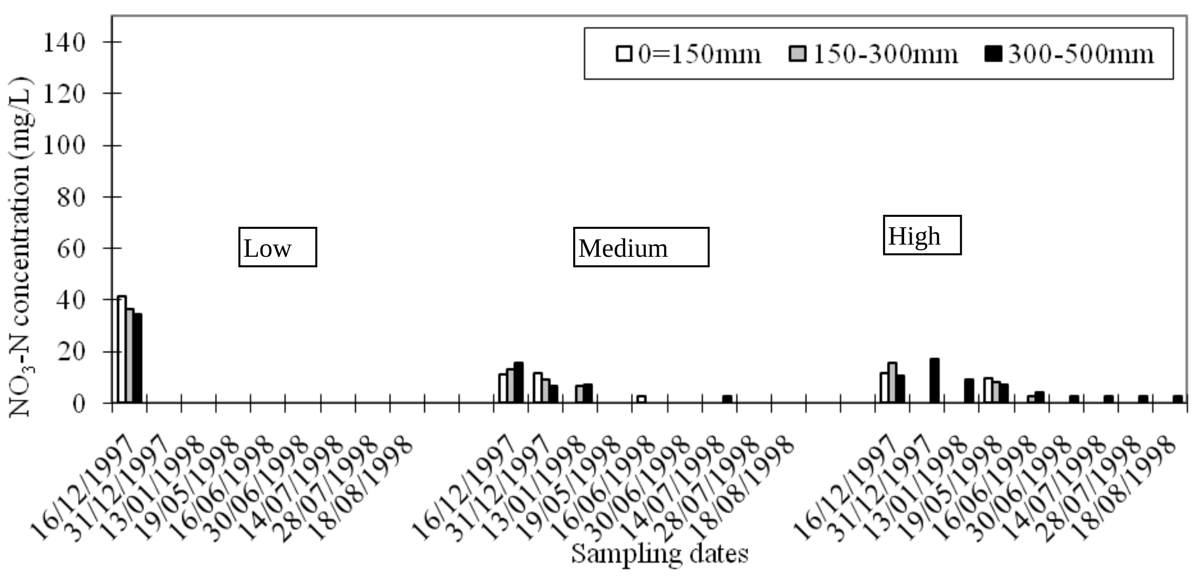


Fig. 3.8: NO_3-N concentrations in soil solution samples taken from the treatments receiving low, medium and high rates of effluent irrigation on the pasture area.

Another possible reason for the high NO_3-N concentrations in the treatments receiving low rates of effluent irrigation could be a lower SMC than in the treatments receiving higher rates of effluent irrigation. While sampling suction cups in the field, it was noticed that a lower volume of soil solution was obtained in the L treatment as compared to the sample volume obtained from the M and H treatments. Presumably the low volume of soil solution collected reflects a lower SMC. This then would increase the NO_3-N concentration in the soil solution that remained.

3.2.3.7 Groundwater Monitoring

Groundwater table monitoring: The results of groundwater depth monitoring at the site are shown in Fig. 3.9. The groundwater table depth dropped in all monitoring wells during summer and autumn, but it started to rise during winter in response to rainfall events. The water table was on average 2 to 3 m below the ground surface at the site. During summer the water table levels (from ground surface) at wells 1, 2 and 3 were 3.15, 2.8 and 2.41 m, respectively. Whereas during winter the water table levels at wells 1, 2 and 3 were 2.57, 1.84 and 1.82 m, respectively (Fig. 3.10).

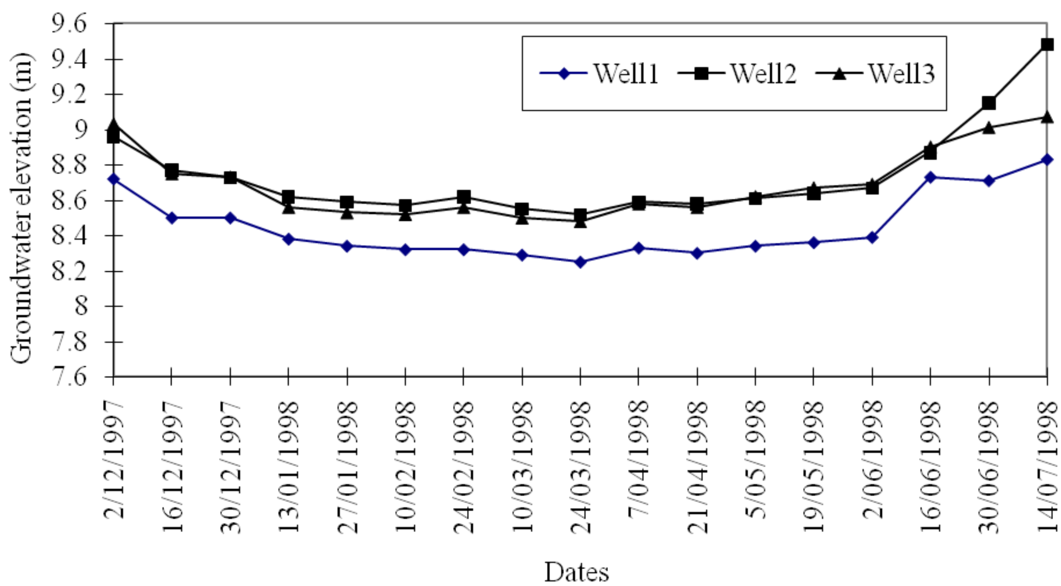


Fig. 3.9: Groundwater table levels (relative to ground level) in low (well 1), medium (well 2) and high (well 3) effluent irrigation treatments.

The rise in groundwater table at the site was not attributed to the irrigation practices but rather there was a general trend of rising groundwater table levels throughout the region due to high rainfall during winter. A survey of ground surface, channel bed, groundwater table and river bed levels shows that seepage is very likely to occur from the oxidation ponds to well 1 (due to an elevation difference of 2 - 3 m between the channel bed and water table level of well 1). Seepage is also likely to occur from the river to well 3 and well 4 (Fig. 3.10). The groundwater table level and ground surface level show that the groundwater flow direction was towards the river.

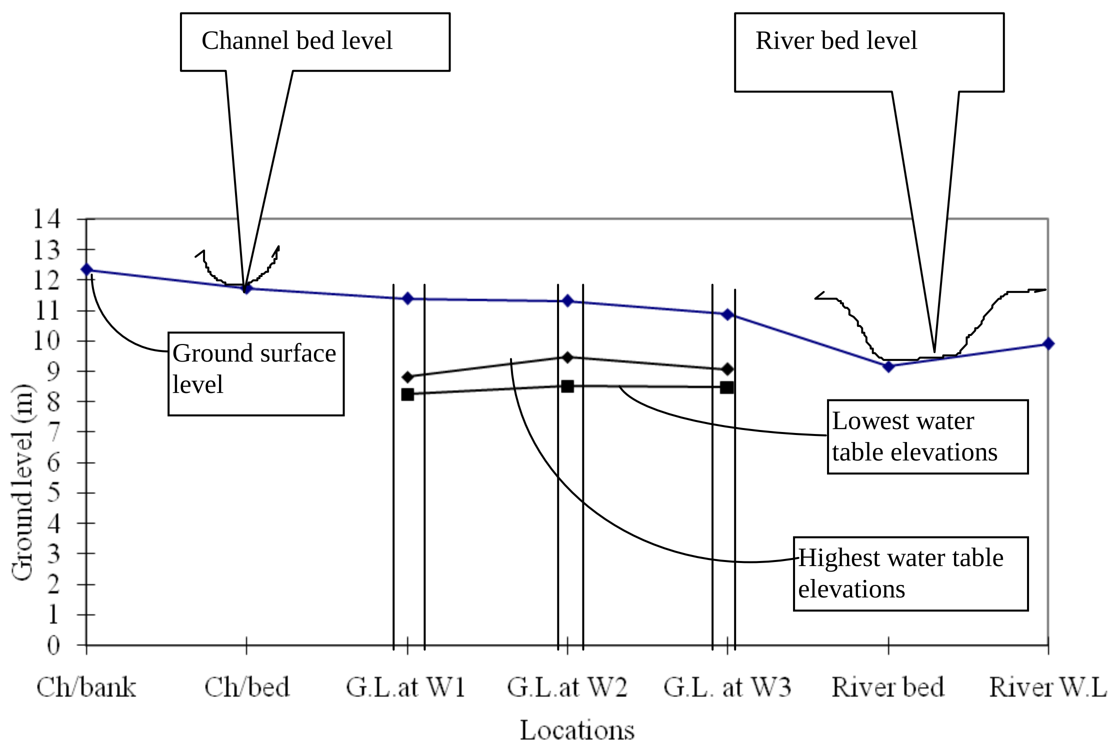


Fig. 3.10: A survey of land and groundwater table levels at the site (*G.L. stands for ground level*).

The EC and pH measurements: The EC measurements of the groundwater samples collected from wells 1, 2, 3, 4, and 5 varied between 98 and 277 μS/cm. The pH measurements of groundwater varied between 6.5 and 7.9, which is within the maximum allowable limit (<8.0) of the European Union (EU). The pH is an important variable in water quality assessment as it influences many biological and chemical processes within a water body, and all processes associated with water supply and treatment. When measuring the effects of an effluent discharge, it can be used to help determine the extent of the effluent plume in the water body (Chapman, 1997).

Groundwater N concentration: The analysis of groundwater samples taken from wells 1, 2, 3, 4 and 5 showed that 80 - 100% of the N present in groundwater samples was in the NO_3-N form. Nitrate concentrations in groundwater samples during summer and autumn remained below the maximum permissible limit (MPL) of 11.3 mg/L in all the treatments. But during winter the NO_3-N level equaled or exceeded the MPL (Fig. 3.11).

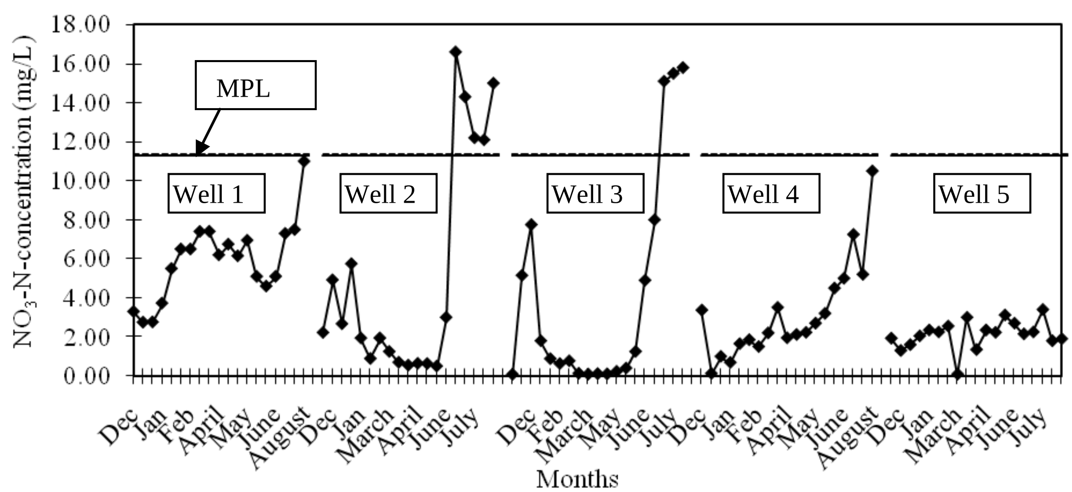


Fig. 3.11: NO_3-N concentrations in groundwater samples collected from wells 1, 2, 3, 4 and 5 at the experimental site.

Results also show that there were fluctuations in NO_3-N concentration levels that appeared to be in response to rainfall events. These are discussed below in more detail for each well.

Well 1*:* This well represented the low effluent irrigation treatment of the tree plantation area which was receiving 130 kg N/ha/38 weeks, and was located close to the oxidation pond. During summer and autumn, NO_3-N concentrations were between 2 and 8 mg/L and these concentrations were higher than the NO_3-N concentrations in the medium and high treatments. These higher NO_3-N concentrations were attributed to the seepage of effluent from the oxidation pond to well 1, due to the elevation difference between the effluent channel and groundwater table level of well 1. During winter, the NO_3-N concentrations were close to the MPL (11.3 mg/L) due to the leaching of NO_3-N from the soil-water matrix to the groundwater. The increase in NO_3-N level was attributed to the flushing through of NO_3-N accumulated in the soil.

Well 2*:* This well represented the medium effluent irrigation treatment of the tree plantation area which was receiving 1.5 times more N and water than the low treatment (i.e. 195 kg N/ha/38 weeks), and was located between the oxidation pond and the river. In this well NO_3-N concentrations were between 0 and 6 mg/L in summer and autumn, and exceeded the MPL in winter. Two high NO_3-N concentrations found during summer were in response to rainfall events that leached accumulated NO_3-N through the soil.

Well 3*:* This well was located in the tree plantation area of the high effluent irrigation treatment which was close to the river, and was receiving almost 3.5 times more N than the low treatment (i.e. 433 kg N/ha/38 weeks). During summer and autumn (when SMC was low and there was little rainfall) the NO_3-N concentrations in groundwater were very low (0 - 2 mg/L), except for one occasion when a value of 8 mg/L was observed in response to a rainfall event (Fig. 3.11). The land and groundwater leveling survey showed that seepage from the river to well 3 was likely to occur due to the elevation difference between the river and groundwater table level of well 3 (Fig. 3.10). The low NO_3-N concentrations during summer and autumn were attributed to the dilution of groundwater from the river water seepage. During winter the NO_3-N concentration exceeded the MPL and went up to 16 mg/L. The groundwater elevation in this well was at the highest level (i.e. 9.07 m) at that time.

Well 4*:* This well was located in the pasture plot receiving the high effluent irrigation treatment, and was closer to the river than well 3. The NO_3-N concentration in this well was below the MPL (i.e. 11.3 mg/L) even during winter. This was attributed to the high level of dilution of groundwater from the river water seepage because well 4 was closer to the river than well 3.

Control Well (Well 5): This well was the upstream bore (receiving no irrigation treatment), located between the oxidation pond and plantation area. The NO_3-N concentration in this well was between 0 and 4 mg/L throughout the whole duration of the trial and had the lowest NO_3-N concentration compared to the other wells.

3.2.3.8 Monitoring of Plants

There was considerable variation in tree growth and size across the trial (data not presented here) but some general trends could be observed. There was an increase in tree height, ranging from 10 to 35 cm during the 38 weeks of effluent irrigation, but no significant change was observed in tree diameter. The dry weight of leaves per tree also increased after effluent-irrigation. The combination of increased tree height and leaf mass over the period of 38 weeks resulted in an overall increase in tree biomass during the monitoring period, but there were no significant differences between irrigation treatments.

The estimated uptakes of nutrients over the period of irrigation are presented in Table 3.4 (for trees) and Table 3.5 (for pasture). There was no significant difference in foliar or wood nutrient concentrations before and after effluent irrigation, or between effluent irrigation treatments. The nutrient concentration levels found in the tree plantation and pasture areas were generally within those expected for healthy growth (e.g. Bell and Ward, 1984; McLaren and Cameron, 1990).

Over the period of the trial, N uptake by trees or pasture was between 56 and 107 kg/ha (average 78 kg N/ha) with no significant differences between trees and pasture, or between effluent irrigation rates. These values for N uptake compare with N application rates of 130, 195 and 433 kg N/ha/38 weeks period.

Table 3.4: Nutrient uptake by trees over the 38-weeks irrigation period.

Treatment	**Plot no**	**Stem uptake of nutrients (kg/ha per 38 weeks)**			**Leaf uptake of nutrients (kg/ha per 38 weeks)**			**Plant uptake of nutrients (kg/ha per 38 weeks)**		
		N	**P**	**K**	**N**	**P**	**K**	**N**	**P**	**K**
Low	p1(Nitens)	20	3	33	38	8	219	58	11	252
	p2 (Ovata)	14	4	21	66	18	195	81	22	217
	p3 (Nitens)	15	2	24	39	23	56	54	25	9
	p4 (Ovata)	18	4	27	68	27	325	86	32	352
Average								*70*	*22*	*225*
Medium	p6 (Ovata)	33	8	50	14	6	51	47	15	101
	p7 (Nitens)	6	1	9	120	39	433	126	39	442
	p8 (Ovata)	26	6	38	22	28	228	48	34	266
	p9 (Nitens)	9	1	15	65	13	312	74	15	327
Average								*74*	*26*	*284*
High	p11 Ovata)	30	8	45	75	24	244	105	32	290
	p12 Nitens)	9	1	15	101	13	167	111	14	182
	p13 Nitens)	9	1	14	98	26	370	107	27	384
	p14 Ovata)	35	9	52	71	0	64	106	29	316
Average								*107*	*26*	*293*

Table 3.5: Nutrient uptake (kg/ha) by pasture over the 38-weeks irrigation period.

Treatments	**Nutrients**	**Pasture Cut** February	August	**Total Uptake (kg/ha)**
Low	Nitrogen	51	32	83
	Phosphorus	11	8	18
	Potassium	64	37	101
Medium	Nitrogen	35	21	56
	Phosphorus	8	5.5	13.5
	Potassium	43	29	72
High	Nitrogen	38	42	80
	Phosphorus	8	11.5	19.5
	Potassium	54	64	118

3.2.4 Summary

- The trial was conducted successfully (at least hydraulically) and the site could handle the high effluent applications rates as no ponding or surface runoff was observed at the site.
- At this site it was not possible to obtain soil solution samples on a regular basis. This could not be explained. It does however place a question over the feasibility of these measurements for management purposes.
- The groundwater table was shallow at the site which varied seasonally. There is evidence that NO_3-N concentrations in the shallow groundwater beneath the effluent disposal area appeared to be affected by the effluent treatments over the course of the trial as the control well had the least NO_3-N concentrations compared to other wells throughout the trial.
- The low effluent application rate treatment increased the concentration of NO_3-N in the groundwater in the winter months, once the soil approached field capacity.

- The medium and high effluent application rates increased the concentration of NO_3-N in the groundwater in the winter months and whenever a significant rainfall event caused the soil to approach field capacity.
- In the winter the NO_3-N concentration in the groundwater exceeded the MPL value in the medium and high application rate treatments applied to the trees.
- The concentration of NO_3-N in the groundwater appeared to be affected by "leakage" from the effluent ponds and the river. The study highlighted the importance of assessing the other inputs to the groundwater at a LTS.
- Wet periods of the year are a crucial time for the manager to manage a land treatment system. During this time of the year drainage losses were observed in tree and pasture areas for all treatments. During this time the manager of the LTS may have to store the effluent for a short or longer time depending on the weather and soil hydrological conditions, or apply to a larger area (results are shown in Chapter 4).

3.2.5 Acknowledgments

I would like to express my sincere thanks to Mr. David Bridges and Ms. Bettina Anderson of *Good Earth Matters* for their interest, valuable help and guidance, and providing the financial support in conducting this experiment. Special thanks to Mr. Peter Leighton of Carterton District Council for his assistance in effluent-irrigation and site maintenance. Thanks to *NIWA* for providing data on pan evaporation.

CHAPTER 4

HYDROLOGICAL MODEL

4.1 Background

Timely prediction of water and NO_3-N movement through the soil-water matrix is an essential pre-requisite for the operation and management of a land treatment system. An exploratory field scale investigation was undertaken (Chapter 3) at a Carterton land treatment site to examine the effect on ground water from the irrigation of sewage effluent on trees and pasture at different application rates. In this study it appeared that increasing application rates of effluent may have resulted in increasing NO_3-N concentrations in the shallow groundwaters immediately below the disposal site. Based on the findings of this experiment a simple hydrological model was constructed to assist short term management decisions at a LTS. This chapter describes that model and is based largely on the published paper that is reproduced below. Once again, some changes have been made to the text of the paper to improve linkages with other chapters.

4.2 A Physical Model to Make Short Term Management Decisions at an Effluent-Irrigated Land Treatment System

by Babar Mahmood, Gavin L. Wall, and John Russell

Agricultural Water Management, 2003, 58, 55-65.

4.2.1 Introduction

The prime objective of management of a LTS is to apply effluent to the pasture or plantation in as close as possible to optimum quantities so that the receiving water bodies are not going to be contaminated. Groundwater is a resource of significant social value - people rely on it for sustaining life and livelihoods. Recognition that this resource is subject to degradation has heightened interest in the formulation of management practices to deal with the short-term and longer-term effects of land application of wastewater on groundwater quality at a LTS.

Management of a LTS includes the scheduling of effluent irrigation (i.e. the amount and timing or the frequency of irrigation). Soil characteristics as well as the plants' capacity to take up the effluent will influence effluent irrigation frequency and duration. Management must avoid creating a risk of run-off, erosion, soil damage or unacceptable leaching. Other factors (e.g. a need to avoid reductions in wood quality due to excessively rapid growth) may be additional considerations. The design and management of land treatment systems should focus on ensuring that they retain their ability to deal with effluent over the long term and have minimal adverse environmental impacts. Particularly important goals are maintaining the quality of ground and surface water resources and avoiding soil contamination or structural change that could reduce plant growth, or constrain land use in the future.

To operate a LTS within its safe limits, it is usually necessary to make some site specific measurements that then enable management decisions as to when (what time of the year) and how much effluent (mm or kg/ha/year) to apply to the land. The management issues associated with the operation of a LTS include the ability to check the short term (daily or weekly) and longer-term (monthly or seasonal) effects of land application of wastewater on groundwater quality in terms of NO_3-N contamination.

Some field scale studies have investigated the impact of effluent leakage from a LTS into groundwater, and parameters to consider for groundwater quality monitoring at a LTS (e.g. Mahmood and Wall, 2001; Rosen, 1995). But little effort has been made to find a relationship between the information produced through the analysis of the SSF and groundwater quality monitoring data at a LTS. The basic goal of monitoring is to provide a check on the potential contamination of groundwater at a LTS, but the circumstances surrounding the need to monitor may vary from site to site. Historically, it has been difficult to demonstrate the relationship between SSF and groundwater quality changes, at least in part because of a lack of well designed water quality and land treatment monitoring efforts. Overcash and Pal (1979) reported that 90% of the failures in land effluent application systems can be directly attributed to the neglect of land treatment site data (including information on soils, climate, crops and hydrology) in the system design.

This paper describes (a) the management issues at a LTS (as highlighted in the above section), (b) the approach used to establish a physical model to make short term management decisions at a LTS, and (c) to provide an overview of the importance and practical feasibility of measuring SSFs and groundwater quality at a LTS.

4.2.2 Physical Model Concept

The rationale underpinning this research was that short-term (daily or weekly) management decisions can be made in response to appropriate monitoring of SSF at a LTS. Of particular concern is the desire to delay leaching of NO_3-N as long as possible to maximize the opportunity for plants growing on the LTS to take up the NO_3-N derived from effluent. This situation applies particularly in the spring/summer/autumn period when careful management should be able to restrict the leaching of NO_3-N. The longer-term (monthly or yearly) management issues associated with designing and operating a LTS are considered in Chapter 5. The proposal is that short-term effects of land application of the planned irrigation volume can be managed and minimized by simply monitoring the SMC and the possibility of rainfall in the immediate future at a LTS (Fig. 4.1).

For each day the irrigation is planned, it is necessary to decide whether the irrigation volume can be safely applied. To do this the current SMC is measured (to check the ability of the soil to absorb the planned irrigation) and examine the possibility of rainfall in the immediate future is determined. If the soil can absorb the planned irrigation volume and the possibility of rainfall is low (drainage is very unlikely to occur and there is no risk of NO_3-N leakage), then the planned irrigation volume may be applied. If the soil cannot absorb the planned irrigation volume (drainage and NO_3-N leakage is likely to occur), then the planned irrigation volume is reduced and the process repeated (Fig. 4.1).

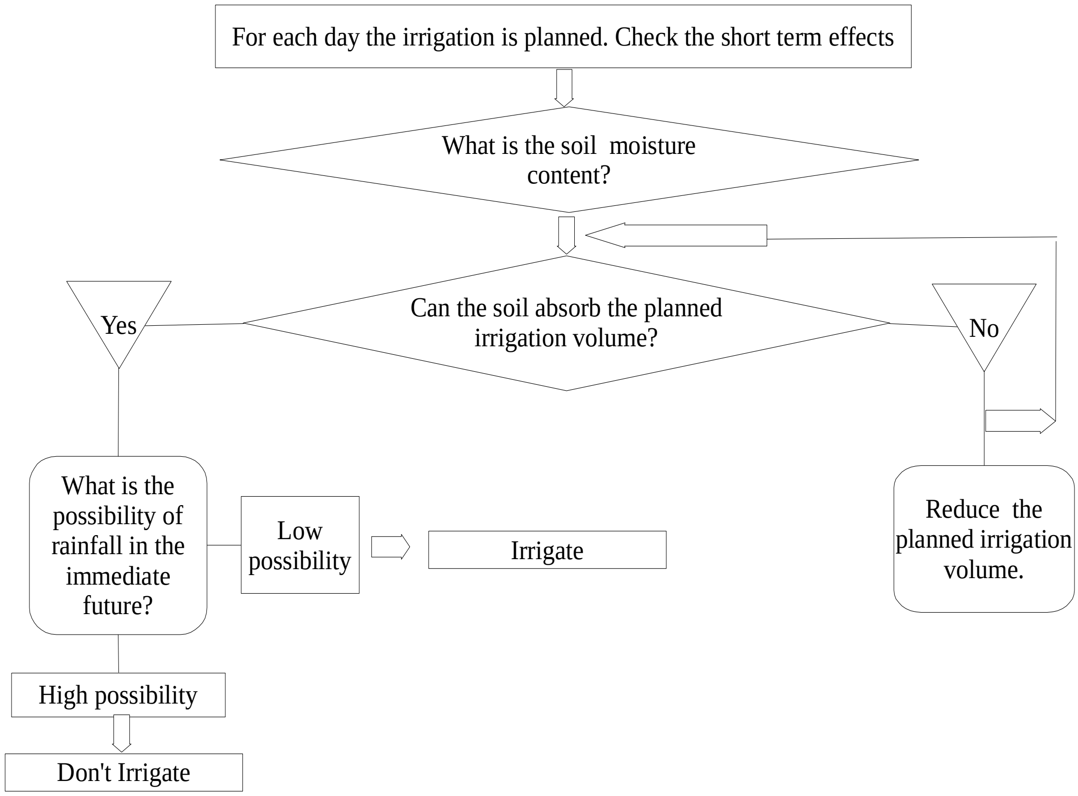


Fig. 4.1: A physical model to make short term management decisions at a LTS.

4.2.3 Data to Support the Concept

This concept was tested using data from the field study at the Carterton land treatment site.

4.2.3.1 Water Balance Approach to Estimate Drainage Losses

The components of the water balance that were monitored at the site were:

- rainfall
- volume of effluent irrigation applied
- stored soil moisture (through soil moisture content measurements)
- pan evaporation data (collected from a weather station close to the site)

The following equation was used to estimate the drainage losses between times t1 and t2. This water balance was calculated biweekly.

$$Drainage\ loss = SSM_{t2} - SSM_{t1} + IRR\ (net) + PPT\ (net) - AET\ (net) \qquad (4.1)$$

Where,

SSM_{t1} = Stored Soil moisture on day t1 (mm)

SSM_{t2} = Stored Soil moisture on day t2 (mm)

IRR (net) = Net irrigation applied during time t1 to t2 (mm)

PPT (net) = Total rainfall that occurred during time t1 to t2 (mm)

AET (net) = Estimated actual water use by trees or pasture during time t1 to t2 (mm)

Drainage losses from the tree plantation area: The drainage loss estimations for the *Eucalyptus* tree plantation area were made using Equation 4.1. The results (Fig. 4.2) indicate that there would be a marked irrigation treatment effect on the estimated drainage losses i.e. the volume of drainage water (mm) increased with the increase of irrigation volume. There were no predicted drainage losses in the L and M treatments during summer and early autumn (December - March) - except for a brief event in early December when there was 67 mm of rain (Fig. 4.2). From late autumn to early spring (April - August), there were predicted drainage losses in the L and M treatments because of high rainfall, and soil moisture contents that were close to FC. There were predicted drainage losses in the H treatment throughout the irrigation period. These were attributed to the rainfall events plus the high effluent application rate. The small negative drainage values over summer in the L and M treatments (Fig. 4.2), result from over-estimation of AET from dry soils, and can be considered to represent nil drainage.

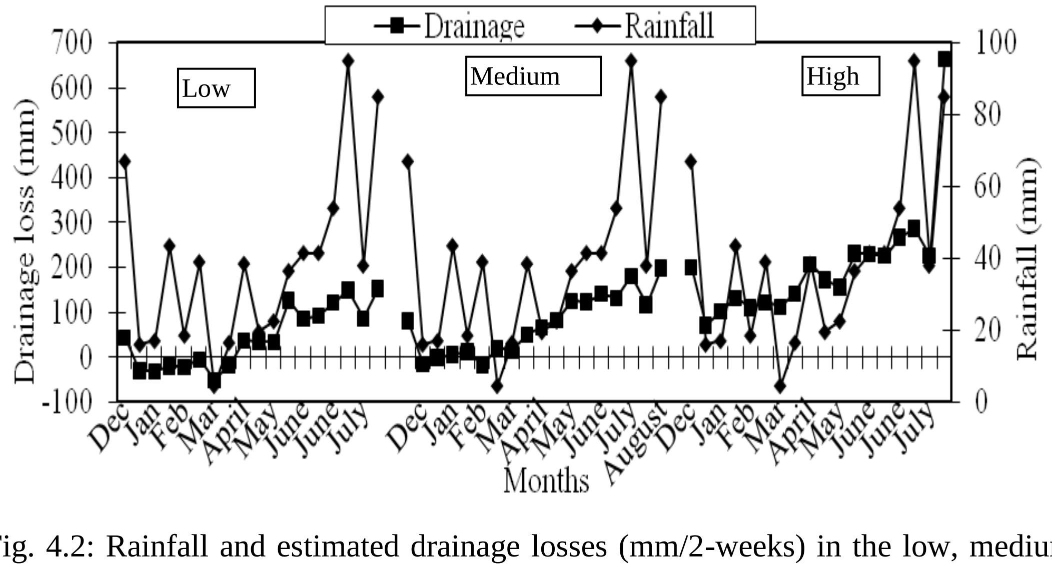


Fig. 4.2: Rainfall and estimated drainage losses (mm/2-weeks) in the low, medium and high effluent irrigation treatments on the tree plantation area.

It is important to note that this water balance approach of estimating drainage losses may result in an under-estimate, because there might be some drainage occurring from preferential flow of effluent during rain-free periods. Nevertheless, it provides a useful first approximation and was used to estimate the drainage losses in all treatment areas. For this particular year, the water balance suggested that up to 45 mm of effluent could be applied to the tree plantation area per week during summer and early autumn without causing drainage that could carry NO_3-N beyond the reach of plant roots. During late autumn and winter leaching was predicted to occur – even in the L treatment.

Drainage losses from the pasture area: The estimated drainage losses from all treatments on the pasture area are shown in Fig. 4.3. Once again there was a predicted irrigation treatment effect on the drainage losses but in all cases the predicted leaching losses from the pasture plots (Fig. 4.3) were higher than from the tree plots (Fig. 4.2). During summer there were only small drainage losses from the L irrigation treatment, and during this time effluent could be applied at a rate of 30 mm/week to pasture without having a risk of major drainage losses, depending again on the soil hydrological and climatic conditions. During autumn and winter, when SMC was high and there were frequent rainfall events, drainage is very likely to occur irrespective of the rate of effluent application.

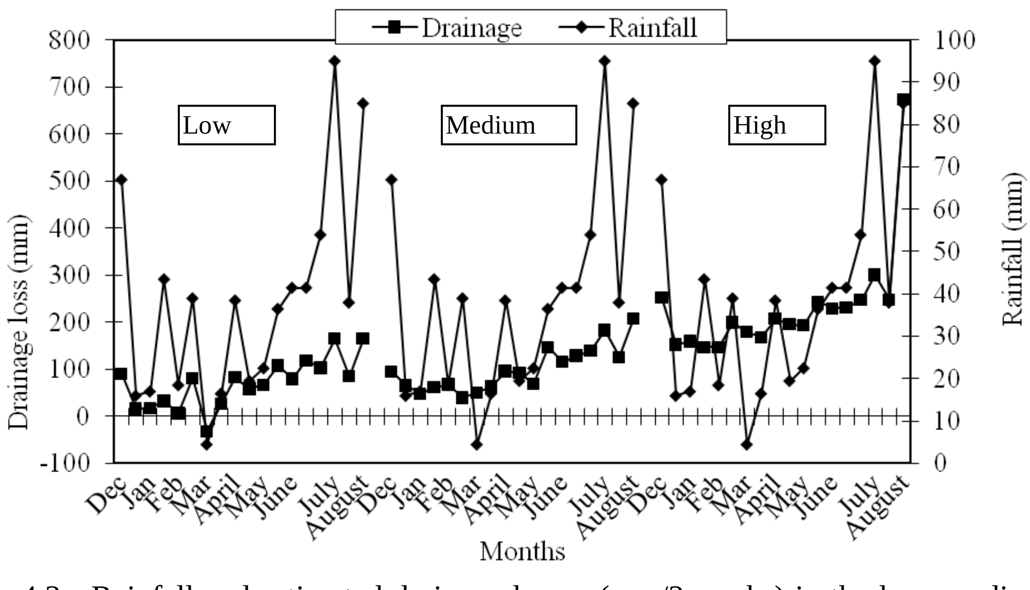


Fig. 4.3**:** Rainfall and estimated drainage losses (mm/2-weeks) in the low, medium and high effluent irrigation treatments on the pasture area.

SMC and NO_3-N concentration in groundwater - Figure 4.4 shows the pattern of SMC and groundwater NO_3-N concentration in the L, M and H effluent irrigation treatments on the tree plantation areas. During summer and autumn, when there was little rain and the SMC was low (10 - 25%) in all treatments of the tree plantation area, the NO_3-N concentration in groundwater was below the MPL (i.e. 11.3 mg/L). During this time, the fortnightly drainage losses were also low (0 - 40 mm, 0 - 77 mm and 50 - 205 mm in the L, M and H effluent irrigation treatments, respectively – see Figure 4.3).

During winter, when rainfall was higher and the soil moisture content was between 32 and 35% (i.e. close to FC) in all treatments of the tree plantation area, the NO_3-N concentration in the shallow groundwater immediately beneath the plots exceeded the MPL. This increase in NO_3-N concentration was attributed to leaching occurring in response to rainfall when the SMC was high.

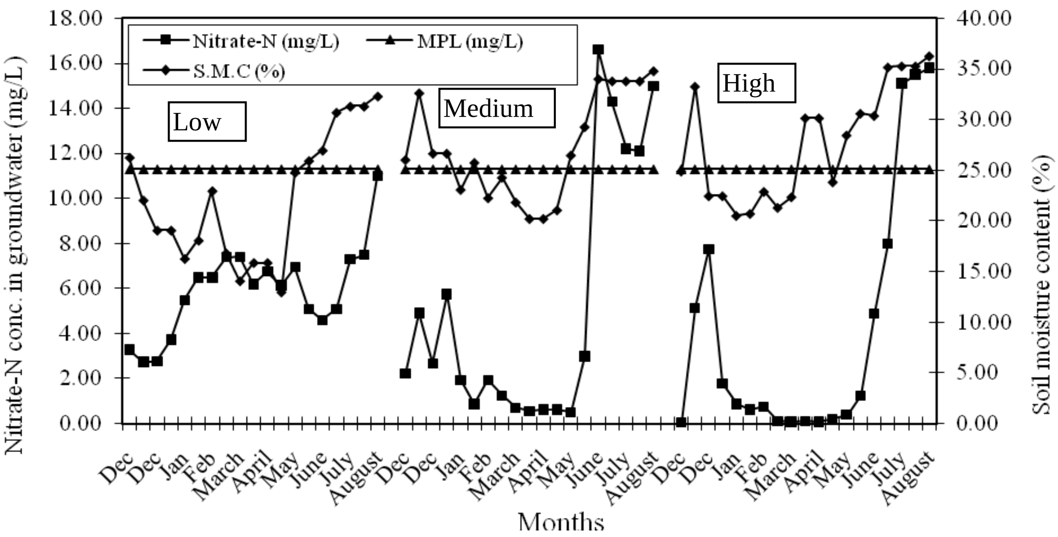


Fig. 4.4: Soil moisture content, NO_3-N concentration in groundwater and the maximum permissible limit for NO_3-N concentration in groundwater for the tree plantation receiving different rates of effluent irrigation.

This exercise helped in understanding the key mechanism governing the distribution and transportation of N through the soil-water matrix into the groundwater at this site. In summary, monitoring of SMC along with predicted rainfall would enable the manager of a LTS to predict early the NO_3-N leakage at the site and make short term management decisions to minimize this.

4.2.4 An Assessment of the Importance and Practical Feasibility of Monitoring Site Specific Factors

4.2.4.1 Monitoring of Climatic Factors (Rainfall and Evapotranspiration)

The field experiment showed that the measurement of rainfall and ET was practicable and feasible. Results can be found in Chapter 3. The ET data was not measured at the site but was estimated from the pan evaporation data collected from a nearby site. This appeared to be a feasible and satisfactory alternative to the collection of ET data at the site itself. The rainfall often controls the maximum amount of effluent that can be applied without overloading the hydrological capacity of the soil or reducing the effluent renovation processes. Also rainfall is partially responsible for the hydrological characteristics of a LTS, and thus influences the degree to which many soil and soil-plant processes occur.

Measurements of the rainfall, ET data and the applied irrigation volume (being the input parameters of the water balance) were necessary in order to estimate the drainage losses at the site under different water and crop regimes. At this site the high concentrations of NO_3-N in groundwater appeared to coincide with the occurrence of drainage.

4.2.4.2 Monitoring of Soil Factors

Soil moisture level: Monitoring of SMC was important to know how the soil hydrological conditions change with effluent irrigation and rainfall events. Monitoring of SMC was also important to determine the soil moisture storage which was an input parameter to the water balance calculation. The study showed that monitoring of SMC using TDR was practicable and feasible at the site.

Soil nutrient level: Measuring the concentration of soil nutrients (N, P) before and at the conclusion of the experiment was important to enable a nutrient balance to be estimated, and to provide information on the renovation processes taking place in the soil (data not presented here). However, field experience of measuring the soil nutrient levels showed that it was not practicable and feasible to measure soil nutrient levels at the site on a regular basis as the large number of samples required to counter soil variability, and the stony nature of the soil made soil sampling a laborious and time consuming job.

Soil pH, temperature, bulk density, and infiltration capacity: Field experience of monitoring soil pH, temperature, bulk density and infiltration capacity showed that monitoring soil temperature on a daily basis was very practicable and feasible, but that although it was very important to monitor changes in bulk density and infiltration capacity over time, it would not be feasible to monitor these more than 2 or 3 times a year. This is because of the large number of samples required to counter soil variability and the time-consuming nature of the sampling procedure. Soil bulk density is very important because it reflects the soil structure and the pore size distribution. The pore size distribution determines the soil moisture characteristics, or moisture release curve (Gradwell and Birrell, 1979). Bulk density was measured before and at the conclusion of the experiment.

4.2.4.3 Monitoring of Water Quality

Monitoring of water quality generally involves the assessment of the quality of waters (effluent and natural) entering, within, and leaving a LTS (McMahon and Thorn, 1990).

Effluent monitoring: Effluent monitoring was done to record what was being applied and how nutrient concentrations fluctuated seasonally. The field study showed that effluent samples could easily be taken and sent to the laboratory for chemical analysis. The study demonstrates that monitoring the effluent quality is important in determining the nutrient loading rates and the seasonal variation of nutrient concentrations throughout the experiment.

Soil water: Several methods are available for measuring the movement of water and solutes in the soil profile. The detail of these methods can be found in Burt et al.

(1993). Addiscott (1990) have reviewed the different techniques. In general there is no single preferred technique, as each method involves different degrees of effort, and each is, to some extent, unsatisfactory. The choice of any one method therefore remains to a degree a compromise between what is ideal and what is practical. In this study, porous ceramic cup samplers were used to extract water from the soil by applying suction, so that solute concentrations in the soil water could be analysed using normal laboratory methods. However, in the field experiment (Chapter 3) the difficulty in obtaining sufficient numbers of soil solution samples cast doubts on the suitability of this technique for routine monitoring at a LTS. There may also be problems with the use of porous ceramic cups (Addiscott, 1990), that are related to the nature of the soil pores that are sampled and the degree to which the water sampled is representative of all the water in the soil and from management point of view, it is too late to take any effective management action by the time the information regarding the nutrient concentration level in the soil water samples is received.

Groundwater: Monitoring of groundwater beneath effluent irrigation sites is an essential indicator of environmental performance (Bond et al., 1998). The field experiment showed that the groundwater samples can easily be taken and sent to the laboratory for the chemical analysis of nutrients, but once the groundwater is contaminated it is too late to rectify the problem in the short term.

The field scale study demonstrates that the knowledge of groundwater conditions and how these conditions change as a result of climate, SMC, and effluent irrigation is necessary for the effective management of LTS, and confirms that monitoring of groundwater within the vicinity of a LTS may provide a useful means of assessing the impacts of effluent irrigation on the local environment (McMahon and Thorn, 1990). In the study reported in Chapter 3 the effluent irrigation was stopped at the site following the information on groundwater NO_3-N concentrations becoming available.

4.2.5 Summary and Conclusions

The rationale for this research was that monitoring of SSF is necessary for the effective control over the impact of leached NO_3-N on groundwater quality. Information received from the analysis and interpretation of SMC, rainfall, and groundwater quality data was helpful in understanding the distribution and movement of N through the soil-water matrix into groundwater at the site. The results of the study showed that short term management decisions can be made by monitoring the capacity of the soil to absorb the designed hydraulic loading rate (by calculating the difference between the SMC of a particular day and the field capacity of the soil) and the possibility of rainfall occurring in the immediate future. The study showed that when the SMC approached a level which was close to or at field capacity, further rainfall caused NO_3-N leakage to occur and resulted in the MPL of 11.3 mg/L being reached or exceeded in the shallow groundwater at this site.

Monitoring the ability of soil to absorb the designed irrigation volume and likely rainfall in the immediate future is a cost-effective approach to estimate the potential for NO_3-N leakage through the soil-water matrix into groundwater. The approach is cost effective in the sense that little time is required to regularly monitor the SMC, and it is easy to check the possibility of rainfall in the near future. If this is done, continuous monitoring of groundwater may not be required, and money can be saved from the laboratory analysis of groundwater samples. Groundwater samples can be taken three or four times a year as a check.

CHAPTER 5

LABORATORY SCALE APPLICATION OF THE DECISION SUPPORT MODEL

5.1 Background

In previous chapters a field experiment was carried out to investigate the leaching of NO_3-N from a LTS and a simple hydrological model was developed to describe when such leaching was likely to occur. This Chapter describes a lysimeter experiment, conducted in a glasshouse that investigated leaching of NO_3-N from applied dairy shed effluent. The data from this experiment were then used to calibrate and test an already-published simulation model (LEACHN). Finally, an evaluation was made of the utility of this model as a decision support tool in designing effluent application strategies that will minimise leaching of NO_3-N at a LTS. This Chapter is based on an already published paper[1] (Appendix 4), but a number of modifications have been made to the original text of the paper to allow better integration with the rest of the thesis.

5.2 Introduction

A major concern in New Zealand is the potential impact of wastewater application to land on groundwater quality (Di et al., 1998). The concern is that land application of wastes at excessive rates may cause leaching of NO_3-N and groundwater contamination. One management tool for the prevention of NO_3-N contamination of groundwater at a LTS is the use of solute transport models to assess the possibility of NO_3-N leaching at the designed effluent irrigation volume. Computer simulation models have become popular tools for studying the transport mechanisms of agricultural chemicals through subsurface zones. Simulation studies can be used as an inexpensive, time saving, and environmentally safe technique to evaluate the effects of

[1]Mahmood, B., Wall, G., and Russell, J. (2002). A new management technique to reduce the risk of groundwater nitrate contamination at a land treatment system (LTS). *International Agricultural Engineering Journal*, 11 (2&3), 157-172.

various agricultural management practices on the subsurface movement of NO_3-N (Singh and Kanwar, 1995). Over the past several years, a large number of computer simulation models have been developed. But few data sets are available for testing a range of models, few models have been tested on a range of soils, and very few models have much demonstrable ability to simulate transient field leaching conditions (Addiscott and Wagenet, 1985). Few of the models developed so far are suitable for solving management related problems because of the lack of thorough validation (de Willigen et al., 1990).

The behaviour of N in the soil-plant-water system is very complex. It is dynamic and involves numerous interactions and transformations. Models are useful tools for integrating the different processes involved in N transport in soil, and can be used in forecasting how a system will behave without actually making measurements in the physical system. Since early ninteens, the development and application of models to predict soil water transport, N transformations, and N transport have increased tremendously (Ahmed et al., 1994). LEACHN is one such model, which can be used to simulate field-scale N transformations and movement in the unsaturated zone of the soil profile. The detailed description of LEACHN is included in one of the peer-reviewed published papers (Appendix 5), and is one of the five sub-models of LEACHM developed by the Department of Agronomy at Cornell University, USA. One particular application of LEACHN is the simulation of the leaching of NO_3-N and NH_4-N from the plant root zone. Leaching of N is of the concern because once beyond the root zone, N is no longer available to plants and thereby has the potential to pollute the groundwater.

Disposal of effluent by irrigation onto land is an increasingly popular treatment option due to concern about nutrient addition to rivers and coastal waters. Studies have shown that irrigation with wastewater can lead to contamination of groundwater resources. Therefore, there is a need for a simulation model that predicts the leaching of NO_3-N on a real time basis, in order to make management decisions about effluent application at a

LTS. There could be significant economic and environmental effects from such decisions, and it is important that they are based on scientific data.

Land treatment of wastewater should aim to produce effluent in the form of leachate from the near-surface soil-plant ecosystem, that has a low nutrient concentration and does not exceed quality parameters set by local authorities. Achievement of this performance is usually attained by controlling the input loading to the LTS. The current resource consent procedures in New Zealand recognise this level of technology by prescribing maximum allowable annual inputs of potential contaminants such as total N.

There may be a requirement to monitor the quality of groundwater likely to be affected by the land treatment of effluent, but any detected contamination indicates damage already done, and is rather too late for effective management of the LTS. Here a decision support system (DSS) based on a simulation model is presented that may assist long-term management decisions based on NO_3-N concentrations in the leachate.

5.2.1 The Management Approach

The proposed DSS was intended to assist decisions on the amount and timing of effluent irrigation by predicting the quantity and timing of future outflows (leachate volume and NO_3-N concentration) from a LTS (Fig. 5.1). This DSS takes account of the soil's ability to absorb the designed irrigation volume, the NO_3-N concentration in the leachate, and the plant's ability to take up the applied amount of water and N.

In this DSS, the LEACHN model provides the predictions on which to base a management decision. Weather data, initial water and nutrient contents (SMC and SNC) of the soil profile, and the design effluent irrigation volume are the input parameters to the model. The model then predicts SMC and SNC, AET, plant uptake of nutrients, leachate volume and nutrient concentration (mg/L) in the leachate.

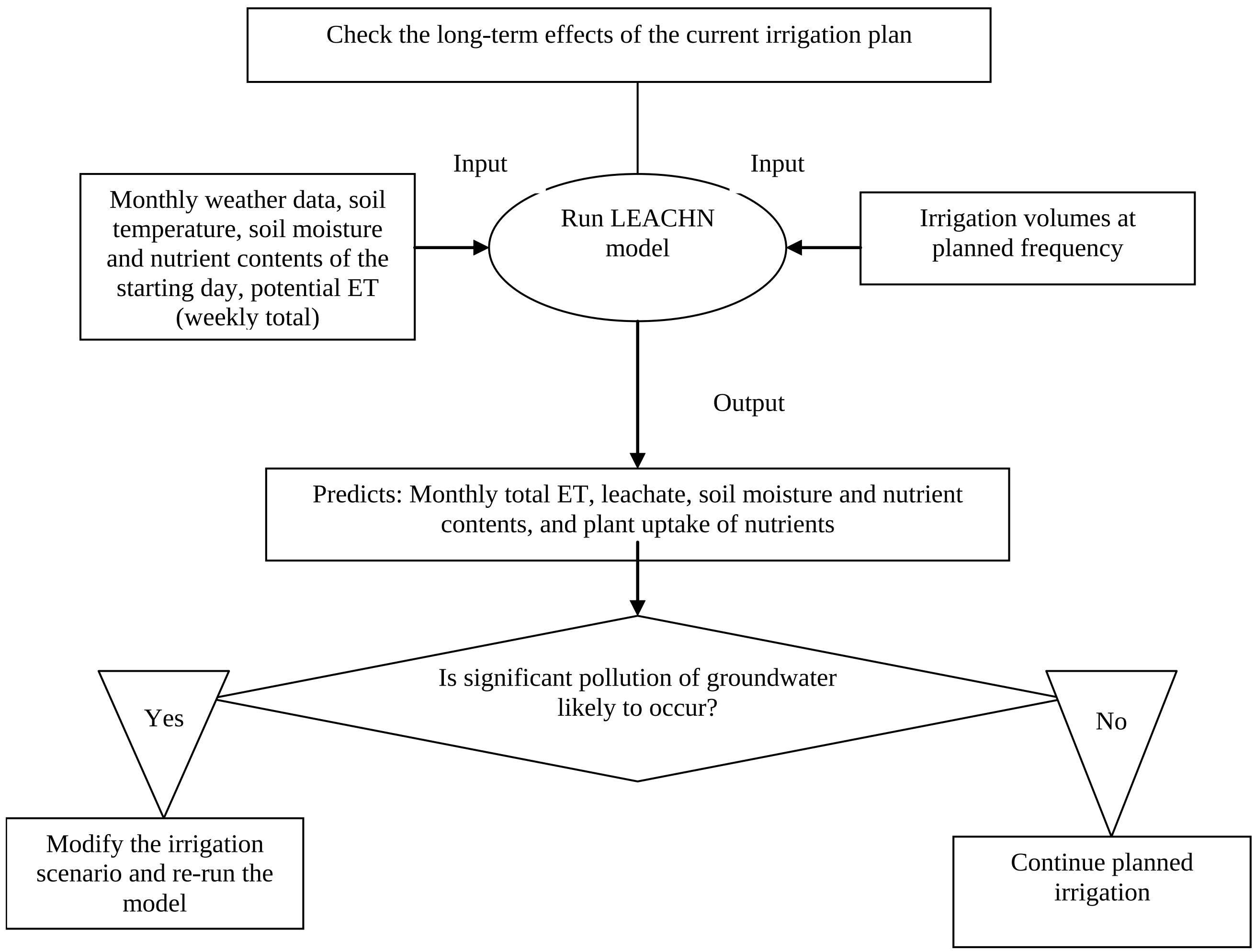


Fig. 5.1: A DSS to reduce the risk of groundwater contamination at a land treatment system.

To check the long-term effects of a proposed irrigation plan, the SMC and SNC on the starting day, along with a likely weather scenario and the proposed irrigation plan are given as the input parameters to the LEACHN model. The predicted leachate volume and the NO_3-N concentration in the leachate are then used to predict the likely impact on groundwater contamination. If the NO_3-N concentration in the leachate is below the WHO maximum permissible limit (MPL) of 11.3 mg/L in drinking water, then the planned irrigation scenario can be implemented. If the model suggests that the leachate NO_3-N concentration is likely to exceed considerably the MPL (i.e. there is a risk of unacceptable groundwater contamination), then the planned irrigation scenario can be modified and the model re-run.

Prior to applying the LEACHN model to the data from the field trial described in Chapter 3 a short term, laboratory-based lysimeter study was used to test the ability of the model to predict the leaching of NO_3-N through soil.

5.3 Methodology

Two "undisturbed" soil lysimeters were collected from a grazed pasture at Massey University. Cameron et al. (1992) have given detailed descriptions of lysimeter collection and installation. The soil was Manawatu fine sandy loam. Particle size distribution data for this soil type were obtained from the National Soils Database (NSD-Landcare Research Institute, New Zealand).

Two independent cylindrical lysimeters (A and B) of 400 mm diameter and 600 mm depth were created by gradually pushing the lysimeter casing down into the ground. The lysimeter casings were made of PVC pipe. The edge of the lysimeter casing was bevelled to a 45^0 angle using an angle grinder. The bottom 3 - 5 cm of soil in the lysimeter was replaced with gravel to hold the soil and collect the filtered leachate. The gap between the soil core and lysimeter casing was sealed using a snow-white petrolatum to prevent edge flow effects (Cameron et al., 1992).

The area of each lysimeter was 0.1256 m^2. The lysimeters were set up on 18th August 1999 in a glasshouse at the Plant Growth Unit at Massey University. Each lysimeter had pasture growing on the soil surface. Prior to the first irrigation, the pasture was cut to a height of 10 mm in each lysimeter and it was then cut again to 10 mm after 10 weeks. The temperature of the air surrounding the lysimeters varied within the range 18 - 25 ^{0}C (Peter Kemp, personal communication, 1999).

Lysimeters A and B were used to calibrate and validate the model, respectively. The calibration period, using lysimeter A, lasted 72 days (from 19th August to 29th October, 1999). Prior to the first irrigation, measurements of SMC, soil nitrogen and carbon contents, bulk density, and soil temperature were made in the top 500 mm of the soil profile for lysimeters A and B. The detail of how the soil samples were collected is given later.

Treated dairy shed effluent (DSE) (collected from the No. 4 dairy farm at Massey University) was applied at the designed rate of 3.6 mm/week on lysimeter A. The depth of effluent irrigation (3.6 mm/week) was calculated on the basis of the estimated average inorganic N concentration of the DSE (0.08 kg/m^3) and the desired N loading rate of 150 kg/ha/year - as per Wellington Regional Council Rule 11 (Annual Wellington Regional Council Plans, 1997).

DSE was applied for 5 - 10 minutes using a hand spray bottle in order to approximately simulate spray irrigation. DSE samples were collected on a weekly basis from the oxidation pond. All the effluent samples were stored at 4 ^{0}C until analysed for NO_3-N and NH_4-N using the standard methods of water analysis (Gillian, 1984). Artificial "rainfall" was applied to the top of lysimeter A, 24 hours after each DSE irrigation event (using the same hand spray bottle). Then sufficient rainfall was added to generate leachate. The amount needed was based on measurements of SMC and PET, and varied between 16 and 56 mm/week.

TDR probes were installed vertically permanently at five depths (0 - 50, 0 - 150, 0 - 250, 0 - 350, and 0 - 450 mm) in lysimeter A. Volumetric SMC measurements were made at these depths prior to irrigation to provide the initial SMC input values to LEACHN. SMC measurements were also made after the rapid drainage of soil water to estimate the FC of the soil. Soil temperature measurements were also made at the same depths prior to the first irrigation and then weekly in the top 150 mm of the soil profile using a temperature probe permanently installed in lysimeter A.

Soil core samples were collected at the same depths (i.e. 50, 150, 250, 350, and 450 mm) from the pasture adjacent to where the lysimeter samples (A and B) were taken, in order to provide initial soil nutrient values for the model. The samples were analysed for NO_3-N, NH_4-N, and organic carbon using the standard methods of Blakemore et al. (1987). The bulk density of the soil samples was determined using the standard method of Blakemore et al. (1987).

A pan evaporator was placed in the glasshouse to measure the pan evaporation rate. A crop factor of 0.75 was used to calculate the PET on a weekly basis for the pasture crop. Leachate samples were collected after each "rainfall" event. All the leachate samples were stored at 4^0 C until analysed for NO_3-N and NH_4-N using the standard methods of water analysis (Gillian, 1984).

5.3.1 Selection of Input Parameters for the LEACHN Model

The input parameters used to calibrate the nitrogen version of LEACHM are given in Table 5.1. The profile depth was divided into five equal soil segments of 100 mm thickness (i.e. the mid-points of the layers were 50, 150, 250, 350 and 450 mm below the soil surface) with a lysimeter boundary condition at the bottom.

The initial values of the parameters within the LEACHN model were those used by the developers of the model based on their experience (Hutson and Wagenet, 1992). The measured "rainfall", amount of effluent irrigation, air temperature, bulk density, and the estimated air entry value (AEV) (a), and the (BCAM) exponent (b) for Campbell's equations (Campbell, 1974) were entered into the hydrology portion of the LEACHN model. The a and b parameters of the Campbell's equation were estimated using the non-linear parameter estimation method (Wraith and Or, 1998), from the soil-water retention curve data for the Manawatu fine sandy loam soil (Clothier et al., 1977).

The soil segments were assigned the clay (%), silt (%), and inorganic N contents (mg/kg of dry soil), organic carbon content (%), SMC (%), and bulk density (kg/dm^3)

values measured prior to the first effluent irrigation. No data was collected for humus-N, as an input to the model. Values for humus-N were estimated from the initial measured organic carbon values using the C:N default ratio (10:1) in the model. Most of the N transformation, dispersivity, and hydraulic conductivity data used by the model developers were retained as no field data were obtained. The measured inorganic N (NO_3-N and NH_4-N) concentrations in the effluent samples were used in the hydrology portion of the model, along with the amount of "rainfall" and irrigation. The N concentration in the "rain water" was assumed to be zero. The weekly estimated PET values were also entered into the hydrology portion of the model, along with the mean weekly maximum/minimum air temperature values.

5.3.2 Calibration

The model was run on a weekly basis. To calibrate and then test the model the leachate data from lysimeter A was divided into two sets: data set 1 (from 19th August to 24th September, 1999) and data set 2 (from 24th September to 29th October, 1999). Data set 1 was used for the initial calibration of the model which was carried out in two stages. During the first stage the b value of the Campbell equation was adjusted for each depth segment (Table 5.1) so that the predicted soil water flow matched the measured drainage times. The resulting estimated values for the b parameter were within the acceptable range (Lesikar et al., 1997).

In the second stage, minor adjustments were made to the default values of various parameters associated with the transport and transformation of N. These included the rate constants for nitrification, denitrification and mineralization (Table 5.1), and the distribution coefficient for NO_3-N (K_d = 0.15 l/kg, Bolan et al., 1999) while the parameters calibrated in the first stage were kept unchanged. All the resulting rate constants were within the ranges reported by other researchers (e.g. Misra et al., 1974; Wagenet et al., 1977; Hagin et al., 1984; Johnsson et al., 1987; Myrold and Tiedje, 1986).

The simulation accuracy of the LEACHN model was evaluated on the basis of its ability to predict NO_3-N concentration in the leachate. When selecting the appropriate values of the various parameters the predicted and measured NO_3-N concentrations were plotted together. In addition to graphical displays of measured and predicted NO_3-N concentrations, the statistical methodologies suggested by Loague and Green (1991) were used to evaluate the prediction capabilities of the model. These statistical measures included root mean square error (RMSE) and modelling efficiency (EF) which are expressed as:

$$RMSE = [\Sigma(P-M)^2/n]^{0.5} \times 100/M_m \qquad (5.1)$$

$$EF = 1- [\Sigma(P-M)^2/[\Sigma(M- M_m)^2 \qquad (5.2)$$

where

P = predicted value,

M = measured value,

M_m = measured mean, and

n = number of observations

Once the model had been calibrated with data set 1, then data set 2 was used as inputs to the model (without changing the calibrated parameters), to compare the measured and predicted values of NO_3-N concentration in the leachate.

5.3.3 Model Testing and Simulation of Alternative Scenarios

Once the LEACHN model had been calibrated, it was then used to simulate leaching in lysimeter B. Lysimeter B had been kept in the same glasshouse as lysimeter A while the calibration experiment described above was being carried out. During this time no effluent was applied to it, and only sufficient water was added to ensure that the pasture remained alive.

After the completion of the calibration experiment on lysimeter A, DSE was applied to lysimeter B at a rate of 3.6 mm/week for 10 weeks (from 15th November 1999 to 23rd January 2000). This 10-week period was divided into a “dry” period of 5 weeks followed by a "wet" period of 5 weeks. During the "dry" period, the amount of "rainfall" applied to the lysimeter was calculated from the PET and SMC measurements (amount of rainfall varies between 0 and 8 mm/week during dry periods), and was just sufficient to keep the pasture plants growing on the lysimeter alive, and to avoid low soil moisture conditions causing cracks that would create rapid drainage. During the "wet" period the amount of "rainfall" applied to the model was greater than the measured PET (with care being taken to avoid any ponding) in order to create wet conditions and induce leaching. The day of the week to apply planned rainfall was determined using the Random Number Generator (RNG) statistical function of a spreadsheet. The amount of rainfall varied between 30 and 56 mm/week during wet periods. Three leachate samples were collected during the "wet" period and the measured and predicted NO_3-N concentration values in the leachate were compared (Fig. 5.3).

Once the calibrated LEACHN model had been tested against the experimental data from lysimeter B, 4 hypothetical scenarios, involving different effluent application strategies (Figs 5.4, 5.5, 5.6, and 5.7) were simulated. In each of these modelled scenarios the PET was kept constant throughout the trial at 20 mm/week. Simulated rainfall was the same in all four scenarios and was 8mm in week 1, 3mm in week 2, no rainfall during week 3, 4 and 5 (i.e. the dry periods), and then 30 mm/week for weeks 6-10 (i.e. the wet periods).

5.4 Results and Discussion

5.4.1 Calibration

The SMC measurements made after the rapid drainage of soil water showed that the FC of the soil was between 29 and 31%. The total N concentration in the DSE was between 30 and 120 mg/L. Almost all (>95%) of this was in the form of NH_4-N. The equivalent N loading rate over the 10 - week calibration period was 31 kg/ha (equivalent to an annual loading rate of 161 kg/ha).

Table 5.1: Parameter values utilised in the LEACHN model of the laboratory scale experiment.

Parameters	Units	Values/Range
*AEV (a)	kPa	-0.71 to -5.47 (different for each soil segment)
**BCAM (b)	dimensionless	2 - 3.17 (different for each soil segment)
Clay particles	(%)	18 - 25 (different for each soil segment)
Silt particles	(%)	31 - 47 (different for each soil segment)
Organic Carbon	(%)	1.56 - 3.16 (different for each soil segment)
Initial SMC	(%v/v)	21 - 30 (different for each soil segment)
Bulk density	(kg/dm^3)	1.14 - 1.50 (different for each soil segment)
C:N	dimensionless	10:1
Initial NH_4-N	(mg N/kg)	2 - 5.4 (different for each soil segment)
Initial NO_3-N	(mg N/kg)	0.2 - 0.7 (different for each soil segment)
PET	(mm/week)	20
Crop factor		0.75
K_d for NO_3-N	l/kg	0.15
Nitrification rate	day^{-1}	0.03 - 0.05 (different for each soil segment)
Denitrification rate	day^{-1}	0.02
Humus mineralization rate	day^{-1}	0.00004

**AEV = Air entry value (a)*
***BCAM = Exponent for Campbell's equation (b) (Campbell, 1974)*

The graphical comparison of measured and predicted values for data set 1 (Fig. 5.2) showed that the predicted NO_3-N concentrations in the leachate were in good agreement with the measured concentrations. This was to be expected because the variables in the model had been adjusted to optimise the fit between measured and predicted data for this data set. For data set 2, the model over-predicted the NO_3-N concentrations at the end of the calibration period. The maximum difference between the predicted and measured NO_3-N concentration values was 3.36 mg/L.

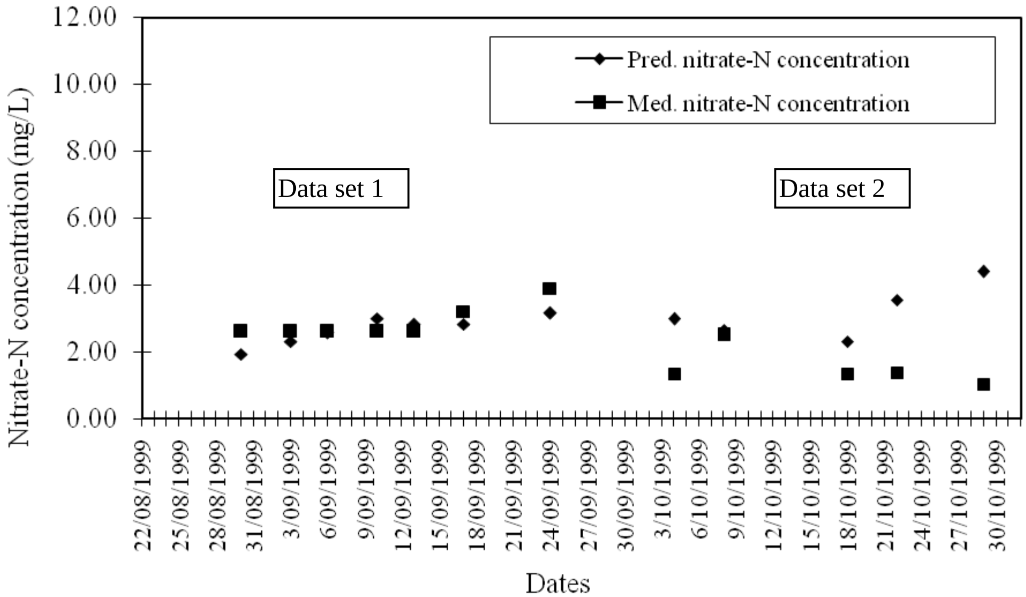


Fig. 5.2: Comparison of the measured and predicted NO_3-N concentrations in the leachate from lysimeter A.

LEACHN (version 3.1) has no capability to model macropore or preferential flow. It is possible that the large amounts of "rainfall" applied at high application rates in the last 3 weeks of the experiment may have caused preferential flow that would have resulted in incomplete leaching of the NO_3-N in the soil, and a lower than expected NO_3-N concentration in the leachate.

5.4.2 Model Testing and Simulation of Alternative Scenarios

The N loading rate in the leaching experiment using lysimeter B was 37 kg/ha/10 weeks (192 kg/ha/year). In this experiment there was no leaching during the initial 5-week "dry" period but there was significant leaching in the second 5-week "wet" period in response to heavy "rainfall" events of 48 mm, 56 mm, and 40 mm. When the data from this experiment were compared with predictions from the LEACHN (Fig. 5.3) model a similar pattern to that obtained during the initial calibration with the data from lysimeter A was obtained. The predicted NO_3-N concentrations followed a similar pattern to the measured concentrations, but the predicted concentrations were again higher than the measured values. As was the case in the earlier calibration experiment this could be due to preferential flow, resulting from the large applications of artificial “rainfall”.

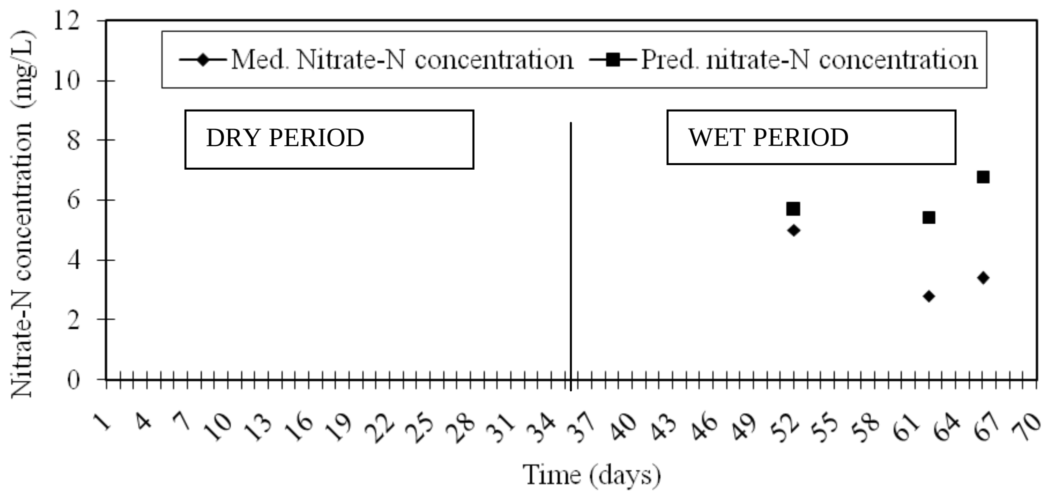


Fig. 5.3: The measured and predicted NO_3-N concentrations in the leachate from lysimeter B.

The simulation results of the four hypothetical irrigation scenarios are shown in Figs. 5.4, 5.5, 5.6, and 5.7. All scenarios assumed the same rainfall distribution with a total of 11 mm of rain during the first 5 “dry” weeks, but from weeks 6 - 10, there was 30 mm/week of rain. For all these scenarios, it was assumed that the predicted NO_3-N concentration in the leachate should not go above the MPL. If it went above the MPL then the planned irrigation volume would need to be modified.

In the first scenario, the application of effluent was increased in regular increments from 3.6 mm/week in week 1, to 19 mm/week in week 4. From weeks 5 to 10 the effluent application rate was 3.6 mm/week. The LEACHN model predicted that there would be no leaching during the dry period and the first two weeks of the wet period (Fig. 5.4). On day 52 (i.e. the 3rd week of the wet period), leaching was predicted to occur, and the predicted NO_3-N concentration in the leachate was 18.5 mg/L, which was above the MPL of 11.3 mg/L.

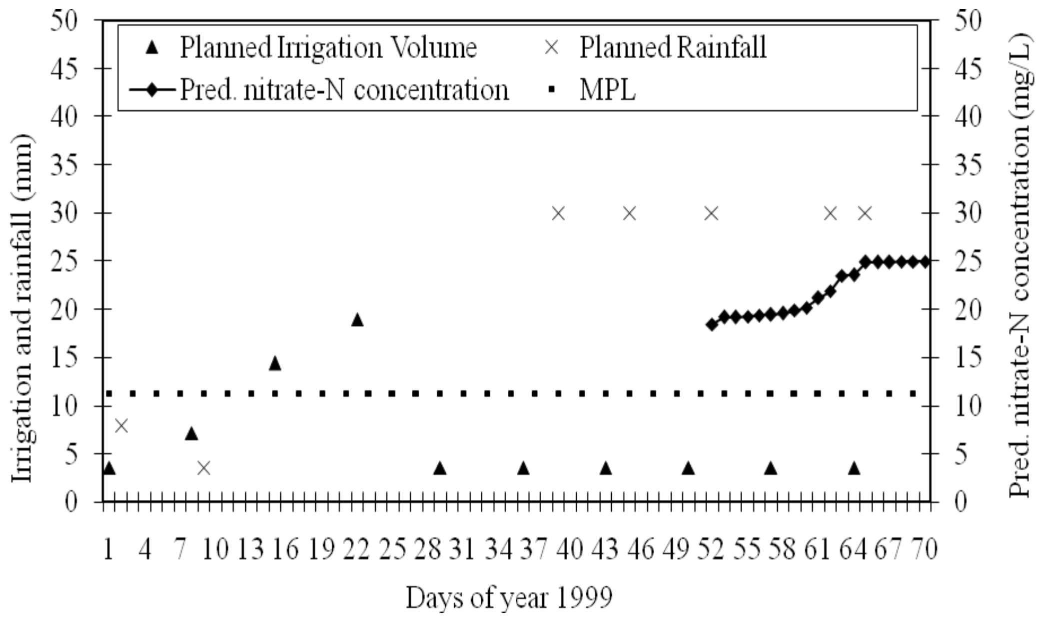


Fig. 5.4: Simulated effluent irrigation volume, rainfall, and predicted NO_3-N concentration in the leachate for scenario 1 in the assessment of the decision support model.

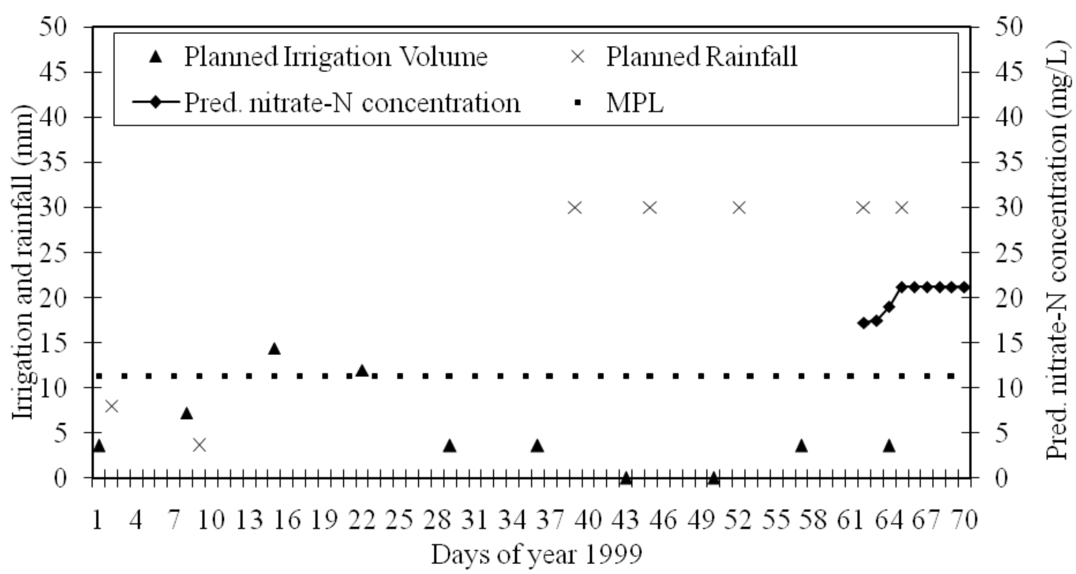


Fig. 5.5: Simulated effluent irrigation volume, rainfall, and predicted NO_3-N concentration in the leachate for scenario 2 in the assessment of the decision support model.

In the second scenario, modifications in the effluent irrigation volume were made for the 4th week of the dry period (on day 22 the irrigation volume was reduced from 19 to 12 mm/week) and the 2nd & 3rd weeks of the wet period (on days 43 and 50 the irrigation volume was reduced from 3.6 to 0.0 mm/week) (Fig. 5.5). This modification was made to reduce the risk of groundwater contamination from day 52 of the first scenario. This modification was predicted to shift the first leaching from day 52 to 62. On day 62, the predicted NO_3-N concentration in the leachate was 17.2 mg/L (Fig. 5.5).

In the third scenario, the irrigation volume was increased in a step-wise fashion from 3.6 mm/week in week 1 up to 14.4 mm/week in week 3 and then reduced to a constant value of 3.6 mm/week for the remaining 7 weeks. The LEACHN model predicted that in this scenario there would be no leaching throughout the dry period and the first three weeks of the wet period (Fig. 5.6). This means that during this time the designed irrigation volume could be applied with no immediate risk of groundwater contamination. On day 62 (4th week of the wet period), the predicted NO_3-N concentration in the leachate was 15.8 mg/L, which was above the MPL of 11.3 mg/L.

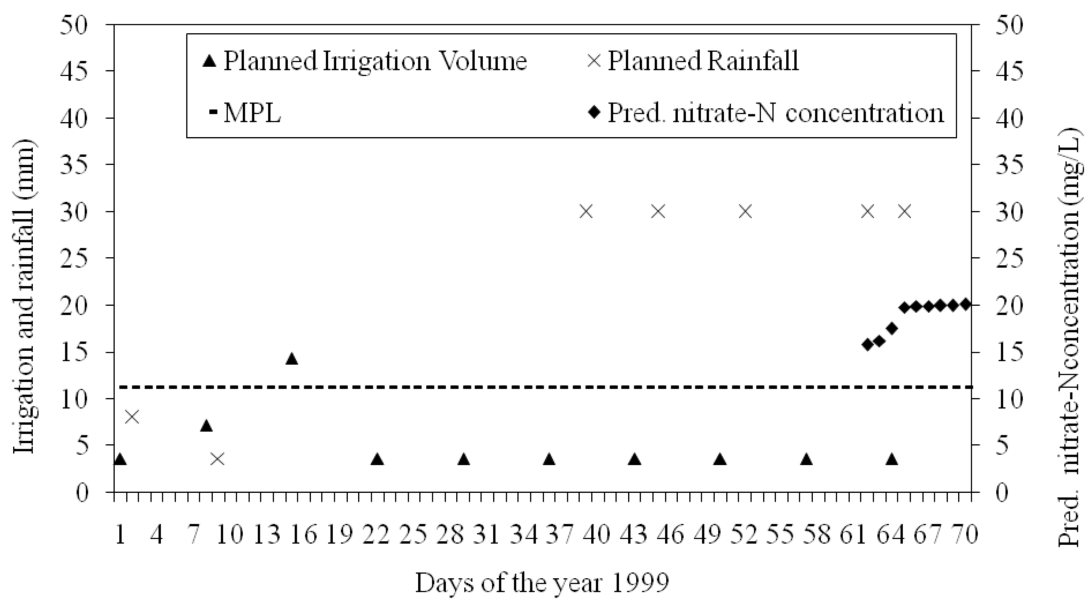


Fig. 5.6: Simulated effluent irrigation volume, rainfall, and predicted NO_3-N concentration in the leachate for scenario 3 in the assessment of the decision support model.

In the fourth scenario, the planned irrigation volume was reduced from 3.6 to 1.5 mm/week for the 4^{th} week of the wet period (Fig. 5.7). This modification was made in order to reduce the predicted risk of groundwater contamination on day 62 in scenario 3. This modification shifted the time of first leaching from day 62 to day 65. On day 65, the predicted NO_3-N concentration in the leachate was 17.1 mg/L (Fig. 5.7) which was higher than the MPL of 11.3 mg/L.

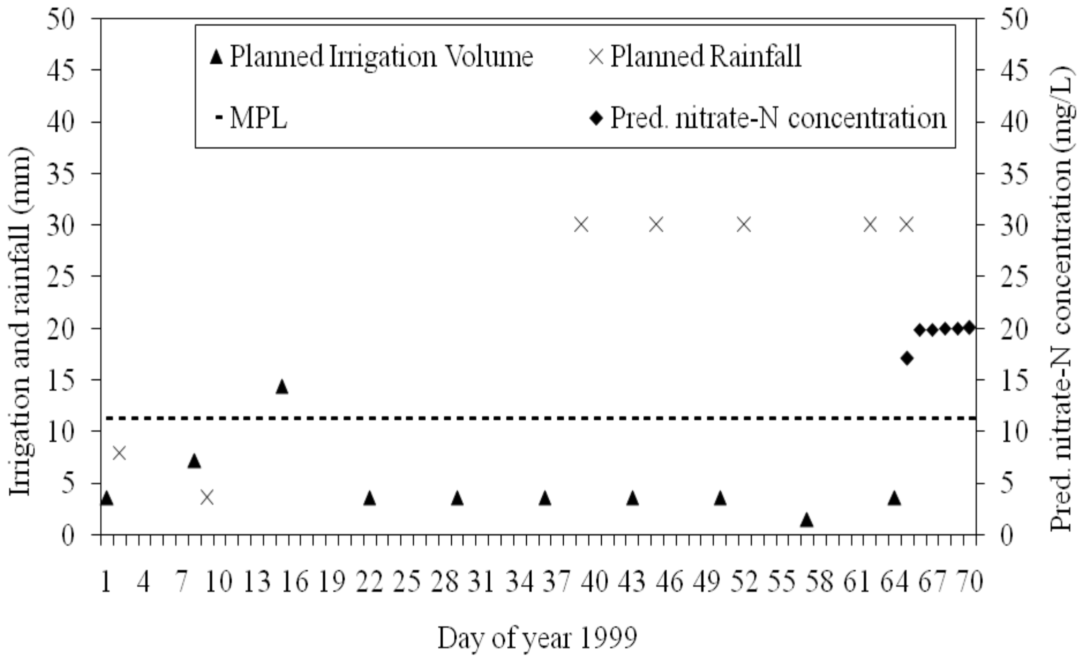


Fig. 5.7: Simulated effluent irrigation volume, rainfall, and predicted NO_3-N concentration in the leachate for scenario 4 in the assessment of the decision support model.

5.4.3 General Discussion

In this initial evaluation the LEACHN model appeared to have potential as a decision support tool to predict the leaching of NO_3-N on a real time basis for supporting and making management decisions at a LTS. This approach could facilitate the management of a LTS in a way that the risk of groundwater contamination can be reduced. This technique is conservative and safe because the long-term management decisions can be made on the basis of the predicted future NO_3-N concentration in the leachate. Whenever NO_3-N concentration in the leachate is predicted to be above the MPL, the planned irrigation volume would be reduced or modified. Using this approach to management of a LTS it might be possible to reduce the number of monitoring wells and the level of monitoring (frequency of monitoring) which means, in future, a reduction in the cost of monitoring. The initial cost may be higher but this investment may have a long term effect on the cost of monitoring.

In this demonstration of the use of the decision support model, it was assumed that whenever the concentration of NO_3-N exceeded the MPL of 11.3 mg/L there was a threat of unacceptable groundwater contamination, and the effluent application strategy was modified accordingly. In real life there may well be a dilution factor involved when the leachate reaches the groundwater. There is therefore a need to develop a relationship or dilution factor between the NO_3-N concentration in the leachate and the resulting NO_3-N concentration in the groundwater. If this is done, then the critical NO_3-N concentration in the leachate may be considerably greater than 11.3 mg/L.

The simulation of the various effluent application strategies in this study demonstrated that reducing effluent application during wet periods could delay the onset of leaching, but was unlikely to affect the NO_3-N concentration in the leachate to any great extent. This was because the NO_3-N in the leachate was already added to the soil some considerable time before the leaching took place.

5.5 Conclusions

It was possible to choose parameters for the LEACHN model that enabled it to closely reproduce the pattern of leachate NO_3-N concentrations in the dataset used for calibration. This is not surprising because LEACHN is a ‘data hungry’ model and there are many input parameters that can be adjusted in order to obtain a good match between the predicted and measured values. In this calibration process, best efforts were made to measure, estimate, or otherwise find appropriate input parameters from the literature in order to obtain a good match between the predicted and measured values.

However, when the calibrated LEACHN model was then used to predict subsequent NO_3-N leaching from the same lysimeter, or a second lysimeter, it consistently overestimated the likely NO_3-N concentration in the leachate. This may be due to preferential flow occurring through macropores in response to high rates of "rainfall" application. This would result in lower than expected NO_3-N concentrations in the leachate as much of the soil NO_3-N would remain in the micropores. If this is the case then the LEACHN model could be viewed as presenting the highest possible NO_3-N concentrations, and that in many cases non-uniform water flow through the soil will result in lower NO_3-N concentrations.

Although in percentage terms the differences between predicted and observed NO_3-N concentrations was sometimes large (11% - 48%), in actual environmental terms the differences were not so great with the model correctly predicting that the leachate NO_3-N concentrations would remain comfortably below the MPL of 11.3 mg/L.

CHAPTER 6

FIELD SCALE APPLICATION OF THE DECISION SUPPORT MODEL

6.1 Introduction

The goal of this study was to develop decision support tools that would assist engineers/managers to reduce the risk of groundwater contamination at a land treatment system. In this chapter the LEACHN model described and tested in the previous chapter is used to (1) simulate NO_3-N leaching in the low, medium, and high effluent irrigation treatments of tree and pasture plots at the Carterton land treatment site and (2) evaluate the sensitivity of the predicted leachate NO_3-N concentration to changes in model parameters. Much of the work presented in this chapter has been described previously in a published paper[2] that is included as Appendix 5. As in previous chapters, some alterations have been made to the text of the published paper, and some additional information included. .

6.2 Methodology

6.2.1 Experimental Site

Details of the experimental site, effluent application treatments and monitoring results are presented in Chapters 3 and 4.

6.2.2 Calibration of the LEACHN Model

It was necessary to modify some of the parameters in the LEACHN model described in Chapter 5 to take account of the different conditions at the Carterton LTS. The original intention was to use a subset of the soil solution NO_3-N concentrations collected from suction cups installed at the effluent disposal site for calibration purposes, and then to check the performance of the resulting model against the remainder of the field data.

[2] Mahmood, B., Russell, J., and Wall, G. (2002). Field scale nitrate simulation. *Transactions of the American Society of Agricultural Engineers (ASAE)*, 45(6), 1835-1842.

However, as noted in Chapter 3, it proved difficult to obtain suction cup samples for analysis and only a few samples were collected. Given that NO_3-N concentrations in the shallow groundwater beneath the effluent disposal area appeared to be affected by the effluent treatments over the course of the trial, it was decided to use the concentrations of NO_3-N in the groundwater from the medium treatment of the tree plots to calibrate the LEACHN model. The medium treatment of the tree plots was chosen because it was in the middle of the tree plantation area, and there would be less effect of seepage from the oxidation pond and the stream on this plot. The data from 2nd June to 18th August, 1998 (77 days) was chosen for the calibration process. The predicted leachate NO_3-N concentrations at a soil depth of 500 mm were compared with the concentration of NO_3-N in the groundwater over this time.

The small number of soil solution samples obtained from the low, medium and high effluent application treatments on both tree and pasture plots during summer (December – 1997, January – 1998), autumn (May – 1998) and winter (June and July of 1998) were then used to provide an independent check on the accuracy of the model predictions.

The input parameter values required by the model were obtained from direct field and laboratory measurements, and literature sources (Table 6.1). Estimates of the air entry value, BCAM, nitrification rate, denitrification rate, and humus mineralisation rate constants were obtained from the literature or default values in the model were used. These were then adjusted one by one within the expected range to minimise the discrepancies between the measured and predicted values.

The measured mineral N (NO_3-N and NH_4-N) concentrations in the applied effluent were entered into the hydrology portion of the model along with the measured rainfall and amount of irrigation. The N concentration in the rainwater was considered to be zero. The estimated weekly PET values were also entered into the hydrology portion of the model along with the mean weekly measured maximum/minimum air temperature values. The soil layers were assigned the measured values of silt and clay content, the initial mineral N contents, the organic carbon contents, and the SMC measured at the site. The AEV (a) and b exponent (BCAM) for Campbell's equation (Campbell, 1974) were taken from the literature

(Lesikar et al., 1997). No measurements were made of soil humus-N content and therefore the initial humus-N contents were estimated from the measured initial soil carbon pools using the C:N default ratio of 10:1 in the model. Most of the N transformation, dispersivity, and hydraulic conductivity data used by the model developers were retained as no site specific data could be obtained. Minor adjustments (from the default values) were made for parameters used in the transport and transformation of N (e.g. nitrification, denitrification and mineralization rate constants). Some of these parameters had different values at different depths down the soil profile and the range of values is shown in Table 6.1.

Once it had been calibrated and checked against the data collected from the soil solution samples, the LEACHN model was applied to the remainder of the Carterton site's data set to predict the leachate NO_3-N concentration in the unsaturated zone (0 - 500 mm soil depth) in the low, medium, and high irrigation treatments of tree and pasture plots. The designed irrigation volume and the estimated PET values for the tree and pasture crops were inputs to the model. The model was run for each irrigation treatment (low, medium, high) for both tree and pasture crops.

Table 6.1: Values assigned to parameters in the LEACHN model for the simulation of NO_3-N leaching at the Carterton LTS.

Parameters	**Source**	**Units**	**Values/Range**
*AEV (a)	literature	kPa	-1.0
**BCAM (b) l	literature	dimensionless	3 - 5 (different for each layer)
Clay content	measured	(%)	20
Silt content	measured	(%)	12
Organic carbon	measured	(%)	2.50 - 4.50 (different for each layer)
Initial SMC	measured	(%v/v)	26 - 29 (different for each layer)
Bulk density	measured	(kg/dm^3)	1.00 - 1.44 (different for each layer)
C/N	literature	dimensionless	10:1
NH_4-N	measured	(mg N/kg)	2.63 - 4.00 (different for each layer)
NO_3-N	measured	(mg N/kg)	3.85 - 6.38 (different for each layer)
Crop cover	literature		0.8
K_d for NO_3-N	literature	l/kg	0.00
Nitrification rate	literature	day^{-1}	0.02 - 0.09 (different for each layer)
Denitrification rate	literature	day^{-1}	0.02
Humus mineralisation rate	literature	day^{-1}	0.00007

**AEV = Air entry value (a)*

***BCAM = Exponent for Campbell's equation (b) (Campbell, 1974).*

6.2.3 Sensitivity Analysis

A sensitivity analysis was performed on the calibrated LEACHN model by changing the value of input parameters including the soil moisture content, the AEV (a), the BCAM exponent for Campbell's equation (b), the soil organic carbon level, the bulk density, the nitrification rate constant, the denitrification rate constant, the humus mineralisation rate constant, base temperature and Q_{10} factor. The value of each input parameter was increased by 30%. Model sensitivity to changes in these parameters was evaluated on the basis of their impact on the concentration of NO_3-N in the leachate at 500 mm depth on the 70th day. Model sensitivity to changes in parameters was evaluated only for the medium effluent treatment of the tree plots.

6.2.4 Use of the LEACHN Model to Evaluate Different Effluent Application Strategies at the Carterton Experimental Site

The model was then used to evaluate the impact of modifying the effluent application strategy on the amounts of NO_3-N leached and the NO_3-N concentration in the drainage water. The simulation focussed on the winter months on the tree treatment receiving the medium rate of effluent application. This was because the groundwater NO_3-N concentration went above 11.3 mg/L during June. Three scenarios were considered. The first scenario was the original effluent application strategy of 45 mm/week. In the second scenario the effluent application rate was reduced to 30 mm/week for 9 weeks during April and May. In the third scenario the application rate was reduced during April and May (as in scenario 2) and was then reduced still further to 25 mm/week for 5 weeks in June. In each case the model was run for the 260 days of the original field assessment at the Carterton LTS (Chapter 3). The rainfall entered into the LEACHN model was the same for all the scenarios and was the rainfall actually measured at the site over the corresponding period (Chapter 3).

6.3 Results and Discussion

6.3.1 Calibration of the LEACHN Model against Groundwater NO_3-N Concentrations

Minor changes to the b (BCAM) exponent were made to adjust the drainage time and these changes were within the range (0.14 - 13.3) as given by Lesikar et al. (1997). All the rate constants used in the model were within the ranges reported by other researchers (Hagin et al., 1984; Johnsson et al., 1987; Misra et al., 1974; Myrold and Tiedje, 1986; Wagenet et al., 1977, and John Hutson - developer of LEACHM, personal communication, 2000). The predicted leachate and measured groundwater NO_3-N concentrations are shown in Fig. 6.1.

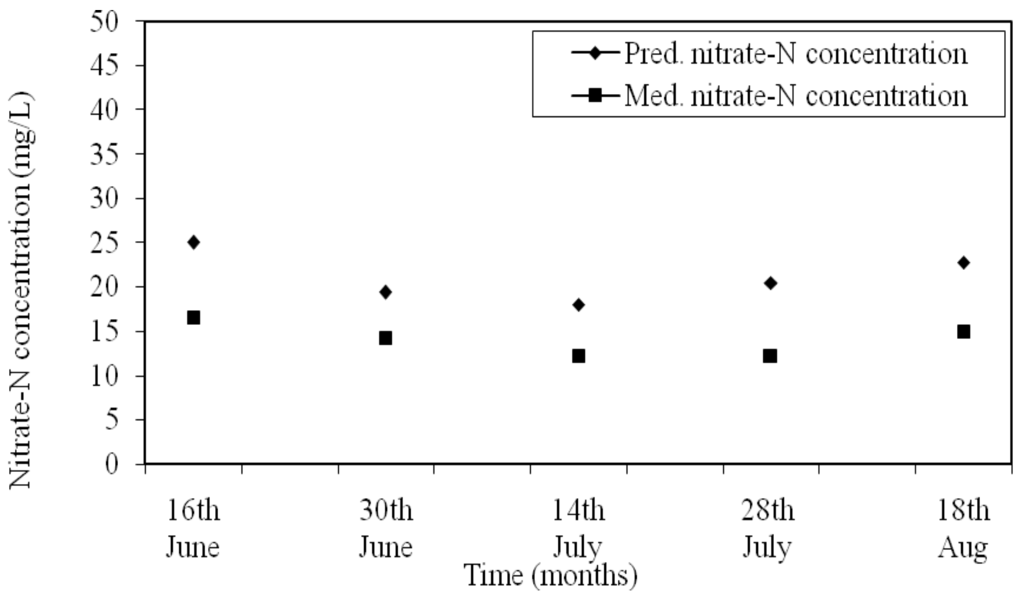


Fig. 6.1: The predicted leachate and measured groundwater NO_3-N concentrations in the tree plot receiving the medium effluent application rate.

The predicted leachate NO_3-N concentration at a depth of 500 mm varied between 18 and 25 mg/L. These predicted NO_3-N concentrations in the leachate followed the same trend as the measured groundwater NO_3-N concentrations throughout the winter period, but the predicted concentrations in the leachate were consistently higher than the measured NO_3-N values in the groundwater. To decrease the predicted concentrations would have required assigning a value to one or more of the model parameters that would have been outside the normally expected range as recorded in the literature. There was also the possibility that the NO_3-N in the groundwater had been diluted by water entering from outside the disposal area, thereby invalidating the assumption that the groundwater NO_3-N concentrations were a fair reflection of the NO_3-N concentrations in the leachate exiting the root zone. Given this uncertainty it was decided to continue to the next stage of the model evaluation with the model parameters as originally assigned. The maximum difference between the predicted and measured concentrations of NO_3-N was 8.3 mg/L.

6.3.2 Comparison of Measured and Predicted Soil Solution Nitrate Concentrations

There was a pattern of high NO_3-N concentrations in the suction cup samples early in the drainage season that tailed off to quite low concentrations as the winter progressed (Table 6.2). Although at first glance there did not appear to be a good correspondence between actual soil solution concentrations and those predicted by the LEACHN model (Table 6.2), plotting the modelled soil solution NO_3-N concentrations over the whole measurement period revealed a similar predicted seasonal pattern of soil solution NO_3-N concentrations (Fig. 6.2) to that observed in the field (Table 6.2). The high (both predicted and observed) soil solution NO_3-N concentrations in autumn and early winter result from a build-up of NO_3-N during summer when there is little or no leaching – particularly in those plots receiving low or medium rates of effluent application. This accumulated NO_3-N is then flushed through the soil-water matrix during the first drainage events, and from then on the system relies on mineralisation to supply NO_3-N for leaching. As the winter temperatures are low, mineralisation is usually slow and the resultant NO_3-N concentrations in the leachate are low.

This movement of a “front” of NO_3-N through the soil profile is further demonstrated by the predicted soil solution NO_3-N concentrations at depths from 0 to 500 mm from the ground surface on 16 June (Fig. 6.3). The LEACHN model predicted that on that date soil solution NO_3-N concentrations would increase with depth down to about 450 mm and then decrease slightly at 500 mm.

Although the LEACHN model predicted a similar seasonal pattern of NO_3-N concentrations to that observed in the field, and also correctly predicted which effluent application rates would generate the highest and the lowest NO_3-N concentrations, the actual soil solution NO_3-N concentrations were much lower than those predicted by the LEACHN model in nearly all cases (Table 6.2 and Fig. 6.2). This is consistent with the apparent over-prediction of the groundwater NO_3-N concentrations reported in the previous section. The possible reason for this is discussed in the following section and also in Chapter 8.

Table 6.2: Average measured NO_3-N concentrations in suction cup samples collected at a soil depth of 500 mm from pasture and tree plots receiving low, medium and high rates of effluent application, and predicted soil solution NO_3-N concentrations using the LEACHN model at the same soil depth on the same days.

	Time		**Measured NO_3-N Concentration (mg/L)**			**Predicted NO_3-N Concentration (mg/L)**		
Pasture	Dates	Days no.	Low	Med	High	Low	Med	High
	16/12/1997	16	34.3	15.6	10.6	33.1	27.6	17.5
	31/12/1997	31		6.5	16.93		25.4	19.5
	13/01/1998	44		6.95	9.1		34.9	26
	19/05/1998	170			7.14			16.8
	16/06/1998	198			3.9		20	17.6
	30/06/1998	212			2.6			16.2
	14/07/1998	226		2.6	2.6		16.6	16.4
	28/07/1998	240			2.6			
Trees	16/12/1997	16	19.5	10.7	8.43	57.3	39.5	23.4
	31/12/1997	31	28.6		8.45	75.9		24.5
	13/01/1998	44			11.7			34.3
	19/05/1998	170						17.7
	16/06/1998	198						18.4
	30/06/1998	212			4.3			17.7
	14/07/1998	226			4.5			22.5
	28/07/1998	240						

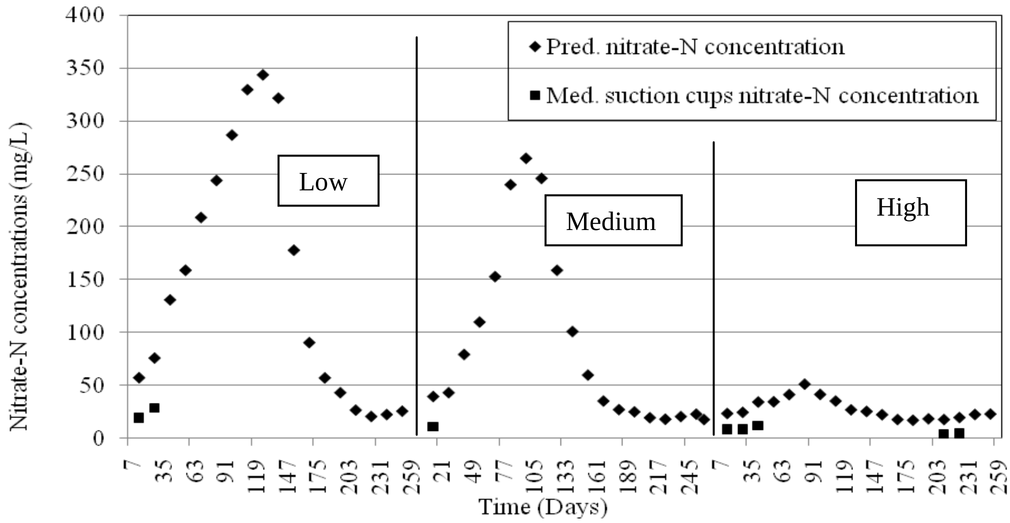


Fig. 6.2: A comparison of the measured and predicted leachate NO_3-N concentrations in the low, medium and high irrigation treaments of the tree plantation area.

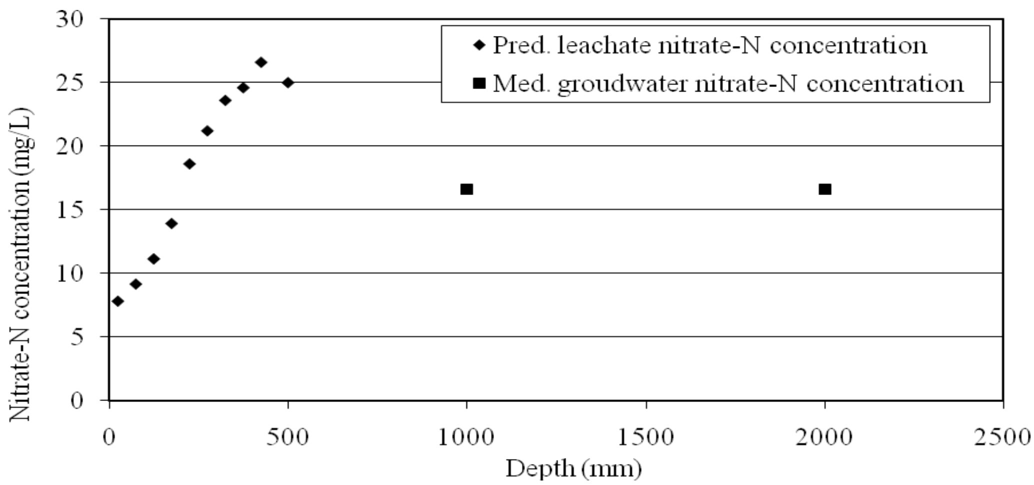


Fig. 6.3: Predicted soil solution NO_3-N concentrations down the soil profile on 16 June 1998 and the measured groundwater concentrations at two depths on the same day.

It was interesting to note that the predicted timing of the movement of the “front” of high soil solution NO_3-N concentrations below 500 mm coincided with a marked increase in groundwater NO_3-N concentrations (Fig 6.4).

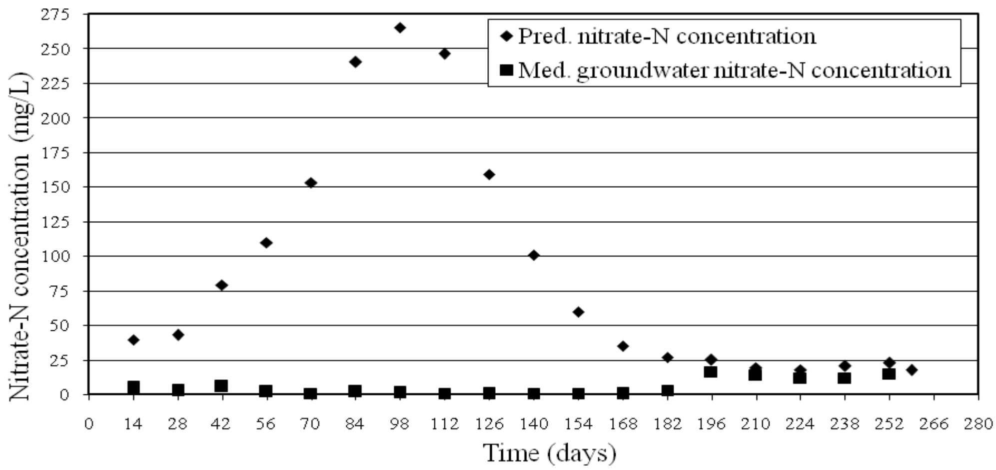


Fig. 6.4: Predicted soil solution NO_3-N concentrations at a depth of 500 mm and measured groundwater nitrate concentrations in the medium effluent application rate treatment of the tree plots (well 2).

6.3.3 Sensitivity Analysis

The model's sensitivity to hydrological and N transformation parameters was evaluated by observing the effect of increasing parameter values, one at a time, by 30% on the subsequent predictions of the NO_3-N concentration in the leachate at 500 mm depth on day 70 (Table 6.3). Changing the values for initial soil moisture content, soil organic carbon, and nitrification and denitrification rate constants had minimal effects on the predicted NO_3-N concentrations. Nitrate-N concentrations predicted with LEACHN were more sensitive to values for the bulk density, AEV (a), BCAM (b), mineralisation rate, base temperature and Q_{10} factor.

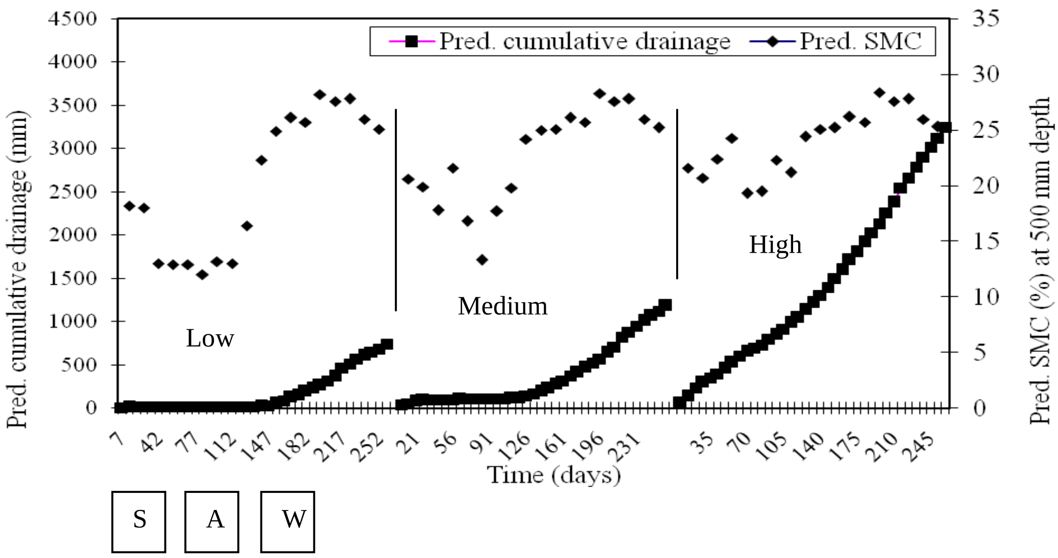
Although the model appears to very sensitive to soil bulk density, the actual bulk density was measured at the site and this should reduce errors in the model predictions due to this parameter. Of more concern is the sensitivity to the humus mineralisation rate. This is extremely difficult to measure and the default value of 0.00007 day^{-1} was used in the model. This equates to an annual mineralisation rate of the soil organic carbon (and N) of 2.55%. If this figure is too high for a site under permanent pasture or trees, it may explain the consistent over-prediction of soil solution NO_3-N concentrations by the LEACHN model in this study. This is discussed further in Chapter 8.

Table 6.3: Sensitivity of the predicted soil solution NO_3-N concentration at 500 mm depth on day 70 in the plot receiving the medium effluent application treatment to changes in model parameters. (A positive variation represents an increase in the predicted soil solution NO_3-N concentration and a negative variation represents a decrease in the predicted NO_3-N concentration).

Parameter	**% Increase in input parameter (A)**	**Output** **NO_3-N (mg/L)**	**% Change in output (B)** **NO_3-N (mg/L)**	**Sensitivity (B/A)**
Initial SMC (v/v)	30	41.6	1.2	0.04
		42.1		
AEV (b)	30	41.6	-6.7	-0.22
		38.8		
b (BCAM)	30	41.6	-8.9	-0.30
		37.9		
Bulk Density (kg/dm^3)	30	41.6	77.6	2.59
		73.9		
Organic Carbon (%)	30	41.6	1.9	0.06
		42.4		
Nitrification rate	30	41.6	1.2	0.04
		42.1		
Denitrification rate	30	41.6	-0.7	-0.024
		41.3		
Humus mineralization rate	30	41.6	18.5	0.62
		49.3		
Base temperature	30	41.60	-20.2	-0.67
		33.20		
Q_{10} factor	30	41.6	12.0	0.40
		46.6		

6.3.4 Application of the Calibrated LEACHN Model to all the Effluent Application Treatments

The LEACHN model developed in the previous section using the measured data for the medium irrigation treatment on the tree plots, was then used to simulate the leaching of NO_3-N in the other treatments. The simulations predicted that there would be a very substantial irrigation effect on the quantity of drainage (Fig. 6.5 and Table 6.4). The simulation predicted that there would be little drainage during summer and early autumn in the low and medium effluent irrigation treatments on the tree plots but that drainage would continue steadily throughout the measurement period on the plots receiving the high effluent application rate. The predicted cumulative drainage over the experimental period on the tree plots ranged from 738 mm on the plot receiving the low rate of effluent application to 3246 mm on the plot receiving the high rate of effluent application.



S=summer, A=autumn, W=winter

Fig. 6.5: Predicted cumulative drainage and soil moisture content in the low, medium, and high effluent irrigation treatments on the tree plots.

The LEACHN model predicted that during summer and autumn the NO_3-N concentrations in the soil solution at 500 mm would build up to a peak and then drop away again (Fig. 6.6). As discussed in the previous section, this results from the passage of a "front" of NO_3-N through the soil. The predicted NO_3-N concentrations in the soil solution at 500 mm were very high - ranging from 17 to 350 mg/L in the low and medium effluent irrigation treatments on the tree plots (Fig. 6.6). This high NO_3-N concentration resulted from the build-up of NO_3-N in the soil from ongoing applications of effluent and the low volumes of drainage water over these months. In contrast, during late autumn and winter, the predicted NO_3-N concentrations in the soil solution were lower (17 - 24 mg/L) in all treatments on the tree plots.

It is interesting to note that both the measured and predicted NO_3-N concentrations in the soil solution were inversely related to the effluent application rate. This is because the increased volume of effluent applied at the high application rate was more than sufficient to dilute the increased quantities of NO_3-N in the soil profile. The predicted total amounts of N leached over the simulation period however, did increase with increasing effluent application rate (Table 6.4).

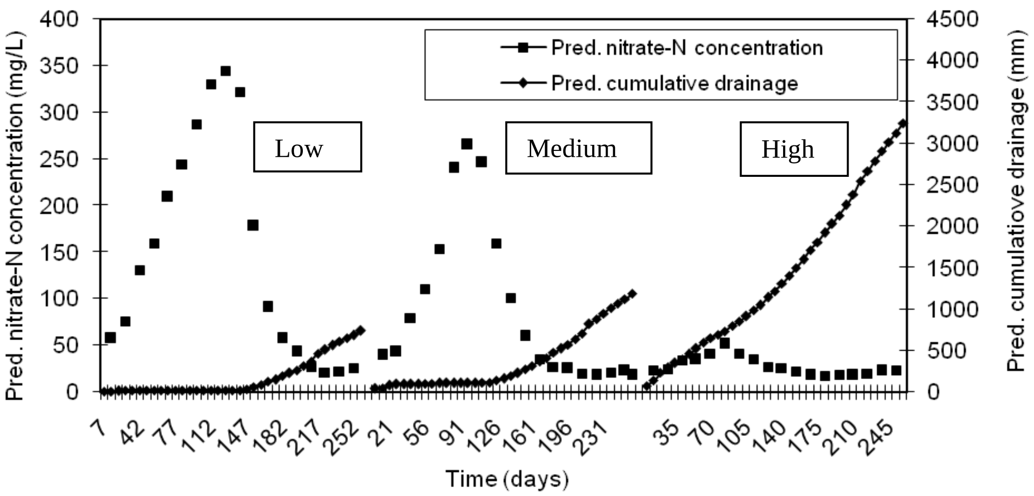


Fig. 6.6: Predicted cumulative drainage and leachate NO_3-N concentration at 500 mm soil depth for all treatments of tree plots.

Similar reasoning can explain the predicted differences in NO_3-N concentrations between the pasture plots and the tree plots. During summer and early autumn, the predicted soil solution NO_3-N concentrations were between 32 and 100 mg/L in the low and medium treatments of the pasture plots (Fig. 6.7). These concentrations were considerably lower than the comparable concentrations in the tree plots (Fig. 6.6).

Table 6.4: Predicted accumulated leachate and NO_3-N leached over the simulated period on tree and pasture plots receiving three rates of effluent application.

Plant Species	**Effluent Application rate**	**Drainage**	**NO_3-N Leached**
		(mm)	**(kg/ha)**
Trees			
	Low	738	312
	Medium	1185	414
	High	3246	782
Pasture			
	Low	1183	380
	Medium	1750	446
	High	3824	770

This reflects the lower evapotranspiration and hence the greater drainage volumes in the pasture plots (Table 6.4). These greater drainage volumes dilute the NO_3-N in the soil resulting in lower predicted NO_3-N concentrations in the drainage.

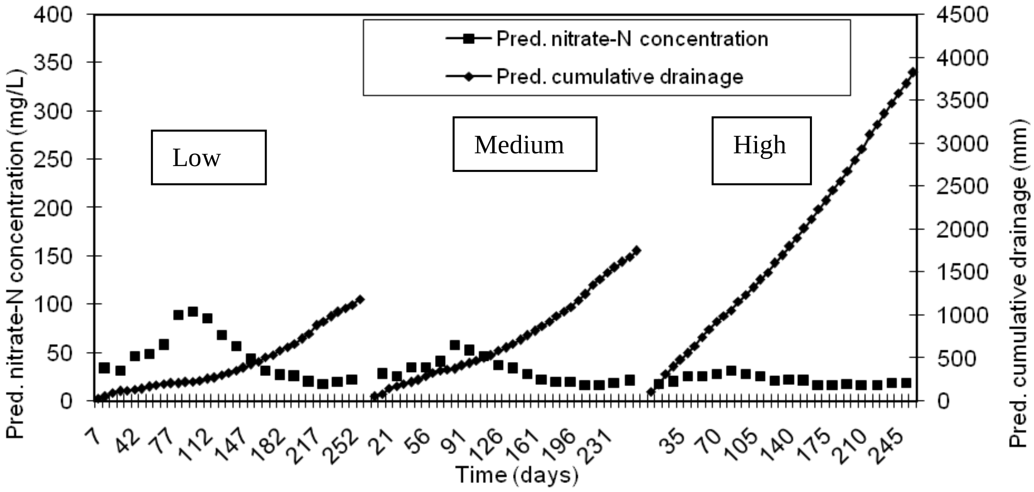


Fig. 6.7**:** Predicted cumulative drainage and leachate NO_3-N concentration at 500 mm soil depth for all treatments of pasture plots.

6.3.5 Application of the Calibrated LEACHN Model to Manage Effluent Irrigation at the Carterton Site

At the Carterton LTS, it was observed (Chapter 3) that the NO_3-N concentration in the groundwater under the treatment receiving the medium rate of effluent application exceeded the MPL of 11.3 mg/L during June and reached a concentration of 16.6 mg/L. Managers at the site would therefore be interested to know whether changing effluent application strategies immediately prior to and during the "problem" month of June could reduce both the concentration of NO_3-N in the leachate, and also the total amount of NO_3-N leached. The LEACHN model was used to carry out this scenario analysis.

The first scenario modelled was the original effluent application strategy of 45 mm per week. In the second scenario, this application rate was reduced to 30 mm per week during April and May (a total of 9 effluent applications). The third scenario was similar to the second scenario except that there was an additional reduction in effluent application to 25 mm per week in June (a total of 5 applications).

Scenarios 2 and 3 had less effluent and effluent N applied than scenario 1 (Table 6.5). Not surprisingly therefore, the LEACHN model predicted that both the quantity of drainage and the amount of leached N would also be less in scenarios 2 and 3 than in scenario 1. However, while the predicted reduction in drainage was very similar to the reduction in effluent added, the predicted reduction in N leached in scenarios 2 and 3 was less than the reduction in effluent N applied. This is because much of N leached during the winter months originates from effluent N applied several months earlier. Therefore, any reduction in N leached, as a consequence of reducing effluent applications, results mainly from the reduction in drainage volume rather than a reduced concentration of NO_3-N in the drainage water. Indeed, the predicted average NO_3-N concentration in the drainage water over the late autumn and winter months is predicted to increase as the volume of effluent applied is decreased (Table 6.5).

Table 6.5: Effluent and effluent N applied in three scenarios, and the resulting predicted drainage, N leached and average N concentration over the whole simulation period (1 December – 18 August) and from 1 April – 18 August.

Scenarios	**Effluent Application (mm)**	**Effluent N Application (kg/ha)**	**Drainage (mm)**	**Drainage (1 Apr.- 18 Aug.) (mm)**	**N leached (kg/ha)**	**N leached (1 Apr.–18 Aug.) (kg/ha)**	**Average N conc. (1 Apr.–18 Aug.) (mg/L)**
1	1710	195	1190	1060	427	353	33.3
2	1575	179	1050	940	418	344	36.6
3	1475	168	940	830	404	330	39.8

Whether this increased NO_3-N concentration in the drainage water has a greater impact on the resulting groundwater NO_3-N concentration than the overall reduction in the weight of NO_3-N leached will depend on the extent to which the groundwater is sourced from areas other than the LTS. However it is apparent that the LEACHN model provides a valuable tool with which the likely impact of management decisions at a LTS can be assessed.

6.4 Conclusions

The LEACHN model has been used with field data to simulate NO_3-N leaching at a land treatment site in Carterton District. Based on the calibration process, simulation results, and sensitivity analysis the following conclusions can be drawn:

- Comparison of the model against the experimental data from the treatments confirmed that by using measured values for some of the model parameters and varying others within the ranges normally found in the literature, the predicted seasonal and between-treatment pattern of NO_3-N concentrations in the soil solution were similar to those observed in the measured soil solution concentrations. However the LEACHN model consistently predicted higher concentrations of NO_3-N in the soil solution than were measured.
- The model was not very sensitive to changes in most of the parameters to do with transformations of inorganic N. But the model was quite sensitive to the values used for bulk density, AEV (a), BCAM (b), N mineralisation rate, base temperature and the Q_{10} factor. The apparent sensitivity of the model to large (30%) changes in bulk density is not of great concern because any errors in the measurement of bulk density are likely to be much smaller than this. Of real concern however is the sensitivity of the model to the value used for the N mineralisation rate constant. The correct value of this parameter is very difficult to determine and the use of the default value for this parameter in the model is thought to perhaps be the reason for the systematic over-estimation of the NO_3-N concentrations by the model.
- The model predicted that the cumulative drainage would increase as effluent application rate increased, and would also be higher from the pasture plots than the tree plots. Although the weight of NO_3-N leached over the

simulation period was predicted to increase with increasing effluent application rate, the concentrations of NO_3-N in the leachate were predicted to be much lower in the treatments receiving the high rates of effluent application. Similarly, the NO_3-N concentrations in the leachate from the pasture plots were predicted to be lower than the NO_3-N concentrations in the leachate from the tree plots.

- Evaluation of two possible alternative application strategies to reduce the impact of effluent application on groundwater NO_3-N concentrations in winter demonstrated the usefulness of the LEACHN model in such scenario analysis. The simulations demonstrated that although the high NO_3-N concentrations in groundwater occurred in winter, simply modifying effluent application rates during late autumn and winter would be unlikely to have a great beneficial effect.

CHAPTER 7

ANALYSIS OF THE DECISION SUPPORT MODEL

7.1 Background

The decision support model developed in earlier chapters has been tested on two different soils (i.e. a stony silt loam from recent fluvial deposits and a sandy loam), two effluents (i.e. municipal wastewater and dairy shed effluent), with different vegetation cover (trees and pasture), and under different effluent irrigation programmes (refer to Chapters 5 and 6). The results of these simulations showed that rainfall patterns and designed effluent hydraulic loading rates are the critical parameters in determining whether environmentally significant leaching of NO_3-N will occur from a LTS.

In real-life management situations, actual rainfall is not known in advance and so the average rainfall at the site is used to do the initial (prior to effluent application) runs of the model in order to decide on a likely safe rate of effluent application. Once the year is underway, and effluent application has started, average rainfall in the model can be replaced by actual rainfall data as it occurs, and effluent application rates can be modified if that is thought necessary. In this Chapter the effect of using actual and average rainfall data on the predictions of the calibrated LEACHN model is investigated. This work is reported in the published paper reproduced below. As in previous chapters, there has been some modification to the text of the paper to improve clarity and linkage with the remainder of the thesis.

7.2 Analysis of a LEACHN-Based Management Technique in 'Predictive Mode'

by Babar Mahmood,

Agricultural Water Management – An International Journal. Elsevier S

7.2.1 Introduction

Computer simulation models have become popular tools for studying the transport mechanisms of agricultural chemicals through subsurface zones. Simulation studies can be used as an inexpensive, time saving, and environmentally safe technique to evaluate the effects of various agricultural management practices on the subsurface movement of NO_3-N (Singh and Kanwar, 1995 and various others e.g. Addiscott et al., 1986; Hutson and Wagenet, 1992; Jarvis et al., 1991; Ramos and Carbonell, 1991). Over the past several years, a large number of computer simulation models have been developed. But few data sets are available for testing a range of models, few models have been tested on a range of soils, and very few models have much demonstrable ability to simulate transient field leaching conditions (Borah and Kalita, 1999).

Few of the models developed so far are suitable for solving management related problems because of the lack of thorough validation. The models vary widely in their conceptual approach and degree of complexity, and are strongly influenced by the environment, training and pre-occupations of their developers (Borah and Kalita, 1999). Models also vary in complexity, output presentation, and input parameter requirements. Spatial and temporal variations involved with the data result in a high degree of uncertainty associated with the results obtained (Verma et al., 1995). Before using a model for a particular soil type and environment, it is important to corroborate the model for these conditions.

The behaviour of N in the soil-plant-water system is very complex. It is dynamic and involves numerous interactions and transformations. Models are useful tools for integrating different processes involved in N transport in soil and can be used in forecasting how a system will behave without actually making measurements in the physical system. In the past decade, development and application of models to predict soil water transport, N transformation, and N transport has increased tremendously (Ahmed et al., 1994). LEACHN is one such model, which can be used to simulate field-scale N transformations and movement in the unsaturated zone of the soil profile. LEACHN is one of the five sub-models of LEACHM, developed by Hutson and Wagenet (1992). One particular application of LEACHN is the simulation of the leaching of NO_3-N and NH_4-N from the plant root zone, which is of concern because once beyond the root zone, N is no longer available to plants and thereby has the potential to pollute the groundwater.

The purpose of this study was to use the LEACHN model in predictive mode in order to check if it is reasonable to use average rainfall data to assess the likelihood of unacceptable levels of NO_3-N leaching as a result of effluent application.

7.2.2 Methodology

The Decision Support System: Based on the findings of a field scale experiment (Chapter 3), a DSS (Fig. 7.1) was developed to assist with long-term management decisions at a LTS on the basis of the predicted concentration of NO_3-N in the leachate. In this DSS the LEACHN model (Hutson and Wagenet, 1992) deals with both the hydrological and biological aspects of the wastewater-irrigated soils.

This DSS takes account of the soil's and plant's ability to take up the applied water and nitrogen. Using this DSS, management decisions can be made at a LTS by looking at the concentration of NO_3-N in the leachate as predicted by the LEACHN model. Monthly weather data, initial SMC and SNC of the soil profile, and the design effluent irrigation volume are the input parameters to the model. It then predicts SMC and SNC, AET, plant uptake of nutrients, leachate volume and nutrient concentration in the leachate. If the leachate NO_3-N concentration is below the WHO's MPL of 11.3 mg/L in drinking water (i.e. no risk of groundwater contamination), then the planned irrigation can be applied. If the leachate NO_3-N concentration is above the MPL (i.e. there is a risk of groundwater contamination), then the irrigation scenario has to be modified and the model rerun.

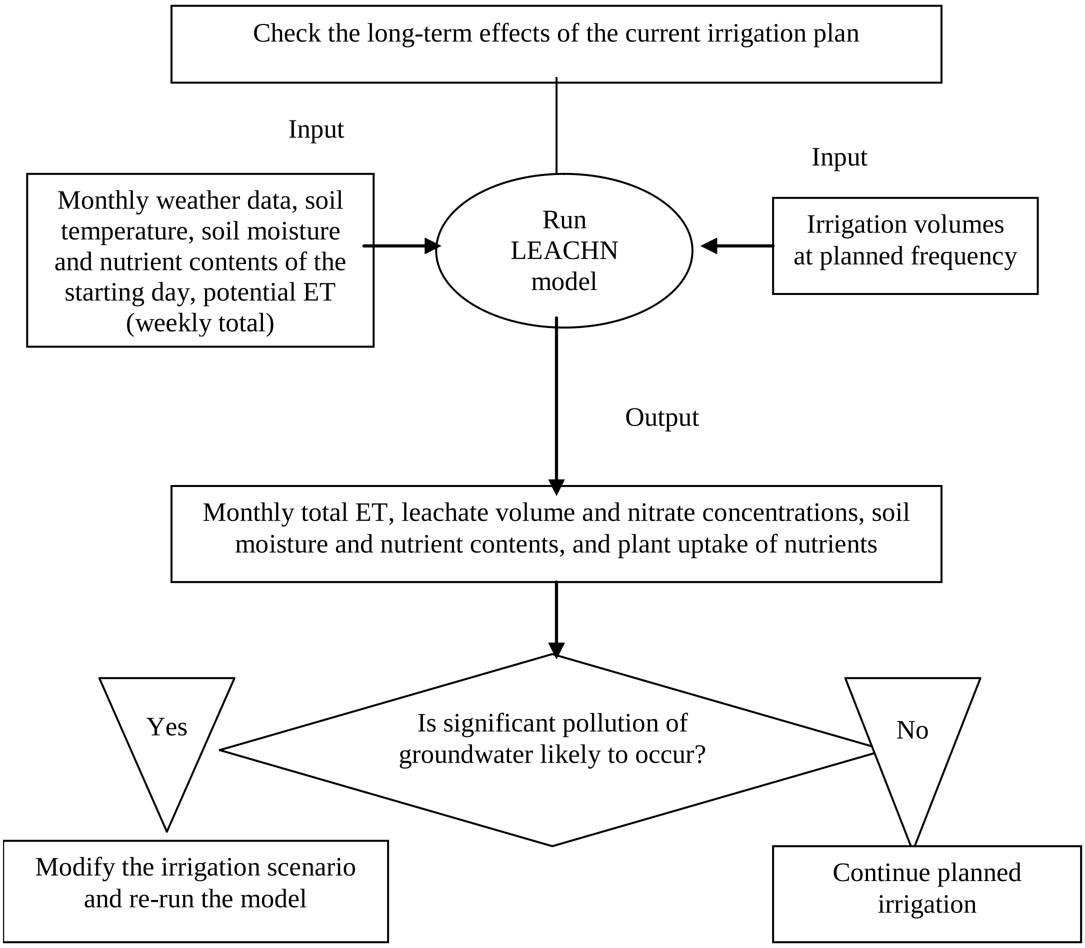


Fig. 7.1**:** A LEACHN-based DSS to reduce the risk of groundwater contamination at a LTS.

Analysis of the DSS in "predictive mode": An analysis of the DSS in 'predictive mode' was undertaken, using the calibrated LEACHN model, for 365 days from 1 January to 31 of December. The question was whether it would be possible to use the DSS to achieve management of the land application of effluent throughout the year in a way that the leachate NO_3-N concentration did not exceed the MPL of 11.3 mg/L.

The LEACHN model was calibrated for the Manawatu fine sandy loam soil (classified as Dystric Fluventic Eutrochrept - Weathered Fluvial Recent Soil in the New Zealand Soil Classification; Hewitt, 1998) and DSE (Chapter 5). The past 26 years (1975 to 2000) of daily rainfall, minimum and maximum temperature, and pan evaporation data for the Massey University site was obtained from the National

Institute of Water and Atmospheric Research (NIWA). A crop factor of 0.75 was used to estimate the PET values for the pasture crop. The designed hydraulic loading rate of 3.6 mm/week (which was calculated from the average inorganic N concentration (0.08 kg/m^3) and a N loading rate of 150 kg/ha/year), daily average rainfall, weekly mean temperature, and estimated weekly PET data for the past 26 years was entered into the model.

The model was run for 365 days using the designed irrigation volume. Further runs of the model were used to modify the weekly irrigation volumes in order to bring the leachate NO_3-N concentration below the MPL. Then the DSS was run on month-by-month basis using planned irrigation volume and actual rainfall data for the wettest year (1996 with total rainfall=1206 mm/year) of the past 26 years. Each run of the DSS was built-on the previous run.

7.2.3 Results and Discussion

Analysis of the decision support model in 'predictive mode': In the first scenario, the design irrigation volume and average rainfall, temperature, and PET data were given to the model (Fig. 7.2).

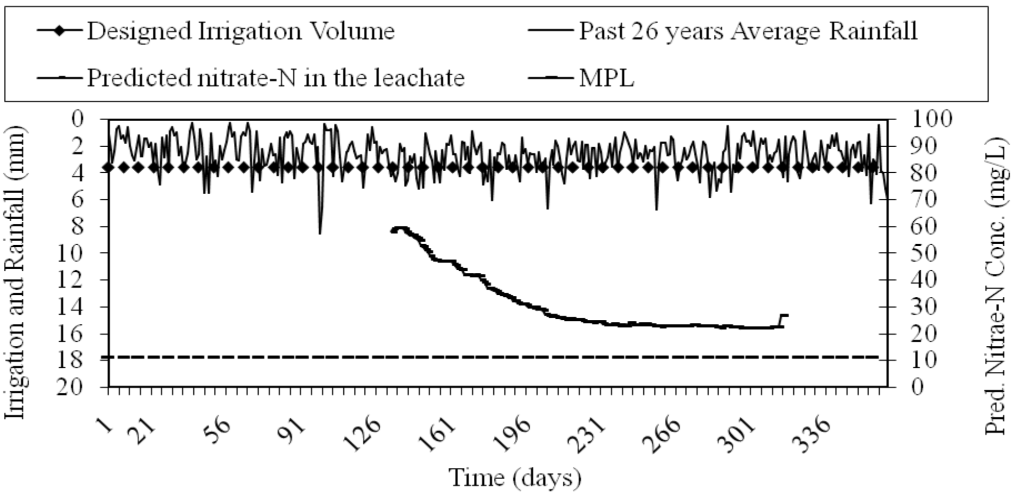


Fig. 7.2: Scheduled effluent irrigation, average daily rainfall and predicted NO_3-N concentrations in the leachate in the first scenario simulated by the LEACHN model.

The results of this scenario showed that there was no leachate from day 1 - 133 and day 317 - 365 (during autumn and the following summer). The leaching occurred from day 134 - 316 (during winter and spring) and the leachate NO_3-N concentration was between 22 and 59.9 mg/L (i.e. >11.3 mg/L which means there would be a risk of groundwater contamination).

In the second scenario, the irrigation volume was reduced from 3.6 to 0.0 mm/week from day 134 - 316 (winter and spring time) to see if this would reduce the risk of groundwater contamination during this time. In response to this modification, the leaching occurred from day 132 - 295 and the leachate NO_3-N concentration was between 23.7 and 59.5 mg/L (Fig. 7.3). The second scenario resulted in a surplus of 80.2 mm of effluent, which had not been applied during spring and winter (from day 134 to 316). The second scenario also indicated that there would be no leachate from days 295 to 365 (summer time).

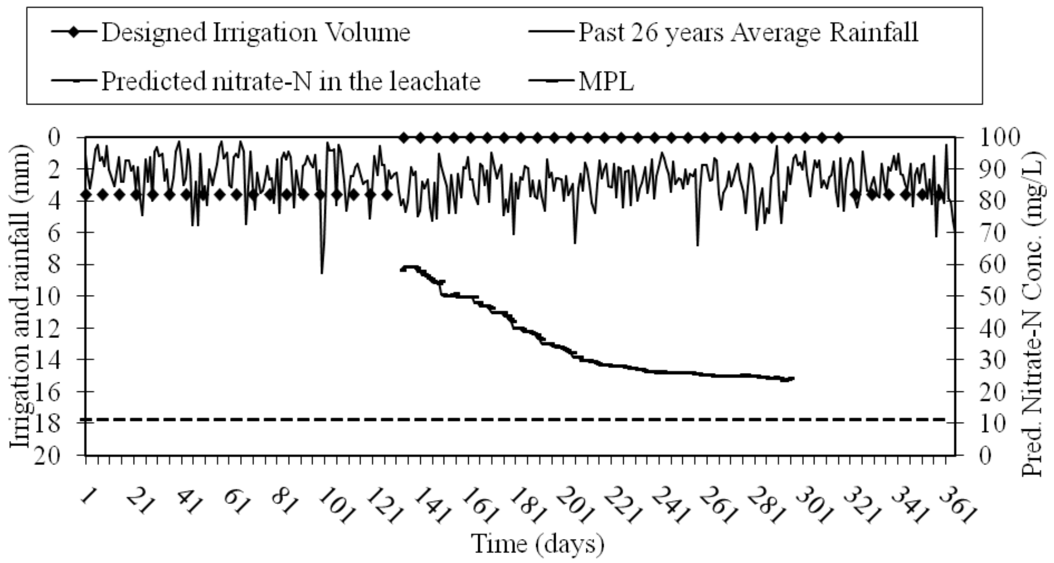


Fig. 7.3: Scheduled effluent irrigation, average daily rainfall and predicted NO_3-N concentrations in the leachate in the second scenario simulated by the LEACHN model.

In the third scenario, the designed irrigation volume was increased from 3.6 to 8.6 mm/week during late spring/summer (days 295 - 365), in order to apply some of the "carried over" effluent not applied in winter in Scenario 2. The model predictions for the third scenario suggested that it would be possible to safely apply 26 mm of extra effluent onto the land without having any leaching during this time (Table 7.1). In this third scenario, there was still 54.2 mm of surplus effluent, carried over from winter, that could not be applied during the year. That extra amount of effluent was rolled forward to the next year.

Table 7.1: Irrigation application rates, leachate NO_3-N concentrations, and days when leaching occurred (numbered from 1 January) in scenarios 1 - 3. These scenarios assumed average rainfall data from the past 26 years (1975 – 2000).

Scenario	**Irrigation application rate (mm/week)**	**Leachate NO_3-N concentrations (mg/L)**	**Days with Leachate**
1	Jan-Dec: 3.6	22.0 – 59.9	134 to 316
2	Jan-May: 3.6 May-Nov: 0.0 Nov–Dec: 3.6	23.7 – 59.5	132 to 295
3	Jan-May: 3.6 May-Nov: 0.0 Nov-Dec: 8.6	23.7 – 59.5	132 to 295

In the fourth to the tenth scenarios the model was run initially with the planned irrigation volume and average rainfall data (Table 7.2), and then with the average rainfall data replaced with actual rainfall data for the appropriate months in1996.

Table 7.2: Irrigation application rates, sources of rainfall data (average for 26 years or actual for 1996), leachate NO_3-N concentrations, and days when leaching occurred (numbered from 1 January) in scenarios 4-10.

Scenario	Irrigation volume (mm/week)	Source of rainfall data (Average - Av or Actual - Act)	Leachate NO_3-N conc. (mg/L)	Days with Leachate
4	Jan: 8.6 Feb-May: 3.6 May-Nov: 0.0 Nov-Dec: 8.6	Jan: Act Feb-Dec: Av	23.7 – 59.5	132 to 295
5	Jan-Feb: 8.6 Mar-May: 3.6 May-Nov: 0.0 Nov-Dec: 8.6	Jan-Feb: Act Mar-Dec: Av	21.5 – 39.0	50 to 53 and 90 to 295
6	Jan-Mar: 8.6 Mar-May: 3.6 May-Nov: 0.0 Nov-Dec: 8.6	Jan-Mar: Act Apr-Dec: Av	21.0 – 40.2	81 and 100 to 295
7	Jan-Mar: 8.6 Mar-Apr: 3.6 May-Nov: 0.0 Nov-Dec: 8.6	Jan-Apr: Act May-Dec: Av	19.0 – 37.0	92 to 96, 105 to 111, and 126 to 295
8	Jan-Mar: 8.6 Mar-Apr: 3.6 May-Nov: 0.0 Nov-Dec: 8.6	Jan-mid Nov: Act Mid Nov-Dec: Av	10.0 - 36.9	50 to 52, 81, 93 to 96, 107 to 111, 121, 136, 141 to 146, 153, 162 to 217, 233 to 235, 248 to 257, 272, 273, and 282 to 288
9	Jan-Mar: 8.6 Mar-Apr: 3.6	Jan-Nov: Act Dec: Av	17.5	323, 324, 329, 330, and 333

	May-mid Nov: 0.0 Mid Nov-end Nov: 42.6 Dec: 8.6			
10	Jan-Mar: 8.6 Mar-Apr: 3.6 May-mid Nov: 0.0 Mid to end of Nov: 42.6 Dec: 10.6	Jan-Dec: Act	17.5 – 21.0	336, 337, 338, and 351

In the fourth scenario, the planned irrigation volume was increased from 3.6 to 8.6 mm/week for the month of January (from day 1 to 31) in order to apply some of the effluent (which had not been applied during wet periods in scenario 2 and was rolled forward). The outcome of the first model run of the fourth scenario (with average rainfall data) suggested that it would be possible to safely apply an additional 30 mm of surplus effluent onto the land without there being any leaching during this month (January).

In Scenario 2 there was 80.2 mm of effluent that could not be applied during winter. If some of this surplus effluent was applied as in Scenario 3 (26 mm in November/Dec) and Scenario 4 (30 mm in January), this would still leave 24.2 mm of surplus effluent from the previous winter, that had not been applied, and this was rolled forward to the next month.

The second run of the fourth scenario (with actual January rainfall data) suggested that there would still be no leaching in January, and there would also be little effect on the time leaching occurred and leachate NO_3-N concentrations throughout the rest of the year (Table 7.2).

In the fifth scenario, the planned irrigation volume was increased from 3.6 to 8.6 mm/week for February (from day 32 to 57) in order to apply most of the remaining "carried forward" effluent. In the first run (with average rainfall data) the model suggested that it would be possible to apply 20 mm of surplus effluent (in addition to the scheduled 3.6 mm per week) onto the land without having any leaching during the month of February. This would still leave 4.2 mm of surplus effluent, which could not be applied during this month and was rolled forward to the next month.

The second run of the fifth scenario (with actual rainfall data for January and February) predicted that leaching would occur on day 50, 51, 52, 53 (third week of February), and then from day 90 to 295 (Fig. 7.4). The leachate NO_3-N concentration would be between 21.5 and 39 mg/L during this time. The predicted leaching in February occurred in response to a large rainfall event and illustrates some of the difficulties associated with using average rainfall data. The average rainfall data used in this study were obtained by averaging the rainfall that fell on a particular day (e.g. 1 February) for each of the past 26 years. This has the effect of producing an even pattern of rainfall from day to day, with no days without rain and no really large amounts of rainfall on any day. In summer this average daily rainfall is likely to be less than evapotranspiration, and thus the model predicts that quite large quantities of effluent can be applied, without any danger of leaching.

Of course, actual rainfall patterns are not like this. On many days no rain falls at all, and on others there can be large rainfall events. This was the case during the 3rd week in February 1996, when very heavy rain fell on several days. In combination with effluent irrigation, this was sufficient for leaching to be predicted.

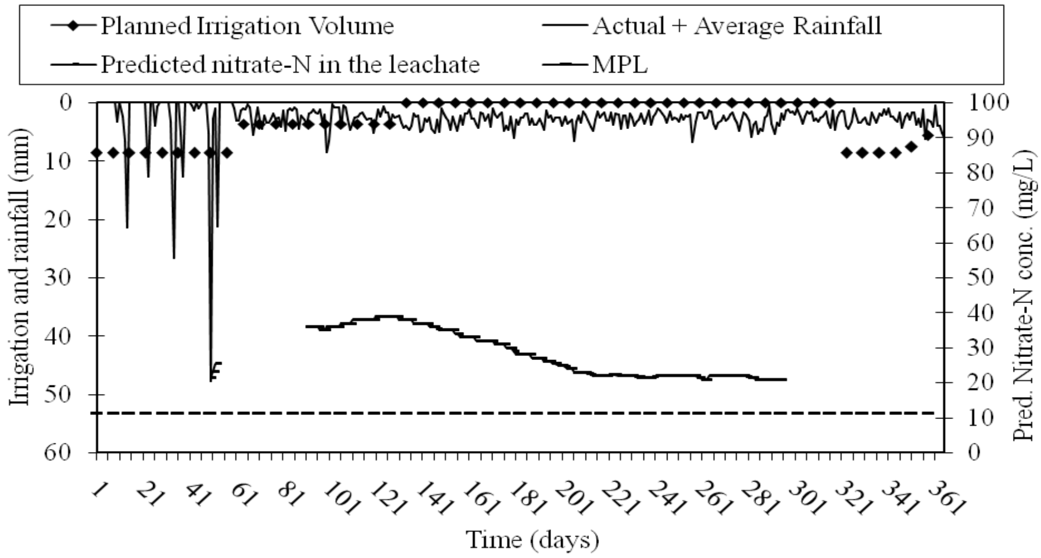


Fig. 7.4: Scheduled effluent irrigation, actual daily rainfall and predicted NO_3-N concentrations in the leachate in the fifth scenario simulated by the LEACHN model.

In the sixth scenario, the planned irrigation volume was increased from 3.6 to 8.6 mm/week for the month of March (from day 57 to 90). This would be more than sufficient to apply the remaining effluent (i.e. 4.2 mm) carried over from the previous winter. The outcome of the first run (with average rainfall data) predicted that it would be possible to safely apply this additional effluent onto the land without having any leaching during the month of March. The leaching was predicted to occur from day 90 to 295, and the predicted leachate NO_3-N concentration was between 21.5 and 39 mg/L (i.e. above 11.3 mg/L).

The results of the second run of the 6th scenario (with actual rainfall data for January, February and March) suggested that leaching occurred on day 81 (third week of March), and then from day 100 to 295 (Table 7.2). The leachate NO_3-N concentration was predicted to be between 21 and 40.2 mg/L during this time. Once again, this highlights the difference in predicted leaching patterns when using actual rather than average rainfall data.

In the seventh scenario, the planned irrigation volume was reduced from 3.6 to 0.00 mm/week from day 99 to 120 (during April) in order to reduce the risk of groundwater contamination during this time. The outcome of the run with average rainfall data suggested that leaching occurred from day 100 to 295, and leachate NO_3-N concentrations were between 21 and 42 mg/L (> MPL). This means either it is necessary to store the effluent (which could not be applied during this time) or discharge the effluent to a river or stream during wet periods of the year.

The results of the second run (with actual rainfall data for January to April) of the 7th scenario (Table 7.2) suggested that leaching would occur on days 92 - 96, 105 - 111, and 126 - 295 (from late autumn to mid spring). The leachate NO_3-N concentrations were reduced, varying between 19 and 37 mg/L during this time.

In the eighth scenario the planned irrigation volume was 0.0 mm/week from day 121 to 316 (from May to mid-November). In the eighth scenario the average rainfall data was replaced with actual rainfall data from May to mid - November (i.e. from day 121 to 316). The results of this scenario suggested that leaching would occur on days 50, 51, 52, 81, 93, 94, 95, 96, 107-111, 121, 136, 141-146, 153, 162-217, 233, 234, 235, 248-257, 272, 273, and 282 - 288. The leachate nitrate-N concentrations were between 10.0 and 36.9 mg/L during these times. This scenario showed that replacing the average rainfall values with actual rainfall have changed the timing of leachate (Table 7.2) and reduced the leachate NO_3-N concentrations. It should be noted that it would have been safe to apply effluent during May to mid – November when the leachate nitrate concentrations were below or just above 11.3 mg/L. It is clear from eighth scenario that 100.8 mm of effluent could not be applied during wet periods (late autumn to winter).

In the ninth scenario, the planned irrigation volume was increased from 8.6 to 42.6 mm during the third and fourth week of November (from day 323 to 330) in order to apply the extra effluent (which was not applied in late autumn and winter in Scenario 8). The first run of the model (with average rainfall data) predicted that no leaching would occur after day 288, which suggests that it was safe to apply the extra effluent during this time of the year.

The second run of the ninth scenario (with actual rainfall data) predicted that leaching would occur on days 323, 324, 329, 330, and 333 (Fig. 7.5) and the leachate NO_3-N concentration was 17.5 mg/L. In this scenario there would still remain a surplus of 58.2 mm of effluent that would not have been applied due to the wet conditions in winter.

In the tenth scenario, the planned irrigation volume was increased from 8.6 to 10.6 mm/week from day 335 to 365 (during December) in order to apply some of the effluent not applied during winter. The outcome of the first run (with average rainfall data) showed that it was possible to safely apply 10.6 mm of surplus effluent (in addition to the normal application of 3.6 mm/week) without having any leaching during this month. The results of the second run of the 10th scenario (with actual rainfall data for December) showed that leaching occurred on days 336, 337, 338, and 351 (Table 7.2).

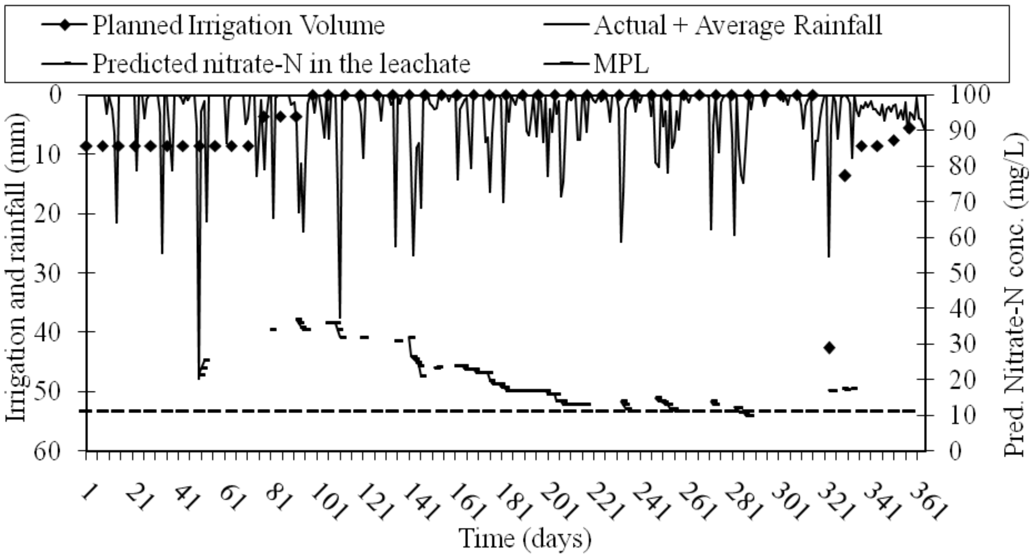


Fig. 7.5: Scheduled effluent irrigation, actual daily rainfall and predicted NO_3-N concentrations in the leachate in the ninth scenario simulated by the LEACHN model.

The leachate NO_3-N concentration was predicted to be between 17.5 and 21 mg/L during this time. Despite this high effluent application rate in December, there would still remain a surplus of 47.6 mm of effluent that could not be applied onto the land during the 1996 year. This extra amount of effluent could be stored and rolled forward to the next year in order to apply over the summer time (low rainfall, high temperature and ET) or perhaps discharged to a river or stream (if there is any) during the wet periods of the year when river flows are high.

The model predicted that the cumulative leachate volume with average rainfall data (313 mm) would be much less than with the actual 1996 rainfall data (607 mm). This is largely because 1996 was deliberately chosen because it was a wet year that could be expected to pose problems for a LTS. A comparison of the monthly average rainfall data from the past 26 years (1975 - 2001) and the actual monthly rainfall data in 1996 (Table 7.3), showed that the actual 1996 rainfall data was greater than the average rainfall data in 10 of the 12 months. The increase in monthly rainfall from the average ranged from 1.4% to 50%. A secondary factor leading to lower predicted leaching volumes from the averaged rainfall records is the absence in the averaged data of large daily rainfalls – particularly in summer. As noted earlier in this Chapter, in February in 1996 there were several days of very heavy rain that caused leaching to take place at a time of the year when averaged data suggested there would be no possibility of this occurring.

The predicted cumulative mass of NO_3-N leached with average rainfall values (120 kg/ha/year) was less than with actual rainfall data (132 kg/ha/year), (Table 7.3). However, because of the much greater predicted leachate volumes using the actual 1996 rainfall data, the predicted leachate NO_3-N concentrations using average rainfall data were 35 % (on average) higher than those predicted using actual rainfall data. In other words, using long-term averaged data resulted in a prediction of a smaller volume of leachate with a higher average concentration of NO_3-N than would be predicted in a very wet year using actual rainfall data.

Given that 1996 was deliberately chosen as an "outlier year" in terms of annual rainfall the difference in predicted leaching using averaged and "actual" climate data is not that large in terms of likely environmental impact, and therefore it would seem reasonable to use the average rainfall data to do the initial feasibility modeling when designing a LTS. When operating the LTS, and making decisions on a daily basis as to how much effluent to apply, the averaged data can be replaced by actual daily rainfall, as it occurs, to update estimations of current SMC and the likelihood of future leaching.

Table 7.3: Comparisons of the rainfall and mass of NO_3-N leached averaged over 26 years and in 1996.

	With average annual rainfall data of past 26 years	**With actual 1996 rainfall data**	**Percentage increase (%)**
Mass of NO_3-N leached (kg/ha/year)	120	132	10
Annual rainfall (mm/year)	965	1206	25

The analysis also showed that it was not possible to manage the land application of effluent throughout the year (i.e. 1 January to 31 December) in a way that the leachate NO_3-N concentration did not exceed the MPL of 11.3 mg/L. It was noted that the leaching occurred during the wet periods of the year and that the NO_3-N concentrations were highest in the first leaching events in autumn and then gradually decreased to reach a reasonably steady "equilibrium" concentration in late winter and early spring (similar to observations made by Magesan et al., 1996). When average rainfall data were used in the simulations, this "equilibrium" NO_3-N concentration was approximately 20 mg/L – i.e. above the MPL of 11.3 mg/L (e.g. Fig 7.4). When the actual rainfall data for autumn, winter and spring in the wet year of 1996 are used in the simulation (Fig. 7.5), the NO_3-N concentration in the leachate was close to the MPL of 11.3 mg/L for much of winter and spring.

This steady leaching of NO_3-N (Figs 7.4 and 7.5) was predicted to occur even though no effluent was being applied over the winter/spring period. The high NO_3-N concentrations in the early leaching events presumably result from NO_3-N that had built up in the soil over the summer/autumn period. During this time little or no leaching is predicted to occur and the nitrification of both soil organic N and added effluent N resulted in NO_3-N being produced at a faster rate than could be taken up by plants or lost through denitrification. However, this accumulated surplus of NO_3-N appeared to be flushed from the soil reasonably quickly and the NO_3-N leached during the remainder of winter and spring is presumably sourced from the mineralization and subsequent nitrification of soil organic N. Changing effluent application strategies will not affect this greatly – as was shown by the various scenarios modeled in this chapter.

The extra amount of effluent (which could not be applied during the wet periods) could be stored and rolled forward to the next year where it would be applied to land over the summer time (low rainfall, high temperature and ET) or discharged to a river or stream (if there is any) during the wet periods of the year when river flows are high.

In the modeling approach used in this and earlier chapters it has been assumed that the major likely environmental impact of effluent application would be on groundwater NO_3-N concentrations, and that therefore the leachate from the LTS should not have a NO_3-N concentration greater than the MPL of 11.3 mg/L. This is a very conservative strategy, because normally leachate from a LTS would be diluted once it mixed with the groundwater. It would therefore be possible to have leachate NO_3-N concentrations considerably in excess of 11.3 mg/L, without causing the groundwater to exceed the MPL. However, in the absence of detailed information on groundwater hydrology, a strategy of ensuring the leachate NO_3-N concentrations are always <11.3 mg/L could be justified, because that way the leachate from the LTS can never cause the groundwater to exceed the MPL.

Using this DSS it would be possible to identify those times of the year (in advance) when it may not be possible to apply the effluent onto the land, so that the manager of the LTS could make contingency plans for that time of the year. At the same time, using this DSS it is possible to identify the times of the year when the surplus effluent (which could not be applied during wet periods) could safely be applied to the land during summer time. Also, if the normal weather pattern changed or unexpected heavy rainfall events occurred during the summer time then the DSS can be used to model the likely effect on NO_3-N leaching, and subsequent effluent application strategies can be modified accordingly.

7.2.4 Summary and Conclusions

1. The LEACHN model, which is the basis of the DSS, predicted that effluent could be irrigated onto the LTS regularly throughout the late spring, summer and early autumn period with little likelihood of leaching – except if there was typically large rainfall event. However during winter the rainfall exceeds evapotranspiration for an extended period and leaching occurs. This is the case irrespective of whether or not effluent is applied over the winter period.
2. The N application rate during the summer period is greater than the N uptake by plants and loss of N by denitrification, and so NO_3-N builds up in the soil. When leaching commences in late autumn, this NO_3-N is flushed through the soil, and this creates very high NO_3-N concentrations in the leachate during the early part of the drainage season. There seems to be little that can be done to avoid this. Several workers (e.g. Sharpley and Syers, 1979; Heng et al., 1991; Houlbrooke et al., 2008) have observed high concentrations of NO_3-N in leachate from the first few leaching events in autumn/winter, even in the absence of effluent application.
3. The model prediction of cumulative leachate volume (313 mm) using average rainfall data was about half that (607 mm) predicted using actual rainfall data for 1996. Most of this difference could be attributed to the fact that 1996 was one of the wettest years on record. However, a second reason for the difference was the way that the average rainfall data in this study was calculated. This was done for each calendar day by averaging the amount of rainfall that fell on that day, over each of the last 26 years. The resulting rainfall pattern has some rain falling on all days, but without there being any days with particularly high rainfall. In winter, when the rainfall is much higher than the evaporation, the daily rainfall pattern has little effect on the cumulative leachate volume over a period of a few weeks. In contrast, in summer the average daily rainfall is much less than the daily evapotranspiration rate and so there would appear to be no possibility of leaching if the average rainfall data are used. Actual rainfall patterns for summer however, have many days without rain and sometimes quite heavy rain over a period of several days. This can sometimes be sufficient to cause significant leaching.

4. The predicted cumulative mass of NO_3-N leached using average rainfall values was 120 kg/ha/year and was less than predicted with actual rainfall data from a wet year (132 kg/ha/year).
5. Thus using average rainfall data resulted in a prediction of a smaller leachate volume, but a higher average NO_3-N concentration, than if actual rainfall data from 1996 were used. The differences however were fairly small from the perspective of environmental management and therefore using average rainfall data for conducting feasibility studies when designing an LTS should be satisfactory.

CHAPTER 8

SOLUTE TRANSPORT MODELLING

8.1 Introduction

Simulation models are useful to understand the interaction between the transformation and transport of N in the field. The growing concern about the environmental impact of effluent irrigated systems has increased the desire to predict the transport and transformation of N in the soil plant system more accurately. Several models simulating N transformations and transport in the soil plant system have been developed and tested over time (e.g. Johnsson et al., 1987, Jabro et al., 1993; Hutson and Wagenet, 1992; Vanclooster et al., 1995; Eckersten et al., 1994; Bryant et al., 2011). Evaluating N transformation and transport models under field conditions is a complex research challenge. A major difficulty in evaluating these models under field conditions results from the strong interaction between physical and biological factors, plant uptake and N cycling processes.

In previous chapters, the LEACHN model was evaluated at both laboratory and field scales. In those chapters, LEACHN appeared to have some promise as a decision support tool but it became apparent that just using this model as a 'black box' was not satisfactory, and it requires an in-depth understanding of how the model works for it be a useful tool for the operators and managers of a LTS.

In Chapter 7 it was identified that using the default value for the mineralisation rate constant inadvertently resulted in predictions that there would be a large net mineralisation of N and a decrease in the quantity of humus N. This was the cause of the excessively high predictions of N leaching. In this chapter the way in which the LEACHN model handles the losses and gains from the organic N pool will be explored more fully.

In Chapter 6 a simple sensitivity analysis was undertaken. To actually use the model as a tool to assist in the management of an LTS this sensitivity analysis needs to be extended to understand why the model is more sensitive to some parameters than others, and to identify what parameters will need to be measured at a site and what parameters can be obtained from the literature.

Therefore, this chapter provides an in-depth analysis of the impacts of the mineralisation (K_{min} – day^{-1}), nitrification (K_{nit} – day^{-1}) and denitrification (K_{den} – day^{-1}) rate constants, and the bulk density on LEACHN predictions of leachate NO_3-N concentrations and the total amounts of NO_3-N leached. To assist in developing this understanding of the model behaviour this chapter includes an assessment of how different irrigation scenarios (i.e. rainfall alone with no effluent irrigation, rainfall and irrigation of effluent containing no N, and rainfall and irrigation with effluent containing N) affect the resulting distribution of N down the soil profile (0 - 500 mm).

8.2 Aims and Objectives

The aims of this study were to undertake a detailed evaluation of the LEACHN model in order to assess its usefulness as a decision support tool for managing an LTS. The specific objectives of this study were to:

1. Evaluate the sensitivity of predicted changes in the amount of humus-N to changes in the mineralisation, nitrification and denitrification rate constants using the CDC field data set. It was apparent in earlier chapters that the model is sensitive to the value used for the N mineralisation rate constant, and use of the default value for this parameter resulted in the model over estimating the leachate NO_3-N concentrations.
2. Evaluate the sensitivity of predicted NO_3-N concentrations and amount of leachate and NO_3-N leached to changes in the soil bulk density. It was apparent in Chapter 6 that the model was sensitive to bulk density values and the question was “why was this the case?”
3. Undertake irrigation scenario analysis (i.e. rainfall alone with no effluent irrigation, rainfall and irrigation of effluent containing no N, and rainfall and

irrigation with effluent containing N) using the parameterised model in order to understand the predicted fate of N added in effluent at a land disposal site.

4. Use the OVERSEER® nutrient budget model, to compare its predictions of the amount of N leached and average NO_3-N concentrations with LEACHN model predictions.
5. Use the parameterised LEACHN model for another field data set.

8.3 Model Descriptions

8.3.1 LEACHN

LEACHN is the nitrogen version of the LEACHM model, which has evolved from modelling efforts in the last twenty years and has successfully been used by several workers to describe nitrate and pesticide movement in the field (Sogbedji et al., 2001; Ramos and Carbonell, 1991; Wagenet and Hutson, 1986 and 1989). It is a research model that can be used for management purposes if some of the required inputs that are difficult to obtain, are estimated and not actually measured (Ramos and Carbonell, 1991). There are five modules of LEACHM. One of these modules, LEACHN, describes nitrogen transport and transformation and it is the one used in the present simulation. A second, LEACHP, deals with pesticides; a third, LEACHC, describes the flow of inorganic ions (calcium, magnesium, sodium, potassium, chloride, carbonate, and bicarbonate); a fourth, LEACHB, describes microbial population dynamics, and a fifth, LEACHW, describes water transport only. LEACHM also has a heat subroutine that allows rate constants to be adjusted according to temperature and water content. In each of the five modules there is a main routine that initialises variables, calls subroutines and performs mass balancing. Subroutines deal with the different processes such as water and solute flow, evapotranspiration, sinks, sources, plant growth, heat flow, and also input and output, and time step calculations. A detailed description of the model is given by Hutson and Wagenet (1992).

The main required inputs for LEACHN are:

- Soil properties and initial conditions for each soil segment (initial water content, hydrological constants for the moisture retentivity curve, chemical contents and soil chemical properties),
- Soil surface boundary conditions (irrigation and rainfall amounts and rates of application, mean temperature, potential evapotranspiration data),
- Crop details (time of planting, root and crop maturity and harvest, root and cover growth parameters),
- Other constants needed include the diffusion co-efficient, dispersivity co-efficient, rate constants, and adsorption co-efficients for each soil segment.

LEACHN (version 3.0) is not intended to:

- Use unequal depth increments,
- Predict surface runoff water quantity or quality,
- Simulate how plants respond to soil or environmental changes,
- Predict crop yield,
- Simulate the transport of immiscible liquids,
- Predict solute distributions in situations subject to two or three dimensional flux patterns.

In addition, LEACHM pays limited attention to farm-management practices and crop development. It does not have built-in capacity to perform parameter estimation or uncertainty analysis (Nolan et al., 2005). This is also true for LEACHN based on personal experience.

Many field processes are represented in LEACHN but some are not, and potential users should be aware of this. The effects of macro pore flow, diffusion into and out of aggregates, small scale gradients in redox conditions which may influence nutrient transformations and a feedback loop between soil water and N status and plant growth are excluded, and could be important for some soils.

However, LEACHN is a useful tool to simulate N and water fluxes in unsaturated soils and also provides an opportunity to test the effects of changes in environmental variables (as shown in Chapters 5, 6 and 7) or conditions under which leaching occurs (Hutson and Wagenet, 1991). It considers water movement and chemical transport through the soil-water matrix. It numerically solves the Richards's equation (as given below) for water flow and the convection dispersion equation (CDE) for nitrate transport and leaching in a one dimensional, vertical layered soil profile (Sogbedji et al., 2001; Jabro et al., 1993). The Richards's equation is given below.

$$\frac{\partial \theta}{\partial t} = \frac{\partial}{\partial z}[K(\theta)\frac{\partial H}{\partial z}] - U(z,t) \qquad (8.1)$$

Where:

(θ) = Volumetric water content (m^3/m^3),

H = Hydraulic head (mm),

K(θ) = Water content dependent hydraulic conductivity (mm/d),

$\partial H/\partial z$ = Hydraulic gradient (mm/mm),

t = time (days),

z = depth (mm), and

U = water extraction by roots (per day)

In the LEACHN model urea, ammonium and nitrate can be sorbed onto the soil surface. The CDE for a sorbing, degrading chemical that is subject to plant uptake is given below (Hutson and Wagenet, 1991).

$$\frac{\partial(\theta c)}{\partial t} + \frac{\partial(\rho s)}{\partial t} = \frac{\partial}{\partial z}[\theta D(\theta,q)\frac{\partial c}{\partial z} - qc] - U(z,t) \pm \phi(z,t) \qquad (8.2)$$

Where:

c = Chemical concentration in liquid phase (mg/dm^3),

s = Chemical concentration in the sorbed phase (mg/kg dry soil),

ρ = soil bulk density (kg/dm^3),

D (θ, q) = Effective dispersion coefficient (mm^2/day), which is a function of water content and pore water velocity,

q = Water flux density (mm/day),

U (z, t) = Plant uptake of nitrogen (mg/dm^3 per day), and

ϕ = A source or sink term (mg/dm^3 per day) accounting for gains or losses through transformation.

Sorbed and solution phase concentrations are related by a linear sorption isotherm (i.e. $C_s = K_dC_L$ - where K_d is the partition coefficient measured in dm^3/kg) in which the concentration of sorbed substance (C_s) is assumed to be proportional to the concentration of that in solution (C_L). For nitrate the K_d is usually, but not necessarily, zero. Further discussion of the development of Equations 8.1 and 8.2 can be found in Wagenet (1993).

8.3.1.1 Nitrogen Cycling in LEACHN

LEACHN uses the concepts and equations described by Johnsson et al. (1987) to represent the transformation of, and flux of nitrogen between three organic pools (litter, degradable faeces or manure, and a relatively stable humus fraction), urea, and mineral ammonium and nitrate (Fig. 8.1). It should be noted that in the modelling reported earlier in the thesis the initial litter, manure and urea pools values were set at the default value of zero because there was no information on litter return at the site and the focus was on the fate of the inorganic N applied in the effluent. The implications of these assumptions are explored later in this chapter.

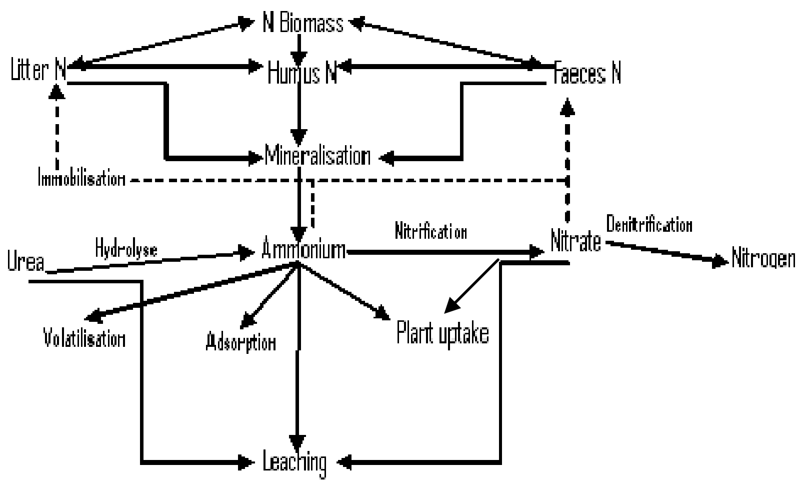


Fig. 8.1: Interactions, transport and transformation of N in LEACHN.

The major N transformation processes are mineralisation for each organic N pool, nitrification, denitrification, and volatilisation. The mineralisation, nitrification and denitrification rate constants are adjusted for temperature and water content (Johnsson et al., 1987). A Q_{10} factor is assumed for temperature, while rate constant values decrease on either side of optimum water content, except for the denitrification rate constant, which increases with increasing water content. The Q_{10} value and the maximum water content range (between a minimum air filled porosity and a minimum matric potential) are specified by the user. C:N ratios are specified for humus and biomass.

Soil NO_3-N concentrations, and hence the likelihood of NO_3-N leaching depends on the balance between supply and demand or removal. If sufficient N for the formation of humus and biomass is not released by mineralisation then mineral N will be immobilised from the soil, which means an insufficient supply of N will reduce the net rate of mineralisation. A lack of sufficient C substrate will inhibit denitrification in the same way.

It is impossible to measure all of the parameters for each site required to run the model. Comparing measured and predicted data to determine different parameters may not be feasible because of the large number of possible combinations of data values and availability of field data. Field data are often incomplete and are measured only occasionally. For example, only one or two N pools, usually mineral N pools, may be measured. Therefore, for calibration purposes there is a series of snapshots of part of the system at certain times. Unfortunately, it is difficult to calculate N fluxes through soil or between N pools using these data.

Mineralisation is a microbial process and depends on the growth of a microbial population. The microbial activity, in turn, depends on the temperature, water content, oxygen and substrate supply. The broad features of these processes are described by the equations of Johnsson et al. (1987). If the rate at which the parent organic N pool disappears is known then the mineralisation rate constant could be approximated, since an estimated half-life enables a corresponding rate constant to be calculated. But, it may not be possible to apply this technique because the estimated constant then fluctuates with soil temperature and water content during a simulation. A Q_{10} value of 2 doubles the rate constant for each 10 degree C rise in temperature and the water content correction factor can vary between 0 and 1.

Therefore, combined temperature and water effects on mineralisation rates could vary widely. In order to arrive at a reasonably correct seasonal N balance it may be wise initially to ignore temperature and water content effects (Hutson and Wagenet, 1991).

The amount of ammonium in the soil profile depends on its rate of generation (mineralisation) minus the rate at which it is removed (through nitrification and immobilisation) (Riha et al. 1986). Similarly, soil nitrate concentrations and the likelihood of nitrate leaching will depend on the balance between supply and demand or removal of nitrate. As the mineralisation rate constant is increased (as in Chapter 6), the residual organic N decreases and the general NO_3-N level increases.

8.3.1.2 Soil Hydrological Properties in LEACHN

The LEACHN model uses the Richards's equation to describe water movement and the CDE to describe chemical movement, and therefore it requires soil retentivity (i.e. θ at several values of matric potential), hydraulic conductivity and dispersivity data in the soil properties section. These data are rarely measured or are difficult to measure and are usually very variable spatially. Hutson and Wagenet (1991) mentioned that the user should not bother with a time consuming and expensive data collection process because chemical transport is often relatively insensitive to these values. In the field situation, water content distributions may vary widely from one soil to another, but the net downward fluxes depend mainly on the balance between rainfall, ET, and runoff. It is interesting to note that none of these is controlled primarily by soil hydraulic properties. Therefore, it is possible to estimate both soil retentivity and conductivity, with little loss of accuracy, when simulating field processes.

Campbell's retentivity equation (Campbell, 1974) is used in LEACHN and is given below.

$$\psi = a(\theta/\theta_s)^{-b} \qquad (8.3)$$

Where,

a and b = Constants,

θ_s = Water content at saturation (m^3/m^3),

ψ = Matric potential (kPa)

The simplest way to estimate a and b is to assume that field capacity is at -5 kPa and wilting point is at -1500 kpa, which may be true for most of soils (Hutson and Wagenet, 1991). By substituting the measured or assumed values of water content at field capacity and wilting point in equation 9.3, a and b values can be obtained that will simulate the plant available water range.

Estimation of $K(\theta)$ is difficult. The relationship between soil texture and $K(\theta)$ is not as close as it is for water content and matric potential because of the influence of soil structure and pore geometry on water flow. Campbell's (1974) estimations for $K(\theta)$ are used in LEACHN as shown in the equation below.

$$K = K_{sat} \, (\theta/\theta_s)^{2b+3} \qquad (8.4)$$

Where, K_{sat} is the saturated hydraulic conductivity, and θ_s and b are the same as in Equation 8.3.

Hutson and Wagenet's (1991) modelling experience showed that realistic simulations of redistribution to field capacity are often obtained if a value of K_{sat} is selected so that equation 8.4 could be applied to most of the $K(\theta)$ curve between saturation and FC, and the K value is equal to 1 mm/day at -20 kPa. However, measured K_{sat} values could be used but these may lead to unrealistic K values at water contents in the vicinity of field capacity. Field soils are rarely at saturation and representing realistic drainage to field capacity is more important than using a measured value of K_{sat}.

8.3.2 OVERSEER®

OVERSEER® is a DSS farm model (as developed by the Fertiliser Association of New Zealand Incorporated and AgResearch Limited – freely available online) used by farmers, advisors, and policy makers. It estimates the nutrient budget for a farm taking into account all inputs and outputs (including internal cycling of nutrients around the farm) to and from the farm. This model is widely used in New Zealand as a decision support tool by consultants and regional councils. This model was based on the knowledge obtained primarily from New Zealand and in consultation with farmers and consultants, and therefore, it is well suited for handling management practices and environmental conditions specific to New Zealand. Initially, this model was used to examine the impacts of land use and management practices on nutrient losses as a way of estimating fertiliser requirements, but recently the model has been used to monitor farm nutrient losses as a tool for applying new environmental policies (Wheeler et al., 2006). OVERSEER® has proved to be a very useful tool to examine nutrient use and flows within a farm (i.e. as products, fertiliser, effluent, supplements or transfer by animals) and to assess nutrient use efficiency and environmental impacts at the farm scale.

As mentioned earlier, it is an empirical model (i.e. based on observations or experiments) and empirical relationships, internal databases, and readily available data from existing farms are used by the model to calculate nutrient budgets at a farm scale level. The overall model contains separate sub-models dealing with pastoral, cropping, and horticulture enterprises. Of these various sub-models, the pastoral model is probably the most robust because of the large quantity of trial data that is available in New Zealand on grazed pasture systems

The model is designed so that the input data required is meaningful to farmers and is easily available. This is in contrast to LEACHN which is a data hungry model. For reliable performance, OVERSEER® requires that reasonable input data are given (Wheeler, 2009). This means that the amount of fertiliser required to support the given level of production needs to be known. The model assumes that the system is in equilibrium (i.e. model doesn't account for transition during a change from one practice to another). The model is designed to predict long term average behaviour of the system (as compared to LEACHN that can predict on a daily, weekly and monthly basis) and therefore it is not suitable for extreme case scenarios or a system in transition, or for estimating nutrient losses from particular years.

In OVERSEER® a dairy farm can be divided into blocks that can include effluent disposal and non-effluent disposal blocks. This provided the opportunity to assess whether the OVERSEER® model could be used to simulate N leaching losses at a LTS by assuming that it was operating as an effluent-irrigated block on a dairy farm. If the OVERSEER® model could be used at a LTS this would have considerable advantages as many regional councils already understand and accept the principals and predictions from the OVERSEER® model.

8.4 Methods and Materials

As noted in Section 8.1, the work described in this chapter covers four main areas.

1. A detailed analysis of the factors affecting the performance of the LEACHN model in simulating N dynamics in the CDC LTS. This was an extension of the work reported in earlier chapters and was intended to explain some of the differences observed between the measured and modelled data at the CDC LTS.
2. To use the LEACHN model to gain a greater understanding of the dynamics of N within the CDC site.
3. To use the OVERSEER® model to simulate the CDC LTS.
4. To use the LEACHN model to simulate another field data set (obtained from Massey University).

8.4.1 Assessment of the LEACHN Model

8.4.1.1 The Sensitivity of the LEACHN Model to Values of K_{min}, K_{nit}, and K_{den}

Initially, a base model was constructed using the CDC field data set for the medium effluent application rate on the tree plots (Chapter 6). Input parameters included the initial SMC, the clay, silt, and organic C contents, the bulk density, and the initial inorganic N concentrations for each layer. The values for these parameters were as described in Chapter 6 (Table 6.1) and were within the ranges used by other researchers (e.g. Ramos and Carbonell, 1991; Hutson and Wagenet, 1992; Johnsson et al., 1987). The quantities and timing of rainfall and effluent application (together with the associated N concentrations), the PET and mean daily temperature data were also inputted to the model. An N balance was then constructed based on the LEACHN model predictions.

In the base model, the K_{nit} and K_{den} default values were kept the same as in Chapter 6 for all the soil layers (Table 6.1). The K_{min} for humus-N was also the same as used in Chapter 6 (Table 6.1).

In the subsequent 7 scenarios all the parameters were kept constant except for K_{min}. The K_{min} value was then reduced in 7 steps from 0.7×10^{-4} to 0.017×10^{-4} day^{-1} (Table 8.1) and the effects on the levels of soil humus N at the end of the trial (Day 252) were noted. The lowest value of 0.017×10^{-4} day^{-1} corresponds with the lowest value used in the literature (Schmied et al., 2000).

For three of the K_{min} values (i.e. the highest, middle, and lowest) the predicted NO_3-N concentrations and predicted cumulative drainage volumes were also calculated.

The sensitivity of the LEACHN model to the values of K_{nit} and K_{den} was tested by increasing each value in three steps from 0.02 to 0.40 day^{-1} (Scenarios 9-11, Table 8.1) and observing the predicted effect on the N balance at the site on day 252. In these scenarios the K_{min} value was kept the same at 0.017×10^{-4} day^{-1}.

8.4.1.2 The Sensitivity of the LEACHN Model to Values for Soil Bulk Density

This investigation involved a comparison between Scenarios 7 and 8 in Table 8.1. In Scenario 8 all the parameters for the LEACHN model were the same as for Scenario 7, except for the value assigned to the soil bulk density. In Scenario 7 the bulk density values (ranging between 1.0 and 1.44 kg/dm^3 in the top 500 mm soil profile) used in the LEACHN model were those measured for different soil layers at the site. In Scenario 8 the bulk density values were increased by 30% (i.e. ranging between 1.3 and 1.87 kg/dm^3) and the predicted effect on the N balance was observed.

8.4.1.3 Ability of the LEACHN Model Parameterised Using Data from the Medium Effluent Irrigation Rate to Predict the Leachate NO_3-N Concentrations in Plots Receiving the High Rate of Effluent Application

Using the LEACHN model parameters in Scenario 7 of Table 8.1, (which had been derived from comparisons with the data from the medium rate of effluent application), the ability of the LEACHN model to then simulate the leachate NO_3-N concentrations measured at 500 mm soil depth in the plots receiving the high rate of effluent application was assessed.

8.4.2 Use of the LEACHN Model to Explore the Effect of Effluent Irrigation on the Dynamics of Water and N at the CDC LTS

It was assumed that the values for K_{min}, K_{nit}, and K_{den} used in scenario 7 were adequate to describe the NO_3-N concentrations at the CDC's land disposal site. The model was then used to investigate in more detail the effect of adding effluent to the soil at the land disposal site. The medium irrigation treatment of the tree plot was used for this analysis and this treatment was compared firstly with the model predictions of what would have occurred at the site under natural rainfall with no added effluent and secondly with the model predictions of what would have occurred if the medium irrigation strategy had been followed, but with pure water, rather than N-containing effluent.

8.4.3 The Ability of the OVERSEER® Nutrient Budget Model to Predict N Dynamics at the CDC LTS

The OVERSEER® model was used to compare the amount of N applied and leached at the CDC LTS. The OVERSEER® model was not originally designed to predict N leaching from a LTS such as the CDC LTS, but as noted in Section 8.3.2, the OVERSEER® model does have the ability to simulate N leaching from effluent disposal areas on dairy farms. In doing this the OVERSEER® model (version 6) does not have a facility to enter the application rate of effluent N directly into the model. Instead, the OVERSEER® model calculates the application rate of effluent N from the number of dairy cows in the herd on the dairy farm and the area of the effluent disposal area. For a given number of cows, if the size of the effluent area is large the application rate of effluent N per hectare will be low, and if the area of the effluent area is very small then the application rate will be correspondingly high.

This feature of OVERSEER® provided the opportunity to use the model to simulate N leaching on an LTS such as that at Carterton. This was done in the OVERSEER® model by setting up the CDC LTS as the effluent disposal block of a notional dairy farm. The notional "size" of the effluent block was adjusted so that the application rate of effluent organic N corresponded with the application rate of organic N in the low irrigation treatment on the pasture plot at the CDC LTS. The reason for matching the application rates of organic N in this way is explained below.

Although by manipulating the size of the notional effluent area it is possible in OVERSEER® to adjust the rate of N application on the effluent block to a value similar to that in a LTS, there are differences between DSE and municipal sewage effluent in the concentration of total N (which affects the amount of water added per kg of N) and the ratio of organic to inorganic N. Fortunately, the OVERSEER® model can account for these differences through its capacity to apply irrigation water containing inorganic N, as well as applying FDE.

DSE is more concentrated than the municipal sewage effluent at the CDC LTS. Its N concentration, on average, varies between 200 and 500 mg/L (Longhurst et al., 2000). Also, it is has been reported that organic N is the main component of total N in DSE (up to approximately 80% - Longhurst et al., 2000). In this analysis, it was assumed that OVERSEER® would use an average total N concentration for DSE of 250 mg/L, of which 80% would be in organic form.

As noted above, the OVERSEER® model (version 6) has the capacity to specify an amount of irrigation on a monthly basis, and also to specify the nutrient concentration in that irrigation water. If a N concentration in the irrigation water is specified, the OVERSEER® model assumes that this N is in the inorganic form. By using this feature of the OVERSEER® model it was possible to mimic the N application regime in the low effluent treatment at the CDC LTS exactly, in terms of the depth of water applied, the total N applied and the ratio of organic to inorganic N. This was done as follows:

1. The quantities of water, organic N and inorganic N applied in the low effluent treatment at the CDC LTS were calculated.
2. The application rate of DSE needed to apply the same amount of organic N as at the CDC LTS was then calculated (assuming that 80% of total N in DSE was organic-N) and the “size” of the notional effluent area was adjusted so that quantity of organic N was being applied.
3. This application rate of DSE was then supplying the correct amount of organic N, but the quantities of water and inorganic N applied were much less than at the CDC LTS. These shortfalls were then added in the OVERSEER® model through irrigation.
4. The amount of additional irrigation water required to match the total water applied in the low effluent treatment at the CDC LTS was then calculated.
5. The inorganic N concentration in this irrigation water needed to ensure that the same quantities of inorganic N were applied as in the low effluent treatment at the CDC LTS was then calculated.

When setting up the notional effluent block in the OVERSEER® model, it was specified in the model that all the pasture grown on the effluent irrigated block was taken off as supplements in order to mimic the “cut & carry” system at the CDC LTS.

The behaviour of N predicted by OVERSEER® for this farm scenario was then compared with the predictions of the LEACHN model. The annual rainfall data for the Carterton land treatment site was used in the scenarios.

Finally, when doing this comparison it was noticed that the OVERSEER® model predicted an overall increase in Humus N whereas the parameterised LEACHN model predicted a small decrease over the 252-day period. To explore this further, the LEACHN model was run with a slightly negative K_{min} value for the low effluent irrigation treatment on the pasture plot, and the effect of this on the comparison between the OVERSEER® and LEACHN models was observed.

8.4.4 Ability of the Parameterised LEACHN Model to Predict the Leachate NO_3-N Concentrations for another Field Data Set

The parameterised model (i.e. Scenario 7) was run using a field data set that was available for a DSE irrigation trial on pasture plots at Massey University. The trial site was on a poorly drained soil that had a mole and tile drain system installed. The tile drain discharge from each pasture plot was monitored for flow and samples collected at regular intervals for analysis of NO_3-N. The effluent irrigation, drainage and leachate NO_3-N concentration data was available for two irrigation events (20th Oct and 9th November, 2007). On each occasion the application of DSE caused the tile drains to flow, and flow was monitored for 3 days following each irrigation.

In the LEACHN model the soil profile was divided into 5 segments of 100 mm each. Input parameters required to run the model included the initial SMC, the clay, silt, and organic C contents, the bulk density, and the initial inorganic N concentrations for each layer. There was no rainfall during the two monitoring periods but average weekly PET at the site was entered into the model. The quantities and timing of the two effluent applications (together with the associated N concentrations) were entered into the model. There were two separate simulations for the two irrigation/drainage events (Table 8.12).

The plant N uptake was estimated from the measured PET during the monitoring periods and assuming that for each 1 mm of PET 12 kg DM/ha of pasture is produced and pasture has an N concentration of 4%. This rate of N uptake was then extrapolated to give a notional annual total N uptake of 150 kg N/ha for use in the LEACHN model.

As is commonly the case when testing models against data produced by other researchers, some of the necessary input data were not available. In this case the initial inorganic N concentrations and SMC for each layer were not available. Although the initial inorganic N concentrations immediately prior to the two irrigations had not been measured, the site had been extensively studied over several years and measurements of soil inorganic N had been made on several occasions. At the time of the year of these irrigations soil inorganic N levels tended to be low and reasonably constant (Hanly pers.com.). Therefore the initial soil inorganic N concentrations were assumed to be the same as in these earlier studies at the same site at a similar time of the year.

Soil moisture content can vary considerably at the time of the year these trials were conducted and so it was not possible to approximate it from earlier work at the site. Instead the initial SMC (%) values were adjusted (between 20 and 25% depending on the soil layer) in order to bring the predicted drainage volumes close to the measured values (Table 8.11). This meant that the accuracy of the water sub-model of the LEACHN model was not being assessed, but it did enable the prediction of N movement to be tested. The parameterised model was then run for 7 days for each of the two events.

8.5 Results and Discussion

8.5.1 Parameterisation of the LEACHN model

K_{min} : The effect of varying the value of the K_{min} rate constant on the N cycle over the 252 days of effluent irrigation is shown in Table 8.2. The predicted amount of NO_3-N leached reduced from 378 to 92.4 kg/ha in response to reducing K_{min} from $0.7x10^{-4}$ (i.e. the value that was used in Chapter 6) to $0.017x10^{-4}$ day^{-1}. As would be expected, changing the K_{min} value had no effect on predicted water movement with the same cumulative leachate (1080 mm) predicted for all the K_{min} values (Fig. 8.2). Therefore, the reduced quantity of NO_3-N leached was a consequence of significantly lower predicted NO_3-N concentrations in the leachate.

A reduction in the K_{min} value also has a significant impact on the predicted quantity of soil humus-N remaining in the soil on day 252. In the initial simulations reported in Chapter 6, the LEACHN model predicted large decreases in the quantities of humus-N when effluent was being applied. Studies by other workers have suggested that total C and N concentrations in soils at effluent irrigated LTS can increase or decrease depending on the local environmental conditions. Some studies have shown that the total C and N concentrations and microbial activities in soils increased due to N and C input in treated wastewater (Quin & Woods, 1978; Friedel et al., 2000). These increases have mainly been observed in long-term experiments.

In contrast, decreases in total soil C and N concentrations have been reported in various other studies and have been attributed to the following factors: (i) most of the effluent-N being in the mineral form (Feigin et al., 1991); (ii) fast mineralization of the effluent organic N fraction, consisting predominantly of dead algae (Snow et al., 1999b); (iii) maintenance of ideal conditions for mineralization of organic matter such as humidity, temperature, re-supply of O_2 (Myers et al., 1982; Stanford and Smith, 1972); (iv) the low C/N ratio of treated effluent (Fonseca et al., 2007); and (v) an increase of the microbial activity encouraging soil organic matter decomposition associated with possible priming effects due to high inputs of effluent-N (Barton et al., 2005).

The difference between initial and final soil humus-N concentrations was reduced from 410 to 9 kg/ha for K_{min} values of 0.7×10^{-4} and 0.017×10^{-4} day^{-1}, respectively. At this lowest value for K_{min} (0.017×10^{-4} day^{-1}) the LEACHN model predicts that the soil organic N pool will be reasonably stable. This is likely to be a more realistic prediction as the LTS was maintained under pasture or forest and there was no cultivation. It suggests that the K_{min} value (0.7×10^{-4} day^{-1}) used in Chapter 6 was too high for this land disposal site. This in turn led to predicted NO_3-N concentrations that were too high.

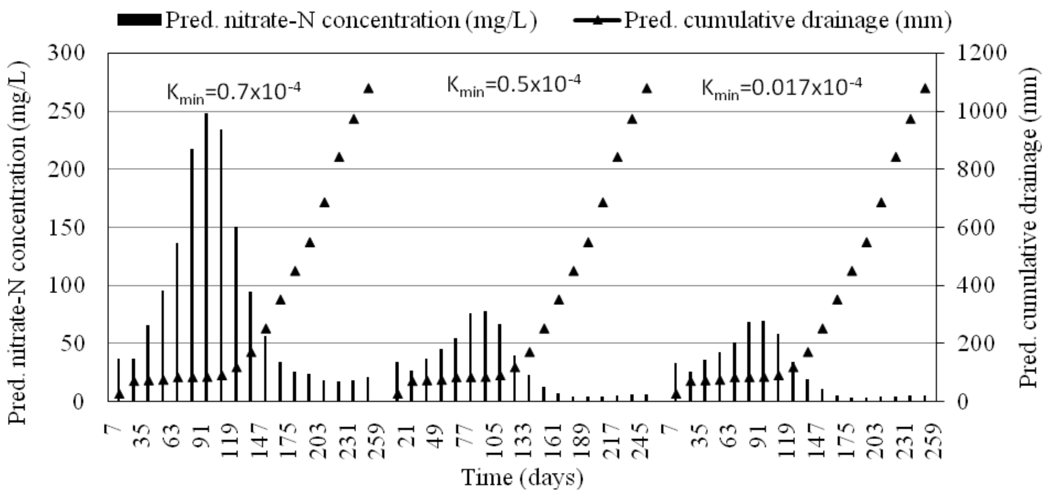


Fig. 8.2: The predicted leachate NO_3-N concentrations and cumulative drainage volume at 500 mm soil depth for different K_{min} values.

K_{nit} and K_{den}: The predicted mass of NO_3-N leached increased from 92 to 136 kg/ha (Table 8.3) in response to an increase in K_{nit} from 0.02 to 0.4 day^{-1}, respectively (Fig. 8.3). This was because ammonium was transformed to nitrate at a faster rate by the soil microbes through nitrification. The predicted mass of NO_3-N leached decreased only slightly from 92 to 89 kg/ha in response to an increase in K_{den} value from 0.02 to 0.4 day^{-1}, respectively. The amounts of N denitrified were 0.22 and 0.43 kg/ha (Table 8.4) for K_{den} values of 0.02 and 0.04 day^{-1} respectively. This shows that denitrification is not predicted to be a major mechanism of N loss and the predicted quantities of NO_3-N leached were therefore not sensitive to the value of $K_{den.}$

The parameterisation of N transformation rate constants shows that K_{min} was the key factor in controlling the N transformations in the soil profile, and had a larger effect on leachate NO_3-N concentrations than the K_{nit} and K_{den}. The K_{den} had the least effect on LEACHN model output (Fig. 8.4). The sensitivity of NO_3-N concentrations to the volatilisation rate constant was not included in the sensitivity analysis although N loss through volatilisation was significant. The amounts of N volatilised ranged between 34 and 27 kg/ha for K_{nit} values of 0.02 and 0.04 day^{-1}, respectively (Table 8.3).

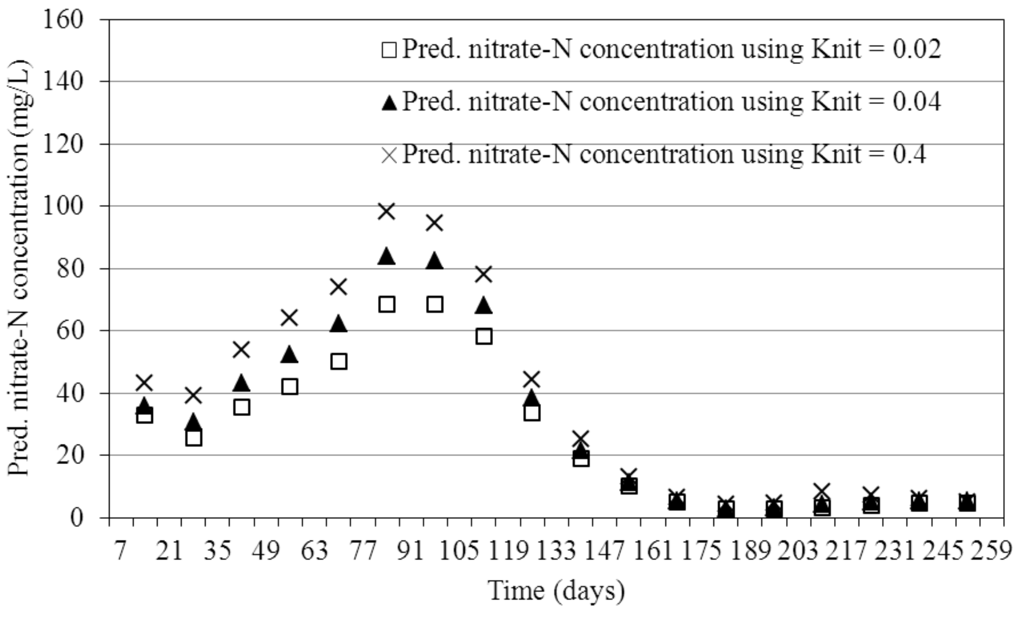


Fig. 8.3: The predicted leachate NO_3-N concentration (mg/L) over time for different K_{nit} values.

Table 8.1: Values of different rate constants used in the parameterisation of the LEACHN model (refer to Table 6.1 for the other parameters).

Scenario Names	Scenario number	Rate constants (per day) K_{nit}	K_{den}	Humus K_{min}	NO_3-N Leached (kg/ha)	Initial Humus (N_i - kg/ha)	Final humus in the profile (N_f - kg/ha)	Net N mineralised (kg/ha)	Comments
NTREEMM1	Base 1	0.02	0.02	0.00007	378	17922	17512	410	Highest K_{min}
NTREEMM2	2	0.02	0.02	0.00006	336	17922	17570	352	
NTREEMM3	3	0.02	0.02	0.00005	295	17922	17627	295	
NTREEMM4	4	0.02	0.02	0.00002	169	17922	17806	116	
NTREEMM5	5	0.02	0.02	0.00001	127	17922	17864	58	
NTREEMM6	6	0.02	0.02	0.000005	106	17922	17894	28	
NTREEMM7	7	0.02	0.02	0.0000017	92	17922	17913	9	Lowest K_{min} – Reasonably a balanced system
NTREEMM9	8	0.02	0.02	0.0000017	118	17922	17913	9	Bulk density was increased by 30%
NTREMM11	9	0.04	0.02	0.0000017	107	17922	17913	9	K_{nit} was increased from 0.02 to 0.04
NTREMM12	10	0.4	0.02	0.0000017	107	17922	17913	9	K_{nit} was increased from 0.04 to 0.4
NTREMM13	11	0.02	0.04	0.0000017	92	17922	17913	9	K_{den} was increased from 0.02 to 0.04
NTREMM14	12	0.02	0.4	0.0000017	92	17922	17913	9	K_{den} was increased from 0.04 to 0.4

Table 8.2: An N balance for the CDC land disposal site on day 252 for different K_{min} values, as calculated by LEACHN.

	Units	**K_{min} (day^{-1})**						
		$0.7x10^{-4}$	**$0.6x10^{-4}$**	**$0.5x10^{-4}$**	**$0.2x10^{-4}$**	**$0.1x10^{-4}$**	**$0.05x10^{-4}$**	**$0.017x10^{-4}$**
Total N applied	(kg/ha)	184	184	184	184	184	184	184
Humus-N_i	(kg/ha)	17922	17922	17922	17922	17922	17922	17922
Humus-N_f	(kg/ha)	17512	17570	17627	17806	17864	17894	17913
N mineralised	(kg/ha)	410	352	295	116	58	28	9
Initial NH_4-N	(kg/ha)	19	19	19	19	19	19	19
Initial NO_3-N	(kg/ha)	31	31	31	31	31	31	31
NH_4-N in the soil profile	(kg/ha)	68	61	54	33	25	21	19
NO_3-N in the soil profile	(kg/ha)	18	16.26	14.33	8.49	6.53	5.54	4.88
NH_4-N leached	(kg/ha)	34	29	25	11	7	5	3
NO_3-N leached	(kg/ha)	378	336	295	169	127	106	92
Plant uptake of NH_4-N	(kg/ha)	19	19	20	22	23	24	25
Plant uptake of NO_3-N	(kg/ha)	72	71.17	70.72	68.63	67.42	66.62	65.98
N volatilised	(kg/ha)	55	52	49	40	37	35	34
N denitrified	(kg/ha)	1.4	1.2	1.00	0.5	0.33	0.26	0.22
Cumulative leachate	(mm)	1080	1080	1080	1080	1080	1080	1080

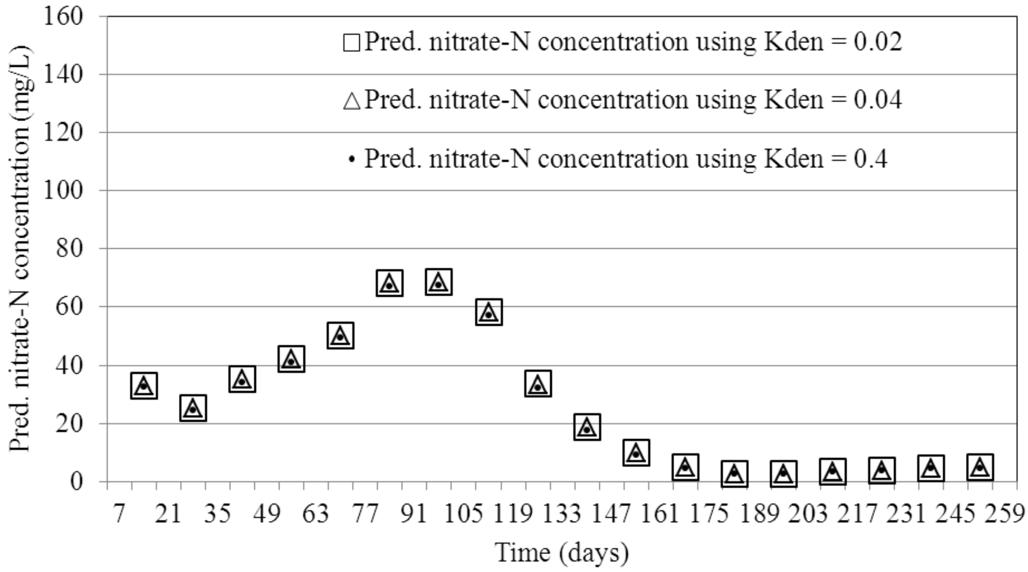


Fig. 8.4: The predicted leachate NO_3-N concentration (mg/L) over time for different K_{den} values.

Table 8.3: An N balance of the LTS using different K_{nit} values, as calculated by LEACHN.

	Units	**K_{nit} (day^{-1})**		
		0.02	**0.04**	**0.4**
Total N applied	(kg/ha)	184	184	184
Humus-N_i	(kg/ha)	17922	17922	17922
Humus-N_f	(kg/ha)	17913	17913	17913
Initial NH_4-N	(kg/ha)	19	19	19
Initial NO_3-N	(kg/ha)	31	31	31
NH_4-N in the soil profile	(kg/ha)	19	11	1.12
NO_3-N in the soil profile	(kg/ha)	5	6	6
NH_4-N leached	(kg/ha)	3	2	1
NO_3-N leached	(kg/ha)	92	107	136
Plant uptake of NH_4-N	(kg/ha)	25	18	4
Plant uptake of NO_3-N	(kg/ha)	66	73	86
Volatilised	(kg/ha)	34	27	9
Denitrified	(kg/ha)	0.2	0.3	0.4
Cumulative leachate	(mm)	1080	1080	1080

Table 8.4: An N balance of the LTS using different K_{den} values as calculated by LEACHN.

	Units	**K_{den} (day^{-1})**		
		0.02	**0.04**	**0.4**
Total N applied	(kg/ha)	184	184	184
Humus-N_i	(kg/ha)	17922	17922	17922
Humus-N_f	(kg/ha)	17913	17913	17913
Initial NH_4-N	(kg/ha)	19	19	19
Initial NO_3-N	(kg/ha)	31	31	31
NH_4-N in the soil profile	(kg/ha)	19	19	19
NO_3-N in the soil profile	(kg/ha)	5	5	5
NH_4-N leached	(kg/ha)	3	3	3
NO_3-N leached	(kg/ha)	92	92	89
Plant uptake of NH_4-N	(kg/ha)	25	25	25
Plant uptake of NO_3-N	(kg/ha)	66	66	66
Volatilised	(kg/ha)	34	34	34
Denitrified	(kg/ha)	0.2	0.4	4
Cumulative leachate	(mm)	1080	1080	1080

Effect of changing bulk density: The sensitivity analysis reported in Chapter 6 suggested that the LEACHN model was very sensitive to changes in soil bulk density. But, it was not clear what processes in the model were being so influenced by changes to soil bulk density. To explore this further the bulk density was increased by 30% and the impact on the predicted NO_3-N concentrations in the leachate and the N balance was studied.

The cumulative NO_3-N leached was increased by 26 kg/ha (from 92 to 118 kg/ha) on day 252 in response to an increase in bulk density. This was an increase of approximately 28%. The total accumulated drainage did increase slightly from 1080 to 1157 mm (Table 8.5 and Fig. 8.5) - an increase of 7%. But, it seems that the sensitivity of NO_3-N leached in the LEACHN model to bulk density was not in the water sub-model.

Table 8.5: An N balance of the LTS using measured and 30% increased bulk density different values, as calculated by LEACHN.

	Units	**Using measured bulk density values**	**Using bulk density values increased by 30% above the measured values**
Total N applied	(kg/ha)	184	184
Humus-N_i	(kg/ha)	17922	23271
Humus-N_f	(kg/ha)	17913	23620
Initial NH_4-N	(kg/ha)	19	25
Initial NO_3-N	(kg/ha)	31	41
NH_4-N in the soil profile	(kg/ha)	19	21
NO_3-N in the soil profile	(kg/ha)	5	4
NH_4-N leached	(kg/ha)	3	3
NO_3-N leached	(kg/ha)	92	118
Plant uptake of NH_4-N	(kg/ha)	25	21
Plant uptake of NO_3-N	(kg/ha)	66	70
Volatilised	(kg/ha)	34	25
Denitrified	(kg/ha)	0	0
Cumulative leachate	(mm)	1080	1157

Instead it appears that the apparent sensitivity of NO_3-N leaching to soil bulk density in the LEACHN model is an artefact of the way the sensitivity analysis was conducted. Increasing the bulk density, without changing any other parameters, would result in a predicted increase in humus-C per unit volume of soil. The humus-N in the soil is calculated from the C:N ratio. For example, if the initial bulk density was 1 kg/dm^3 and the initial concentration of humus-C was 1% w/w (i.e. 10 g humus-C per kg soil), then the initial humus-N concentration would be 1 g/kg w/w (based on a C:N ratio of 10:1) and 1 g/dm^3 w/v . If the bulk density was increased to 1.3 kg/dm^3 and the concentration of organic C was assumed to still be 1% w/w then the predicted humus-N concentration would still be 1 g/kg w/w but would be 1.3 g/dm^3 w/v. It is this apparent increase in humus N in the soil profile with increasing bulk density that leads to a predicted increase in N mineralisation and a predicted large increase in NO_3-N leaching.

Although at first glance the sensitivity of the LEACHN model to bulk density appears to be an artefact, it does highlight the importance of accurate measurement of bulk density. In the laboratory, soil C and N measurements are made routinely on a w/w basis and the bulk density of the soil must be known to convert these concentrations to a soil profile basis. As has been demonstrated in this sensitivity analysis, errors in the bulk density values and the resulting errors in the estimated amount of total N contained in the soil profile can have a large effect on the predicted quantities of NO_3-N leached.

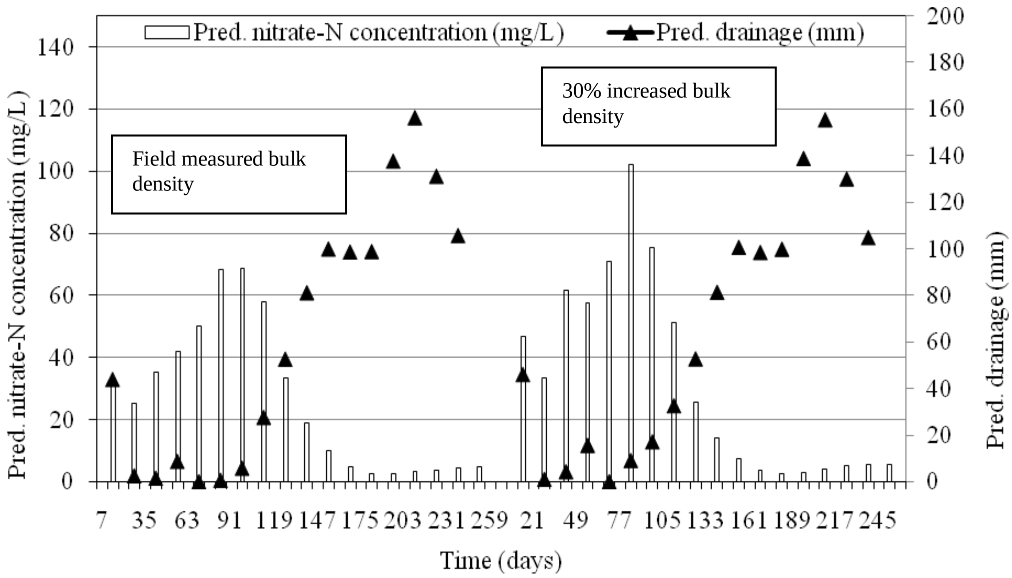


Fig. 8.5: The predicted drainage volume and NO_3-N concentrations for the field measured and 30% increased bulk density values (as used in LEACHN model).

8.5.2 Ability of the LEACHN Model Parameterised Using Data from the Medium Effluent Irrigation Rate to Predict the Leachate NO_3-N Concentrations in Plots Receiving the High Rate of Effluent Application

The predicted and measured leachate NO_3-N concentrations were compared for the high irrigation treatment for the tree plots (Fig. 8.6) using the LEACHN model with the parameter values detailed in Scenario 7 (Table 8.1). The predicted and measured leachate NO_3-N concentrations were reasonably close, and much better than in the initial simulations described in Chapter 6.

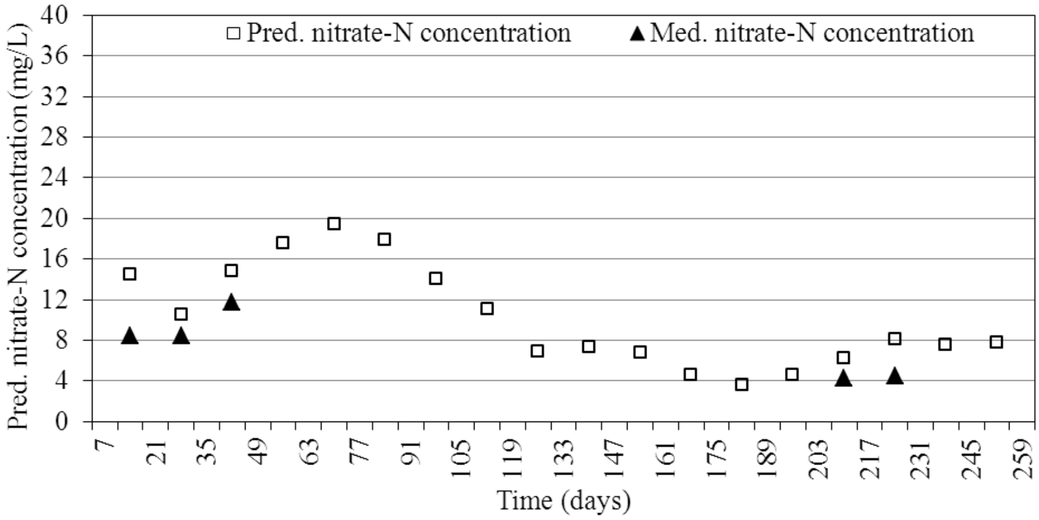


Fig. 8.6: Comparison of predicted and measured leachate NO_3-N concentration (mg/L) for the high irrigation treatment on the tree plot.

8.5.3 The Ability of the LEACHN Model to Predict N Dynamics at the CDC LTS

This study used the LEACHN model to investigate the fate of the added water and N in the effluent at the CDC LTS (Table 8.6). The model predicted that under natural rainfall and with no effluent irrigation (Scenario S1 in Table 8.6) there would be very little accumulated leachate from day 1 to 196. However, there was some leaching from day 210 onwards (a total of 177 mm during winter). The predicted leachate NO_3-N concentrations were very low (< 1 mg/L) and the cumulative mass of NO_3-N leached for this scenario was < 1 kg/ha (Fig. 8.8) for the simulation period (260 days). The reason for these low predicted NO_3-N concentrations in the drainage water are discussed later in this section. The predicted net N mineralised in scenario S1 was 6 kg/ha.

When LEACHN was used to simulate the addition of irrigation water containing no N at the medium effluent application rate (Scenario S2 in Fig. 8.7), the irrigation caused drainage from day 28 onward (i.e. during summer). The accumulated drainage for scenario S2 was 1080 mm as compared to 177 mm for S1. The amount of water added (as rainfall and irrigation water) in scenario S2 was 2256 mm as compared to 681 mm added in scenario S1 as rainfall only. In Scenario 2 the predicted leachate NO_3-N concentrations were high (up to 32.0 mg/L) at the start of the drainage period but then dropped away to much lower concentrations (1 mg/L) towards the end of the simulation period (Fig. 8.7). The cumulative mass of NO_3-N leached was just over 20 kg/ha (Table 8.6 and 8.7; Fig. 8.7 and 8.8), and net N mineralised was 9 kg/ha (i.e. 3 kg/ha more than when no irrigation water was applied).

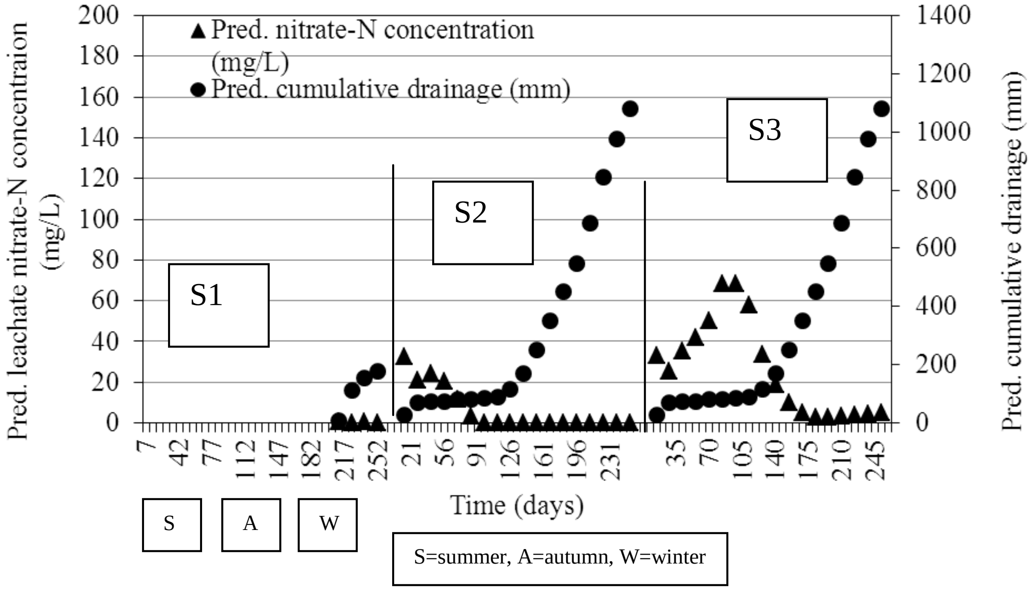


Fig. 8.7: The predicted leachate NO_3-N concentrations and cumulative drainage volume for three irrigation scenarios (S1 = rainfall alone and no effluent irrigation, S2 = rainfall and irrigation with effluent containing no N, and S3 = rainfall and irrigation with effluent that contains N) using the LEACHN model.

When the normal N-containing effluent was added (Scenario S3 in Fig. 8.7) LEACHN predicted that the cumulative drainage would be 1080 mm (the same as in scenario S2). The NO_3-N concentrations at the commencement of drainage were similar to those at the start of scenario S2 (32 mg/L) but then increased to a maximum of 68.6 mg/L after about 90 days. From then on, the NO_3-N concentrations dropped steadily to a minimum value of 2.75 mg/L. The net N mineralised was the same as in scenario 2 (i.e. 9 kg/ha) but the cumulative mass of

NO_3-N leached was greater (92 kg/ha – Table 8.6 and 8.7; Fig. 8.8) than that in scenario 2 (20 kg/ha; Fig. 8.8). The pattern of NO_3-N concentrations over time was reflected in the cumulative loads of NO_3-N which were similar to those in scenario S2 until about day 115. After this date however, the cumulative load of leached NO_3-N in scenario S3 increased rapidly – presumably reflecting the arrival of the first of the effluent N at the drainage depth.

Table 8.6: The amount drainage, NO_3-N leached, and net N mineralised in response to three different irrigation scenarios, as calculated by LEACHN.

Irrigation Scenarios	**Cumulative mass of NO_3-N Leached**	**Cumulative drainage**	**Net N mineralised**
	(kg/ha)	**(mm)**	**(kg/ha)**
S1 - Just rainfall and no effluent	< 1	177	6
S2 - Rainfall and effluent with no N	20	1080	9
S3 - Rainfall and effluent with N	92	1080	9

Although LEACHN predicted that more NO_3-N would be leached in scenario S3 (92 kg/ha) than in scenario S2 (20 kg/ha) the difference (72 kg/ha) was much less than the amount of N added in the effluent (184 kg/ha; Table 8.7). The net mineralisation was predicted to be the same in both scenarios and so the remainder of the added N was divided between increased volatilisation and plant uptake, and a greater amount of inorganic N remaining in the soil profile (Table 8.7).

The predicted differences in plant N uptake and soil mineral N concentrations are interesting. It is apparent from Tables 8.7 and 8.8, and Fig. 8.9 that the LEACHN model predicts that the quantities of inorganic N in the profile will decrease during the period of the simulation – particularly in scenarios S1 and S2. This is a result of plant uptake being greater than the supply of inorganic N through mineralisation. By day 56 in scenario S1 (Table 8.8), the amount of inorganic N in the upper layers of the soil had been depleted but most of the inorganic N in the lower layers remained. In contrast, by day 56 in scenario S2, irrigation had moved inorganic N down the soil profile and some had leached beyond the root zone. As a result inorganic N levels were low throughout the soil profile and plants were unable to access sufficient N for maximum growth. As a result plant uptake by day 56 was predicted to be lower in scenario S2 than in scenario S1 (data not presented).

By day 126 virtually all the inorganic N in the profile in scenarios S1 and S2 had been exhausted by a combination of plant uptake (S1 and S2) and leaching (S2). As a result the predicted plant N uptake in scenarios S1 and S2 was considerably less than in scenario S3 in which effluent containing N was irrigated on to the soil. Such a plant response to added N is commonly observed in New Zealand agricultural systems.

Table 8.7: A quantitative N balance for day 252 for three irrigation scenarios, as calculated by LEACHN.

	Units	**Irrigation Scenarios**		
		Rainfall alone (S1)	**Rainfall and Irrigation without N (S2)**	**Rainfall and Effluent Irrigation with N (S3)**
Total N applied	(kg/ha)	0	0	184
Humus-N_i	(kg/ha)	17922	17922	17922
Humus-N_f	(kg/ha)	17916	17913	17913
Initial NH_4-N	(kg/ha)	19	19	19
Initial NO_3-N	(kg/ha)	31	31	31
NH_4-N in the soil profile	(kg/ha)	0.4	0.6	19
NO_3-N in the soil profile	(kg/ha)	0.00	0.05	5
NH_4-N leached	(kg/ha)	0.2	1	3
NO_3-N leached	(kg/ha)	< 1	20	92
Plant uptake of NH_4-N	(kg/ha)	8	8	25
Plant uptake of NO_3-N	(kg/ha)	47	30	66
Volatilised	(kg/ha)	<1	<1	34
Denitrified	(kg/ha)	0.00	0.05	0.22
Cumulative leachate	(mm)	177	1080	1080

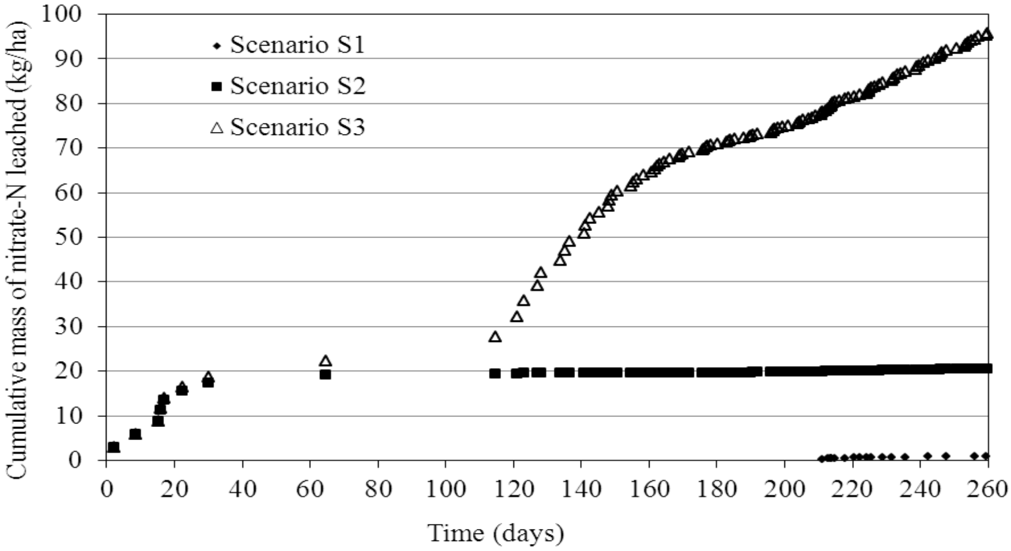


Fig. 8.8: The cumulative mass of NO_3-N leached over the simulation time period of 260 days for the three irrigation scenarios.

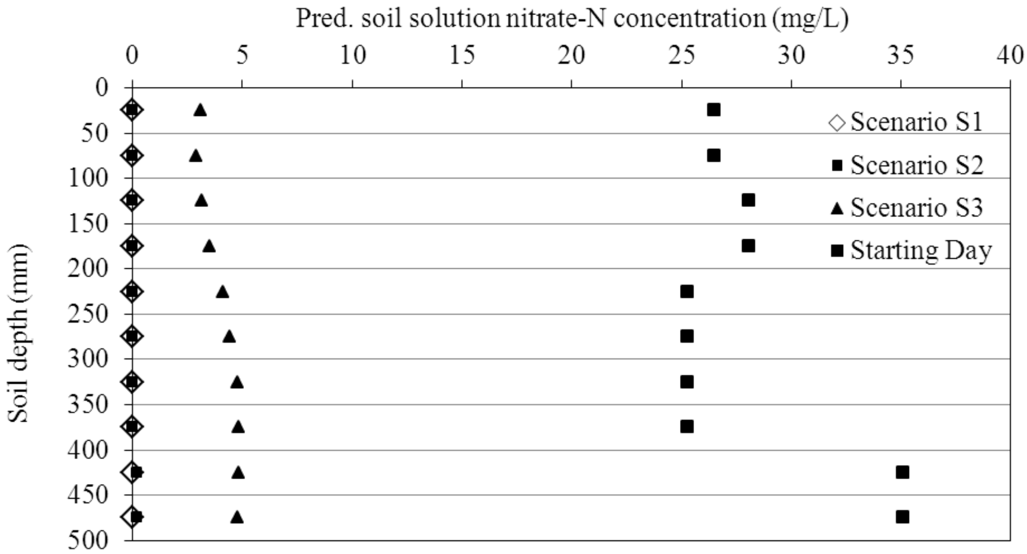


Fig. 8.9: Predicted soil solution NO_3-N concentrations down the soil profile on the starting day and day 252 for the three irrigation scenarios (i.e. S1 = just rainfall and no effluent irrigation, S2 = rainfall and irrigation without N, and S3 = rainfall and irrigation with effluent that contains N).

Table 8.8: The distribution of NO_3-N in the soil profile (mg/dm^3) on day 0, 56, 126 and 252 for the three irrigation scenarios (i.e. S1 = just rainfall and no effluent irrigation, S2 = rainfall and irrigation without N, and S3 = rainfall and irrigation with effluent that contains N).

Day no.	**Soil depth**	**Soil NO_3-N for scenario S1**	**Soil NO_3-N for irrigation S2**	**Soil NO_3-N for irrigation S3**
	(mm)	**(mg/dm^3)**	**(mg/dm^3)**	**(mg/dm^3)**
0	100	6.63	6.63	6.63
	200	7.02	7.02	7.02
	300	5.31	5.31	5.31
	400	5.31	5.31	5.31
	500	7.03	7.03	7.03
56	100	1.44	0.12	4.16
	200	2.94	0.47	4.85
	300	4.97	1.50	6.56
	400	5.71	2.94	7.90
	500	6.61	4.36	9.09
126	100	0.00	0.00	2.90
	200	0.00	0.00	3.82
	300	0.02	0.00	5.75
	400	0.10	0.00	7.29
	500	0.18	0.00	8.11
252	100	0.00	0.00	0.68
	200	0.00	0.00	0.81
	300	0.00	0.00	1.10
	400	0.00	0.03	1.18
	500	0.00	0.05	1.21

8.5.4 The Ability of the OVERSEER® Nutrient Budget Model to Predict N dynamics at the CDC LTS

The focus of this exercise was to explore if it is possible to use the OVERSEER® model (with some modifications to input parameters) to simulate N dynamics at an effluent irrigated LTS (similar to CDC). As described in Section 8.3.4, the amounts of water, organic N and inorganic N applied in the low irrigation treatment of the pasture plot at CDC land disposal site (Table 8.9) were entered into the OVERSEER® model as a notional combination of DSE and irrigation water containing a low concentration of inorganic N

Table 8.9: The amounts of water, inorganic and organic N that were applied as DSE and irrigation in the OVERSEER® model to create a notional effluent disposal site similar to the low irrigation treatment pasture plot at CDC.

Input	**Unit**	**CDC low effluent irrigation treatment**		**OVERSEER® model**	
Water	mm/year	Effluent application	1560	DSE application	67
				Irrigation	1493
Total water	mm/year		**1560**		**1560**
Organic N	kg/ha/year	Effluent application	131	DSE application	134
Total organic N	kg/ha/year		**131**		**134**
Inorganic N	kg/ha/year	Effluent application	180	DSE application	34
				Irrigation	146
Total inorganic N	kg/ha/year		**180**		**180**

The outputs from the OVERSEER® model were compared with the outputs from the LEACHN model parameterised as described in Section 5.5.1 (Table 8.10). Both models predicted similar annual total leachate (1629 mm and 1682 mm for the LEACHN and OVERSEER® models respectively). The inputs of inorganic N in the applied effluent (and irrigation) were as measured at the CDC LTS and were therefore the same for both models. Similarly, in both models the plant uptake of N was set at 132 kg N/ha which was measured at the CDC site. Given these similar inputs, the OVERSEER® model predicted a leaching loss of 69 kg N/ha, which was reasonably similar to the 85 kg N/ha predicted by the LEACHN model. The OVERSEER® model predicted very small emissions of N to the atmosphere (4 kg N/ha) whereas the LEACHN model predicted that 32 kg N/ha would be lost to the atmosphere. Most of the predicted emissions in the LEACHN model were as ammonia volatilisation.

Table 8.10: Comparison of the predictions from the LEACHN and OVERSEER® models in terms of N leached (kg/ha/year) and average NO_3-N concentrations (mg/L) when the amount of N applied is the same.

Inputs	**Units**	**LEACHN model**	**OVERSEER® model**
Inorganic N applied	kg/ha	180	179
Organic N applied	kg/ha	(131)	132
N fixation	kg/ha	-	2
<u>Total N inputs</u>	<u>kg/ha</u>	<u>311</u>	<u>313</u>
Outputs			
N leached	kg/ha	85	69
Plant N uptake	kg/ha	132	133
Loss to atmosphere	kg/ha	32	4
<u>Total N outputs</u>	<u>kg/ha</u>	<u>249</u>	<u>206</u>
<u>Change in soil N</u>			
Soil inorganic N	kg/ha	-39	0
Soil organic N	kg/ha	(101)	107
Total change in soil N	kg/ha	(62)	107
<u>Water</u>			
Rainfall	mm		
Effluent and irrigation	mm	1560	1560
Cumulative leachate	mm	1629	1682
Average NO_3-N concentration	mg/L	5.2	4.1

The OVERSEER® model predicted that there would be no change in soil inorganic N over the year but that the organic N would increase by 107 kg N/ha. The LEACHN model predicted a decrease in inorganic N, but prediction of changes in organic N by the LEACHN model is difficult. At the CDC LTS the effluent applied contained both inorganic and organic N (Table 8.9) and the organic N was included in the inputs to the OVERSEER® model. It is however, not easy to include inputs of organic N in the LEACHN model, and therefore only the inorganic N content of the CDC effluent was included in the LEACHN simulations described throughout the thesis.

In Table 8.10 a notional N balance was calculated for the LEACHN model by adding the input of organic N (number in brackets) and then calculating the final increase or decrease in organic N by difference. When this was done, the predicted increase in organic N was less than predicted by the OVERSEER® model – reflecting the greater losses by leaching and volatilisation.

It would therefore appear that the existing OVERSEER® model can be adapted to simulate a LTS and gives a similar prediction to a more detailed process-based model such as LEACHN. This could be of value because regional councils are increasingly recognising the use of OVERSEER® as a monitoring tool to demonstrate compliance with environmental regulations.

8.5.5 Ability of the Parameterised LEACHN Model to Predict the Leachate NO_3-N Concentrations for another Data Set

The initial SMC and soil inorganic N concentrations entered into the LEACHN model for the two irrigation events are given in Table 8.11. The amounts and inorganic N concentrations of the DSE for the two irrigation events are given in Table 8.12. The LEACHN simulations were continued until drainage was predicted to cease. This was 2 days for irrigation 1 and 3 days for irrigation 2. The predicted quantity of drainage in the two simulations is also given in Table 8.12.

Table 8.11: Initial SMC and soil inorganic N concentrations in the LEACHN model simulations of the two irrigation events.

Soil layers	**Initial SMC for event 1**	**Initial SMC for event 2**	**Initial soil inorganic N Conc. (mg/kg) for both events**	
(mm)	**% (v/v)**	**% (v/v)**	**NH_4-N**	**NO_3-N**
100	0.20	0.20	10	2
200	0.20	0.20	8	1
300	0.20	0.20	7	0.5
400	0.22	0.22	7	0.5
500	0.24	0.23	7	0.5

Table 8.12: The amounts and inorganic N concentrations of DSE applied in the parameterised LEACHN model, and the predicted total drainage volume (mm) by the end of the simulations for the two irrigation events.

Events	Amount of DSE added	DSE N Conc. (mg/L)		Predicted total drainage volume (mm) at the end of simulation
	(mm)	NH_4-N	NO_3-N	
1	5.4	96.5	0.16	1.8
2	27.5	105.3	0.25	9.3

The predicted total drainage for irrigation event 1 was 1.8 mm as compared to 1.72 mm measured at the site (Table 8.13). For the irrigation event 2, the predicted amount of total drainage was 9.3 mm, compared to the measured drainage of 9.71 mm at the site.

The predicted leachate concentrations of NO_3-N from the two drainage events were somewhat higher than the measured values (Table 8.13), but were of a similar order of magnitude. This is encouraging because it suggests that by using historical soil N data the LEACHN model can predict the likely NO_3-N concentrations in leachate following effluent irrigation in subsequent years. The difference between the measured and modelled values is discussed further below.

Table 8.13: The actual and predicted (by the LEACHN model) drainage volume and NO_3-N concentrations in drainage from irrigation of a pasture plot at Massey University.

Irrigation Events	Date	Effluent Application Depth	Measured Data		Predicted Data	
			Drainage Volume	Leachate NO_3-N Conc.	Drainage Volume	Leachate NO_3-N Conc.
		(mm)	(mm)	(mg/L)	(mm)	(mg/L)
1	20/10/07	5.40	1.72	3.46	1.8	5.58
2	9/11/07	27.50	9.71	3.41	9.3	6.85

Table 8.14: A comparison of soil and soil solution NH_4-N and NO_3-N concentrations down the soil profile at the beginning and end of simulations for the two irrigation events.

Irrigation Events	Soil depth (mm)	Soil NH_4-N concentration (mg/kg)	Soil solution NH_4-N concentration (mg/L)	Soil solution NO_3-N concentration (mg/L)
Event 1- Initial	25	10		
	75	10	3.1	11
	125	8		
	175	8	2.5	6.8
	225	7		
	275	7	2.2	3.6
	325	7		
	375	7	2.2	3.5
	425	7		
	475	7	2.2	3.3
Final	25	17		
	75	10	3.1	9.7
	125	7.6		
	175	7.6	2.4	8.5
	225	6.7		
	275	6.7	2.1	6.0
	325	6.7		
	375	6.7	2.1	5.8
	425	6.6		
	475	6.6	2.1	5.6
Event 2 - Initial	25	10		
	75	10	3.1	11
	125	8		
	175	8	2.5	6.8
	225	7		
	275	7	2.2	3.6
	325	7		
	375	7	2.2	3.5
	425	7		
	475	7	2.2	3.4
Final	25	43		
	75	17.5	5.5	8.8
	125	9.2		
	175	7.7	2.4	7.8
	225	6.6		
	275	6.6	2.1	7.4
	325	6.6		
	375	6.6	2.1	7.1
	425	6.5		
	475	6.5	2.1	6.9

Table 8.15: A cumulative total N mass balance at the start and end of simulation for the two irrigation events. (**Note:** The irrigation was applied on day 1 for both events).

Event 1	**Units**	**Day 0**	**Day 1**	**Day 2**
Cumulative irrigation applied	(mm)	0	5.4	5.4
Cumulative NH_4-N applied	(kg/ha)	0	5.2	5.2
Humus-Ni	(kg/ha)	7420	7420	7420
Humus-Nf	(kg/ha)	7420	7420	7420
Initial NH_4-N	(kg/ha)	53.9		
Initial NO_3-N	(kg/ha)	5.8		
Cum. NH_4-N in the soil profile	(kg/ha)	53.9	57.4	55.8
Cum. NO_3-N in the soil profile	(kg/ha)	5.8	7.0	7.9
Cum. NH_4-N leached	(kg/ha)	0	0.02	0.04
Cum. NO_3-N leached	(kg/ha)	0	0.04	0.08
Cum. plant uptake of NH_4-N	(kg/ha)	0	0.15	0.27
Cum. plant uptake of NO_3-N	(kg/ha)	0	0.25	0.54
Cum. volatilised	(kg/ha)	0	0.21	0.44
Cum. denitrified	(kg/ha)	0	0	0.0
Cumulative leachate	(mm)	0	1.0	1.8
Event 2	**Units**	**Day 0**	**Day 1**	**Day 3**
Cumulative irrigation applied	(mm)	0	27.5	27.5
Cumulative NH_4-N applied	(kg/ha)	0	28.9	28.9
Humus-Ni	(kg/ha)	7420	7420	7420
Humus-Nf	(kg/ha)	7420	7420	7420
Initial NH_4-N	(kg/ha)	53.9		
Initial NO_3-N	(kg/ha)	5.8		
Cum. NH_4-N in the soil profile	(kg/ha)	53.9	80.3	75.1
Cum. NO_3-N in the soil profile	(kg/ha)	5.8	7.4	9.8
Cum. NH_4-N leached	(kg/ha)	0	0.03	0.2
Cum. NO_3-N leached	(kg/ha)	0	0.05	0.5
Cum. plant uptake of NH_4-N	(kg/ha)	0	0.23	0.63
Cum. plant uptake of NO_3-N	(kg/ha)	0	0.10	0.52
Cum. volatilised	(kg/ha)	0	0.6	1.97
Cum. denitrified	(kg/ha)	0	0	0.0
Cumulative leachate	(mm)	0	1.2	9.3

A comparison of modelled soil and soil solution NH_4-N and NO_3-N concentrations at the start and end of simulations for the two events is given in Table 8.14. The initial soil solution NH_4-N and NO_3-N concentrations for both events were similar (refer to Table 8.14). For the top layer, the soil solution NO_3-N concentrations at the end of the simulation were slightly lower for both events than the initial values. The difference between the initial and final soil NO_3-N concentrations was slightly greater for irrigation 2 than irrigation 1 (Table 8.14). This was probably due to a dilution effect, because the effluent had very low NO_3-N concentrations and more irrigation was applied in event 2.

Further down the profile, the LEACHN model predicted that the final soil solution NO_3-N concentrations would be increasing during the simulation period (Table 8.14). A further analysis (data not presented) was undertaken to see why this is the case. The analysis showed that nitrification of soil ammonium N in the bottom layers could account for the observed increase in soil NO_3-N concentration in the bottom layers.

In these simulations it was assumed that the K_{nit} value was the same (0.02 per day) for the entire soil profile. In reality under field conditions, it is highly likely that the nitrification rate constant decreases down the soil profile. In long term simulations this probably does not matter too much as the ammonium N will be nitrified eventually and leached. However in short term simulations such as these, it does appear as if the predicted NO_3-N concentrations in the leachate are quite sensitive to the nitrification rate constant in the subsoil. It is interesting to note from Tables 8.13 and 8.14 that the predicted NO_3-N concentrations in the leachate were similar to the final soil solution NO_3-N concentrations in the lower soil depths. This is not surprising. If the assigned K_{nit} values at depth had been close to zero then the soil solution NO_3-N concentrations would have remained close to their original values and the predicted leachate NO_3-N concentrations would have been very close to the observed values of 3 - 4 mg/L.

The initial soil NH_4-N was 53.9 kg/ha prior to irrigation 1 (Table 8.15), and this increased by 3.5 kg/ha to 57.4 kg/ha when 5.2 kg/ha of NH_4-N was added in the DSE on day 1. The rest of the applied NH_4-N was volatilised (0.21 kg/ha), taken up taken by plants (0.15 kg/ha), and nitrified. Very little was leached (< 0.1 kg/ha). The next day the cumulative soil NH_4-N had dropped slightly from 57.4 to 55.8 kg/ha, as a result of uptake by the plants (0.27 kg/ha) or volatilised (0.44 kg/ha) (Table 8.15). For irrigation 2, the amount of initial soil NH_4-N was same as for irrigation 1 (Table 8.15). The amount of NH_4-N applied as effluent in irrigation 2 was 28.9 kg/ha and the soil NH_4-N increased to 80.3 kg/ha, and then dropped to 75.1 kg/ha on day 3 (at the end of simulation) as some of soil N was used by plants (0.63 kg/ha) and volatilised (1.97 kg/ha). A small amount (0.2 kg/ha) was leached.

It is apparent (Table 8.14) that virtually all the added NH_4-N remained in the top 100 mm of soil – even in irrigation 2 when 27.5 mm of effluent was applied. This suggests that the composition of the effluent has little effect on NO_3-N concentrations in the drainage water in short term simulations such as this. In this simulation, the soil NO_3-N concentration mainly depends on the concentrations of NO_3-N already present in the soil solution at depth.

8.6 Summary and Conclusions

A detailed evaluation of the LEACHN model was under taken in this Chapter. Based on this evaluation the following conclusions can be drawn:

1. The LEACHN model is sensitive to the K_{min} values and this parameter has a significant impact on soil humus-N and the amount of NO_3-N leached. The size and rate of breakdown of humus-N are important parameters in determining the amount of N leached, and more work is required to identify appropriate values for K_{min} in LTSs under New Zealand conditions.
2. The total amount of NO_3-N leached was slightly affected by K_{nit} values with the predicted amount of NO_3-N leached increasing with increasing K_{nit} value. However, the K_{den} value had no significant impact on the total amount of NO_3-N leached, even when its value was increased by 100%. It was apparent that denitrification was not predicted by LEACHN to be a major mechanism of N loss but volatilisation seemed to be a significant mechanism of N loss and it varied considerably as the K_{nit} value changed.

3. The apparent sensitivity of predicted NO_3-N concentrations and the amount of NO_3-N leached to changes in the soil bulk density was due to an artefact within the sensitivity analysis. Increasing the soil bulk density implied a greater weight of soil within the soil profile and a consequent increase in the weight of soil constituents – such as humus-N. This, in turn resulted in a predicted increase in N mineralisation, and an increase in NO_3-N leaching as predicted by the model. This analysis did however highlight the need for accurate measurements of bulk density as inputs to the LEACHN model.
4. The irrigation scenario showed that the accumulated leachate was almost nil from day 1 to 196 under natural rainfall (with no N effluent irrigation). There was a total of 177 mm leaching during winter from day 210 onwards and leachate NO_3-N concentration were quite low (< 1 mg/L) during that time. The accumulated mass of NO_3-N leached in the first 177 mm of S1 and S2 was 0.85 and 19.5 kg/ha.
5. The soil solution NO_3-N concentrations did not change greatly in the top 500 mm soil depth when more water (i.e. rainfall and irrigation with water containing no N) was added to the system and consequently the soil N was leached and therefore there was a lack of available soil N for plants uptake in irrigation scenario 2. The model predicted that more nitrate would be in the soil profile and the soil solution NO_3-N concentration would be significantly increased (up to 5.0 mg/L) in irrigation scenario 3 (i.e. rainfall and irrigation with effluent containing N).
6. The LEACHN model was useful in helping to understand the fate of water and N added to a LTS by comparing the scenario with effluent added to other scenarios in which the only input was natural rainfall, or rainfall plus irrigation with water that contained no added N.
7. The OVERSEER® model can be modified to simulate a LTS. The predicted amount of N leaching by the OVERSEER® model (69 kg/ha) was reasonably similar to that of predicted by the LEACHN model (85 kg/ha).
8. In the simulations reported in this thesis the focus in the LEACHN model was on the fate of inorganic N added in effluent. There is however the capacity within the LEACHN model to include the addition of plant residues, manure and urea. It would be useful in future studies to explore ways in which the

ability of LEACHN to include different types of N input could be used to mimic more exactly the management regimes at LTSs.

9. The testing of the LEACHN model using another dataset showed that the model had the ability to predict leachate NO_3-N concentrations in short term effluent irrigation events. The key driver of the leachate NO_3-N concentration in such events is the initial NO_3-N concentration in the soil solution near the base of the soil profile. This information is not usually routinely available, but for long periods of the year in winter and spring the concentrations of NO_3-N in the soil solution are similar from year to year. This provides the opportunity to use historic soil test data and achieve reasonably accurate predictions of likely leachate NO_3-N concentrations.
10. In short term simulations the rate of nitrification throughout the soil profile is very important in determining how much NO_3-N will be leached. More research is required on nitrification rates at deeper soil depths.
11. The strength of LEACHN lies in its ability to predict what would happen to soil N down the profile when irrigation with or without N is added at a LTS and when, where and how much NO_3-N is going to leach. This is very helpful information for the operators/managers of a LTS. Therefore, a LEACHN based DSS could be a useful decision making tool for the operator(s) or manager(s) of a land treatment site.

CHAPTER 9

SUMMARY AND CONCLUSIONS

9.1 Summary

The goal of this study was to develop and assess decision support tools that could assist managers to reduce the risk of groundwater contamination at a land treatment system. A review of the literature revealed that there was limited published information on the effects of land treatment of effluent on NO_3-N contamination of groundwater in New Zealand. Although there is some information on NO_3-N leaching through soils, quantitative links between these leaching processes and contamination of groundwater have not been well established either in New Zealand or overseas.

An exploratory field scale investigation was undertaken at a land treatment site of Carterton District Council (Chapter 3) to examine how the irrigation of sewage effluent on trees and pasture at different application rates affected the concentration of NO_3-N in groundwater. A trickle irrigation system was designed to apply sewage effluent at three hydraulic loading rates. The hydraulic loading rate used for design is based on the more restrictive of two limiting conditions - the capacity of the soil profile to transmit water (soil permeability) or the limiting constituent concentration in the effluent applied. In municipal wastewater land treatment systems, N is often usually the limiting constituent when protection of potable groundwater is a concern. The lowest hydraulic loading rate in this study was chosen so that the N loading (150 kg N/ha) complied with the limit imposed by the local regional council. .

The groundwater NO_3-N concentrations at the site varied spatially between monitoring wells and also over time. Some of this spatial variation appeared to be related to the proximity of individual monitoring wells to the effluent ponds or the river. Despite this spatial variation, NO_3-N concentrations in groundwater across the site were usually less than the MPL throughout most of the year. In late winter however, NO_3-N concentrations in the groundwater increased and frequently exceeded the MPL. This effect was most marked on the plots receiving the higher rates of effluent application, but even on the plot receiving the low effluent application rate (150 kg N/ha) the groundwater NO_3-N concentration equalled the MPL.

At this site it was not possible to obtain soil solution samples on a regular basis using suction cup samplers. The reasons for this were not clear but it does place a question over the feasibility of these measurements for management purposes.

The next phase of the study was to develop a simple hydrological model to describe the short term effects of applying effluent to a LTS and to identify the parameters that are practicable and feasible to monitor at a LTS in order to implement such a model. It was suggested that simply by monitoring soil moisture content and using forecasts of likely future rainfall a LTS can be managed in such a way that leaching of NO_3-N immediately after effluent application is minimised (Fig. 4.1).

To provide longer term predictions of likely leaching losses of NO_3-N the LEACHN model was chosen. As a first step, this decision support model was tested in a short-term laboratory scale experiment using a different soil (Manawatu fine sandy loam) and effluent (dairy shed effluent) than the soil and effluent at the Carterton site. The details of calibration and testing of the LEACHN model are presented in Chapter 5.

In the laboratory scale testing of the LEACHN model, it proved possible to choose values for the model parameters that enabled it to reproduce closely the leachate NO_3-N concentrations in the dataset used for calibration. However, when the calibrated LEACHN model was then used to predict subsequent NO_3-N leaching from the lysimeters, the model consistently overestimated the likely NO_3-N concentration in the leachate. This may be due to preferential flow occurring through macropores in response to high rates of "rainfall" application. This would result in lower than expected NO_3-N concentrations in the leachate as much of the soil NO_3-N would remain in the micropores. If this is the case then the LEACHN model could be viewed as presenting the highest possible NO_3-N concentrations and that in many cases non-uniform water flow through the soil will result in lower NO_3-N concentrations. Although in percentage terms the differences between predicted and observed NO_3-N concentrations was sometimes large (11% - 48%), in environmental terms the differences were not so great with the model correctly predicting that the leachate NO_3-N concentrations would remain comfortably below the MPL of 11.3 mg/L.

The LEACHN model was then evaluated as a possible decision support system (Fig. 6.1) to assist with management of the Carterton LTS. The LEACHN model was calibrated for the Carterton LTS, and was used with the Carterton field data set (Chapter 3) to simulate leaching of NO_3-N at the site. The details of the calibration process, a sensitivity analysis of the calibrated model, and the application of the calibrated model at the Carterton site are presented in Chapter 6.

Calibration of the model against the measured NO_3-N concentrations in groundwater from one of the treatments confirmed that by using measured values for some of the model parameters and varying others within the ranges normally found in the literature, the model could predict the seasonal pattern of NO_3-N concentrations observed in the groundwater. However, although the model could reproduce the seasonal pattern of NO_3-N concentrations in the groundwater the predicted NO_3-N concentrations were consistently higher than the measured values in the groundwater. Similarly, the LEACHN model predicted higher soil solution NO_3-N concentrations than those measured in samples collected from suction cups. The LEACHN model did however predict the observed seasonal pattern in NO_3-N concentrations collected from the sample cups and also the relative ranking of the soil solution NO_3-N concentrations in the three effluent application treatments.

The model was not very sensitive to changes in most of the parameters to do with inorganic nitrogen transformations. The model was relatively more sensitive to the bulk density, air entry value (a), BCAM (b), mineralisation rate, base temperature, and Q_{10} factor. Of particular importance is the sensitivity of the model to the mineralisation rate. The default value proposed by the model developers was used in these simulations, but if this value was too high it might explain the consistent over-estimation of soil solution NO_3-N concentrations.

Once the calibration and testing had been completed the LEACHN model was used to simulate the leaching of NO_3-N in the tree and pasture plots. The model predicted that the cumulative drainage would be higher from the pasture plots than the tree plots for all the effluent irrigation treatments. It is predicted that the concentrations of NO_3-N in drainage from the tree plots would be higher than from the pasture plots because of dilution in the greater volume of drainage water. The model also predicted that the NO_3-N concentrations in drainage water would decrease as the rate of effluent application increased despite the greater quantities of nitrogen being applied. This again results from the dilution of the leached NO_3-N in a greater volume of drainage water.

It had been observed in the field experiment (Chapter 3) that the NO_3-N concentration in the groundwater under several of the treatments exceeded the MPL during June. The LEACHN model was therefore used to evaluate different effluent application strategies in an attempt to reduce the amounts and concentrations of NO_3-N leached from the tree plot. This analysis suggested that reduction of effluent application rates during late autumn and early winter would have little effect on the amounts and concentrations of NO_3-N leached during winter. This was because the NO_3-N leached from the soil during this period had built up in the soil over the summer months.

The results of the both laboratory and field applications (Chapters 5 and 6) of the decision support model showed that rainfall and hydraulic loading rates were the critical parameters in determining whether environmentally significant leaching of NO_3-N will occur from a LTS. The final phase of this study investigated the effect of using actual and average rainfall data on the predictions of the calibrated LEACHN model. The modelled scenario involved the application of dairy shed effluent to the Manawatu fine sandy loam as described in Chapter 5. In this case however, the simulated scenario was a field based LTS rather than the lysimeters described in Chapter 5. Rainfall data from the Manawatu region was used in the simulation

In this analysis it was assumed that the leachate NO_3-N concentration should not exceed 11.3 mg/L (MPL). If it did then the effluent irrigation strategy was modified and the model re-run to see if the predicted leachate NO_3-N concentration remained below the MPL. The details of the different effluent irrigation strategies, the average daily rainfall and the predicted NO_3-N concentrations in the leachate in the different scenarios simulated by the LEACHN model can be found in Chapter 7. In summary, this analysis showed that effluent could be irrigated onto the LTS regularly throughout the late spring, summer and early autumn period with little likelihood of leaching – unless there was an atypically large rainfall event. However during winter the rainfall exceeds evapotranspiration for an extended period and leaching occurs. This is the case irrespective of whether or not effluent is applied over the winter period. The N application rate during the summer period is greater than the N uptake by plants and loss of N by denitrification, and so NO_3-N builds up in the soil. When leaching commences in late autumn, this NO_3-N is flushed through the soil, and this creates very high NO_3-N concentrations in the leachate during the early part of the drainage season. There seems to be little that can be done to avoid this. Other workers (e.g. Sharpley and Syers, 1979; Heng et al., 1991; Houlbrooke et al., 2008) have also observed high concentrations of NO_3-N in leachate from the first few leaching events in autumn/winter, even in the absence of effluent application.

The model prediction of cumulative leachate volume (313 mm) using average rainfall data was about half that (607 mm) predicted using actual rainfall data for 1996. Most of this difference could be attributed to the fact that 1996 was one of the wettest years on record. However, a second reason for the difference was the way that the average rainfall data in this study was calculated. This was done for each calendar day by averaging the amount of rainfall that fell on that day, over each of the last 26 years. The resulting rainfall pattern has some rain falling on all days but no days have particularly high rainfall. In winter, when the rainfall is much higher than the evaporation, the daily rainfall pattern has little effect on the cumulative leachate volume over a period of a few weeks. In contrast, in summer the average daily rainfall is much less than the daily evapotranspiration rate and so there would appear to be no possibility of leaching if the average rainfall data are used. Actual rainfall patterns for summer however, have many days without rain and sometimes quite heavy rain over a period of several days. This can sometimes be sufficient to cause significant leaching.

The predicted cumulative mass of nitrate leached using average rainfall values was 120 kg/ha/year and was less than predicted with actual rainfall data from a wet year (132 kg/ha/year). Thus, using average rainfall data resulted in a prediction of a smaller leachate volume, but a higher average NO_3-N concentration, than if actual rainfall data from 1996 were used. The differences however were fairly small from the perspective of environmental management and therefore using average rainfall data for conducting feasibility studies when designing an LTS should be satisfactory.

In conclusion, it was not possible to manage the land application of effluent throughout the year (i.e. 1 January to 31 December) in a way that the leachate NO_3-N concentration did not exceed the MPL of 11.3 mg/L at some stage. The decision support model simulations showed that rainfall and hydraulic loading rates are the critical parameters in determining whether environmentally significant leaching of NO_3-N will occur from a LTS. In all simulations it has been assumed that the major likely environmental impact of effluent application would be on groundwater NO_3-N concentrations, and that therefore the leachate from the LTS should not have a NO_3-N concentration greater than the MPL of 11.3 mg/L. This is a very conservative strategy, because normally leachate from a LTS would be diluted once it is mixed with the groundwater. It may, therefore, be possible to have leachate NO_3-N concentrations considerably in excess of 11.3 mg/L, without causing the groundwater to exceed the MPL. However, in the absence of detailed information on groundwater hydrology, a strategy of ensuring the leachate NO_3-N concentrations are always <11.3 mg/L could be justified, because that way the leachate from the LTS can never cause the groundwater to exceed the MPL. It should be noted that since the commencement of this study the environmental concern associated with the leaching of NO_3-N has shifted from impacts on drinking quality, to stimulating algal growth in lakes and rivers. This shift in environmental priorities over the time will change the emphasis in land treatment systems from the concentration of NO_3-N in leachate leaving the system, to the total amount of NO_3-N leaving the system.

Using this LEACHN based decision support system it might be possible to reduce the number of monitoring wells and the frequency of monitoring which means, in future, a reduction in the cost of monitoring. The initial cost may be high but there will be long term effects on the cost of monitoring.

9.2 Modelled Nitrogen Budget at a LTS

The LEACHN model described in Chapters 5, 6 and 7 consistently over-predicted soil solution NO_3-N concentrations and indicated that it would be very difficult to manage the LTS in a way that avoided exceeding the MPL of 11.3 mg/L NO_3-N in the leachate during at least part of the year. To investigate further why this was the case the LEACHN model was used to construct a nitrogen balance (Fig 8.1) for one of the LTS scenarios (the tenth) investigated in Chapter 7.

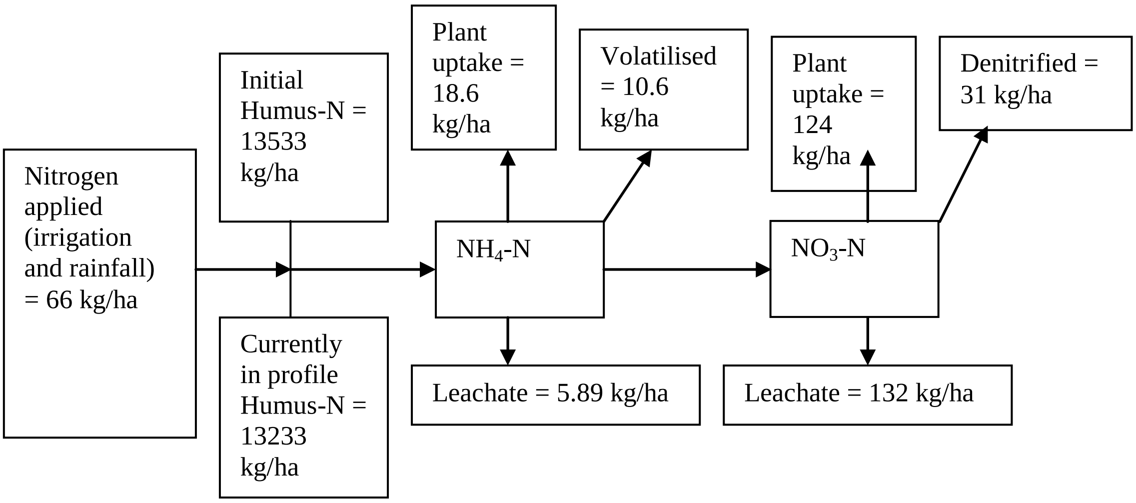


Fig. 9.1: A simple N balance of a LTS as calculated by the LEACHN model.

The total amount of N applied in effluent irrigation in the tenth Scenario (Chapter 7) was 66 kg/ha/year. The N concentration in rainfall was considered to be zero. The N uptake by the trees predicted by the model was 124 Kg/ha/year, which suggested that there should perhaps be little or no leaching of NO_3-N. However despite this large predicted N uptake by the trees, the model predicted that the total amount of NO_3-N leached would be 132 kg/ha/year. Soil organic nitrogen (Humus-N) was predicted to be the main source (Fig. 8.1) of this leached N. This predicted large net mineralisation of soil organic nitrogen is unexpected in a site receiving effluent and under a tree plantation. This predicted outcome may be a result of the mineralisation rate constant in the LEACHN model being set too high. The sensitivity analysis described in Chapter 6 highlighted the sensitivity of the model to the value used for the mineralisation rate constant. As noted earlier, a mineralisation rate constant set at too high a level could explain why the model consistently overestimated the soil solution concentrations at the Carterton LTS.

In order to understand LEACHN better and also to check its usefulness as a decision support tool for an LTS, the last phase of the study as to undertake a detailed analysis of the parameters (i.e. K_{min}, K_{nit}, K_{den} rate constants, and the bulk density) affecting on LEACHN predictions of leachate NO_3-N concentrations and the total amounts of NO_3-N leached. In this last phase of the study, an assessment of how different irrigation scenarios (i.e. rainfall alone with no effluent irrigation, rainfall and irrigation of effluent containing no N, and rainfall and irrigation with effluent containing N) impact on resulting distribution of N down the soil profile (0 - 500 mm) was also undertaken. Another model i.e. OVERSEER® was used to simulate the CDC LTS, and also the parameterised LEACHN model was used to simulate another field data set as part of the last phase of the study. The details of the detailed analysis of the LEACHN model can be found in Chapter 8. This detailed evaluation of LEACHN showed that it could be very useful decision support tool to simulate N dynamics at an effluent disposal site.

9.3 Conclusions

The study presented here is largely based on peer reviewed journal papers (except Chapter 8). Based on that published work, together with some other reported results, the following conclusions can be drawn:

1. A field study showed that land application of effluent is likely to cause N leaching that can result in NO_3-N concentrations in shallow groundwaters reaching or exceeding the MPL - even when the nitrogen loading rate (i.e. 150 Kg N/ha/year) was within application guidelines.
2. A simple hydrological model is a useful tool to predict the occurrence of leaching events, in the short term, by assessing the ability of the soil to absorb the designed irrigation volume and likelihood of possibility in the immediate future. Such a model is capable of predicting the timing and extent of leaching, but it is not able to quantify the nitrogen loss that will occur during these leaching events. Such a model is a good management tool

to avoid leaching in the short term, in order to maximise the plant uptake of effluent nutrients.

3. Both laboratory and field scale testing of a nutrient movement model based on the LEACHN programme showed that the model was able to predict the seasonal pattern of nitrate leaching from a LTS, and also the qualitative effects of changes in effluent application rates and plant species on the concentrations of NO_3-N in soil solution and leachate.
4. Although the LEACHN model proved capable of assessing the qualitative effects of changing management strategies at a LTS, it consistently overestimated the measured concentrations of NO_3-N in soil solution and in leachate. A complete nutrient budget (Fig. 9.1) constructed for one of the LTS management scenarios indicated that the default proportionality constant value in the model for calculating the rate soil organic breakdown (and hence nitrogen mineralisation) was too high for New Zealand soils.
5. The study demonstrated that when using the LEACHN based DSS to explore the consequences of different effluent application strategies care must be taken to ensure that realistic rainfall patterns are input into the model – rather than estimates of long-term averages.
6. The detailed analysis of LEACHN model showed that the model is sensitive to K_{min} values and has significant impact on soil humus-N and the amount of NO_3-N leached. The size and rate of breakdown of humus-N are the most important parameters to determine the amount of N leached, and this is a very useful piece of information. The total amount of NO_3-N leached was slightly affected by $_{Knit}$ values, and $_{Kden}$ value has no significant impact on the total amount of NO_3-N leached even when its value was increased by 100%.
7. The sensitivity of predicted NO_3-N concentrations and amount of leachate and NO_3-N leached to changes in the soil bulk density was due to the fact that increasing the soil bulk density would increase the soil humus-N, with the resulting in N mineralisation, which would lead to an increase in NO_3-N leaching as predicted by the model.
8. The irrigation scenario analysis showed that the LEACHN model is able to build up a very detailed picture of what is happening to soil N in the soil profile in response to three irrigation scenarios i.e. no effluent application (i.e. just natural rainfall), application of pure water (i.e. rainfall and effluent

without N) and effluent application with N composition. Again, this is very useful piece of information.

9. The comparison of two different models showed the amount of N leached, as predicted by parameterised OVERSEER® model, was 69 kg/ha/year which was reasonably close to that of predicted by LEACHN model (85 kg/ha/year). The comparison showed that, depending on the level of precision required, the OVERSEER® model could be used to simulate N dynamics at a LTS similar to CDC.
10. The detailed analysis showed that LEACHN model is not designed for an effluent-irrigated land treatment system and also this model doesn't allow adding organic form of N. Also, LEACHN is not designed to predict crop yield or development; however, this comparison showed that the LEACHN does have the flexibility of creating a system like an effluent-irrigated LTS.
11. It seems that LEACHN model was originally designed for a cropping regime in which any N in the irrigation water could be in the form of urea, ammonium or nitrate (i.e. no complex organic N). When first examined or looked at, this would seem to be major limitation for its possible application to effluent irrigated sites under pasture or forest. But, in fact, it seems that by creative management of the input parameters the model could be used other situations than irrigated cropping.
12. The testing of the parameterised LEACHN model using another short term data set showed that the model appeared to have promise in terms of predicting leachate NO_3-N concentrations. The key driver of leachate NO_3-N concentrations in short term events was the initial soil solution NO_3-N concentration in the soil profile, and this information is not usually available. But, for long periods of the year in winter and spring the soil solution NO_3-N concentrations are similar from year to year. The testing of the model showed that historical soil data could be used to achieve reasonably accurate predictions of likely leachate NO_3-N concentrations.
13. Therefore it can be concluded that LEACHN provides useful information on how much NO_3-N is going to leach and when this is likely to happen. At the same time it provides useful insights into the movements and transformation of soil and effluent N down the profile, which could be very helpful for the operators/managers of land treatment systems.

9.4 Recommendations and Future Work

1. This study highlights that prior to commencing the design of a LTS there needs to be a clear understanding of what environmental outcomes the LTS is intended to achieve. If the aim is to minimise the total weight of N leached from the LTS then the effluent application strategy may be different than if the aim is to keep the concentration of nitrate-N in a shallow groundwater as low as possible. If it is the latter aim that is considered most important, it is important to understand how much of the groundwater is contributed by leachate from the LTS and therefore the extent of dilution of the leachate. When this is known, a target maximum concentration of NO_3-N in the leachate can be established.
2. The study demonstrated that although monitoring the concentration of nitrate in the groundwater under the LTS is important to gauge the long term performance of the treatment system the results of such monitoring cannot be used as a management tool to make short term changes in effluent application strategies. By the time any problem is recognised it is too late because there is already several months' worth of nitrate accumulated in the soil profile and is ready to leach. It is therefore recommended that a mechanistic model of nitrate leaching (such as LEACHN) is used when designing the LTS (size of the area and vegetation type) and the best effluent application strategy to meet the environmental goals.
3. In this study the LEACHN model included the default and literature values for the mineralisation rate constant. The detailed analysis indicated that the outputs from the LEACHN model are influenced greatly by the assumed annual rate of organic matter mineralisation at the site. This is because of the high organic matter content of many New Zealand soils. It is suggested that further research is required to measure the actual mineralisation rate likely to be occurring at an LTS.
4. Testing of the model on another short term data set showed that the rate of nitrification throughout the soil profile is very important in determining how much NO_3-N will be leached. More research is needed on nitrification rates at deeper soil depths.

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